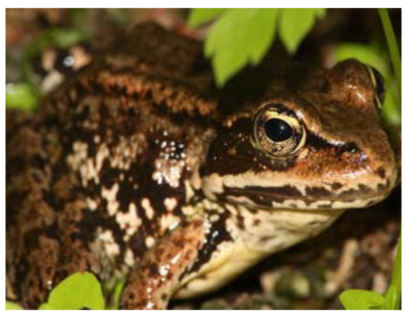
BEFORE THE SECRETARY OF THE INTERIOR

PETITION TO LIST 53 AMPHIBIANS AND REPTILES IN THE UNITED STATES AS THREATENED OR ENDANGERED SPECIES UNDER THE ENDANGERED SPECIES ACT








CENTER FOR BIOLOGICAL DIVERSITY

JULY 11, 2012

Notice of Petition

Ken Salazar, Secretary
U.S. Department of the Interior
1849 C Street NW
Washington, D.C. 20240
exsec@ios.doi.gov

Dan Ashe, Director
U.S. Fish and Wildlife Service
1849 C Street NW
Washington, D.C. 20240
Dan_Ashe@fws.gov

Gary Frazer, Assistant Director for Endangered Species
U.S. Fish and Wildlife Service
1849 C Street NW
Washington, D.C. 20240
gary_frazer@fws.gov

Nicole Alt, Chief
Division of Conservation and Classification, Endangered Species Program
U.S. Fish and Wildlife Service
4401 N. Fairfax Drive, Room 420
Arlington, VA 22203
nicole_alt@fws.gov

Douglas Krofta, Chief
Branch of Listing, Endangered Species Program
U.S. Fish and Wildlife Service
4401 North Fairfax Drive, Room 420
Arlington, VA 22203
Douglas_Krofta@fws.gov

AUTHORS

Collette L. Adkins Giese
Herpetofauna Staff Attorney
Center for Biological Diversity
P.O. Box 339
Circle Pines, MN 55014-0339
CAdkinsGiese@biologicaldiversity.org

D. Noah Greenwald
Endangered Species Program Director
Center for Biological Diversity
P.O. Box 11374
Portland, OR 97211-0374
NGreenwald@biologicaldiversity.org

Tierra Curry
Conservation Biologist
P.O. Box 11374
Portland, OR 97211-0374
TCurry@biologicaldiversity.org

PETITIONERS

The Center for Biological Diversity. The Center for Biological Diversity ("Center") is a non-profit, public interest environmental organization dedicated to the protection of native species and their habitats through science, policy, and environmental law. The Center is supported by over 375,000 members and on-line activists throughout the United States. The Center and its members are concerned with the conservation of endangered species, including amphibians and reptiles, and the effective implementation of the ESA.

C. Kenneth Dodd, Jr.
University of Florida; FWS - Office of Endangered Species Staff Herpetologist 1976-1984
[redacted]

Kenney Krysko
Senior Biological Scientist, Division of Herpetology, Florida Museum of Natural History
kenneyk@flmnh.ufl.edu

Michael J. Lannoo
Professor, Department of Anatomy and Cell Biology, Indiana University School of Medicine
mlannoo@iupui.edu

Thomas Lovejoy
Biodiversity Chair, H. John Heinz III Center for Science, Economics and Environment
tlovejoy@gmu.edu

Allen Salzberg
Publisher, HerpDigest
asalzberg@herpdigest.org

Edward O. Wilson
Professor, Museum of Comparative Zoology, Harvard University
ewilson@oeb.harvard.edu

Submitted this 11th day of July, 2012

Pursuant to Section 4(b) of the Endangered Species Act ("ESA"), 16 U.S.C. § 1533(b); Section 553(e) of the Administrative Procedure Act, 5 U.S.C. § 553(e); and 50 C.F.R. § 424.14(a), the Center for Biological Diversity (through Collette L. Adkins Giese, D. Noah Greenwald and Tierra Curry), Kenneth Dodd Jr., Kenney Krysko, Michael Lannoo, Thomas Lovejoy, Allen Salzberg, and Edward O. Wilson hereby petition the Secretary of the Interior, through the United States Fish and Wildlife Service ("FWS"), to list 53 species of amphibians and reptiles as threatened or endangered and to designate critical habitat to ensure their recovery. This petition sets in motion a process placing definite response requirements on the FWS and specific time constraints upon those responses.

The FWS has long recognized that providing protection for multiple species improves efficiency of listing and recovery. For example, in 1994, the FWS specifically stated its policy to undertake "Group listing decisions on a geographic, taxonomic, or ecosystem basis where possible." 59 Fed. Reg. 34724. In furtherance of this policy, the FWS developed listing guidance that specifically encourages "Multi-species listings . . . when several species have common threats, habitat, distribution, landowners, or features that would group the species and provide more efficient listing and subsequent recovery" (FWS 1994, p. iv). Consistent with this policy, this petition identifies more than 50 species of herpetofauna that face common threats and need ESA protection.

TABLE OF CONTENTS

I. INTRODUCTION

The Earth is facing the largest mass extinction in 65 million years (Lawton and May 1995, Vitousek et al. 1997, Wilson 1999, Myers and Knoll 2001, Balmford et al. 2003, Chivian and Bernstein 2008). Current global extinction rates for animals and plants are estimated to be up to 10,000 times higher than the background rate in the fossil record (Chivian and Bernstein 2008). Amphibians and reptiles – collectively, herpetofauna – are some of the most imperiled of all taxa. According to the International Union for the Conservation of Nature ("IUCN"), nearly 30 percent of the nearly 10,000 species of amphibians and reptiles in the world are endangered or vulnerable to extinction (IUCN 2011). To be sure, scientists have observed widespread amphibian and reptile population declines in the United States and globally (Barinaga 1990, Vitt et al. 1990, Wyman 1990, Wake 1991, Vial and Saylor 1993, Corn 1994, Fisher and Shaffer 1996, Alford and Richards 1999, Gibbons et al. 2000, Houlahan et al. 2000, Stuart et al. 2004).

Declining populations of amphibians and reptiles are a concern for many reasons. To begin, herpetofauna are important elements of our national biological heritage. They are crucial to the natural functioning of many ecological processes and play key roles as both predators and prey. For example, many reptiles and amphibians are essential links between aquatic and terrestrial food webs as they transfer energy from aquatic prey to terrestrial predators (Wilbur 1997). The importance of adult amphibians and reptiles in terrestrial food webs is highlighted by their efficiency at converting the prey they consume to new animal tissue; as ectotherms, they are more than 25 times more efficient than mammals or birds (Pough 1980, 1983). Moreover, amphibians are important in the control of insect pests such as mosquitoes, many snakes control rodents that threaten crop damage (Pough et al. 1998), and the presence of lizards even helps reduce exposure of lyme disease to humans (Casher et al. 2002).

The loss of amphibians and reptiles is a symptom of environmental degradation, as amphibians and reptiles are valuable bioindicators of environmental health. Amphibians have highly permeable skin and egg membranes and complex life cycles with aquatic and terrestrial life history stages that make them sensitive to environmental change, and both amphibians and reptiles are often philopatric to specific breeding, foraging, and overwintering habitats that must be connected by habitats suitable for migration (Bauerle et al. 1975, Duellman and Trueb 1986, Weygoldt 1989, Wake 1991, Olson 1992, Blaustein 1993, 1994, Welsh and Ollivier 1998). As a result, populations are easily harmed by habitat fragmentation and become vulnerable to extinction (Dodd and Cade 1998).

The factors implicated in the disappearance of amphibian and reptiles are discussed in the following sections of this petition. A primary threat to nearly all of the petitioned amphibians and reptiles is habitat loss from human activities such as development, logging, agriculture, and mining (e.g., Bury et al. 1980, Ballinger and Watts 1995, Lind et al. 1996). Many of the petitioned amphibians and reptiles are also threatened by overutilization, especially collection for the pet trade (e.g., Jennings and Hayes 1985, Weir 1992, Wilkinson 1996b). Disease threatens some of the petitioned species, such as the amphibians that are susceptible to the chytrid fungus epidemic (e.g., Dodd 1988, Carey 1993, Berger et al. 1998). Other factors implicated in observed declines include: introduced species (e.g., Bradford 1989, Kupferberg 1996, Adams 1997, Hecnar and M`Closkey 1997), environmental pollutants (e.g., Dunson et al. 1992, Guillete et al. 1994, Sparling et al. 2001), increased ambient UV-B radiation (e.g., Blaustein et al. 1995), and climate change (e.g., Pounds and Crump 1994). Because of these threats and others, all of the species included in this petition qualify as endangered or threatened under the Endangered Species Act.

II. METHODS

We identified species for petitioning based on an iterative process utilizing information from available databases and literature cataloging information on species' habitat preferences, status and threats, including NatureServe, IUCN, and AmphibiaWeb. We formed an initial list by searching NatureServe for amphibian and reptile species (including turtles) that occur in the United States and appear to be imperiled. We considered species imperiled if they were classified as G1, G2, or G3 by NatureServe or included on the IUCN's Red List as Near Threatened or worse. Once we had an initial list, we further narrowed the list by including only those species with some information documenting declines and demonstrating threats. We largely avoided species that have yet to be fully described or would best be listed as a distinct population segment. Then we consulted with numerous scientific experts specializing in amphibians or reptiles to obtain their feedback on whether listing of the species may be warranted, and we removed many species based on expert advice.

Once species were identified for the petition, we created a database structured for entering the basic information necessary to show that listing of the species may be warranted, including fields on range, habitat, status, and the five factors under the ESA for determining whether a species is threatened or endangered. 16 U.S.C. § 1533(a)(1). We then systematically searched available literature on the species and created the individual species accounts contained in this petition. Using the information entered in our database, we then tabulated the numbers of species impacted by each threat discussed in the petition. (A spreadsheet showing the species and the threats they face will be provided electronically.) The numbers reported here represent a minimum because we only included those threats to the species identified in the literature.

III. THREATS

Section 4 of the Endangered Species Act and its implementing regulations set forth the procedures for adding species to the federal list of endangered and threatened species. 16 U.S.C. § 1533; 50 C.F.R. Part 424. FWS may determine a species to be endangered or threatened due to one or more of the five factors described in section 4(a)(1) of the Act. Each of these factors is discussed below, as all of these threaten amphibians and reptiles in the United States.

A. THE PRESENT OR THREATENED DESTRUCTION, MODIFICATION, OR CURTAILMENT OF ITS HABITAT OR RANGE

Habitat destruction, alteration, and fragmentation are the most serious causes of amphibian and reptile population declines and extinctions (Wyman 1990, Blaustein et al. 1994, Beebee 1996, Alford and Richards 1999, Dodd and Smith 2003, Beebee and Griffiths 2005). It is well established that habitat loss and fragmentation can reduce amphibian and reptile abundance, species richness, and genetic diversity (e.g., Saunders et al. 1991, deMaynadier and Hunter 1995, Turner 1996, Wind 1996, Hitchings and Beebee 1998, Vos and Chardon 1998, Kolozsvary and Swihart 1999, Dupuis and Bunnell 1999, Joly et al. 2001, Scribner et al. 2001, Laurance et al. 2002). All of the petitioned species are threatened by habitat loss, degradation, or fragmentation.

Habitat fragmentation is a secondary effect of habitat loss, and it can be difficult to separate the effects of habitat loss from fragmentation (Fahrig 1997, Wilkinson 2008). Habitat fragmentation reduces the amount of available habitat, degrades habitat quality through edge effects, and impedes dispersal between habitat remnants (Saunders et al. 1991, Fahrig and Merriam 1994). Numerous studies document the threat of habitat fragmentation on amphibians and reptiles (Duellman and Trueb 1986, Laan and Verboom 1990, Branch et al. 1996, Turner 1996, Marsh and Pearman 1997, Hokit et al. 1999, Cosson et al. 1999, Kjoss and Litvaitis 2001, Schlaepfer and Gavin 2001, MacNally and Brown 2001, Lehtinen et al. 2003, Driscoll 2004, Marsh et al. 2005, Semlitsch et al. 2007).

As discussed below, humans harm habitats of amphibians and reptiles for residential and commercial development, recreation, logging, mining, grazing, agriculture, roads, and through physical alterations of aquatic habitats such as impoundments, channelization, and diversions.

1. <u>Urbanization</u>

Urbanization is a major threat to amphibians and reptiles (Reinelt et al. 1998, McKinney 2002, Rubbo and Kiesecker 2005, McKinney 2006), and this petition identifies 23 species threatened by urbanization, which represents a minimum of 40 percent of the petitioned species. The most obvious impact of urbanization is the direct loss and fragmentation of habitat. Indirect impacts include human-subsized or introduced predators, changes in water quality or flow, and road construction – all of these are discussed in other sections of this petition. In addition, noise (Patricelli and Blickley 2006) and light pollution (Moore et al. 2000) from human development can interfere with behavior patterns and disrupt breeding and feeding activities, particularly for amphibians (Dodd 1997).

Another major impact of urbanization is that increased human density often precludes many beneficial forms of habitat restoration and management (Sutherland 2009). Prescribed burning is a prime example because many terrestrial ecosystems need fire to maintain healthy plant communities and burning may be impossible in small remnant tracts of habitat surrounded by human land use (NC WRC 2005). Fire suppression is a threat to several of the petitioned species, including the Key ringneck snake, short-tailed snake, Florida pine snake, Apalachicola kingsnake, Carolina gopher frog, Cascades frog, and Florida scrub lizard. To be sure, the greatest threat to the Carolina gopher frog is the loss and alteration of both upland and wetland habitats resulting from development and fire suppression (Jensen and Richter 2005).

Many studies have determined that ponds and wetlands situated in more urbanized landscapes contain fewer species or lower abundances of frogs, toads, and salamanders (Delis 1993, Richter and Azous 1995, Delis et al. 1996, Chin 1996, Reinelt et al. 1998, Knutson et al. 1999, Vos and Chardon 1998, Knutson et al. 1999, Lehtinen et al. 1999, Carr and Fahrig 2001, Herrig and Shute 2002, Houlahan and Findlay 2003, Pellet et al. 2004, Rubbo and Kiesecker 2005, Pillsbury and

Miller 2008). Snakes and turtles (Gibbs and Shriver 2002, Aresco 2005) also seem to be quite sensitive to urbanization, especially to road impacts (Findlay and Houlahan 1997, Andrews and Gibbons 2005, Roe et al. 2006, Row et al. 2007). For example, many populations of the western pond turtle have been lost as a result of urbanization in the area south of central California (Rathbun et al. 1992).

The threat of urbanization increases with our growing population. The more people there are, the greater the demand there is on the land and water. As of February of 2012, the United States has a total resident population of 313,031,000, making it the third most populous country in the world (U.S. Census Bureau 2012) with growth among the highest in industrialized countries (Central Intelligence Agency 2012). The American population more than tripled during the 20th century – at a growth rate of about 1.3 percent a year – from about 76 million in 1900 to 281 million in 2000. The Census Bureau projects a U.S. population of over 400 million in 2050 (U.S. Census Bureau 2009).

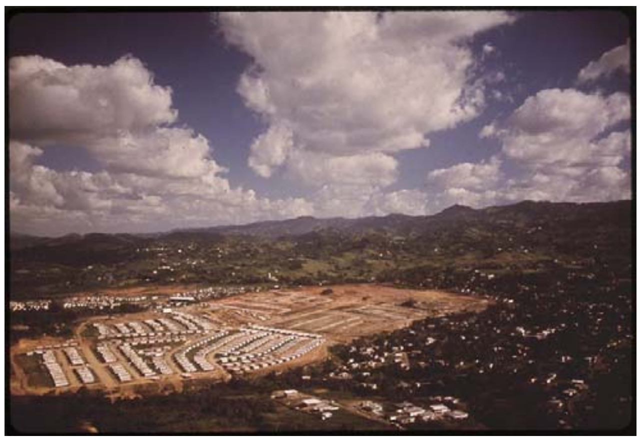


Figure 1. Urbanization threatens 23 species included in the petition.

2. Recreation

The petition identifies 16 species threatened by recreation (30 percent of the petitioned species), which is a minimum figure because it only represents those species where recreation is documented as a threat in the scientific literature. The impacts of off-road vehicles (ORVs) are discussed here. The impacts of fisheries management are discussed in the "Other Factors" section below. Water impoundments are created to provide a variety of recreational opportunities including fishing, hunting, swimming, boating, and camping; the harmful impacts of impoundments are discussed in the "Alteration of Aquatic Habitats" section below.

Several of the petitioned species are impacted by ORVs, particularly through trampling of organisms, habitat degradation, and noise disturbance (see Taylor undated, Maxell and Hokit 1999, Himes et al. 2002, Munger et al. 2003, Wuerthner 2007). These include the Carolina gopher frog, foothill yellow-legged frog, Cedar Key mole skink, Colorado Desert fringe-toed lizard, Panamint alligator lizard, Yuman Desert fringe-toed lizard, and Inyo Mountains salamander.

Luchenbach and Bury (1983) studied the impacts of ORVs on lizards, including the Colorado Desert fringe-toed lizard. They determined that there were frequent encounters between ORVs and lizards that resulted in the animal's death, and they found significantly fewer Colorado Desert fringe-toed lizards on ORV-impacted plots than control plots. In addition, the researchers found an increase in the frequency of tail loss among lizards following an increase in ORV activity in the area; tail loss likely leads to reduced survivorship and fecundity.

Numerous studies have shown that ORVs can degrade habitat to the point that it becomes unusable by herpetofauna (Speight 1973, Busack and Bury 1974, Liddle 1975, Bury et al. 1977, Jennings 1997, Munger et al. 2003). For example, the fragile sand dune habitats of the Colorado Desert fringe-toed lizard and Yuman Desert fringe-toed lizard have been negatively impacted by off-road vehicle (Jennings and Hayes 1994). And ORV use has degraded habitats of the Carolina gopher frog (Hammerson and Jensen 2004).

Figure 2. Off-road vehicles can degrade habitat and disturb herpetofauna.

3. <u>Timber Harvest</u>

Many amphibians and reptiles, including over half of the petitioned species, are threatened by timber harvest. More than 95 percent of the original forest in the 48 conterminous states has been lost (Noss et al. 1995), including 99 percent of eastern deciduous forest (Allen and Jackson 1992). Much of this loss can be attributed to timber harvest.

There is general agreement that timber harvest in temperate regions can have numerous negative effects on species richness and abundance of forest-dependent species, including amphibians in particular (e.g., Bury 1983, Petranka et al.1993, Petranka et al. 1994, deMaynadier and Hunter 1995, Dupuis et al. 1995, Ash 1997, Dodd 1997, Herbeck and Larsen 1999, Grialou et al. 2000, Ross et al. 2000, DeGraaf and Yamasaki 2002, Adams and Bury 2002, Herrig and Shute 2002, Ford et al. 2002, Knapp et al. 2003, Russel et al. 2004, Karraker and Welsh 2006, Olson et al. 2007, Semlitsch et al. 2009).

In particular, studies by Semlitsch et al. (2009) generated dozens of statistically significant negative effects of timber harvest treatments on a broad range of pond-breeding amphibian responses. Removal of the forest canopy or course woody debris exposes amphibians to warmer and drier microclimate conditions (Keenan and Kimmins 1993, Ash 1995, Harpole and Haas 1999, Chen et al. 1999, Zheng et al. 2000), eventually reducing leaf litter (Hughes and Fahey 1994, Ash 1995) and food resources (Seastedt and Crossley 1981). These changes eventually lead to lower survival (Todd and Rothermel 2006) or higher evacuation of habitats (Semlitsch et al. 2008). A large portion of the amphibian population dies if they stay in clearcut areas, especially small juveniles (Rothermel and Luhring 2005, Todd and Rothermel 2006, Harper 2007, Patrick et al. 2008, Todd et al. 2008).

Several of the petitioned species impacted by logging are salamanders, including the relictual slender salamander, Kern Canyon slender salamander, Oregon slender salamander, California giant salamander, Shasta salamander, Blue Ridge gray-cheeked salamander, Caddo Mountain salamander, Cheoah Bald salamander, Fourche Mountain salamander, Peaks of Otter salamander South Mountain gray-cheeked salamander, Pigeon Mountain salamander, white-spotted salamander, Weller's salamander, green salamander, Cascade torrent salamander, Columbia torrent salamander, and Olympic torrent salamander. Salamander responses to logging have been extensively studied. Salamander populations in timbered areas are usually lower, and sometimes absent, when compared to untimbered areas (Blymer and McGinnes 1977, Bury 1983, Enge and Marion 1986, Pough et al. 1987, Ash 1988, Bury and Corn 1988, Stiven and Bruce 1988, Corn and Bury 1989, Welsh 1990, Raymond and Hardy 1991, Petranka et al. 1993, Petranka et al. 1994, Dupuis et al. 1995, de Maynadier and Hunter 1995, Ash 1997, de Maynadier and Hunter 1998, Bast and Maret 1998, Sattler and Reichenbach 1998, Herbeck and Larsen 1999, Grialou et al. 2000, Rocco and Brooks 2000, Barr and Babbitt 2002, Willson and Dorcas 2003). To be sure, across 16 research projects, control stands had about 4.3 times more captures of salamanders than clearcut stands (deMaynadier and Hunter 1995). In one study, Petranka et al. (1993) compared species richness and abundance of salamanders on six recent clearcuts with salamander densities in mature forest stands in the Appalachian Mountains. They found that salamander densities in the mature stands were five times higher than those in the recently cut plots. From these surveys, Petranka et al. (1993) estimated that timber harvesting in the Appalachian Mountains resulted in the loss of 14 million salamanders annually.

Many studies have dealt only with clearcutting, but some studies have analyzed alternative silvicultural practices such as shelterwoods (Mitchell et al. 1996, Sattler and Reichenbach 1998) and single-tree selection or thinning (Pough et al. 1987, Messere and Ducey 1998, Grialou et al. 2000). Research by Homyack and Haas (2009) indicates that a range of forest management techniques may cause lasting reductions of terrestrial salamander populations likely due to both low population growth rates and changes to habitat. Methods such as group selection and shelterwood involve several entries into the stand (Knapp et al. 2003). This not only exposes the salamander community to a reopening of the canopy and the associated drying of the environment, but it also results in recompaction or disturbance of the soil and leaf litter from tree felling and logging traffic (Knapp et al. 2003).

Logging has other indirect negative effects on amphibians and reptiles. Erosion from poor forestry practices causes sedimentation and degrades water quality (Williams et al. 1993). Herbicides used after timber harvests also negatively affect amphibians and other aquatic organisms (Dodd 1997, Hayes et al. 2002, Herrig and Shute 2002, Cauble and Wagner 2005, King and Wagner 2010). In some regions, clearcutting may also result in soil compaction and disturbance to the soil profile during the course of timber extraction and postharvest site preparations, such as burning. In addition, behavioral studies show that both juvenile and adult amphibians often avoid entering clearcuts when given a choice (Rittenhouse and Semlitsch 2006, Patrick et al. 2008, Todd et al. 2009), causing habitat fragmentation and isolation of populations.

The harmful effects of timber harvest on amphibians are long lasting. Scientists have concluded that population recovery from clearcutting requires 50–70 years (Petranka et al. 1993) or even longer (Petranka et al. 1994, Homyak and Haas 2009, Semlitsch et al. 2007).

Figure 3. Logging destroys and degrades habitat of over half of the petitioned species. Photo by Rvannatta.

4. <u>Mining</u>

Mining threatens 14 of the petitioned species (about 25 percent). Impacts on herpetofauna from coal mining have been extensively documented (Riley 1952, 1960; Myers and Klimstra 1963; Redmond 1980; Turner and Fowler 1981; Matter and Ney 1981; Gore 1983; Fowler et al. 1985; Dodd 1997; Folkerts 1997; Middelkoop et al. 1998; Soucek et al. 2003), as has mining for other materials (Porter and Hakanson 1976, Schnoes and Humphrey 1987, Saugey et al. 1988, Twigg and Fox 1991).

Mining negatively impacts amphibians and reptiles, and other biota, through direct habitat destruction, decreased water availability, variations in flow and thermal gradients, and chronic and acute pollution of surface and ground water (Linder et al. 1992, FWS 1996, Hamilton 2002, Williams 2003, Pond et al. 2008, Palmer and Bernhardt 2009, Pomponio 2009, Wood 2009, Loughman and Welsh 2010, Bernhardt and Palmer 2011, Lindberg et al. 2011). In addition, surface coal mining and associated road-building increase human access to imperiled species, which can lead to poaching and contribute to the spread of invasive species (FWS 1996). Macroinvertebrate communities are also seriously degraded in mining tributaries (Carlisle et al. 2008, Pond et al. 2008, Wood 2009), which has negative consequences for the herpetofauna that rely on these organisms for prey.

Contaminants from coal mining and processing operations include sediments, metals, hydraulic fluids, frothing agents, modifying reagents, pH regulators, dispersing agents, flocculants, and media separators (Cherry et al. 2001, Soucek et al. 2003, Ahlstedt et al. 2005). Surface waters receiving mine discharge commonly have extremely low pH levels, below 3.0, with toxic impacts extending several miles downstream (Soucek et al. 2003). Acid mine runoff from abandoned mines has completely destroyed stream biotas in many areas (Folkerts 1997). Pollution from mining leads to less diverse and more pollution-tolerant species (Cherry et al. 2001, EPA 2005, Lemly 2009, Pomponio 2009).

The Inyo Mountain salamander, one of the petitioned species, provides an example of mining impacts. More than 360 mining claims are located near Inyo Mountains salamander populations (Papenfuss and Macey 1986). Even when mining lands are reclaimed, habitat is unlikely to be restored to suitable conditions and recolonization is unlikely to occur given that remaining localities for the salamander are highly isolated (see Hanson and Wake 2005). The green salamander, one of the petitioned salamanders, is a species threatened by mountaintop removal coal mining, as are many high elevation endemics (Gatwicke 2008, Wood 2009). Concerning the concentration of endemic salamanders in coal mining areas, Palmer and Bernhardt (2009) state: “Where mining activities destroy stream habitat and degrade stream water quality, many of these taxa become locally extinct, and for species with small geographic distributions, mining activities will contribute to their global extinction.”


Figure 4. Mountaintop removal and other mining threatens several petitioned species, especially the salamanders.

5. Livestock Grazing

Livestock grazing has a profound impact on native biota including nine species of herpetofauna included in this petition. In the United States, grazing has contributed to the demise of 22 percent of federal threatened and endangered species – nearly equal to logging and mining combined (USDI-BLM and USDA-Forest Service 1995, Czech et al. 2000, USDA-NRCS 1997).

Grazing is particularly widespread in the western U.S (Fleischner 1994). Approximately 70 percent of the 11 western states (Montana, Wyoming, Colorado, New Mexico, and westward) is grazed by livestock (Council for Agricultural Science and Technology 1974, Crumpacker 1984). Grazing occurs on the majority of federal lands in the West, including most of the domains of the U.S. Bureau of Land Management (BLM) and the U.S. Forest Service, as well as in many national wildlife refuges, federal wilderness areas, and even some national parks.

Several studies have documented grazing impacts on herpetofauna (Busack and Bury 1974, Jones 1981, Szaro et al. 1985, Jones 1988, Bock et al. 1990, Munger et al. 1994). For example, Borisenko and Hayes (1999) found locations with foothill yellow-legged frogs (a species included in this petition) had significantly less grazing than locations without frogs. And heavy grazing by cattle in summer in dried pond basins likely reduces or eliminates oviposition sites for the Carolina gopher frog, another petitioned species (Hammerson and Jensen 2004).

Indeed, amphibians and water-associated reptiles are particularly vulnerable to grazing near streams and other waterways. Livestock grazing has damaged 80 percent of the streams and riparian ecosystems in the arid West (Belsky et al. 1999; see also Hendrickson and Minckley 1984, Ohmart et al. 1988, Zwartjes et al. 2005). A survey of peer-reviewed studies on the effects of livestock grazing on stream and riparian ecosystems found that grazing negatively affects water quality and quantity, channel morphology, hydrology, soils, instream and streambank vegetation, and aquatic and riparian wildlife (Belsky et al. 1999). In addition, diversion of water from western streams for livestock watering and forage production reduces water quantity (and even entirely dewaters streams) (Wuerthner 2002).

Indeed, livestock congregate along streambeds, eat or trample the vegetation, accelerate bank erosion, and contaminate the stream with feces and urine (Behnke and Raleigh 1978, Hafner and Brittingham 1973, Kauffman and Kruger 1984, Belding et al. 2000, Schoonover et al. 2006). When vegetation is trampled, plant diversity decreases and cover for herpetofauna and other wildlife is reduced (Behnke and Raleigh 1978, Bulow-Olsen 1980, Kauffman and Krueger 1984, Belding et al. 2000). Such impacts from grazing are a threat to the Inyo Mountain salamander and Kern Canyon slender salamander, two of the petitioned species.

Livestock grazing may also impact herpetofauna by altering the composition and community structure of aquatic fauna, which form the food base for many vertebrates (e.g. Covich et al. 1999). The aquatic invertebrate community may change because of altered stream channel characteristics, higher water temperatures induced by loss of riparian vegetation, sediment deposition or substrate size changes, or nutrient impoverishment or enrichment (Rinne 1988, Jones et al. 1997).

Finally, grazing harms herpetofauna by contributing to the spread of invasive species. Livestock can cause weed invasion by grazing and trampling native plants; clearing vegetation, destroying the soil crust and preparing weed seedbeds through hoof action; and transporting and dispersing seeds on their coats and through their digestive tracks (Belsky and Gelbard 2000). In addition, ranchers sometimes plant exotic species for cattle forage, such as buffelgrass, which clogs habitat of wildlife such as the reticulate collared lizard, one of the petitioned species (Scott 1996). Livestock production can also facilitate the spread of invasive species through the construction of reservoirs, irrigation canals and watering holes for cattle, which create permanent waters utilized by invasive bullfrogs (Bury and Whelan 1984)

Figure 5. Poor grazing practices can degrade habitat for amphibians and reptiles. Photo by Billy Hathorn.

6. Agricultural Practices

Agricultural practices are a threat to 21 of the amphibian and reptile species included in the petition (about 40 percent). Like other land uses such as grazing and mining, farming impacts herpetofauna primarily through habitat loss and fragmentation. Agricultural practices have contributed to widespread loss of wetlands, which is discussed below in the "Inadequate Regulatory Mechanisms" section. Disking of depressed wet areas to promote drying is another common agricultural practice that eliminates breeding habitats for amphibians (Trauth et al. 2006), including the western spadefoot, a petitioned species (Davidson et al. 2002, Fisher and Shaffer 1996).

More recently, farmers have begun precision leveling their fields to enhance the equal distribution of irrigation water throughout the fields, improve crop yields, and reduce nonpoint source pollution from sediment and agricultural chemicals (Natural Resources Conservation Service 2002, Trauth et al. 2006). This now common practice removes the depressions that serve as amphibian breeding pools as well as rearranges the top layers of soil housing underground burrows and their resident frogs. Prior to the introduction of precision land-leveling, the natural depressions in the farm fields held water long enough for frog recruitment even in years of below-average rainfall (Trauth 1992). One of the petitioned species, the Illinois chorus frog, is primarily threatened by precision land leveling (Trauth et al. 2006).

In addition, both traditional farming practices and confined animal feeding operations contribute to water quality degradation through erosion, sedimentation, and pollution from point and non-point sources (Patrick 1992, Morse et al. 1997, Neves et al. 1997, Peterson et al. 2000, Irwin et al. 2001, Buckner et al. 2002, Herrig and Shute 2002, Mallin and Cahoon 2003, Orlando et al. 2004). Agricultural pollution carries sediment, pesticides, fertilizers, animal wastes, pathogens, salts, and petroleum particles into waterways (Morse et al. 1997, EPA 2009). Non-point source pollution from agriculture is the leading source of water quality impairment in lakes and rivers in the United States, and is also a major contributor to groundwater contamination and wetlands degradation (EPA 2009). Agricultural runoff is a significant threat to many of the petitioned species, including the Blanding's turtle. These turtles are known to be sensitive to use of herbicides, which destroy aquatic vegetation and likely affect the turtle itself (Kofron and Schreiber 1985). Blanding's turtles were nearly extirpated in Missouri due to marsh drainage and use of pesticides (Kofron and Schreiber 1985). Impacts of pollution on herpetofauna are discussed below in the "Other Factors" section.

Finally, agricultural practices can also cause direct mortality. For example, mortality from mowing of agricultural fields is a significant threat to the wood turtle, one of the petitioned species (Akre and Ernst 2006, Castellano et al. 2008, Erb and Jones 2011).

Figure 6. Agriculture threatens 40 percent of the petitioned species through habitat loss, exposure to pollutants, and direct mortality. Photo by Nigel Mykura.

7. Roads

Roads are a significant threat for amphibian and reptile populations, including 28 (over half) of the petitioned species. Roads are significant features of most landscapes, ecologically influencing an estimated 15-20 percent of the United States land area (Jochimsen et al. 2004). Many studies have documented the importance of road density, traffic density, and urbanization variables in limiting amphibian and reptile populations and causing declines (Rosen and Lowe 1994, Mitchell 1994, Buhlmann 1995, Ashley and Robinson 1996, Kline and Swann 1998, Rudolph et al. 1999, Knutson et al. 1999, Carr and Fahrig 2001, Mazerolle 2004, Pellet et al. 2004, Jochimsen et al. 2004, Rubbo and Kiesecker 2005, Aresco 2005, Glista et al. 2007, Pillsbury and Miller 2008, Sutherland 2009).

The most direct impact of roads on herpetofauna is roadkills. A growing literature suggests that a significant amount of amphibian and reptile mortality is associated with road kill (e.g., Campbell 1956, Van Gelder 1973, Dodd et al. 1989, Bernardino and Dalrymple 1992, Fahrig et al. 1995, Rosen and Lowe 1994, Ashley and Robinson 1996, Vos 1997, Kline and Swann 1998, Rudolph et al. 1999, Enge and Wood 2002, Smith and Dodd 2003, Aresco 2005, Orlowski 2007). Rosen and Lowe (1994) estimate that tens to hundreds of millions of snakes have been killed by automobiles in the United States. Studies provide evidence that road mortality is most detrimental to populations of amphibian and reptile species with low reproductive rates (Rosen and Lowe 1994, Ruby et al. 1994, Fowle 1996, Kline and Swann 1998, Gibbs and Shriver 2002).

Amphibians are particularly vulnerable to road effects because their life histories involve migrating between wetlands and upland habitat. As such, the highest levels of mortality for amphibians are usually found in places where roads intersect major wetlands or other aquatic breeding habitats (van Gelder 1973, Kuhn 1987, Reh and Seitz 1990, Oldham and Swan 1991, Ashley and Robinson 1996, Findlay and Houlahan 1997, Trombulak and Frissell 2000, Glista et al. 2007). Reptiles also exhibit migratory behaviors that increase suspectibility to road mortality, including movements related to fluctuations in water level (Bernardino and Dalrymple 1992, Aresco 2003, Smith and Dodd 2003), adult males searching for mates (Bonnet et al. 1999, Whitaker and Shine 2000), nesting migrations of adult females in the spring (Fowle 1996, Bonnet et al. 1999, Haxton 2000, Baldwin et al. 2004), and neonatal dispersal during late summer or early autumn (Bonnet et al. 1999, Enge and Wood 2002, Smith and Dodd 2003).

Several other behaviors and characteristics may also increase susceptibility of herpetofauna to road-related mortality. For example, some species of snakes may be attracted to road surfaces to thermoregulate (Klauber 1939, McClure 1951, Sullivan 1981, Rosen and Lowe 1994, Ashley and Robinson 1996) or scavenge from carcasses (Smith and Dodd 2003), and some species of toads may use roads under streetlights to forage for insects (Neill 1950). Other studies have demonstrated that female turtles may be attracted to roads for nesting purposes (Wood and Herlands 1997, Marchand and Livatis 2004). Many species of snakes present a relatively large target as they crawl across roadways, which may affect the frequency of intentional killing (Whitaker and Shine 2000) or collecting by humans (Dodd et al. 1989).

Roads also pose a suite of indirect threats on amphibians and reptiles, including barrier effects, edge effects, pollution, and sedimentation (Seigel 1986, Dalrymple and Reichenbach 1984, Kjoss and Litvaitis 2001). For example, disappearance of populations of the relictual slender salamander (a petitioned species) in the lower Kern River Canyon likely resulted from habitat changes caused during construction of State Route 178, as road margin habitat has been severely degraded by road maintenance and related construction activities (Hansen and Wake 2005).

Finally, roads can threaten herpetofauna through road avoidance (see Weatherhead and Prior 1992, Gibbs 1998a, Fitch 1999, Sealy 2002, Shine et al. 2004, Andrews 2004). In particular, barrier effects from roads have been observed in terrestrial and aquatic salamanders causing habitat fragmentation and isolation of populations (Gibbs 1998, DeMaynadier and Hunter 2000, Forman and Deblinger 2000, Jones et al. 2000, Marsh and Beckman 2004, Marsh et al. 2004, Marsh et al. 2005, Cushman 2006, Semlitsch et al. 2007). Correlations between road density and genetic distance in amphibians have been found (Reh and Seitz 1990, Hitchings and Beebee 1996, Boarman et al. 1997).

Figure 7. Road mortality threatens over half of the petitioned species of amphibians and reptiles.

8. Physical Alteration of Aquatic Habitats

Humans have drastically altered freshwater habitats (EPA 2004, Strayer 2006), which is harming aquatic herpetofauna across the United States, including 19 of the petitioned species. The threats posed by impoundments, dredging and channelization, and water loss are discussed below.

a. Impoundments

Impoundment is a threat to amphibians, turtles, and other species dependant on aquatic systems (Kupferberg 1994, Koch et al. 1996, Maxwell and Hoyitt 1999), including 12 of the petitioned species. Scientists have established that impoundments impact herpetofauna through the alteration of water level and flow patterns (Richter and Azous 1995, Delis et al. 1996, Riley et al. 2005, Schoonover et al. 2006). Dams modify habitat conditions and aquatic communities both upstream and downstream of the impoundment (Mulholland and Lenat 1992, Soballe et al. 1992, Neves et al. 1997). Downstream of dams, flow regime fluctuations cool water temperature and affect dissolved oxygen levels, scouring the substrate and eroding downstream tributaries (Schuster 1997, Buckner et al. 2002). Colder water temperatures may increase mortality by decreasing larval growth rates and causing decreased immunity (Wilbur 1980, Nyman 1986, Carey 1993, Maniero and Carey 1997). Dams also fragment habitat by blocking corridors for migration and dispersal, resulting in population isolation and heightened susceptibility to extinction (Neves et al. 1997).

In addition, manipulation of water levels in water impoundments can destroy habitat and result in direct and indirect mortality of amphibian larvae and eggs. For example, the construction of Shasta Dam created Shasta Lake, which submerged key habitats for the Shasta salamander (a petitioned species) and caused population declines (Hansen and Papenfuss 1994, Wake and Papenfuss 2005, NatureServe 2011).

Another petitioned species, the Arizona toad, is now absent from historical localities where the riparian corridor has been altered dramatically through the construction of impoundments (Sullivan 1986, 1993). As another example, Lind et al. (1996) found that reduced water flows below dams on the Trinity River in California resulted in the loss of floodplain breeding pools and vegetational overgrowth of riparian areas used for basking and foraging by amphibians and reptiles, including the western pond turtle and foothill yellow-legged frog, which are petitioned species. To be sure, impoundments are a major threat to turtle populations, especially in the Southeast where turtle richness is high but there are few major rivers that have not been impounded (Shute et al. 1997).
.

b. Dredging and channelization

Dredging and channelization is another threat to amphibians and reptiles, harming six of the petitioned species (about 10 percent). Many rivers are continually dredged to maintain a channel for shipping traffic (Abell et al. 2000). Dredging and channelization modify and destroy habitat for aquatic species by destabilizing the substrate, increasing erosion and siltation, removing woody debris used for basking, decreasing habitat heterogeneity, and stirring up contaminants that settle onto the substrate (Buckner et al. 2002, Bennett et al. 2008). In particular, dredging and channelization are contributing to the decline of turtles (Buhlmann and Gibbons 1997), including these petitioned species: alligator snapping turtle, Blanding's turtle, wood turtle, and Rio Grande cooter.

c. Water Diversion and Decreased Water Availability

The diminishing availability of freshwater poses a present and increasing threat to amphibians, turtles and other aquatic species in the United States (Benz and Collins 1997, Buckner et al. 2002, Herrig and Shute 2002, Hutson et al. 2005, Lysne et al. 2008), including six of the petitioned species. Human population growth is increasing demand for freshwater resources (Postel 2000, Jackson et al. 2001, Strayer 2006). Increasing drought due to global climate change is also expected to exacerbate the threat of limited water availability to aquatic and riparian species (Karl et al. 2009).

Surface diversion of streams is a particular threat to amphibians and reptiles (Abell et al. 2000, Buckner et al. 2002, Herrig and Shute 2002). Besides the loss of aquatic habitats, reduced water volume also increases the concentration of pollutants (Abell et al. 2000, Herrig and Shute 2002). An additional threat is groundwater overdraft, which threatens spring flow and species that are dependent on consistent spring flow conditions (Herrig and Shute 2002, Strayer 2006, Deacon et al. 2007). Plus, many springs have been drastically altered to supply water for human uses (Etnier 1997). Spring development and diversion can alter flow regime and water quality parameters, lead to substrate disturbance and erosion, and alter the structure and composition of vegetative cover with effects on freshwater herpetofauna (Shepard 1993, Frest and Johannes 1995, Frest 2002). For example, loss of spring flows from groundwater overdraft is a primary threat to the Cascade Caverns salamander, one of the petitioned species.

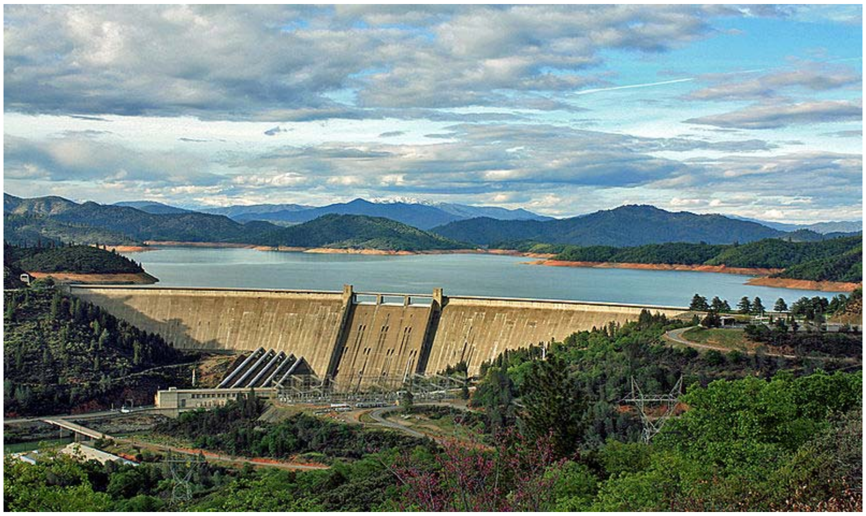


Figure 8. Water impoundments, such as the Shasta Dam pictured here, can flood key habitats for amphibians and reptiles. Photo by Apaliwal.

B. OVERUTILIZATION FOR COMMERCIAL, RECREATIONAL, SCIENTIFIC, OR EDUCATIONAL PURPOSES

Worldwide collection and harvest of amphibians and reptiles for food, sport, and commerce is extensive (Maxwell and Hokit 1999, Schlaeper et al. 2005, Crump 2011) and is a threat to 20 petitioned species. Hundreds of millions of herpetofauna are removed from the wild and killed each year for these activities, and annual worldwide commerce in herpetofauna may be valued in the hundreds of millions, perhaps even billions, of dollars (e.g., Williams 1995, Wilkinson 1996b, Buck 1997, Pough et al. 1998). Many amphibian and reptile species predictably aggregate in small areas during breeding or hibernation, making them particularly vulnerable to intensive harvest efforts (Klemens and Thorbjarnarson 1995, Milner-Gulland 2001).

Throughout the United States, amphibians and reptiles are exploited for all these same reasons, which may cause significant population declines (Emmons 1973, Jennings and Hayes 1985, Salzberg 1995, Williams 1995, Buhlmann and Gibbons 1997, Dodd 1997, Gibbons 2000, Herrig and Shute 2002, Schlaepher et al. 2005, Means 2009). The threats posed by overharvest of turtles, the pet trade, and scientific/educational collection are discussed below. In addition, nearly all of the petitioned snakes are threatened by wanton or malicious killing by those that perceive snakes as a threat.

1. Overharvest of Turtles

Overharvest is a primary threat to all of the petitioned turtles, and population declines occur when adult turtles are harvested (Brooks et al. 1991; Heppell 1998; Congdon et al. 1993, 1994). To be sure, overharvest has caused population declines in almost all turtle species that are now extinct, critically endangered, or rare (Klemens and Thorbjarnarson 1995). The trade of freshwater turtles destined for the Southeast Asian market has been well documented (van Dijk et al. 2000).

Turtles are characterized by a suite of life history characteristics that predispose populations to rapid declines in the face of harvest. Among these characters are delayed maturity, high annual survivorship of adults, and high natural levels of nest mortality (Reed and Gibbons 2003). Stable turtle populations are dependent on sufficiently long-lived breeding adults to offset the effects of high egg and nestling mortality and delayed sexual maturity (Wilbur and Morin 1988, Congdon et al. 1993). Removing even a few adults from a population can have effects lasting for decades because each adult turtle removed eliminates the reproductive potential over a breeding life that may exceed 50 years. As such, scientists warn that freshwater turtles cannot sustain any significant level of harvest from the wild without leading to population crashes (Congdon et al. 1993, 1994; Heppell 1998; Reed et al. 2002; van Dijk 2010).

As an example, the alligator snapping turtle – one of the petitioned species and one of the largest turtles in the world – historically faced very high levels of collection for food and pets (Roman et al. 1999). Consequently, the species has been drastically reduced in numbers in the southeastern rivers it once inhabited (Moler 1992, Jensen 1998). Reed et al. (2002) found that the removal of as few as two female adult alligator snapping turtles could halve a population of 200 turtles within 50 years (see also Congdon et al. 1994).

2. Pet Trade

The pet trade is a significant threat to amphibians and reptiles across the United States, and threatens 19 of the petitioned species, which is 35 percent. More of the petitioned species are likely threatened by the pet trade but the threat is documented for just these 19 species. Millions of amphibians and reptiles are being kept as pets in the United States and commercial interest in amphibians and reptiles has grown rapidly (Hoover 1998, Franke and Telecky 2001, Crump 2011). Because there is no way (short of DNA finger-printing) to know whether the animal is captive bred, commercial dealers have been known to supplement their legal captive born stock with wild-caught specimens. As an example, the Apalachicola kingsnake is threatened by private and commercial collecting (NatureServe 2011); this beautiful snake is regularly offered for sale on the Internet (see, e.g., Reptiles-n-critters.com, I-herp.com, thegrandreptile.weebly.com). Similarly, the southern hog-nosed snake is prized by the pet industry (Nickel 2010), and collection for the pet trade is likely a threat (Enge and Wood 2003).

3. Scientific and Educational Collecting

Amphibians and reptiles are in great demand for medical and biological research, and are widely used in teaching for dissections and demonstrations (Crump 2011). Researchers and teachers generally buy amphibians and reptiles from biological supply houses, which buy the animals from people who make a living by collecting them from the wild (Nace et al. 1979, Crump 2011). For example, collection of wood turtles (included in this petition) by biological supply houses likely led to population declines and extirpations, particularly in Wisconsin (Harding and Bloomer 1979, Vogt 1981). In addition, overcollection for scientific study is a threat to some plethodontid salamanders, including the Caddo Mountain salamander, one of the petitioned species (Jensen and Camp 2005, Huston 2009), as well as the Florida scrub lizard, another petitioned species (Enge et al. 1986).


Figure 9. The petitioned turtles are especially vulnerable to overexploitation for the pet trade. Photo by Vmenkov.

C. DISEASE AND PREDATION

1. Amphibian Diseases

Disease has been implicated as a factor in the decline of amphibian populations worldwide (Bradford 1991, Blaustein et al. 1994b, Laurance et al. 1996, Kiesecker and Blaustein 1997, Berger et al. 1998, Daszak et al. 1999, Daszak et al. 2000, Fellers et al. 2001, Kiesecker et al. 2001, Kiesecker et al. 2004, Briggs et al. 2005). Disease is also a threat to several of the petitioned amphibian species, including the Carolina gopher frog, foothill yellow-legged frog, and Cascades frog. Disease likely works synergistically with other threats to amphibians (Fellers et al. 2001, Kiesecker at al. 2001). In some cases, sublethal environmental stressors may suppress immune systems (Carey 1993) and allow disease agents to kill weakened animals (Alford and Richards 1999).

Numerous diseases are contributing to amphibian declines. These include infections of chytrid fungus (*Batrachochytrium dendrobatidis*) and white fungus (*Saprolegnia ferax*), ranaviruses, bacterial infections, and trematodes (Dodd 1997, Daszak et al. 1999, Briggs et al. 2005, Davis et al. 2007, Peterson et al. 2007). The range of bacteria reported to cause disease in amphibians is small, but red leg disease is the one bacterial disease associated with significant mortality in wild amphibians (Bradford 1991, Carey 1993, Hogrefe et al. 2005, Densmore and Green 2007) and could pose a future threat to some of the petitioned amphibian species, such as the foothill yellow-legged frog. Trematode infestation has been implicated in limb deformities in several species of amphibians (Johnson et al. 1999, Johnson et al. 2002) but is not known to currently threaten any of the petitioned species. The three biggest pathogenic threats to the petitioned species are chytrid fungus, white fungus, and ranaviruses (Carey et al. 2003, Collins et al. 2003b), and each of these is discussed below.

a. Chytrid Fungus

Chytrid fungus has been implicated as a primary or suspected cause of disease epidemics and subsequent population declines of amphibians in many parts of the world (Berger et al. 1998, Lips 1998, Young et al. 2001), including the United States (Fellers et al. 2001). Chytrid fungus affects not only frogs and toads but has also been reported in both aquatic and terrestrial salamanders (Davidson et al. 2003, Cummer et al. 2005, Padgett-Flohr and Longcore 2007, Byrne et al. 2008).

Two of the petitioned species, the Cascades frog and the foothill yellow-legged frog, are presently threatened by chytrid. Frogs infected with chytrid have been found in the wild in both species (Lowe 2007, Gaulke et al. 2011, Piovia-Scott et al. 2011. While laboratory studies have demonstrated the harmful effects of the fungus on these species (Garcia et al. 2006, Davidson et al. 2007, Piovia-Scott et al. 2011), population effects in the wild are unknown (Fellers 2005), as many infected frogs appear asymptomatic (Gaulke et al. 2011) and many extant populations appear to be coexisting with the pathogen (Piovia-Scott et al. 2011, Reeder et al. 2012). Chytrid fungus is also a potential threat to the Carolina gopher frog because it has been detected in Mississippi gopher frogs, which are a related species (USFWS 2009a,b).

Chytrid attacks the keratin and skin of amphibians and has caused 90-100 percent mortality rates in metamorphosed amphibians (McGee and Keinath 2004). Adult amphibians infected with chytrid exhibit symptoms such as lethargy and reluctance to flee, skin abnormalities, loss of righting reflex, and extended back legs (Fellers et al. 2001, National Wildlife Health Center 2001). In tadpoles infected with chytrid fungus, jaw sheaths and tooth rows are abnormally formed or lack pigment, and this type of deformity likely inhibits tadpole foraging ability (Fellers et al. 2001). The exact mechanism of chytrid is not well understood (Carey 1993, Berger et al. 1998, Pessier et al. 1999, Sredl 2000). Voyles et al. (2012) found that infection by the fungus seems to disrupt balance of fluids and electrolytes, which are minerals found in the blood that are crucial for muscle function, proper blood pH, and hydration.

b. White Fungus

Outside of chytrid, the primary fungal agent impacting amphibians is white fungus or *Saprolegnia ferax* (Blaustein et al. 1994, Kiesecker and Blaustein 1997, Kiesecker et al. 2001). *Saprolegnia* is commonly carried by fish and may be introduced to amphibian habitats via sport fish stocking (Kiesecker et al. 2001). Once introduced to a system, individual amphibians may transmit the pathogen to other populations as they migrate or disperse. *Saprolegnia ferax* is suspected in the decline of foothill yellow-legged frog (*Rana boylii*), which is one of the petitioned species (Blaustein et al. 1994, Kiesecker and Blaustein 1997).

c. Ranaviruses

Ranavirus is the name of one of the genera in the family Iridovirus that contains emerging disease pathogens with the potential to affect fish, amphibians, and reptiles (Collins et al. 2003b).

In the United States, ranaviruses have been associated with mass mortality in the federally listed Sonoran tiger salamanders (*Ambystoma tigrinum stebbinsi*) and several other amphibians (Collins et al. 1988, Cunningham et al. 1996, Jancovich et al. 1997, Chinchar 2002, Schumacher 2006). Ranaviruses also threaten the Carolina gopher frog, one of the petitioned species. A die-off of hundreds of ranid tadpoles, including gopher frogs, in two ponds in Withlacoochee State Forest, Hernando County, Florida, was apparently caused by a ranavirus (Davis et al. 2007, Rothermel et al. 2008). In addition, a newly identified mesomycetozoan pathogen, *Anuraperkinsus emelandra*, has been the cause of massive ranid tadpole mortalities in 10 states, including a 2003 die-off of almost all tadpoles at a breeding pond of the federally endangered Mississippi gopher frog (FFWCC 2011). The foothill yellow-legged frog – because it is also a ranid frog – is a second petitioned species susceptible to ranaviruses.

2. <u>Reptilian Diseases</u>

Disease threatens reptile species too (Schumacher 1996, Diemer Berish et al. 2000, Gibbons et al. 2000). For example, shell diseases have been implicated in the decline of turtles (e.g., shell lesions on sliders, Lovich et al. 1996; cutaneous abnormal keratinization affecting the shell and thickened forelimb scutes of desert tortoises, Jacobson 1994; and emaciation and lesions of the plastron of federally listed flattened musk turtles, *Sternotherus depressus*, Dodd 1988).

Decline of one of the petitioned species, the western pond turtle, has been caused by disease and is a threat. An upper respiratory disease epidemic in Washington in 1990 left a total population of fewer than 100 western pond turtles in the wild (Andelman and Gray 1992). Observations have suggested the potential occurrence of a similar disease syndrome in one northern California population of the turtle (Jennings and Hayes 1994).

Recently, a soil-dwelling fungus, *Chrysosporium*, has been diagnosed as the cause of lethal facial lesions on eastern massasauga snakes (*Sistrurus catenatus catenatus*) (Parry 2012). Scientists fear that this fungus could "represent a new and devastating superbug," that threatens many imperiled reptile species (Parry 2012), including the petitioned species.

Figure 10. A chytrid-infected frog. Photo by Forrest Brem.

3. <u>Predation</u>

Amphibians and reptiles are prey for a variety of organisms but such natural predation is not usually a threat. However, human-subsidized mesopredators, such as raccoons and skunks (Bury and Germano 2008), prey upon and threaten many of the petitioned species, including the Key ringneck snake, Florida pine snake, Arizona toad, as well as all of the turtles (see Christiansen and Gallaway 1984, Browne and Hecnar 2007). The mesopredators can also drive smaller animals to extinction, affecting the petitioned species by causing cascading ecological effects on the other trophic levels in these systems (Crooks and Soule 1999, Henke and Bryant 1999, Marzluff et al. 2001).

The following examples demonstrate the harmful impacts of human-subsidized predators on the petitioned species. Ross and Anderson (1990) found that all the Blanding's turtle nests on their study site in Wisconsin were destroyed by predators; 75 percent were destroyed when first discovered and the remaining nests were destroyed within 24 hours of nesting. Similarly, predation by raccoons accounts for the loss of a majority of alligator snapping turtle eggs in Florida (about 2/3 along the lower Apalachicola River) (Florida Fish and Wildlife Conservation Commission 2011). Brooks et al. (1992) found that 60 percent of adult wood turtles and about 30 percent of juveniles bore wounds from predatory attempts. Schwaner and Sullivan (2005) observed that raccoons (*Procyon lotor*) commonly consume Arizona toads, a petitioned species, during the breeding season.

The threats posed by introduced species are discussed below in the "Other Factors" section.

D. INADEQUATE REGULATORY MECHANISMS

There are no existing regulatory mechanisms at the federal, state, or regional levels that adequately protect the petitioned species, all of which are at risk of extinction and would benefit from the protections of the ESA.

1. <u>The Clean Water Act</u>

Pollution and habitat loss are two of the largest threats facing the petitioned species, many of which depend on healthy riparian and aquatic habitats for survival. The federal Clean Water Act ("CWA"), 33 U.S.C. § 1251 et seq. (1972), provides a basic level of water quality and wetland protection for aquatic species, but it is inadequate to ensure their continued survival without the addition of Endangered Species Act protection. As explained below, pollution from point and non-point sources is causing ongoing degradation of water quality and loss of wetland habitat continues.

a. Water pollution

Although the CWA resulted in an overall gain in water quality, degraded water quality still is a significant factor affecting aquatic organisms, including many of the petitioned species, because a number of activities responsible for habitat degradation are outside of regulatory oversight.

Nonpoint pollution sources (for example, animal and human waste, agricultural practices, road construction, logging, oilfields) are causing degraded water quality and are generally approached in a nonregulated, voluntary manner. For example, a multitude of waterways within the range of the wood turtle (a petitioned species) are polluted or significantly degraded by runoff by agricultural and industrial areas (see, e.g., Jones et al. 1997).

The Environmental Protection Agency and individual states regulate point sources of pollution with the National Pollution Discharge Elimination System (NPDES). The NPDES system is not adequate to protect the petitioned species from the negative effects of point source pollution because permits may be issued with few restrictions, cumulative effects of all the point sources within a watershed are not taken into consideration when permits are issued, and state governments often lack the resources or political will to monitor and enforce permits (Morse et al. 1997, Buckner et al. 2002). Moreover, even if existing laws were strictly enforced, current water quality standards are not sufficient to protect sensitive species or sensitive life-stages of species. Water-quality standards are not based on toxicity testing of rare species, and some aquatic organisms are more sensitive to pollutants than the organisms that are used to establish the standards (Herrig and Shute 2002).

b. Loss of Wetlands

One of the greatest threats to amphibians and reptiles in North America is the loss of wetland areas, especially small, temporary or isolated wetlands (Buhlmann and Gibbons 1997, LaClaire 1997, Semlitsch and Brodie 1998, Trauth 2006). Tile and open-ditch drainage of wetland areas, once considered a conservation practice, was prevalent from the mid-1950s to the mid-1970s and resulted in the conversion of millions of hectares of wetland to farmland (Dahl and Allord 1997, Crump 2011). Continuing wetland conversion through agriculture or urban expansion invariably results in habitat loss and severe changes in water regimes (Rubec et al. 1988, Wilen and Frayer 1990), and puts amphibian and reptile populations at risk (Gibbs 1993). Loss of wetlands is a documented threat to six petitioned species (more than ten percent of the petitioned species).

Under the CWA, actions that result in discharge of dredged or fill material in wetlands and certain other waters require a permit from the United States Army Corps of Engineers. However, despite the protections of the CWA, the permitted filling of wetlands is ongoing. As an initial matter, many small or isolated wetlands that provide essential habitat for many herpetofaunal species are exempt from section 404 (Semlitsch and Bodie 1998). But even wetlands regulated under section 404 are lost. Section 404 sets a goal of no net loss of wetlands, but this is not a required outcome of permit decisions (Connolly et al. 2005). The Corps still has the authority to issue general permits on a state, regional, or nationwide basis for activities in wetlands that the agency determines will have minimal effects on wetland habitat (Mitsch and Gosselink 2000). For example, in fiscal year 2003, the Army Corps of Engineers issued 4,035 permits for the destruction of natural wetlands, while denying only 299 permits (Connolly et al. 2005). Permits by the Corps, however, may not authorize activities that will jeopardize the continued existence of a threatened or endangered species or destroy or adversely modify its critical habitat, which is why ESA protection for the petitioned species is necessary (USACE 2001).

Lost wetlands are required to be replaced by mitigation wetlands, but mitigation wetlands often differ in structure, function, and community composition from the natural wetlands which are destroyed (Holland et al. 1995). Mitigation requirements are also not strictly enforced. Mitigation "represents a promise that the permittees will perform the mitigation in the future. Unfortunately, permittees are often unable or unwilling to comply with compensatory mitigation requirements" (Connolly et al. 2005, p. 262). As such, mitigation is rarely effective in preserving biodiversity (Cubbage et al. 1993, Water Environment Federation 1993).

In sum, the Clean Water Act is not adequate to protect the petitioned species from the threats of habitat loss and degradation and pollution.

2. Management of Federal Lands

Habitat protection is essential to conserve the petitioned species. Many lands in federal ownership are protected from development, such as refuges, recreation areas, and national forests. These protected lands alone, however, are insufficient to protect imperiled species due to threats from a host of other factors including climate change, poaching, pollution, and genetic isolation due to lack of habitat connectivity. For example, Browne and Hecnar (2007) document the decline of spotted turtles (a petitioned species) in protected habitat due to low-level recreational collection. They conclude, "Our study illustrates that habitat protection provides no guarantee for species persistence when multiple threats exist." In addition, management of federal lands for timber, recreation, mining, and other activities is often not compatible with the habitat needs of wildlife. The inadequacy of the National Wildlife Refuge System, military lands, and national forests/BLM lands to protect the petitioned species is discussed below.

a. National Wildlife Refuge System

The Department of the Interior, through the FWS, administers the National Wildlife Refuge System. The National Wildlife Refuge System Administration Act of 1966 (NWRAA; 16 U.S.C. §§ 668dd–668ee) provides legislation for the administration of a national network of lands and water for the conservation, management, and restoration of fish, wildlife, and plant resources and their habitats for the benefit of the American people. Each refuge must implement a comprehensive conservation plan ("CCP"). The CCP must identify and describe the wildlife and related habitats in the refuge and actions needed to correct significant problems that may adversely affect wildlife populations and habitat. 16 U.S.C. § 668dd(e).

Several of the petitioned species are found within wildlife refuges, including the Key ringneck snake, Blanding's turtle, Cedar Key mole skink, western spadefoot, Rim Rock crowned snake, and Yuman Desert fringe-toed lizard. Habitat of amphibians and reptiles within national wildlife refuges is protected from loss due to urban development. However, these species are still threatened with extinction or endangerment for several reasons. Although refuges are managed under conservation plans that provide guidance for planning and management decisions, the plans do not constitute a commitment for staffing or funding. Refuge budget and staffing levels are usually inadequate to implement preferred management actions, and management priority is generally given to more charismatic species. Species that occur on refuges also face threats from historical habitat degradation, climate change, invasive species, recreation, and poaching.

b. *Military Lands*

On military installations, the Department of Defense ("DOD") must conserve and maintain native ecosystems, viable wildlife populations, listed species, and habitats as vital elements of its natural resource management programs, to the extent these requirements are consistent with the military mission (DOD Instruction 4715.3). Amendments to the Sikes Act (16 U.S.C. § 670 et seq.) require each military department to prepare and implement an integrated natural resources management plan ("INRMP") for each installation under its jurisdiction.

The effectiveness of individual INRMPs to protect amphibians and reptiles vary between and within military departments. Yet when considered together, the INRMPs do not provide adequate regulatory mechanisms to prevent extinction of the petitioned species because these protected lands constitute just a fraction of the habitats needed by these species. Moreover, most of the INRMPs do not have specific management direction aimed at preserving the petitioned species and are largely voluntary. For example, the Yuman Desert fringe-toed lizard is found on land within the Barry M. Goldwater Air Force Range, which is subject to monitoring to help protect the desert ecosystem (Villarreal et al. 2011). However, any efforts to protect the lizard on this land are purely voluntary and therefore inadequate.

c. *National Forests and BLM Lands*

Many of the petitioned species occur on National Forests or lands managed by the Bureau of Land Management ("BLM"). The Federal Land Policy Management Act of 1976 (43 U.S.C. § 1701 et seq.) and the National Forest Management Act of 1976 (16 U.S.C. § 1600 et seq.) direct Federal agencies to prepare programmatic-level management plans to guide long-term resource management decisions on the lands they manage. But these plans do not adequately protect the petitioned amphibians and reptiles. To begin, the petitioned species are not often designated as sensitive species or management indicator species so specific planning guidance is not provided. Even when the petitioned species are specifically addressed in planning documents, many of these guidelines are discretionary. Plus, national forests are managed pursuant to multiple use mandates and activities (such as mining, timber, grazing, and recreation) that are often incompatible with the protection of imperiled wildlife.

In some cases, agencies effectively manage federal lands to protect the petitioned species but such lands make up just a fraction of the species' range. Additionally, many activities that affect the petitioned species and their habitats are beyond Forest Service or BLM control. For instance, the Forest Service does not have the authority to regulate off-site activities such as pesticide applications that may be responsible for amphibian and reptile declines, and the Forest Service has only limited ability to regulate introductions or stockings of nonnative species that prey on the petitioned species. Moreover, despite extensive planning efforts by federal land managers and implementation of management actions, loss of the petitioned species continues.

3. The Wild and Scenic Rivers Act

The Wild and Scenic Rivers Act of 1968 protects selected rivers in free-flowing condition and protects their immediate environments to safeguard water quality and to fulfill national conservation purposes. Wild and Scenic designation provides some protection for the species that occur within these reaches, such as some populations of the western pond turtle. It does not adequately protect the petitioned species, however, because there are very few designated Wild and Scenic stretches, they do not provide habitat protection beyond a narrow corridor, and because many of the areas of highest aquatic biodiversity are not included in the system (Neves et al. 1997).

4. National Environmental Policy Act

The National Environmental Policy Act of 1969 (NEPA) (42 U.S.C. §§ 4321– 4370a) requires Federal agencies to consider the environmental impacts of their actions. NEPA requires Federal agencies to describe the proposed action, consider alternatives, identify and disclose potential environmental impacts of each alternative, and involve the public in the decision-making process. Many actions taken by the Forest Service, the Bureau of Land Management, and other Federal agencies that affect the petitioned species (such as logging) are subject to NEPA. NEPA is inadequate to protect the petitioned species, however, because federal agencies are not required to select the alternative having the least significant environmental impacts. The agency may select an action that will adversely affect sensitive species provided that these effects were known and identified in a NEPA document.

5. State Fish and Wildlife Departments

Nanjappa and Conrad (2012) provide a summary of state laws affecting herpetofauna. Many of the petitioned species are listed as threatened or endangered by state wildlife agencies, but state endangered and threatened species designations generally do not provide species with meaningful regulatory protection. And only California affords some protection against habitat destruction for state-listed wildlife.

In addition, many of the species are classified as Species of Conservation Priority or Species of Greatest Conservation Need under state Wildlife Action Plans or Wildlife Conservation Strategies. These documents provide a framework for conservation, but are not regulatory documents and do not contain mandatory or enforceable provisions to protect the species or their habitat. For example, the Florida scrub lizard (a petitioned species) is considered a species of greatest conservation need in Florida (FFWCC 2005), but current law allows unlimited collection of the lizards (Nanjappa and Conrad 2012). Further, the implementation of conservation strategies is dependent on the cooperation of resource managers and stakeholders, making their effectiveness uncertain. State conservation priorities and initiatives are also sharply limited by funding, with charismatic and game species generally receiving the majority of resources.

Some states have rules that regulate the take of some of the petitioned species, but these rules are not comprehensive, are generally poorly enforced, and are not adequate to protect wildlife from other threats (USFWS 1997). For example, no permit is required to collect native amphibians and reptiles for personal use in Virginia with a daily bag limit of five animals (Nanjappa and Conrad 2012), and under these regulations the Peaks of Otter salamander faces high collection pressure at some sites (NatureServe 2011).

6. Lacey Act

Under section 3372(a)(1) and section 3372(a)(2)(A) of the Lacey Act Amendments of 1981 (16 U.S.C. § 3371–3378), it is unlawful to import, export, transport, sell, receive, acquire, or purchase any wildlife taken, possessed, transported, or sold in violation of law. However, the effectiveness of the Lacey Act is limited because of poor enforcement. For example, wildlife inspectors are often unable to identify amphibians and reptiles to the species level and therefore are unable to determine whether the specimen is a member of a prohibited taxon. Additionally, because collection of some of the petitioned species is allowed in some states but prohibited in others, illegal collectors can misrepresent the state of origin to get around the prohibition of the Lacey Act. For example, take of wood turtles is prohibited across their range, except in Maine, where wood turtles have no protection (Maine Dept. of Inland Fisheries and Wildlife 2010), and as such, state protection appears to have "done little to curb collection of this species" (NatureServe 2011). Thus, although the prohibitions and penalties of the Lacey Act Amendments of 1981 provide some protection for the petitioned species, this law, by itself, does not adequately prevent their illegal commercial trade.

7. Convention on International Trade in Endangered Species

The Convention on International Trade in Endangered Species (CITES) is an international agreement between governments that aims to ensure that the international trade of wild animals and plants does not threaten their survival. Species protected under CITES are listed in one of three appendices. Trade is generally prohibited under Appendix I, which includes species in danger of extinction. Trade is permitted at levels that do not threaten the survival of the species listed under Appendix II. Under Appendix III, trade is not restricted but is permitted and closely monitored.

CITES conveys some degree of protection to two of the petitioned turtle species, the wood turtle and the alligator snapping turtle, but is inadequate to ensure their continued survival. The alligator snapping turtle is threatened by the international pet trade despite being protected under Appendix III (USFWS 2010). And the wood turtle, which is listed on Appendix II, is still threatened by international trade because many collectors misrepresent wild caught turtles as captive bred. In addition, several petitioned species deserve protection under CITES but do not currently receive it, including the spotted turtle and Blanding's turtle (Adkins Giese 2011).

In sum, existing regulatory mechanisms are not adequate to protect the petitioned species. Without the effective protection of the Endangered Species Act, these species are likely to become endangered or extinct.

E. OTHER FACTORS

1. Exotic Species

Introduced species affect amphibians and reptiles – including 19 of the petitioned species – through direct predation, competition, disease introduction, and ancillary effects of control actions (Dodd 1997, Jolly et al. 2010). Moreover, recent studies have found that other factors of decline, such as habitat modification (Adams 2000), chemical contaminants (Relyea Rick and Mills 2001), UV-B radiation (Kats et al. 2000), and disease (Kiesecker et al. 2001) work synergistically to exacerbate the negative effects of introduced species.

The spread of invasive species is an indirect effect of urbanization. Invasive species tend to become more common in urban environments often because of human releases, and these nonnative species out-compete and over-consume native plants and animals (Loewenstein and Loewenstein 2005, Riley et al. 2005, Seabloom et al. 2006, Jodoin et al. 2008). For example, urbanization results in increasing numbers of cats, which are known to consume large numbers of native snakes and lizards (McMurry and Sperry 1941, Parmalee 1953, Whitaker and Shine 2000), such as the short-tailed snake, one of the petitioned species (Highton 1956, Godley et al. 2008). As another example, surveys have shown that introduced turtles are increasing in abundance in urban areas and may contribute to population declines of native turtles (Bury 2008), including the western pond turtle, another petitioned species (Spinks et al. 2003, Patterson 2006).

The threat of invasive species is expected to increase in the future as the climate warms and as habitat availability shrinks. Even taxa which are not currently threatened by invasive species are expected to disappear due to future biological invasions as species adjust their ranges and humans continue to accidentally and intentionally transport nonnative species (Ricciardi and Rasmussen 1998). Details on the threats posed by introduced fishes, bullfrogs, fire ants, and invasive plants are discussed below.

a. Introduced Fishes

The introduction of fish into historically fishless habitats is widely recognized as a threat to amphibians (e.g., Maxwell and Hokit 1999, Lawler et al. 1999, Goodsell and Kats 1999, Knapp and Matthews 2000, Pilliod and Peterson 2001, Kats and Ferrer 2003, Vredenburg 2004, Knapp 2005, Welsh et al. 2006, Welsh et al. 2006), and is a threat to 15 percent of the petitioned species.

Egg, larval, and adult amphibians may be subject to direct predation by introduced warm and cold water fishes (e.g., Korschgen and Baskett 1963, Licht 1969, Simons 1988, Semlitsch and Gibbons 1988, Liss and Larson 1991, Vredenburg 2004). Plus, all three amphibian life history stages are likely to be indirectly effected by the threat of predation due to (1) adult avoidance of oviposition sites where predators are present (e.g. Resetarits and Wilbur 1989, Hopey and Petranka 1994), (2) decreased larval foraging and, therefore, growth rates as a result of staying in refuges to avoid predators (e.g., Figiel and Semlitsch 1990, Skelly 1992, Kiesecker and Blaustein 1998, Tyler et al. 1998), and (3) decreased adult foraging, growth rates, and overwinter survival as a result of avoiding areas with fishes (e.g., Bradford 1983). In addition, introduced fish compete with amphibians for their invertebrate source of prey (Herrig and Shute 2002, Kats and Ferrer 2003, Strayer 2006). As one example, Joseph et al. (2010) suggest that reductions in the availability of emerging aquatic insects cause Cascades frogs (*Rana cascadae*), one of the petitioned species, to consume more terrestrial prey where trout are present.

While impacts of introduced fishes on reptiles is not as severe, younger age classes of reptiles are likely to be directly preyed on by nonindigenous fish and are also likely to be negatively affected indirectly as a result of the loss of amphibians, which they depend on as prey (e.g., Jennings et al. 1992, Koch et al. 1996). The western pond turtle is one of the petitioned reptile species threatened by introduced fish; bass (*Micropterus* spp.) are known to prey on the smallest juveniles (Holland 1991a).

In addition, commercial piscicides (e.g. rotenone) have often been used to remove unwanted fish stocks from a variety of aquatic habitats (Schnick 1974), and these chemicals are known to be toxic to amphibians and reptiles. The impacts of rotenone-containing piscicides on amphibians and turtles were reviewed by Fontenot et al. (1994) and McCoid and Bettoli (1996). Another piscicide, antimycin, is also likely toxic to turtles and amphibian larvae (Patla 1998).

b. Bullfrogs

Bullfrogs (*Rana catesbeiana*) have been implicated in the declines of a number of amphibian and reptile species (Moyle 1973, Hammerson 1982, Bury and Whelan 1984, Kupferberg 1994, Rosen et al. 1995, Kupferberg 1997, Lawler et al. 1999). They are a threat to four of the petitioned species: western pond turtle, western spadefoot, Illinois chorus frog, and foothill yellow-legged frog. Bullfrogs are native to the eastern United States (Bury and Whelan 1984), but they have now been widely introduced into permanent waters in all lower forty-eight states, with the possible exception of North Dakota. Their large size, high mobility, generalized eating habits, and huge reproductive capabilities have made bullfrogs extremely successful invaders and a threat to biodiversity (AmphibiaWeb 2012).

All three life history stages of amphibians, as well as smaller aquatic reptiles, may be subject to direct predation by adult bullfrogs (Korschgen and Baskett 1963, Carpenter and Morrison 1973, Bury and Whelan 1984, Clarkson and DeVos 1986). Additionally, both the eggs and larvae of native amphibians may be preyed upon by larval bullfrogs (e.g., Ehrlich 1979, Kiesecker and Blaustein 1997b). As just one example, recruitment of the western spadefoot – a petitioned species – is likely unsuccessful in pools with bullfrogs (Santos-Barrera et al. 2004).

Reptile predators that are dependent on larval or adult amphibians as a food source may also be impacted as a result of the loss of native amphibian larvae from bullfrogs and the presence of larger bullfrog tadpoles and adults upon which they are unable to efficiently prey (e.g., Kupferberg 1994). In addition, chytrid has been found on bullfrogs farmed for international food trade (Mazzoni et al. 2003). Infected bullfrogs released into the wild may contribute to the spread of chytrid, impacting other amphibians susceptible to the fungal disease.

c. Fire Ants

Fire ants threaten six of the petitioned species (more than ten percent). Humans accidentally introduced red imported fire ants to Alabama decades ago, and the ants have since spread across the southern U.S. (Wojick et al. 2001), where they are contributing to the decline of native species (Reagan et al. 2000, Allen et al. 2004). Fire ants have been reported to prey on both eggs (Moulis 1997) and young (Allen et al. 1997) of reptiles. For example, the fire ant has invaded the Lower Keys of Florida, and predation has been suggested as a reason for declines in some snake populations in the Southeastern Coastal Plain (Mount 1981), including the following petitioned species: Key ringneck snake, Florida pine snake, Rim Rock crowned snake, short-tailed snake, and southern hog-nosed snake.

d. Plants

Invasive plants are a threat to 15 percent of the petitioned species because they displace native plants and interfere with the food web. These weeds often form dense stands that can exclude amphibian and reptile species that are sensitive to changes in microhabitat (Germano and Hungerford 1981, Scott 1996, Maxwell and Hokit 1999). For example, the Yuman Desert fringe-toed lizard, a petitioned species, is threatened by non-native annual mustard (*Brassica*), which recently invaded southwestern Arizona. The mustard forms thick carpets and degrades habitat (Arizona Game and Fish Department, as cited in NatureServe 2011).

The hydrophilic salt cedar (*Tamarix ramosissima*) provides another example of an exotic plant that is threatening a petitioned species. Salt cedar has become widespread in mesic desert habitat and is a serious threat to springs and seeps in the region (DeDecker 1991). Because the panamint alligator lizard requires moist riparian habitats, introduction of tamarisk degrades its habitats and is a threat.

In addition to directly harming native species, the introduction of weed species may enhance the probability of successful introduction of other exotic species. For example, there is some evidence that the survival of exotic bullfrogs is enhanced by the presence of exotic aquatic vegetation, which provides habitat more suitable to the bullfrogs (Kupferberg 1996).

Figure 11. Introduced species, such as the bullfrog pictured here, threaten about one-third of the petitioned species.

2. Pollutants

It is well documented that pollutants are contributing to amphibian and reptile declines (e.g. Buhlmann and Gibbons 1997, Blaustein et al. 2003, Borja et al. 2006), and 16 of the petitioned species are threatened by pollutants (30 percent). Many substances can be toxic for amphibians and aquatic reptiles, including heavy metals, pesticides, phenols, fertilizers, roadsalt, mining waste, and chemicals in runoff (Dodd 1997). Amphibians are particularly vulnerable because the life history of most amphibians involves both aquatic larvae and terrestrial adults, allowing exposure to toxicants in both habitats. Plus, many amphibians have skin with vascularization in the epidermis and little keratinization, allowing easy absorption of many toxicants (Stebbins and Cohen 1995).

Pollutants can come from point or nonpoint sources. Point source pollution from manufacturing sites, power plants, and sewage treatment plants is a major cause of aquatic habitat degradation (Morse et al. 1997, Buckner at al. 2002, Kolpin et al. 2002). Non-point source pollution also degrades aquatic habitats, particularly when the runoff includes sediments and chemicals from timber operations, agriculture, or urban and industrial areas. Runoff from urban areas includes many substances that are harmful for amphibians, turtles and other aquatic organisms, such as petroleum particles, highway salts, silt, fertilizers, pesticides, surfactants, heavy metals, and pet wastes (deMaynadier and Hunter 1995, Neves et al. 1997, Turtle 2000, Buckner et al. 2002, Karraker and Gibbs 2011). These substances accumulate in aquatic herpetofauna and cause chronic effects. For example, eggs of the western pond turtle (a petitioned species) tested in Oregon contained low concentrations of organochlorine pesticides and PCBs, along with heavy metals mercury and chromium (Henny et al. 2003).

Many studies have demonstrated the harmful effects of chemical contamination on amphibians (reviewed in Cooke 1981, Hall and Henry 1992, Boyer and Grue 1995, and Carey and Bryant

1995). The consequences of chemical stressors are lethal, sublethal, direct and indirect. The sublethal affects of contaminants include hampered growth, development and behavior, which could lead to developmental and behavioral abnormalities (Bridges 1997, Bridges 2000) and may alter susceptibility to predation (Bridges 1999a) and competition and decrease reproductive success (Bridges 1999b, Relyea and Mills 2001, Boone and Semlitsch 2002). Chemical contaminants also weaken the immune system making amphibians and reptiles more susceptible to parasites, disease, and UV radiation (Blaustein et al. 2003, Christin et al. 2003, Daszak et al. 2003, Gendron et al. 2003). Endocrine disruption caused by contaminants can also lead to demographic shifts in aquatic reptile populations (EPA undated, Guillette et al. 1994, Gibbons et al. 2000). Other contaminates indirectly affect amphibians by altering food web dynamics (Boone and Bridges 2003).

Pesticides, in particular, threaten 13 petitioned species, which represents 25 percent of the petitioned species. For example, Blanding's turtles in Nebraska were found to be highly susceptible to the highly persistent pesticide dieldrin that was applied to cornfields for insect control and accumulated in wetland habitat (Congdon and Keinath 2006). To be sure, numerous studies have examined the harmful effects of pesticides on amphibians and reptiles (e.g. Saunders 1970, Harfenist et al. 1989, Johnson and Prine 1976, Bergeron et al. 1994, Gendron et al. 1997, Bridge et al. 2002). Many pesticides result in decreased growth rate and inhibition of a predator response in amphibians (e.g., Berrill et al. 1993 and Berrill et al. 1994). Other pesticides, such as atrazine, are endocrine disrupters that can cause demasculination and feminization of male frogs (Hayes et al. 2002a, 2002b, 2003, 2006). In a U.S. Geological Survey study of agricultural areas, 75 percent of stream water samples nationwide contained atrazine (Gilliom et al. 2006).

The impacts of the insecticide carbaryl on amphibians are well studied (e.g. Reylea and Mills 2001). Bridges (1999a, 1999b, 1997, 2000) found that sublethal concentrations of carbaryl alters tadpole behavior, making them more vulnerable to predation, and decrease feeding rates resulting in a smaller size at metamorphosis. Davidson et al. (2007) found that sublethal exposure to carbaryl likely inhibits the innate immune defense of foothill yellow-legged frogs, one of the petitioned species, and increases susceptibility to disease.

3. Sedimentation

Sedimentation is a threat to several of the petitioned species, including the wood turtle, Carolina gopher frog, foothill yellow-legged frog, California giant salamander, Cascade Caverns salamander, green salamander, Cascade torrent salamander, Columbia torrent salamander, and Olympic torrent salamander. Numerous studies have documented reductions of amphibian densities or populations of their invertebrate prey in streams experiencing sediment loading (Morse et al. 1997, Richter 1997, Welsh and Ollivier 1998, Semlitsch 2000). In particular, studies have shown that fine sediments, which may enter the stream through logging, mining, grazing, roads, and construction, may reduce salamander densities (Hawkins et al. 1983, Southerland 1986, Smith and Grossman 2003, White 2004). The impacts of sedimentation may be further heightened if the sediments contain toxic materials.


Figure 12. Pollutants such as pesticides threaten 30 percent of the petitioned species, especially the amphibians.

4. UV Radiation

UV radiation threatens some of the petitioned species, including the Cascades frog and foothill yellow-legged frog. Ambient levels of UV-B radiation in the atmosphere have risen significantly over the past few decades most likely due to decreases in stratospheric ozone, climate warming, and lake acidification (AmphibiaWeb 2012). Because amphibian eggs lack shells and adults and tadpoles have thin delicate skin, they are extremely vulnerable to increased levels of UV-B radiation (AmphibiaWeb 2012).

Researchers have found that UV-B radiation can kill amphibians directly, cause sublethal effects such as slowed growth rates and immune dysfunction, and work synergistically with contaminants, pathogens, and climate change (Kiesecker and Blaustein 1995; Long et al. 1995; Anzalone et al. 1998; Blaustein et al. 1996, 1998; Belden and Blaustein 2002a). For example, in the Cascades frog, exposure to UV-B increases embryo mortality, causes retinal damage, developmental and physiological abnormalities, and hampers antipredator behavior (Blaustein et al. 1994, Hays et al. 1996, Fite et al. 1998, Kats et al. 2000, Hatch and Blaustein 2000). It is likely that increases in ambient levels of UV-B radiation have contributed to some amphibian population declines (Blaustein and Wake 1995).

5. Isolation

We identified isolation as a threat to 31 of the petitioned species, especially many of the salamander species, which generally have low dispersal abilities and often exist in remnants of suitable habitat. Habitat connectivity is a key to regional viability of amphibian and reptile populations (Hecnar and M'CIoskey 1996, Semlitsch et al. 1996, Semlitsch and Bodie 1998, Skelly et al. 1999, Marsh and Trenham 2001, Rothermel and Semlitsch 2002). Isolated populations are more likely to go extinct in the long run than populations that are slightly connected (Hanski 1999). This is because small and isolated populations are more susceptible to extirpations due to stochastic events, human impacts, and environmental factors (Soulé 1987, Begone et al. 1990). And isolated populations are unlikely to be recolonized following a local extinction (Semlitsch and Bodie 1998). In addition, lack of gene flow may cause loss of genetic variability due to random genetic drift (Wright 1931) and inbreeding depression may occur (Franklin 1980). And the loss of genetic diversity can affect a population's ability to respond to environmental changes, confounding the effects of climate change, contaminants, and introduced species.

6. Climate Change

Climate change is a threat to 20 of the petitioned amphibian and reptile species, and it is likely responsible for observed population declines in some amphibians and reptiles (Donnelly and Crump 1998, Pounds et al. 1999, Araujo et al. 2006, Daszak et al. 2005, Pounds et al. 2006).

Climate change is already causing a rise in temperatures across the United States and an increase in extreme weather events, such as droughts and floods (Parmesan et al. 2000, NSC 2003, CCSP 2008, Karl et al. 2009). This is particularly problematic for amphibian and reptile populations because they are ectothermic. As such, they are sensitive to changes in air and water temperature, precipitation, and the hydroperiod (length of time and seasonality of water presence); their body temperatures and activity cycles are dependent on the presence of optimal environmental conditions (Carey and Alexander 2003, Lind 2008). For example, an early breeding season for the Illinois chorus frog, one of the petitioned species, makes it susceptible to sudden freezes (Packard et al. 1998, Tucker 2000).

Some amphibians have shown a trend towards earlier breeding, apparently in response to global warming (Beebee 1995, Blaustein et al. 2001, Gibbs and Breisch 2001). If such shifts in activities occur inconsistently with other ecological events (e.g., emergence of their insect prey), growth and survival rates would be affected. Plus, some reptile species, such as the spotted turtle (one of the petitioned species), exhibit temperature-dependent sex determination during egg incubation that could be influenced by changes and variability in global climates (Gibbons et al. 2000, Hawkes et al. 2007). Temperatures outside of their thermal optima will also cause physiological stresses for amphibians and reptiles (Lind 2008) and may affect body size, which in turn affects reproductive rate (Reading 2007).

Also, many amphibians require aquatic habitats for egg laying and larval development and moist environments for post-metamorphic life stages (Deullman and Trueb 1986, Wells 2007). Pond-breeding amphibian species require water bodies that do not dry up before their tadpoles can metamorphose (Lind 2008). Species associated with ephemeral waters, such as shallow ponds and intermittent streams, may be particularly vulnerable to altered precipitation patterns (Dodd 1997, Lind 2008, McMenamina et al. 2008). These include several of the petitioned species, including the western spadefoot, Carolina gopher frog, and Illinois chorus frog. Long-term droughts likely already have caused some gopher frog populations to disappear because of insufficient population recruitment (Florida Fish and Wildlife Conservation Commission 2011).

In general, particular ecological communities are expected to move upward in both elevation and latitude in response to climate change (Walther et al. 2002). For example, van Dijk and Harding (2010) suggest that global warming is likely to tip southern populations of the wood turtle (a petitioned species) towards extinction. As with other species, montane and higher-latitude populations of amphibians and reptiles are most at risk (Root et al. 2003). An already fragmented landscape will impede the ability of species to respond to climate-induced habitat changes (Halpin 1997).

Many of the petitioned species, especially the salamanders, have narrow elevational ranges along altitudinal gradients, with essentially no latitudinal or longitudinal ranges at all; those restricted to near the tops of mountains may experience range collapse under a climatic warming scenario, because suitable environmental conditions no longer exist locally (Rovito et al. 2009, Early and Sax 2011). The following petitioned salamander species are threatened by climate change: Inyo Mountains salamander, lesser slender salamander, Kings River slender salamander, Caddo Mountain salamander, Fourche Mountain salamander, Peaks of Otter salamander, Weller's salamander, green salamander, Cascade torrent salamander, Columbia torrent salamander, and Olympic torrent salamander.

Sea level rise due to climate change is a threat to several of the petitioned species, including the Key ringneck snake, Rim Rock crowned snake, and Cedar Key mole skink. For example, the Florida Fish and Wildlife Conservation Commission (2011a) explains that habitat on Big Pine Key – essential for the Key ringneck snake and Rim Rock crowned snake – is being lost due to sea level rise. In the best-case scenario, a sea level rise of 18 cm (7 inches) by Year 2100 would inundate 34 percent of Big Pine Key, resulting in the loss of 11 percent of the island's upland habitat (FFWCC 2011a). In the worst-case scenario, a sea level rise of 140 cm (4.6 feet) by Year 2100 would inundate 96 percent of Big Pine Key.

Climate-driven changes are likely to combine with other human-induced stresses to further increase the vulnerability of natural ecosystems to pests, invasive species, and loss of native species (Karl et al. 2009). For example, changes in climatic regimes are likely to increase pathogen virulence and amphibian and reptile susceptibility to pathogens (Pounds et al. 2006, Pounds et al. 2007, Gervasi et al. 2008, Alford 2011). Similarly, warm water invasive species (e.g., bullfrogs, some fishes in the western United States) are a concern to native species and may expand their ranges given warming trends. In a changing climate, populations of some pests such as red fire ants and rodents, better adapted to a warmer climate, are projected to increase (Karl et al. 2009).

Figure 13. Increased droughts is one effect of climate change, which impacts many of the petitioned species.

7. Synergies and Multiple Causes

The risk of extinction for the petitioned species is heightened by synergies between threats, as most species face multiple threats and these threats interact and magnify each other (see, e.g. Kiesecker et al. 2001, Gendron et al. 2003, Pounds et al. 2006). For example, UV-B rays have been directly implicated as a cause of increasing bacterial *Saprolegina* infections in the Cascades frog (a petitioned species) that lead to mass population declines in some areas (Kiesecker and Blaustein 1995). In addition, Kupferberg et al. (2009a) presented data supporting a link between periods of unusually warm summer water temperatures in a northern California river, outbreaks of the parasitic copepod *Lernaea cyprinacea*, and malformations in tadpoles and young of the year foothill yellow-legged frogs, another petitioned species.

***

All of the petitioned species are threatened by one or more of the above factors and many face numerous overlapping threats to their continued existence. Due to the magnitude and imminence of the threats facing the amphibians and reptiles in this petition, they all warrant protection under the Endangered Species Act. The Act has been 99 percent successful at preventing the extinction of listed species and is the best tool available for reversing the amphibian and reptile extinction crisis.

IV. LITERATURE CITED

Abell, R.A., D.M. Olson, E. Dinerstein, P. Hurley, J.T. Diggs, W. Eichbaum, S. Walters, W. Wettengel, T. Allnutt, C.J. Loucks, P. Hedao. 2000. Freshwater ecoregions of North America: a conservation assessment. Washington, DC: Island Press. 319 p.

Adams, L.W. 1994. Urban habitats: a landscape perspective. Minneapolis, MN: University of Minnesota Press. 186 pp.

Adams, M .J. 1997. Experimental analysis of tadpole mortality factors: effects of bullfrogs and exotic fish in the Puget Sound Lowlands, Washington. Dissertation, University of Washington, Seattle, Washington.

Adams, M. J. 2000. Pond permanence and the effects of exotic vertebrates on anurans. Ecological applications 10: 559-568.

Adams, M.J., and R.B. Bury. 2002. The endemic headwater stream amphibians of the American Northwest: Associations with environmental gradients in a large forested preserve. Global Ecology and Biogeography 11: 169–178.

Adkins Giese, C. 2011. Species Proposals for Consideration at CoP16, *available at* http://www.biologicaldiversity.org/campaigns/southern_and_midwestern_freshwater_turtles/pdfs/Freshwater_turtles-CITES_petition_Aug_8.pdf (last visited July 4, 2012).

Ahlstedt, S.A., M.T. Fagg, R.S. Butler, and J.F. Connell. 2005. Long-term information for freshwater mussel populations at twelve fixed-station monitoring sites in the Clinch and Powell Rivers of Eastern Tennessee and Southwestern Virginia. Final report submitted to U.S. Fish and Wildlife Service February 16, 2005.

Akre, T. and C. Ernst. 2006. Final Report Population Dynamics, Habitat Use, and Home Range of the Wood Turtle, Glyptemys (=Clemmys) insculpta, in Virginia, submitted to the Virginia Department of Game and Inland Fisheries, contract ed0820ce_080.

Alford, R. A., and S. J. Richards. 1999. Global amphibian declines: a problem in applied ecology. Annual Review of Ecology and Systematics 30: 133–165.

Alford, R. 2011. Bleak future for amphibians. Nature 480: 461.

Allen, C.R., D.M. Epperson, and A.S. Garmestani. 2004. Red imported fire ant impacts on wildlife: a decade of research. American Midland Naturalist 152: 88-103.

Allen, E. B., and L.L. Jackson. 1992. The arid west: here the challenge is to reverse the downward spiral of desertiĐcation over vast areas. Restoration and Management Notes 10: 56-59.

Allen, K. G. Rice, D. P. Wojcik, and H. F. Percival. 1997b. Effect of red imported fire ant envenomization on neonatal American alligators. J. Herp. 31: 318–321.

Amézquita, A., H. Lüddecke. 1999. Correlates of intrapopulational variation in metamorphic size of the high-Andean frog *Hyla labialis*. Herpetologica 55: 295-303.

AmphibiaWeb: Information on amphibian biology and conservation. [web application]. 2012. Berkeley, California: AmphibiaWeb, *available at*: http://amphibiaweb.org/ (last visited Apr 3, 2012).

Andelman, S. J., and E. Gray. 1992. Viability and recovery of western pond turtle (*Clemmys marmorata*) populations in Washington state. Abstract, Society for Conservation Biology, 6th Annual Meeting, p. 33.

Andrews, K. M. 2004. Interspecific comparisons of behavioral responses of southeastern snakes to roads. M.S. Thesis. University of Georgia. Athens, GA.

Andrews, K. M., and J. W. Gibbons. 2005. How do highways influence snake movement? Behavioral responses to roads and vehicles. Copeia 2005: 771-781.

Ankney, C. D. 1996. An embarrassment of riches: too many geese. Journal of Wildlife Management 60: 217-223.

Anzalone, C. R., L. B. Kats, and M. S. Gordon. 1998. Effects of solar UV-B radiation on embryonic development in Hyla cadaverina, Hyla regilla, and Taricha torosa. Conservation Biology 12: 646-653.

Araújo M.B., W. Thuiller, and R.G. Pearson. 2006. Climate warming and the decline of amphibians and reptiles in Europe. J. Biogeogr. 33: 1712-1728.

Aresco, M. J. 2003. Highway mortality of turtles and other herpetofauna at lake Jackson, Florida, USA, and the efficacy of a temporary fence/culvert system to reduce roadkills. Pp. 433-449 *In*: Proceedings of the International Conference on Ecology and Transportation, edited by C. Leroy Irwin, Paul Garrett, and K. P. McDermott, Raleigh, NC: Center for Transportation, and the Environment, North Carolina State University.

Aresco, M. J. 2005. Mitigation measures to reduce highway mortality of turtles and other herpetofauna at a north Florida lake. Journal of Wildlife Management 69: 549-560.

Ash, A.N. 1988. Disappearance of salamanders from clearcut plots. Journal of the Elisha Mitchell Scientific Society 104: 116–122.

Ash, A.N. 1995. Effects of clear-cutting on litter parameters in the southern Blue Ridge Mountains. Castanea 60: 89–97.

Ash, A.N. 1997. Disappearance and return of plethodontid salamanders to clearcut plots in the southern Blue Ridge Mountains. Conservation Biology 11: 983–989.

Ashley, E. P., and J. T. Robinson. 1996. Road mortality of amphibians, reptiles and other wildlife on the long point causeway, Lake Erie, Ontario. Canadian Field-Naturalist 110: 403-412.

Ashton, D. T., A. J. Lind, and K. E. Schlick. 1998. *Rana boylii* – foothill yellow-legged frog. Natural history review. USDA Forest Service, Pacific Southwest Research Station, Redwood Sciences Laboratory, Arcata, California.

Associated Press. 2006. U.S. population hits 300 million mark, *available at* http://www.msnbc.msn.com/id/15298443 (last visited Feb. 17, 2012).

Baldwin, E. A., M. N. Marchand, and J. A. Litvaitis. 2004. Terrestrial habitat use by nesting painted turtles in landscapes with different levels of fragmentation. Northeastern Naturalist 11(1): 41-48.

Ballinger, R .E. and K. S. Watts. 1995. Path to extinction: impact of vegetational change on lizard populations on Arapaho Prairie in the Nebraska Sandhills. American Midland Naturalist 134: 413-417.

Balmford, A., R.E. Green, M. Jenkins. 2003. Measuring the changing state of nature. Trends in Ecology and Evolution 18(7): 326–330.

Barinaga, M. 1990. Where have all the froggies gone? Science 247: 1033–1034.

Barr, G. E., and K. J. Babbitt. 2002. Effects of biotic and abiotic factors on the distribution and abundance of larval two-lined salamanders (Eurycea bislineata) across spatial scales. Oecologia 133: 176–185.

Bast, K., and T.J. Maret. 1998. Ecological correlates of stream salamander abundance and distribution. Journal of the Pennsylvania Academy of Science 71: 164.

Bauerle, B., D.L. Spencer, and W. Wheeler. 1975. The use of snakes as indicator species. Copeia 1975: 366-368.

Beebee T.J.C., and R.A Griffiths. 2005. The amphibian decline crisis: a watershed for conservation biology? Biol Conserv 125: 271–285.

Beebee, T.J.C. 1995. Amphibian Breeding and Climate. Nature 374:219-220.

Beebee, T.J.C. 1996. Ecology and Conservation of Amphibians. Chapman and Hall, London.

Begone, M., J.L. Harper, and C.R. Townsend. 1990. Ecology: Individuals, populations, and communities. Blackwell Scientific Publications.

Behnkne, J., and R.F. Raleigh. 1978. Grazing and the riparian zone: impact on management perspectives. Pages 263-267 in R.Johnson and J. F McCormick, editors. Strategies for protection and management of floodplain wetlands and other riparian ecosystems. United States Forest Service, General Technical Report WO-12.

Belden, L.K. and A. R. Blaustein. 2002. UV-B Induced Skin Darkening in Larval Salamanders Does Not Prevent Sublethal Effects of Exposure on Growth. Copeia 3: 748–754.

Belding, R, B., W M. Giulianod, D. Putnam, and J. Taracido. 2000. Integrating wildlife management and agriculture: conserving biodiversity through long-term partnerships. Pages 48-51 in G. D. Therres, editor. Conservation of biodiversity: a key to the restoration of the Chesapeake Bay ecosystem and beyond. Maryland Department of Natural Resources, Annapolis, USA.

Bellis, E.D. 1962. The influence of humidity on wood frog activity. American Midland Naturalist 68: 139–148.

Belsky, A.J. and J. L. Gelbard. 2000. Livestock grazing and weed invasions in the arid west. Oregon Natural Desert Association. Bend, OR.

Belsky, A.J., A. Matzke, S. Uselman. 1999. Survey of livestock influences on stream and riparian ecosystems in the western United States. J. Soil & Water Conserv. 54(1): 419.

Bender, D.J., T. A. Contreras, and L. Fahrig. 1998. Habitat loss and population decline: a meta-analysis of the patch size effect. Ecology 79: 517–533.

Bennett, M.G., B.R. Kuhajda, and J.H. Howell. 2008. Status of the imperiled Frecklebelly Madtom, *Noturus munitus* (Siluriformes: Ictaluridae): A review of data from field surveys, museum records, and the literature. Southeastern Naturalist 7(3): 459-474.

Benz, G.W. and D.E. Collins (editors). 1997. Aquatic Fauna in Peril: The Southeastern Perspective. Southeast Aquatic Research Institute Special Publication 1, Lenz Design and Communications, Decatur, GA. 553 pp.

Berger, L., R. Speare, P. Daszak, D. E. Green, A. A. Cunningham, C. L. Goggin, R. Slocombe, M. A. Ragan, A. D. Hyatt, K. R. McDonald, H. B. Hines, K. R. Lips, G. Marantelli, and H. Parkes. 1998. Chytridiomycosis causes amphibian mortality associated with population declines in the rain forests of Australia and Central America. Proceedings of the National Academy of Sciences 95: 9031-9036.

Bergeron, J.M., D. Crews, and J.A. McLachlan. 1994. PCBs as environmental estrogens: turtle sex determination as a biomarker of environmental contamination. Environmental Health Perspectives 102: 780-781.

Bernardino, Jr., F.S. and G.H. Dalrymple. 1992. Seasonal activity and road mortality of the snakes of the Pa-hay-okee wetlands of Everglades National Park, USA. Biological Conservation 61: 71-75.

Bernhardt, E.S. and M.A. Palmer. 2011. The environmental costs of mountaintop mining valley fill operations for aquatic ecosystems of the Central Appalachians. Annals of the New York Academy of Sciences 1223: 39–57.

Berrill, M., S. Bertram, A. Wilson, S. Louis, D. Brigham, and C. Stromberg. 1993. Lethal and sublethal impacts of pyrethroid insecticides on amphibian embryos and tadpoles. Environmental Toxicology and Chemistry 12(3): 525-539.

Berrill, M., S. Bertram, L. McGillivray, M. Kolohon, and B. Paul. 1994. Effects of low concentrations of forest-use pesticides on frog embryos and tadpoles. Environmental Toxicology and Chemistry 13(4): 657-664.

Bervan, K.A. 1981. Mate choice in the wood frog, Rana sylvatica. Evolution 35: 707–722.

Birdsall, C.W., C.E. Grue, and A. Anderson. 1986. Lead concentrations in bullfrog Rana catesbeiana and green frog R. clamitans tadpoles inhabiting highway drainages. Environ. Pollut. Ser. A 40: 233-247.

Black, J.H. 1981. Oklahoma rattlesnake hunts. Bulletin of the Oklahoma Herpetological Society 6: 39-43.

Blaustein, A.R., D.G. Hokit, R.K. O'Hara. 1994. Pathogenic fungus contributes to amphibian losses in the Pacific Northwest. Biol. Cons. 67: 251-254.

Blaustein, A.R. 1993. Declining amphibian populations in perspective. Park Science 1993(Fall): 8-9.

Blaustein, A.R. 1994. Chicken little or Nero's fiddle: a perspective on declining amphibian populations. Herpetologica 50: 85-87.

Blaustein, A.R. and P.T.J. Johnson. 2003. Explaining frog deformities. *Scientific American* 288: 60-65.

Blaustein, A. R., and D. B. Wake. 1995. The Puzzle of Declining Amphibian Populations. Scientific American 272: 52-57.

Blaustein, A. R., B. Edmond, J. M. Kiesecker, J. J. Beatty, and D. G. Hokit. 1995. Ambient ultraviolet radiation causes mortality in salamander eggs. Ecological Applications 5: 740–743.

Blaustein, A. R., D. B. Wake, and W. P Sousa. 1994. Amphibian declines: judging stability, persistence, and susceptibility of populations to local and global extinctions. Conservation Biology 8: 60-71.

Blaustein, A. R., D. G. Hokit, R. K. O'Hara, and R. A. Holt. 1994b. Pathogenic fungus contributes to amphibian losses in the Pacific Northwest. Biological Conservation 67(3):251-254.

Blaustein, A. R., J. M. Kiesecker, D. P. Chivers, D. G. Hokit, A. Marco, L. K. Belden, and A. Hatch. 1998. Effects of ultraviolet radiation on amphibians: Field experiments. American Zoologist 38: 799-812.

Blaustein, A. R., J. M. Romansic, J. M. Kiesecker, and A. C. Hatch. 2003. Ultraviolet radiation, toxic chemicals and amphibian population declines. Diversity & Distributions [print] 9:123-140.

Blaustein, A. R., L. K. Belden, D. H. Olson, D. M. Green, T. L. Root, and J. M. Kiesecker. 2001. Amphibian breeding and climate change. Conservation Biology 15: 1804-1809.

Blaustein, A. R., R D. Hoffman, J. M. Kiesecker, and J. B. Hays. 1996. DNA repair activity and resistance to solar UV-B radiation in eggs of the red legged frog. Conservation Biology 10:1398-1402.

Blaustein, A.R., D.B. Wake, and W.P Sousa. 1994. Amphibian declines: judging stability, persistence, and susceptibility of populations to local and global extinctions. Conservation Biology 8: 60–71.

Blymer, M. J., and B. S. McGinnes. 1977. Observations on possible detrimental effects of clearcutting on terrestrial amphibians. Bull. Maryland Herpetol. Soc. 13: 79-83.

Bock, C. E., H. M. Smith, and J. H. Bock. 1990. The effect of livestock grazing upon the abundance of the lizard, *Sceloporus scalaris*, in southeastern Arizona. Journal of Herpetology 24: 445-446.

Bondello, M.C. 1976. The effects of high intensity motorcycle sounds on the acoustical sensitivity of the desert iguana, Dipsosaurus dorsalis Masters Thesis, California State University, Fullerton, California.

Bonnet, X., G. Naulleau and R. Shine. 1999. The dangers of leaving home: Dispersal and mortality in snakes. Biological Conservation 89: 39-50.

Boone, M. D., and C. M. Bridges. 2003. Effects of pesticides on amphibian populations. Pages 152-167 in R. D. Semlitsch, editor. Amphibian Conservation. Smithsonian Institution, Washington. toxic chemicals and amphibian population declines. Diversity & Distributions [print] 9: 123-140.

Boone, M. D., and R. D. Semlitsch. 2002. Interactions of an insecticide with competition and pond drying in amphibian communities. Ecological Applications 12: 307-316.

Borisenko A.N. and M.P. Hayes. 1999. Status of the foothill yellow-legged frog (Rana boylii) in Oregon. Final Report to The Nature Conservancy under contract to the US Fish and Wildlife Service with assistance from the Oregon Department of Fish and Wildlife, the US Army Corps of Engineers, and the United States Geological Survey. 39 p. Available from: Oregon State Office of US Fish and Wildlife Service, 2600 SE 98th Avenue, Portland, OR 97266.

Borja, A., I. Muxika, J. Franco. 2006. Long-term recovery of softbottom benthos following urban and industrial sewage treatment in the Nervion estuary (southern Bay of Biscay). Marine Ecology Progress Series 313: 43–55.

Boyer, R., and C. E. Grue. 1995. The need for water quality criteria for frogs. Environmental Health Perspectives 103: 352-357.

Bradford, D. E. 1991. Mass mortality and extinction in a high-elevation population of Rana muscosa. Journal of Herpetology 25: 174-177.

Bradford, D. F. 1983. Winterkill, oxygen relations, and energy metabolism of a submerged dormant amphibian, *Rana muscosa*. Ecology 64: 1171-1183.

Branch, L. C., D. G. Hokit, and B. M. Stith. 1996. Landscape dynamics and Florida scrub lizards: assessing the effects of landscape structure on populations at multiple scales. Final report, RWO 122, Biological Resources Division, U.S. Department of Interior, Washington, D.C., USA.

Brattstrom, B. H., and M. C. Bondello. 1979. The effects of dune buggy sounds on the telencephalic auditory evoked response in the Mojave fringe-toed lizard, Uma scoparia Unpublished Report to U.S. Bureau of Land Management, California Desert Program, Riverside, California. 31pp. Contract CA-060-CT7-2737.

Bridges, C. M. 1997. Tadpole swimming performance and activity affected by acute exposure to sublethal levels of carbaryl. Environmental Toxicology & Chemistry 16: 1935-1939.

Bridges, C. M. 1999a. Effects of a pesticide on tadpole activity and predator avoidance behavior. Journal of Herpetology 33: 303-306.

Bridges, C. M. 1999b. Predator-prey interactions between two amphibian species: Effects of insecticide exposure. Aquatic Ecology 33: 205-211.

Bridges, C. M. 2000. Long-term effects of pesticide exposure at various life stages of the southern leopard frog (Rana sphenocephala). Archives of Environmental Contamination and Toxicology 39: 91-96.

Bridges, C. M., F. J. Dwyer, D. K. Hardesty, and D. W. Whites. 2002. Comparative contaminant toxicity: Are amphibian larvae more sensitive than fish? Bulletin of Environmental Contamination & Toxicology [print] 69: 562-569.

Briggs, C. J., V. T. Vredenburg, R. A. Knapp, and L. J. Rachowicz. 2005. Investigating the population level effects of chytridiomycosis: An emerging infectious disease of amphibians. Ecology 86: 3149–3159.

Brooks, R.J., C.M. Shilton, G.P. Brown, and N.W.S. Quinn. 1992. Body size, age distribution, and reproduction in a northern population of wood turtles (Clemmys insculpta). Can. J. Zool. 69: 1,314-20.

Brooks, R.J., G.P. Brown, and D.A. Galbraith. 1991. Effects of a sudden increase in natural mortality of adults on a population of the common snapping turtle (*Chelydra serpentina*). Canadian Journal of Zoology 69: 1314-1320.

Brown, J.H., A. Kodric-Brown. 1977. Turnover rates in insular biogeography: effect of immigration on extinction. Ecology 58: 445–449.

Browne, C.L. and S.J. Hecnar. 2007. Species loss and shifting population structure of freshwater turtles despite habitat protection. Biological Conservation 138(3-4): 421-429.

Buckner, M.M., W. Smith, J.A. Takats. 2002. Tennessee, Cumberland, and Mobile River Basins at Risk. A Biological Assessment and Vision for the World Wildlife Fund's Southeast Rivers and Streams Project. Nashville, TN. 52 pp., *available at* www.worldwildlife.org/what/wherewework/sers/WWFBinaryitem2738.pdf (last visited April 4, 2012).

Buhlmann, K.A. and J.W. Gibbons. 1997. Imperiled aquatic reptiles of the Southeast. Pp. 201-231 *In:* Benz, G.W. and D.E. Collins (editors). 1997. Aquatic Fauna in Peril: The Southeastern Perspective. Southeast Aquatic Research Institute Special Publication 1, Lenz Design and Communications, Decatur, GA. 553 pp.

Bulow-Olson, A. 1980. Changes in the species composition in an area dominated by Deschampsia flexuosa (L.) trin. as a result of cattle grazing. Biological Conservation 18: 257-270.

Burke, D. M., and E. Nol. 1998. Influence of food abundance, nest-site habitat, and forest fragmentation on breeding ovenbirds. The Auk 115: 96–104.

Burke, V.J. and J.W. Gibbons. 1995. Terrestrial buffer zones and wetland conservation: a case study of freshwater turtles in Carolina bay. Conservation Biology 9: 1365-1369.

Bury, R.B. 1983. Differences in amphibian populations in logged and old-growth redwood forests. Northwest Science 57: 167–178.

Bury, and P. S. Corn. 1988. Responses of aquatic and streamside amphibians to timber harvest: a review. In K. Raedeke (ed.), Streamside Management: Riparian Wildlife and Forestry Interactions, pp. 165-181. Institute of Forest Resources, Cont. No. 59. Univ. Washington, Seattle.

Bury, B. R., and A. J. Whelan. 1984. Ecology and Management of the Bullfrog. United States Department of the Interior, Fish and Wildlife Service, Resource Publication 155, Washington D.C.

Bury, R. B., C. K. Dodd, Jr., and G. M. Fellers. 1980. Conservation of the amphibia of the United States: a review. Department of the Interior, Fish and Wildlife Service Publication Number 134: 1-34, Washington, D.C., USA.

Bury, R. B., R. A. Luckenbach, and S. O. Busack. 1977. Effects of off-road vehicles on vertebrates in the California Desert. Washington, D.C. U.S. Fish and Wildlife Service. Wildlife Research Report 8: 1-23.

Bury, R.B, and D.J. Germano. 2008. Actinemys marmorata (Baird and Girard 1852) – Western pond turtle, Pacific pond turtle. Conservation Biology of Freshwater Turtles and Tortoises: A compilation project of the IUCN/SSC Tortoise and Freshwater Turtle Specialist Group. Chelonian Research Monographs No. 5.

Bury, R.B. 2008, Amphibians in the Willamette Valley, Oregon- Survival in a rapidly urbanizing and developing region. In Mitchell, J.C., Brown, R.E., Bartholomew, B., eds., Urban Herpetology: Salt Lake City, UT, Herpetological Conservation No. 3. Society for Study of Amphibians and Reptiles, p. 435-444.

Busack, S. D., and R. B. Bury. 1974. Some effects of off-road vehicles and sheep grazing on lizard populations in the Mojave Desert. Biological Conservation 6(3): 179-83.

Byrne, M.W., E.P. Davie, and J.W. Gibbons. 2008. *Batrachochytrium dendrobatidis* Occurrence in *Eurycea cirrigera*. Southeastern Naturalist 7(3): 551-555.

Campbell, F.T. 1988. The desert tortoise. Pages 567-581 in W.J. Chandler, editor. Audubon Wildlife Report 1988/1989. Academic Press, San Diego, California.

Campbell, H. 1956. Snakes found dead on the roads of New Mexico. Copeia 1956: 124-125.

Campbell, J. A., D. R. Formanowicz, Jr., and E. D. Brodie, Jr. 1989. Potential impact of rattlesnake roundups on natural populations. Texas Journal of Science 41: 301-317.

Carey, C. 1993. Hypothesis concerning the causes of the disappearance of Boreal toads from the mountains of Colorado. Conservation Biology 7(2): 355-362.

Carey, C., and Bryant, C. J. 1995. Possible interrelations among environmental toxicants, amphibian development, and decline of amphibian populations. Environmental Health Perspectives 103 (Suppl. 4): 13-17.

Carlisle, D.M., C.P. Hawkins, M.R. Meadori, M. Potapova, and J. Falcone. 2008. Biological assessments of Appalachian streams based on predictive models for fish, macroinvertebrate, and diatom assemblages. Journal of the North American Benthological Society 27(1): 16-37.

Carpenter, H. L., and E. O. Morrison. 1973. Feeding behavior of the bullfrog, *Rana catesbeiana*, in north central Texas. Bios 44: 188-193.

Carr, L. W., and L. Fahrig. 2001. Effect of road traffic on two amphibian species of differing vagility. Conservation Biology 15: 1071-1078.

Cary, C., M.A. Alexander. 2003. Climate change and amphibian declines: Is there a link? Diversity and Distributions. 9: 111-121.

Casher L., R. Lane R, Barrett, and L. Eisen. Relative importance of lizards and mammals as hosts for ixodid ticks in northern California. Exp Appl Acarol. 26(1-2): 127-43.

Castellano, C.M., J.L. Behler, and G.R. Ultsch. 2008. Terrestrial movements of hatchling Wood Turtles (*Glyptemys insculpta*) in agricultural fields in New Jersey. Chelonian Conservation and Biology 7(1): 113–118.

Cauble, K. and R.S. Wagner. 2005. Sublethal effects of the herbicide glyphosate on amphibian metamorphosis and development. Bulletin of environmental contamination and toxicology 75: 429–435.

CCSP (U.S. Climate Change Science Program). 2008. Weather and Climate Extremes in a Changing Climate, Regions of Focus: North America, Hawaii, Caribbean, and U.S. Pacific Islands. T.R. Karl, G.A. Meehl, C.D. Miller, S.J. Hassol, A.M. Waple, and W.L. Murray (eds.). Washington, DC: Department of Commerce, NOAA’s National Climate Data Center.

Central Intelligence Agency. 2012. The World Factbook -- Field Listing - Population growth rate, *available at* https://www.cia.gov/library/publications/the-world-factbook/fields/2002.html (last visited Feb. 17, 2012).

Chan-McLeod, A.C.A. 2003. Factors affecting the permeability of clearcuts to red-legged frogs. J. Wildl. Manag. 67: 663–671.

Chen, J., S.C. Saunders, T.R. Crow, R.J. Naiman, K.D. Brosofske, G.D. Mroz, B.L. Brookshire, J.F Franklin. 1999. Microclimate in forest ecosystem and landscape ecology. BioScience 49: 288–297.

Cherry D.S., R.J. Currie, D.J. Soucek, H.A. Latimer, and G.C. Trent. 2001. An integrative assessment of a watershed impacted by abandoned mined land discharges. Environmental Pollution 111:377–388.

Chin, N. T. 1996. Watershed Urbanization Effects on Palustrine Wetlands: A Study of Hydrologic, Vegetative, and Amphibian Community Response Over Eight Years. Master's thesis, University of Washington.

Chinchar, V.G. 2002. Ranaviruses (family Iridoviridae): emerging cold-blooded killers. Arch. Virol. 147: 447-470.

Chivian, E., and A. Bernstein. 2008. Sustaining Life: How Human Health Depends upon Biodiversity. Oxford University Press.

Christiansen, J. L., and B. J. Gallaway. 1984. Raccoon removal, nesting success, and hatchling emergence in Iowa turtles with special reference to Kinosternon flavescens (Kinosternidae). Southwestern Naturalist 29: 343-348.

Christin, M.-S., D. Gendron Andree, P. Brousseau, L. Menard, J. Marcogliese David, D. Cyr, S. Ruby, and M. Fournier. 2003. Effects of agricultural pesticides on the immune system of Rana pipiens and on its resistance to parasitic infection. Environmental Toxicology & Chemistry [print] 22: 1127-1133.

Clark, K. L., and B. D. Lazerte. 1985. A laboratory study of the effects of aluminum and pH on amphibian eggs and tadpoles. Canadian Journal of Fisheries & Aquatic Sciences 42: 1544-1551.

Clark, K. L., and R. J. Hall. 1985. Effects of elevated hydrogen ion and aluminum concentrations on the survival of amphibian embryos and larvae. Canadian Journal of Zoology 63: 116-123.

Clark, V. S. 1991. A report on rattlesnake roundups in Oklahoma and other parts of the United States. Herpetology 21: 12-15.

Clarkson, R. W., and J. C. DeVos, Jr. 1986. The bullfrog, *Rana catesbeiana* Shaw, in the lower Colorado River, Arizona, California. Journal of Herpetology 20: 42-49.

Corkran, C.C., T.R. Curry, and C. Fielder. 2006. Is recreation impacting western toad populations on the Mt. Hood National Forest? Abstracts from the 2006 Annual Meetings of the Society for Northwestern Vertebrate Biology and the Washington Chapter of the Wildlife Society. Olympia, WA. March 27, 2006. Northwestern Naturalist 87(2):153-194.

Colborn, T. 2004. Commentary: setting aside tradition when dealing with endocrine disruptors. ILAR J 45(4): 394–400.

Colborn, T. 2006. A case for revisiting the safety of pesticides: a closer look at neurodevelopment. Environ Health Perspect 114(1): 10–17.

Colborn, T., F.S. vom Sall, and A.M. Soto. 1993. Developmental effects of endocrine disrupting chemicals in humans and wildlife. Environmental Health Perspectives 101: 378-384.

Collins, J.P., T.R. Jones. and H.A Berna. 1988. Conserving genetically distinctive populations: the case of the Huachuca tiger salamander (*Ambystoma tigrinum stebbinsi*). Management of amphibians, reptiles and small mammals in North America (ed. by R.C. Szaro, K.C. Severson and D.R. Patton), pp. 45–53. USDA Forest Service GTR-RM-166, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO.

Collins, J.P., N. Cohen, E.W. Davidson, J.E. Longcore, and A. Storfer. 2003b. Global amphibian declines: an interdisciplinary research challenge for the 21st century. Status and conservation of U.S. amphibians, Vol. 1. Conservation essays (ed. by M.J. Lannoo), pp. 43–52. University of California Press, Berkeley, CA, in press.

Congdon, J.D., and D.A. Keinath. 2006. Blanding's Turtle (*Emydoidea blandingii*): A Technical Conservation Assessment, USDA Forest Service, Rocky Mountain Region, *available at* http://www.fs.fed.us/r2/projects/scp/assessments/blandingsturtle.pdf (last visited July 5, 2011).

Congdon J.D., A.E. Dunham, and R.C. van Lobels Sels. 1993. Delayed Sexual Maturity and Demographics Blanding's Turtles (*Emydoidea blandingii*): Implications for conservation and management of long-lived organisms. Conservation Biology 7(4).

Congdon J.D., Dunham AE, van Lobels Sels RC. 1994. Demographics of Common Snapping Turtles (*Chelydra serpentine*): Implications for conservation and management of long-lived organisms. Amer. Zool. 34: 397-408.

Connolly, K.D., D.R. Williams, and S.M. Johnson. 2005. Wetlands Law and Policy: Understanding Section 404. Chicago: Section of Environment, Energy, and Resources, American Bar Association.

Connor, E. F., and E. D. McCoy. 1979. The statistics and biology of the species-area relationship. The American Naturalist 113: 791–833.

Cooke, A. K. 1981. Tadpoles as indicators of harmful levels of pollution in the field. Environmental Pollution (Series A) 25: 123-133.

Cooper N., J. Bielby, G.H. Thomas, and A. Purvis. 2008. Macroecology and extinction risk correlates of frogs. Global Ecology and Biogeography 17(2): 211-221.

Corn, P. S. 1994. What we know and don't know about amphibian declines in the west. Pages 59-67 in W. Covington and L. DeBano (technical coordinators.), Sustainable ecological systems: implementing an ecological approach to land management. U.S.D.A. Forest Service, Rocky Mountain Forest and Range Experimental Station, Ft. Collins, Colorado. General Technical Report RM-247.

Corn, P. S., and R. B. Bury. 1989. Logging in western Oregon: responses of headwater habitats and stream amphibians. Forest Ecology and Management 29: 39–57.

Cory, L., P. Fjeld, W. Serat, 1970. Distribution patterns of DDT residues in the Sierra Nevada Mountains. Pesticides Monitoring Journal 3: 204-211.

Cosson, J. F., S. Ringuet, O. Claessens, J. C. de Massary, A. Dalecky, J. F. Villiers, L. Granjon, and J. M. Pons. 1999. Ecological changes in recent land-bridge islands in French Guiana, with emphasis on vertebrate communities. Biological Conservation 91: 213–222.

Council for Agricultural Science and Technology. 1974. Livestock grazing on federal lands in the 11 western states. Journal of Range Management 27: 174-181.

Covich, A.P., M.A. Palmer; T.A. Crowl. 1999. The role of benthic invertebrate species in freshwater ecosystems: zoobenthic species influence energy flows and nutrient cycling. BioScience 49(2): 119.

Cox, G. W. 1999. Alien species in North America and Hawaii : impacts on natural ecosystems. Island Press, Washington, D.C.

Crooks, K. R., and M. E. Soule. 1999. Mesopredator release and avifaunal extinctions in a fragmented system. Nature 400: 563-566.

Crump, M. 2011. Amphibians and Reptiles: An introduction to their natural history and conservation. McDonald and Woodward Publishing Co.

Crumpacker, D.W. 1984. Regional riparian research and a multi-university approach to the special problem of livestock grazing in the Rocky Mountains and Great Plains. Pages 413- 422 in R. E. Warner, and K. Hendrix, editors. California riparian systems: Ecology, conservation, and productive manage merit. University of California Press, Berkeley, California.

Cubbage, F.W. and C.H. Flather. 1993. Forested wetland distribution: A detailed look at the south. Journal of Forestry 91(5): 35-40.

Cummer, M. R., D. E. Green, and E. M. O'Neill. 2005. Aquatic chytrid pathogen detected in terrestrial plethodontid salamander. Herpetological Review 36: 248–249.

Cunningham, A.A., T.E.S. Langton, P.M. Bennett, J.F. Lewin, S.E.N. Drury, R.E. Gough, S.K. Macgregor. 1996a. Pathological and microbiological findings from incidents of unusual mortality of the common frog (Rana temporaria). Philo. Trans. Biol. Sci. 351: 1539-1557.

Cushman, S.A. 2006. Effects of habitat loss and fragmentation on amphibians: A review and prospectus. Biological Conservation 128: 231-40.

Czech, B., P. R. Krausman, and P.K. Devers. 2000. Economic associations among causes of species endangerment in the United States. BioSci. 50(7): 594

Dahl, T. E., and G. J. Allord. 1997. Technical aspects of wetlands: history of wetlands in the conterminous United States. National Water Summary on Wetland Resources, United States Geological Survey Water Supply Paper 2425, Washington, D.C., USA.

Dalrymple, G. H. and N. G. Reichenbach. 1984. Management of an endangered species of snake in Ohio, USA. Biological Conservation 30: 195-200.

Daszak P., L. Berger, A.A. Cunningham, A.D. Hyatt, D.E. Green, R. Speare. 1999. Emerging infectious diseases and amphibian population declines. Emerg. Infect. Dis. 5: 735-748.

Daszak, P., Cunningham, A.A. and Hyatt, A.D. 2000. Emerging infectious diseases of wildlife- threats to biodiversity and human health. Science 287: 443–449.

Daszak, P., Cunningham, A.A. and Hyatt, A.D. 2003. Infectious disease and amphibian population declines. Diversity and Distributions 9, in press.

Daszak, P., D.E. Scott, A.M. Kilpatrick, C. Faggioni, J.W. Gibbons, and D. Porter. 2005. Amphibian population declines at Savannah River Site are linked to climate, not chytridiomycosis. Ecology 86: 3232-3237.

Davidson, C., H. B. Shaffer, and M. R. Jennings. 2002. Spatial tests of the pesticide drift, habitat destruction, UV-B, and climate-change hypotheses for California amphibian declines. Conservation Biology 16: 1588-1601.

Davidson, C., M.F. Bernard, H.B. Shaffer, J.M. Parker, C. O'Leary, J.M. Conlon, and L.A. Rollins-Smith. 2007. Effects of chytrid and carbaryl exposure on survival, growth and skin peptide defenses in foothill yellow-legged frogs. Environmental Science and Technology 41: 1771-1776.

Davis, A.K., M.J. Yabsley, M.K. Keel, and J.C. Maerz. 2007. Discovery of a novel alveolate pathogen affecting southern leopard frogs in Georgia: description of the disease and host effects. EcoHealth 4(3): 310-317.

de Solla, S.R. and K.J. Fernie. 2004. Characterization of contaminants in snapping turtles (Chelydra serpentina) from Canadian Lake Erie Areas of Concern: St. Clair River, Detroit River, and Wheatley Harbour. Environmental Pollution 132: 101-112.

Deacon, J.E., A.E. Williams, C.D. Williams, and J.E. Williams. 2007. Fueling population growth in Las Vegas: how large-scale groundwater withdrawal could burn regional biodiversity. BioScience 57(8): 688-698.

DeDecker, M. 1991. Shrubs and flowering plants. pp. 108-241. *In:* C.A. Hall, Jr. (ed.), Natural History of the White-Inyo Range, Eastern California. California Nat. Hist. Guides (55), Univ. California Press, Berkeley, California.

DeGraaf R.M., Yamasaki M. 2002. Effects of edge contrast on redback salamander distribution in even-aged northern forests. Forest Science 48: 351–363.

Delis, P. 1993. Effects of urbanization of the community of anurans of a pine flatwood habitat in west central Florida. M.S. Thesis, University of South Florida, Tampa, FL.

Delis, P.R., H.R. Mushinsky and E.D. McCoy. 1996. Decline of some west-central Florida anuran populations in response to habitat degradation. Biodiversity and Conservation 12: 1579–1595.

deMaynadier, P. G., and M. L. Hunter. 1995. The relationship between forest management and amphibian ecology: a review of the literature. Environmental Reviews 3: 230–261.

deMaynadier, P. G., and M. L. Hunter. 1998. Effects of silvicultural edges on the distribution and abundance of amphibians in Maine. Conservation Biology 12: 340–352.

deMaynadier, P. G. and M. L. Hunter, Jr. 2000. Road effects on amphibian movements in a forested landscape. Natural Areas Journal 20: 56-65.

deMaynadier, P.G., and Hunter, M.L., Jr. 1995. The relationship between forest management and amphibian ecology: a review of the North American literature. Environ. Rev. 3: 230–261.

Densmore, C.L. and D.E. Green. 2007. ILAR Journal 48(3): 235-54.

Deullman, W.E.; Trueb, L. 1986. Biology of amphibians. New York: McGraw-Hill. 670 p.

Diemer Berish, J.E., L.D. Wendland, and C.A. Gates. 2000. Distribution and Prevalence of Upper Respiratory Tract Disease in Gopher Tortoises in Florida. Journal of Herpetology 34(1): 5-12.

Dimond, J.B., Kadunce, R.E., Getchell, A.S., Blease, J.A., 1968. DDT residue persistence in red-backed salamanders in a natural environment. Bulletin of Environmental Contamination and Toxicology 3: 194-202.

Dodd C. K., Jr., K. M. Enge, and J. N. Stuart. 1989. Reptiles on highways in north-central Alabama, USA. Journal of Herpetology 23: 197-200.

Dodd, C. K., and L. L. Smith. 2003. Habitat destruction and alteration: historical trends and future prospects for amphibians. Pages 94-112 in R. D. Semlitsch, editor. Amphibian Conservation. Smithsonian Institution, Washington.

Dodd, C. K., Jr. 1988. Disease and population declines in the flattened musk turtle *Sternotherus depressus*. American Midland Naturalist 119(2): 394-401.

Dodd, C.K. Jr. 1997. Imperiled Amphibians: A Historical Perspective. p. 165-200 *In:* Benz, G.W. and D.E. Collins (editors). 1997. Aquatic Fauna in Peril: The Southeastern Perspective. Southeast Aquatic Research Institute Special Publication 1, Lenz Design and Communications, Decatur, GA. 553 pp.

Dodd, C.K., Jr. 1990. Effects of habitat fragmentation on a stream-dwelling species, the flattened musk turtle *Sternotherus depressus*. Biological Conservation 54: 33-45.

Dodd, C.K., Jr. and B.S. Cade. 1998. Movement patterns and the conservation of amphibians breeding in small, temporary wetlands. Conservation Biology April 12: 331−339.

Donnelly, M. A., and M. L. Crump. 1998. Potential effects of climate change on two neotropical amphibian assemblages. Climate Change 39: 541-561.

Drake, J. A., and International Council of Scientific Unions. Scientific Committee on Problems of the Environment. 1989. Biological invasions : a global perspective. Published on behalf of the Scientific Committee on Problems of the Environment (SCOPE) of the International Council of Scientific Unions (ICSU) by Wiley, Chichester; New York.

Driscoll,D. A. 2004. Extinction and outbreaks accompany fragmentation of a reptile community. Ecological Applications 14: 220–240.

Duellman, W. E., and L. Trueb. 1986. Biology of amphibians. Johns Hopkins University Press, Baltimore, Maryland, USA.

Dunson, W. A., R. L. Wyman, and E. S. Corbett. 1992. A symposium on amphibian declines and habitat acidification. Journal of Herpetology 26(4): 349-442.

Dupuis, L. A., and F. L. Bunnell. 1999. Effects of stand age, size and juxtaposition on abundance of western redback salamanders (Plethodon vehiculum) in coastal British Columbia. Northwest Sci. 73(1):27–33.

Dupuis, L.A., J.N.M. Smith, and F. Bunnell. 1995. Relation of terrestrial-breeding amphibian abundance to tree-stand age. Conservation Biology 9: 645–653.

Early, R. and D. F. Sax. 2011. Ecol. Lett. 14: 1125.

Ehrlich, D. 1979. Predation by bullfrog tadpoles (*Rana catesbeiana*) on eggs and newly hatched larvae of the plains leopard frog (*Rana blairi*). Bulletin of the Maryland Herpetological Society 15: 25-26.

Elton, C. S. 1958. The ecology of invasions by animals and plants. Methuen, London, England.

Enge, K. M. and K. N. Wood. 2002. A pedestrian road survey of an upland snake community in Florida. Southeastern Naturalist 1: 365-380.

Enge, K. M., and W. R. Marion. 1986. Effects of clearcutting and site preparation on herpetofauna of a north Florida flatwoods. For. Ecol. & Mgmt. 14: 177-192.

Enge, K. M., M. M. Bentzien, and F. Percival. 1986. Florida scrub lizard status survey. Tech. Rept. No. 26 to U.S. Fish Wildl. Serv., Jacksonville End. Sp. Office, Florida, *available at* http://ufdc.ufl.edu/UF00067421/00001 (last visited June 26, 2012).

Erb, L. and M. Jones. 2011. Can Turtle Mortality be Reduced in Managed Fields? Northeastern Naturalist 18(4): 489.

Etnier, D.A. 1997. Jeopardized southeastern freshwater fishes. p. 87-104 *In:* Benz, G.W. and D.E. Collins (editors). 1997. Aquatic Fauna in Peril: The Southeastern Perspective. Southeast Aquatic Research Institute Special Publication 1, Lenz Design and Communications, Decatur, GA. 553 pp.

Fahrig, L. 1997. Relative effects of habitat loss and fragmentation on population extinction. J. Wildl. Manage. 61: 603–610.

Fahrig, L., and G. Merriam. 1994. Conservation of fragmented populations. Conservation Biology 8: 50–59.

Fahrig, L., J. H. Pedlar S. E. Pope P. D. Taylor and J. F. Wegner. 1995. Effect of Road Traffic on Amphibian Density. Biological Conservation 73: 177-182.

Fellers, G. M. 2005. *Rana boylii* Baird, 1854(b). Pages 534-536 in M. Lannoo, editor. Amphibian declines: the conservation status of United States species. University of California Press, Berkeley.

Fellers, G. M., D. E. Green, and J. E. Longcore. 2001. Oral chytridiomycosis in the mountain yellow-legged frog (Rana muscosa). Copeia 2001: 945-953.

Figiel, C. R., Jr., and R. D. Semlitsch. 1990. Population variation in survival and metamorphosis of larval salamanders (Ambystoma maculatum) in the presence and absence of fish predation. Copeia 1990: 818-826.

Findlay, C. S. and J. Houlahan. 1997. Anthropogenic correlates of species richness in southeastern Ontario wetlands. Conservation Biology 11: 1000-1009.

Fisher, R. N., and H. B. Shaffer. 1996. The decline of amphibians in California's Great Central Valley. Conservation Biology 10(5): 1387-1397.

Fitch, H. S. 1999. A Kansas Snake Community: Composition and Changes Over 50 Years. Krieger Publishing Company, Melbourne, FL.

Fite, K. V., A. R. Blaustein, L. Bengston, and H. E. Hewitt. 1998. Evidence of retinal light damage in Rana cascadae: A declining amphibian species. Copeia 1998: 906-914.

Fitzgerald LA, Painter CW. 2000. Rattlesnake commercialization: Long-term trends, issues, and implications for conservation. Wildlife Society Bulletin 28: 235–253.

Fleischner, T.L. 1994. Ecological Costs of Livestock Grazing in Western North America. Conservation Biology 8(3): 629-644.

Florida Fish and Wildlife Conservation Commission (FFWCC). 2005. Florida's wildlife legacy initiative. Florida's comprehensive wildlife strategy. Florida Fish and Wildlife Conservation Commission, Tallahassee (FL).

Florida Fish and Wildlife Conservation Commission. 2011. Alligator Snapping Turtle Biological Status Review Report, *available at* http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=1&ved=0CCcQFjAA&url=http%3A%2F%2Fmyfwc.com%2Fmedia%2F1351487%2FAlligator%2520Snapping%2520Turtle%2520Final%2520BSR.pdf&ei=-Ss0T6eLE6jg2AWi1P2WAg&usg=AFQjCNGYayvoVFuOCrP0Rbu6ANBojCeE5Q (last visited Feb. 9, 2012).

Florida Fish and Wildlife Conservation Commission. 2011. Key Ringneck Snake Biological Status Review Report, *available at* http://www.myfwc.com/media/1351568/Key%20Ringneck%20Snake%20Final%20BSR.pdf

Florida Fish and Wildlife Conservation Commission. 2011. Gopher Frog Biological Status Review Report, *available at* http://www.myfwc.com/media/1351559/Gopher%20Frog%20Final%20BSR.pdf (last visited Oct. 14, 2011).

Florida Fish and Wildlife Conservation Commission (FFWCC). 2011b. Florida's Endangered and Threatened Species, *available at* http://myfwc.com/media/1515251/Threatened_Endangered_Species.pdf (last visited Oct. 4, 2011).

Folkerts, G.W. 1997. State and fate of the world's aquatic fauna. p. 1-16 *In:* Benz, G.W. and D.E. Collins (editors). 1997. Aquatic Fauna in Peril: The Southeastern Perspective. Southeast Aquatic Research Institute Special Publication 1, Lenz Design and Communications, Decatur, GA. 553 pp.

Fontenot, L. W., G. P. Noblet, and S. G. Platt. 1994. Rotenone hazards to amphibians and reptiles. Herpetological Review 25(4): 150-156.

Ford, W.M., Chapman, B.R., Menzel, M.A., Odom, R.H., 2002. Stand age and habitat influences on salamanders in Appalachian cove hardwood forests. Forest Ecology and Management 155: 131–141.

Forman, R. T. T. and L. E. Alexander. 1998. Roads and their major ecological effects. Annual Review of Ecology and Systematics 29: 207-231.

Forman, R. T. T. and R. D. Deblinger. 2000 . The ecological road-effect zone of a Massachusetts (U.S.A.) suburban highway. Conservation Biology 14: 36-46.

Fowle, S. C. 1996. Effects of roadkill mortality on the western painted turtle (*Chrysemys picta belli*) in the Mission Valley, western Montana. Pp. 205-223 *In* G.L. Evink, P. Garrett, D. Zeigler and J. Berry (eds.) Trends in Addressing Transportation Related Wildlife Mortality, Proceedings of the Transportation Related Wildlife Mortality Seminar. State of Florida Department of Transportation, Tallahassee, Florida. FL - ER - 58 - 96

Fowler, D. F., D. M. Hill, and L. J. Fowler. 1985. Colonization of coal surface mine sediment ponds in southern Appalachia by aquatic organisms and breeding amphibians. *In* Brooks R. P., J. B. Hill, F. J. Brenner, and S. Carpets (eds.), Wetlands and Water Management on Mined Lands: Proceedings of a Conference, pp. 261-280. The Pennsylvania State University, State College.

Franke, J., T.M. Telecky. 2001. Reptiles as Pets: An Examination of the Trade in Live Reptiles in the United States. Washington (DC): Humane Society of the United States.

Franklin, I. R. 1980. Evolutionary changes in small populations. In Conservation Biology: An evolutionary-ecological perspective (ed. a. B. A. W. M. E. Soule). pp. 135-149.

Fraser, D. F. 1976. Empirical evaluation of the hypothesis of food competition in salamanders of the genus Plethodon. Ecology 57: 459–471.

Freda, J., and D. G. McDonald. 1990. Effects of aluminum on the Leopard Frog Rana pipiens: Life stage comparisons and aluminum uptake. Canadian Journal of Fisheries & Aquatic Sciences 47: 210-216.

Freda, J., V. Cavdek, and D. G. McDonald. 1990. Role of Organic Complexation in the Toxicity of Aluminum to Rana-Pipiens Embryos and Bufo-Americanus Tadpoles. Canadian Journal of Fisheries & Aquatic Sciences 47: 217-224.

Freed, P. S. and K. Neitman. 1998. Notes on predation on the endangered Houston toad, *Bufo houstonensis*. Texas J. Sci. 40: 454–456.

Frest, T.J. 2002. Native snails: Indicators of Ecosystem Health. p. 211-215, in: Welfare Ranching, G. Wuerthner and M. Matteson (Eds.). Island Press, Sausalito, California.

Frest, T.J. and E.J. Johannes. 1995. Interior Columbia Basin Mollusk Species of Special Concern. Final Report prepared for Interior Columbia Basin Ecosystem Management Project Contract # 43-OE00-4-9112. January 15, 1995. Deixis Consultants, Seattle, WA. 274 pp.

Gabor, C. R. 1995. Correlational test of Mathis' hypothesis that bigger salamanders have better territories. Copeia 1995: 729–735.

Garber, S.D., and J. Burger. 1995. A 20-yr study documenting the relationship between turtle decline and human recreation. Ecological Applications 5(4): 1151-1162.

Garcia, T.S., J.M. Romansic, and A.R. Blaustein. 2006. Survival of three species of anuran metamorphs exposed to UV-B radiation and the pathogenic fungus Batrachochytrium dendrobatidis. Diseases of Aquatic Organisms; 72(2): 163-169.

Gaulke, C.A., J.T. Irwin and R.S. Wagner. 2011. Prevalence and Distribution of *Batrachochytrium dendrobatidis* at Montane Sites in Central Washington State, USA. Herpetol. Rev. 42(2): 209-211.

Gendron, A. D., C. A. Bishop, R. Fortin, and A. Hontela. 1997. In vivo testing of the functional integrity of the corticosterone-producing axis in mudpuppy (Amphibia) exposed to chlorinated hydrocarbons in the wild. Environmental Toxicology and Chemistry 16(8): 1694-1706.

Gendron, A. D., D. J. Marcogliese, S. Barbeau, M. S. Christin, P. Brousseau, S. Ruby, D. Cyr, and M. Fournier. 2003. Exposure of leopard frogs to a pesticide mixture affects life history characteristics of the lungworm Rhabdias ranae. Oecologia 135: 469-476.

Germano, D. J., and C. R. Hungerford. 1981. Reptile population changes with manipulation of Sonoran Desert shrub. Great Basin Naturalist 41: 129-138.

Gervasi S.S., J. Foufopoulos. 2008. Costs of plasticity: responses to desiccation decrease post-metamorphic immune function in a pond-breeding amphibian. Functional Ecology 22: 100-108.

Gibbons, J.W., D.E. Scott, T.J. Ryan, K.A. Buhlman, T.D. Tuberville, B.S. Mettis, J.L. Greene, T. Mills, Y. Leiden, S. Poppy, and C.T. Winne. 2000. The global decline of reptiles, déjà vu amphibians. BioScience. 50(8): 653-666. August.

Gibbs, J. P., and A. R. Breisch. 2001. Climate warming and calling phenology of frogs near Ithaca, New York, 1900-1999. Conservation Biology 15: 1175-1178.

Gibbs, J. P., and W. G. Shriver. 2002. Estimating the effects of road mortality on turtle populations. Conservation Biology 16: 1647-1652.

Gibbs, J.P., 1993. Importance of small wetlands for the persistence of local populations of wetland-associated animals. Wetlands 13: 25–31.

Gilliom R.J. et al. 2006. The Quality of Our Nation's Waters: Pesticides in the Nation's Streams and Ground Water, 1992–2001. U.S. Geological Survey Circular 1291.

Glista, D.J., T.L. Devaul, and J.A. Dewoody. 2007. Vertebrate Road Mortality Predominantly Impacts Amphibians. Herpetological Conservation and Biology 3(1): 77-87.

Godley, J.S., S.M. Gonzalez, and M.C. Gonzalez. 2008. *Stilosoma extenuatum* (short-tailed snake). Diet and predation. Herpetological Review 39: 473–474.

Goodsell, J., A., and L. Kats, B. 1999. Effect of introduced mosquitofish on Pacific treefrogs and the role of alternative prey. Conservation Biology 13: 921-924.

Gore, J. A. 1983. The distribution of Desmognathine larvae (Amphibia: Plethodontidae) in coal surface mine impacted streams of the Cumberland Plateau, USA. Journal of Freshwater Ecology 2: 13-23.

Greene, H. W. 1997. Snakes: the evolution of mystery and nature. University of California Press, Berkeley, California.

Grialou J.A., S.D. West, and R.N. Wilkins. 2000. The effects of forest clearcut harvesting and thinning on terrestrial salamanders. Journal of Wildlife Management 64: 105–113.

Guillette, L. J., Jr., T. S. Gross, G. R. Masson, J. M. Matter, H. F. Percival, and A. R. Woodward. 1994. Developmental abnormalities of the gonad and abnormal sex hormone concentrations in juvenile alligators from contaminated and control lakes in Florida. Environmental Health Perspectives 102: 680-688.

Hafner, C.L., and M .C. Brittingham. 1993. Evaluation of a steam-bank fencing program in Pennsylvania. Wildlife Society Bulletin 21: 307-315.

Hall, R. J., and P. F. P. Henry. 1992. Assessing effects of pesticides on amphibians and reptiles: status and needs. Herpetological Journal 2: 65-71.

Hall, R.J., and D.R. Clark. 1982. Responses of the iguanid lizard *Anolis carolensis* to four organophosphorus pesticides. Env. Pollution (Series A) 28: 45-52.

Hall, R.J., R.E. Ashton, and R.M. Prouty. 1985. Pesticide and PCB residues in the Neuse River waterdog, Necturus lewisi. Brimleyana 10: 107-109.

Halley, J. M., R. S. Oldham, and J. W. Arntzen. 1996. Predicting the persistence of amphibian

Halpin, P.N. 1997. Global climate change and natural-area protections: management responses and research directions. Ecological Applications. 7(3): 828-843.

Hamer, A. J., and M. J. McDonnell. 2008. Amphibian ecology and conservation in the urbanising world: A review. Biological Conservation 141: 2432-2449.

Hamilton, M. S. 2002. Effects of developmental activities on streamside salamander communities in Boone County, West Virginia. M. S. Thesis. Marshall University, Huntington, West Virginia. 81 pp.

Hammerson, G. A. 1982. Bullfrog eliminating leopard frogs in Colorado? Herpetological Review 13: 115-116.

Hammerson, G., and J. Jensen 2004. *Lithobates capito*. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.1. <www.iucnredlist.org>. Downloaded on 14 October 2011.

Hanski, I., 1999. Metapopulation Ecology. OxfordUniversity Press, New York.

Hanski, I., and Gilpin, M. 1991. Metapopulation dynamics: brief history and conceptual domain. Biol. J. Linn. Soc. 42: 3–16.

Harding, J. H. 1985. Life history notes: *Clemmys insculpta*: predation - mutilation. Herp. Rev. 16(1): 30.

Harding, J.H. and T.J. Bloomer. 1979. The Wood turtle, *Clemmys insculpta* . . . a natural history. Bulletin of the New York Herpetological Society 15: 9-26.

Harfenist, A., T. Power, K. L. Clark, and D. B. Peakall. 1989. A review and evaluation of the amphibian toxicological literature. Canadian Wildlife Service Technical Report Series 61. Canadian Wildlife Service, Ottawa, Canada. 222pp.

Harper, E.B. 2007. The role of terrestrial habitat in the population dynamics and conservation of pond-breeding amphibians. PhD dissertation. University of Missouri, Columbia.

Harpole, D. N., and C. A. Haas. 1999. Effects of seven silvicultural treatments on terrestrial salamanders. Forest Ecology and Management 114: 245–252.

Hatch, A. C., and A. R. Blaustein. 2000. Combined effects of UV-B, nitrate, and low pH reduce the survival and activity level of larval cascades frogs (*Rana cascadae*). Archives of Environmental Contamination and Toxicology 39: 494-499.

Havlick, D.G. 2002. No Place Distance: Roads and Motorized Recreation on America's Public Lands. Island Press. Washington D.C. 297 pp.

Hawkes, L.A., A.C. Broderick, M.H. Godfrey, and B.J. Godley. 2007. Investigating the potential impacts of climate change on a marine turtle population. Global Change Biology. 13: 923-932.

Hawkins, C. P., M. L. Murphy, N. H. Anderson, and M. A. Wilzbach. 1983. Density of fish and salamanders in relation to riparian canopy and physical habitat in streams of the northwestern United States. Canadian Journal of Fisheries and Aquatic Sciences 40: 1173–1185.

Haxton, T. 2000. Road mortality of snapping turtles, *Chelydra serpentina*, in central Ontario during their nesting period. Canadian Field Naturalist 114: 106-110.

Hayes, T.B., A.A. Stuart, M. Mendoza, A. Collins, N. Noriega, A. Vonk, G. Johnson, R. Liu, and D. Kpodzo. 2006. Characterization of atrazine-induced gonadal malformations in African Clawed Frogs (Xenopus laevis) and comparisons with effects of an androgen antagonist (cyproterone acetate) and exogenous estrogen (17 beta-estradiol): Support for the demasculinization/feminization hypothesis. Environmental Health Perspectives 114: 134-141.

Hayes, T. B., A. Collins, M. Lee, M. Mendoza, N. Noriega, A. A. Stuart, and A. Vonk. 2002b. Hermaphroditic, demasculinized frogs after exposure to the herbicide atrazine at low ecologically relevant doses. Proceedings of the National Academy of Sciences of the United States of America 99: 5476-5480.

Hayes, T., K. Haston, M. Tsui, A. Hoang, C. Haeffele, and A. Vonk. 2002a. Herbicides: Feminization of male frogs in the wild. Nature 419: 895-896.

Hayes, T., K. Haston, M. Tsui, A. Hoang, C. Haeffele, and A. Vonk. 2003. Atrazine-induced hermaphroditism at 0.1 ppb in American Leopard Frogs (Rana pipiens): Laboratory and field evidence. Environmental Health Perspectives [print] 111: 568-575.

Hayes, T.B., P. Case, S. Chui, D. Chung, C. Haeffele, K. Haston, M. Lee, V.P. Mai, Y. Marjuoa, J. Parker, M. Tsui. 2006. Pesticide mixtures, endocrine disruption, and amphibian declines: are we underestimating the impact? Environ Health Perspect. 114: 40-50.

Hays, J. B., A. R. Blaustein, J. M. Kiesecker, P. D. Hoffman, I. Pandelova, D. Coyle, and T. Richardson. 1996. Developmental response to amphibians to solar and artificial UVB sources: A comparative study. Photochemistry and Photobiology 64: 449-456.

Hecnar, S.J., and M'Closkey, R.T., 1996. Regional dynamics and the status of amphibians. Ecology 77: 2091–2097.

Hecnar, S. J. and R. T. M'Closkey. 1997. Changes in the composition of a Ranid Frog Community Following Bullfrog Extinction. American Midland Naturalist 137(1): 145-150.

Hecnar, S. J., and R. T. M'Closkey. 1997. The effects of predatory fish on amphibian species richness and distribution. Biological Conservation 79: 123-131.

Hels, T. and E. Buchwald. 2001. The effect of road kills on amphibian populations. Biological Conservation 99: 331-340.

Hendrickson, D. A. and W. L. Minckley. 1984. Ciénegas-vanishing climax communities of the American Southwest. Desert Plants 6: 131-175.

Henke, S. E., and F. C. Bryant. 1999. Effects of coyote removal on the faunal community in western Texas. Journal of Wildlife Management 63: 1066-1081.

Heppell, S. S. 1998. Application of life-history theory and population model analysis to turtle conservation. Copeia. 1998: 367-375.

Herbeck, L.A., and D.R. Larsen. 1999. Plethodontid salamander response to silvicultural practices in Missouri Ozark Forests. Conservation Biology 13: 623–632.

Herrig, J. and P. Shute. 2002. Chapter 23: aquatic animals and their habitats. Southern Region, USDA Forest Service and Tennessee Valley Authority. 45 pp. *In:* Wear, David N.; Greis, John G., eds. 2002. Southern forest resource assessment. Gen. Tech. Rep. SRS-53. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 635 pp., *available at* www.srs.fs.usda.gov/sustain/report/pdf/chapter_23e.pdf (last visited March 23, 2012).

Highton, R. 1956. Systematics and variation of the endemic Florida snake genus *Stilosoma*. Bulletin of the Florida State Museum, Biological Sciences 1: 73–96.

Himes, J.G., L.M. Hardy, D.C. Rudolph, and S.J. Burgdorf. 2002. Growth rates and mortality of the Louisiana Pine Snake (Pituophis ruthveni). Journal of Herpetology 36(4): 683-687.

Hitchings, S. P. and T. J. C. Beebee. 1997. Genetic substructuring as a result of barriers to gene flow in urban Common Frog populations: implications for biodiversity conservation. Heredity 79: 1l7- 127.

Hitchings, S.P., and T.J.C. Beebee. 1998. Loss of genetic diversity and fitness in common toad (Bufo bufo) populations isolated by inimical habitat. J. Evol. Biol. 11: 269–283.

Hogrefe, T.C. 2005. Boreal Toad (Bufo boreas boreas) Conservation Plan in the State of Utah. Utah Division of Wildlife Resources. 65 pp.

Hokit, D. G., B. M. Stith, and L. C. Branch. 1999. Effects of landscape structure in Florida scrub: a population perspective. Ecological Applications 9: 124-134.

Holland, D. C. 1991. A synopsis of the ecology and status of the western pond turtle (*Clemmys marmorata*) in 1991. Report prepared for the United States Fish and Wildlife Service, National Ecology Research Center, San Simeon Field. Station, San Simeon, California.

Holland, C.C., J. Honea, S.E. Gwin, and M.E. Kentula. 1995. Wetland degradation and loss in the rapidly urbanizing area of Portland, Oregon. Wetlands 15(4): 1995.

Homyack, Jessica A., and Carola A. Haas. 2009. Long-term effects of experimental forest harvesting on abundance and reproductive demography of terrestrial salamanders. Biological Conservation 142: 110-121.

Hoover C. 1998. The U.S. Role in the International Live Reptile Trade: Amazon Tree Boas to Zululand Dwarf Chameleons.Washington (DC): TRAFFIC North America,WWF. Jenkins MD. 1995.Tortoises and Freshwater Turtles: The Trade in Southeast Asia. Cambridge (United Kingdom): TRAFFIC International.

Hopey, M. E., and J. W. Petranka. 1994. Restriction of wood frogs to fish-free habitats: how important is adult choice? Copeia 1994: 1023-1025.

Houlahan, J. E., and C.S. Findlay. 2003. The effects of adjacent land use on wetland amphibian species richness and community composition. Canadian Journal of Fisheries and Aquatic Sciences 60(9): 1078-1094.

Houlahan, J.E., Findlay, C.S., Schmidt, B.R., Meyer, A.H., and Kuzmin, S.L. 2000. Quantitative evidence for global amphibian population declines. Nature (Lond.) 404: 752–755.

Hughes JW, and T.J. Fahey. 1994. Litterfall dynamics and ecosystem recovery during forest development. Forest Ecology and Management 63: 181–198.

Huhta, V. 1976. Effects of clear-cutting on numbers, biomass and community respiration of soil invertebrates. Annales Zoologici Fennici 13: 63–80.

Huston, R. 2009. Biological Evaluation for Two Mile Watershed Project, *available at* http://www.fs.fed.us/outernet/r8/ouachita/projects/two_mile/Two_Mile_Biological_Evaluation.doc (last visited Jan. 31, 2012).

Hutson, S. S., N. L. Barber, J. F. Kenny, K. S. Linsey, D. S. Lumia, And M. A. Maupin. 2005. Estimated use of water in the United States in 2000. U.S. Geological Survey Circular 1268. US Geological Survey, Denver, Colorado.

Irwin, L.K., S.L. Gray, E. Oberdorster. 2001. Vitellogenin induction in painted turtle, Chrysemys picta, as a biomarker of exposure to environmental levels of estradiol. Aquat Toxicol 55: 49–60.

IUCN 2011. IUCN Red List of Threatened Species. Version 2011.2, *available at* www.iucnredlist.org (last visited April 3, 2012).

Iverson, J.B. and G.R Smith. 1993. Reproductive ecology of the Painted Turtle (Chrysemys picta) in the Nebraska sandhills and across its range. Copeia 1993: 1–21.

Jackson, R. B., S. R. Carpenter, C.N. Dahm, D. M. Mcknight, R. J. Naiman, S. L. Postel, and S. W. Running. 2001.Water in a changing world. Ecological Applications 11: 1027–1045.

Jacobson, E.R. 1993. Implications of infectious diseases for captive propagation and introduction programs of threatened/endangered reptiles. Journal of Zoo and Wildlife Medicine 24: 245-55.

Jacobson, E.R. 1994. Causes of mortality and disease in tortoises: a review. Journal of Zoo and Wildlife Medicine 25: 2 –17.

Jaeger, R.G. 1980. Fluctuations in prey availability and food limitation for a terrestrial salamander. Oecologia 44: 335–341.

Jancovich, J.K., E.W. Davidson, J.F. Morado, B.L. Jacobs, J.P. Collins. 1997. Isolation of a lethal virus from the endangered tiger salamander Ambystoma tigrinum stebbinsi. Dis Aquat Organ 31: 161-167.

Jennings, M. R., and M. P. Hayes. 1985. Pre-1900 overharvest of California red-legged frogs (*Rana aurora draytoni*): the inducement for bullfrog (Rana catesbeiana) introduction. Herpetologica 41: 94-103.

Jenkins, M., and S. Broad. 1994. International Trade in Reptile Skins: A Review and Analysis of the Main Consumer Markets, 1983–1991.Cambridge (United Kingdom): TRAFFIC International.

Jennings, M. R., D. F. Bradford, and D. F. Johnson. 1992. Dependence of the garter snake *Thamnophis elegans* on amphibians in the Sierra Nevada of California. Journal of Herpetology 26: 503-505.

Jennings, W.B. 1997. Habitat Use and Food Preferences of the Desert Tortoise, Gopherus agassizii, in the Western Mojave Desert and Impacts of Off-Road Vehicles. In: Proceedings of

Conservation, Restoration, and Management of Tortoises and Turtles—An International Conference 1997: 42-45.

Jennings, M. R., and M. P. Hayes. 1994. Amphibian and reptile species of special concern in California. Final Report submitted to the California Department of Fish and Game, Inland Fisheries Division. Contract No. 8023. 255 pp.

Jensen, J.B. 1998. Distribution and status of the alligator snapping turtle (*Macroclemys temminckii*) in Georgia. Paper presented at the 78th Annual Meeting of the American Society of Ichthyologists and Herpetologists; 16-22 Jul 19989, University of Guelph, Ontario, Canada.

Jensen, J.B. and C. Camp. 2005. *Plethodon petraeus* - Pigeon Mountain Salamander. In M. Lannoo, editor. Amphibian declines: the conservation status of United States species. University of California Press, Berkeley.

Jensen, J.B. and S. Richter. 2005. Gopher Frog (*Rana capito*). Status and Conservation of U.S. Amphibians. Volume 2: Species Accounts. Lannoo, M.J.,editor. University of California Press. Berkeley, California.

Jochimsen, M. Denim, C.R. Peterson, K.M. Andrews, and J.W. Gibbons. 2004. A Literature Review of the Effects of Roads on Amphibians and Reptiles and the Measures Used to Minimize Those Effects. Idaho Fish and Game Department USDA Forest Service, 78 pp.

Jodoin, Y., C. Lavoie, P. Villeneuve, M. Theriault, J. Beaulieu, and F. Belzile. 2008. Highways as corridors and habitats for the invasive common reed Phragmites australis in Quebec, Canada. Journal of Applied Ecology 45: 459-466.

Johnson, P.T.J., K.B. Lunde, E.G. Ritchie, A.E. Launer. 1999. The effect of trematode infection on amphibian limb development and survivorship. Science 284: 802-804.

Johnson, C. R., and J. E. Prine. 1976. The effects of sublethal concentrations of organophosphorus insecticides and an insect growth regulator on temperature tolerance in hydrated and dehydrated juvenile western toads, *Bufo boreas*. Comparative Biochemistry and Physiology 53A: 147-149.

Johnson, J. E., D. W. Smith, and J. A. Burger. 1985. Effects on the forest floor of whole-tree harvesting in an Appalachian oak forest. American Midland Naturalist 114: 51–61.

Johnson, P. T. J., Lunde, K. B., Ritchie, E. G., and Launer, A. E. 1999. The effect of trematode infection on amphibian limb development and survivorship. Science (Washington D. C.) 284: 802-804.

Johnson, P.T. and D.R. Sutherland. 2003. Amphibian deformities and Ribeiroia infection: an emerging helminthiasis. Trends in Parasitology 19(8): 332-335.

Johnson, P.T.J., K.B. Lunde, E.M. Thurman, E.G. Ritchie, S.N. Wray, D.R. Sutherland, J.M. Kapfer, T.J. Frest, J. Bowerman, and A.R. Blaustein. 2002. Parasite (Ribeiroia ondatrae) infection linked to amphibian malformations in the western United States. Ecol. Monogr. 72: 151–168.

Johnston, B., and L. Frid. 2002. Clearcut logging restricts the movements of terrestrial Pacific giant salamanders (Dicamptodon tenebrosus Good). Can. J. Zool. 80: 2170–2177.

Joly, P., Miaud, C., Lehmann, A., and Grolet, O. 2001. Habitat matrix effects on pond occupancy in newts. Conserv. Biol. 15: 239–248.

Jolley, D. Buck, Stephen S. Ditchkoff, Bill D. Sparklin, Laura B. Hanson, Michael S. Mitchell, and James B. Grand. 2010. Estimate of herpetofauna depredation by a population of wild pigs. Source: Journal of Mammalogy, 91(2): 519-524. 2010.

Jones, K.B. 1981. Effects of grazing on lizard abundance and diversity in western Arizona. Southwestern Naturalist 26: 107-115.

Jones, K.B. 1988. Comparison of herpetofaunas of a natural and altered riparian ecosystem. Pages 222-227 in R. C. Szaro, K. E. severson, and D. P,. Patton, technical coordinators. Management of amphibians, reptiles, and small mammals in North America. General Technical Report RM-166. U.S. Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado.

Jones, C.G., J.H. Lawton, and M. Shachak. 1997. Positive and negative effects of organisms as ecosystem engineers. Ecology 78: 1946–1957.

Jones, J. A., F. J. Swanson, B. C. Wemple, and K. U. Snyder. 2000. Effects of roads on hydrology, geomorphology, and disturbance patches in stream networks. Conservation Biology 14: 76–85.

Jones, K.B., K.H. Riitters, J.D. Wickham, R.D. Tankersley, R.V. O'Neill, D.J. Chaloud, E.R. Smith, and A.C. Neale. 1997. An Ecological Assessment of the United States Mid-Atlantic Region: A Landscape Atlas. USEPA/600/R-97/130. U.S. Environmental Protection Agency. Washington, D.C. 104 pp.

Joseph, M.B., J. Piovia-Scott, S.P. Lawler, and K.L. Pope. 2010. Indirect effects of introduced trout on Cascades frogs (Rana cascadae) via shared aquatic prey. Freshwater Biology (2010).

Karl, T.R., J.M. Melillo, and T.C. Peterson (eds.). 2009. Global Climate Change Impacts in the United States. New York, NY: Cambridge University Press.

Karraker, N.E., and H.H. Welsh Jr. 2006. Long-term impacts of even-aged timber management on abundance and body condition of terrestrial amphibians in northwestern California. Biological Conservation 131: 132–140.

Karraker, Nancy E., and James P. Gibbs. 2011. Road deicing salt irreversibly disrupts osmoregulation of salamander egg clutches. Environmental Pollution 159 (2011): 833-835.

Kats, L. B., J. M. Kiesecker, D. P. Chivers, and A. R. Blaustein. 2000. Effects of UV-B radiation on anti-predator behavior in three species of amphibians. Ethology 106: 921-931.

Kats, L.B. and R.P. Ferrer. 2003. Alien predators and amphibian declines: review of two decades of science and the transition to conservation. Diversity and Distributions 9(2): 99-110.

Kauffman, B., and W C. Kruger. 1984. Livestock impacts on riparian ecosystems and streamside management implications: a review. Journal of Range Management 37: 430-437.

Keenan R.J., and J.P. Kimmins. 1993. The ecological effects of clear-cutting. Environmental Review 1: 121–144.

Kerbes, R. H., P. M. Kotanen, and R. J. Jefferies. 1990. Destruction of wetland habitats by lesser snow geese: a keystone species on the west coast of Hudson Bay. Journal of Applied Ecology 27: 242-258.

Kiesecker, J. M., and A. R. Blaustein. 1997a. Influences of egg laying behavior on pathogenic infection of amphibian eggs. Conservation Biology 11: 214-220.

Kiesecker, J. M., and A. R. Blaustein. 1997b. Population differences in responses of red-legged frogs (*Rana aurora*) to introduced bullfrogs. Ecology 78: 1752-1760.

Kiesecker, J. M., and A. R. Blaustein. 1998. Effects of introduced bullfrogs and smallmouth bass on microhabitat use, growth, and survival of native red-legged frogs (Rana aurora). Conservation Biology 12: 776-787.

Kiesecker, J. M., A. R. Blaustein, and C. L. Miller. 2001. Transfer of a pathogen from fish to amphibians. Conservation Biology 15: 1064-1070.

Kiesecker, J. M., A. R. Blaustein, and L. K. Belden. 2001. Complex causes of amphibian population declines. Nature 410: 681-684.

Kiesecker, J.M., L.K. Belden, L. Shea, M.J. Rubbo. 2004. Amphibian decline and emerging disease. Am Scientist 92: 138-147.

King, J.J. and R.S. Wagner. 2010. Toxic effects of the herbicide Roundup Regular on Pacific Northwestern amphibians. Northwestern Naturalist 91: 318-324.

Kjoss, V. A. and J. A. Litvaitis. 2001. Community structure of snakes in a human-dominated landscape. Biological Conservation 98: 285-292.

Klauber, L. M. 1939. Studies of reptile life in the arid southwest, Part 1. Night collecting on the desert with ecological statistics. Bulletin of the Zoological Society of San Diego 14: 2-64.

Klemens, M.W., and J.B. Thorbjarnarson. 1995. Reptiles as a food source. Biodivers. Conserv. 4: 281-98.

Kline, N. C. and D. E. Swann. 1998. Quantifying Wildlife Road Mortality in Saguaro National Park. Pp. 23-31 *In*: G.L. Evink, P. Garrett, D. Zeigler and J. Berry (eds.) Proceedings of the International Conference on Wildlife Ecology and Transportation. Florida Department of Transportation, Tallahassee, Florida.

Knapp S.M., C.A. Haas, D.N. Harpole, R.L. Kirkpatrick. 2003. Initial effects of clearcutting and alternative silvicultural practices on terrestrial salamander abundance. Conservation Biology 17: 752–762.

Knapp, R.A. 2005. Effects of nonnative fish and habitat characteristics on lentic herpetofauna in Yosemite National Park, USA. Biological Conservation 121: 265-279.

Knapp, R.A., and K.R. Matthews. 2000. Non-native fish introductions and the decline of the Mountain Yellowlegged Frog from within protected areas. Conservation Biology 14: 428-438.

Knutson, M. G., J. R. Sauer, D. A. Olsen, M. J. Mossman, L. M. Hemesath, and M. J. Lannoo. 1999. Effects of landscape composition and wetland fragmentation on frog and toad abundance and species richness in Iowa and Wisconsin, USA.

Koch, E. D., G. Williams, C. R. Peterson, and P. S. Corn. 1996. Conference on declining and sensitive amphibians in the Northern Rockies and the Pacific Northwest: a summary paper. Idaho

Herpetological Society Technical Bulletin and U.S. Fish and Wildlife Service Report, Boise, Idaho.

Kolozsvary, M.B., and R.K. Swihart. 1999. Habitat fragmentation and the distribution of amphibians: patch and landscape correlates in farmland. Can. J. Zool. 77: 1288–1299.

Kolpin D.W., E.T. Furlong, M.T. Meyer, E.M. Thurman, S.D. Zaugg, L.B. Barber et al. 2002. Pharmaceuticals, hormones, and other organic wastewater contaminants in U.S. streams, 1999–2000: a national reconnaissance. Environ Sci Technol 36: 1202–1211.

Korschgen, L. J., and T. S. Baskett. 1963. Foods of impoundment and stream dwelling bullfrogs in Missouri. Herpetologica 19: 88-99.

Kucken, D. J., J. S. Davis, J. W. Petranka, and C. K. Smith. 1994. Anakeesta stream acidification and metal contamination: effects on a salamander community. Journal of Environmental Quality 23: 1311–1317.

Kuhn, J. 1987. Strassentod der erdkroete (*Bufo bufo L.*): verlustquoten und verhehsaufkommen, verhalten auf der strasse. Naturschutz und Landschaftspflege in Baden - Wurttemberg 41: 175-186.

Kupferberg, S. J. 1994. Exotic larval bullfrogs (*Rana catesbeiana*) as prey for native garter snakes: functional and conservation implications. Herpetological Review 25(3): 95-97.

Kupferberg, S. J. 1996. The ecology of native tadpoles (*Rana boylii* and *Hyla regilla*) and the impacts of invading bullfrogs (*Rana catesbeiana*) in a northern California river. Dissertation. University of California, Berkeley, California, USA.

Kupferberg, S. J. 1997. Bullfrog (*Rana catesbeiana*) invasion of a California river: the role of larval competition. Ecology 78: 1736-1751.

Kupferberg, S., Catenazzi, A., Lunde, K., Lind, A., and Palen, W. 2009a. [North Coast] Parasitic copepod (*Lernaea cyprinacea*) outbreaks in Foothill Yellow-legged Frogs (*Rana boylii*) linked to unusually warm summers and amphibian malformations in Northern California. Copeia 2009(3): 529-537.

Laan, R., and B. Verboom. 1990. Effects of pool size and isolation on amphibian communities. Biological Conservation 54: 251-262.

LaClaire, L.V. 1997. Amphibians in Peril: Resource Management in the Southeast. p. 307-321 *In:* Benz, G.W. and D.E. Collins (editors). 1997. Aquatic Fauna in Peril: The Southeastern Perspective. Southeast Aquatic Research Institute Special Publication 1, Lenz Design and Communications, Decatur, GA. 553 pp.

Laurance, W. F., T. E. Lovejoy, H. L. Vasconcelos, E. M. Bruna, R. K. Didham, P. C. Stouffer, C. Gascon, R. O. Bierregaard, S. G. Laurance, and E. Sampaio. 2002. Ecosystem decay of Amazonian forest fragments: a 22-year investigation. Conservation Biology 16: 605–618.

Laurance, W.F., McDonald, K.R. and Speare, R. 1996. Epidemic disease and the catastrophic decline of Australian rain forest frogs. Conservation Biology 10: 406–413.

Lawler, S. P., D. Dritz, T. Strange, and M. Holyoak. 1999. Effects of introduced mosquitofish and bullfrogs on the threatened California red-legged frog. Conservation Biology 13:613-622.

Lawton, J.H., and R.M. May. 1995. Extinction Rates. Oxford University Press.

Lehtinen, R. M., S. M. Galatowitsch, and J. R. Tester. 1999. Consequences of habitat loss and fragmentation for wetland amphibian assemblages. Wetlands 19: 1-12.

Lemly, A.D. 2009. Aquatic Hazard of Selenium Pollution From Mountaintop Removal Coal Mining. Report for The Appalachian Center for the Economy and the Environment. April 2009. 17 pp., *available at*: wvgazette.com/static/lemley2009.pdf (last visited Feb. 17, 2012).

Licht, L. E. 1969. Palatability of Rana and Hyla eggs. American Midland Naturalist 82: 296-298.

Liddle, M. 1975. A selective review of the ecological effects of human trampling on natural ecosystems. Biological Conservation 7: 17-36

Lind, A. J., H. H. Welsh, Jr., and R. A. Wilson. 1996. The effects of a dam on breeding habitat and egg survival of the Foothill Yellow-legged frog (*Rana boylii*) in Northwestern California. Herpetological Review 27(2): 62-67.

Lind, A.J., J.B. Bettaso and S.M. Yarnell. 2003. Natural history notes: *Rana boylii* (foothill yellow-legged frog) and *Rana catesbeiana* (bullfrog). Reproductive behavior. Herpetological Review 34: 234–235.

Lind, A.J. 2008. Amphibians and Reptiles and Climate Change. (May 20, 2008). U.S. Department of Agriculture, Forest Service, Climate Change Resource Center, *available at* http://www.fs.fed.us/ccrc/topics/wildlife/amphibians-reptiles.shtml (last visited April 2, 2012).

Lindberg, T.T., E.S. Bernhardt, R. Bier, A.M. Helton, R.B. Merola, A. Vengosh, and R.T. Di Giulio. 2011. Cumulative impacts of mountaintop mining on an Appalachian watershed. Proceedings of the National Academy of Sciences 108(52), *available at* http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=1&ved=0CFQQFjAA&url=http%3A%2F%2Fwww.nicholas.duke.edu%2Fnews%2Fpnas%2520201112381.pdf&ei=_2zfT9OSCYKC8ATskY3ZCg&usg=AFQjCNEmxdnwsVrSNP8OhnioKMnGGik_-g&sig2=5iA9PSESsjMIydDARpmWCQ (last visited June 18, 2012).

Linder, G. J. Wyant, R. Meganck, B. Williams. 1992. Evaluating amphibian responses in wetlands impacted by mining activities in the western United States. Technical Report. Pub. in Proceedings: Thorne Symposium, Apr 91. Prepared in cooperation with ManTech Environmental Technology, Inc., Corvallis, OR.

Lips, K.R. 1998. Decline of a tropical montane amphibian fauna. Conservation Biology 12: 106–117.

Liss, W. J., and G. L. Larson. 1991. Ecological effects of stocked trout on North Cascades naturally fishless lakes. Park Science 11: 22-23.

Lodé, T. 2000. Effect of a motorway on mortality and isolation of wildlife populations. Ambio 29: 163-166.

Loewenstein, N.J., E.F. Loewenstein. 2005. Non-native plants in the understory of riparian forests across a land use gradient in the Southeast. Urban Ecosystems 8: 79–91.

Long. L. E., L. S. Saylor, and M. E. Sould. 1995. A pH/UV-B synergism in amphibians. Conservation Biology 9: 1301-1303.

Loughman, Z.J. and Stuart A. Welsh. 2010. Distribution and Conservation Standing of West Virginia Crayfishes. Southeastern Naturalist, 9(3): 63-78.

Lovich J.E., S.W. Gotte, C.H. Ernst, J.C. Harshbarger, A.F. Laemmerzahl, J.W. Gibbons. 1996. Prevalence and histopathology of shell disease in turtles from Lake Blackshear, Georgia. Journal of Wildlife Diseases 32: 259-65.

Lowe, J. 2007. Chytrid Distribution in Foothill Yellow-legged Frogs and Pacific Tree Frogs in the Diablo Mountains and San Joaquin Valley Foothills of Central Coastal California. Presented at California-Nevada Amphibian Populations Task Force Meeting, January 18-20, 2007, Las Vegas, Nevada. Bureau of Land Management, Hollister Field Office, CA.

Luchenbach, Roger A. and R. Bruce Bury. 1983. Effects of Off-Road Vehicles on the Biota of the Algondones Dunes, Imperial County, California. The Journal of Applied Ecology 20(1): 265-286.

Lysne, S.J., K.E. Perez, K.M. Brown, R.L. Minton, and J.D. Sides. 2008. A review of freshwater gastropod conservation: challenges and opportunities. Journal of the North American Benthological Society 27(2): 463-470.

MacArthur, R. H., and E. O. Wilson. 1967. The Theory of Island Biogeography. Princeton University Press, Princeton, NJ.

Mackessy, S.P. 2005. Desert Massasauga Rattlesnake (Sistrurus catenatus edwardsii): a technical conservation assessment. USDA Forest Service, Rocky Mountain Region, *available at* http://www.fs.fed.us/r2/projects/scp/assessments/massasauga.pdf (last visited Oct. 7, 2011).

MacNally, R., and G.W. Brown. 2001. Reptiles and habitat fragmentation in the box–ironbark forests of central Victoria, Australia: predictions, compositional change, and faunal nestedness. Oecologia 128: 116–125.

Mahaney, P. A. 1994. Effects of freshwater petroleum contamination on amphibian hatching and metamorphosis. Environmental Toxicology and Chemistry 13: 259-265.

Maine Dept. of Inland Fisheries and Wildlife. 2010. Maine Endangered Species Program/State list, *available at* http://www.maine.gov/ifw/wildlife/species/endangered_species/state_list.htm (last visited October 3, 2011).

Mallin, M.A. and L.B. Cahoon. 2003. Industrialized animal production—a major source of nutrient and microbial pollution to aquatic ecosystems. Population and Environment 24(5): 369-385.

Maniero, G. D., and C. Carey. 1997. Changes in selected aspects of immune function in the leopard frog, Rana pipiens, associated with exposure to cold. Journal of Comparative Physiology 167: 256-263.

Manny, B. A., W. C. Johnson, and R. G. Wetzel. 1994. Nutrient additions by waterfowl to lakes and reservoirs: predicting their effects on productivity and water quality. Hydrobiologia 279/280: 121-132.

Marchand, M. N. and J. A. Litvaitis. 2004. Effects of habitat features and landscape composition on the population structure of a common aquatic turtle in a region undergoing rapid development. Conservation Biology 18: 758-767.

Marsh, D. M., and N. G. Beckman. 2004. Effects of forest roads on the abundance and activity of terrestrial salamanders. Ecological Applications 14: 1882–1891.

Marsh, D. M., and P. B. Pearman. 1997. Effects of habitat fragmentation on the abundance of two species of leptodactylid frogs in an Andean montane forest. Conservation Biology 11: 1323-1328.

Marsh, D.M., and P.C. Trenham. 2001. Metapopulation dynamics and amphibian conservation. Conservation Biology 15: 40–49.

Marzluff, J.M., R. Bowman, and R. Donnelly. 2001. A historical perspective on urban bird research: Trends, topics, and definitions. Pages 1-18 in J.M. Marzluff, R. Bowman, and R. Donnelly, editors. Avian ecology and conservation in an urbanizing world. Kluwer Academic Publishers, Norwell.

Mathews, A. E. 1989. Conflict, controversy, and compromise: the Concho water snake (Nerodia harteri paucimaculata) versus the Stacy Dam and Reservoir. Environmental Management 13(3): 297-307.

Matter, W.J. and J.J. Ney. 1981. The impact of surface mine reclamation on headwater streams in southwest Virginia. Hydrobiologia 28: 63-71.

Maxwell, B. and G. Hokit. 1999. Amphibians and reptiles. In Joslin, G., and H. Youmans, coordinators. 1999. Effects of recreation on Rocky Mountain wildlife: A Review for Montana. Committee on Effects of Recreation on Wildlife, Montana Chapter of The Wildlife Society. 307pp.

Mazzoni, R., A.A. Cunningham, P. Daszak, A. Apolo, and E. Perdomo. 2003. Emerging pathogen of wild amphibians in frogs (*Rana catesbeiana*) farmed for international trade. Emerging Infectious Diseases 9: 3-30.

McClure, H. E. 1951. An analysis of animal victims on Nebraska's highways. Journal of Wildlife Management 15: 410-420.

McCoid, M. J., and P. W. Bettoli. 1996. Additional evidence for rotenone hazards to turtles and amphibians. Herpetological Review 27(2): 70-71.

McGee, M. and D. Keinath. 2004. Species Assessment for Boreal Toad (*Bufo boreas boreas*) in Wyoming. Prepared for U.S. Dept. of Interior, Bureau of Land Management.

McKinney, M. L. 2002. Urbanization, biodiversity, and conservation. Bioscience 52: 883-890.

McKinney, M. L. 2006. Urbanization as a major cause of biotic homogenization. Biological Conservation 127: 247-260.

McMenamina, Sarah K., Elizabeth A. Hadlya, and Christopher K. Wright. 2008. Climatic change and wetland desiccation cause amphibian decline in Yellowstone National Park. PNAS 105(44): 16988–16993.

McMurry, F. B., and C. C. Sperry. 1941. Food of feral house cats in Oklahoma, a progress report. Journal of Mammalogy 22(2): 185-190.

Means, D.B. 2009. Effects of rattlesnake roundups on the eastern diamondback rattlesnake (*Crotalus adamanteus*). Herpetological Conservation and Biology 4(2):132-141.

Messere, M., and P. K. Ducey. 1998. Forest floor distribution of northern redback salamanders, Plethodon cinereus, in relation to canopy gaps: first year following selective logging. Forest Ecology and Management 107: 319–324.

Meyers Schöne, L., and B.T. Walton. 1994. Turtles as monitors of chemical contaminants in the environment. Rev Environ Contam Toxicol 135: 93-153.

Middelkoop, J., T. Watts, and M. Schorr. 1998. Acid mine drainage and its effects on physicoche mical conditions and salamander populations in a Cumberland Plateau stream. Journal of the Tennessee Academy of Science 73: 36.

Mitchell, J. C., J. A. Wicknick, and C. D. Anthony. 1996. Effects of timber harvesting practices on Peaks of Otter salamander (*Plethodon hubrichti*) populations. Amphibian and Reptile Conservation 1: 15–19.

Mitsch, W. J., and J. G. Gosselink. 2000. Wetlands. Third edition. John Wiley & Sons, New York, New York, USA.

Moler, P. E. (ed.). 1992. Rare and Endangered Biota of Florida. Volume 3. Amphibians and Reptiles. University Press of Florida.

Moore, M.V., S.M. Pierce, H.M. Walsh, S.K. Kvalvik, J.D. Lim. 2000. Urban light pollution alters the diel vertical migration of daphnia. Verhandlungen der Internationalen Vereinigung fur Limnologie 27: 1–4.

Morse, J.C., B.P. Stark, W. P. McCafferty, and K.J. Tennessen. 1997. Southern Appalachian and other Southeastern streams at risk: Implications for mayflies, dragonflies and damselflies, stoneflies, and caddisflies. p. 17-42 *In:* Benz, G.W. and D.E. Collins (editors). 1997. Aquatic Fauna in Peril: The Southeastern Perspective. Southeast Aquatic Research Institute Special Publication 1, Lenz Design and Communications, Decatur, GA. 553 pp.

Moulis, R. A. 1997. Predation by the red imported fire ant (*Solenopsis invicta*) on loggerhead sea turtle (*Caretta caretta*) nests on Wassaw National Wildlife Refuge, Georgia. Chel. Conserv. Biol. 2: 433–436.

Mount, R. H. 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. Journal of the Alabama Academy of Science 52: 71–78.

Moyle, P. B. 1973. Effects of introduced bullfrogs, Rana catesbeiann. on the native frogs of the San Joaquin valley, California. Copeia 1973: 18-22.

Mulholland, P.J. and D.R. Lenat. 1992. Streams of the southeastern piedmont, Atlantic drainage. *In*: Biodiversity of the Southeastern United States: Aquatic Communities. C.T. Hackney, S.M. Adams, and W.H. Martin (eds.). John Wiley and Sons, New York, NY, p. 193-231.

Munger, J.C., L. Heberger, D. Logan, W. Peterson, L. Mealy, and M. Cauglin. 1994. A survey of the herpetofauna of the Bruneau Resource Areas, with focus on the spotted frog, *Rana pretiosa*. Bureau Land Management Tech. Bull. 94.

Munger, James C., Bruce R. Barnett, Stephen J. Novak and Aaron A. Ames. 2003. Impacts of Off-Highway Motorized Vehicle Trails on the Reptiles and Vegetation of the Owyhee Front. Idaho Bureau of Land Management Technical Bulletin 03-3: 1-23.

Murcia, C. 1995. Edge effects in fragmented forests: implications for conservation. Trends in Ecology & Evolution 10: 58–62.

Myers, C. W., and W. D. Klimstra. 1963. Amphibians and reptiles of an ecologically disturbed (strip- mined) area in southern Illinois. American Midland Naturalist 70: 127-132.

Myers, N., Knoll, A.H. 2001. The biotic crisis and the future of evolution. Proceedings of the National Academy of Science 98 (10): 5389–5392.

Nace, G. W., and J. K. Rosen. 1979. Source of amphibians for research II. Herpetological Review 10: 8-15.

Nanjappa, P. and Paulette M. Conrad 2012. State of the Union: Legal Authority Over the Use of Native Amphibians and Reptiles in the United States, *available at* http://www.fishwildlife.org/files/SOU_FULL-lo-res.pdf (last visited April 30, 2012).

Nash, R. F., G. G. Gallup, Jr., and M. K. McClure. 1970. The immobility reaction in leopard frogs (Rana pipiens) as a function of noise-induced fear. Psychonometric. Science 21(3): 155-156.

National Wildlife Health Center. 2001. Amphibian diseases. National Wildlife Health Center information sheet. U.S. Geological Survey.

Natural Resources Conservation Service. 2002. Precision land forming. Code 462. National conservation practice standard—national handbook of conservation practices, *available at* http://www.nrcs.usda.gov/technical/standards/nhcp.html (last visited Feb. 21, 2012).

NC WRC. 2005. NC Wildlife Action Plan. NC Wildlife Resources Commission, Raleigh, NC USA.

Neill, W. T. 1950. Reptiles and amphibians in urban areas of Georgia. Herpetologica 6: 113-116.

Neves, R.J., A.E. Bogan, J.D. Williams, S.A. Ahlstedt, P.W. Hartfield. 1997. Status of aquatic mollusks in the southeastern United States: a downward spiral of diversity. p. 43-85 In: Benz, G.W. and D.E. Collins (editors). 1997. Aquatic Fauna in Peril: The Southeastern Perspective. Southeast Aquatic Research Institute Special Publication 1, Lenz Design and Communications, Decatur, GA. 553 pp., *available at* http://www.sherpaguides.com/southeast/aquatic_fauna/chapter_3/ (last visited April 4, 2012).

Nickel, N. 2010. Keeping and Breeding the Southern Hognose Snake. Reptilia 68: 25-35.

Noss R.F., E.T. LaRoe, J.M. Scott. 1995. Endangered Ecosystems of the United States: A Preliminary Assessment of Loss and Degradation.Washington (DC): U.S. Department of the Interior. Biological Report 28.

NSC (National Safety Council). 2003. Reporting on Climate Change: Understanding the Science. Washington, DC: National Safety Council, Environmental Health Center.

Nyman, S. 1986. Mass mortality in larval *Rana sylvatica* attributable to the bacterium, *Aeromonas hydrophila*. Journal of Herpetology 20: 196-201.

Ohmart, R.D., B.W. Anderson, and W.C. Hunter. 1988. The Ecology of the lower Colorado River from Davis Dam to the Mexico-United States International Border: A Community Profile. USDI Fish and Wildlife Service. Biological Report 85(7.19).

Oldham, R. S. and Swan, M. J. S. 1991. Conservation of amphibian populations in Britain. In Species conservation: a population-biological approach, ed. A. Seitz & V. Lowschcke. Birkh~iuser Verlag, Basel, pp. 141-57.

Olson, D. H. 1992. Ecological susceptibility of amphibians to population declines. Pages 55-62 in R. R. Harris, D. E. Erman (technical Coordinators) and H. M. Kerner (editors). Proceedings of symposium on biodiversity of northwestern California. University of California Wildland Resources Center, Report 29, Davis, California.

Olson, D.H., P.D. Anderson, C.A. Frissell,H.H. Welsh Jr., D.F. Bradford. 2007. Biodiversity management approaches for stream-riparian areas: Perspectives for Pacific Northwest headwater forests, microclimates, and amphibians. Forest Ecology and Management 246: 81–107.

Orlando, E.F., A.S. Kolok, G.A. Binzcik, J.L. Gates, M.K. Horton, C.S. Lambright, L.E. Gray, A.M. Soto, and L.J. Guillette. 2004. Endocrine-Disrupting Effects of Cattle Feedlot Effluent on an Aquatic Sentinel Species, the Fathead Minnow. Environmental Health Perspectives 112(3): 353-358.

Orlowski, G., 2007. Spatial distribution and seasonal pattern in road mortality of the common toad Bufo bufo in an agricultural landscape of south–western Poland. Amphibia–Reptilia, 28: 25–31.

Orser, P. N. and D. J. Shure. 1972. Effects of urbanization on the salamander *Desmognathus fuscus fuscus*. Ecology 53: 1148-1154.

Packard, G.C., J. K. Tucker, and L. D. Lohmiller. 1998. Distribution of Strecker's chorus frogs (*Pseudacris streckeri*) in relation to their tolerance for freezing. Journal of Herpetology 32: 437-440.

Padgett-Flohr, G. E. and J. E. Longcore. 2007. *Taricha torosa*. Fungal infection. Herpetological Review 38:176–177.

Palmer, M.A. and E.S. Bernhardt. 2009. Mountaintop Mining Valley Fills and Aquatic Ecosystems: A Scientific Primer on Impacts and Mitigation Approaches, *available at*: wvgazette.com/static/mtrwhitepaper.pdf (last visited July 5, 2012).

Palmer, M.A., E.S. Bernhardt, W.H. Schlesinger, K.N. Eshleman, E. Foufoula-Georgiou, M.S. Hendryx, A.D. Lemly, G.E. Likens, O.L. Loucks, M.E. Power, P.S. White, and P.R. Wilcock. 2010. Mountaintop mining consequences. Science 327: 148-149.

Paoletti, D. P. 2009. Responses of Foothill Yellow-legged Frog (Rana boyliz) larvae to an introduced predator. Unpubl. Master's thesis, Oregon State University, Corvallis, Oregon.

Paoletti, D.J., Deanna H. Olson, and Andrew R. Blaustein. 2011. Responses of Foothill Yellow-legged Frog (Rana boylii) Larvae to an Introduced Predator. Copeia 2011 (1): 161–168.

Papenfuss, T.J. and J.R. Macey. 1986. A review of the population status of the Inyo Mountains salamander. U.S. Fish and Wildlife Service, Order Number 10188-5671-5, Endangered Species Office, Sacramento, California.

Parmalee, P. W. 1953. Food habits of the feral house cat in east-central Texas. Journal of Wildlife Management 17(3): 375-376.

Parmesan, C., T.L. Root, and M.R. Willig. 2000. Impacts of extreme weather and climate on terrestrial biota. Bulletin of the American Meteorological Society. 81(3): 443-450.

Parry, W. 2012. Mysterious Rattlesnake-Killing Infection Emerges. Live Science (Feb. 24, 2012), available at http://www.livescience.com/18634-endangered-rattlesnake-fungal-infection.html (last visited March 29, 2012).

Pathogenicity and transmission of chytridiomycosis in Tiger Salamanders (*Ambystoma tigrinum*). Copeia 2003: 601–607.

Patla, D. A. 1998. Potential effects of native fish restoration projects on amphibians in Yellowstone National Park Part I. Report to National Park Service, Yellowstone National Park. 20 November 1998. 26pp.

Patricelli, G.L., J.L. Blickley. 2006. Avian communication in urban noise: causes and consequences of vocal adjustment. Auk 123: 639–649.

Patrick, D.A., E.B. Harper, M.L. Hunter Jr., A.J.K. Calhoun. 2008. Terrestrial habitat selection and strong density-dependent mortality in recently metamorphosed amphibians. Ecology 89: 2563–2574.

Patrick, R. 1992. Surface Water Quality: Have the Laws Been Successful? Princeton University Press, Princeton, N.J.

Patterson, L.C. 2006. Life history and ecology of an introduced population of red-eared sliders (*Trachemys scripta elegans*) in the Central Valley of California with implications for the conservation of the western pond turtle (*Emys marmorata*). M.S. Thesis, California State University, Sacromento.

Pellet, J., A. Guisan, and N. Perrin. 2004. A concentric analysis of the impact of urbanization on the threatened European tree frog in an agricultural landscape. Conservation Biology 18: 1599-1606.

Pessier AP, Nichols DK, Longcore JE, Fuller MS. 1999. Cutaneous chytridiomycosis in poison dart frogs (*Dendrobates* spp.) and White's tree frogs (*Litoria caerulea*). J. Vet. Diag. Invest. 11: 194-199.

Peterson E.W., R.K. Davis, and H.A. Orndorff. 2000. 17β-Estradiol as an indicator of animal waste contamination in mantled karst aquifers. J. Environ. Qual. 29: 826–834.

Peterson, J. D., M. B. Wood, W. A. Hopkins, J. M. Unrine, and M. T. Mendonca. 2007. Prevalence of *Batrachochytrium dendrobatidis* in American Bullfrog and Southern Leopard Frog larvae from wetlands on the Savannah River Site, South Carolina. Journal of Wildlife Diseases 43: 450–460.

Petranka, J. W., M. E. Eldridge, and K. E. Haley. 1993. Effects of timber harvesting on southern Appalachian salamanders. Conservation Biology 7: 363–370.

Petranka, J. W., M. P. Brannon, M. E. Hopey, and C. K. Smith. 1994. Effects of timber harvesting on southern Appalachian salamanders. Forest Ecology and Management 67: 135–147.

Pilliod, D. S., C. R. Peterson, and P. I. Ritson. 2002. Seasonal migration of Columbia Spotted frogs (Rana luteiventris) among complementary resources in a high mountain basin. Canadian Journal of Zoology 80: 1849-1862.

Pillsbury, F. C., and J. R. Miller. 2008. Habitat and landscape characteristics underlying anuran community structure along an urban-rural gradient. Ecological Applications 18: 1107-1118.

Pimental, D. 1971. Ecological effects of pesticides on non-target organisms. U.S. Government Printing Office, Washington D.C. 220pp.

Piovia-Scott, J., K. L. Pope, S. P. Lawler, E. M. Cole, and J. E. Foley. 2011. Factors related to the distribution and prevalence of the fungal pathogen *Batrachochytrium dendrobatidis* in *Rana cascadae* and other amphibians in the Klamath Mountains. Biological Conservation 144: 2913-2921.

Pomponio, J.R. 2009. Testimony of John “Randy” Pomponio, Director, Environmental Assessment and Innovation Division, U.S. Environmental Protection Agency Mid-Atlantic Region, before the Committee On Environment And Public Works Subcommittee on Water And Wildlife, U.S. Senate. June 25, 2009.

Pond, G.J., M.E. Passmore, F.A. Borsuk, L. Reynolds, and C. J. Rose. 2008. Downstream effects of mountaintop coal mining: comparing biological conditions using family- and genus-level macroinvertebrate bioassessment tools. Journal of North American Benthological Society. 27(3): 717-737.

Ponsero, A., and P. Joly. 1998. Clutch size, egg survival and migration distance in the agile frog (*Rana dalmatina*) in a floodplain. Archiv fur Hydrobiologie 142: 343–352.

Porter, K. R., and D. E. Hakanson. 1976. Toxicity of mine drainage to embryonic and larval boreal toads (Bufonidae: *Bufo boreas*). Copeia 2: 327-331.

Post, D. M., J. P. Taylor, J. F. Kitchell, M. H. Olson, D. E. Schindler, and B. R. Herwig. 1998. The role of migratory waterfowl as nutrient vectors in a managed wetland. Conservation Biology 12: 910-920.

Postel, S.L. 2000. Entering an era of water scarcity: the challenges ahead. Ecological Applications 10: 941–948.

Pough, F. H. 1980. The advantages of ectothermy for tetrapods. American Naturalist 115: 92-112.

Pough, F. H. 1983. Amphibians and reptiles as low-energy systems. Pages 141-188 in Behavioral Energetics: the cost of survival in vertebrates. Ohio State University Press. Columbus, Ohio.

Pough, F. H., R. M. Andrews, J. E. Cadle, M. L. Crump, A. H. Savitzky, and K. D. Wells. 1998. Herpetology. Prentice-Hall Inc., Upper Saddle River, New Jersey. 577pp.

Pough, F.H., E.M. Smith, D.H. Rhodes, A. Collazo. 1987. The abundances of salamanders in forest stands with different histories of disturbance. Forest Ecology and Management 20: 1–9.

Pough, F.H., R.M. Andrews, J.E. Cadle, M.L. Crump, A.H. Savitzky, K.D. Wells. 2001. Herpetology. Upper Saddle River, NJ: Prentice-Hall, Inc. 736 p.

Pounds, J. A., and M. L. Crump. 1994. Amphibian declines and climate disturbance: the case of the golden toad and harlequin frog. Conservation Biology 8(1): 72-85.

Pounds, J. A., M. P. L. Fogden, and J. H. Campbell. 1999. Biological response to climate change on a tropical mountain. Nature 398: 611-615.

Pounds, J.A., M.R. Bustamante, L.A. Coloma, J.A. Consuegra, M.P.L. Fogden, P.N. Foster, E. La Marca, K.L. Masters, A. Merino-Viteri, R. Puschendorff, S.R. Ron, G.A. Sánchez-Azofeifa, C.J. Still, and B.E. Young. 2006. Widespread amphibian extinctions from epidemic disease driven by global warming. Nature 439: 161-167.

Pounds, A., A.C.O.Q. Carnaval and S. Corn. 2007. Climate Change, Biodiversity Loss, and Amphibian Declines. In IUCN's Amphibian Conservation Action Plan (Claude Gascon et al. editors).

Price, S.J., Dorcas, M.E., Gallant, A.L., Klaver, R.W., Willson, J.D. 2006. Three decades of urbanization: estimating the impact of land-cover change on stream salamander populations. Biological Conservation 133: 436–441.

Puky, M. 2006. Amphibian road kills: a global perspective. Pages 325-338 in Proceedings of the 2005 International Conference on Ecology and Transportation. Center for Transportation and the Environment, North Carolina State University, Raleigh, N.C.

Rathbun, G. B., N. Siepel, and D. C. Holland. 1992. Nesting behavior and movements of western pond turtles (Clemmys marmorata). The Southwestern Naturalist 37(3): 319-324.

Raymond, L. R., and L. M. Hardy. 1991. Effects of a clearcut on a population of the mole salamander, Ambystoma talpoideum, in an adjacent unaltered forest. J. Herpetol. 25: 509-512.

Reading, C.J. 2007. Linking global warming to amphibian declines through its effects on female body condition and survivorship. Oecologia 151: 125–131.

Reagan, S.R., J.M. Ertel, V.L. Wright. 2000. David and Goliath retold: fire ants and alligators. Journal of Herpetology. 34: 475-478.

Redmond, W. H. 1980. Notes on the distribution and ecology of the black mountain dusky salamander *Desmognathus welteri* Barbour (Amphibia: Plethodontidae) in Tennessee. Brimleyana 4: 123-131.

Reed, R. N., and J. W. Gibbons. 2003. Conservation status of live U.S. nonmarine turtles in domestic and international trade. Report to United States Fish and Wildlife Service, Arlington, Virginia. 92 pp, *available at* http://www.google.com/url?sa=t&source=web&cd=1&ved=0CEEQFjAA&url=http%3A%2F%2Fwww.graptemys.com%2Fturtle_trade.doc&rct=j&q=Conservation%20status%20of%20live%20U.S.%20nonmarine%20turtles%20in%20domestic%20and%20international%20trade&ei=ggc3TrWFAoqGsgLd09A4&usg=AFQjCNEEfB93Ot2yKXLV9p67Y93RXlu1qA&cad=rja (last visited Aug. 1, 2011).

Reed, R. N., J. Congdon, and J. W. Gibbons. 2002. The alligator snapping turtle [*Macrochelys (Macroclemys) temminckii*]: a review of ecology, life history, and conservation, with demographic analyses of the sustainability of take from wild populations. Report to United States Fish and Wildlife Service, Arlington, Virginia. 17 pp.

Reeder, N.M.M., A.P. Pessier, V.T. Vredenburg. 2012. A Reservoir Species for the Emerging Amphibian Pathogen *Batrachochytrium dendrobatidis* Thrives in a Landscape Decimated by Disease. PLoS ONE 7(3), *available at* http://www.plosone.org/article/info%3Adoi%2F10.1371%2Fjournal.pone.0033567 (last visited April 30, 2012).

Reh, W. and Seitz, A. 1990. The influence of land use on the genetic structure of populations of the common frog Rana temporaria. Biol. Conserv. 54: 239-49.

Reinelt, L., R. Horner, and A. Azous. 1998. Impacts of urbanization on palustrine (depressional freshwater) wetland- research and management in the Puget Sound region. Urban. Ecosystems 2:219-236.

Reinking, L. N., C. H. Daugherty, and L. B. Daugherty. 1980. Plasma aldosterone concentrations in wild and captive western spotted frogs (Rana pretiosa). Comparative Biochemistry and Physiology 65A(4): 517-518.

Relyea Rick, A., and N. Mills. 2001. Predator-induced stress makes the pesticide carbaryl more deadly to gray treefrog tadpoles (Hyla versicolor). Proceedings of the National Academy of Sciences of the United States of America [print] 98: 2491-2496.

Resetarits, W. J., Jr., and H. M. Wilbur. 1989. Choice of oviposition site by *Hyla chrysoscelis*: role of predators and competitors. Ecology 70: 220-228.

Ricciardi, A. and J.B. Rasmussen. 1998. Predicting the identity and impact of future biological invaders: a priority for aquatic resource management. Canadian Journal of Fisheries and Aquatic Sciences 55: 1759-1765.

Richter, K. O. 1997. Criteria for the restoration and creation of wetland habitats of lenticbreeding amphibians of the Pacific Northwest. Pg. 72-94 *In*: Macdonald, K. B. And F. Winmann (eds.) Wetland and Riparian Restoration: Taking a Broader View. Publication EPA 910-R-97-007. USEPA, Region 10, Seattle, Washington.

Richter, K. O. and A. L. Azous. 1995. Amphibian occurrence and wetland characteristics in tile Puget Sound basin. Wetlands 15: 305 312.

Riley, C. V. 1952. An evaluation of reclaimed coal strip mined lands as wildlife habitat. Unpubl. Ph. D. Diss., The Ohio State University, Columbus. 289 pp.

Riley, C. V. 1960. The ecology of water areas associated with coal strip-mined lands in Ohio. The Ohio Journal of Science 60: 106-121.

Riley, S. P. D., G. T. Busteed, L. B. Kats, T. L. Vandergon, L. F. S. Lee, R. G. Dagit, J. L. Kerby, R. N. Fisher, and R. M. Sauvajot. 2005. Effects of urbanization on the distribution and abundance of amphibians and invasive species in southern California streams. Conservation Biology 19: 1894-1907.

Rinne, J.N., J.A. Stefferud, D.A. Clark, and P.J. Sponholtz. 1998. Fish community structure in the Verde River, Arizona, 1974-1997. Hydrology and Water Resources in Arizona and the Southwest 28:75-80, *available at* http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=1&ved=0CCkQFjAA&url=http%3A%2F%2Fwww.rmrs.nau.edu%2Fawa%2Fpublications%2Frinne_etal_fishcommstructureverde.pdf&ei=CAHUTu6zGIe62gWN67y2Dw&usg=AFQjCNHxXGKXDiViQUIS2C5ojdWEDEbS3A (last visited Nov. 28, 2011).

Rittenhouse, T.A.G., E.B. Harper, L.R. Rehard, R.D. Semlitsch. 2008. The role of microhabitats in the desiccation and survival of anurans in recently harvested oak-hickory forest. Copeia 2008: 807–814.

Rittenhouse, TAG, R.D. Semlitsch. 2006.Grasslands as movement barriers for a forest-associated salamander:Migration behavior of adult and juvenile salamanders at a distinct habitat edge. Biological Conservation 131: 14–22.

Rocco, G. L., and R. P. Brooks. 2000. Abundance and distribution of a stream plethodontid salamander assemblage in 14 ecologically dissimilar watersheds in the Pennsylvania central Appalachians: final report. Penn State Cooperative Wetlands Center, Forest Resources Laboratory, Report 2000–4. Pennsylvania State University, University Park, USA.

Roe, J. H., J. Gibson, and B. A. Kingsbury. 2006. Beyond the wetland border: Estimating the impact of roads for two species of water snakes. Biological Conservation 130: 161-168.

Rohr, J.R., A.M. Schotthoefer, T.R. Raffel, H.J. Carrick, N. Halstead, J.T. Hoverman, C.M. Johnson, L.B. Johnson, C. Lieske, M.D. Piwoni, P.K. Schoff, and V.R. Beasley. 2008. Agrochemicals increase trematode infections in a declining amphibian species. Nature 455(30): 1235-1240.

Roman, J. S.D. Santhuff, P.E. Moler, and B.W. Bowen. 1999. Population structure and cryptic evolutionary unites in the alligator snapping turtle. Conservation Biology 13: 135-42.

Root, T.L., J.T. Price, K.R. Hall, S.H. Schneider, C. Rosenzweig, J.A. Pounds. 2003. Fingerprints of global warming on wild animals and plants. Nature 421: 57-60.

Rosen, P. C. and C. H. Lowe. 1994. Highway mortality of snakes in the Sonoran Desert of southern Arizona. Biological Conservation 68: 143-148.

Rosen, P. C., C. R. Schwalbe, D. A. Parizek Jr., P. A. Holm, and C. H. Lowe. 1995. Introduced aquatic vertebrates in Chiricahua region: effects on declining native ranid frogs. U.S. Forest Service General Technical Report RM 264: 251-261.

Rosen, P.C., and C. H. Lowe. 1994. Highway mortality of snakes in the Sonoran Desert of southern Arizona. Biological Conservation 68: 143-148.

Ross B., T. Fredericksen, E. Ross, W. Hoffman, M.L. Morrison, J. Beyea, M.B. Lester, B.N. Johnson, and N.J. Fredericksen. 2000. Relative abundance and species richness of herpetofauna in forest stands in Pennsylvania. Forest Science 46: 139–146.

Ross, D. A., and R. K. Anderson. 1990. Habitat use, movements, and nesting of *Emydoidea blandingii* in central Wisconsin. J. Herpetol. 24: 6-12.

Rothermel B.B., and T.M. Luhring. 2005. Burrow availability and desiccation risk of mole salamanders (Ambystoma talpoideum) in harvested versus unharvested forest stands. Journal of Herpetology 39: 619–626.

Rothermel, B.B., and R.D. Semlitsch. 2002. An experimental investigation of landscape resistance of forest versus old-field habitats to emigrating juvenile amphibians. Conserv. Biol. 16: 1324–1332.

Rothermel, B. B., S. C. Walls, J. C. Mitchell, C. K. Dodd, Jr, L. K. Irwin, D. E. Green, V. M. Vazquez, J. W. Petranka, and D. J. Stevenson. 2008. Widespread occurrence of the amphibian chytrid fungus *Batrachochytrium dendrobatidis* in the southeastern USA. Diseases of Aquatic Organisms 82: 3–18.

Rouse, J. D., C. A. Bishop, and J. Struger. 1999. Nitrogen pollution: An assessment of its threat to amphibian survival. Environmental Health Perspectives 107: 799-803.

Rovito, S.M., G. Parra-Olea, C. R. Vásquez-Almazán, T. J. Papenfuss, and D. B. Wake. 2009. Proc. Natl. Acad. Sci. U.S.A. 106: 3231.

Row, J. R., G. Blouin-Demers, and P. J. Weatherhead. 2007. Demographic effects of road mortality in black ratsnakes (Elaphe obsoleta). Biological Conservation 137:117.

Rowe, C. L., W. A. Hopkins, and V. R. Coffman. 2001. Failed recruitment of southern toads (*Bufo terrestris*) in a trace element-contaminated breeding habitat: Direct and indirect effects that may lead to a local population sink. Archives of Environmental Contamination & Toxicology [print] 40: 399-405.

Rubbo, M.J., and J.M. Kiesecker. 2005. Amphibian breeding distribution in an urbanized landscape. Conservation Biology 19: 504–511.

Rubec, C.D.A., P. Lynch-Stewart, G.M. Wickware, I. Kessel-Taylor. 1988. Utilisation des terres humides au Canada. In: Groupe de Travail national sur les Terres humides (Ed.), Terres humides du Canada. Polyscience Publications, Montre´ al, pp. 379–412.

Ruby, D. E., J. R. Spolita, S. K. Martin, and S. J. Kemp. 1994. Behavioural responses to barriers by desert tortoises: implications for wildlife management. Herpetological Monographs 8: 144-160.

Rudolph, D. C., S. J. Burgdorf, R. N. Conner, and R. R. Schaefer. 1999. Preliminary evaluation of roads and associated vehicular traffic on snake populations in eastern Texas. Pages 128–135 in G. L. Evink, P. Garrett, and D. Ziegler, editors. Proceedings of the Third International Conference on Wildlife Ecology and Transportation, Florida Department of Transportation, Tallahassee, Florida, USA.

Russell, K.R., T.B. Wigley; W.M. Baughman, M. William, H.G. Hanlin, W. M. Ford. 2004. Responses of Southeastern amphibians and reptiles to forest management: A review. In: Gen. Tech. Rep. SRS–75. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. Chapter 27. p. 319-334.

Santos-Barrera, G., G. Hammerson, and S. Morey 2004. *Spea hammondii*. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.1. <www.iucnredlist.org>. Downloaded on 13 October 2011.

Sattler, P., and N. Reichenbach. 1998. The effects of timbering on Plethodon hubrichti: short-term effects. Journal of Herpetology 32: 399–404.

Saugey, D. A., G. A. Heidt, and D. R. Heath. 1988. Utilization of abandoned mine drifts and fracture caves by bats and salamanders: unique subterranean habitat in the Ouachita Mountains. In: R.C. Szaro, K.E. Severson, and D.R. Patton. Eds. Management of amphibians, reptiles, and mammals in North America. General technical report RM 166 - Rocky Mountain Forest and Range Experiment Station, U.S. Department of Agriculture, Forest Service.

Saunders, D. A., R. J. Hobbs, and C. R. Margules. 1991. Biological consequences of ecosystem fragmentation: a review. Conservation Biology 5: 18–30.

Saunders, H. O. 1970. Pesticide toxicities to tadpoles of the western chorus frog, Pseudacris triseriata, and Fowler's toad, Bufo woodhousei fowleri. Copeia 1970(2): 246-251.

Schlaepfer, M. A., C. Hoover, and C. K. Dodd, Jr. 2005. Challenges in Evaluating the Impact of the Trade in Amphibians and Reptiles on Wild Populations. BioScience 55(3): 256.

Schlaepfer, M.A., and T.A. Gavin. 2001. Edge effects on lizards and frogs in tropical forest fragments. Conservation Biology 15: 1079–1090.

Schnick, R. A. 1974. A review of the literature on use of rotenone in fisheries. U.S. Fish and Wildlife Service, Literature Review 74-15. No. PB-235 454/AS. 130pp.

Schnoes, R. S., and S. R. Humphrey. 1987. Terrestrial plant and wildlife communities onphosphate- mined lands in central Florida. Bulletin of the Florida State Museum, Biological Sciences. 30(3): 53-116.

Schoonover, J.E., B.G. Lockaby, B.S. Helms. 2006. Impacts of land cover on stream hydrology in the West Georgia Piedmont, USA. Journal of Environmental Quality 35: 2123–2131.

Schumacher J. 1996. Viral diseases, in Mader DR (ed): Reptile Medicine and Surgery. Philadelphia, PA, WB Saunders, pp 224-234.

Schumacher J. 2003. Fungal diseases of reptiles. Vet. Clin. Exot. Anim. 6: 327-335.

Schumacher, J. 2006. Selected Infectious Diseases of Wild Reptiles and Amphibians. Journal of Exotic Pet Medicine 15 (1): 18-24.

Schuster, G.A. 1997. Crustacean Resource Management in the Southeast. p. 269-282 *In:* Benz, G.W. and D.E. Collins (editors). 1997. Aquatic Fauna in Peril: The Southeastern Perspective. Southeast Aquatic Research Institute Special Publication 1, Lenz Design and Communications, Decatur, GA. 553 pp.

Schwaner, T.D., and B.K. Sullivan. 2005. Bufo microscaphus Cope 1867 “1866” Arizona toad. Pages 422-424 in M.J. Lannoo (ed), Amphibian Declines: The Conservation Status of United States Species. University of California Press, Berkeley, CA.

Scott, N. J., Jr. 1996. Evolution and management of the North American grassland herpetofauna. Pages 40-53 in D. M. Finch editor. Ecosystem disturbance and wildlife conservation in western grasslands: a symposium proceedings. General Technical Report RM-GTR-285. U.S. Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado.

Scribner, K.T., Arntzen, J.W., Cruddace, N., Oldham, R.S., and Burke, T. 2001. Environmental correlates of toad abundance and population genetic diversity. Biol. Conserv. 98: 201–210.

Seabloom, E. W., J. W. Williams, D. Slayback, D. M. Stoms, J. H. Viers, and A. P. Dobson. 2006. Human impacts, plant invasion, and imperiled, plant species in California. Ecological Applications 16: 1338-1350.

Sealy, J. B. 2002. Ecology and behavior of the timber rattlesnake (Crotalus horridus) in the upper Piedmont of North Carolina: Identified threats and conservation recommendations. *In:* G. W. Schuett, M. Hoggren, M. E. Douglas, and H. W. Greene (eds.) Biology of the Vipers. Eagle Mountain Publishing. Eagle Mountain, Utah.

Seastedt, T.R.,and D.A. Crossley Jr. 1981.Microarthopod response following cable logging and clear-cutting in the southern Appalachians. Ecology 62: 126–135.

Seigel, R. A. 1986. Ecology and conservation of an endangered rattlesnake, *Sistrurus catenatus*, in Missouri, USA. Biological Conservation 35: 333-346.

Semlitsch, R. D. 2000. Principles for management of aquatic-breeding amphibians. Journal of Wildlife Management 64: 615-631.

Semlitsch, R. D., and J. W. Gibbons. 1988. Fish predation in size-structured populations of treefrog tadpoles. Oecologia 75: 321-326.

Semlitsch, R.D. and J.R. Bodie. 1998. Are small, isolated wetlands expendable? Conservation Biology 12(5): 1129-1133.

Semlitsch, R.D., D.E. Scott, J.H.K. Pechmann and J.W. Gibbons. 1996. Structure and dynamics of an amphibian community: evidence from a 16-year study of a natural pond. Pp. 217–248. *In* Cody, M.L. and J.A. Smallwood (Eds.), Long-term Studies of Vertebrate Communities. Academic Press, San Diego, California.

Semlitsch, R. D., T. J. Ryan, K. Hamed, M. Chatfield, B. Drehman, N. Pekarek, M. Spath, and A. Watland. 2007. Salamander abundance along road edges and within abandoned logging roads in Appalachian forests. Conservation Biology 21: 159–167.

Semlitsch R.D., C.A. Conner, D.J. Hocking, T.A.G. Rittenhouse, and E.B. Harper. 2008. Effects of timber harvesting on amphibian persistence: Testing the evacuation hypothesis. Ecological Applications 18: 283–289.

Semlitsch, Raymond D., Brian D. Todd, Sean M. Blomquist, Aram J. K. Calhoun, J. Whitfield Gibbons, James P. Gibbs, Gabrielle J. Graeter, Elizabeth B. Harper, Daniel J. Hocking, Malcolm L. Hunter Jr., David A. Patrick, Tracy A. G. Rittenhouse, and Betsie B. Rothermel. 2009. Effects of Timber Harvest on Amphibian Populations: Understanding Mechanisms from Forest Experiments. BioScience 59(10): 853-862.

Shaffer, M. L. 1981. Minimum population sizes for species conservation. BioScience 31:131–134.

Shepard, W.D. 1993. Desert springs—both rare and endangered. Aquatic Conservation: Marine and Freshwater Ecosystems. 3: 351–359.

Shine, R., M. Lemaster, M. Wall, T. Langkilde, and R. Mason. 2004. Why did the snake cross the road? Effects of roads on movement and location of mates by garter snakes (*Thamnophis sirtalis parietalis*), available at http://www.ecologyandsociety.org/vol9/iss1/art9/ (last visited July 5, 2012).

Shute, P.W., R.G. Biggins, and R.S. Butler. 1997. Management and Conservation of Rare Aquatic Resources: A Historical Perspective and Recommendations for Incorporating Ecosystem Management. p. 445-466 *In:* Benz, G.W. and D.E. Collins (editors). 1997. Aquatic Fauna in Peril: The Southeastern Perspective. Southeast Aquatic Research Institute Special Publication 1, Lenz Design and Communications, Decatur, GA. 553 pp.

Simons, L. H. 1998. *Rana cascadae* (Cascades frog): Predation. Herpetological Review 29: 232.

Sinsch, U., 1990. Migration and orientation in anuran amphibians. Ethology, Ecology & Evolution 2: 65–79.

Sjrgren, R. 1991, Extinction and isolation gradients in metapopulations: the case of the pool frog (Rana lessonae). Biological Journal of the Linnean Society 42:135-147.

Skelly, D. K. 1992. Field evidence for a cost of behavioral antipredator response in a larval amphibian. Ecology 73: 704-708.

Skelly, D.K., E.E. Werner and S.A. Cortwright. 1999. Long-term distributional dynamics of a Michigan amphibian assemblage. Ecology 80: 2326–2337.

Smith, R.B., R.A. Seigel, K.R. Smith. 1998. Occurrence of Upper Respiratory Tract Disease in gopher tortoise populations in Florida and Mississippi. Journal of Herpetology 32: 426–30.

Smith, L. and C. K. Dodd Jr. 2003. Wildlife mortality on U.S. highway 441 across Paynes Prairie, Alachua County, Florida. Florida Scientist 66: 128-140.

Smith, S., and G. D. Grossman. 2003. Stream microhabitat use by larval southern two-lined salamanders (Eurycea cirrigera) in the Georgia piedmont. Copeia 2003: 531–543.

Soucek, D.J., D.S. Cherry, and C.E. Zipper. 2003. Impacts of mine drainage and other nonpoint source pollutants on aquatic biota in the Upper Powell River System, Virginia. Human and Ecological Risk Assessment 9(4):1059-1073.

Soulé, M. 1987. Viable populations for conservation. Cambridge University Press, Great Britain. 189 pp.

Soulé, M. E. 1990. The onslaught of alien species, and other challenges in the coming decades. Conservation Biology 4: 233-239.

Southerland, M. T. 1986. The effects of variation in streamside habitats on the composition of mountain salamander communities. Copeia 1986: 731–741.

Sparling, D. W., G. M. Fellers, and L. L. McConnell. 2001. Pesticides and amphibian population declines in California, USA. Environmental Toxicology and Chemistry 20: 1591–1595.

Speight, M. (1973). Outdoor Recreation and its Ecological Effects: a bibliography and review. Discussion Papers in Conservation No. 4, University College, London. 35 pp.

Spencer, R.-J. 2002. Experimentally testing nest site selection: fitness trade-offs and predation risk in turtles. Ecology 83: 2136–2144.

Spencer, R.-J. and Thompson, M.B. 2003. The significance of predation in nest site selection of turtles: an experimental consideration of macro- and microhabitat preferences. Oikos 102(3): 592-600.

Spinks, P.Q, G.B. Pauly, J.J. Crayon, and H.B. Shaffer. 2003. Survival of the western pond turtle (*Emys marmorata*) in an urban California environment. Biol. Conserv. 113: 257-67.

Stebbins, R.C., and N.W. Cohen. 1995. A Natural History of Amphibians. Princeton University Press, Princeton, NJ.

Stiven, A. E., and R. C. Bruce. 1988. Ecological genetics of the salamander Desmognathus quadramaculatus from disturbed watersheds in the southern Appalachian biosphere reserve cluster. Conserv. Biol. 2: 194-205.

Strayer, D.L. 2006. Challenges for freshwater invertebrate conservation. Journal of the North American Benthological Society 25(2): 271–287.

Steen, D.A., J.P. Gibbs, K.A. Buhlmann, J.L. Carr, B.W. Compton, J.D. Congdon, J.S. Doody, J.C. Godwin, K.L. Holcomb, D.R. Jackson, F.J. Janzen, G. Johnson, M.T. Jones, J.T. Lamer, T.A. Langen, M.V. Plummer, J.W. Rowe, R.A. Asumure, J.K. Tucker, and D.S. Wilson. 2012. Terrestrial habitat requirements of nesting freshwater turtles. Biological Conservation 150: 121–128.

Stuart, S., J.S. Chanson, N.A. Cox, B.E. Young, A.S.L. Rodrigues, D.L. Fishman, and R.W. Waller. 2004. Status and trends of amphibian declines and extinctions worldwide. Science 306: 1783-1786.

Sullivan, B. K. 1981. Distribution and relative abundance of snakes along a transect in California. Journal of Herpetology 15: 247-248.

Sutherland, R.W. 2009. The Effects of Urbanization on Reptiles and Amphibians in the Sandhills Region of North Carolina, *available at* http://dukespace.lib.duke.edu/dspace/bitstream/handle/10161/3199/MicrosoftWord-RonaldSutherlandDissertationFinalDraft20096_1_09.pdf?sequence=1 (last visited Feb. 17, 2012).

Szaro, P.C., S.C. Belfit, J. K. Aitldn, and J.N. Rinne. 1985. Impact of grazing on a riparian garter snake. Pages 359-363 in R. R. Johnson, C. D. Ziebeil, D. R. Patton, P. F. Ffoiliott, and F. H. Hamre, technical coordinators. Riparian ecosystems and their management: Reconciling conflicting uses. General Technical Report RM-120. U.S. Forest service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado.

Tacon, A.G.J. and I.P. Forster. 2003. Aquafeeds and the environment: policy implications. Aquaculture 226: 181–189

Taylor, R.B. Undated. The effects of off-road vehicles on ecosystems, *available at* http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=1&ved=0CC4QFjAA&url=http%3A%2F%2Fwww.tpwd.state.tx.us%2Fpublications%2Fpwdpubs%2Fmedia%2Fpwd_rp_t3200_1081.pdf&ei=k95DT_X3K8XJqgGmhPncCg&usg=AFQjCNEpkEWgr_8vRMt_dAoZGJXg7NX7wQ (last visited Feb. 21, 2012).

Todd B.D., B.B Rothermel. 2006. Assessing quality of clearcut habitats for amphibians: Effects on abundances versus vital rates in the southern toad (Bufo terrestris). Biological Conservation 133: 178–185.

Todd B.D., and K.M. Andrews. 2008. Response of a reptile guild to forest harvesting. Conservation Biology 22: 753–761.

Todd B.D., B.B. Rothermel, R.N. Reed, T.M. Luhring, K. Schlatter, L. Trenkamp, J.W. Gibbons. 2008. Habitat alteration increases invasive fire ant abundance to the detriment of amphibians and reptiles. Biological Invasions 10: 539–546.

Todd, B.D., T.M. Luhring, B.B. Rothermel, and J.W. Gibbons. 2009. Effects of forest removal on amphibian migrations: Implications for habitat and landscape connectivity. Journal of Applied Ecology 46: 554–561.

Townsend, P.A., D.P. Helmers, C.C. Kingdon, B.E. McNeil, K.M de Beurs, and K.N. Eshleman. 2009. Changes in the extent of surface mining and reclamation in the Central Appalachians detected using a 1976–2006 Landsat time series. Remote Sensing of Environment 113: 62–72.

Trauth, J.B, S.E. Trauth, and R.L. Johnson. 2006. Best Management Practices and Drought Combine to Silence the Illinois Chorus Frog in Arkansas. Wildlife Society Bulletin 34(2): 514.

Tucker, J. K. 2000a. Growth and survivorship in the Illinois chorus frog. Transactions of the Illinois State Academy of Science 93:63-68.

Turner, I. M. 1996. Species loss in fragments of tropical rain forest: a review of the evidence. Journal of Applied Ecology 33: 200–209.

Turner, L. J., and D. K. Fowler. 1981. Utilization of surface mine ponds in east Tennessee by breeding amphibians. U. S. Fish and Wildlife Service Report FWS/OBS-81/08. 13 pp.

Turtle, S. L. 2000. Embryonic survivorship of the spotted salamander (*Ambystoma maculatum*) in roadside wand woodland vernal pools in southeastern New Hampshire. Journal of Herpetology 34: 60-67.

Twigg, L. E., and B. J. Fox. 1991. Recolonization of regenerating open forest by terrestrial lizards following sand mining. Australian Journal of Ecology 16: 137-148.

Tyler, T. J., W. J. Liss, R. L. Hoffman, and L. M. Ganio. 1998. Experimental analysis of trout effects on survival, growth, and habitat use of two species of ambystomatid salamanders. Journal of Herpetology 32(3): 345-349.

Urbina-Cardona, J.N., M. Olivares-Perez, V.H. Reynoso. 2006. Herpetofauna diversity and yellow-legged frog (Rana muscosa). Copeia 2001(4): 945-953.

U.S. Census Bureau. 2005. Statistical Abstract of the United States, *available at* http://www.census.gov/prod/2005pubs/06statab/pop.pdf (last visited Feb. 17, 2012).

U.S. Census Bureau. 2009. Projected Population by Single Year of Age, Sex, Race, and Hispanic Origin for the United States: July 1, 2000 to July 1, 2050, *available at* http://www.census.gov/population/www/projections/downloadablefiles.html (last visited Feb. 17, 2012).

U.S. Census Bureau. 2012. U.S. and World Population Clocks, *available at* http://www.census.gov/main/www/popclock.html (last visited Feb. 17, 2012).

U.S. Environmental Protection Agency (EPA). 2003. National management measures for the control of nonpoint pollution from agriculture. EPA-841-B-03-004. Washington, D.C., USA.

U.S. Environmental Protection Agency (EPA). 2004. National Water Quality Inventory: Report to Congress, *available at* http://www.epa.gov/owow/305b/2004report (last visited March 23, 2012).

U.S. Environmental Protection Agency (EPA). 2005. Mountaintop Mining/Valley Fills in Appalachia Final Programmatic Environmental Impact Statement. October 2005. EPA 9-03-R-05002.

U.S. Environmental Protection Agency (EPA). 2009. Managing Nonpoint Source Pollution from Agriculture, *available at*: http://www.epa.gov/nps/facts/point6.htm (last visited April 4, 2012).

U.S. Environmental Protection Agency. Undated. Protecting Wetlands for Amphibians and Reptiles.

U.S. Fish and Wildlife Service (USFWS). 1994. Procedural Guidance for the Preparation and Processing of Rules and Notices Pursuant to the Endangered Species Act, *available at* http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=4&ved=0CDsQFjAD&url=http%3A%2F%2Ftraining.fws.gov%2FEC%2FResources%2FES_Listing_and_Candidate_Assessment%2FESA%2520Folder%2FFWS%2520Listing%2520Handbook.pdf&ei=XgqDT7uoI5K

c8QSkidjUBw&usg=AFQjCNFyHDRVaKzf5PcTKaDMxXBQe18DMQ (last visited April 9, 2012).

U.S. Fish and Wildlife Service (USFWS). 1996. Formal Section 7 Biological Opinion and Conference Report on Surface Coal Mining and Reclamation Operations Under the Surface Mining Control and Reclamation Act of 1977. September 24, 1996. 15 pp.

U.S. Fish and Wildlife Service (USFWS). 1997. Endangered and Threatened Wildlife and Plants; Determination of Endangered Status for the Cumberland Elktoe, Oyster Mussel, Cumberlandian Combshell, Purple Bean, and Rough Rabbitsfoot, Final Rule. 62 FR 1647. January 10, 1997.

U.S. Fish and Wildlife Service (USFWS). 2009. Mississippi Gopher Frog Recovery Action Plan, *available at* http://ecos.fws.gov/docs/action_plans/doc3100.pdf (last visited Oct. 14, 2011).

U.S. Fish and Wildlife Service (USFWS). 2009b. Determination of Endangered Status for Reticulated Flatwoods Salamander; Designation of Critical Habitat for Frosted Flatwoods Salamander and Reticulated Flatwoods Salamander. 74 Fed. Reg. 6702 (Feb. 10, 2009).

U.S. Fish and Wildlife Service (USFWS). 2010. Conservation, Status & Monitoring Working Group Findings and Recommendations, *available at* http://www.fws.gov/international/DMA_DSA/CITES/Appendix_III/turtles_ws.html (last visited Feb. 8, 2011).

United States Army Corps of Engineers. 2001. Draft, nationwide permits. Programmatic environmental impact statement. The Institute for Water Resources, Alexandria, Virginia, USA.

USDA-NRCS. 1997. America's private land: a geography of hope. Program Aid 1548. USDA-Natural Resources Conservation Service: 154.

USDI-BLM, USDA-Forest Service. 1995. Rangeland Reform '94 Final Environmental Impact Statement. USDIBLM. Washington, DC: 26.

van Dijk, P.P. and Harding, J. 2010. *Glyptemys insculpta*. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.1, *available at* www.iucnredlist.org (last visited Oct. 3, 2011).

van Dijk, P.P., B.L. Stuart,A.G.J. Rhodin, eds. 2000. Asian Turtle Trade: Proceedings of a Workshop on Conservation and Trade of Freshwater Turtles and Tortoises in Asia. Lunenberg (MA): Chelonian Research Foundation. Chelonian Research Monographs, vol. 2.

van Dijk, P.P. 2010. Impacts of non-harvest anthropogenic activities on freshwater turtles in the United States in U.S. Fish and Wildlife Service, Conservation and Trade Management of Freshwater and Terrestrial Turtles in the United States: Workshop Presentation Abstracts, available at http://www.fws.gov/international/DMA_DSA/CITES/Appendix_III/turtles_ws.html (last visited Feb. 8, 2011).

van Gelder, J. J. 1973. A quantitative approach to the mortality resulting from traffic in a population of *Bufo bufo*). Oecologia 13: 93-95.

Vial, J. L., and L. Saylor. 1993. The status of amphibian populations: a compilation and analysis. Working Document No. 1. The International Union for the Conservation of Nature and Natural Resources/Species Survival Commission, Declining Amphibian Populations Taskforce, Buckinghamshire, United Kingdom.

Villarreal, M.L.,Charles van Riper III, Robert E. Lovich, Robert L. Palmer, Travis Nauman Sarah E. Studd, Sam Drake, Abigail S. Rosenberg, Jim Malusa, and Ronald L. Pearce. 2011. An Inventory and Monitoring Plan for a Sonoran Desert Ecosystem: Barry M. Goldwater Range–West, *available at* http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=71&ved=0CCYQFjAAOEY&url=http%3A%2F%2Fpubs.usgs.gov%2Fof%2F2011%2F1232%2Fof2011-1232.pdf&ei=pcPfTqIRiqraBbaL6asF&usg=AFQjCNGFrpTGf_qPC-6JHrB2BABhbo9xag (last visited Dec. 7, 2011).

Vitousek, P.M., H.A. Mooney, J. Lubchenco, J.M. Melillo, 1997. Human domination of Earth's ecosystems. Science 277: 494–499.

Vitt, L.J., Caldwell, J.P., Wilbur, H.M., and Smith, D.C. 1990. Amphibians as harbingers of decay. Bioscience 40: 418.

Vogt, R.C. 1981. Natural History of Amphibians and Reptiles in Wisconsin. Milwaukee Public Museum, Milwaukee, Wisconsin. 205 pp.

Vos, C. C. 1997. Effects of road density: A case study of the moor frog. Fragmentation & Infrastructure: 93-97.

Vos, C. C., and J. P. Chardon. 1998. Effects of habitat fragmentation and road density on the distribution pattern of the moor frog Rana arvalis. Journal of Applied Ecology 35:44-56.

Voyles, J., Vance T. Vredenburg, Tate S. Tunstall, John M. Parker, Cheryl J. Briggs, Erica Bree Rosenblum. 2012. Pathophysiology in Mountain Yellow-Legged Frogs (Rana muscosa) during a Chytridiomycosis Outbreak. PLOS One, *available at* http://www.plosone.org/article/info%3Adoi%2F10.1371%2Fjournal.pone.0035374.

Vredenburg, V. T. 2004. Reversing introduced species effects: experimental removal of introduced fish leads to rapid recovery of a declining frog. Proceedings of the National Academy of Sciences 101: 7646-7650.

Wake, D. 2012. Facing Extinction in Real Time. Science 335: 1052.

Wake, D.B. 1991. Declining amphibian populations. Science 253: 860.

Walther, G-R., E. Post, P. Convey, A. Menzel, C. Parmesan, T.J.C Beebee, J.-M. Fromentin, O. Hoegh-Guidberg, F. Bairlein. 2002. Ecological responses to recent climate change. Nature. 416: 389-395.

Ward, K. Jr. 2009. Mining: West Virginia mountains go unrestored—OSM report. Charleston Gazette July 27, 2009.

Warwick, C. 1990. Disturbance of natural habitats arising from rattlesnake round-ups. Environmental Conservation 17: 172-174.

Water Environment Federation, The. 1993. The Clean Water Act of 1987. The Water Environment Federation 318 pp.

Weatherhead, P. J. and K. A. Prior. 1992. Preliminary observations of habitat use and movements of the eastern massasauga rattlesnake. Journal of Herpetology 26: 447-452.

Weir, J. 1992. The Sweetwater rattlesnake round-up: A case study in environmental ethics. Conservation Biology 6: 116-127.

Wells, K.D. 2007. The ecology and behavior of amphibians. Chicago, IL: University of Chicago Press. 1400 p.

Welsh, H. H. Jr. 1990. Relictual amphibians and old-growth forests. Conserv. Biol. 4: 309-319.

Welsh, H. H. Jr., and L. M. Ollivier. 1998. Stream amphibians as indicators of ecosystem stress: a case study from California's redwoods. Ecological Applications 8(4): 1118-1132.

Welsh, H.H. Jr., and Droege S. 2001. A case for using plethodontid salamanders for monitoring biodiversity and ecosystem integrity of North American forests. Conservation Biology 15: 558–569.

Welsh, H. H. Jr., K. L. Pope, and D. Boiano. 2006. Sub-alpine amphibian distributions related to species palatability to non-native salmonids in the Klamath Mountains of northern California. Diversity and Distributions 12: 298-309.

Weygoldt, P. 1989. Changes in the composition of mountain stream frog communities in the Atlantic Mountains of Brazil: frogs as indicators of environmental deteriorations? Studies on Neotropical Fauna and Environment 243: 249-255.

Whitaker, P. B. and R. Shine. 2000. Sources of mortality of large elapid snakes in an agricultural landscape. Journal of Herpetology 34: 121-128.

White, J. A. 2004. Geomorphic analysis of stream crossings in a portion of the upper Cheat River basin. Thesis, West Virginia University, Morgantown, USA.

Wilbur, H.M. 1997. Experimental ecology of food webs: complex systems in temporary ponds. Ecology 78: 2279–2302.

Wilbur, H.M. and P.J. Morin. 1988. Life history evolution in turtles, p. 387-439. In C. Gans and R.B. Huey (eds.), Biology of the Reptilia, Vol. 16. Alan R. Liss, New York.

Wilbur, H.M. 1980. Complex life cycles. Annual Review of Ecology and Systematics 11: 67–93.

Wilen, B.O., and W.E. Frayer. 1990. Status and trends of US wetlands and deepwater habitats. Forest Ecology and Management 33–34: 181–192.

Wilkinson, J. 2008. Habitat destruction, alteration and fragmentation, *available at* http://amphibiaweb.org/declines/HabFrag.html (last visited Feb. 15, 2012).

Wilkinson, T. 1996a. Utah ushers its frogs toward oblivion. High Country News 19: 1, 10-13. May 27, 1996.

Wilkinson, T. 1996b. Reptile rustlers. National Parks 1996 (November/December): 36-39.

Williams, T. 1995. The terrible turtle trade. Audubon. 5 pp.

Williams, J.M. 2003. Impacts on Terrestrial and Streamside Herpetofauna by Mountaintop Removal Mining in Southern West Virginia. Ph.D. Dissertation, *available at* https://eidr.wvu.edu/eidr/documentdata.eIDR?documentid=3231 (last visited Feb. 17, 2012).

Willson, J. D., and M. E. Dorcas. 2003. Effects of habitat disturbance on stream salamanders: Implications for buffer zones and watershed management. Conservation Biology 17: 763-771.

Wilson, E.O., 1999. The Diversity of Life. Longitude, New York, USA.

Wind, E. 1996. Habitat associations of wood frogs (Rana sylvatica), and effects of fragmentation, in boreal mixedwood forests. M.Sc. Thesis, Univ. British Columbia, Vancouver, BC.

Wojcik, D.P., C.R. Allen, R.J. Brenner, E.A. Forys, D.P. Jouvenaz, and R.S. Lutz. 2001. Red imported fire ants: impact on biodiversity. American Entomologist. 47(1): 16-23.

Wood, D. 2009. Memorandum to J. Bailey, J. Wirts, and P. Campbell. Evidence for Secretary Randy Huffman. June 30, 2009. West Virginia Department of Environmental Protection, Charleston, WV.

Wood, R.C. and R. Herlands. 1997. Turtles and tires: the impact of roadkills on Northern Diamondback Terrapin, Malaclemys terrapin terrapin, populations on the Cape May Peninsula, Southern New Jersey, USA. Pgs. 46–53. In: J. Van Abbema, ed. Proceedings: Conservation, Restoration and Management of Tortoises and Turtles—An International Conference. New York Turtle and Tortoise Society, New York.

Wright, S. 1931. Evolution in Mendelian populations. Genetics 16:97-159.

Wu, M., M. Quirindongo, J. Sass, and A. Wetzler. 2009. Poisoning the Well: How the EPA is Ignoring Atrazine Contamination in Surface and Drinking Water in the Central United States. Natural Resources Defense Council Report. August 2009, *available at* http://www.nrdc.org/health/atrazine/default.asp (last visited April 3, 2012).

Wuerthner. G. 2002. Guzzling the West's water: squandering a public resource at public expense. Pages 185-187 in G. Wuerthner and M. Matteson (eds.). WELFARE RANCHING. Island Press. Covelo, CA.

Wuerthner, G. 2007. Thrillcraft: The Environmental Consequences of Motorized Recreation.

Wyman, R.L. 1990. What's happening to the amphibians? Conserv. Biol. 4: 350–352.

Yahner, R.H. 1988. Changes in wildlife communities near edges. Conservation Biology 2: 333–339.

Young, B.E., K.R. Lips, J.K. Reaser, R. Ibanez, A.W. Salas, J.R. Cedeno, L.A. Coloma, S. Ron, E. La Marca, J.R. Meyer, A. Munoz, F. Bolanos, G. Chaves, and D. Romo. 2001. Population declines and priorities for amphibian conservation in Latin America. Conservation Biology 15: 1213–1223.

Zheng, D., J. Chen, B. Song, M. Xu, P. Sneed, R. Jensen. 2000. Effects of silvicultural treatments on summer forest microclimate in southeastern Missouri Ozarks. Climate Research 15: 45–59.

Zhou Z, and Z. Jiang. 2004. International trade status and crisis for snake species in China. Conservation Biology 18: 1386–1394.

Zwartjes, P.W., J.E. Cartron, P.L. Stoleson, W.C. Haussamen, T.E. Crane. 2008. Assessment of Native Species and Ungulate Grazing in the Southwest: Terrestrial Wildlife. USDA Forest Service Gen. Tech. Rep. RMRS-GTR-142, *available at* http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=1&ved=0CB0QFjAA&url=http%3A%2F%2Fwww.fs.fed.us%2Frm%2Fpubs%2Frmrs_gtr142.pdf&ei=AQbUTqjrM8aI2gWev4XJDw&usg=AFQjCNHk_lE8EFe1YWKl-3OkfNrFYgH5CQ (last visited Nov. 28, 2011).

V. SPECIES ACCOUNTS

TURTLES

Western Pond Turtle (*Actinemys marmorata*)

Wood Turtle (*Glyptemys insculpta*)

Spotted Turtle (*Clemmys guttata*)

Rio Grande Cooter (*Pseudemys gorzugi*)

Alligator Snapping Turtle (*Macrochelys temminckii*)

Blanding's Turtle (*Emydoidea blandingii*)

Scientific Name:

Actinemys marmorata (formerly *Clemmys marmorata*)

Common Name:

Western Pond Turtle or Pacific Pond Turtle

G Rank:

G3

IUCN Red List:

Vulnerable (assessed in 1996, now under review)

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

Distribution and abundance of the western pond turtle have declined as a result commercial exploitation for the pet trade, habitat loss and degradation, introduced species, and (locally) disease. The western pond turtle is discontinuously distributed and generally uncommon or rare from western Washington (Puget Sound region, at least formerly) south to northwestern Baja California (known only from the Pacific slopes of the San Pedro Martir) (Buskirk 1990, Bury and Germano 2008, NatureServe 2011). In Washington, the turtle has been essentially extirpated from historic habitat in the lower Puget Sound, and only two populations remain in the Columbia River Gorge (Allen et al. 2003). In May 2002, the Canadian Species at Risk Act listed the Pacific pond turtle (another common name for the western pond turtle) as being extirpated in Canada.

Habitat:

The western pond turtle is found in permanent and intermittent waters of rivers, creeks, small lakes and ponds, marshes, irrigation ditches, and reservoirs (NatureServe 2011). It is sometimes found in brackish water. In a northern California stream, deep large pools with logs, branches, or boulders were favored sites (Bury 1972). The turtle commonly basks on land, near or away from water (Rathbun et al. 2002). The name "pond" turtle is something of a misnomer because this species more frequently lives in lotic habitats and spends a lot of time in terrestrial habitats (Ernst and Lovich 2009, p. 182).

To be sure, terrestrial habitat may be just as important as aquatic habitat for this turtle (Ernst and Lovich 2009). In some populations, males utilize terrestrial habitat for some portion of ten months annually, while females are on land during some of all months because of nesting and overwintering activities (Reese and Welsch 1997). In San Luis Obispo County, California, radio-tracked turtles spent 34-191 (mean 111) days in terrestrial refuges, generally under leaf litter in woodland and coastal sage scrub habitats, mainly from October to February (n = 43 turtle-years) (Rathbun et al. 2002). However, some did not leave aquatic habitat, and this flexibility occurs throughout the range of the species (Rathbun et al. 2002).

The western pond turtle usually nests on sandy banks near water or in fields or sunny spots up to a few hundred meters from water (Storer 1930, Nussbaum et al. 1983). In San Luis Obispo County, California, females nested in open areas with little vegetative cover that were 6-80 m (mean 28.2 m) (possibly up to 170 m) from water, 0.5-17.5 m in elevation above creek beds (Rathbun et al. 2002).

Biology and Taxonomy:

Mating, which has been rarely observed, typically occurs in late April or early May, but may occur year-round (Holland 1985a, 1991b). Females emigrate from the aquatic site to an upland location that may be a considerable distance (400 m or more) from the aquatic site to nest, but is often less, and deposit from 1-13 eggs that have a thin, but hard (calcified) outer shell in a shallow (ca. 10-12 cm deep) nests excavated by the females (Holland 1991a; Rathbun et al. 1992, 1993). Females may lay more than one clutch a year (Rathbun et al. 1993). Most oviposition occurs during May and June, although some individuals may deposit eggs as early as late April and as late as early August (Storer 1930; Buskirk 1992; Rathbun et al. 1992, 1993; D. Holland, pers. comm. as cited in Jennings and Hayes 1994).

Age and size at reproductive maturity varies with latitude (Jennings and Hayes 1994). In California, reproductive maturity occurs at between 7 and 11 years of age, and approximately 110-120 mm CL, with turtles maturing at a larger size and a more advanced age as one moves north, and males generally maturing at a slightly smaller sizes and younger ages than females (D. Holland, pers. comm. as cited in Jennings and Hayes 1994). Data on longevity are lacking, but western pond turtles are thought to be long-lived since the minimum age of a recaptured individual was 42 years from a population studied in northern California (Trinity County: B. Bury and D. Holland, pers. comm. as cited in Jennings and Hayes 1994).

Western pond turtles are dietary generalists and highly opportunistic (Holland 1991a), and will consume almost anything that they are able to catch and overpower. The most prominent part of western pond turtle behavior is the activities they perform to thermoregulate, which vary with ambient temperature based on time of day and season (Jenning and Hayes 1994). Ernst and Lovich (2009, p. 173-182) and Bury and Germano (2008) summarize additional information on reproduction and other aspects of western pond turtle biology.

NatureServe (2011) explains that based on morphological data, Holman and Fritz (2001) split *Clemmys* as follows: *Clemmys guttata* was retained as the only member of the genus; *Clemmys insculpta* and *C. muhlenbergii* were placed in the genus *Glyptemys* (as first reviser, Holman and Fritz gave *Glyptemys* Agassiz, 1857, precedence over the simultaneously published genus *Calemys* Agassiz, 1857); and *Clemmys marmorata* was transferred to the monotypic genus *Actinemys*.

Population Status:

The western pond turtle qualifies for endangered species status because it is declining rangewide with many areas experiencing extirpations because of past collection pressure and ongoing habitat destruction. The western pond turtle is declining in abundance rangewide, especially in the northernmost part and southern one-third of the range (NatureServe 2011). Specifically, three areas show marked and significant declines in populations: southern California from Baja up to Ventura, the Central Valley of California, and the northernmost populations in Washington and perhaps Oregon (Bury and Germano 2008). Today, only northern California and southern Oregon support large populations but even in those areas their status is uncertain (Ernst and Lovich 2009, p. 181). The species was a candidate for federal protection until the FWS eliminated the C2 category, but it currently receives no federal protection under the ESA.

Moreover, areas where the turtles have experienced significant declines and extirpations must be considered significant portions of the range because of the unique genetic variation in those areas that is essential to longterm viability of the species. Spinks and Shaffer (2005) identified four major clades of the western pond turtle. These included a large Northern clade composed of populations from Washington south to San Luis Obispo County, California, west of the Coast Ranges; a San Joaquin Valley clade from the southern Great Central Valley; a geographically restricted Santa Barbara clade from a limited region in Santa Barbara and Ventura counties; and a Southern clade that occurs south of the Tehachapi Mountains and west of the Transverse Range south to Baja California, Mexico.

In Oregon, the western pond turtle occurs widely but in low to very low densities (Holland 1993). Researchers observed the turtles in 83 of 313 sites surveyed in 1991 (Holland 1993). In the Willamette Valley in Oregon, western pond turtles appear to have declined to a level that represents roughly one percent of historic levels (Holland 1991a).

In Washington, the total population in the early 1990s was fewer than 100 individuals in the wild (Andelman and Gray 1992). The species survives in the state only in two populations in the Columbia River Gorge (Allen et al. 2003).

In California, there are probably a couple hundred extant occurrences statewide (Jennings and Hayes 1994). Many populations in California are small and declining with low viability (Jennings and Hayes 1994). In California's Central Valley, Germano and Bury (2001) surveyed 55 sites and they detected turtles at only 15 sites, and only 5 sites had sizable populations. Much of the natural habitat for the species in the Central Valley has been eliminated (Bury and Germano 2008). Brattstrom and Messer (1988) found that few viable populations remain in southern California and that only 6-8 viable populations remain south of the Santa Clara River system (including the desert slope) in California (Holland 1991a). As Jennings and Hayes explains: "[M]ost western pond turtle populations examined in this region appear to show an age (size) structure increasingly biased toward adults, indicating little or no recruitment is taking place." The decline in southern California has been rapid (Brattstrom 1988). In Baja California, most historic populations have been extirpated and only a few populations remain at remote localities (Holland 1991a).

The situation has undoubtedly declined since these studies published their findings. Moreover, the western pond turtle populations in some areas of northern California (e.g., the drainages entering Clear Lake, and portions of the Klamath River system in California) "are in equally serious or worse condition than those in southern California" (Jennings and Hayes 1994).

THREATS

Habitat alteration and destruction:

Decline of the western pond turtle is largely due to alteration, loss, and fragmentation of habitat (Bury and Germano 2008). Many populations have been lost as a result of urbanization and agricultural development in the area south of central California (Rathbun et al. 1992). And extensive draining of wetlands and habitat alteration in the last 100 years have left few aquatic areas in the Central Valley of California, where the species has seriously declined (Germano and Bury 2001, Bury and Germano 2008).

Massive water development projects have changed the location, flow, and use of water across most of the range of the species, particularly in the Central Valley of California (Bury and Germano 2008). Construction of dams on many rivers results in cooler water temperatures and faster flowing water, which is likely detrimental to turtle populations (Reese and Welsh 1998a,b). Also, the reservoirs behind these dams are likely unsuitable habitat because recreational activities such as fishing and skiing likely disturb the turtles (Bury and Germano 2008). In addition, some of these reservoirs have large draw downs seasonally, which inhibits growth of aquatic vegetation and associated invertebrate populations that are prey for turtles (Bury and Germano 2008). In northern California, damming of the mainstem Trinity River has likely negatively impacted juveniles (Reese and Welsh 1998). Specifically, the artificially colder thermal regime created by the hypolimnetic releases from the dam appears to be influencing the turtles' thermoregulatory behavior and forcing these animals to compensate by seeking alternative aquatic thermal refugia (Bettaso et al. 2006).

In addition, Jennings and Hayes (1994) explains that abusive grazing practices have eliminated many populations (see also Holland 1991a). And according to Holland (1994), road mortality probably matches or exceeds all other anthropogenic effects.

Overutilization:

Many populations declined as a result of historical commercial exploitation (Bury and Germano 2008, NatureServe 2011). As a relatively large turtle, the species was once widely utilized for food (Bury and Germano 2008, Ernst and Lovich 2009). Populations around San Francisco and in the Sacramento-San Joaquin River Delta were persecuted mercilessly for the food market (Bettelheim 2005).

The pet trade has also seriously reduced some populations. Bury (1989) reported that one pet wholesaler obtained about 500 individuals from a southern California lake. Betteleheim (2005) reports that large numbers were removed from southern California rivers and streams in the 1970s and 1980s.

In addition, some turtles are deliberately shot while basking and others are inadvertently caught while fishing (Bury and Germano 2008). Surveys in Oregon also indicate that western pond turtles are frequently caught on baited hooks and are subsequently released carrying a hook that can significantly impair or entirely prevent normal feeding (Mader 1988). Based on the weight loss observed in such turtles, a high likelihood exists that most of the individuals caught in this manner ultimately perish if released without removal of the hook.

Disease or predation:

In Washington, decline was exacerbated by an upper respiratory disease epidemic in 1990, leaving a total population of fewer than 100 individuals in the wild (Andelman and Gray 1992). Observations also suggest the potential occurrence of a similar disease syndrome in one northern California population (Jennings and Hayes 1994). As such, disease should be considered a threat to the western pond turtle.

The native raccoon can be a problem in situations where turtle habitat occurs in urban environments because of artificially high raccoon populations associated with supplemental food from human habitations (Bury and Germano 2008).

Inadequacy of existing regulatory mechanisms:

The western pond turtle has special status under state law within its entire native range (Bury and Germano 2008). It is listed as state endangered in Washington, sensitive/critical in Oregon, and a species of special concern in California. But none of these laws confer effective protection of habitat. Some successful recovery actions have occurred in Washington and a conservation strategy drafted for California, but these initiatives are merely voluntary and must be considered inadequate (Allen et al. 2001, Ashton and Welsh Jr. 2009, vander Haegen et al. 2009). Federal protection would lead to more resources being devoted to these and other recovery actions.

Some habitat for the turtle is protected in waters designated as Wild and Scenic Rivers but this protection applies to few rivers within the turtle's range. In addition, the turtle occurs in some state and federal parks but habitat management practices that harm the turtles are not prohibited. Moreover, only a small fraction of the turtle's habitat is found in these areas.

In 1991, a Western Pond Turtle Group was established to foster communication and coordinate research and develop a strategic plan for its conservation (Bury and Germano 2008). Despite these efforts, the western pond turtle continues to decline.

Other factors:

Nonnative turtles, including *Trachemys scripta*, likely have negatively affected western pond turtles through competition for basking sites and introduction of disease (Spinks et al. 2003, Patterson 2006).

In addition, introductions of non-native predators (bullfrogs and bass) have likely been detrimental (NatureServe 2011). Bullfrogs prey on hatchling or juvenile turtles (Moyle 1973, Holland 1991a). Bass (*Micropterus* spp.) are also known to prey on the smallest juveniles (Holland 1991a).

Chemical contaminants also threaten the western pond turtle. Spills of diesel fuel and other chemicals have harmed turtles in California and Oregon locations in the last few decades (Holland 1994, Luke and Sterner 2000, Lovich and Meyer 2002), sometimes with disastrous effects (Ernst and Lovich 2009, p. 181). Eggs of this turtle tested in Oregon contained low concentrations of organochlorine pesticides and PCBs, along with heavy metals mercury and chromium (Henny et al. 2003).

Finally, remaining isolated clusters of the turtles are separated by inhospitable areas of cities and roads (Bury and Germano 2008). As such, isolation should be considered a threat to the turtle.

References:

Allen, H.L., F. Slavens, K. Slavens, D. Anderson, K. McAllister, J. Lewis, and D. Shepherdson. 2001. Recovery program for the endangered western pond turtle in Washington state. Northwestern Naturalist 82(2): 65.

Andelman, S. J., and E. Gray. 1992. Viability and recovery of western pond turtle (*Clemmys marmorata*) populations in Washington state. Abstract, Society for Conservation Biology, 6th Annual Meeting, p. 33.

Ashton, Don T., and H.H. Welsh Jr. 2009. A Conservation Strategy for Western Pond Turtles in California. Turtle Newsletter 14: 21-23.

Bettaso, J.B., D.T. Ashton, H.H. Welsch, and R.M. Sullivan. 2006. Basking patterns and thermal regulatory behaviors of western pond turtles (*Clemmys marmorata*): comparing responses to thermal regimes in dammed and undammed tributaries of the Trinity River. Northwestern Naturalist 87(2): 155.

Bettelheim, M. 2005. Marmorata: The famed mud turtle of the San Francisco market. California Hist. 82: 26-42, 65-68.

Brattstrom, B.H. 1988. Habitat destruction in California with special reference to *Clemmys marmorata*: a perspective. In H.F. DeLisle, et al. (eds.), Proceedings of the Conference on California Herpetology. Southwest. Herpetol. Soc., Van Nuys, California.

Brattstrom and Messer. 1988. Current status of the southwestern pond turtle, *Clemmys marmorata pallida*, in southern California. Final report for California Dept. Fish and Game, Contract C-2044. 47 pp.

Bury, R. B. 1972. Habits and home range of the Pacific pond turtle, *Clemmys marmorata*, in a stream community. Ph.D. dissertation, Univ. California, Berkeley.

Bury, R.B, and D.J. Germano. 2008. *Actinemys marmorata* (Baird and Girard 1852) – Western pond turtle, Pacific pond turtle. Conservation Biology of Freshwater Turtles and Tortoises: A compilation project of the IUCN/SSC Tortoise and Freshwater Turtle Specialist Group. Chelonian Research Monographs No. 5.

Buskirk, J. R. 1990. Geographic distribution: *Clemmys marmorata*. SSAR Herpetol. Rev. 21(1):26.

Buskirk, J. R. 1991. An overview of the western pond turtle, *Clemmys marmorata*. Pages 16-23 in Beaman, K. R., F. Caporaso, S. McKeown, and M. D. Graff, editors. Proceedings of the First International Symposium on Turtles and Tortoises: Conservation and Captive Husbandry. Chapman University, Orange, California.

Buskirk, J. R. 1992. An overview of the western pond turtle, *Clemmys marmorata*. pp. 16-23 In: K. R. Beaman, F. Caporaso, S. McKeown, and M. Graff (editors), Proceedings of the first international symposium on turtles and tortoises: Conservation and captive husbandry. California Turtle and Tortoise Club, Van Nuys, California.

Ernst, C.H. and J.E. Lovich. 2009. Turtles of the United States and Canada. The John Hopkins University Press. 827 pages.

Germano, D.J. and R.B. Bury. 2001. Western pond turtles (*Clemmys marmorata*) in the Central Valley of California: Status and population structure. Trans. West. Sect. Wildl. Soc. 37: 22-36.

Henny, C.J., K.F. Beal, R.B. Bury, and R. Goggans. 2003. Organochlorine pesticides, PCBs, trace elements and metals in western pond turtle eggs from Oregon. Northwest Sci. 77: 46-53.

Holland, D. C. 1985a. An ecological and quantitative study of the western pond turtle (*Clemmys marmorata*) in San Luis Obispo County, California. MA Thesis, Fresno State

University, Fresno, California.

Holland, D. C. 1991a. A synopsis of the ecology and status of the western pond turtle (*Clemmys marmorata*) in 1991. Report prepared for the United States Fish and Wildlife Service, National Ecology Research Center, San Simeon Field. Station, San Simeon, California.

Holland, D. C. 1991b. Status and reproductive dynamics of a population of western pond turtles (*Clemmys marmorata*) in Klickitat County, Washington in 1991. Report prepared for the Washington Department of Wildlife, Olympia, Washington.

Holland, D. C. 1993. A synopsis of the distribution and current status of the western pond turtle (*Clemmys marmorata*) in Oregon. Unpublished report, Nongame Division, Oregon Department of Fish and Wildlife.

Holland, D.C. 1994. The western pond turtle: habitat and history. Final Rep. U.S. Dept. of Energy, Bonneville Power Adm. Proj. 92-068, Contr. DE-BI79-92B62137.

Jennings, M.R., and M.P. Hayes. 1994. Amphibian and reptile species of special concern in California. California Dept. of Fish Game, Rancho Cordova.

Lovich, J.E. and K. Meyer. 2002. The western pond turtle (*Clemmys marmorata*) in the Mojave River, California, USA: Highly adapted survivor or tenous relict? J. Zool. (London) 256: 537-45.

Luke, C. and D. Sterner. 2000. Possible impacts of the Cantara spill on reptile populations along the upper Sacramento River. California Fish Game 86: 61-71.

Mader, D. R. 1988. Herpetological medicine: Off-feed animals. The Vivarium 1(1): 31-34.

Moyle, P. B. 1973. Effects of introduced bullfrogs, *Rana catesbeiana*, on the native frogs of the San Joaquin Valley, California. Copeia 1973(1): 18-22.

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, *available at* http://www.natureserve.org/explorer.

Nussbaum, R.A., E.D. Brodie, Jr., and R.M. Storm. 1983. Amphibians and Reptiles of the Pacific Northwest. University Press of Idaho, Moscow, Idaho. 332 pp.

Patterson, L.C. 2006. Life history and ecology of an introduced population of red-eared sliders (*Trachemys scripta elegans*) in the Central Valley of California with implications for the conservation of the western pond turtle (*Emys marmorata*). M.S. Thesis, California State University, Sacromento.

Rathbun, G. B., M. R. Jennings, T. G. Murphey, and N. R. Siepel. 1993. Status and ecology of sensitive aquatic vertebrates in lower San Simeon and Pica Creeks, San Luis Obispo County, California. Unpublished Report, National Ecology Research Center, Piedras Blancas Research Station, San Simeon, California, under Cooperative Agreement (14-16-0009-91-1909).

Rathbun, G. B., N. Siepel, and D. C. Holland. 1992. Nesting behavior and movements of western pond turtles (Clemmys marmorata). The Southwestern Naturalist 37(3): 319-324.

Rathbun, G. B., N. J. Scott, Jr., and T. G. Murphey. 2002. Terrestrial habitat use by Pacific pond turtles in a Mediterranean climate. Southwestern Naturalist 47: 225-235.

Reese, D. A., and H. H. Welsh, Jr. 1997. Use of terrestrial habitat by western pond turtles, *Clemmys marmorata*: implications for management. Pages 352-357 in J. V. Abbema, editor. Proceedings: conservation, restoration, and management of tortoises and turtles--an international conference. New York Turtle and Tortoise Society and Wildlife Conservation Society, New York.

Reese, D. A., and H. H. Welsh, Jr. 1998a. Comparative demography of *Clemmys marmorata* populations in the Trinity River of California in the context of dam-induced alterations. Journal of Herpetology 32: 505-515.

Reese, D. A., and H. H. Welsh, Jr. 1998b. Habitat use by western pond turtles in the Trinity River, California. Journal of Wildlife Management 62: 842-853.

Spinks, P.Q. and B.H Shaffer. 2005. Range-wide molecular analysis of the western pond turtle (*Emys marmorata*): cryptic variation, isolation by distance, and their conservation implications. Molecular Ecology 14(7): 2047.

Spinks, P.Q, G.B. Pauly, J.J. Crayon, and H.B. Shaffer. 2003. Survival of the western pond turtle (*Emys marmorata*) in an urban California environment. Biol. Conserv. 113: 257-67.

Storer, T. I. 1930. Notes on the range and life-history of the Pacific fresh-water turtle, *Clemmys marmorata*. Univ. Calif. Publ. Zool. 32(5): 429-441.

vander Haegen, W.M., S.L. Clark,K.M. Perillo, D.P. Anderson, and H.L. Allen. 2009. Survival and Causes of Mortality of Head-Started Western Pond Turtles on Pierce National Wildlife Refuge, Washington. Journal of Wildlife Management 73(8): 1402-7.

Scientific Name:

Glyptemys insculpta

Common Name:

Wood Turtle

G Rank:

G3

IUCN Red List:

Endangered

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

The wood turtle is found in eastern North America, from Cape Breton Island, Nova Scotia, New Brunswick, and Quebec in Canada; south to northern Virginia and Eastern Panhandle of West Virginia; west through the Great Lakes region (including southern Ontario) to eastern Minnesota, northeastern Iowa, and western Pennsylvania (Bleakney 1963, Gilhen and Grantmyer 1973, Green and Pauley 1987, Quinn and Tate 1991, Conant and Collins 1991, Harding 1997). It may have once occurred in northeastern Ohio (Thompson 1953). Multiple factors such as dam construction, agriculture, and urban expansion have contributed to the reduction of suitable sites across its distribution range (Litzgus and Brooks 1996, Mitchell and Klemens 2000).

Habitat:

Wood turtles live along permanent streams during much of each year but in summer may roam widely overland and can be found in a variety of terrestrial habitats adjacent to streams, including deciduous woods, cultivated fields, and woodland bogs, marshy pastures. Use of woodland bogs and marshy fields is most common in the northern part of the range (NatureServe 2011). Basking sites include emergent logs over deep stream channels, stream banks, and woodland openings with low ground cover (Ernst and Lovich 2009).

Wood turtles overwinter in bottoms or banks of streams where water flows all winter, including pools underneath a layer of ice (NatureServe 2011). Underwater muskrat burrows, beaver lodges, or over-bank root systems also may be used as winter hibernation sites (Ernst 1986).

Biology and Taxonomy:

The wood turtle is endemic to North America. It is in the genus *Glyptemys*, a designation given to only one other turtle: the bog turtle (*Glyptemys muhlenbergii*).

Like most turtle populations, the viability of wood turtle populations is dependent upon long-lived adults. Wood turtles have delayed maturity, low fecundity (small clutch size and one clutch per year), high nest predation, and high juvenile mortality rates. As a result, recruitment in most populations is quite low. This low recruitment must be balanced by high adult survival rates, and therefore stable populations depend on stable numbers of adults reproducing at fairly constant rates (see also Harding 1991, Harding 1997). Combining an increase in adult mortality rate (even a very small increase) with a high juvenile mortality rate can lead to rapid decline and extirpation (Congdon et al. 1993, 1994; Compton 1999).

The combination of late maturity, low reproductive success, and long-lived adults also results in a population structure skewed heavily toward adults (NatureServe 2011). These characteristics combine to delay the detection of population declines (NatureServe 2011). A given population may in fact be a "ghost population" in which adults are surviving from year to year but there is no successful reproduction (Bowen and Gillingham 2004). In addition, low mobility and the tendency to home reduces the probability of recolonization of decimated wood turtle populations (NatureServe 2011).

Unusual feeding behavior has been observed in wood turtles: wood turtles sometimes cause earthworms to come to the surface by stomping their front feet (Ernst and Lovich 2009).

Population Status:

The wood turtle qualifies for endangered species status because it is rare and declining across its range and it is subject to continuing threats of habitat destruction, overcollection, and predation. It is considered to be one of the most endangered freshwater turtles in North America and is classified as Endangered in most of the states in which it occurs (Garber et al. 1994, Klemens

2000, Castellano et al. 2009). The principal factor leading to the decline of this species is habitat loss caused by dam construction, agriculture, and urban development (Saumure and Bider 1998, Klemens 2000).

Available evidence indicates that the wood turtle is declining across its range (Ernst and McBreen 1991, Harding 1991, Ernst and McBreen 1991c, Farrell and Graham 1991, Klemens 1993, Garber and Burger 1995, Lovich 1995, Burger and Garber 1995, Dahl 1996, Oldfield 1996, Litzgus and Brooks 1996, Levell 1997, Buech et al. 1997a, Burke et al. 2000, Ernst 2001b, Harding 2002, Bowen and Gillingham 2004, Daigle and Jutras 2005, Daigle and Jutras 2005, Akre and Ernst 2006, Jones 2009, NatureServe 2011). Decline in population size over the past three generations (which likely exceeds 50 years) probably has been substantial (NatureServe 2011, van Dijk and Harding 2010). It is still extant in all 21 states and Canadian provinces from which recorded but it is not known to be stable or increasing in any substantial portion of the range (NatureServe 2011).

Harding and Bloomer (1979) describe the loss of the turtle from the Northeast: "This species was common in many parts of northern New Jersey in the 1950's. Today it is almost totally eradicated from eastern and north-central New Jersey due to urban development, and is declining rapidly in much of northwestern New Jersey, southeastern New York, and eastern Pennsylvania." It is widespread but apparently rare in Maine (Hunter et al. 1992).

The wood turtle is also declining from the southern parts of its range. The situation has undoubtedly worsened since 1991 when Ernst and McBreen wrote: "Recent development in Fairfax and Loudoun counties threatens the existence of 33 percent of the reported localities in Virginia, and three additional unreported populations in Fairfax and Arlington counties have disappeared since 1979 because of habitat destruction. . . . As the human population of northern Virginia continues to spread westward, colonies of the Wood Turtle in the Shenandoah Valley may be threatened or eliminated."

In the Great Lakes region, this species is generally uncommon to rare (Harding 1997). It is rare in Minnesota and uncommon even in suitable habitat (Oldfield and Moriarty 1994). Field data suggest a nearly complete lack of recruitment in Iowa (Spradling et al. 2011).

The species has disappeared from the southern parts of both Ontario and Quebec in conjunction with development and high road densities (COSEWIC 2007). In southern Quebec, a local population in an agricultural area along the Sutton River declined by 50 percent over seven years (Daigle and Jutras 2005). Indeed, much of the turtle's habitat in Canada has been permanently modified by human activity (Kerr and Deguise 2004).

Ernst and Lovich (2009, p. 262) summarize available literature on wood turtle density across its range.

THREATS

As explained below, habitat destruction, human exploitation, agricultural accidents, and highway mortality are the greatest causes of adult mortality (Ernst and McBreen 1991c, Harding 1991, Garber and Burger 1995, Buech et al. 1997a, Ernst 2001b, Wilson et al. 2003a, Daigle and Jutras 2005, Desroches and Picard 2005, VDGIF 2005, Saumeure et al. 2007).

Habitat alteration and destruction:

Habitat degradation, fragmentation and destruction are serious problems across the wood turtle's range. Since the submission of a wood turtle listing petition in 1994 (RESTORE et al. 1994), human populations and development within the turtle's range have substantially increased and the amount of habitat destroyed, degraded, and/or fragmented has gotten substantially worse.

On private land, specific threats include residential and recreational developments (particularly second homes/cabins) and associated infrastructure (van Dijk and Harding 2010). To be sure, in the eastern part of its range, urbanization is responsible for destroying much wood turtle habitat (Harding and Bloomer 1979, Ernst and McBreen 1991).

Land management practices in the public lands where the main populations are located also threaten the turtles. Potential threats include damming, streambank stabilization, and intensive timber harvesting activities within 300 m of inhabited wetlands (Harding and Bloomer 1979, Harding 1991, Buech and Nelson 1997, Bower and Gillingham 2004). For example, Compton (1999) found that damming and subsequent flooding through water release destroyed 25 percent of nests in Maine. Damming and channelization of rivers and streams is destroying wood turtle habitat across its range (Harding and Bloomer 1979, Harding 1991, Buech and Nelson 1997).

In addition, certain fisheries management practices, such as sand bank stabilization and the digging of sand traps in streams, can eliminate nesting sites and reduce preferred turtle habitat (Harding 1997, Bower and Gillingham 2004). Wusterbarth (2000) found that wood turtles appeared to prefer unstabilized rather than stabilized banks for nesting along the Manistee River.

Development near wood turtle habitat also increases human access and likely leads to overexploitation, which is discussed below. In Connecticut, two formerly stable wood turtle populations declined drastically after a protected drinking water supply area was opened to recreational use (Garber and Burger 1995). Presumably, most of the turtles that disappeared were taken by people (NatureServe 2011). As another example, wood turtle decline have been observed at sites in Minnesota subject to a variety of recreational uses (Oldfield 1996).

With increasing development, adult mortality due to road traffic also increases (Harding 1997). Fatal encounters with recreational vehicles and agricultural machinery also increase with development (van Dijk and Harding 2010). Populations of turtles cannot remain viable in the face of the additional adult mortality caused by road-kill, and as such, road kill from vehicle use on roads is a primary threat (Gibbs and Shriver 2002, Steen et al. 2006, Langen et al. 2009). Numerous observers have seen wood turtles killed on roads (see, e.g., Krichbaum 2009, Akre and Ernst 2006, Ernst 2001a, and Langen et al. 2007). Biased sex ratios are a harmful secondary effect of road mortality because female turtles are more likely to cross roadways and be hit than are males (Steen et al. 2006).

Mortality from mowing of agricultural fields in another significant threat (Akre and Ernst 2006, Castellano et al. 2008, Erb and Jones 2011). In fact, studies have reported substantially higher rates of mortality resulting from agricultural activities than from automobiles (Saumure 2004, Saumure et al. 2007, Jones 2009). A study in Quebec indicated that agricultural practices resulted in reduced growth rates and recruitment as well as increased adult mortality (Saumure and Bider 1998). Moreover, Daigle and Jutras (2005) reported a 50 percent decline in the number of wood turtles at an agricultural site over a seven-year period (see also Saumure et al. 2007). Significant annual mortality (13.4 percent) due to agricultural mowing has been reported in Massachusetts (Jones 2009b). Tingley et al. (2009) reported that mowing is the major source of human-induced mortality in wood turtles in Nova Scotia.

At a region-wide scale, much wood turtle habitat has been taken over by agricultural operations (Sanderson et al. 2002). In many places (e.g., the Shenandoah Valley of Virginia, eastern Ontario, southern Wisconsin) the turtles have apparently been extirpated (Brewster 1985, Kerr and Deguise 2004, Akre and Ernst 2006).

In addition, areas in Pennsylvania within the turtle's range (e.g., Indiana County) are severely impacted by acid mine drainage and atmospheric deposition of acids (Lovich and Lovich 1996).

Overutilization:

Experts in most states surveyed mentioned collecting as a major threat in their state (NatureServe 2011). In the past, collection of wood turtles by biological supply houses likely led to population declines and extirpations, particularly in Wisconsin (Harding and Bloomer 1979, Vogt 1981). Collection for food has been a threat in the eastern part of the range (Harding 1991), and is likely responsible to some extent for current population reductions. A turtle researcher in Wisconsin tracked a transmittered turtle to a dumpster and found remains of over 60 other wood turtles that had been killed for food by one individual (COSEWIC 2007).

Presently, overcollection for the pet trade is seriously impacting the wood turtle (Harding 1991), which is known as one of the most handsome of all North American turtles (Ernst and Lovich 2009). Burger and Garber (1995) stated that humans find wood turtles "irresistible" and generally remove them or at least displace them when they are found. The turtle's apparent intellect (Tinklepaugh 1932) and "striking appearance" (Carr 1952) have certainly boosted its popularity as a pet. Wood turtles are perhaps the most valuable legally traded native species of turtle in the country (Franke and Telecky 2001, Reed and Gibbons 2003).

Entire populations along some streams have been eliminated (NatureServe 2011). Collectors can easily remove an entire population along many miles of stream in only one or two seasons of collecting by timing collection to coincide with the turtles' emergence from hibernation (NatureServe 2011). The wood turtles tend to clump in numbers at certain spots during hibernation and are therefore "very vulnerable to collectors" (Farrell and Graham 1991). As a result of collection pressure, the distribution is discontinuous and gene flow has certainly been reduced in some areas (NatureServe 2011).

Most states and provinces in the range now have laws prohibiting mass collection and commercial use. Yet illegal collection is ongoing. For example, in March of 2009, 18 individuals were charged in New York for the illegal sale of reptiles and amphibians, including wood turtles (NYDEC 2009). Enforcement of laws protecting the turtles is difficult because it is not illegal to sell wood turtles in the rest of the United States, or to export them. Wood turtles commonly show up in pet stores on the west coast, and they are also shipped to Japan and Europe (NatureServe 2011). Sale prices in Europe were reported to exceed US $100 (NatureServe 2011). The selling price and apparent ease of collection will continue to put pressure on this species until sales are effectively regulated. Levell (2000) discussed commercial exploitation for the live animal trade.

As with most species of freshwater turtles, the wood turtle is also shot by human vandals (Harding 1991).

Disease and predation:

Nesting success for wood turtles generally is very low, with egg predators taking a heavy toll and apparently responsible for observed population declines (Brooks et al. 1992, Buech et al. 1997b, Hunter et al. 1999, Siart 1999, Harding 2002, Paradis et al. 2004, Bowen and Gillingham 2004, Tamplin 2005, Akre and C. Ernst 2006). High predation rates is another detrimental aspect of development and intense recreational use because egg predators such as skunks and raccoons commonly increase in abundance with surrounding development and degradation of natural habitat (van Dijk and Harding 2010, NatureServe 2011). One report conservatively estimated egg and hatchling mortality at 98 percent (Harding 1990). And it is believed that predation by raccoons in Michigan results in zero recruitment for the species (NatureServe 2011). Foscarini (1994) reported 80-83 percent of nests destroyed at her Ontario study site. Harding and Bloomer (1979) and Harding (1992, 2002) have reported a 70 to 100 percent predation rate, mostly from raccoons. Buech (1991, 1992) and Buech et al. (1993) reported a 75 to 100 percent predation rate on Minnesota wood turtle nests. Hunter et al. 1999 explained: “Nest predation by mammalian predators is a major factor in nest mortality in some localities and may approach 100% in some years . . . Combined effects of all mortality factors may be quite high, and a low percentage of Wood Turtle nests may hatch in any given year. . . . It is believed that predation is the main reason for low recruitment in Wood Turtle populations.”

Raccoons and other predators also increase adult mortality (Farrell and Graham 1991). In Quebec it was estimated that predators killed 40 percent of the nesting females in just a few years (COSEWIC 2007). Predator-related injuries have been widely observed for wood turtles and commonly include limb amputations, bobtails, and damaged shells (Saumure and Bider 1998, Ernst 2001a, Walde et al. 2003, Saumure et al. 2007, Krichbaum 2009, Carroll 2009, Greaves and Litzgus 2009). Brooks et al. (1992) found that 60 percent of adults and 28.6 percent of juveniles at a site in Ontario bore wounds from predatory attempts. At another Ontario site (where researchers referred to its “remoteness”) the frequency of limb amputations in the population was 14.5 percent (Greaves and Litzgus 2009). Limb loss is likely to significantly affect survivorship (Harding 1985). Harding (1985) provided further information on predation and injuries.

Inadequacy of existing regulatory mechanisms:

The Committee on the Status of Endangered Wildlife in Canada (COSEWIC 2007) lists the species as threatened, and in Quebec the species has been added to the list of vulnerable species (Gouvernement du Québec 2005). In addition, Ontario has developed a recovery strategy under its Endangered Species Act (Ontario Wood Turtle Recovery Team 2009). Implementation of the recovery strategy is not guaranteed, however, as it is subject to appropriations, priorities, and budgetary constraints of the participating jurisdictions and organizations.

The wood turtle is listed on CITES Appendix II, which means that export permits should only be issued if trade is not detrimental to the species. Unfortunately, it is likely that many wild caught wood turtles are sold as captive bred to avoid this requirement.

The wood turtle is found on some lands protected from development, such as national forests. Indeed, the wood turtle is designated as a Regional Forester Sensitive Species on the Chequamegon-Nicolet, Green Mountain, Huron-Manistee, Ottawa, Superior, and White Mountain national forests in the Eastern Region of the Forest Service (Region 9) (Bowen and Gillingham 2004). But sensitive species designations only require that impacts of land management activities be considered and do not require that harmful actions be avoided. In addition, habitat for the wood turtle is probably not suitable on a large portion of "protected" lands due to factors such as elevation and steepness of waterways (see, e.g., Scott 2001, NH WMNF paper, Jones 2009, Krichbaum 2009). Federal protection would ensure that actions do not jeopardize the species and protect essential habitats. Such protection is necessary. As Ernst (2001) explains: "[Wood turtle] habitats must be protected as refuges with no, or extremely little, interference from humans if breeding populations are to survive."

Most states and provinces within the range of the wood turtle list the turtles as endangered or threatened and provide protection from take (Garber et al. 1994, Klemens 2000, Castellano et al. 2009). One exception is Maine, where wood turtles have no protection (Maine Dept. of Inland Fisheries and Wildlife 2010). State protection appears to have "done little to curb collection of this species" (NatureServe 2011). Declines have not abated because illegal collection continues to be a threat and protected status at the state level does not require protection of wood turtle habitat. Federal protection would deter collection through increased penalties and additional enforcement.

Moreover, given the fact that most wood turtle populations are in a state of decline, there is at present no reason to conclude that current regulations are sufficient (Bowen and Gillingham 2004).

Other factors:

Climate change is another factor threatening the wood turtle. van Dijk and Harding (2010) suggest that global warming is likely to tip southern populations towards extinction (van Dijk and Harding 2010).

Wood turtles are found in disjunct, isolated populations (e.g. Tessier et al. 2005). Plus, many populations of wood turtles are already very small (Brooks et al. 1992, Ernst 2001a, Daigle and Jutras 2005, Akre and Ernst 2006, Sweeten 2008). As such, isolation must be considered a threat to the wood turtle (see Marchand and Litvaitis 2004a). Small, isolated populations are more susceptible to extirpations due to stochastic events, human impacts, and environmental factors (Soulé 1987, Begone et al. 1990). Lack of gene flow may cause loss of genetic variability due to random genetic drift (Wright 1931) and inbreeding depression may occur (Franklin 1980). Once these populations are extirpated, their isolation precludes genetic interchange and recolonization of habitat.

Pollution of the wood turtle's aquatic habitats is also a threat. Because of its need for clear, flowing water, the wood turtle is "pollution intolerant" and is affected by pesticide use and other runoff (Harding and Bloomer 1979, Burger and Garber 1995). Wood turtles showed declines in some areas in the 1950s and 1960s, probably in response to increasing insecticide use (Harding and Bloomer 1979, NatureServe 2011). A multitude of waterways within the wood turtle's range are polluted or significantly degraded in various ways (e.g., fecal coliform, agricultural and industrial chemicals, sedimentation, and acidic deposition) (see, e.g., Jones et al. 1997). For example, Virginia has numerous "impaired (category 5) waters" in the "Potomac River and Shenandoah River Basins," the watersheds where the turtle resides in the state (see Virginia DEQ 2006).

Finally, wood turtles are likely to face threats from litter such as plastics and fishing gear (Burger and Garber 1995, Wusterbarth 2000, Ernst 2001).

Acknowledgment:

The following individual contributed to this account:

[redacted]

References:

Akre, T. and C. Ernst. 2006. Final Report Population Dynamics, Habitat Use, and Home Range of the Wood Turtle, Glyptemys (=Clemmys) insculpta, in Virginia, submitted to the Virginia Department of Game and Inland Fisheries, contract ed0820ce_080.

Begone, M., J. L. Harper, and C. R. Townsend. 1990. Ecology: Individuals, populations, and communities. Blackwell Scientific Publications.

Bleakney, J. S. 1963. Notes on the distribution and life histories of turtles in Nova Scotia. Canadian Field Naturalist 77(2): 67-76.

Bowen, K.D. and J.C. Gillingham. 2004. R9 Species Conservation Assessment for Wood Turtle – *Glyptemys insculpta* (Le Conte, 1830). USDA Forest Service, Milwaukee, WI. 39 pp.

Brooks, R.J., C.M. Shilton, G.P. Brown, and N.W.S. Quinn. 1992. Body size, age distribution, and reproduction in a northern population of wood turtles (*Clemmys insculpta*). Can. J. Zool. 69: 1,314-20.

Buech, R.R. and M.D. Nelson. 1997. Conservation of Wood turtles in Minnesota. Pages 15-21 in J.J. Moriarty and D. Jones (editors). Minnesota's Amphibians and Reptiles: Their Conservation and Status. Serpent's Tale Natural History Book Distributors, Lanesboro, Minnesota.

Buech, R.R., M.D. Nelson, L.G. Hanson, and B. Brecke. 1997. Wood turtle habitat research, p. 475-76. In J. Van Abbema (ed.), Proceedings: Conservation, Restoration, and Management of Tortoises and Turtles – An International Conference. New York Turtle Tort. Soc., New York.

Burger, J. and S.D. Garber. 1995. Risk assessment, life history strategies, and turtles: could declines be prevented or predicted? J. Toxciol. Environ. Health 46: 483-500.

Burke, V.J., J.E. Lovich, and J.W. Gibbons. 2000. Conservation of freshwater turtles. Pp. 156-179 in M.W. Klemens (ed.), Turtle Conservation. Smithsonian Institution Press, Washington, D.C.

Carr, A.F. 1952. Handbook of Turtles. Cornell University Press, Ithaca, New York.

Carroll, D.M. 2009. Following the Water: A Hydromancer's Notebook. Houghton Mifflin Harcourt, Boston, MA. 186 pp.

Castellano, C.M., J.L. Behler, and G.R. Ultsch. 2008. Terrestrial movements of hatchling Wood Turtles (*Glyptemys insculpta*) in agricultural fields in New Jersey. Chelonian Conservation and Biology 7(1): 113–118.

Castellano, C.M., J.L. Behler, and G. Amato. 2009. Genetic diversity and population genetic structure of the Wood Turtle (Glyptemys insculpta) at Delaware Water Gap National Recreation Area, USA. Conservation Genetics 10(6): 1783.

Committee on the Status of Endangered Wildlife in Canada ("COSEWIC"). 2007. COSEWIC Status Assessment, Turtle, Wood, Glyptemys insculpta, *available at* http://www.cosewic.gc.ca/eng/sct1/searchdetail_e.cfm?id=286&StartRow=1&boxStatus=All&boxTaxonomic=All&location=All&change=All&board=All&commonName=wood%20turtle&scienceName=&returnFlag=0&Page=1 (last visited Oct. 3, 2011).

Compton, B.W. 1999. Ecology and Conservation of the Wood turtle (*Clemmys insculpta*) in Maine. M.S. Thesis, University of Maine, Orono, Maine. 91 pp.

Conant, R. and J. T. Collins. 1991. A field guide to reptiles and amphibians: eastern and central North America. Third edition. Houghton Mifflin Co., Boston, Massachusetts. 450 pp.

Congdon, J.D., A.E. Dunham, and R.C. van Loben Sels. 1993. Delayed sexual maturity and demographics of Blanding's turtles (*Emydoidea blandingii*): implications for conservation and management of long-lived organisms. Conservation Biology 7: 826-833.

Congdon, J.D., A.E. Dunham, and R.C. van Loben Sels. 1994. Demographics of Common Snapping turtles: implications for conservation and management of long-lived organisms. American Zoologist 34: 397-408.

Dahl, L. 1996. Wood Turtle (*Clemmys insculpta*) Survey [redaction] 1996 Report [redaction] Minnesota, unpublished report dated August 22, 1996 submitted to Minnesota Department of Natural Resources Nongame Program. 9 pp.

Daigle, C., and J. Jutras. 2005. Quantitative evidence of decline in a southern Quebec wood turtle (*Glyptemys insculpta*) population. Journal of Herpetology 39: 130-132.

Desroches, J.F., and I. Picard. 2005. Mortalite des tortues sur les routes de L'Outaouais. Nat. Can. 129: 35-41.

Erb, L. and M. Jones. 2011. Can Turtle Mortality be Reduced in Managed Fields? Northeastern Naturalist 18(4): 489.

Ernst, C. H. 2001. An overview of the North American turtle genus *Clemmys* Ritgen, 1828. Chelonian Conservation and Biology 4: 211-216.

Ernst, C.H. and J.E. Lovich. 2009. Turtles of the United States and Canada. The John Hopkins University Press. 827 pages.

Ernst, C. H. and J. F. McBreen. 1991. Wood turtle, *Clemmys insculpta*. pp455-457. In: J. N. McDonald and T. Skwara (eds.) Virginia's Endangered Species. Blacksburg, VA: McDonald and Woodward.

Farrell, R. F. and T. E. Graham. 1991. Ecological notes on the turtle *Clemmys insculpta* in northwestern New Jersey. J. Herp. 25(1): 1-9.

Franke, J., and T.M. Telecky. 2001. Reptiles as pets: An examination of the trade in live reptiles in the United States. The Humane Society of the United States. Washington, D.C. 146 pp.

Franklin, I. R. 1980. Evolutionary changes in small populations. In Conservation Biology: An evolutionary-ecological perspective (ed. a. B. A. W. M. E. Soule). pp. 135-149.

Garber, S. D., and J. Burger. 1995. A 20-yr study documenting the relationship between turtle decline and human recreation. Ecological Applications 5: 1151-1162.

Garber S.D., J. Burger, J.H. Harding, C. Ernst, S.E. Tuttle, and J. Davis. 1994. Petition for a rule to list the North American Wood Turtle (*Clemmys insculpta*) as Threatened under the Endangered Species Act 16 U.S.C. Sec. 1531 et seq. (1973) as amended.

Gibbs, J.P. and W.G. Shriver. 2002. Estimating the effects of road mortality on turtle populations. Conservation Biology 16: 1647-1652.

Gilhen, J. and B. Grantmyre. 1973. The wood turtle, Clemmys insculpta (LeConte): An addition to the herpetofauna of Cape Breton Island, Nova Scotia. Can. Field-Nat. 87: 308-310.

Green, N. B., and T. K. Pauley. 1987. Amphibians and reptiles in West Virginia. University of Pittsburg Press, Pittsburg, Pennsylvania. xi + 241 pp.

Gouvernement du Québec. 2005. Communiqué N° 00–233: Le ministre des Ressources naturelles et de la Faune désigne six nouvelles espèces et un habitat. Cabinet du ministre des Ressources naturelles et de la Faune, Québec, 16 March 2005.

Greaves, W.F. and J.D. Litzgus. 2009. Variation in life-history characteristics among populations of North American wood turtles: a view from the north. Journal of Zoology 2009: 1-12.

Harding, J. H. 1985. Life history notes: *Clemmys insculpta*: predation - mutilation. Herp. Rev. 16(1): 30.

Harding, J.H. 1991. A twenty year wood turtle study in Michigan: Implications for conservation, p. 31-35. In. K.R. Beaman, F. Caporaso, S. McKeown, and M.D. Graff (eds.), Proceedings of the First International Symposium on Turtles and Tortoises: Conservation and Captive Husbandry. Chapman Univ., Orange, California.

Harding, J. H. 1997. Amphibians and reptiles of the Great Lakes region. University of Michigan Press, Ann Arbor. xvi + 378 pp.

Harding, J. 2002. *Clemmys insculpta* North American wood turtle. University of Michigan Museum of Zoology Animal Diversity Web.

Harding, J.H. and T.J. Bloomer. 1979. The Wood turtle, *Clemmys insculpta* . . . a natural history. Bulletin of the New York Herpetological Society 15: 9-26.

Hunter, M. L., J. Albright, and J. Arbuckle, editors. 1992. The Amphibians and Reptiles of Maine. Maine Agricultural Experiment Station Bulletin 838. 188 pp.

Jones, M.T. 2009. Spatial ecology, population structure, and conservation of the Wood Turtle (Glyptemys inseulpta) in central New England. Ph.D. Dissertation. University of Massachusetts, Amherst, MA. 233 pp,

Jones, K.B., K.H. Riitters, J.D. Wickham, R.D. Tankersley, R.V. O'Neill, D.J. Chaloud, E.R. Smith, and A.C. Neale. 1997. An Ecological Assessment of the United States Mid-Atlantic

Region: A Landscape Atlas. USEPA/600/R-97/130. U.S. Environmental Protection Agency. Washington, D.C. 104 pp.

Kerr, J.T. and I. Deguise. 2004. Habitat loss and the limits to endangered species recovery. Ecology Letters 7: 1163–1169.

Krichbaum, S. 2009. "Wood Turtle Conservation on the GWNF", unpublished report submitted to the U.S. Forest Service/George Washington National Forest, Virginia Division of Natural Heritage, and Virginia Department of Game and Inland Fisheries, dated January 9, 2009. 34 pp.

Klemens, M.W. 1993. Amphibians and Reptiles of Connecticut and Adjacent Regions. State Geological and Natural History Survey of Connecticut. Bulletin No. 112.

Klemens, M.W. 2000. Turtle conservation. Smithsonian Institution Press, Washington, DC

Langen, T.A., A. Machniak, E.K. Crowe, C. Mangan, D.F. Marker, N. Liddle, and B. Roden. 2007. Methodologies for surveying herpetofauna mortality on rural highways. Journal of Wildlife Management 71(4): 1361–1368.

Langen, T.A., K.M. Ogden, and L.L. Schwarting. 2009. Predicting hot spots of herpetofauna road mortality along highway networks. Journal of Wildlife Management 73(1): 104-114.

Levell, J. P. 2000. Commercial exploitation of Blanding's turtle, *Emydoidea blandingii*, and the wood turtle, *Clemmys insculpta*, for the live animal trade. Chelonian Conservation and Biology 3:665-674.

Litzgus, J.D., and J.R. Brooks. 1996. Status report on the wood turtle, *Clemmys insculpta*, in Canada. Committee on the Status of Endangered Wildlife in Canada, Ottawa, Ont.

Lovich, J.E. 1995. Turtles. Pp. 118-121 in E.T. LaRoe, G.S. Farris, C.E. Puckett, P.T. Doran, and M.J. Mac (eds.), Our Living Resources. U.S. Dept. of Interior, National Biological Survey, Washington, DC.

Maine Dept. of Inland Fisheries and Wildlife. 2010. Maine Endangered Species Program/State list, *available at* http://www.maine.gov/ifw/wildlife/species/endangered_species/state_list.htm (last visited October 3, 2011).

Marchand, M.N. and J.A. Litvaitis. 2004. Effects of habitat features and landscape composition on the population structure of a common aquatic turtle in a region undergoing rapid development. Conservation Biology 18(3): 758–767.

Mitchell, J.C., and Klemens, M.W. 2000. Primary and secondary effects of habitat alteration. In Turtle conservation. Edited by M.W. Klemens. Smithsonian Institution Press, Washington, D.C. pp. 5–32.

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application].

Version 7.1. NatureServe, Arlington, Virginia, *available at* http://www.natureserve.org/explorer.

New York State Department of Environmental Conservation. 2009. Black market animal trade busted: DEC's in-depth undercover investigation nets 18 arrests. Press release dated March 19, 2009. Yancey Roy, contact, 518-402-8000.

Oldfield, B. 1996. Report of 1995 Wood Turtle (*Clemmys insculpta*) Survey on the Lower Cannon River & Middle Fork of the Zumbro River, unpublished report dated April 15, 1996 submitted to Minnesota Department of Natural Resources Nongame Program. 12 pp.

Oldfield, B., and J. J. Moriarty. 1994. Amphibians & reptiles native to Minnesota. University of Minnesota Press, Minneapolis. xv + 237 pp.

Ontario Wood Turtle Recovery Team. 2009. Draft Recovery strategy for the Wood Turtle (*Glyptemys insculpta*) in Ontario. Ontario Recovery Strategy Series. Prepared for the Ontario Ministry of Natural Resources. vi + 23 pp.

Quinn, N. W. S., and D. P. Tate. 1991. Seasonal movements and habitat of wood turtles (*Clemmys insculpta*) in Algonquin Park, Canada. J. Herpetol. 25: 217-220.

Reed, R.N. and J.W. Gibbons. 2003. Conservation status of live U.S. non-marine turtles in domestic and international trade. Report to US Department of the Interior and US Fish and Wildlife Service. 92 pp.

Restore: The North Woods et al. 1994. "Petition for a Rule to List the North American Wood Turtle (*Clemmys insculpta*) as Threatened under the Endangered Species Act 16 U.S.C. Sec. 1531 et seq. (1973) as Amended". Submitted to Office of Endangered Species, U.S. Fish and Wildlife Service, U.S. Department of Interior. Dated December 27, 1994. 40 pp.

Sanderson, E.W., J. Malanding, M.A. Levy, K.H. Redford, A.V. Wannebo, and G. Woolmer. 2002. The human footprint and the last of the wild. BioScience 52: 891–904.

Saumure, R.A. 2004. Spatial ecology and conservation of the North American Wood Turtle (*Glyptemys insculpta*) in a fragmented agri-forested landscape, Ph.D, Dissertation, McGill University, Montréal, QC, Canada, 134 pp.

Saumure, R.A., and J.R. Bider. 1998. Impact of agricultural development on a population of Wood Turtles (*Clemmys insculpta*) in Southern Quebec, Canada. Chelonian Conserv Biol 3: 37–45.

Saumure, R.A., T.B. Herman, and R.D. Titman. 2007. Effects of haying and agricultural practices on a declining species: The North American wood turtle, *Glyptemys insculpta*. Biol. Cons. 135: 581-591.

Scott, J.M., F.W. Davis, R.G. McGhie, R.G. Wright, C. Groves, and J. Estes. 2001. Nature reserves: Do they capture the full range of America's biological diversity? Ecological Applications 11(4): 999–1007.

Siart, L. 1999. An investigation of wood turtle (*Clemmys insculpta*) nest success in southwestern Massachusetts. M.S. Thesis, Antioch University, Yellow Springs, OH.

Soulé, M. 1987. Viable populations for conservation. Cambridge University Press, Great Britain. 189 pp.

Spradling, T.A., J.W. Tamplin, S. Dow, and K. Meyer. 2010. Conservation genetics of a peripherally isolated population of the wood turtle (*Glyptemys insculpta*) in Iowa. Conservation Genetics 11(5): 1667.

Steen, D.A. M.J. Aresco, S.G. Bielke, B.W. Compton, E.P. Condon, C.K. Dodd Jr., H. Forrester, J.W. Gibbons, J.L. Greene, G. Johnson, T.A. Langen, M.J. Oldham, D.N. Oxier, R.A. Saumure, F.W. Shueler, J.M. Sleeman, L.L. Smith, J.K. Tucker, and J.P. Gibbs. 2006. Relative vulnerability of female turtles to road mortality. Animal Conservation 9: 269-273.

Sweeten, S.E. 2008. "Home range, hibernacula fidelity, and best management practices for Wood Turtles (*Glyptemys insculpta*) in Virginia". Masters thesis, James Madison University, Harrisonburg, VA. 47 pp.

Tamplin, J. 2005. Microhabitat selection of free-ranging Wood Turtles (Glyptemys insculpta) in Butler County, Iowa. Unpublished report to Iowa Department of Natural Resources. 9 pp.

Tessier, N., S.R. Paquette, and F.J. Lapointe. 2005. Conservation genetics of the wood turtle (*Glyptemys insculpta*) in Quebec, Canada. Canadian Journal of Zoology 83(6): 765.

Thompson, F. G. 1953. Further evidence of the occurrence of the wood turtle, Clemmys insculpta (Le Conte) in northeastern Ohio. Herpetologica 9: 74.

Tingley, R., D.G. McCurdy, M.D. Pulsifer, and T.B. Herman, 2009. Spatio-temporal differences in the use of agricultural fields by male and female Wood Turtles (*Glyptemys insculpta*) inhabiting an agri-forest mosaic. Herpetological Conservation and Biology 4: 185-190.

Tinkelpaugh, O.L. 1932. Maze learning of a turtle. Journal of Comparative Psychology 13: 201-206.

van Dijk, P.P. and Harding, J. 2010. *Glyptemys insculpta*. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.1, *available at* www.iucnredlist.org (last visited Oct. 3, 2011).

Virginia Department of Environmental Quality. 2006. Final 2006 305(b)/303(d) Water Quality Assessment Integrated Report, Approved by EPA on October 16, 2006.

Virginia Department of Game and Inland Fisheries [VDGIF]. 2005. Virginia's comprehensive wildlife conservation strategy. Virginia Department of Game and Inland Fisheries, Richmond, Virginia. 717 pp. plus appendices.

Vogt, R.C. 1981. Natural History of Amphibians and Reptiles in Wisconsin. Milwaukee Public Museum, Milwaukee, Wisconsin. 205 pp.

Walde, A.D., J.R. Bider, C. Daigle, D. Masse, J.-C. Bourgeios, J. Jutras, and R.D. Titman. 2003. Ecological aspects of a Wood Turtle, Glyptemys insculpta, population at the northern limit of its range in Quebec. The Canadian Field Naturalist 117: 377-388.

Wilson, T.P., J.C. Mitchell, and T.S. Akre. 2003. Status and conservation of the genus Clemmys in Virginia: Prospects for the future, p. 112. In C.W. Swarth, W.M. Roosenburg, and E. Kiviat (eds.), Conservation and ecology of turtles of the Mid-Atlantic Region: A symposium. Bibliomania, Salt Lake City, Utah.

Wright, S. 1931. Evolution in Mendelian populations. Genetics 16: 97-159.

Wusterbarth, T.L. 2000. Nest Predation and effects of human recreation on the Wood turtle (*Clemmys insculpta*) of a western Michigan river. M.S. Thesis, Central Michigan University, Mt. Pleasant, Michigan. 44 pp.

Scientific Name:

Clemmys guttata

Common Name:

Spotted Turtle

G Rank:

G5

IUCN Red List:

Endangered

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

The spotted turtle inhabits the Great Lakes region of Canada and the United States, occurring from the southern tip of Lake Michigan to the St. Lawrence valley, as well as the upper reaches of the Ohio River system. It also occurs in the Atlantic coastal lowlands and foothills from New Hampshire (possibly southern Maine), southwards to northern Florida (Iverson 1992, Meylan 2006, Ernst and Lovich 2009).

Local extirpations have caused the geographic range to contract and fragment. For example, the spotted turtle's historic range in Illinois likely included much of the Chicago metropolitan area (Cook Co.), but no individuals have been discovered in Cook County since the early 1950s (Dreslik et al. 1998). In Maine, the species has disappeared from historic range in southern Cumberland Co. (USFWS 2000).

Habitat:

The spotted turtle occupies a wide variety of shallow wetland habitats across its range and during the year (Joyal et al. 2001). Habitat requirements include clear, clean water; a soft substrate; and aquatic or emergent vegetation (Ernst and Lovich 2009, p. 214). In some parts of the range and during certain times of the year, it spends considerable time on land (Ward et al. 1976, Milam and Melvin 2001). They often bask along the water's edge, on brush piles in water, or on logs or vegetation clumps (NatureServe 2011).

Cold season hibernation occurs in the muddy bottoms of waterways or bogs in communal hibernacula. Hibernacula usually have water depths of 55 to 95 cm (22 to 37 in) with a slow but steady flow or drift of water through densely vegetated wetlands with a deep, soft, mucky substrate (NatureServe 2011). The bogs and marshes that it inhabits are fragmented and disappearing (COSEWIC 2009).

Biology:

The spotted turtle is a typical K-selected species with population dynamics that emphasize the long-term reproductive contributions of adult animals over time. It occurs at low density, has an unusually low reproductive potential, and a long-lived life history (COSEWIC 2009). The age of sexual maturity is probably more closely related to reaching a specific size than age, although this length is usually obtained by 10 years of age (Ernst 1975). The maximum life span of adults is at least 26 years but may be as high as 50 (Tyning 1990).

Population Status:

The spotted turtle deserves federal protection because its range has contracted with most remaining populations declining and suffering from isolation. The species faces ongoing threats such as habitat destruction, introduction of invasive plant species, collection for the pet trade, and mortality from vehicular encounters (Ernst and Lovich 2009, p. 221; van Dijk 2010).

The spotted turtle generally occurs in small localized populations (van Dijk 2010). A survey of populations presented by Milam and Melvin (2001) and Litzgus and Mousseau (2004) shows that the estimated population sizes reported are usually small.

Although some populations are in protected areas, many likely have a low probability of persistence, especially because small numbers and isolation reduce population viability. In Canada, the low frequency of juveniles in most studied populations suggests these populations are composed largely of remnant, aged cohorts with low reproductive success (COSEWIC 2009). Serious declines easily go unnoticed because adult turtles can live many decades after recruitment problems start, thereby masking impending local extinctions with their presence (Klemens 2000).

The size of the U.S. population is declining throughout much of its range due to threats that include habitat destruction and overcollecting (Ashley and Robinson 1996, Buhlmann and Gibbons 1997, Dresklik et al. 1998, Brodman et al. 2002, Ernst and Lovich 2009, van Dijk 2010, NatureServe 2011). The species is likely to have suffered more than 50 percent overall reduction in population size, with much of this loss irreversible given habitat loss (van Dijk 2010). Recolonization of any new sites is slow and constrained by subsidized predators and possibly climate change (van Dijk 2010).

Many populations have been documented as in decline through loss of adults or lack of recruitment (Meylan 2006, Ernst and Lovich 2009). Turtle populations in areas with heavy development likely have suffered the greatest declines in numbers. Populations in northeastern Illinois have declined such that, at present, there are relatively few spotted turtles, and the numbers are also dropping in other Midwestern states and the Mid-Atlantic region (Wilson 2003). In Connecticut, spotted turtles are considered to be declining in the Quinnipiac River watershed (USFWS 2000). Historically, the spotted turtle was considered the most abundant turtle in Massachusetts, but populations have declined substantially in the past century (Milam and Melvin 1997). Lovich (1989) documented the decline of spotted turtles in Cedar Bog, Champaign County, Ohio. He concluded that "the spotted turtle population at Cedar Bog has declined dramatically during this century to what may be a critical level" (Lovich 1989). The species declined in northwestern Indiana between the 1930s and 1990s (Brodman et al. 2002). In New York, the spotted turtle was considered to be perhaps the most common turtle in the New York City area at the turn of the century, but today occurs in only a few isolated populations in protected areas (USFWS 2000).

THREATS

Habitat alteration and destruction:

Habitat alteration and fragmentation is a major threat to the spotted turtle (NatureServe 2011). The species is reasonably specialized in its habitat requirements and is not a good disperser. As a result, habitat destruction and fragmentation leads to disappearance of populations (van Dijk 2010).

Increased development pressures in wetlands since European settlement have contributed to the national trend of decreasing populations. It has been documented that increasing human populations and associated development in the last two decades have reduced the quantity and quality of the spotted turtle habitat in southern Maine (McCollough 1991) and southeastern New Hampshire, as well as in many other parts of its range (NatureServe 2011). Habitat succession is a challenge even in areas unaffected by development (van Dijk 2010).

Ohio provides one example of the devastating spotted turtle habitat loss. Lewis et al. (2004) visited 48 of 50 previously identified Ohio spotted turtle habitats, of which 8 had been developed and were no longer habitable. Of the remaining sites, 57 percent had significant invasive species, 64 percent were regionally fragmented, and 51 percent showed signs of intrasite fragmentation. Only five percent (two sites) showed no site-specific threats. Thus, most Ohio habitats were marginal for spotted turtle populations.

Additionally, the complex movement ecology and habitat requirements of spotted turtles make their populations especially vulnerable to road mortality: over the course of a year, they typically visit multiple wetlands to forage, mate, thermoregulate, and overwinter (Joyal et al. 2001, Grgurovic and Sievert 2005), requiring frequent overland migrations and road crossings. In addition, warm-season draw-downs of wetlands for game management can initiate emigrations of turtles that result in significant road kills (NatureServe 2011).

Overutilization:

Legal and illegal commercial exploitation (for both domestic use and export) and incidental collecting have impacted and continue to impact spotted turtle populations in many parts of the species' range (Smith et al. 1973, Minton et al. 1982, Hunter et al. 1992, COSEWIC 2009, NatureServe 2011).

Approximately 1,500 live spotted turtle individuals are exported from the United States per year (Weissgold 2010). The number of spotted turtles exported has been steadily increasing since 1995 (Weissgold 2010). This trend likely reflects increasing demand for the pet trade. Spotted turtles have sold for $219 each online (www.TurtleSale.com). The number of wild caught spotted turtles reportedly exported from the United States is much less. Trade data from 2006-2010 show that 176 wild caught spotted turtles were exported from the U.S (data on file with C. Adkins Giese). However, it is possible that many wild caught individuals are falsely reported as captive bred.

Although protected from harvest across most of its range in the northeastern United States, illegal trade has been documented. For example, in June 1998, state and federal agents raided a house in Bedford County, Pennsylvania and confiscated more than 60 illegally-held turtles, including 28 spotted turtles (Blankenship 1999). In 2009, an undercover investigation by the New York Department of Environmental Conservation called "Operation Shellshock" led to the arrest of two dealers illegally selling spotted turtles (Livingston County News 2009). In addition, overcollection has been suggested as a reason for spotted turtle population declines in Indiana and Ohio since the 1970s and 1980s (Smith et al. 1973, Minton et al. 1982).

Herpetologists report losses of known spotted turtle populations from overcollection. Carl Ernst reports that a population in Lancaster County, Pennsylvania had 300-400 individuals in 1980, but none were found at the site in 1999 (USFWS 2000). Two other, similarly-sized populations in northern Virginia have lacked a significant presence of spotted turtles since the 1980s (USFWS 2000). James Harding, a herpetologist with the Michigan State University Museum, has strong circumstantial evidence that collectors wiped out his study population of spotted turtles in south-central Michigan in the early 1970s (USFWS 2000). Alvin Braswell of the North Carolina State Museum reports that spotted turtles were difficult to locate in Hyde and Tyrrell counties, North Carolina, after a collector removed more than 1,100 from the wild in 1993-94 (USFWS 2000). Herpetologists have even encountered turtle poachers on study sites (Wilson 1999).

Disease and predation:

Subsidized predators (i.e., unnaturally large populations of predators subsidized by easily available resources near human settlements) probably represent a further impact on eggs and juveniles, and likely reduce recruitment into existing populations (van Dijk 2010). This threat increases as the habitat becomes more and more fragmented by urbanization (NatureServe 2011).

Inadequacy of existing regulatory mechanisms:

The spotted turtle was recommended for inclusion in CITES Appendix II by the Conservation, Status & Monitoring Working Group that the FWS convened during the September 2010 conference entitled "Conservation and Trade Management of Freshwater and Terrestrial Turtles in the United States" (USFWS 2010). The Center for Biological Diversity has also recommended listing on Appendix II. But currently, the species receives no protection under CITES.

In Canada, the Committee on the Status of Endangered Wildlife in Canada ("COSEWIC") designated the species as "Special Concern" in April 1991 but re-examined the status and designated it as "endangered" in May 2004. The designation was prompted in part by the "clear threat" from the pet trade (COSEWIC 2009). Only a fraction of the species' range occurs in Canada, and federal protection is needed in United States to ensure viability.

The spotted turtle is listed as endangered, threatened, or a species of special concern at the State/provincial level throughout its range (USFWS 1999). The species is protected as "threatened" or "endangered" in Maine, Vermont, New Hampshire, Illinois, Indiana, South Carolina, Michigan under the respective state endangered species laws (New Hampshire Fish and Game 2008, Michigan DNR 2009, SC DNR 2010, Maine Dept. of Inland Fisheries and Wildlife 2010, Vermont Fish and Wildlife Dept. 2011, Illinois Endangered Species Protection Board 2011, Indiana DNR 2011). This turtle is a species of "special concern" in New York and Georgia (GA DNR 2010, New York Dept. of Environmental Conservation 2011). In Rhode Island, state regulations prohibit taking spotted turtles from the wild or possession of one without a permit issued by the state wildlife agency (Rhode Island Dept. of Environmental Management 2006). The spotted turtle continues to suffer from population declines despite the regulatory mechanisms in these states, which demonstrates that they are inadequate.

To be sure, the species is without special status in Massachusetts, Connecticut, New Jersey, Delaware, Pennsylvania, Maryland, Ohio (New Jersey Division of Fish and Wildlife 2004, Massachusetts Dept. of Fish and Game 2008, Connecticut Dept. of Energy and Environmental Protection 2010, Maryland Dept. of Natural Resources 2010, Ohio DNR 2010, Delaware Division of Fish and Wildlife 2011, Pennsylvania Natural Heritage Program 2011). Federal protection would ensure that the species is protected from take across its range and likely deter illegal collection.

The spotted turtle occurs in a number of protected areas; however, because of vegetation dynamics, pollution and potential collection impacts, such protected populations are not necessarily secure in the long-term (van Dijk 2010). For example, the spotted turtle was extirpated from a national park in Canada despite habitat protection because of heavy predation on turtle nests, road mortality, habitat succession, and possibly chemical contamination (Browne and Hecnar 2007).

Moreover, while laws protecting wetlands provide some benefit to spotted turtles, they are insufficient to ensure viability. For example, Milam and Melvin (2001) found that 24 of 26 turtles nested or estivated well outside the 30 and 60 m-wide upland buffers protected under Massachusetts' Wetlands Protection Act. The authors conclude that "protection of complexes of seasonal pools and permanent wetlands bordered by substantially larger areas of upland habitat will be necessary if viable populations of spotted turtles are to be protected."

Other factors:

Global warming is another threat to the spotted turtle. The spotted turtle has temperature-dependent sex determination. Should its nesting environment become hotter in the future, the sex ratio is likely to be skewed toward primarily female clutches (the normal sex ratio is 1:1). Also, the spotted turtle is a cold-adapted species (Ernst 1976, Ernst 1982). Warming will adversely affect its behavior and possibly dry up many of the shallow wetlands where it occurs.

Many remaining spotted turtles populations are isolated in habitat fragments. Parker and Whiteman (1993) found that spotted turtles drawn from a small isolated wetland were significantly less diverse at these loci that those from a large wetland complex. To be sure, small, isolated populations like that of the spotted turtle are more susceptible to extirpations due to stochastic events, human impacts, and environmental factors (Soulé 1987, Begone et al. 1990). Lack of gene flow may cause loss of genetic variability due to random genetic drift (Wright 1931) and inbreeding depression may occur (Franklin 1980). Once these populations are extirpated, their isolation precludes genetic interchange and recolonization of habitat.

In addition, invasive plant species affecting wetland vegetation structure are a contributing threat factor (van Dijk 2010).

References:

Ashley, E.P., and J.T. Robinson. 1996. Road mortality of amphibians, reptiles and other wildlife on the Long Point Causeway, Ontario. Can. Field-Nat. 110: 403-12.

Begone, M., J. L. Harper, and C. R. Townsend. 1990. Ecology: Individuals, populations, and communities. Blackwell Scientific Publications.

Blankenship, K. 1999. Poaching and the illegal sale of reptiles and amphibians. Pennsylvania Angler & Boater.

Brodman, R., S. Cortwright, and A. Resetar. 2002. Historical changes of reptiles and amphibians of northwest Indiana fish and wildlife properties. American Midland Naturalist 147: 135-144.

Browne, C.L., and S.J. Hecnar. 2007. Species loss and shifting population structure of freshwater turtles despite habitat protection. Biological Conservation 138(3/4): 421-30.

Buhlmann, K. A., and J. W. Gibbons. 1997. Imperiled aquatic reptiles of the southeastern United States: historical review and current conservation status. Pages 201-232 in G. Benz and D. E.

Collins, editor. Conference on aquatic fauna in peril: the southeastern perspective. Southeast Aquatic Research Institute, Lenz Design & Communications, Decatur, GA. 554 pp.

Committee on the Status of Endangered Wildlife in Canada ("COSEWIC"). 2009. Wildlife Species Search, *available at* http://www.cosewic.gc.ca/eng/sct1/index_e.cfm (last visited Oct. 3, 2011).

Connecticutt Dept. of Energy and Environmental Protection. 2010. Endangered, Threatened & Special Concern Reptiles, *available at* http://www.ct.gov/dep/cwp/view.asp?a=2702&q=323484&depNav_GID=1628 (July 5, 2011).

Delaware Division of Fish and Wildlife. 2011. Delaware's Endangered Species, *available at* http://www.dnrec.delaware.gov/fw/NHESP/information/Pages/Endangered.aspx (last visted July 5, 2011).

Dreslik, M. J., E.O. Moll, C.A. Phillips, and T.P. Wilson. 1998. The endangered and threatened turtles of Illinois. Illinois Audubon, Winter 1997-98: 10-15.

Enneson, J. J., and J.D. Litzgus, J. D. 2009. Stochastic and spatially explicit population viability analyses for an endangered freshwater turtle, Clemmys guttata. Canadian Journal of Zoology 87(12): 1241.

Ernst, C. H. 1975. Growth of the spotted turtle, *Clemmys guttata*. Journal of Herpetology 9(3): 313-8.

Ernst, C. H. 1976. Ecology of the spotted turtle, Clemmys guttata, (Reptilia, Testudines, Testudinidae), in southeastern Pennsylvania. J. Herpetology 10: 25-33.

Ernst, C. H. 1982. Environmental temperatures and activities in wild spotted turtles, *Clemmys guttata*. Journal of Herpetology 16(2): 112-20.

Ernst, C.H. and J.E. Lovich. 2009. Turtles of the United States and Canada. The John Hopkins University Press. 827 pages.

Franklin, I. R. 1980. Evolutionary changes in small populations. In Conservation Biology: An evolutionary-ecological perspective (ed. a. B. A. W. M. E. Soule). pp. 135-149.

Georgia Department of Natural Resources (GA DNR). 2008. Special Concern Animal Species in Georgia, *available at* http://georgiawildlife.dnr.state.ga.us/content/specialconcernanimals.asp (last visited June 27, 2011).

Hunter, M. L., J. Albright, and J. Arbuckle, editors. 1992. The Amphibians and Reptiles of Maine. Maine Agricultural Experiment Station Bulletin 838. 188 pp.

Illinois Endangered Species Protection Board. 2011. Checklist of endangered and threatedn animals and plants of Illinois, *available at* http://www.dnr.state.il.us/espb/ (last visited July 5, 2011).

Indiana Dept. of Natural Resources. 2011. Indiana's Species of Greatest Conservation Need, *available at* http://www.in.gov/dnr/fishwild/2356.htm (last visited July 5, 2011).

Iverson, J. B. 1992. A revised checklist with distribution maps of the turtles of the world. Privately printed. Earlham College, Richmond, Indiana

Joyal, L. A., M. McCollough, and M. L. Hunter, Jr. 2001. Landscape ecology approaches to wetland species conservation: a case study of two turtle species in southern Maine. Conservation Biology 15: 1755-1762.

Klemens, M.W. 2000. Turtle Conservation. Smithsonian Institution Press, Washington (2000)

Lewis, T.L., J.M. Ullmer, and J. Mazza. 2004. Threats to Spotted Turtle (*Clemmys guttata*) Habitat in Ohio. Ohio Journal of Science 104(3): 65-72.

Litzgus, J. D., and T. A. Mousseua. 2003. Multiple clutching in southern spotted turtles, *Clemmys guttata*. Journal of Herpetology 37: 17-23.

McCollough, M. 1991. Spotted turtle assessment. Maine Endangered and Nongame Wildlife Project, Resource Assessment Section, Wildlife Division, Department of Inland Fish and Wildlife.

Milam, Joan C., and Scott M. Melvin. 2001. Density, habitat use, movements, and conservation of spotted turtles (*Clemmys guttata*) in Massachusetts. Journal of Herpetology 35(3): 418-28.

Lovich, J. E. 1989. The spotted turtles of Cedar Bog: Historical analysis of a declining population. Pages 23-28 in R. C. Glotzhober, A. Kochman, and W. T. Schultz (eds.) Proceedings of Cedar Bog Symposium II.

Maine Dept. of Inland Fisheries and Wildlife. 2010. Maine Endangered Species Program/State list, *available at* http://www.maine.gov/ifw/wildlife/species/endangered_species/state_list.htm (last visited July 5, 2011).

Maryland Dept. of Natural Resources. 2010. Rare, Threatened, and Endangered Animals of Maryland, *available at* http://www.dnr.state.md.us/wildlife/Plants_Wildlife/rte/rteplants.asp (last visited July 5, 2011).

Massachusetts Dept. of Fish and Game. 2008. Massachusetts List of Endangered, Threatened and Special Concern Species, *available at* http://www.mass.gov/dfwele/dfw/nhesp/species_info/mesa_list/mesa_list.htm (last visited July 5, 2011).

Meylan, P.A. 2006. *Clemmys guttata* – Spotted turtle. Chelon. Res. Monogr. 3: 226-34.

Michigan Dept. of Natural Resources. 2009. Endangered and threatened species, *available at* http://www.michigan.gov/dnr/0,1607,7-153-10370_12141_12168---,00.html (last visited July 5, 2011).

Milam, J. C., and S. M. Melvin. 2001. Density, habitat use, movements, and conservation of spotted turtles (*Clemmys guttata*) in Massachusetts. Journal of Herpetology 35: 418-427.

Minton, S. A., J. C. List, and M. J. Lodato. 1982. Recent records and status of amphibians and reptiles in Indiana. Proc. Indiana Acad. Sci. 92: 489-498.

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, *available at* http://www.natureserve.org/explorer (last visited Oct. 3, 2011).

New Hampshire Fish and Game. 2008. Endangered and threatened wildlife of NH, *available at* http://www.wildlife.state.nh.us/Wildlife/Nongame/endangered_list.htm (last visited July 5, 2011).

New Jersey Division of Fish and Wildlife. 2004. New Jersey's Endangered and Threatened Wildlife, *available at* http://www.state.nj.us/dep/fgw/tandespp.htm (last visited July 5, 2011).

New York Dept. of Environmental Conservation. 2011. List of Endangered, Threatened and Special Concern Fish & Wildlife Species of New York State, *available at* http://www.dec.ny.gov/animals/7494.html (last visited July 5, 2011).

Ohio Dept. of Natural Resources. 2010. Ohio's endangered species, *available at* http://www.dnr.state.oh.us/wildlife/home/resources/mgtplans/endangered/tabid/6005/default.aspx (last visited July 5, 2011).

Parker, Patricia G., and Howard H. Whiteman. 1993. Genetic diversity in fragmented populations of Clemmys guttata and Chrysemys picta marginata as shown by DNA fingerprinting. Copeia 1993(3): 841-7.

Pennsylvania Natural Heritage Program. 2011. Species of Special Concern Lists, *available at* http://www.naturalheritage.state.pa.us/Species.aspx (last visited July 5, 2011).

Rhode Island Dept. of Environmental Management. 2006. Rare native animals of Rhode Island, *available at* http://www.dem.ri.gov/programs/bpoladm/plandev/heritage/ (last visited July 5, 2011).

Smith, H. G., R. K. Burnard, E. E. Good, and J. M. Keener. 1973. Rare and endangered vertebrates in Ohio. Ohio J. Sci. 73: 257-271.

Soulé, M. 1987. Viable populations for conservation. Cambridge University Press, Great Britain. 189 pp.

South Carolina Department of Natural Resources (SC DNR). 2010. SC Rare, Threatened & Endangered Species Inventory, *available at* http://www.dnr.sc.gov/species/pdf/SC_state_wide.pdf (last visited June 27, 2011).

Tyning, T. F. 1990. A guide to amphibians and reptiles. Strokes Nature Guides. Little, Brown and Company, Boston, Massachusetts.

U.S. Fish and Wildlife Service. 1999. Conference of the Parties to the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES); Eleventh Regular Meeting; Proposed Resolutions and Agenda Items Being Considered; Species Being Considered for Amendments to the CITES Appendices; Public Meeting; Observer Information. 64 Fed. Reg. 36902 (July 8, 1999).

U.S. Fish and Wildlife Service. 2000. Proposal: Inclusion of *Clemmys guttata* in Appendix II, *available at* www.cites.org/eng/cop/11/prop/37.pdf (last visited June 5, 2012).

U.S. Fish and Wildlife Service. 2010. Conservation, Status & Monitoring Working Group Findings and Recommendations, *available at* http://www.fws.gov/international/DMA_DSA/CITES/Appendix_III/turtles_ws.html (last visited Feb. 8, 2011).

van Dijk, P.P. 2010. *Clemmys guttata*. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.1, *available at* www.iucnredlist.org (last visited Oct. 3, 2011).

Vermont Fish and Wildlife Dept. 2011. Endangered and threatened animals of Vermont, *available at* http://www.vtfishandwildlife.com/wildlife_nongame.cfm (last visited July 5, 2011).

Ward, F.P., C.J. Hohmann, J.F. Ulrich, and S.E. Hill. 1976. Seasonal microhabitat selections of spotted turtles (*Clemmys guttata*) in Maryland elucidated by radioisotope tracking. Herpetologica 32: 60-64.

Weissgold, B. 2010. U.S. Turtle Exports and Federal Trade Regulations: A Snapshot in U.S. Fish and Wildlife Service, Conservation and Trade Management of Freshwater and Terrestrial Turtles in the United States: Workshop Presentation Abstracts, *available at* http://www.fws.gov/international/DMA_DSA/CITES/Appendix_III/turtles_ws.html (last visited Feb. 8, 2011).

Wilson, T. P. 1999. SSAR Conservation Forum – Forum on the biology and conservation of North American turtles of the genus Clemmys: A review. Held at Pennsylvania State University, June 29-30, 1999. Unpublished manuscript. 17 pp. + tables.

Wilson, T.P. 2003. Habitat change detection in an isolated population of Spotted turtle in northern Virginia: Concepts and applications. *Conservation and Ecology of Turtles of the Mid-Atlantic region (1999).* Paxtuxent Wildlife Refuge, Laurel, Maryland: Bibliomania 43–52.

Wright, S. 1931. Evolution in Mendelian populations. Genetics 16: 97-159.

Scientific Name:

Pseudemys gorzugi

Common Name:

Rio Grande Cooter or Western River Cooter

G Rank:

G3

IUCN Red List:

Near Threatened

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

With collection pressure and degradation and loss of the Rio Grande cooter's habitats, extirpations throughout its range have occurred. It inhabits the Pecos-lower Rio Grande basin from New Mexico through Texas, as well as Coahuila, Nuevo Leon and Tamaulipas, Mexico (Iverson 1992, Seidel 1994). It is not recorded from the Rio Grande at or above the Big Bend region, nor from the San Fernando (van Dijk 2011). Bailey et al. (2008) found the Rio Grande cooter at low density and with a significant hiatus in the middle reaches of the Pecos River from the New Mexico border southward to the confluence with Independence Creek in Terrell County, but they were unable to locate individuals in the main channel of the Rio Grande south of Roma, Texas. It is apparently absent from the Texas side of the lower Rio Grande (Ward 1984). See Ernst and Lovich (2009) for a map of the known distribution.

Habitat:

Habitat includes rivers and their more permanent tributary streams (Garrett and Barker 1987), particularly larger, deeper stream pools with relatively clear water and sandy or rocky bottoms (Degenhardt and Christiansen 1990, Degenhardt et al. 1996). Individuals often bask on logs, overhanging vegetation, or muddy banks, or at the water's surface (Degenhardt et al. 1996). Eggs are buried in soil near water. The turtle occurs as high as 1,082 m altitude in New Mexico (Degenhardt and Christiansen, in Ernst et al. 1994).

Biology and Taxonomy:

One study showed that the gut contents were entirely vegetarian (Legler, as cited by Ernst and Lovich 2009), although the species has also been thought to be omnivorous (Degenhardt et al. 1996, Ernst and Lovich 2009). Limited information is available on reproduction: a clutch size of nine eggs has been reported; whether females produce multiple clutches remains unknown. Age and size at maturity remain unrecorded. Ernst and Lovich (2009) describe available information on the life history characteristics.

See Stephens and Wiens (2003) for genetic data supporting the distinctiveness of the Rio Grande cooter. This species has been treated in the literature also under the names *Pseudemys concinna texana* and *Pseudemys floridana concinna*; *Pseudemys texana* also has been treated under these names.

Population Status:

The Rio Grande cooter qualifies for endangered species status because it is rare and declining across its small range where it is experiencing the ongoing threats of habitat degradation and overcollection. Forstner et al. (2006) carried out range-wide surveys and found populations of the species concentrated in only a few stretches of the U.S. tributaries, and the species has minimal genetic structure across its range (Bailey et al. 2008). Declines apparently have occurred in New Mexico (Biota Information System of New Mexico, version 1/2004). Bailey et al. (2008) also reported an apparent absence of juveniles in the Texas populations, whereas their searches for juveniles in New Mexico were successful; they expressed their concerns for future population dynamics of Texas populations.

NatureServe (2011) explains that the species is uncommon in its small range. The IUCN Red List ranks the species as Near Threatened but explains that it is close to qualifying for Vulnerable.

THREATS

Habitat alteration and destruction:

The habitat of the Rio Grande cooter has been degraded by dams, water diversion, invasive plants, pollution, and other factors (NatureServe 2011). Specifically, riverine and riparian habitat along the lower Rio Grande has been degraded by dams, flood control practices, stream channelization, water diversion, and tamarisk invasion; water quality has declined, and streamflow has become increasingly intermittent (Blackstun et al. 1998, Bailey et al. 2008).

Bailey et al. (2008) explains that modification of flow rates drastically impacts the environment and has negative consequences for organisms inhabiting these drainage systems (Edwards and Conteras-Balderas 1991, Bunn and Arthington 2002). The Rio Grande is increasingly intermittent due to construction of dams, flood-control practices, channelization, water diversions, and introduction of exotic tamarisk. These practices significantly degrade water quality and result in little-to-no surface flow of water. Consequently, there is no longer enough water to support aquatic and riparian habitats while simultaneously meeting current levels of consumption by humans (United States Department of the Interior 1998, van Dijk 2011).

An apparent absence of the species over a 160-km stretch of the lower Pecos was attributed to river pollution by natural gas and oilfield runoff (Ward 1984). In New Mexico, apparent declines have been attributed to degradation of habitat through stream dewatering, loss of vegetation, and pollution (Schmitt et al. 1985, MacRae et al. 2001).

Overutilization:

This beautifully marked turtle is under serious threat from wild collection for the pet trade (Bailey et al. 2008, NatureServe 2011). The species is present in the international turtle trade (World Chelonian Trust 2012), and it is widely available for purchase on the Internet (*see, e.g.*, Turtle and Tortoise Inc. 2012). Disappearance from a locality in Texas likely was associated with commerical exploitation and exportation (Dixon 2000).

Some turtles are killed wantonly by anglers or gunners. Bailey et al. (2008) observed several that had been shot or otherwise killed at fishing camps in Texas.

Disease or predation:

Predation by raccoons and skunks – and facilitated by human presence – is likely problematic for this species as with other turtles.

Inadequacy of existing regulatory mechanisms:

Fitzgerald et al. (2004) provides an analysis of the exploitation of reptiles in the Chihuahuan Desert Ecoregion, including the Rio Grande cooter. The Rio Grande cooter is listed as an Endangered Species in New Mexico, which makes collection illegal (Fitzgerald et al. 2004).

The cooter has no protected status in Texas. However, Texas regulations (Appendix 3) limit individuals to possession of ten specimens of a species and 25 total individuals without a commercial collection or dealer permit. To sell or trade animals previously purchased requires a Resident or Nonresident Nongame Dealer's Permit (Fitzgerald et al. 2004). Take is extensive despite these regulations (Fitzgerald et al. 2004).

It is unknown whether the species occurs in protected areas within its range (van Dijk 2011). Surveys of the Canon de Santa Elena Flora and Fauna Protection Area have failed to record the species (MX Red List Workshop participants 2005). Even within protected areas, the species suffers from habitat degradation and illegal collection.

References:

Bailey, L.A., J. R. Dixon, R. Hudson, and M.R.J. Forstner. 2008. Minimal genetic structure in the Rio Grande cooter (*Pseudemys gorzugi*). Southwestern Naturalist 53: 406-411.

Blackstun, D., L. Woosley, and M. Flora. 1998. Water-resources issues in the Rio Grande--Rio Conchos to Amistad Reservoir subarea. U.S. Department of the Interior U.S.-Mexico Border Field Coordinating Committe.

Bundy, R. E. 1951. New locality records of reptiles in New Mexico. Copeia 1951: 314.

Bunn, S. E., and A. H. Arthington. 2002. Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. Environmental Management 30: 492-507.

Degenhardt, W. G., C. W. Painter, and A. H. Price. 1996. Amphibians and reptiles of New Mexico. University of New Mexico Press, Albuquerque. xix + 431 pp.

Degenhardt, W. G., and J. L. Christiansen. 1974. Distribution and habitats of turtles in New Mexico. Soutwestern Naturalist 19: 21-46.

Dixon, J. R. 2000. Amphibians and reptiles of Texas. Second edition. Texas A & M University Press, College Station. 421 pp.

Edwards, R. J., and S. Conteras-Balderas. 1991. Historical changes in the ichthyofauna of the lower Rio Grande (Rio Bravo del Norte), Texas and Mexico. Southwestern Naturalist 36: 201-212.

Ernst, C.H. and J.E. Lovich. 2009. Turtles of the United States and Canada. The John Hopkins University Press. 827 pages.

Fitzgerald, Lee A., Charles W. Painter, Adrian Reuter, and Craig Hoover. 2004. Collection, trade, and regulation of reptiles and amphibians of the Chihuahuan Desert Ecoregion. North America. Washington D.C.: World Wildlife Fund., *available at* http://herpetology.tamu.edu/Fitzgerald/Publications/Fitzgerald_etal_2004.pdf (last visited Feb. 10, 2012).

Forstner, M. R. J., J. R. Dixon, T. M. Guerra, J. L. McKnight, J. Stuart, and S. K. Davis. 2006. Current status and risks for the Big Bend slider (*Trachemys gaigeae*). Chihuahuan Desert Research Institute Symposium Proceedings.

Garrett, J. M., and D. G. Barker. 1987. A Field Guide to Reptiles and Amphibians of Texas. Texas Monthly Press, Austin, Texas. 225 pp.

Iverson, J. B. 1992. A revised checklist with distribution maps of the turtles of the world. Privately printed. Earlham College, Richmond, Indiana.

MacRae, R.K., J.D. Lusk, and W.R. Radke. 2001. Investigation of the role of chemical contaminants upon ecological sentinel species and their habitats at Bitter Lake National Wildlife Refuge, New Mexico, *available at* http://www.fws.gov/southwest/es/Documents/R2ES/Bitter_Lake_NWR_2000_Report-FINAL.pdf (last visited Feb. 10, 2012).

Seidel, M. E. 1994. Morphometric analysis and taxonomy of cooter and red-bellied turtles in the North American genus *Pseudemys* (Emydidae). Chelonian Conservation and Biology 1(2): 117-130.

Stephens, P. R., and J. J. Wiens. 2003. Ecological diversification and phylogeny of wmydid turtles. Biological Journal of the Linnean Society 79: 577-610.

Turtle and Tortoise Inc. 2012. Rio Grande Cooters (*Pseudemys gorzugi*) CB 2011, *available at* http://www.turtlesandtortoises.com/Rio-Grande-Cooters-Pseudemys-gorzugi-CB-2011.html (last visited Feb. 10, 2012).

United States Department of the Interior. 1998. Water-resources issues in the Rio Grande-Rio Conchos to Amistad Reservoir subarea. United States Department of the Interior, U. S.-Mexico Border Field Coordinating Committee, Washington, D. C. 3:1-8.

van Dijk, P.P. 2011. Pseudemys gorzugi. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.2. <www.iucnredlist.org>. Downloaded on 09 February 2012.

Ward, J. P. 1984. Relationships of chrysemyd turtles of North America (Testudines: Emydidae). Spec. Publ. Museum, Texas Tech Univ. (21): 1-50.

World Chelonian Trust. 2012. Declared Turtle Trade from the United States, *available at* http://www.chelonia.org/articles/us/USmarket_37.htm (last visited Feb. 10, 2012).

Scientific Name:

Macrochelys temminckii (previously *Macroclemys temminckii*)

Common Name:

Alligator Snapping Turtle

G Rank:

G3

IUCN Red List:

Vulnerable (assessed in 1996, now under review)

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

The range of the alligator snapping turtle is principally in the southeastern United States in river systems that drain into the Gulf of Mexico (Lovich 1993), including rivers from southern Georgia (Johnson 1989, Jensen and Birkhead 2003) and northwestern Florida (see Pritchard 1992), west to Louisiana (Boundy and Kennedy 2006) and eastern Texas (San Antonio River), and extending north to southeastern Kansas, southeastern Iowa, Illinois, and southern Indiana (Conant and Collins 1991).

Loss and degradation of habitat in many historically occupied sites and reductions in trapping success in remaining suitable habitat indicate that a large decline in area of occupancy and abundance has occurred across most of the range (Pritchard 1989, Moler 1996, Heck 1998, Reed et al. 2002, Jensen and Birkhead 2003, Riedle et al. 2005, Shipman and Riedle 2008). This species is likely extirpated from Indiana and Iowa, and the Kansas records show no evidence of a viable breeding population (NatureServe 2011). In addition, Bluett et al. (2011) conclude that the species is likely extirpated from Illinois after extensive surveys observed no alligator snapping turtles.

Commercial exploitation and other harvest for human consumption (and to a much lesser extent the pet trade) undoubtedly reduced populations of this species in much of its range (Pritchard 1992, Trauth et al. 1998, Reed et al. 2002, Riedle et al. 2005, Shipman and Riedle 2008). For example, targeted exploitation of the species has depleted the Flint River population (van Dijk and Rhodin 2010).

Habitat:

Adults are usually found in deeper water of large rivers and their major tributaries in floodplain swamp forest (van Dijk and Rhodin 2010) and are also found in lakes, canals, oxbows, swamps, ponds, and bayous associated with river systems (Ernst and Lovich 2009, p. 141), sometimes including swift upland streams (Phelps 2004). This turtle sometimes enters brackish waters near river mouths. Usually it occurs in water with a mud bottom and some aquatic vegetation but may use sand-bottomed creeks. Within streams, alligator snapping turtles may occur under or in logjams, beneath undercut banks, under rock shelters, or in deep holes (Jensen et al. 2008).

Much of the natural habitat of this species in northeast Arkansas and southeast Missouri has been drained and replaced by farm fields. A survey of alligator snapping turtle populations in New Madrid, Mississippi and Dunkin and Pemmiscott counties in Missouri revealed that in this four-county area, 90 percent of the habitat for the species is gone (NatureServe 2011).

In addition, the species is impacted by habitat degradation, primarily river engineering, which reduces silt load below dams, lowering the main channel and depriving connecting swamp channels (van Dijk and Rhodin 2010).

Biology:

The alligator snapping turtle is a very large turtle with a huge head, strongly hooked jaws, an extra row of scutes along each side of the shell (between the costals and marginals), three keels along the carapace, and a long tail (NatureServe 2011). The eyes are placed laterally on the head so that they cannot be seen from above, and a wormlike process on the tongue is used to lure prey within biting range (Ernst and Lovich 2009, p. 138).

Sexual maturity in the alligator snapping turtle is attained at about 11-13 years in both sexes (Ernst et al. 1994). Because of the species' slow life history, collection of breeding adults quickly becomes unsustainable (Reed et al. 2002). To maintain a stable population using biologically realistic values for fecundity, age at maturity, and survival of nests and juveniles, annual adult survivorship of females must be 98 percent. Reducing adult survivorship by two percent (to 96 percent) -- equivalent to annually removing only two adult females from a total population size of 200 turtles (assuming even sex ratios) -- will halve the population in only 50 years. Reed et al. (2002) found no evidence that sustainable exploitation of adults would be possible.

Additional information on the life history and habitat of the alligator snapping turtle have been summarized by Ewert et al. (2006), Pritchard (2006), and Ernst and Lovich (2009).

This species represents one of only two living genera (each with one living species) in the family. Previously, this turtle was included in the genus *Macroclemys*. However, Webb (1995) demonstrated that the generic name *Macrochelys* has priority over *Macroclemys*. Crother et al. (2000) and Crother (2008) agreed with this conclusion and treated this species as a member of *Macrochelys*.

Population Status:

The alligator snapping turtle qualifies for endangered species protection because it is rare or extirpated across most of its range with ongoing threats of habitat degradation and loss. The species is under review and may qualify for an "Endangered" rating from the IUCN Red List (van Dijk and Rhodin 2010). The species was a candidate for federal protection until the FWS eliminated the C2 category, but it currently receives no federal protection under the ESA.

Judging from past harvest rates in Louisiana and Georgia (Johnson 1989), some populations historically must have been very large. One individual trapper legally harvested 4,000-5,000 adult alligator snapping turtles from the Flint River and its tributaries between 1971 and 1983 (Johnson 1989).

Although rigorous data does not exist, it is well understood that extensive harvest, both commercial and personal, depleted populations in many rivers (Florida Fish and Wildlife Conservation Commission 2011). Typically, after periods of heavy effort, declining yields forced harvesters to move on to other sites (Pritchard 2006). This is not unexpected given the long generation time of alligator snapping turtles and the normally low rates of recruitment of virtually all turtles (Florida Fish and Wildlife Conservation Commission 2011).

Recent population surveys of alligator snapping turtles demonstrate populations are depleted throughout its range and even likely extirpated in its historic range in Iowa, Illinois, Kentucky, Missouri and Tennessee. Pritchard (1989) speculated that this turtle has declined up to 95 percent over much of its range. To be sure, the combined effects of targeted harvest, habitat loss, degradation and fragmentation, and perhaps increased predation pressure on nests and juveniles from human-subsidized predators, likely amounts to a halving of total populations since 1950 with some of these impacts being irreversible (van Dijk and Rhodin 2010).

THREATS

As explained below, overharvesting and habitat alteration are the major threats to the species (Reed et al. 2002, Riedle et al. 2005).

Habitat alteration and destruction:

Water pollution and erosion have altered the food chain and otherwise degraded the turtle's habitat in many areas (Heck 1998, Riedle et al. 2005).

In addition, dredging river bottoms to maintain shipping channels destroys habitat, although the subsequent spoil may be utilized for nesting along certain rivers. Riedle et al. (2005) noted a drastic decline of alligator snapping turtles in Oklahoma, due in part to habitat degradation because of stream channelization. Jensen and Birkhead (2003) stated that stream dredging likely inhibits recovery in Georgia. In southeastern Missouri, Shipman and Riedle (2008) found that most sites had been manipulated for channelization or drained and converted to agricultural fields.

Dams have blocked passage of alligator snapping turtles on many rivers. Riedle et al. (2005) noted a drastic decline of alligator snapping turtles in Oklahoma, due in part to thermal alteration by hypolimnetic releases from impoundments. Long-term impacts of microhabitat changes and shoreline development are unstudied but likely negative (Ewert et al. 2006).

Overutilization:

Given its size and catchability, the alligator snapping turtle has a long history of both commercial and personal harvest throughout its range (Dobie 1971, Sloan and Lovich 1995, Reed et al. 2002, Ewert et al. 2006, Pritchard 2006, van Dijk and Rhodin 2010).

Juveniles are extensively traded in the pet trade and to supply East Asian aquaculture operations (van Dijk and Rhodin 2010). Hatchlings are sold online for $28-50 each (www.netpetfinder.com; www.turtlesellers.com). Most hatchlings offered by dealers are said to have been ''captive-bred,'' although these are likely to have been hatched from eggs collected from nests in the wild (USFWS 2000). Larger individuals can sell for hundreds of dollars (www.turtleforum.com). In Asia, dealers sell adult alligator snapping turtles to private turtle collectors, private and public zoos and aquariums because of their huge size and dragon-like appearance.

The alligator snapping turtle meat trade is much larger than the pet trade. In the 1960s and early 1970s, alligator snapping turtles were intensively trapped for the meat trade in Mississippi, Louisiana, Georgia, Alabama, and Texas. In the 1970s alligator snapping turtles were hunted by trappers for a Campbell's soup product, after marine turtles were afforded federal protection under the Endangered Species Act in 1973 (Jensen et al. 2008). In 1982, alligator snapping turtle meat sold for $3.50-$4.50 per pound; a 100 pound turtle can produce 30 pounds of meat (Pritchard 1989).

According to Santhuff (1993), the most serious problem with the commercial take of these turtles is that the efforts of very few trappers can deplete population levels far below self-sustaining levels. If an area is worked for only two nights, then the population is so severely depleted that it is no longer self-sustaining.

Reed et al. (2002) concluded that "many populations were decimated by commercial harvest in the 1960s and 1970s." Various commercial turtle dealers have indicated that populations in Louisiana and other southern states are seriously depleted (Holt and Tolson 1993). From 1994-2007 turtle dealers in Missouri, Arkansas and Louisiana raced to stockpile adult alligator snapping turtles from the wild under the auspices that they were saving them from being sold at seafood markets in Louisiana, when dealers were genuinely targeting the species and buying adults from collectors as broodstock to support an international food and turtle trade market.

Analysis of trade data obtained from the FWS Office of Law Enforcement showed that live alligator snapping turtles have been exported in increasing numbers. In 2009, data indicates that approximately 41,000 live alligator snappers were exported from the United States, with 98 percent of these shipped to China (Weissgold 2010). In contrast, only 1,016 alligator snapping turtles were exported in 1990 (Weissgold 2010). Exports of wild caught alligator snapping turtles have also remained high since the 2006 listing on Appendix III of CITES. From 2006-2010, over 140,000 live, wild caught alligator snapping turtles were exported from the United States.

Although commercial harvest of alligator snapping turtle is now prohibited across its range in states along the Gulf Coast and Mississippi River, wild caught adults are legally sold by licensed turtle dealers in Louisiana, who allege to have possessed the adults prior to November 2004 when Louisiana closed commercial harvest. Adults are also legally sold by a Missouri turtle dealer who utilizes the same allegation. Hatchlings from wild caught adults appear to be the majority of exports to Asia.

The FWS and state agencies have documented illegal hunting of adults to supply the international food and turtle and turtle trade. *See United States v. Guthrie*, 50 F3d 936 (11th Cir. 1995) (defendant conspired to sell alligator snapping turtles in violation of the Lacey Act). In a 2008 incident, a Florida pet shop owner was charged with possession of alligator snapping turtles (O'Connor 2008). In 2009, a New York man was arrested for felony commercialization of wildlife involving a federal protected endangered species and also illegally possessed alligator snapping turtles (Auer 2009).

In addition, inadvertent mortality is a threat. Threats include both trot lines (long lines of submerged baited hooks) and bush lines (single hooks suspended from tree branches) (Ewert et al. 2006, Pritchard 2006). The latter may be more widely used in rivers and hence likely present a greater problem for the alligator snapping turtle (Florida Fish and Wildlife Conservation Commission 2011). Unattended trotlines have, through inadvertent snagging of turtles, resulted in mortality in Missouri (Santhuff 1993). Jensen and Birkhead (2003) stated that mortality on set-lines and trotlines likely inhibits recovery in Georgia.

Disease and predation:

As for all turtles, predation, particularly by human-subsided raccoons, accounts for the loss of a majority of alligator snapping turtle eggs in Florida (about 2/3 along the lower Apalachicola River) (Florida Fish and Wildlife Conservation Commission 2011). Human-subsidized predators are undoubtedly a threat in other parts of the range as well.

Inadequacy of existing regulatory mechanisms:

In 2006, the alligator snapping turtle was listed on Appendix III of CITES. Since that listing, available data conclusively demonstrates that international trade remains extensive. The species has been proposed or recommended for listing on Appendix II of CITES by several groups, including the IUCN/SSC Tortoise and Freshwater Turtle Specialist Group and the Center for

Biological Diversity. In 2010, the FWS convened a Status and Monitoring Working Group during a conference entitled "Conservation and Trade Management of Freshwater and Terrestrial Turtles in the United States" (USFWS 2010). That group also recommended an Appendix II CITES listing.

The alligator snapper is listed or a species of concern in every state within its range (Buhlmann and Gibbons 1997). For example, it is listed as endangered in Indiana and Illinois (Illinois Endangered Species Protection Board 2011, Indiana Dept. of Natural Resources 2011) and harvest is prohibited there (PARC 2011). And it is threatened in Georgia and Texas (GA DNR 2008, Texas Parks and Wildlife 2010).

It is a species of special concern in Florida (Florida Fish and Wildlife Conservation Commission 2011). Beginning in 1973, enactment of a series of protective rules in Florida reduced the species' rate of decline in Florida, although harvest (legal and illegal) still occurred. Rule changes in 2009 prohibited take of all snapping turtles and ended legal harvest. Whether the statewide population has stabilized or begun to increase is undetermined (Ewert et al. 2006).

The alligator snapping turtle is legally harvested in Louisiana (Louisiana Dept. of Wildlife and Fisheries 2011). Louisiana prohibits commercial harvest but allows recreational harvest of 1 per day per boat or vehicle. Harvest is regulated as a nongame species in Kansas and as a game species in Nebraska (PARC 2011). Although commercial use is prohibited in most states, people can take the species for personal use in most States, and there is almost no management of the species by State agencies (USFWS 2000).

In sum, although the alligator snapping turtle is now protected from take across most of its range, collection remains a threat. Endangered Species Act protection would prohibit all take, and increase fines and resources for enforcement that would likely deter illegal collection. In addition, federal protection would help reduce the loss and degradation of the turtle's habitat, which is now the primary factor causing observed population declines. Plus, endangered species status would likely lead to additional conservation actions, such as releases in areas of extirpations (see, e.g., Ream and Scott 2008).

Other factors:

Chemical pollution (from industries such as pulp mills, and waste products from cities and agricultural activities) poses an ongoing threat to riverine fauna (Florida Fish and Wildlife Conservation Commission 2011). For example, the industrially degraded Fenholloway River in Taylor County, Florida once may have supported the alligator snapping turtle but presumably no longer does (Jackson 1999). Because this turtle has the capability of achieving weights in excess of 200 pounds and the potential for its life to span a number of decades, it is a primary target for the bioaccumulation of organochlorines (Holt and Tolson 1993).

Siltation from road crossings, borrow pits, or other development reduces the suitability of smaller streams, such as the clear seepage streams on Eglin Air Force Base in Florida, that the species utilizes (Florida Fish and Wildlife Conservation Commission 2011).

Exotic predators also threaten the alligator snapping turtle. These include wild hogs and imported fire ants (Florida Fish and Wildlife Conservation Commission 2011). In addition, Holcomb and Carr (2011) found larvae of the phorid fly *Megaselia scalaris* in eggs in a naturally incubated nest of an alligator snapping turtle, which contributed to the failure of the nest.

Genetic studies have documented past population bottlenecks and extremely low dispersal by females from one drainage basin into another (Echelle et al. 2009). As such, isolation must also be considered a threat.

References:

Auer, D. 2009. Staten Islander traded in illegal animals, authorities say. Staten Island Advance (Mar. 27, 2009).

Bluett, R.D., D.A. Woolard, J.G. Palis, and J.A. Kath. 2011. Survey for *Macrochelys temminckii* in Southern Illinois: Implications for Recovery Actions. Transactions of the Illinois State Academy of Science 104(1/2): 63-71.

Boundy, J., and C. Kennedy. 2006. Trapping survey results for the alligator snapping turtle (*Macrochelys temminckii*) in southeastern Louisiana, with comments on exploitation. Chelonian Conservation and Biology 5: 3-9.

Buhlmann, K. A., and J. W. Gibbons. 1997. Imperiled aquatic reptiles of the southeastern United States: historical review and current conservation status. Pages 201-232 in G. Benz and D. E. Collins, editor. Conference on aquatic fauna in peril: the southeastern perspective. Southeast Aquatic Research Institute, Lenz Design & Communications, Decatur, GA. 554 pp.

Conant, R. and J.T. Collins. 1991. A field guide to reptiles and amphibians of eastern and central North America. 3rd ed. Houghton-Mifflin Co., Boston, MA. 450pp.

Crother, B. I. (editor). 2008. Scientific and standard English names of amphibians and reptiles of North America north of Mexico, with comments regarding confidence in our understanding. Sixth edition. Society for the Study of Amphibians and Reptiles Herpetological Circular 37: 1-84.

Crother, B. I., J. Boundy, J. A. Campbell, K. de Queiroz, D. R. Frost, R. Highton, J. B. Iverson, P. A. Meylan, T. W. Reeder, M. E. Seidel, J. W. Sites, Jr., T. W. Taggart, S. G. Tilley, and D. B. Wake. 2000. Scientific and standard English names of amphibians and reptiles of North America north of Mexico, with comments regarding confidence in our understanding. Society for the Study of Amphibians and Reptiles, Herpetological Circular No. 29. 82 pp.

Dobie, J. L. 1971. Reproduction and growth in the alligator snapping turtle *Macroclemys temminckii* (Troost). Copeia 1971: 645-658.

Echelle, A.A., J.C. Hackler, J.B. Lack, S.R. Ballard, J. Roman, S.F. Fox, D.M. Leslie, Jr., R.A. Van Den Bussche. 2009. Conservation genetics of the alligator snapping turtle: cytonuclear evidence of range-wide bottleneck effects and unusually pronounced geographic structure. Conservation Genetics.

Ernst, C.H., J.E. Lovich, and R.W. Barbour. 1994. Turtles of the United States and Canada. Smithsonian Institution Press, Washington. 578 pp.

Ernst, C.H. and J.E. Lovich. 2009. Turtles of the United States and Canada. The John Hopkins University Press. 827 pages.

Ewert, M. A., D. R. Jackson, and P. E. Moler. 2006. *Macrochelys temminckii* – alligator snapping turtle. Pages 58–71 *in* P. A. Meylan, editor. Biology and conservation of Florida turtles. Chelonian Research Monographs No. 3.

Florida Fish and Wildlife Conservation Commission. 2011a. Florida's Endangered and Threatened Species, *available at* http://myfwc.com/media/214168/Threatened_Endangered_Species.pdf (last visited June 27, 2011).

Florida Fish and Wildlife Conservation Commission. 2011b. Alligator Snapping Turtle Biological Status Review Report, *available at* http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=1&ved=0CCcQFjAA&url=http%3A%2F%2Fmyfwc.com%2Fmedia%2F1351487%2FAlligator%2520Snapping%2520Turtle%2520Final%2520BSR.pdf&ei=-Ss0T6eLE6jg2AWi1P2WAg&usg=AFQjCNGYayvoVFuOCrP0Rbu6ANBojCeE5Q (last visited Feb. 9, 2012).

Georgia Department of Natural Resources (GA DNR). 2008. Special Concern Animal Species in Georgia, *available at* http://georgiawildlife.dnr.state.ga.us/content/specialconcernanimals.asp (last visited June 27, 2011).

Heck, B. A. 1998. The alligator snapping turtle (*Macroclemys temminckii*) in southeast Oklahoma. Proceedings of the Oklahoma Academy of Science 78: 53-58.

Holcomb, S.R., and J.L. Carr. 2011. Infestation of a naturally incubated nest of the alligator snapping turtles (*Macroclemys temminckii*) by the phorid fly *Megaselia scalaris*. Southwestern Naturalist 56(3): 427-30.

Holt, C. J. and K. M. Tolson, Ph. D. 1993. A Research Proposal Submitted to: The Chelonina Research Foundation. Northeast Louisiana University, Monroe. 8 pp.

Illinois Endangered Species Protection Board. 2011. Checklist of endangered and threatedn animals and plants of Illinois, *available at* http://www.dnr.state.il.us/espb/ (last visted July 5, 2011).

Indiana Dept. of Natural Resources. 2011. Indiana's Species of Greatest Conservation Need, *available at* http://www.in.gov/dnr/fishwild/2356.htm (last visited July 5, 2011).

Jackson, D. R. 1999. Survey of an important distributional "gap" in the Florida range of the river cooter and other freshwater turtles. Final Report to Nongame Wildlife Program, Florida Game and Fresh Water Fish Commission.

Jensen, J. B., and W. S. Birkhead. 2003. Distribution and status of the alligator snapping turtle (Macrochelys temminkii) in Goergia. Southeastern Naturalist 2: 25-34.

Jensen, J.B., C.D. Camp, J.W. Gibbons, and M.J. Elliott. 2008. Amphibians and Reptiles of Georgia. University of Georgia Press, Athens, Georgia, USA.

Johnson, S. 1989. Population status of the alligator snapping turtle (Macroclemys temminckii) in the Flint River. Annual performance report, Georgia Department of Natural Resouces, Forsyth, GA. 11 pp.

Lovich, Jeffrey E. and Clarence J. McCoy. 1993. Graptemys pulchra Baur 1893, Alabama Map Turtle. The Conservation Biology of Freshwater Turtles (IUCN/SSC Tortoise and Freshwater Turtle Specialist Group Action Plan Rating). 2 pp.

Louisiana Department of Fisheries and Wildlife. 2011. Tracking list and fact sheets, *available at* http://www.wlf.louisiana.gov/wildlife/rare-animals-fact-sheets (last visited June 27, 2011).

Moler, P. E. 1996. Alligator Snapping Turtle Distribution and Relative Abundance. Final report for the Florida Game and Freshwater Fish Commission.

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, *available at* http://www.natureserve.org/explorer .

O'Connor, L. 2008. Florida Wildlife Commission investigating pet shop owner. The Palm Beach Post News (July 15, 2008).

Partners in Amphibian and Reptile Conservation. 2011. Midwest State Turtle Harvest Regulations, *available at* http://www.mwparc.org/products/turtles/ (last visited July 18, 2011).

Phelps, J. P. 2004. Aquatic turtles of diversely managed watersheds in the Ouachita Mountains, Arkansas. Pages 183-186 in J. M. Guldin (editor). Ouachita and Ozark Mountains Symposium: Ecosystem Management Research. General Technical Report SRS-74. Department of Agriculture, Forest Service, Southern Research Station, Asheville, NC.

Pritchard, P. C.H. 1989. The alligator snapping turtle, biology and conservation. Milwaukee Public Museum, Milwaukee, Wisconsin. 104 pp.

Pritchard, P. C. H. 1992. Alligator snapping turtle *Macrochelys temminckii* (Harlan). Pages 171-177 in P. E. Moler, editor. Rare and endangered biota of Florida. Vol. III. Amphibians and reptiles. Univ. Press of Florida.

Pritchard, P. C. H. 2006. The alligator snapping turtle: biology and conservation. Second edition. Krieger Publishing Company, Malabar, Florida, USA. 140pp.

Ream, J.T., and A.F. Scott. 2008. Preliminary observations on habitat preference, movement patterns, and survival of introduced juvenile alligator snapping turtles [*Macrochelys temminckii*] in Fayette County, Tennessee. Southeastern Biology 55(3): 305.

Reed, R. N., J. Congdon, and J. W. Gibbons. 2002. The alligator snapping turtle [Macrochelys (Macroclemys) temminckii]: a review of ecology, life history, and conservation, with demographic analyses of the sustainability of take from wild populations. Report to United States Fish and Wildlife Service, Arlington, Virginia. 17 pp.

Riedle, J. D., P. A. Shipman, S. F. Fox, and D. M. Leslie, Jr. 2005. Status and distribution of the alligator snapping turtle *Macrochelys temminckii* in Oklahoma. Southwestern Naturalist 50:79-84.

Santhuff, S. 1993. Letter to Steve N. Wilson, et al., Arkansas Game and Fish Commission, 14 September 1993.

Shipman, P. A., and J. D. Riedle. 2008. Status and distribution of the alligator snapping turtle (*Macrochelys temminckii*) in southeastern Missouri. Southeastern Naturalist 7: 331-338.

Sloan, K.N., and J. E. Lovich. 1995. Exploitation of the alligator snapping turtle, *Macroclemys temminckii,* in Louisiana: a case study. Chelonian Conservation and Biology 1: 221-222.

Texas Parks and Wildlife. 2010. Endangered and Threatened Reptiles and Amphibians in Texas and the United States, *available at* http://www.tpwd.state.tx.us/huntwild/wild/species/endang/animals/reptiles_amphibians/ (last visited June 27, 2011).

Trauth, S.E., J.D. Wilhide, and A. Holt. 1998. Population structure and movement patterns of alligator snapping turtles (Macroclemys temminckii) in northeastern Arkansas. Chelon. Conserv. Biol. 3: 64-70.

U.S. Fish and Wildlife Service. 2000. Proposed Rule: Notice of Intent To Include Several Native U.S. Species in Appendix III to the Convention on International Trade in Endangered Species of Wild Fauna and Flora. 65 Fed. Reg. 4220 (Jan. 26, 2000).

U.S. Fish and Wildlife Service. 2010. Conservation, Status & Monitoring Working Group Findings and Recommendations, *available at* http://www.fws.gov/international/DMA_DSA/CITES/Appendix_III/turtles_ws.html (last visited Feb. 8, 2011).

van Dijk, P.P. and A.G.J. Rhodin. 2010. Draft IUCN Red List Assessments for Freshwater Turtle Species of the United States. August 2010.
Webb, R.G. 1995. The date of publication of Gray's Catalogue of Shield Reptiles. Chelonian Conservation and Biology 1: 322-323.

Weissgold, B. 2010. U.S. Turtle Exports and Federal Trade Regulations: A Snapshot in U.S. Fish and Wildlife Service, Conservation and Trade Management of Freshwater and Terrestrial Turtles in the United States: Workshop Presentation Abstracts, *available at* http://www.fws.gov/international/DMA_DSA/CITES/Appendix_III/turtles_ws.html (last visited Feb. 8, 2011).

Scientific Name:

Emydoidea blandingii

Common Name:

Blanding's Turtle

G Rank:

G4

IUCN Red List:

Endangered

<u>NATURAL HISTORY, BIOLOGY, AND STATUS</u>

Range:

Blanding's turtles are found in Canada and the Midwest and Northeast regions of the United States. The species is distributed disjunctly from southeastern Ontario, adjacent Quebec, and southern Nova Scotia, south into New England, and west through the Great Lakes region to western Nebraska, Iowa, and extreme northeastern Missouri (Congdon et al. 2008). With the exception of two populations in the western portion of their range (Minnesota and Nebraska), populations are frequently small, discontinuous, and often isolated and suffer from significant nest predation and habitat loss (Congdon et al. 2008).

Habitat:

In general, Blanding's turtles occupy a variety of eutrophic wetlands such as swamps, marshes, beaver dams, ponds, and slow moving streams (see Wieten et al. 2012). Blanding's turtles frequently emerge from water to bask on logs and tussocks, or sedge clumps (Congdon et al. 2008). Blanding's turtles nest in well-drained soils with low vegetation cover near wetlands (Congdon et al. 2008).

Blanding's turtle habitat also has a large terrestrial component that consists of nesting areas and movement corridors. The terrestrial component of the core habitat is larger than that of many other aquatic turtle species, and both sexes use terrestrial corridors for movements among wetlands and for nesting migrations (Congdon and Keinath 2006). In a Michigan study, Congdon et al. (2011) found that terrestrial protection zones 300 and 450 m around all wetlands (residence and temporary) protect 90 percent and 100 percent of nests, respectively.

Biology:

Three aspects of the biology of Blanding's turtles give them a slow rate of potential recovery and make them particularly susceptible to disturbance and increased rates of adult mortality (Congdon and Keinath 2006, COSEWIC 2009, van Dijk 2010q).

First, Blanding's turtles have temperature-dependent sex determination and some populations have biased adult sex ratios (e.g., a population in southeastern Michigan has an adult sex ratio close to 1 male to 4 females) (Congdon and Keinath 2006).

Second, reproductive output of Blanding's turtles is low. Females do not begin to reproduce until they are between 14 and 20 years old, do not reproduce every year, and have small clutch sizes, thus resulting in low fecundity. This means that annual survivorship between ages 1 and maturity must average at least 60 percent to maintain population stability (Congdon and Keinath 2006).

Third, Blanding's turtles are long-lived (even compared to other turtles), reaching 77 years in the wild (Ernst and Lovich 2009). Primary reproductive importance is placed upon older adults, and because potential reproductive life spans are longer than generation times, it increases the likelihood of inbreeding in isolated populations (Congdon and Keinath 2006).

Population Status:

Blanding's turtles qualify for endangered species protection because the species is experiencing significant and ongoing population declines across a significant portion of its range due to habitat loss, predation, and other factors. van Dijk (2010q) explains that Blanding's turtles have suffered extensive slow declines of most of its populations from habitat loss and direct removal, accidental mortality, and increased predation. The species was a candidate for federal protection until the FWS eliminated the C2 category, but it currently receives no federal protection under the ESA.

While Blanding's turtles are secure in Nebraska, they range from being vulnerable to threatened, or endangered throughout most of the rest of their distribution (Smith et al. 2006, Congdon and Keinath 2006). The largest population of Blanding's turtles presently known is on the Valentine National Wildlife Refuge in north central Nebraska (Lang 2004). The next largest population exists at Weaver Dunes in southeastern Minnesota (Pappas et al. 2000).

Areas outside of Nebraska and Minnesota constitute a significant portion of the range of this species because those areas make up the majority of the species' historic range and the species' longterm viability cannot be sustained by the remaining remnant populations in Nebraska and Minnesota. Moreover, Mockford et al. (2007) concluded that populations separated by the Appalachian Mountains as well as the highly disjunct Nova Scotian populations of Blanding's turtle should be recognized as evolutionarily significant units, as areas in Appalachian Mountains and the Hudson River appear to present major barriers to gene flow in Blanding's turtle and the extent of fine-scale genetic structure previously reported in the Nova Scotian populations was not found in other parts of the species' range.

In Canada, two populations of Blanding's turtles are recognized: Nova Scotia and Great Lakes. The three small subpopulations of this species found in central southwest Nova Scotia total fewer than 250 mature individuals. Although the largest subpopulation occurs in a protected area, its numbers are still declining. The other subpopulations are also susceptible to increasing habitat degradation, mortality of adults and predation on eggs and hatchlings (COSEWIC 2009). The Great Lakes/St. Lawrence population of this species is declining (COSEWIC 2010).

<u>THREATS</u>

Habitat alteration and destruction:

Blanding's turtles are suffering from degradation of wetlands and the terrestrial portion of their core habitat. These impacts are especially severe near large metropolitan areas (Dorff 1992).

Destruction of resident aquatic habitat is of primary conservation concern because it impacts all stages of the life cycle. Reduction in the numbers of such wetlands can increase risks of mortality for adults and reduce hatchling recruitment into populations. For example, loss of their wet prairie habitat has depleted populations in Illinois (Nyboer 1992, Phillips et al. 1999). And inundation or drainage of wetland habitats for agriculture, river channelization, and water impoundment have reduced available habitat in Minnesota (Coffin and Pfannmuller 1988).

Cultivation to the edge of wetlands and the use of fertilizers, pesticides, and herbicides that wash into wetlands can also degrade aquatic habitats. Blanding's turtles are known to be sensitive to use of herbicides, which destroy aquatic vegetation and likely affect the turtle itself (Kofron and Schreiber 1985). Blanding's turtles were nearly extirpated in Missouri due to marsh drainage and use of pesticides (Kofron and Schreiber 1985).

Water management activities related to fish management and agriculture can be detrimental to overwintering Blanding's turtle populations if they are conducted during winter (Congdon and Keinath 2006; see also Ashley and Robinson 1996, Levell 2000). Drawdown activities to remove undesired fishes such as carp and vegetation in lakes were cited in both Illinois and Minnesota as detrimental due to death from predation, road mortality, freezing when the substrate was exposed in late winter, and poisoning from pesticides sprayed on the exposed lake bottom after the turtles were already moving in late spring (Nyboer 1992, Dorff 1992, Hall and Cuthbert 2000).

Subpopulations are increasingly fragmented by the extensive road network that crisscrosses all of this turtle's habitat, and Blanding's turtles have been reported as being impacted by road mortality (van Dijk 2010q). Indeed, the complex movement ecology and habitat requirements of Blanding's turtles make their populations especially vulnerable to road mortality: over the course of a year, they typically visit multiple wetlands to forage, mate, thermoregulate, and overwinter (Harding 1992, Grgurovic and Sievert 2005), requiring frequent overland migrations and road crossings. Nesting females are especially susceptible to roadkill because they often attempt to nest on gravel roads or on shoulders of paved roads (Congdon and Keinath 2006). Populations in Minnesota suffer from loss of individuals killed on roads (Dorff 1992). Road mortality has also been suggested as one of the greatest threats to the species in Michigan (Harding 1992), second to habitat destruction.

Sex ratios can be affected by habitat changes. Habitat succeeding to shrubs creates a cooler incubation environment for the eggs so that hatchlings appear to be predominantly males in Iowa (Nyboer 1992). Male biased sex ratios are likely to limit the reproductive capacity of the species.

Overutilization:

While Blanding's turtles are not consumed, the pet trade is a serious ongoing threat (COSEWIC 2009, van Dijk 2010q). The species is the second commonest turtle in bycatch of commercial trapping of snapping turtles using baited traps and a ready market exists (van Dijk and Rhodin 2010). Collectors may earn $45 for a 15-20 cm turtle (Coffin and Pfannmuller 1988). In conjunction with their extended longevity and long reproductive lives, collection of adults, juveniles, and hatchlings from small and isolated populations for the pet trade can result in severe reductions and extirpation of populations (Congdon and Keinath 2006).

International trade in Blanding's turtles was bigger in the past. 329 Blanding's turtles were exported from the United States between 1989 and 1997 (Franke and Telecky 2001). Since then, trade appears to have slowed, with available data reporting that just six wild-caught turtles were traded from 2006-2010 (on file with Adkins Giese), but it is possible that many wild caught Blanding's turtles are falsely reported as captive stock. As populations of wood turtles and box turtles become depleted, pet collectors will likely shift their attention to Blanding's turtles. Lovich observed more than 50 Blanding's turtles in the collection of a Georgia turtle dealer in the 1980s (Ernst and Lovich 2009). Levell (2000) discussed commercial exploitation for the live animal trade.

Disease or predation:

Eggs and young hatchlings are highly vulnerable to predation by birds, mammals, and predatory fishes. In Michigan, Congdon et al. (1983) found that nesting success rate was only 22 percent and the probability of surviving to emergence was 0.18. Forty-nine of the 73 nests observed by Congdon were destroyed by predators, and 47 percent of the predation occurred within the first 24 hours after completion of nesting and 84 percent within the first 5 days after completion. Raccoons (*Procyon lotor*) and foxes (*Vulpes vulpes* and *Urocyon cinereoargenteus*) were the most common predators. Distance from water seems to have no correlation with predation

(Congdon et al. 1983), but nests in open areas are more vulnerable than those located on roadsides or ditch banks.

Petokas (1986) found a variation in nest predation during his 6-year study in Ontario. The first year, 17 percent of the nests were destroyed by predators. During the next 4 years, the researchers moved nests and applied protective coverings. However, some nests were still damaged by raccoons that removed the covers, and striped skunks (*Mephitis mephitis*) that tunneled beneath the covers. In the sixth year, the nests were not moved or monitored and Petokas found evidence that 100 percent of the season's nests were likely destroyed.

Ross and Anderson (1990) found that all of 16 nests on their study site in Wisconsin were destroyed by predators. Nine of the 16 nests were destroyed by skunks (*Mephitis mephitis*), the other predators were unknown although red fox, raccoons, and badgers (*Taxidea taxus*) occurred in the area. Ten of the nests were within 50 m of habitat edge.

Jones and Sievert (2012) studied nesting success in a Massachusetts study area. They found nests within residential landscapes to have higher mortality rates than nests in nonresidential landscapes. Most observed or inferred mortalities resulted from eastern chipmunk (*Tamias striatus*) attacks, while cars, birds, and domestic horses caused the remaining mortalities.

Inadequacy of existing regulatory mechanisms:

Currently, the Blanding's turtle receives no protection under CITES from unsustainable international trade. Yet the Blanding's turtle (*Emydoidea blandingii*) was recommended for inclusion in Appendix II of CITES by the Conservation, Status & Monitoring Working Group that the FWS convened during the September 2010 conference entitled "Conservation and Trade Management of Freshwater and Terrestrial Turtles in the United States" (USFWS 2010). The Center for Biological Diversity also recommended that the species by included on CITES Appendix II based on population declines and threat of the international pet trade.

The Blanding's turtle is protected from take by statute in several states. In Minnesota, Wisconsin, Illinois, Indiana, Maine, Massachusetts, New Hampshire, and New York, it is protected as "endangered" or "threatened" (MN DNR 2011, WI DNR 2011, Illinois Endangered Species Protection Board 2011, Indiana Dept. of Natural Resources 2011, Maine Dept. of Inland Fisheries and Wildlife 2010, Massachusetts Dept. of Fish and Game 2008, New Hampshire Fish and Game 2008, New York Dept. of Environmental Conservation 2011). It is a species of special concern in Pennsylvania (Pennsylvania Natural Heritage Program 2011). It has no special status in Michigan, Nebraska, Ohio, South Dakota, and Iowa (Nebraska Game and Parks 2009, Iowa DNR 2009a, Michigan DNR 2009, Ohio Dept. of Natural Resources 2010, South Dakota Game Fish and Parks 2010), but most of these states have some restrictions on harvest of freshwater turtles.

The Committee on the Status of Endangered Wildlife in Canada recognizes two populations of Blanding's turtle. Since May of 2005, the Nova Scotia population is listed as endangered and the Great Lakes population is listed as threatened (COSEWIC 2009).

Federal protection is needed to ensure that take is prohibited across the turtle's range. Moreover, federal protection is needed to ensure that essential habitats of the turtle are protected. While some remaining populations are found on lands protected from development, such as Valentine National Wildlife Refuge in north central Nebraska and Weaver Dunes in southeastern Minnesota, no state laws prevent degradation of the turtle's habitat, which is a primary threat to its viability.

Other factors:

The Blanding's turtle is also threatened by degraded water quality from pesticides and other pollutants. Blanding's turtles in Nebraska were found to be highly susceptible to the pesticide dieldrin that was applied to cornfields for insect control and accumulated in wetland habitats. Although the use of this pesticide was halted in 1974, the chemical is very persistent in the environment (Congdon et al. 2006).

References:

Ashley, E.P., and J.T. Robinson. 1996. Road mortality of amphibians, reptiles and other wildlife on the Long Point Causeway, Lake Erie, Ontario. Can. Field Nat. 110: 403-412.

Beaudry, F., P.M. Demaynadier, and M.L. Hunter, Jr. 2010. Identifying Hot Moments in Road-Mortality Risk for Freshwater Turtles. Journal of Wildlife Management 74(1): 152-60.

Coffin, B., and L. Pfannmuller, eds. 1988. Minnesota's endangered flora and fauna. Univ. Minnesota Press, Minneapolis. 473 pp.

Committee on the Status of Endangered Wildlife in Canada ("COSEWIC"). 2009. Wildlife Species Search, *available at* http://www.cosewic.gc.ca/eng/sct1/index_e.cfm (last visited July 11, 2011).

Congdon, J. D., D. W. Tinkle, G. L. Brettenbach, and R. C. van Loben Sels. 1983. Nesting ecology and hatching success in the turtle *Emydoidea blandingii*. Herpetologica 39(4): 417-429.

Congdon, J.D., T.E. Graham, T. B. Herman, J.W. Lang, M.J. Pappas, and B.J. Brecke. 2008. *Emydoidea blandingii* (Holbrook 1938) – Blanding's Turtle. Chelonian Research Monographs 5, *available at* http://www.iucn-tftsg.org/ind-com/ (last visited February 9, 2011).

Congdon, J.D., O.M. Kinney, and R.D. Nagle. 2011. Spatial ecology and core-area protection of Blanding's Turtle (*Emydoidea blandingii*). Canadian Journal of Zoology 89(11): 1098.

Congdon, J.D., and D.A. Keinath. 2006. Blanding's Turtle (*Emydoidea blandingii*): A Technical Conservation Assessment, USDA Forest Service, Rocky Mountain Region, *available at* http://www.fs.fed.us/r2/projects/scp/assessments/blandingsturtle.pdf (last visited July 5, 2011).

Committee on the Status of Endangered Wildlife in Canada ("COSEWIC"). 2009. Wildlife Species Search, *available at* http://www.cosewic.gc.ca/eng/sct1/index_e.cfm (last visited July 11, 2011).

Ernst, C.H. and J.E. Lovich. 2009. Turtles of the United States and Canada. The John Hopkins University Press. 827 pp.

Frank, J. and T.M. Telecky. 2001. Reptiles as pets: An examination of the trade in live reptiles in the United States. Humane Soc. U.S.

Grgurovic, M. and P.R. Sievert. 2005. Movement patterns of Blanding's turtles (*Emydoidea blandingii*) in the suburban landscape of eastern Massachusetts. Urban Ecosystems 8: 203–213.

Gruchow, P. 1988. A rare turtle crossing. Minnesota Monthly October: 19-23.

Hall, C. D., and F. J. Cuthbert. 2000. Impact of a controlled wetland drawdown on Blanding's turtles in Minnesota. Chelonian Conservation and Biology 3: 643-649.

Illinois Endangered Species Protection Board. 2011. Checklist of endangered and threatened animals and plants of Illinois, *available at* http://www.dnr.state.il.us/espb/ (last visted July 5, 2011).

Indiana Dept. of Natural Resources. 2011. Indiana's Species of Greatest Conservation Need, *available at* http://www.in.gov/dnr/fishwild/2356.htm (last visited July 5, 2011).

Iowa Dept. of Natural Resources. 2009a. Endangered, threatened, and special concern animals, *available at* http://www.iowadnr.gov/Environment/ThreatenedEndangered.aspx (last visited July 5, 2011).

Jones, M.T. and P.R. Sievert. 2012. Elevated Mortality of Hatchling Blanding's Turtles (*Emydoidea blandingii*) in Residential Landscapes. Herpetological Conservation and Biology 7(1): 89−94.

Kofron, C. P., and A. A. Schreiber. 1985. Ecology of two endangered aquatic turtles in Missouri: *Kinosternon flavescens* and *Emydoidea blandingii*. J. Herpetol. 19:27-40.

Lang, J.W. 2004. Blanding's turtles on Valentine NWR, Nebraska: population status, estimate of population size, and road mortality. Final Report for 2002-2003 Nebraska Dept. of Roads: Project EACNH-STPB-83-4.

Levell, J. P. 2000. Commercial exploitation of Blanding's turtle, *Emydoidea blandingii*, and the wood turtle, *Clemmys insculpta*, for the live animal trade. Chelonian Conservation and Biology 3: 665-674.

Maine Dept. of Inland Fisheries and Wildlife. 2010. Maine Endangered Species Program/State list, *available at* http://www.maine.gov/ifw/wildlife/species/endangered_species/state_list.htm (last visited July 5, 2011).

Massachusetts Dept. of Fish and Game. 2008. Massachusetts List of Endangered, Threatened and Special Concern Species, *available at* http://www.mass.gov/dfwele/dfw/nhesp/species_info/mesa_list/mesa_list.htm (last visited July 5, 2011).

Michigan Dept. of Natural Resources. 2009. Endangered and threatened species, *available at* http://www.michigan.gov/dnr/0,1607,7-153-10370_12141_12168---,00.html (last visited July 5, 2011).

Minnesota Department of Natural Resources (MN DNR). Minnesota's List of Endangered, Threatened, and Special Concern Species, *available at* http://www.dnr.state.mn.us/rsg/index.html (last visited Aug. 1, 2011).

Mockford, S. W., T.B. Herman, M. Snyder, and J.M. Wright. 2007. Conservation genetics of Blanding's turtle and its application in the identification of evolutionarily significant units. Conservation Genetics 8(1): 209-20.

Nebraska Game and Parks. 2009. Nebraska Endangered and Threatened Species, *available at* http://outdoornebraska.ne.gov/wildlife/programs/nongame/pdf/E&T_SPECIES_LIST_7-09%5B1%5D.pdf (last visited July 5, 2011).

New Hampshire Fish and Game. 2008. Endangered and threatened wildlife of NH, *available at* http://www.wildlife.state.nh.us/Wildlife/Nongame/endangered_list.htm (last visited July 5, 2011).

New York Dept. of Environmental Conservation. 2011. List of Endangered, Threatened and Special Concern Fish & Wildlife Species of New York State, *available at* http://www.dec.ny.gov/animals/7494.html (last visited July 5, 2011).

Ohio Dept. of Natural Resources. 2010. Ohio's endangered species, *available at* http://www.dnr.state.oh.us/wildlife/home/resources/mgtplans/endangered/tabid/6005/default.aspx (last visited July 5, 2011).

Pappas, M.J., B.J. Brecke, and J.D. Congdon. 2000. The Blanding's turtle of Weaver Dunes, Minnesota. Chelonian Conservation and Biology 3: 557-68.

Pennsylvania Natural Heritage Program. 2011. Species of Special Concern Lists, *available at* http://www.naturalheritage.state.pa.us/Species.aspx (last visited July 5, 2011).

Petokas, P. J. 1986. Patterns of reproduction and growth in the freshwater turtle *Emydoidea blandingii*. Ph.D. dissertation, Graduate School of the University Center at Binghamton, New York.

Phillips, C. A., R. A. Brandon, and E. O. Moll. 1999. Field guide to amphibians and reptiles of Illinois. Illinois Natural History Survey Manual 8. xv + 282 pp.

Ross, D. A., and R. K. Anderson. 1990. Habitat use, movements, and nesting of *Emydoidea blandingii* in central Wisconsin. J. Herpetol. 24: 6-12.

Smith, G.R., J.B. Iverson, and J.E. Rettig. 2006. Changes in a turtle community from a northern Indiana lake: A long-term study. Journal of Herpetology 40: 180–185.

South Dakota Game, Fish, and Parks. 2010. Endangered, threatened, and candidate species of South Dakota, *available at* http://gfp.sd.gov/wildlife/threatened-endangered/threatened-species.aspx (last visited July 5, 2011).

U.S. Fish and Wildlife Service. 2010. Conservation, Status & Monitoring Working Group Findings and Recommendations, *available at* http://www.fws.gov/international/DMA_DSA/CITES/Appendix_III/turtles_ws.html (last visited Feb. 8, 2011).

van Dijk, P.P. 2010q. *Emydoidea blandingii*. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.1, *available at* www.iucnredlist.org (last visited July 18, 2011).

van Dijk, P.P. and A.G.J. Rhodin. 2010. Draft IUCN Red List Assessments for Freshwater Turtle Species of the United States. August 2010.

Wieten, A., M. Cooper, A. Parker, and D. Uzarski. 2012. Great Lakes coastal wetland habitat use by seven turtle species: influences of wetland type, vegetation, and abiotic conditions. Wetlands Ecology & Management 20(1): 47-59.

Wisconsin DNR. 2011. Wisconsin Endangered and Threatened Species Laws and Lists, *available at* http://dnr.wi.gov/org/land/er/ (last visited July 5, 2011).

SNAKES

Key Ring-necked Snake (*Diadophis punctatus acricus*)

Florida Pine Snake (*Pituophis melanoleucus mugitus*)

Rim Rock Crowned Snake (*Tantilla oolitica*)

Apalachicola Kingsnake (*Lampropeltis getula meansi*)

Short-tailed Snake (*Lampropeltis extenuata*)

Southern Hog-nosed Snake (*Heterodon simus*)

Southern Rubber Boa (*Charina umbratica*)

Scientific Name:

Diadophis punctatus acricus

Common Name:

Key Ring-necked Snake or Key Ringneck Snake

G Rank:

T1

IUCN Red List:

Subspecies not reviewed

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

The Key ringneck snake is endemic to the Lower Keys of Monroe County, Florida, and as such, it has an extremely restricted distribution (NatureServe 2011). The Key ringneck snake has been found on five keys: Key West, Big Pine, Little Torch, Middle Torch, and No Name keys (Weaver et al. 1992, Auth and Scott 1996, FFWCC 2011a). It has been speculated that, based upon suitable habitat, it might also occur on Ramrod, Cudjoe, Summerland and Sugarloaf keys (Paulson 1968, Weaver et al. 1992).

Habitat:

The Key ringneck snake inhabits pine rockland habitat and the edges or disturbed portions of rockland hammocks (i.e., tropical hammocks) (Lazell 1989, Weaver et al. 1992). It is also found around limestone outcroppings (NatureServe 2011). It seems to be restricted to areas in the vicinity of permanent fresh water, which often occur as small holes in the limestone (Lazell 1989, Florida Natural Areas Inventory 2001). All *Diadophis* apparently require moist microhabitats to balance evaporative water loss from the body (Myers 1965, Clark 1967).

Biology and Taxonomy:

The Key ringneck snake is considered to be the most taxonomically distinct of the Keys snakes (NatureServe 2011). Collins (1991) proposed that this subspecies be recognized as a distinct species, but he did not present supporting data.

Average adult size is 6 inches (15.2 cm). Adults are small and slender-bodied with a slate gray body. Unlike other ringneck snakes, the ring normally present around the neck is indistinct or faded. The belly is bright orangish-yellow, fading to orange-red beneath the tail. There is a single row of half-moon spots down the center on the belly. The scales are smooth, and there are 15-17 dorsal scale rows at midbody. The pupil is round. Juvenile color is similar to that of the adult (Florida Museum of Natural History 2011).

The diet of *Diadophis* elsewhere consists of small amphibians, lizards, snakes, insects, slugs, and earthworms (*see* Ernst and Ernst 2003). The main foods for the Key ringneck snake specifically are likely worms, geckos, and insects (R. Ehrig, personal comm. 2012). There is no information on reproduction in this subspecies (FFW 2011), but *Diadophis* typically lay clutches of 1–10 eggs and may produce more than 1 clutch annually (*see* Ernst and Ernst 2003).

Population Status:

The Key ringneck snake qualifies for endangered species status because it is rare and declining across its very small range where its habitat is experiencing rapid loss due to development. NatureServe (2011) considers the Key ringneck snake to be critically imperiled because of its small range in southern Florida. The species is rare and known from only a few keys in a region where habitat destruction is occuring at a rapid pace, and no occurrence is fully protected (NatureServe 2011). It is likely that habitat destruction and alteration have resulted in population declines (FFWCC 2011a). The species was a candidate for federal protection until the FWS eliminated the C2 category, but it currently receives no federal protection under the ESA.

The Florida Fish and Wildlife Conservation Commission (2011a) prepared a biological status report that summarizes the available information on the species. They note that it is rare but probably still occurs on all five keys within its historic range. The Key ringneck snake was first documented in 2003 on Key West from the grounds of the Banyan Resort in a highly developed portion of the City of Key West; a total of 4 snakes were found at this site as of 2007 (Florida Museum of Natural History records, cited in FFWCC 2011a). The most recent records from other keys are 1980 on Middle Torch Key, 1984 on Little Torch Key, 1995 on No Name Key, and 2007 on Big Pine Key (museum and Florida Natural Areas Inventory records, cited in FFWCC 2011a). These records represent incidental observations; no surveys have been conducted for the taxon since the 1980s (see Lazell 1989). It is possible that the record for Big Pine Key came from an introduction through potted landscape plants (R. Ehrig, personal comm. 2012).

THREATS

Habitat alteration and destruction:

The Key ringneck snake is a habitat specialist that is experiencing rapid loss of its rockland habitats. Clearing of pine rockland and rockland hammock habitats has likely eliminated Key ringneck snakes from some areas (Enge et al. 2003, FFWCC 2011a). Real estate development is a primary threat to this snake (NatureServe 2011).

According to Bentzien (1987), over 98 percent of pine rockland habitat has been lost. These rockland habitats are highly fragmented and are embedded in a matrix of agricultural and residential landscapes (O'Brien 1998). Many of the hammocks on the Keys and mainland were cleared for agriculture, firewood, and charcoal in the 1800s, and almost all pinelands were clearcut by 1950 (Snyder et al. 1990). In the Keys, most rockland hammocks are privately owned and are in demand for commercial and residential development (Enge et al. 2003).

The snakes appear able to tolerate some habitat disturbance and can survive in vacant lots (Lazell 1989, NatureServe 2011). Yet continuing decline in extent of habitat is projected because of development of vacant lots (FFWCC 2011a).

Snakes are found on roads, and road mortality is likely a factor contributing to declines, particularly in areas on Big Pine Key with a dense network of roads (FFWCC 2011a).

Moreover, pine rockland is a fire "subclimax" community that is dependent upon frequent fires to retard hardwood succession (Enge et al. 2003). Intense development precludes beneficial fire management of remaining rocklands.

Overutilization:

Although Key ringneck snakes are non-venomous, malicious killings by humans likely contribute to mortality. "Ringneck snakes are valued in the pet trade for their attractive coloration" (Yung 2000). As such, collection of these snakes for the pet trade is also likely a threat but this has not been documented.

Inadequacy of existing regulatory mechanisms:

The Key ringneck snake is listed as threatened in Florida (FFWCC 2011b). Protected status makes killing or collecting the snakes illegal but does not prevent habitat destruction, which is the biggest threat to the snake.

The snake's rockland hammock habitats on the Keys are mostly privately owned and will probably be cleared for commercial or residential developments if not protected (Enge et al. 2003). Regulatory authority over rocklands in the Keys is found in the local comprehensive plan, which is enforced by the Department of Community Affairs (Enge et al. 2003). Development of rockland habitats in the Keys is not precluded, and property owners compete for

255 permits annually through the Rate of Growth Ordinance (Anonymous 1999). Federal protection is required to ensure that remaining habitats are adequately protected from development.

The snake is found on Key Deer and Great White Heron national wildlife refuges (NatureServe 2011). These few protected occurrences are not enough to ensure population viability. As Enge et al. (2003) explains: "The rockland forest fragments presently protected in the Keys are inadequate to prevent the extinction of some taxa from catastrophic events like hurricanes, so a network of protected fragments is necessary to provide refugia and the corridors necessary for populations to later recolonize devastated areas."

Moreover, even if habitat is protected within the refuges, the snakes are still subject to other threats within those areas, including road mortality. The snakes are consumed by fire, and inappropriate controlled burns on the Key Deer Refuge are another threat (R. Ehrig, personal comm. 2011, 2012).

Other factors:

The Keys are mostly only 1–2 m above sea level and are prone to inundation by seawater during tropical storms and hurricanes (Snyder et al. 1990). Hurricanes hit South Florida about every three years (Gentry 1974), and associated seawater surges and short-term flooding of upland habitats in the Keys probably kill some snakes and their prey (FFWCC 2011a). In October of 2005, Big Pine Key was hit by a 8 foot storm surge from Hurricane Wilma (R. Ehrig, personal comm. 2012). The storm surge covered 98 percent of the northern island for at least an hour, which killed thousands of trees and harmed the snake and its prey (*id*.). Climate change could increase the frequency of these storm events.

In addition, a sea level rise due to climate change could significantly impact this taxon (FFWCC 2011a). In the best-case scenario, a sea level rise of 18 cm (7 inches) by Year 2100 would inundate 34 percent of Big Pine Key, resulting in the loss of 11 percent of the island's upland habitat (FFWCC 2011a). In the worst-case scenario, a sea level rise of 140 cm (4.6 feet) by Year 2100 would inundate 96 percent of Big Pine Key.

The red imported fire ant (*Solenopsis invicta*) has invaded the Lower Keys, and predation by this nonnative species has been suggested as a reason for declines in some oviparous snake populations in the Southeastern Coastal Plain (Mount 1981). In a study conducted in the Lower Keys, transects with the highest probability of the presence of fire ants were those closest to roads and with the most development within a 150-m radius (Forys et al. 2002). Because of its terrestrial nature and small size, the ringneck snake and its prey would appear to be particularly susceptible to fire ants (FFWCC 2011a).

Nonnative predators include the cane toad (*Rhinella marina*) and Cuban treefrog (*Osteopilus septentrionalis*) (Meshaka et al. 2004, Krysko and Halvorsen 2010). In 2005, opossums from the northern Keys or mainland were introduced to Big Pine Key, and they are an abundant possible predator of the snake, along with the native raccoon (R. Ehrig, personal comm. 2012). Overbrowsing by Key deer is also likely a threat to the snake (*id*.).

In addition, human cultivation has resulted in the establishment of nonnative trees, such as citrus and banana, in many rockland hammocks (Gunderson 1994). Exotic plants have invaded more than 2,833 ha of upland habitat in the Keys (Kruer et al. 1998). Areas of disturbed substrate adjacent to hammocks are often heavily infested with exotic plants that then rapidly invade the hammocks (Anonymous 1999).

References:

Anonymous. 1999. South Florida multiple-species recovery plan. Prepared for the U.S. Fish and Wildlife Service, Southeast Region, Atlanta, Georgia, USA.

Auth, D. L., and C. Scott. 1996. Geographic distribution: *Diadophis punctatus acricus* (Key ringneck snake). Herpetological Review 27: 33.

Bentzien, M. M. 1987. Agency draft recovery plan for five rockland plant species. U.S. Fish and Wildlife Service, Atlanta, Georgia, USA.

Clark, D. R., Jr. 1967. Experiments into selection of soil type, soil moisture level, and temperature by five species of snakes. Transactions of the Kansas Academy of Science 70: 490–496.

Collins, J. T. 1991. Viewpoint: a new taxonomic arrangement for some North American amphibians and reptiles. SSAR Herpetol. Review 22: 42-43.

Enge, K. M., B. A. Millsap, T. J. Doonan, J. A Gore, N. J. Douglass, and G. L. Sprandel. 2003. Conservation plans for biotic regions in Florida containing multiple rare or declining wildlife taxa. Florida Fish and Wildlife Conservation Commission, Bureau of Wildlife Diversity Conservation Final Report, Tallahassee, Florida, USA. 146pp.

Ernst, C. H., and E. M. Ernst. 2003. Snakes of the United States and Canada. Smithsonian Books, Washington, D.C., USA. 668pp.

Florida Fish and Wildlife Conservation Commission. 2011a. Key Ringneck Snake Biological Status Review Report, *available at* http://www.myfwc.com/media/1351568/Key%20Ringneck%20Snake%20Final%20BSR.pdf (last visited Oct. 4, 2011).

Florida Fish and Wildlife Conservation Commission. 2011b. Florida's Endangered and Threatened Species, *available at* http://myfwc.com/media/1515251/Threatened_Endangered_Species.pdf (last visited Oct. 4, 2011).

Florida Museum of Natural History. 2011. Key Ringneck Snake (*Diadophis punctatus acricus*, *available at* http://www.flmnh.ufl.edu/herpetology/fl-guide/diadophispacricus.htm (last visited Oct. 4, 2011).

Florida Natural Areas Inventory. 2011. Key Ringneck Snake - *Diadophis punctatus acricus*, *available at* http://fwcg.myfwc.com/docs/ringneck_snake.pdf (last visited Oct. 4, 2011).

Forys, E. A., C. R. Allen, and D. P. Wojcik. 2002. Influence of the proximity and amount of human development and roads on the occurrence of the red imported fire ant in the Lower Florida Keys. Biological Conservation 108: 27–33.

Gentry, R. C. 1974. Hurricanes in South Florida. Pages 73–81 *in* P. J. Gleason, editor. Environments of South Florida: present and past. Miami Geological Society Memoirs No. 2, Miami, Florida, USA.

Gunderson, L. H. 1994. Vegetation of the Everglades: determinants of community composition. Pages 323–340 in S. M. Davis and J. C. Ogden, editors. Everglades: the ecosystem and its restoration. St. Lucie Press, Delray Beach, Florida, USA.

Kruer, C. R., T. Armstrong, and P. Braisted. 1998. Invasive exotic vegetation mapping in the Florida Keys. Report and 122-MB CD prepared for the Florida Keys Invasive Exotics Task Force, Summerlin Key, Florida, USA.

Kruer, C. R. 1992. An assessment of Florida’s remaining coastal upland natural communities: Florida Keys. Prepared for Florida Natural Areas Inventory, Tallahassee, Florida, USA.

Krysko, K. L., and M. D. Halvorsen. 2010. *Osteopilus septentrionalis* (Cuban treefrog). Prey. Herpetological Review 41: 339–340.

Lazell, J. D., Jr. 1989. Wildlife of the Florida Keys: a natural history. Island Press, Covelo, California, USA. 254pp.

Meshaka, W. E., Jr., B. P. Butterfield, and J. B. Hauge. 2004. The exotic amphibians and reptiles of Florida. Krieger, Melbourne, Florida, USA. 166pp.

Mount, R. H. 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. Journal of the Alabama Academy of Science 52:71–78.

Myers, C. W. 1965. Biology of the ringneck snake, *Diadophis punctatus*, in Florida. Bulletin of the Florida State Museum, Biological Sciences 10: 43–90.

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, available at http://www.natureserve.org/explorer . (last visited Oct. 4, 2011).

O’Brien, J. J. 1998. The distribution and habitat preferences of rare Galactia species (Fabaceae) and Chamaesyce deltoidea subspecies (Euphorbiaceae) native to southern Florida pine rockland. Natural Areas Journal 18: 208–222.

Paulson, D. R. 1968. Variation in some snakes from the Florida Keys. Quarterly Journal of the Florida Academy of Sciences 29: 295–308.

Snyder, J. R., A. Herndon, and W. B. Robertson, Jr. 1990. South Florida rockland. Pages 230–277 in R. L. Myers and J. J. Ewel, editors. Ecosystems of Florida. University of Central Florida Press, Orlando, Florida, USA.

Weaver, W. G., S. P. Christman, and P. E. Moler. 1992. Big Pine Key ringneck snake, *Diadophis punctatus acricus* Paulson. Pages 146–149 *in* P. E. Moler, editor. Rare and endangered biota of Florida. Volume III. Amphibians and reptiles. University Press of Florida, Gainesville, Florida, USA.

Yung, J. 2000. "*Diadophis punctatus*" (On-line), Animal Diversity Web, *available at* http://animaldiversity.ummz.umich.edu/site/accounts/information/Diadophis_punctatus.html (last visited July 6, 2012).

Scientific Name:

Pituophis melanoleucus mugitus

Common Name:

Florida Pine Snake

G Rank:

T3

IUCN Red List:

Subspecies not reviewed

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

The Florida pine snake is associated with the Coastal Plain physiographic region, occurring in the extreme southeastern United States from southwestern South Carolina southwestward to Mobile Bay in southern Alabama, and south into Florida, excluding the Everglades (Conant and Collins 1991, Ernst and Ernst 2003). Pine snakes once occurred along the Atlantic Coastal Ridge as far south as Miami (Duellman and Schwartz 1958, FFWCC 2011), but urban development in southeastern Florida has eliminated some of these populations. There are no recent museum or Florida Natural Areas Inventory records south of Martin County, but the species apparently still occurs in Palm Beach County (FFWCC 2011).

Conant and Collins (1991) indicated that Alabama populations assigned to this subspecies by Mount (1975) were actually intergrades between *Pituophis melanoleucus mugitus* and other subspecies. The core of the Florida pine snake's range is in Florida and Georgia (NatureServe 2011).

Habitat:

The Florida pine snake prefers habitats with well-drained, sandy soils and moderate to open canopy cover (Franz 1992, Ernst and Ernst 2003). The most common natural habitat of pine snakes in Florida is sandhills, but they also are found in scrub, xeric hammock, scrubby flatwoods, and mesic pine flatwoods and dry prairie with dry soils (Allen and Neill 1952, Enge 1997, Franz 2005).

During a telemetry study in northern peninsular Florida, 69 percent of Florida pine snake observations were in sandhill (i.e., high pine), followed by ruderal habitats (i.e., pastures, former orange groves, and old hay fields), xeric hammock, and lake edge (Franz 2005). During a telemetry study in southwestern Georgia, pine snakes used habitats relative to their availability within their home ranges, but at a landscape level, they had a significant preference for mixed pine-hardwood forests, whereas all other habitats (i.e., pine regeneration plots and plantations, scrub/shrub or fallow land, agricultural fields or wildlife food plots, urban areas, hardwood forest, natural pine forest, and aquatic habitat) were used relative to their availability (Miller 2008).

Florida pine snakes are fossorial, spending approximately 80 percent of their time in underground retreats, primarily burrows of the southeastern pocket gopher (*Geomys pinetis*) (Franz 2005, Miller 2008). Other retreats used are stumpholes, mole runs, and burrows of gopher tortoises (*Gopherus polyphemus*), nine-banded armadillos (*Dasypus novemcinctus*), and mice (Franz 2005, Miller 2008). Florida pine snakes are diurnally active and occasionally climb into shrubs and small trees (Franz 2005).

Biology:

The Florida pine snake commonly uses burrows of the gopher tortoise and southeastern pocket gopher for shelter (Franz 1992). Its diet includes the southeastern pocket gopher, immature rabbits, mice, and other small mammals, ground-dwelling birds, and bird eggs (Alabama Dept. of Conservation and Natural Resources 2011). As with other pine snakes, it spends much of its time underground (Alabama Dept. of Conservation and Natural Resources 2011). Observers have reported clutches of from four to eight large white, leathery eggs laid from June to August and hatching in September and October (Mount 1975). Additional biological information on the Florida pine snake has been summarized by Franz (1992), Ernst and Ernst (2003), Franz (2005), and Miller (2008).

Population Status:

The Florida pine snake qualifies for endangered status because it is suffering significant declines and range contractions and faces the ongoing threats of habitat destruction and overcollection.

Franz (1992) reported that pine snakes had seriously declined in the last 20 years (Franz 1992). The populations are likely to have since suffered from further declines with the loss of additional habitat. The species was a candidate for federal protection until the FWS eliminated the C2 category, but it currently receives no federal protection under the ESA.

In Florida, observations at the Ordway-Swisher Biological Station in Putnam County suggest a major decline in the population (Franz 2005). Additionally, of 464 records from museums, Florida Natural Areas Inventory (FNAI), and the literature, 105 records were from the 1990s and just 64 from the 2000s, which is also likely indicative of declining populations.

In Alabama, the pine snake is of local occurrence and cannot be said to be common anywhere (Alabama Department of Conservation and Natural Resources 2011). In South Carolina, pine snakes are also not abundant in any particular area of the state (SC DNR undated).

THREATS

Population declines are likely due to excessive collecting, road mortality, and habitat loss (Franz 1992), which also threaten other subspecies of pine snakes (Golden et al. 2009).

Habitat alteration and destruction:

A primary threat to the Florida pine snake is habitat loss due to conversion for agriculture, silviculture, mining, roads, and commercial/residential development (NatureServe 2011, FFWCC 2011). The sandhill habitat is being lost, altered, and fragmented at a rate dangerous to all of its biotic components (Landers and Speake 1980).

Longleaf pine-dominated sandhill as well as scrub habitat on the ridges of central Florida and along both coasts have suffered serious losses (Means and Grow 1985, Myers 1990, Kautz 1998, Enge et al. 2003). Enge et al. (2003) provided descriptions of sandhill habitat and threats to its wildlife community. By 1987, 88 percent of Florida's sandhill habitat had been lost (Kautz et al. 1993), and scrub habitat has also experienced serious losses (Enge et al. 2003). It is estimated that more than 97 percent of the original longleaf pine (*Pinus palustris*) ecosystem has been converted to agriculture, pine plantations, and urban areas (Noss et al. 1995). From 1985–89 to 2003, 15.5 percent of Florida's sandhill habitat, 12.4 percent of its scrub habitat, and 9.2 percent of its pinelands were converted to other uses, primarily urban or other developed uses (Kautz et al. 2007). Shrub and brushland, a semi-natural cover type often used by pine snakes, lost 36.3 percent of its acreage to intensive human uses (Kautz et al. 2007).

Altered fire regimes in sandhill habitat and resulting hardwood encroachment also degrade habitat conditions for pine snakes, although they will use xeric hammocks (FFWCC 2011). Also, pine snakes do poorly in dense pine plantations, particularly sand pine (*Pinus clausa*) plantations on former sandhill sites in the Florida Panhandle (FFWCC 2011). Stumpwood removal likely affects pine snake subpopulations by decreasing underground habitat structure (Means 2005); this is particularly detrimental in areas where pocket gophers are absent (FFWCC 2011).

Paved roads, sand roads, and trails are prevalent on most federal and state lands, making pine snakes susceptible to mortality from off-road vehicles and other human use of these areas (FFWCC 2011). Large, slow-moving snakes, such as the Florida pine snake, are highly susceptible to road mortality (Andrews and Gibbons 2005); in eastern Texas, populations of large snakes were 50 percent less abundant up to 450 m from roads than they were 850 m from roads (Rudolph et al. 1999).

Florida pine snakes likely avoid crossing major highways abutting their home ranges (Miller 2008); data using the random path generator in ArcGIS versus actual snake movements suggest that pine snakes cross paved roads and wide, graded roads significantly less than expected. Thus, the species is likely vulnerable to effects of habitat fragmentation, including isolation effects.

Overutilization:

Excessive collecting is an additional threat in some areas (Franz 1992). Collection of pine snakes for pets in Florida presumably decreased when they were listed as a Species of Special Concern because commercialization was prohibited and a personal possession limit of one snake was imposed (FFWCC 2011). Commercialization is permitted for “albino” (i.e., amelanistic and leucistic) individuals, however (FFWCC 2011).

In addition, because pine snakes are large and conspicuous, populations in residential areas are threatened by killing by humans and domestic pets (Jordan 1998). Reduced populations in Alabama is likely related in part to the practice of “gassing” tortoise burrows (Mount 1975). Research on some ecological effects of gassing tortoise burrows has shown that Florida pine snakes subjected to gasoline fumes died within 24 days (Speake and Mount 1973).

Inadequacy of existing regulatory mechanisms:

The Florida pine snake is listed as a species of special concern in Florida (FFWCC 2011b) and has been recommended for threatened status (FFWCC 2011a). Special concern status prohibits commercialization and imposes a personal possession limit of one snake (Nanjappa and Conrad 2012). In Georgia and Alabama, the snake is protected from take as a nongame species. Ala. Admin. Code r. 220-2-.92; O.C.G.A. § 27-1-28. It lacks protected status in South Carolina (SC DNR 2010) but is of conservation concern (SC DNR 2004).

Although the Florida pine snake is protected from take across most of its range, the habitat of the Florida pine state is inadequately protected across its range and is being rapidly lost. In Florida, the snakes are found on some state and federal public lands, including Apalachicola National Forest and Ocala National Forest (FFWCC 2011). But even within these lands, the snakes are not often detected, possibly because the lands provide largely marginal habitat for the snakes (FFWCC 2011). As an example, during the past 23 years, a researcher spent an average of 1−2 weeks per year conducting herpetofaunal surveys in Apalachicola National Forest, Tate’s Hell State Forest, and Apalachicola Bluffs and Ravines Preserve in Florida but found only 2 live pine snakes, 3 road-killed snakes, and 1 shed skin (FFWCC 2011). Their rarity on these protected lands could be the result of road mortality or habitat degradation in the absence of fire. Some land uses within the national forests are also likely incompatible with preservation of snakes including intensive timber management.

Other factors:

Because of their association with pocket gophers, pine snake populations might be expected to decline in response to declines in pocket gopher populations, such as from pest control programs (FFWCC 2011). Pocket gopher populations have apparently declined in Alabama, Georgia, and to a lesser extent, Florida (Georgia Department of Natural Resources 2008, Miller et al. 2008), and a subspecies of pocket gopher in Florida is now extinct (Humphrey 1992).

Finally, pine snakes might be experiencing increased rates of predation of adults, hatchlings, or eggs from subsidized mammalian predators, nine-banded armadillos (*Dasypus novemcinctus*), feral hogs (*Sus scrofa*), domestic cats and dogs, and red imported fire ants (*Solenopsis invicta*) (FFWCC 2011).

References:

Alabama Dept. of Conservation and Natural Resources. 2011. Florida Pine Snake, *available at* http://www.outdooralabama.com/watchable-wildlife/what/Reptiles/Snakes/fps.cfm (last visited Oct. 6, 2011).

Allen, E. R., and W. T. Neill. 1952. The southern pine snake. Florida Wildlife 5: 18–19.

Andrews, K. M., and J. W. Gibbons. 2005. How do highways influence snake movement? Behavioral responses to roads and vehicles. Copeia 2005: 772–782.

Conant, R., and J. T. Collins. 1991. A field guide to amphibians and reptiles of eastern and central North America. Third edition. Houghton Mifflin, Boston, Massachusetts, USA. 450pp.

Duellman, W. E., and A. Schwartz. 1958. Amphibians and reptiles of southern Florida. Bulletin of the Florida State Museum, Biological Sciences 3: 181–324.

Enge, K. M. 1997. A standardized protocol for drift-fence surveys. Florida Game and Fresh Water Fish Commission Technical Report No. 14, Tallahassee, Florida, USA. 68pp.

Enge, K. M., B. A. Millsap, T. J. Doonan, J. A Gore, N. J. Douglass, and G. L. Sprandel. 2003. Conservation plans for biotic regions in Florida containing multiple rare or declining wildlife taxa. Florida Fish and Wildlife Conservation Commission, Bureau of Wildlife Diversity Conservation Final Report, Tallahassee, Florida, USA. 146pp.

Ernst, C. H., and E. M. Ernst. 2003. Snakes of the United States and Canada. Smithsonian Books, Washington, D.C., USA. 668pp.

Florida Fish and Wildlife Conservation Commission (FFWCC). 2011. Florida Pine Snake Biological Status Review Report, *available at* http://www.myfwc.com/media/1351568/Key%20Ringneck%20Snake%20Final%20BSR.pdf (last visited Oct. 4, 2011).

Florida Fish and Wildlife Conservation Commission (FFWCC). 2011b. Florida's Endangered and Threatened Species, *available at* http://myfwc.com/media/1515251/Threatened_Endangered_Species.pdf (last visited Oct. 4, 2011).

Franz, R. 1992. Florida pine snake *Pituophis melanoleucus mugitus* Barbour. Pages 254-258 in P. E. Moler, editor. Rare and endangered biota of Florida. Vol. III. Amphibians and reptiles. Univ. Press of Florida.

Franz, R. D. 2005. Up close and personal: a glimpse into the life of the Florida pine snake in a North Florida sand hill. Pages 120–131 *in* W. E. Meshaka, Jr., and K. J. Babbitt, editors. Amphibians and reptiles: status and conservation in Florida. Krieger, Malabar, Florida, USA.

Georgia Department of Natural Resources (GA DNR). 2008. Special Concern Animal Species in Georgia, *available at* http://georgiawildlife.dnr.state.ga.us/content/specialconcernanimals.asp (last visited June 27, 2011).

Golden, D. M., P. Winkler, P. Woerner, G. Fowles, W. Pitts, and D. Jenkins. 2009. Status assessment of the northern pine snake (*Pituophis m. melanoleucus*) in New Jersey: an evaluation of trends and threats. New Jersey Department of Environmental Protection, Trenton, New Jersey, USA. 53pp.

Humphrey, S. R. 1992. Goff's pocket gopher, *Geomys pinetis goffi*. Pages 11–18 *in* S. R. Humphrey, editor. Rare and endangered biota of Florida. Volume I. Mammals. University Press of Florida, Gainesville, Florida, USA.

Jordan, R. A. 1998. Species profile: pine snake (*Pituophis melanoleucus* ssp.) on military installations in the southeastern United States. Technical Report SERDP-98-5, U.S. Army Corps of Engineers Waterways Experiment Station, Vicksburg, Mississippi, USA. 24pp.

Kautz, R. S. 1998. Land use and land cover trends in Florida 1936–1995. Florida Scientist 61: 171–187.

Kautz, R. S., D. T. Gilbert, and G. M. Mauldin. 1993. Vegetative cover in Florida based on 1985–1989 Landsat Thematic Mapper Imagery. Florida Scientist 56: 135–154.

Kautz, R., B. Stys, and R. Kawula. 2007. Florida vegetation 2003 and land use change between 1985–89 and 2003. Florida Scientist 70: 12–23.

Landers, J.L. and D.W. Speake. 1980. Management needs of sandhill reptiles in southern Georgia. Proceedings of the Annual Conference of the Southeastern Association of Fish and Wildlife Agencies 34: 515-29.

Means, D. B. 2005. The value of dead tree bases and stumpholes as habitat for wildlife. Pages 74–78 in W. E. Meshaka, Jr., and K. J. Babbitt, editors. Amphibians and reptiles: status and conservation in Florida. Krieger, Malabar, Florida, USA.

Means, D. B., and G. O. Grow. 1985. The endangered longleaf pine community. ENFO (Environmental Information Center of the Florida Conservation Foundation) Report (September): 1–12.

Miller, G. J. 2008. Home range size, habitat associations and refuge use of the Florida pine snake, *Pituophis melanoleucus mugitus,* in southwest Georgia, U.S.A. M.S. Thesis, University of Florida, Gainesville, Florida, USA. 72pp.

Mount, R. H. 1975. The reptiles and amphibians of Alabama. Alabama Agricultural Experiment Station, Auburn University, Auburn, Alabama, USA. 347pp.

Myers, R. L. 1990. Scrub and high pine. Pages 150–193 *in* R. L. Myers and J. J. Ewel, editors. Ecosystems of Florida. University of Central Florida Press, Orlando, Florida, USA.

Nanjappa, P. and P.M. Conrad. 2012. State of the Union: Legal Authority Over the Use of Native Amphibians and Reptiles in the United States, *available at* http://www.fishwildlife.org/files/SOU_FULL-lo-res.pdf.

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, available at http://www.natureserve.org/explorer. (last visited Oct. 4, 2011).

Noss, R. F, E. T. LaRoe III, and J. M. Scott. 1995. Endangered ecosystems of the United States: a preliminary assessment of loss and degradation. U.S. Department of the Interior National Biological Services Biological Report 28. 81pp.

Rudolph, D. C., S. J. Burgdorf, R. N. Conner, and R. R. Schaefer. 1999. Preliminary evaluation of roads and associated vehicular traffic on snake populations in eastern Texas. Pages 128–135 *in* G. L. Evink, P. Garrett, and D. Ziegler, editors. Proceedings of the Third International Conference on Wildlife Ecology and Transportation, Florida Department of Transportation, Tallahassee, Florida, USA.

South Carolina Department of Natural Resources (SC DNR). Undated. Longleaf Pine Reptile Guild, *available at* http://www.dnr.sc.gov/cwcs/pdf/LongleafPineReptilesGuild.pdf (last visted Oct. 6, 2011).

South Carolina Department of Natural Resources (SC DNR). 2004. Priority Species for the South Carolina State Comprehensive Wildlife Conservation Plan, *available at* http://www.dnr.sc.gov/cwcs/pdf/priorityspecies.pdf (last visited Oct. 6, 2011).

South Carolina Department of Natural Resources (SC DNR). 2010. SC Rare, Threatened & Endangered Species Inventory, *available at* http://www.dnr.sc.gov/species/pdf/SC_state_wide.pdf (last visited June 27, 2011).

Speake, D. W., and R. H. Mount. 1973. Some possible ecological effects of "Rattlesnake Roundups" in the southeastern Coastal Plain. Proc. 27th Ann. Conf. S. E. Assoc. Game Fish Comm. 1973: 267-277.

[redacted]

Scientific Name:

Tantilla oolitica

Common Name:

Rim Rock Crowned Snake

G Rank:

G1

IUCN Red List:

Endangered

<u>NATURAL HISTORY, BIOLOGY, AND STATUS</u>

Range:

The rim rock crowned snake has a restricted and fragmented range in habitat subject to rapid development that directly limits its viability. It is endemic to southern Florida (NatureServe 2011), and its range includes eastern Dade County and Monroe County, Florida, including the Eastern Rock Rim of Miami and the Florida Keys (Hammerson 2007, FFWCC 2011, Yirka et al. 2010). On the mainland, the rim rock crowned snake is known from various localities in Miami, including Brownsville, Coconut Grove, Coral Gables, Cutler, Cutler Ridge, Kendall, Leisure City, North Miami, and Perrine (Duellman and Schwartz 1958, FFWCC 2011). Most of the range in Dade County has been lost (Hammerson 2007, Porras and Wilson). Rangewide, the distribution is severely fragmented (Hammerson 2007).

The Florida Fish and Wildlife Conservation Commission summarizes the recent occurences as follows: "The most recent mainland records are from the Barnacle Historic State Park in 2007 (Hines and Bradley 2009) and Zoo Miami (formerly Miami Metrozoo) property in 2009

(FFWCC 2011). The Barnacle Historic State Park supports a population despite containing only 1.6 ha (4 acres) of hammock (Hines and Bradley 2009). The most recent records from the Keys are 1988 on Upper and Lower Matecumbe keys, 1998 on Grassy Key, 2002 on Vaca Key (Marathon), and 2007 on Big Pine Key and Key Largo (Hines and Bradley 2009, FFWCC 2011). Records compiled by Hines and Bradley (2009) show 6 observations in 1930–50, 6 in 1951–70, 18 in 1971–90, and 12 since 1991 (does not include the Zoo Miami record)." Although the snake might be currently found in some small fragments, longterm viability of these isolated populations is unlikely.

Habitat:

The natural habitats of the rim rock crowned snake are pine rockland and rockland hammock (also called tropical hammock) in the Miami area and Florida Keys, but there are records from human-altered habitats such as roadsides, vacant lots, and pastures with shrubby growth and slash pines (*Pinus elliottii*) (Duellman and Schwarz 1958, Campbell and Moler 1992, Hines and Bradley 2009).

Rockland hammock is a hardwood forest that represents an advanced successional stage of pine rockland (FFWCC 2011). Refugium for the snakes are required from hot, dry weather and from potential predators (Porras and Wilson 1979). Pine rocklands have sparse soils; refugia are provided by holes and crevices in the limestone, piles of rock rubble, and pockets of organic matter accumulating in solution holes and shallow depressions in the oolitic limestone (Enge et al. 2003, FFWCC 2011). Specifically, the snake can sometimes be found in rotten stumps and under anthropogenic surface detritus (like rotting boards), fallen logs, palmetto leaves, and rocks (Hammerson 2007, Duellman and Schwarz 1958, Bartlett 2002, Hines and Bradley 2009, Rochford et al. 2010, Yirka et al. 2010). The snake apparently comes to the surface after rains (Porras and Wilson 1979), possibly because of flooding of its underground refugia (FFWCC 2011).

The type specimen came from a now-vanished vacant lot in Miami, where it was found on the Miami Rim Rock made up of oolitic limestone, hence the name *Tantilla oolitica*. Only 26 specimens are known to exist (FMNH 2011).

Biology:

Almost nothing is known regarding the rim rock crowned snake's reproduction, longevity, or diet (FFWCC 2011). Estimates can be made based on the closely related southeastern crowned snake (Todd et al. 2008). The rim rock crowned snake probably reaches sexual maturity at 3 years of age, and there may be 3 eggs in a clutch (*see* Ernst and Ernst 2003, Behler 1979). Prey probably consists of centipedes, insects, and other small invertebrates (Ernst and Ernst 2003).

Population Status:

The rim rock crowned snake qualifies for endangered species status because it is extremely rare across its small and fragmented range in southern Florida where its habitat is being rapidly lost due to urban sprawl. NatureServe (2011) explains that "intense development within this range threatens the existence of nearly all occurrences; the current pace of development may eliminate even marginally suitable habitat." It further explains that the "[a]rea of occupancy, number of subpopulations, and population size probably have declined significantly compared to the historical situation" and that "the remaining populations are declining or deteriorating in quality as very rapid loss of habitat continues" (see also Hammerson 2007).

Sites previously occupied by the species have been developed (FFWCC 2011). For example, the vacant lot in Marathon where several rim rock crowned snakes have been found under trash was cleared in the past decade and turned into a ball park (FFWCC 2011), and a lot with an abandoned house in Miami where snakes were found has been developed (Hines and Bradley 2009).

Very few individuals of this species have ever been found (Hammerson 2007), and the species is represented by a small number of known occurrences (subpopulations) (Telford 1980, Campbell and Moler 1992). The species is considered "extremely rare" (FMNH 2011). The species was a candidate for federal protection until the FWS eliminated the C2 category, but it currently receives no federal protection under the ESA.

THREATS

Habitat alteration and destruction:

The biggest threat to the rim rock crowned snake is habitat loss and fragmentation from development (Porras and Wilson 1979, Campbell and Moler 1992). Rapid and intense habitat modification has occurred and continues within the Miami-Key Largo area (FFWCC 2011, NatureServe 2011).

Enge et al. (2003) provided descriptions of the rockland habitats of South Florida, their threats, and their wildlife communities. Human development and clearing of these habitats, particularly in the Miami area, has severely fragmented populations of the rim rock crowned snake; these rockland habitats are now embedded in a matrix of agricultural and residential landscapes (O'Brien 1998). Approximately 98 percent of the original Miami Rock Ridge pinelands outside of Everglades National Park has been lost (Snyder et al. 1990). Many of the rockland hammocks on the Keys and mainland were cleared for agriculture, firewood, and charcoal in the 1800s, and almost all pinelands were clearcut by 1950 (Snyder et al. 1990).

Overutilization:

The rim rock crowned snake is so rare that humans are very unlikely to encounter it. But as with all snakes, humans kill them out of fear or maliciously or collect them for pets.

Disease or predation:

It is unlikely that natural predators are threatening the rim rocked crowned snake. But Porras and Wilson (1979) suggest that large scorpions (*Centruroides gracilis*) are potential predators of these snakes. And two rim rock crowned snakes were found inside a road-killed eastern coral snake (*Micrurus fulvius*) (Hines and Bradley 2009).

Inadequacy of existing regulatory mechanisms:

The rim rock crowned snake was listed as Threatened by the State of Florida in 1975, and it is considered a Species of Management Concern by the U.S. Fish and Wildlife Service (Scott 2004). The state listing prohibits take but does not protect its habitat. The primary protection need is the preservation of suitable habitat because the current pace of development threatens to eliminate even marginally suitable habitat (NatureServe 2011).

The snake is found on some protected sites, including Crocodile Lake National Wildlife Refuge, John Pennekamp Coral Reef State Park, Key Largo Hammocks, and Port Bougainville. But given the rarity of this snake, all occupied lands must be protected to prevent extirpation. Federal status would likely generate interest and funding needed for conservation actions and would ensure that remaining essential habitats are protected.

Other factors:

A sea level rise due to climate change could significantly impact this species, particularly in the Florida Keys (FFWCC 2011). In the best-case scenario, a sea level rise of 18 cm (7 inches) by Year 2100 would inundate 34 percent of Big Pine Key, resulting in the loss of 11 percent of the island's upland habitat (FFWCC 2011). In the worst-case scenario, a sea level rise of 140 cm (4.6 feet) by Year 2100 would inundate 96 percent of Big Pine Key (FFWCC 2011).

Seawater surges from hurricanes and tropical storms in the relatively xeric Keys can cause rockland habitats to become flooded with salt water for 1−3 days following hurricanes (Enge et al. 2003), and this would impact rim rock crowned snake populations in the short term. Hurricanes strike South Florida about every three years (Gentry 1974), and there is a 1 in 7 chance of Dade or Monroe County being struck in any given year (Fernald and Purdum 1992). These storm events might become more frequent with climate change.

Introduced species are also likely threatening the snake. The Cuban treefrog (*Osteopilus septentrionalis*), cane toad (*Rhinella marina*), and several of the introduced lizard species are capable of preying on small snakes (Meshaka et al. 2004). Some introduced lizard species, especially the litter dwellers, might compete for food with the rim rock crowned snake (FFWCC 2011). As such, the increasing numbers of introduced lizard species in the Miami area and on some of the Keys (Meshaka et al. 2004) should be considered a threat (FFWCC 2011).

The red imported fire ant (*Solenopsis invicta*) has invaded South Florida and the Keys, and predation by this nonnative species has been suggested as a reason for declines in some oviparous snake populations in the Southeastern Coastal Plain (Mount 1981). In a study conducted in the Lower Keys, transects with the highest probability of the presence of fire ants were those closest to roads and with the largest amount of development within a 150-m radius (Forys et al. 2002). Because of its fossorial nature and small size, the rim rock crowned snake would appear to be particularly susceptible to fire ants (FFWCC 2011).

References:

Bartlett, D. 2002. Notes from the field. Krazy for the Keys: to see uncommon herps, take a trip way down south. Reptiles Magazine 10(4): 22–26.

Behler, J. L., and King F. W. 1979. The Audubon Society Field Guide to North American Reptiles and Amphibians. New York: Alfred A. Knopf.

Campbell, H. W, and P. E. Moler. 1992. Rim rock crowned snake, *Tantilla oolitica* Telford. Pages 158–161 *in* P. E. Moler, editor. Rare and endangered biota of Florida. Volume III. Amphibians and reptiles. University Press of Florida, Gainesville, Florida, USA.

Conant, R., and Collins J. T. (1991). A Field Guide to Reptiles and Amphibians of Eastern and Central North America, 3rd edition. Boston: Houghton.

Duellman, W. E., and A. Schwartz. 1958. Amphibians and reptiles of southern Florida. Bulletin of the Florida State Museum, Biological Sciences 3: 181–324.

Enge, K. M., B. A. Millsap, T. J. Doonan, J. A Gore, N. J. Douglass, and G. L. Sprandel. 2003. Conservation plans for biotic regions in Florida containing multiple rare or declining wildlife taxa. Florida Fish and Wildlife Conservation Commission, Bureau of Wildlife Diversity Conservation Final Report, Tallahassee, Florida, USA. 146pp.

Ernst, C. H., and E. M. Ernst. 2003. Snakes of the United States and Canada. Smithsonian Books, Washington, D.C., USA. 668pp.

Fernald, E. A., and E. D. Purdum, editors. 1992. Atlas of Florida. University Press of Florida, Gainesville, Florida, USA.

Florida Fish and Wildlife Conservation Commission (FFWCC). 2011. Rim Rock Crowned Snake Biological Status Review Report, *available at* http://myfwc.com/media/1351610/Rim%20Rock%20Crowned%20Snake%20Final%20BSR.pdf (last visited Oct. 4, 2011).

Florida Fish and Wildlife Conservation Commission (FFWCC). 2011b. Florida's Endangered and Threatened Species, *available at* http://myfwc.com/media/1515251/Threatened_Endangered_Species.pdf (last visited Oct. 4, 2011).

Florida Museum of Natural History. 2011. Rock rim crowned snake, *available at* http://www.flmnh.ufl.edu/herpetology/fl-guide/tantillaoolitica.htm (last visited October 6, 2011).

Forys, E. A., C. R. Allen, and D. P. Wojcik. 2002. Influence of the proximity and amount of human development and roads on the occurrence of the red imported fire ant in the lower Florida Keys. Biological Conservation 108: 27–33.

Gentry, R. C. 1974. Hurricanes in South Florida. Pages 73–81 *in* P. J. Gleason, editor. Environments of South Florida: present and past. Miami Geological Society Memoirs No. 2, Miami, Florida, USA.

Hammerson, G.A. 2007. *Tantilla oolitica*. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.1. <www.iucnredlist.org>. Downloaded on 06 October 2011.

Hines, K. N., and K. A. Bradley. 2009. Assessment of the status and distribution of the endemic rim rock crowned snake (*Tantilla oolitica*) in Miami-Dade and Monroe counties, Florida. Final Report Grant Agreement No. 401817G006. The Institute for Regional Conservation, Miami, Florida, USA. 22pp.

Meshaka, W. E., Jr., B. P. Butterfield, and J. B. Hauge. 2004. The exotic amphibians and reptiles of Florida. Krieger, Melbourne, Florida, USA. 166pp.

Mount, R. H. 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. Journal of the Alabama Academy of Science 52: 71–78.

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, *available at* http://www.natureserve.org/explorer . (last visited Oct. 4, 2011).

O'Brien, J. J. 1998. The distribution and habitat preferences of rare *Galactia* species (Fabaceae) and *Chamaesyce deltoidea* subspecies (Euphorbiaceae) native to southern Florida pine rockland. Natural Areas Journal 18: 208–222.

Porras, L., and Wilson L. D. (1979). N ew Distributional Records for Tantilla oolitica Telford (Reptilia, Serpentes, Colubridae) from the Florida Keys. Journal of Herpetology 13: 218–220.

Rochford, M., K. Hines, and F. J. Mazzotti. 2010. *Tantilla oolitica* (rim rock crowned snake). Defensive behavior. Herpetological Review 41: 99.

Scott, C. 2004. Endangered and Threatened Animals of Florida and their Habitats. Austin, Texas: University of Texas Press.

Snyder, J. R., A. Herndon, and W. B. Robertson, Jr. 1990. South Florida rockland. Pages 230–277 *in* R. L. Myers and J. J. Ewel, editors. Ecosystems of Florida. University of Central Florida Press, Orlando, Florida, USA.

Telford, J.. S. R. 1966. Variation among the southeastern crowned snakes, genus Tantilla. Bulletin of the Florida State Museum, Biological Sciences 10: 261-304.

Todd, B. D., J. D. Willson, C. T. Winne, R. D. Semlitsch, and J. W. Gibbons. 2008. Ecology of the southeastern crowned snake, *Tantilla coronata*. Copeia 2008: 388–394.

Yirka, M. A., J. N. Flowers, M. D. Martin, K. R. Messenger, and N. A. Shepard. 2010. Geographic distribution: *Tantilla oolitica* (rim rock crowned snake). Herpetological Review 41: 382.

Scientific Name:

Lampropeltis getula meansi

Common Name:

Apalachicola Kingsnake

G Rank:

T2

IUCN Red List:

Not reviewed

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

The Apalachicola kingsnake is rare across its small range in the Eastern Apalachicola Lowlands in the Florida panhandle between the Apalachicola and Ochlockonee rivers and south of Telogia Creek, in Franklin and Liberty counties (Krysko and Judd 2006).

Morphological intermediates (i.e., *L. g. goini*) between *L. g. meansi* and *L. g. getula* are found mostly in the surrounding region from southern Gulf and Franklin counties to the west, north to Calhoun County, and east into northern Liberty (north of Telogia Creek), in Gadsden, Leon, Wakulla, and Jefferson counties (Krysko and Judd 2006).

Habitat:

The Apalachicola kingsnake is found in flatwoods of the Gulf Coastal Lowlands, primarily along wetland margins of bayheads, creek swamps, acid bogs, savannas, roadside ditches, dwarf cypress stands, and evergreen shrub communities (Moler 1992). It occasionally wanders into adjacent longleaf pine flatwoods and has been found at the freshwater marsh zone fringing the Apalachicola River and Bay estuary and in freshwater wetlands immediately behind the beachfront in Franklin County (Moler 1992).

Taxonomy and Biology:

Little is known about the life history of the Apalachicola kingsnake. It is believed that individuals take three years to become sexually mature (Means, in Moler 1992). The snake lays eggs in early summer, and hatching occurs in July or August (Means, in Moler 1992).

Long argued as to whether or not it is a sub-species, the Apalachicola kingsnake was formerly named *Lampropeltis getula goini*. After years of research and many more specimens examined, in 2006 it was renamed to *Lampropeltis getula meansi* after D. Bruce Means, in recognition of his work on this species.

Specifically, Krysko and Judd (2006) examined morphological characters and color pattern of *L. getula* throughout its range, particularly those populations in the eastern United States, and made comparisons to molecular data. They describe a subspecies in the Eastern Apalachicola Lowlands as *L. g. meansi*. They justify this subspecies based on the fact that this panhandle clade is diagnosed by more synapomorphies than any other currently recognized taxon of *L. getula*, and overlaps in distribution with numerous other endemic plants and animals. All molecular analyses produced very similar tree topologies in their morphological dataset (Krysko and Judd 2006).

The Apalachicola kingsnake is recognized by the Catalogue of Life (2009). But if the FWS does not accept the Apalachicola kingsnake as a subspecies, we hereby petition for the distinct population segment ("DPS") of common kingsnake (*Lampropeltis getula*) found in the Eastern Apalachicola Lowlands. This population meets the definition of a DPS, as provided in the DPS Policy (61 Fed. Reg. 4725) because it is both discrete and significant. The population is discrete because it is found within the eastern Apalachicola lowlands region between the Ochlocknee (in the east) and Apalachicola Rivers (in the west) and and south of Telogia Creek, which physically separates the Apalachicola kingsnake population from other kingsnake populations. In addition, the DPS Policy provides that "quantitative measures or genetic . . . discontinuity may provide evidence of this [required] separation." 61 Fed. Reg. 4725. Krysko and Judd (2006) observed more synapomorphies than any other currently recognized taxon of *L. getula*, which provides evidence of discreteness. While some hybridization has been documented, the DPS Policy makes clear that complete genetic separation is not required (61 Fed. Reg. 4724). The primary reason why the Apalachicola kingsnake population is "significant" is because of the research by Krysko and Judd (2006) showing that it differs markedly from other kingsnake populations in its genetic characteristics, which is one of the factors for determining significance recognized in the DPS Policy. *See* 61 Fed. Reg. 4725. The Apalachicola kingsnake population is also significant because it persists in an ecological setting unusual or unique for the taxon. As Krysko and Judd (2006) explain, this population "overlaps in distribution with numerous other endemic plants and animals."

Population Status:

Federal protection is needed for the Apalachicola kingsnake because it is declining and rare across its small range in the Florida panhandle where it is exposed to rapid habitat loss and overcollection. In the early to mid 1970s, it was not uncommon to encounter up to five Apalachicola kingsnakes crossing roads during the spring mating season (Krysko and Smith 2005). However, after travelling thousands of kilometers and hours on these same roads during the 1990s, Krysko found only one individual, which had just been killed by a vehicle (Krysko and Smith 2005). Due to the rarity and severely declining populations of nearly all *Lampropeltis getula* in Florida (Krysko 2001, 2002; Krysko and Smith 2005), Krysko and Judd (2006) conclude that the Apalachicola kingsnake should be listed at the state and/or federal level. NatureServe (2011) explains that although surveys are needed, available data shows that the Apalachicola kingsnake is declining and considered rare.

THREATS

Habitat alteration and destruction:

During the 1970s, it was not uncommon to encounter five or more kingsnakes crossing roads during the breeding season in a single day in the Apalachicola National Forest and Tate's Hell State Forest (D.B. Means, unpubl. data, as cited in Krysko and Smith 2005). Yet near the end of the 1970s, Livingston (1977) reported that the southern half of the Apalachicola region (i.e. Apalachicola Lowlands) had been under extensive development, including agricultural activities, timber harvesting, dredging, and damming. By the early 1980s, kingsnakes had begun to decline drastically in this region (Krysko 2001, Krysko and Smith 2005). The decline is presumed to be the result of poor silivicultural practices that are resulting in the loss and degradation of the species' longleaf pine habitats (Krysko and Smith 2005, NatureServe 2011). Today, less than two or three percent of the longleaf pine savanna habitat remains (Ware et al. 1993, Noss et al. 1995, Platt 1999).

In addition, much of the remaining longleaf pine habitats are becoming degraded in the absence of fire. Without active fire management, remnant longleaf pine ecosystems convert into closed canopy forests and become unsuitable for snakes. In the past 200 years, human settlement of the Coastal Plain has drastically altered the normal, summertime fire cycle. Not only have wildfires been actively suppressed following ignition, but roads, towns, agricultural fields, and other developments have impeded the widespread, weeks-long fires that swept the Coastal Plain regularly in pre-settlement times (Means 2011). The disruption of the natural fire cycle has resulted in an increase in slash and loblolly pine on sites formerly dominated by longleaf pine, an increase in hardwood understory, and a decrease in herbaceous ground cover (Wolfe et al. 1988, Yager et al. 2007). On public lands, prescribed burning is a significant part of many habitat management plans. However, implementation of prescribed burning has been inconsistent due to financial constraints and limitations of weather (drought, wind direction, etc.) that restrict the number of opportunities to burn (USFWS 2009).

Some management practices for restoring native longleaf pine have been initiated in Apalachicola National Forest, but habitats in state and private ownership continue to be lost (Krysko and Smith 2005). On Tate's Hell State Forest, there is no effective program for restoration of native habitat and the management plan still allows extensive clearcutting, timber harvesting, and replacing of native longleaf pine forests with slash pine (*Pinus elliottii*) tree farms (Krysko and Smith 2005). In addition, nearly all of the efforts to protect longleaf pine habitats are purely voluntary and without dedicated funding, which means that it is uncertain whether beneficial management actions will continue in the future.

Overutilization:

Threats include private and commercial collecting (NatureServe 2011). This beautiful snake is regularly offered for sale on the Internet (*see, e.g.*, Reptiles-n-critters.com, I-herp.com, thegrandreptile.weebly.com).

Inadequacy of existing regulatory mechanisms:

The Apalachicola kingsnake is not state listed, and as such, as with most other nongame reptiles, it may be taken throughout the year in any manner without limit (Nanjappa and Conrad 2012). Federal protection is needed to prohibit take of this rare snake.

A substantial part of population lives in Apalachicola National Forest (NatureServe 2011), which protects core habitat from development. However, projects for restoring longleaf pine on this forest and limited by available funding and largely voluntary. Moreover, many management actions within the forest are not compatible with snake preservation, including intense timber management and ORV use.

References:

Catalogue of Life. 2009. *Lampropeltis getula meansi*, *available at* http://www.catalogueoflife.org/annual-checklist/2009/show_species_details.php?record_id=4469531 (last visited Feb. 10, 2012).

Krysko, K.L. 2001. Ecology, Conservation, and Morphological and Molecular Systematics of the Kingsnake, *Lampropeltis getula* (Serpentes: Colubridae) (M.S. Thesis).

Krysko, K.L. 2002. Seasonal activity of the Florida kingsnake, *Lampropeltis getula floridana* (Serpentes: Colubridae), in southern Florida. American Midland Naturalist 148: 102–114.

Krysko, K.L. and D.J. Smith. 2005. The decline and extirpation of kingsnakes, *Lampropeltis getula*, in Florida. In: Meshaka, W.E., Jr. & Babbitt, K.J. (Eds.), Status and Conservation of Florida Amphibians and Reptiles. Krieger Publ., Malabar, Florida, pp. 132–141.

Krysko, K.L. and W.S. Judd. 2006. Morphological systematics of kingsnakes, lampropeltis getula complex (Serpentes: Colubridae), in the eastern United States. Zootaxa 1193: 1–39.

Livingston, R. J. 1977. The apalachicola dilemma: wetlands, priorities, development stress and management initiatives. Proceedings of the National Wetland Protection Symposium, June 6-8, Reston, VA., USA. pp. 163-177.

Means, D.B. 2011. Diamonds in the Rough (unpublished manuscript).

Means, D.B., and K. L. Krysko. 2001. Biogeography and pattern variation of kingsnakes, lampropeltis getula, in the Apalachicola region of Florida. Contemporary Herpetology 5.

Moler, P. E., editor. 1992. Rare and endangered biota of Florida. Volume III. Amphibians and reptiles. University Press of Florida, Gainesville. xviii + 291 pp.

Nanjappa, P. and P.M. Conrad. 2012. State of the Union: Legal Authority Over the Use of Native Amphibians and Reptiles in the United States, *available at* http://www.fishwildlife.org/files/SOU_FULL-lo-res.pdf.

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia. Available http://www.natureserve.org/explorer. (Accessed: February 10, 2012).

Noss, R.F., E.T. LaRoe, and J.M. Scott. 1995. Endangered ecosystems of the United States: a preliminary assessment of loss and degradation. USDI National Biological Service, Biological Report 28.

Platt, W.J. 1999. Southeastern pine savannas. In R.C. Anderson et al. (eds.), Savannas, barrens, and rock outcrop plant communities of North America: 23-51, Cambridge: Cambridge University Press.

Pyron, R. Alexander and Frank T. Burbrink. 2009. Systematics of the Common Kingsnake (Lampropeltis getula: Serpentes:Colubridae) and the burden of heritage in taxonomy. Zootaxa 2241: 22-32.

U.S. Fish and Wildlife Service. 2009. Determination of Endangered Status for Reticulated Flatwoods Salamander; Designation of Critical Habitat for Frosted Flatwoods Salamander and Reticulated Flatwoods Salamander; Final Rule. 74 Fed. Reg. 6702 (Feb. 10, 2009).

Ware, S. C. Frost, and P.D. Doerr. 1993. Southern mixed hardwood forest: the former longleaf pine forest. In W.H. Martin et al. (eds.) Biodiversity of the southeastern United States, lowland terrestrial communities: 447-93, New York: John Wiley and Sons.

Wolfe, S. H., J.A. Reidenauer, and D. B. Means. 1988. An ecological characterization of the Florida panhandle. U.S. Fish and Wildlife Service Biological Report 88(12). 277pp.

Yager, L.Y., M.G. Hinderliter, C.D. Heise, and D.M. Epperson. 2007. Gopher tortoise response to habitat management by prescribed burning. Journal of Wildlife Management 71(2): 428-434.

Scientific Name:

Lampropeltis extenuata (formerly *Stilosoma extenuatum*)

Common Name:

Short-tailed Snake

G Rank:

G3

IUCN Red List:

Near threatened

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

The short-tailed snake is endemic to peninsular Florida, from Suwanee and Columbia counties to Hillsborough, Orange, and Highlands counties (Franz et. al. 1992). It is primarily confined to the central ridges, but its range extends west to the Gulf Coast from Levy County southward to Hillsborough and Pinellas counties (Campbell and Moler 1992). The species is not known from west of the Suwannee River (NatureServe 2011).

The species has been recorded from the following counties since 2008: Alachua, Citrus, Hernando, Levy, Marion, and Pasco (museum and Florida Natural Area Inventory records, as cited in FFWCC 2011). There are 38 museum records from Alachua County, but only 1 record exists from Columbia (1975) and Seminole (1892) counties (FFWCC 2011). Although this fossorial species is rarely found, residents living in suitable habitat occasionally find them in carports, woodsheds, foundation excavations, driveways, and yards (FFWCC 2011). Across its small range, the short-tailed snake is suffering from rapid habitat loss.

Habitat:

The short-tailed snake primarily inhabits areas with well-drained sandy soils, particularly longleaf pine (*Pinus palustris*)/xeric oak (*Quercus* spp.) sandhills, but also scrub and xeric hammock habitats (Van Duyn 1939, Carr 1940, Campbell and Moler 1992, Enge 1997). This snake also has been found in sphagnum bog adjacent to typical habitat (Carr and Goin 1955, Ashton and Ashton 1981, Campbell and Moler 1992, Ernst and Ernst 2003). Campbell and Christman (1982) reported that this species is more likely to be found in early successional stages in pine scrub than in advanced stages with a full pine canopy, dense evergreen shrub layer, and matted ground cover.

It is primarily fossorial and spends most of its time burrowed in sand (FFWCC 2011). It has been plowed up by farmers and dug up by gardeners and builders (Van Duyn 1939, Highton 1956, Woolfenden 1962). Some specimens have been found under fallen logs or other cover, including sphagnum moss (Carr 1940), and one was seen entering a gopher tortoise (*Gopherus polyphemus*) burrow (FFWCC 2011).

Biology and taxonomy:

NatureServe (2011) describes the short-tailed snake as “geographically and taxonomically one of the most interesting of North American snakes.” It is perhaps allied with the kingsnakes (*Lampropeltis*) (Dowling and Maxson 1990, Campbell and Moler 1992). Mitochondrial DNA and nuclear DNA data indicate that the nominal genus *Stilosoma* is nested within *Lampropeltis* and thus does not warrant recognition as a distinct genus (Pyron and Burbrink 2009). Collins and Taggart (2009) classify the snake as *Stilosoma extenuatum*. Some accounts recognize three subspecies (Highton 1956).

Biological information on the short-tailed snake has been summarized by Campbell and Moler (1992) and Ernst and Ernst (2003). Nothing is known regarding its reproduction or clutch size (FFWCC 2011). Its prey is mostly small, smooth-scaled snake species, particularly crowned snakes (*Tantilla relicta*) (Carr 1934, Mushinsky 1984, Campbell and Moler 1992, Rossi and Rossi 1993). The nonnative Brahminy blind snake (*Ramphotyphlops braminus*) provides an additional food source (Godley et al. 2008).

Population Status:

The short-tailed snake qualifies for endangered status because it is rare across its small range and is suffering ongoing and significant declines due to habitat loss from rapid development. NatureServe (2011) explains that the short-tailed snake is “rare even where it occurs” and that “area of occupancy, number of subpopulations, and population size likely have declined and probably continue to do so.” Declines are the result of habitat rapidly being altered for agricultural and urban development or other uses (NatureServe 2011, FFWCC 2011, FNAI undated, Scott 2004). The IUCN Red List ranks the species as Near Threatened but explains that it is close to qualifying for Vulnerable.

THREATS

Habitat alteration and destruction:

The short-tailed snake is primarily threatened by loss of habitat due to residential and agricultural development, mining, as well as clearcutting and other timber management programs in sand pine scrub (Campbell and Moler 1992, Hammerson 2007, NatureServe 2011). Its upland habitats are in great demand for development (FFWCC 2011). The high, dry woodlands preferred by the snake are also favored by humans for citrus production (Scott 2004).

To be sure, populations of xeric-adapted species have declined in Florida because of substantial habitat loss due to human activity (Enge et al. 2003). Sandhill habitat in 1987 covered only 2.4 percent of Florida, which represented an 88 percent loss of habitat since European settlement (Kautz et al. 1993). The remnant sandhills of south-central Florida are small, isolated, and faunally depauperate (Humphrey et al. 1985, Cox et al. 1994). In addition, only about one percent of Florida is now covered by scrub habitat, which has declined 59 percent since European settlement (Kautz et al. 1993).

The Florida Fish and Wildlife Conservation Commission (2011) describes the threats to its habitat as follows: "The greatest threat to short-tailed snakes is loss and alteration of xeric upland habitats resulting from commercial and residential development, silviculture, agriculture, and mining. Intact xerophytic upland ecosystems inhabited by short-tailed snakes have suffered severe losses in Florida, including longleaf pine-dominated sandhill as well as scrub habitat on the ridges of central Florida and the Gulf Coast of Florida (Means and Grow 1985, Myers 1990, Kautz 1998, Enge et al. 2003, Kautz et al. 2007)."

It appears that short-tailed snake populations can coexist with human development in some areas as long as some natural ground cover is retained (FFWCC 2011). However, it is unlikely that such habitat fragments could support populations in the longterm, especially considering the increased threat of road kill and malicious killing of the snakes in areas of human development. To be sure, highway mortality is likely a threat during periods of surface activity, and dead snakes have been found on driveways and roads (FFWCC 2011).

Overutilization:

As with all snakes, humans likely kill the snakes out of fear and maliciously, and these killings are contributing to population declines (FFWCC 2011). Interactions with humans are likely because the snake is found in residential areas as long as enough natural ground cover is maintained (FFWCC 2011). Some illegal collection for pets is also likely to occur.

Inadequacy of existing regulatory mechanisms:

The short-tailed snake is protected as threatened in Florida (FFWCC 2011b). This designation prohibits take but does not require protection of the snake's habitat.

Protection of substantial tracts of xeric habitat with element occurrences is essential (NatureServe 2011). In the absence of federal protection, habitat of the short-tailed snake is being rapidly lost. Most of the snake's potential habitat is privately owned (57 percent), and it inhabits upland habitats that are in great demand for development (FFWCC 2011).

This snake occurs in Ocala National Forest and in some small state parks, preserves, and geological sites (Wekiwa Springs, Ichetucknee Spring, San Felasco Hammock, Devil's Millhopper) (Hammerson 2007). But the snake is inadequately protected even on these lands from threats such as intensive timber management (Hammerson 2007). The condition of sandhill habitats on protected lands may improve in the future because of the Gopher Tortoise Management Plan (FWC 2007) and various projects to restore degraded sandhill and scrub habitats (FFWCC 2011). But these efforts are largely voluntary and occur on only a fraction of the snake's potential habitat.

Other factors:

Predation by red imported fire ants (*Solenopsis invicta*) has been suggested as a reason for declines in some oviparous snake populations in the Southeastern Coastal Plain (Mount 1981). Because of their fossorial nature and small size, short-tailed snakes would appear to be particularly susceptible to fire ants (FFWCC 2011). In addition, domestic cats and dogs are known predators (Highton 1956, Godley et al. 2008).

Some sandhills are threatened by invasion of the nonnative cogongrass (*Imperata* sp.), which forms dense stands that are probably unsuitable for many native plant and animal species, including the short-tailed snake (Simons 1990). Cogongrass is difficult to eradicate by mechanical means, herbicides, and fire; the highly flammable fuel loads of cogongrass likely eventually eliminates most of the fire-adapted woody plant species (Simons 1990).

The short-tailed snake is also likely threatened by isolation (Enge et al. 2003). Extensive habitat destruction has increasingly isolated scrubs so they cannot be recolonized if populations become extinct through some catastrophic event (Enge et al. 2003). Populations that appear to exist as stable metapopulations in a patchy landscape may actually be on their way to extinction because a large population is sustaining smaller populations, and insufficient movements occur between patches for patch recolonization to exceed patch extinction over long periods of time (Harrison 1994).

References:

Ashton, R. E., Jr., and P.S. Ashton. 1981. Handbook of reptiles and amphibians of Florida. Part one: the snakes. Windward, Miami, Florida, USA. 176pp.

Campbell, H.W., and S.P. Christman. 1982. The herpetological components of Florida sandhill and sand pine scrub associations. Pages 163-171 in N. J. Scott (editor). Herpetological communities. U.S. Fish and Wildlife Service, Wildlife Res. Rep. 13.

Campbell, H.W., and P.E. Moler. 1992. Short-tailed snake, *Stilosoma extenuatum* Brown. Pages 150–153 *in* P. E. Moler, editor. Rare and endangered biota of Florida. Volume III. Amphibians and reptiles. University Press of Florida, Gainesville, Florida, USA.

Carr, A.F., Jr. 1934. Notes on the habits of the short-tailed snake, *Stilosoma extenuatum* Brown. Copeia 1934: 138–139.

Carr, A.F., Jr. 1940. A contribution to the herpetology of Florida. University of Florida Publications, Biological Sciences 3:1–118.

Carr, A., and C.J. Goin. 1955. Guide to the reptiles, amphibians and fresh-water fishes of Florida. Univ. Florida Press, Gainesville.

Collins, J. T., and T. W. Taggart. 2009. Standard common and current scientific names for North American amphibians, turtles, reptiles, and crocodilians. Sixth edition. The Center for North American Herpetology, Lawrance, Kansas. iv + 44 pp.

Cox, J., R. Kautz, M. MacLaughlin, and T. Gilbert. 1994. Closing the gaps in Florida's wildlife habitat conservation system. Florida Game and Fresh Water Fish Commission Office of Environmental Services, Tallahassee, Florida, USA.

Dowling, H.G., and L.R. Maxson. 1990. Genetic and taxonomic relations of the short-tailed snakes, genus *Stilosoma*. Journal of Zoology, London 221(1): 77–86.

Enge, K.M. 1997. A standardized protocol for drift-fence surveys. Florida Game and Fresh Water Fish Commission Technical Report No. 14, Tallahassee, Florida, USA. 68pp.

Enge, K.M., B.A. Millsap, T.J. Doonan, J.A. Gore, N.J. Douglass, and G.L. Sprandel. 2003. Conservation plans for biotic regions in Florida containing multiple rare or declining wildlife taxa. Florida Fish and Wildlife Conservation Commission, Bureau of Wildlife Diversity Conservation Final Report, Tallahassee, Florida, USA. 146pp.

Ernst, C.H., and E.M. Ernst. 2003. Snakes of the United States and Canada. Smithsonian Books, Washington, D.C., USA. 668pp.

Florida Fish and Wildlife Conservation Commission (FFWCC). 2011. Short-tailed Snake Biological Status Review Report, *available at* http://myfwc.com/media/1351634/Short-tailed%20Snake%20Final%20BSR.pdf (last visited Oct. 4, 2011).

Florida Fish and Wildlife Conservation Commission (FFWCC). 2011b. Florida's Endangered and Threatened Species, *available at* http://myfwc.com/media/1515251/Threatened_Endangered_Species.pdf (last visited Oct. 4, 2011).

Florida Natural Areas Inventory (FNAI). Undated. Field guide to the rare animals of Florida: Short-tailed snake, *available at* http://www.fnai.org/FieldGuide/pdf/Stilosoma_extenuatum.PDF (last visited Oct. 6, 2011).

Franz, R., C.K. Dodd, Jr., and A.M. Bard. 1992. The non-marine herpetofauna of Egmon Key, Hillsborough County, Florida. Florida Scientist 55(3): 179-83.

FWC. 2007. Gopher tortoise management plan, *Gopherus polyphemus*. Florida Fish and Wildlife Conservation Commission, Tallahassee, Florida, USA. 127pp.

Godley, J.S., S.M. Gonzalez, and M.C. Gonzalez. 2008. *Stilosoma extenuatum* (short-tailed snake). Diet and predation. Herpetological Review 39: 473–474.

Hammerson, G.A. 2007. *Stilosoma extenuatum*. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.1. <www.iucnredlist.org>. Downloaded on 06 October 2011.

Harrison, S. 1994. Metapopulations and conservation. Pages 111–128 in P. J. Edwards, R. M. May, and N. R. Webb, editors. Large-scale ecology and conservation biology. Blackwell Scientific, Oxford, United Kingdom.

Highton, R. 1956. Systematics and variation of the endemic Florida snake genus *Stilosoma*. Bulletin of the Florida State Museum, Biological Sciences 1: 73–96.

Humphrey, S. R, J. F. Eisenberg, and R. L. Franz. 1985. Possibilities for restoring wildlife of a longleaf pine savanna in an abandoned citrus grove. Wildlife Society Bulletin 13: 487–496.

Kautz, R.S. 1998. Land use and land cover trends in Florida 1936–1995. Florida Scientist 61: 171–187.

Kautz, R. S., D. T. Gilbert, and G. M. Mauldin. 1993. Vegetative cover in Florida based on 1985–1989 Landsat Thematic Mapper Imagery. Florida Scientist 56: 135–154.

Kautz, R., B. Stys, and R. Kawula. 2007. Florida vegetation 2003 and land use change between 1985–89 and 2003. Florida Scientist 70: 12–23.

Means, D.B., and G.O. Grow. 1985. The endangered longleaf pine community. ENFO (Environmental Information Center of the Florida Conservation Foundation) Report (September): 1–12.

Mount, R.H. 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. Journal of the Alabama Academy of Science 52: 71–78.

Mushinsky, H.R. 1984. Observations on the feeding habits of the short-tailed snake, *Stilosoma extenuatum* in captivity. Herpetological Review 15: 67–68.

Myers, R.L. 1990. Scrub and high pine. Pages 150–193 *in* R. L. Myers and J. J. Ewel, editors. Ecosystems of Florida. University of Central Florida Press, Orlando, Florida, USA.

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, *available at* http://www.natureserve.org/explorer (last visited Oct. 4, 2011).

Pyron, R.A., and F.T. Burbrink. 2009. Neogene diversification and taxonomic stability in the snake tribe Lampropeltini (Serpentes: Colubridae). Molecular Phylogenetics and Evolution 52: 524–529.

Rossi, J.V., and R. Rossi. 1993. Notes on the captive maintenance and feeding behavior of a juvenile short-tailed snake (*Stilosoma extenuatum*). Herpetological Review 24: 100–101.

Scott, C. 2004. Endangered and Threatened Animals of Florida and their Habitats. Austin, Texas: University of Texas Press.

Simons, R. W. 1990. Terrestrial and freshwater habitats. Pages 99–157 in S. H. Wolfe, editor. An ecological characterization of the Florida Springs Coast: Pithlachascotee to Waccasassa rivers. U.S. Fish and Wildlife Service Biological Report 90(21).

Trescott, E. 1998. The short-tailed snake (*Stilosoma extenuatum*). Reptiles Magazine 6(3): 32–37.

Van Duyn, G. 1939. Extension in range of *Stilosoma extenuatum*. Copeia 1939: 51–52.

Woolfenden, G.E. 1962. A range extension and subspecific relations of the short-tailed snake, *Stilosoma extenuatum*. Copeia 1962: 648–649.

Scientific Name:

Heterodon simus

Common Name:

Southern Hog-nosed Snake or Southern Hognose Snake

G Rank:

G2

IUCN Red List:

Vulnerable

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

The southern hog-nosed snake occurs on the Coastal Plain from eastern North Carolina to southern Florida (Lake Okeechobee), west to southeastern Mississippi (Conant and Collins 1991, Palmer and Braswell 1995, Tennant 1997, Ernst and Ernst 2003). It is now very rare (or possibly extirpated) in the western part of the range in Mississippi and Alabama (NatureServe 2011, Alabama Dept. of Conservation and Natural Resources 2011). No recent (1983-1998) records are available for much of the historical range within Georgia (NatureServe 2011). Across its range, the snake's habitat is fragmented and threatened with development.

Habitat:

According to NatureServe (2011), the southern hog-nosed snake inhabits open, xeric habitats with well-drained, sandy or sandy-loam soils such as sand ridges, stabilized coastal sand dunes, pine flatwoods, mixed oak-pine woodlands and forests, scrub oak woods, and oak hammocks, as well as old fields and river floodplains (Ashton and Ashton 1981, Palmer and Braswell 1995, Tennant 1997, Ernst and Ernst 2003). This snake spends considerable time burrowed in the soil (NatureServe 2011).

Biology:

Edgren (1955) provides information on the natural history of the genus *Heterodon* but little is known about the southern hog-nosed snake in particular. Price and Carr (1943) report a clutch of 6 laid by a captive Florida female. Carr (1940) states that toads are the chief item of diet. Goin (1947) described a southern hog-nosed snake about 10 inches in length digging out and eating a spadefoot toad (*Scaphiopus h. holbrooki*).

Population Status:

The southern hog-nosed snake qualifies for endangered species status as it is now rare because it has suffered significant population declines and extirpations across its range and continues to be threatened by loss of habitat. There has been a substantial decline throughout the range (Tuberville et al. 2000, Hammerson 2007, NatureServe 2011). This snake appears to be declining in area of occupancy, number of subpopulations, and abundance (Tuberville et al. 2000). The species was a candidate for federal protection until the FWS eliminated the C2 category, but it currently receives no federal protection under the ESA.

This species is represented by a large number of collection sites, but the species appears to be no longer extant at many of these (Tuberville et al. 2000). No recent (1983-1998) records are available for Mississippi, Alabama, and much of the historical range within Georgia (NatureServe 2011). The species was considered quite common in southwestern Georgia in the 1940s and 1950s, but only a few occurrences have been noted in this area since 1970 (Tuberville et al. 2000). In South Carolina, the species has been seen since 1980 in only 10 of the 20 counties from which it was once known (NatureServe 2011). Palmer and Braswell (1995) mapped several dozen collection sites in North Carolina, but it is unknown whether these are still extent. The Florida Museum of Natural History (2011) considers the snake uncommon to rare in Florida.

THREATS

Habitat alteration and destruction:

As with most species restricted to the Coastal Plain, the southern hog-nosed snake is threatened by habitat lost due to intensive agricultural/silvicultural activities (Hammerson 2007). In Florida, the principle cause of decline has been attributed to the extensive loss of xeric upland habitat (Jackson and Printiss 2000, FNAI 2001, Enge et al. 2003).

This species has been found in some areas of fragmented and altered upland habitats, although cumulative road mortality is likely a significant factor, especially for hatchlings (Enge and Wood 2003). It is unknown whether the snake can persist longterm in such fragments. As Enge et al. (2003) explains: "Landscape development and fragmentation potentially pose demographic, genetic, and environmental stochasticity threats to xeric-adapted taxa. Land clearing has fragmented and isolated sandhill habitat, reducing the size and increasing the distance between remaining patches of habitat until they no longer can support viable populations of some species . . . . Even species with smaller home ranges, such as the southern hognose snake and short-tailed snake, may be experiencing population declines due to habitat destruction or degradation and road mortality (Tuberville et al. 2000)."

Overutilization:

Mortality from human persecution is likely contributing to population declines (Hammerson 2007). The snake is prized by the pet industry (Nickel 2010). Collection for the pet trade is documented in Florida and is likely a threat (Enge and Wood 2003).

Inadequacy of existing regulatory mechanisms:

The USFWS considers the southern hognose snake a "species of concern" but this status does not afford any protections. The southern hog-nosed snake is state listed as endangered in Mississippi (Mississippi Natural Heritage Program 2002), but not elsewhere in its range (Tuberville 2002, NCWRC 2008, SCDNR 2010, FFWCC 2011). Because it is not state listed in Florida, it may be taken throughout the year in any manner without limit (Nanjappa and Conrad 2012). The snake is protected from take as a nongame species in Georgia and Alabama. Ala. Admin. Code r. 220-2-.92; O.C.G.A. § 27-1-28. These state laws do nothing to protect habitats for the southern hog-nosed snake, which is the key to preventing extinction of this snake. Federal protection would prohibit take across the range.

Some southern hog-nosed snakes are in protected areas (Hammerson 2007). For example, it is present on some large state and national forests as well as Eglin Air Force Base in Florida (FNAI 2001). But even on protected habitats, threats such as intensive timber management, road mortality, and imported fire ants can seriously impact the snakes. Indeed, this snake appears to have disappeared in some large protected areas with relatively pristine habitats (Gibbons et al. 2000). Meegan (2002) reports that after sixty hours and 1,056 miles of road cruising from 2001 September to 2002 December, as well as the operation of two fifty-foot drift fences for one month during prime southern hognose movement period, no southern hog-nosed snakes were captured in Apalachicola National Forest in northern Florida.

Other factors:

It is likely that widespread pesticide application is harming the snakes (Hammerson 2007). In addition, predation of eggs and hatchlings by red imported fire ants is likely a factor in the decline (Tuberville et al. 2000). The snake's disappearance from certain areas is associated with heavy red imported fire ant infestations (Hammerson 2007, NatureServe 2011).

References:

Alabama Dept. of Conservation and Natural Resources. 2011. Southern hog-nosed snake, *available at* http://www.outdooralabama.com/watchable-wildlife/what/Reptiles/Snakes/shns.cfm (last visited Oct. 6, 2011).

Ashton, R.E., Jr., and P.S. Ashton. 1981. Handbook of reptiles and amphibians of Florida. Part One: The Snakes. Windward Publishing Company, Miami, Florida. 176 pp.

Carr, A. F., Jr. 1940. A contribution to the herpetology of Florida. Univ. Florida Publ. 3(1): 1-118.

Conant, R. and J.T. Collins. 1991. A field guide to reptiles and amphibians: eastern and central North America. Third edition. Houghton Mifflin Co., Boston, Massachusetts. 450 pp.

Edgren, R.A. 1955. The natural history of the hog-nosed snakes, genus Heterodon: a review. Herpetologica 11: 105-117.

Enge, K.M., and K.N. Wood. 2002. A pedestrian road survey of an upland snake community in Florida. Southeastern Naturalist 1: 365-380.

Enge, K.M., B.A. Millsap, T.J. Doonan, J.A. Gore, N.J. Douglass, and G.L. Sprandel. 2003. Conservation plans for biotic regions in Florida containing multiple rare or declining wildlife taxa. Florida Fish and Wildlife Conservation Commission, Bureau of Wildlife Diversity Conservation Final Report, Tallahassee, Florida, USA. 146pp.

Ernst, C.H., and E.M. Ernst. 2003. Snakes of the United States and Canada. Smithsonian Books, Washington, D.C.

Florida Fish and Wildlife Conservation Commission (FFWCC). 2011-12. Florida's Hunting Regulations, *available at* http://www.eregulations.com/florida/hunting/nongame-wildlife-regulations/ (last visited Oct. 6, 2011).

Florida Fish and Wildlife Conservation Commission (FFWCC). 2011b. Florida's Endangered and Threatened Species, *available at* http://myfwc.com/media/1515251/Threatened_Endangered_Species.pdf (last visited Oct. 4, 2011).

Florida Museum of Natural History. 2011. Southern hog-nosed snake, *available at* http://www.flmnh.ufl.edu/herpetology/fl-guide/heterodonsimus.htm (last visited October 6, 2011).

Florida Natural Areas Inventory. 2001. Southern hog-nosed snake, *available at* http://www.fnai.org/FieldGuide/pdf/Heterodon_simus.pdf (last visited Oct. 6, 2011).

Gibbons, J.W., D.E. Scott, T.J. Ryan, K.A. Buhlman, T.D. Tuberville, B.S. Mettis, J.L. Greene, T. Mills, Y. Leiden, S. Poppy, and C.T. Winne. 2000. The global decline of reptiles, déjà vu amphibians. BioScience. 50(8): 653-666.

Goin, C. J. 1947. A note on the food of *Heterodon sirus*. Copeia 1947(4): 275.

Hammerson, G.A. 2007. *Heterodon simus*. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.1. <www.iucnredlist.org>. Downloaded on 06 October 2011.

Jackson, D.R. and D. Printiss. 2000. Status of habitat at known localities of the southern hognose snake in Florida. Report to the U.S. Fish and Wildlife Service. Florida Natural Areas Inventory, Tallahassee, FL.

Mississippi Natural Heritage Program. 2002. Endangered Species of Mississippi. Museum of Natural Science, Mississippi Dept. of Wildlife, Fisheries, and Parks, Jackson, MS. 2 pp.

Nanjappa, P. and P.M. Conrad. 2012. State of the Union: Legal Authority Over the Use of Native Amphibians and Reptiles in the United States, *available at* http://www.fishwildlife.org/files/SOU_FULL-lo-res.pdf.

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, *available at* http://www.natureserve.org/explorer (last visited Oct. 4, 2011).

Nickel, N. 2010. Keeping and Breeding the Southern Hognose Snake. Reptilia 68: 25-35.

North Carolina Wildlife Resources Commission (NCWRC). 2008. Protected Wildlife Species of North Carolina, *available at* http://www.ncwildlife.org/Portals/0/Conserving/documents/protected_species.pdf (last visited Dec. 28, 2010).

Palmer, W.M., and A.L. Braswell. 1995. Reptiles of North Carolina. North Carolina State Museum of Natural Sciences, University of North Carolina Press, Chapel Hill, North Carolina.

Price, W. H. and L. G. Carr. 1943. Eggs of *Heterodon simus*. Copeia 1943(3): 193.

South Carolina Department of Natural Resources (SCDNR). 2010. SC Rare, Threatened and Endangered Species Inventory, *available at* http://www.dnr.sc.gov/species/index.html (last visited Dec. 28, 2010).

Tennant, A. 1997. A field guide to snakes of Florida. Gulf Publishing Company, Houston, Texas. xiii + 257 pp.

Tuberville, T.D. 2002. Upland Snake Species Profile: Southern Hognose Snake, *available at* http://www.gophertortoisecouncil.org/dox/simus.pdf (last visited Oct. 6, 2011).

Tuberville, T.D., J.R. Bodie, J.B. Jensen, L. LaClaire, and J.W. Gibbons. 2000. Apparent decline of the southern hog-nosed snake, *Heterodon simus*. Journal of the Elisha Mitchell Scientific Society 116: 19-40.

U.S. Fish and Wildlife Service. 1996. Endangered and threatened wildlife and plants:review of plant and animal taxa that are candidates for listing as endangered or threatened species, proposed rule. Federal Register 61(40): 7596-7613.

Scientific Name:

Charina umbratica (or in the alternative, *Charina bottae umbratica*)

Common Name:

Southern Rubber Boa

G Rank:

G2

IUCN Red List:

Not reviewed because the IUCN considered it a subspecies of *Charina bottae*

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

The southern rubber boa's viability is threatened by its very restricted range where it faces a number of threats. Stewart et al. (2005) explains that the southern rubber boa is restricted to the San Bernardino and San Jacinto Mountains at elevations between about 1,540 m and 2,460 m (see also Stewart 1988, 1991). It is known from approximately 40 localities in the San Bernardinos (Stewart 1991) and eight in the San Jacintos (Keasler 1982, Loe 1985). Twenty-six of the San Bernardino localities occur in a 16-km-long strip of habitat in the Lake Arrowhead area between Twin Peaks on the west and Green Valley on the east (Stewart 1988, 1991).

Habitat:

Typical habitat for this snake is mixed conifer-oak forest or woodland dominated by two or more of the following species: Jeffrey pine (*Pinus jeffreyi*), yellow pine (*P. ponderosa*), sugar pine (*P. lambertiana*), incense cedar (*Calo-cedrus decurrens*), white fir (*Abies concolor*), and black oak (*Quercus kellog-gii*) (Stewart 1988). A relatively open canopy seems to be preferred, at least during the spring months (April and May), and rock outcrops evidently are important as hibernacula (Keasler 1982, Stewart 1988). In the Heaps Peak area near Lake Arrowhead, the boa occurs at rock outcrops in open areas characterized by mixed grasses and bracken fern (*Pteridium aquilinum*) together with variable numbers of shrubs and small trees (Hoyer and Stewart 2000a). In all habitat types, rock outcrops and surface materials (such as rocks, logs, and a well-developed litter/duff layer) are important habitat components because they provide cover and maintain soil moisture (Loe 1985).

Because it is semi-fossorial, primarily crepuscular or nocturnal, and highly secretive, its seasonal activity and habitat use are difficult to determine (Stewart et. al 2005). Boas are most easily found during the spring months at rock outcrops where scattered surface rocks and wood debris provide movable cover under which to search (Keasler 1982, Stewart 1988, Hoyer and Stewart 2000a). The snakes essentially disappear during the summer months, although they may emerge for surface activity on humid nights (Alten and Keasler 1978) or after a rain (Stewart pers. observ., as cited in Stewart et. al 2005). It is likely that some individuals remain at the rock outcrops during the summer and retreat deeper into crevices (Stewart pers. observ., as cited in Stewart et al 2005), but others apparently move into cooler, moister forest and riparian habitats (Loe 1985, Stewart 1988).

Biology and Taxonomy:

Rodriguez-Robles et al. (2001) used mtDNA data to examine phylogeography of *C. bottae* and concluded that "*C. b. umbratica* is a genetically cohesive, allopatric taxon that is morphologically diagnosable" (using a suite of traits) and that "it is an independent evolutionary unit that should be recognized as a distinct species, *Charina umbratica*." Crother et al. (2003) listed *C. umbratica* as a species whereas Stebbins (2003) mentioned the proposal but did not adopt the split. Crother (2008) and Collins and Taggart (2009) also recognize *Charina umbratica* as a distinct species.

The southern rubber boa is mainly crepuscular during warmer periods of spring, summer, and fall but has some nocturnal and diurnal activity (Morey and Basey 2002). The snake probably has little seasonal movement but may migrate short distances to and from suitable hibernacula at higher elevations (Morey and Basey 2002). It has not been observed aggressively defending territories in the wild (Morey and Basey 2002). Breeding occurs from April to June (Morey and Basey 2002). Young are born alive from late summer (Erwin 1964) to late November (Hudson 1957), and number of young is two to eight (Stebbins 1972). Adults and young may occasionally be taken by hawks and owls or by predatory mammals such as skunks and raccoons (Morey and Basey 2002). More information on diet, predators and other biological information is provided by Hoyer and Stewart (2000) and Sudkamp (2002).

Population Status:

The southern rubber boa needs ESA protection because it is very rare across its restricted range and is experiencing declines due to habitat loss and degradation. It is known only from a very small number of individuals and localities, and it is threatened by development and increased recreational use in forested areas where it occurs (NatureServe 2011). Stewart (1991) describes the snake as rare. During Keasler's 1981 and 1982 springtime searches, he found mountain kingsnakes (a Forest Service Sensitive Species) at a frequency about 10 times that of rubber boas. Factors that put the southern rubber boa at risk include that it seems to prefer flat productive areas that are prime for development and recreation and that it is sensitive to ground disturbance (USFS undated). The species was a candidate for federal protection until the FWS eliminated the C2 category, but it currently receives no federal protection under the ESA.

THREATS

Habitat alteration and destruction:

The southern rubber boa is declining due to habitat loss and degradation (NatureServe 2011). The greatest threats are resort development, off-road vehicle use, and logging/wood collecting (Stewart et al. 2005, NatureServe 2011). The removal of vegetation for fuel breaks and forest thinning projects adversely affects southern rubber boas and their habitat by removing cover and drying out the soil (USFS undated).

The majority of known rubber boa locations are on private lands. Private land capable of supporting rubber boas continues to develop at a rapid pace. Large amounts of fuels work are also taking place on private land. For example, most of the prime habitat in the 16-km strip near Lake Arrowhead is private land subject to residential and commercial development (Stewart 1988, 1991). In addition, Crestline to the Snow Valley Ski Area has long been considered the best southern rubber boa habitat in the San Bernardino Mountains. Forty-four percent of this area is private land subject to development (Stewart 1991). Twenty-seven of the 40 recorded rubber boa localities are in this area and 16 of the 27 are on private property.

Substantial areas of known or potential habitat are on San Bernardino National Forest lands (Loe 1985, Stewart 1991). Specifically, of all the known and potential habitat in the San Bernardino Mountains, roughly 81 percent is on public lands managed by the Forest Service. While these areas are protected from development, Forest Service-permitted activities, such as personal fuelwood harvesting, off-highway vehicle use, and commercial timber sales, increasingly degrade and fragment what once were large, contiguous tracts of rubber boa habitat (Loe 1985, Stewart 1991). In addition, fuels work is being done around the mountain communities and other developments in the San Bernardino and San Jacinto mountains. This is resulting in unprecedented ground disturbance. It has long been assumed that ground disturbance with heavy equipment was detrimental to the species (USFS undated).

Stewart (1991) estimated that most of the suitable southern rubber boa habitat on private lands would be lost in the next 20-40 years, and in a worse case scenario, most of the habitat that is heavily impacted by OHVs and fuelwood harvest could also be lost. In his opinion, if this happened, the resulting loss of 50-60 percent of the suitable habitat would endanger the San Bernardino Mountains southern rubber boa population.

Since the assessment by Stewart (1991), the San Bernardino National Forest has made some progress in controlling unauthorized OHV use and impacts from fuelwood harvest. A system of authorized OHV trails have been built and designated, and enforcement using State OHV funds has increased. Fuelwood gathering has been more strictly controlled and is now restricted to roads (USFS undated). While these changes likely benefit the snake, other activities harmful to the snake continue on federal and private lands, such as removal of vegetation for fuel breaks and forest thinning projects.

Overutilization:

As with all snakes, humans likely kill them out of fear or maliciously or collect them for pets.

Inadequacy of existing regulatory mechanisms:

The southern rubber boa is listed by the State of California as Threatened (CFG 2011). This status prohibits take of the species and provides some protection for habitat. However, its state threatened status has not prevented rapid development of its habitat. Federal ESA protection would add additional protections on federal lands through section 7 consultation and critical habitat designation and would allow for use of section 6 funds to recover the species.

The sourthern rubber boa is a Forest Service Region 5 Sensitive Species (Stephenson and Calcarone 1999). Although this status has resulted in some protections, Forest Service-permitted activities, such as personal fuelwood harvesting, off-highway vehicle use, and commercial timber sales, continue to degrade and fragment what once were large, contiguous tracts of rubber boa habitat (Loe 1985, Stewart 1991).

There are some enforceable standards for the snake on the San Bernardino National Forest. These include:

SBNF S4 - Where available, in suitable southern rubber boa habitat retain a minimum of nine down logs per acre (minimum 12 inches diameter and 180 total linear feet) except in Wildland/Urban Interface Defense Zones and fuelbreaks. Give preference to large diameter logs (Arrowhead, Big Bear, Big Bear Back Country, Front Country, Garner Valley, Idyllwild, Silverwood, San Gorgonio, and Santa Rosa and San Jacinto National Monument Places).

S11: When occupied or suitable habitat for a threatened, endangered, proposed, candidate or sensitive (TEPCS) species is present on an ongoing or proposed project site, consider species guidance documents (see Appendix H) to develop project-specific or activity-specific design criteria. This guidance is intended to provide a range of possible conservation measures that may be selectively applied during site-specific planning to avoid, minimize or mitigate negative long-term effects on threatened, endangered, proposed, candidate or sensitive species and habitat. Involve appropriate resource specialists in the identification of relevant design criteria. Include review of species guidance documents in fire suppression or other emergency actions when and to the extent practicable.

Standard S11 directs new projects to consider species guidance documents (Appendix H-Species Guidance Summary) to develop project-specific or activity-specific design criteria. Both the species account for the southern rubber boa and the San Bernardino National Forest Species Management Guide for the southern rubber boa are included in the species guidance documents (Loe 1985). The species management guide and the down log standard (San Bernardino National Forest, Part 2, S4) have been used for many years to guide management on the San Bernardino National Forest. The species management guide and down log standards were jointly developed with the California Department of Fish and Game and the Southern Rubber Boa Advisory Committee. The recently developed "Southern Rubber Boa Avoidance Measures for Removal of Dead, Dying and Diseased Trees" has been incorporated into the southern rubber boa species account and will be used to design projects.

While these standards will result in important protections for the snakes, they only apply in the San Bernardino National Forest, which makes up a minority of its habitat. Most of the known locations of the snake occur on private lands, which will not benefit from these protections. Moreover, these standards do not prohibit all practices on the forest that are threatening the snakes. For example, the standards do not prohibit logging but just seek to mitigate some logging impacts.

Other factors:

The combined factors of habitat degradation, fragmentation, and loss increase the likelihood of local extirpations and restrict gene flow, potentially leading to reduced genetic diversity and greater vulnerability to chance environmental catastrophes (Stewart et al. 2005). As such, isolation should be considered a threat.

References:

Alten, G.R., and G. Keasler. 1978. Southern rubber boa study, Mt. Pinos Ranger District, Los Padres National Forest. Report prepared for the U.S. Department of Agriculture, Forest Service, Los Padres National Forest, Goleta, California. 38 p.

California Dept. of Fish and Game. 2011. State and Federally Listed Endangered and Threatened Animals of California, *available at* http://www.dfg.ca.gov/biogeodata/cnddb/pdfs/TEAnimals.pdf (last visited Oct. 7, 2011).

California Dept. of Fish and Game. 2002. California Wildlife Habitat Relationships System, Life History Accout – Rubber Boa, *available at* http://www.dfg.ca.gov/biogeodata/cwhr/cawildlife.aspx (last visited Oct. 7, 2011).

Collins, J. T., and T. W. Taggart. 2009. Standard common and current scientific names for North American amphibians, turtles, reptiles, and crocodilians. Sixth edition. The Center for North American Herpetology, Lawrance, Kansas. iv + 44 pp.

Crother, B.I., J. Boundy, J.A. Campbell, K. de Quieroz, D. Frost, D.M. Green, R. Highton, J.B. Iverson, R.W. McDiarmid, P.A. Meylan, T.W. Reeder, M.E. Seidel, J.W. Sites, Jr., S.G. Tilley, and D.B. Wake. 2003. Scientific and standard English names of amphibians and reptiles of North America north of Mexico: update. Herpetological Review 34: 198-203.

Crother, B. I. (editor). 2008. Scientific and standard English names of amphibians and reptiles of North America north of Mexico, with comments regarding confidence in our understanding. Sixth edition. Society for the Study of Amphibians and Reptiles Herpetological Circular 37: 1-84.

Erwin, D. B. 1964. Some findings on newborn rubber boas, *Charina b. bottae*. Copeia 1964: 222-223.

Hoyer, R. F. and G.R. Stewart. 2000. Biology of the Rubber Boa (*Charina bottae*), with Emphasis on C. b. umbratica. Part I: Capture, Size, Sexual Dimorphism, and Reproduction. Journal of Herpetology. 34: 354-60.

Hudson, G. E. 1957. Late parturition in the rubber snake. Copeia 1957: 51-52.

Keasler, G.L. 1981. Rubber boa survey for the San Bernardino National Forest. San Bernardino National Forest, San Bernardino, CA. 26 p.

Keasler, G.L. 1982. Eastern San Bernardino Mountain southern rubber boa survey. Report prepared for the U.S. Department of Agriculture, Forest Service, San Bernardino National Forest, San Bernardino, California; 22 p. and 2 maps.

Loe, S.A. 1985. Habitat management guide for southern rubber boa *(Charina bottae umbratica)* on the San Bernardino National Forest. San Bernardino National Forest, San Bernardino, CA. 9 p

Morey, S. and H. Basey. 2002. Life history account for rubber boa, *available at* http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=19&ved=0CGsQFjAIOAo&url=http%3A%2F%2Fnrm.dfg.ca.gov%2FFileHandler.ashx%3FDocumentVersionID%3D17981&ei=_S45T8uaOI3kggeazP3nBQ&usg=AFQjCNG2ESIz-GnIcs3X3CHHswIapA85Cw (last visited Feb. 13, 2012).

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, *available at* http://www.natureserve.org/explorer (Accessed: February 13, 2012).

Rodriguez-Robles, J. A., G. R. Stewart, and T. J. Pappenfuss. 2001. Mitochondrial DNA-based phylogeography of North American rubber boas, *Charina bottae* (Serpentes: Boidae). Molecular Phylogenetics and Evolution 18: 227-237.

Stebbins, R. C. 1972. California amphibians and reptiles. Univ. California Press, Berkeley. 152pp.

Stebbins, R.C. 2003. A field guide to western reptiles and amphibian. 3rd ed. New York: Houghton Mifflin Company.

Stephenson, J.R.; Calcarone, G.M. 1999. Southern California mountains and foothills assessment: Habitat and species conservation issues. General Technical Report PSW-GTR-172.

Albany, CA: Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture.

Stewart, G.R. 1988. The rubber boa *(Charina bottae)* in California, with particular reference to the southern subspecies, *C. b. umbratica.* In: H.F. DeLisle, P.R. Brown, B. Kaufman, and B.M. McGurty, editors. Proceedings of the Conference on California Herpetology. Southwestern Herpetologists Society: 131–138.

Stewart, G.R. 1990 and 1991. Status of the southern rubber boa (Charina bottae umbratica) in the San Bernardino Mountains of southern California. Report Submitted to U.S. Fish and Wildlife Service, Laguna Niguel Field Office. Contract number 10120-87-00342. October 1, 1990, Revised March 1, 1991.

Stewart, G.R., M.R. Jennings, and R.H. Goodman, Jr.. 2005. Sensitive Species of Snakes, Frogs, and Salamanders in Southern California Conifer Forest Areas: Status and Management. USDA Forest Service Gen. Tech. Rep. PSW-GTR-195, *available at* https://www.google.com/url?sa=t&source=web&cd=26&ved=0CEEQFjAFOBQ&url=http%3A%2F%2Fwww.fs.fed.us%2Fpsw%2Fpublications%2Fdocuments%2Fpsw_gtr195%2Fpsw_gtr195_2_11_Stewart.pdf&rct=j&q=%22Southern%20Rubber%20Boa%22&ei=JwePTqOgK4b9sQLF0fXMAQ&usg=AFQjCNGAxNwPJ_G7Er3LifCmrpT9lD6TWQ&cad=rja (last visited Oct. 7, 2011).

Sudkamp, A. 2002. A Natural History of the Rubber Boa, Charina bottae, *available at* pages.uoregon.edu/titus/herp_old/andy.htm (last visited Feb. 13, 2012).

U.S. Forest Service. Undated. Species account for southern rubber boa, *available at* http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=29&ved=0CGcQFjAIOBQ&url=http%3A%2F%2Fwww.lpfw.org%2Fdocs%2Fspecies%2FSouthernRubberBoa.pdf&ei=gS85T5vmEtDdgQef76zqBQ&usg=AFQjCNFP8_oCR_IXYRe8yjdPr_K_K-pKVw (last visited Feb. 13, 2012).

TOADS

Western Spadefoot (*Spea hammondii*)

Arizona Toad (*Anaxyrus microscaphus*)

Scientific Name:

Spea hammondii (also known as *Scaphiopus hammondii*)

Common Name:

Western Spadefoot

G Rank:

G3

IUCN Red List:

Near Threatened

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

The range of the western spadefoot includes the Central Valley and bordering foothills of California and the Coast Ranges (south of San Francisco Bay) and extends southward into northwestern Baja California, Mexico (NatureServe 2011, Santos-Barrera et al. 2004). The species has been extirpated throughout much of lowland southern California (NatureServe 2011). Elevational range extends from near sea level to elevations of up to about 1,363 m (Ervin et al. 2001), but usually below 910 m (Stebbins 1985).

Habitat:

The western spadefoot lives in a wide range of habitats: lowlands to foothills, grasslands, open chaparral, pine-oak woodlands, but it prefers shortgrass plains, sandy or gravelly soil (e.g., alkali flats, washes, alluvial fans) (Santos-Barrera et al. 2004, NatureServe 2011). It is fossorial and breeds in temporary rain pools and slow-moving streams (NatureServe 2011, Santos-Barrera et al. 2004). It also breeds in stock tanks and other artificial water bodies as long as the surrounding habitat is not developed for human settlement or irrigated agriculture (Santos-Barrera et al. 2004).

Taxonomy and Biology:

The western spadefoot is placed in the genus *Scaphiophus* by some authors (NatureServe 2011). Hall (1998) argued against the recognition of *Spea* as a distinct genus, but most authors have accepted the split of *Spea* from *Scaphiophus* (Collins and Taggart 2009, NatureServe 2011).

Figure 14. Spadefoot toads get their name from the spade on each hind foot that the toads use for digging.

Few movements of western spadefoots occur during most of the year, but they will travel up to several meters on rainy nights (Morey 2000). Movements to and from breeding ponds are rarely extensive (Morey 2000). The western spadefoot is not territorial during most of the year, but aggressive encounters between calling males at a breeding site suggest a degree of territoriality (Whitford 1967). Breeding and egg laying normally occur from late winter to the end of March (Morey 2000). Females lay numerous small, irregular clusters containing 10 to 42 eggs, and eggs hatch rapidly, normally within two weeks (Morey 2000). Additional information on the biology of the species is summarized by Morey (2000) and Lannoo (2005).

Population Status:

The western spadefoot qualifies for endangered status because it is experiencing rapid and significant population declines due to habitat destruction. To be sure, since the 1950s, drastic declines have been noted in the Central Valley and in southern California (Santos-Barrera et al. 2004, NatureServe 2011). The toad is extirpated across much of southern California, and more than 80 percent of the previously occupied habitat has been developed or converted to incompatible uses (Santos-Barrera et al. 2004). The species is lost from more than 30 percent of its habitat in northern and central California (Jennings and Hayes 1994). Western spadefoot toads formerly were widespread but apparently are now absent on the Los Angeles coastal plain

(Jennings and Hayes 1994) and much of lowland southern California. Habitat conversion has also caused losses in the Great Valley and its associated foothills (Jennings and Hayes 1994, Fisher and Shaffer 1996). The IUCN Red List ranks the species as Near Threatened but explains that it is close to qualifying for Vulnerable (Santos-Barrera et al. 2004).

THREATS

Habitat alteration and destruction:

Habitat loss and degradation due to urban and agricultural development is the primary threat to the survival of the western spadefoot (Fisher and Shaffer 1996, Davidson et al. 2002). Much of the spadefoot's range has already been lost, and habitat loss is expected to continue (Lannoo 2005).

Agriculture threatens the toad's habitat in several ways. Irrigation agriculture destroys the toad's terrestrial habitat and can change the hydroperiod of temporary breeding pools. Farming practices such as disking and livestock trampling also degrade and destroy the toad's habitat.

Urban development threatens the toad in numerous ways, including through habitat fragmentation and direct mortality (Jennings and Hayes 1994, Stebbins 2003). Morey and Guinn (1992) found both juvenile and adult toads on rainy nights on roads intersecting a vernal pool complex. Spadefoot toad congregation on roads undoubtedly leads to mortality from vehicle strikes and can contribute to genetic isolation.

A specific threat to the toad's remaining habitat is the Madera Irrigation District (MID) Water Supply Enhancement Project (WSEP), west of the city of Madera, in Madera County, California (EPA 2009). The project encompasses an area of 128,292 acres and delivers water to its service area as part of the Hidden Unit (Fresno River) and Friant Division (San Joaquin River) long-term water supply contracts with the Bureau of Reclamation. The project would permanently remove vernal pools and harm the western spadefoot (EPA 2009). A record of decision approving the project was released in the summer of 2011 (Bureau of Reclamation 2011).

Disease or predation:

Disease has not been identified as a major threat to spadefoot toads at this time. Goldburg and Bursey (2002) observed western spadefoots infected with trematodes. In the wild, these infections apparently do not lead to significant disease (Tinsley 1995), although they represent a potential threat, especially in conjunction with other stressors.

Inadequacy of existing regulatory mechanisms:

Western spadefoots are listed as a Species of Special Concern in California (Jennings and Hayes 1994, CDFG 2011). This status recognizes that the toads are suffering drastic declines but does not afford any legal protection.

The toads are found on some BLM lands (Santos-Barrera et al. 2004), where they are recognized as a sensitive species (BLM 2006). But a sensitive species designation only requires analysis of the impacts of management actions and does not prevent actions that might harm the toads.

The habitat of the toads is protected on a few small Nature Conservancy preserves (Santos-Barrera et al. 2004), but just a fraction of the toad's available habitat enjoys such protection, which is not enough to ensure the survival of the toads.

The species is also found on some U.S. Department of Defense and Department of Energy lands, National Monuments, and National Wildlife Refuges (Santos-Barrera et al. 2004). It also occurs within the University of California's Natural Reserve System (Santos-Barrera et al. 2004). Although these lands are protected from the threats of urbanization and agriculture, even on these lands the toads can be impacted from other threats such as roads and introduced predators.

This species is also covered in some federal Habitat Conservation Plans (FWS 2011). But again, these plans cover just a fraction of the toad's habitat. Federal listing would ensure that all essential habitats are adequately protected.

Other factors:

Recruitment is likely unsuccessful in pools with bullfrogs (*Rana catesbeiana*) or introduced fish (for example, those containing mosquitofish (*Gambusia*) historically used for mosquito abatement) (Santos-Barrera et al. 2004). As such, most successful breeding is limited to ephemeral pools that do not support many introduced predators and competitors (Lannoo 2005). Yet even within ephemeral pools the toads are often threatened because of abundant exotic tiger salamanders (*Ambystoma tigrinum*), which are established in some California localities and can breed in ephemeral pools (Lannoo 2005). Larval tiger salamanders likely prey upon larval western spadefoot toads (Lannoo 2005).

Climate change could impact the toads by causing ephemeral pools to dry up faster. Spadefoot toads are of great interest to scientists because of their ability to accelerate metamorphosis when subject to food and water volume reduction (Boorse and Denver 2004). When a pond begins to dry-up, the tadpoles develop faster. However, at least 30 days are required for successful metamorphosis (Morey 1998).

References:

Boorse, G. C. and R. J. Denver. 2004. Endocrine Mechanisms Underlying Plasticity in Metamorphic Timing in Spadefoot Toads.

Bureau of Land Management (BLM). 2006. California-BLM Animal Sensitive Species List, *available at* http://www.blm.gov/ca/pdfs/pa_pdfs/biology_pdfs/SensitiveAnimals.pdf (last visited Oct. 13, 2011).

Bureau of Reclamation. 2011. Reclamation Issues a Record of Decision for the Madera Irrigation District Water Supply Enhancement Project, *available at* http://www.usbr.gov/newsroom/newsrelease/detail.cfm?RecordID=37025 (last visited May 7, 2012).

California Dept. of Fish and Game. 2011. Special Animals List, *available at* http://www.dfg.ca.gov/biogeodata/cnddb/pdfs/SPAnimals.pdf (last visited Oct. 13, 2011).

Davidson, C., H. B. Shaffer, and M. R. Jennings. 2002. Spatial tests of the pesticide drift, habitat destruction, UV-B, and climate-change hypotheses for California amphibian declines. Conservation Biology 16: 1588-1601.

Ervin, E.E. and R.N. Fisher. 2001. Natural history notes: *Thamnophis hammondii* (two-striped garter snake). Prey. Herpetological Review. 32: 265–266.

Ervin, E. L., A. E. Anderson, T. L. Cass, and R. E. Murcia. 2001. *Spea hammondii*. Elevation record. Herpetological Review 32: 36.

Feaver, P.E. 1971. Breeding pool selection and larval mortality of three California amphibians: *Ambystoma tigrinum californiense* Gray, *Hyla regilla* Baird and Girard, and *Scaphiopus hammondi hammondi* Girard. Master's thesis. Fresno State College, Fresno, California.

Fisher, R.N. and H.B. Shaffer. 1996. The decline of amphibians in California's Great Central Valley. Conservation Biology 10: 1387–1397.

Goldberg, S.R. and C.R. Bursey. 2002. Helminths of the Plains Spadefoot, Spea bombifrons, the western spadefoot, Spea hammodii, and the Great Basin spadefoot, Spea intermontana (Pelobatidae). Western North American Naturalist 62(4): 491–495.

Hall, J.A. 1998. *Scaphiopus intermontanus*. Catalogue of American Amphibians and Reptiles. 650: 1-17.

Hayes, M.P. and J. Warner. 1985. Life history notes: *Rana catesbeiana* (bullfrog). Food. Herpetological Review 16: 109.

Jennings, M. R., and M. P. Hayes. 1994. Amphibian and reptile species of special concern in California. Final Report submitted to the California Department of Fish and Game, Inland Fisheries Division. Contract No. 8023. 255 pp.

Lannoo, M. (ed.). 2005. Amphibian Declines: The Conservation Status of United States Species. University of California Press.

Morey, S. R. 1998. Pool Duration Influences Age and Body Mass at Metamorphosis in the Western Spadefoot Toad: Implications for Vernal Pool Conservation. In C. W. Witham, E. T. Bauder, D. Belk, W. R. Ferren Jr., and R. Ornduff (Editors). Ecology, Conservation, and Management of Vernal Pool Ecosystems – Proceedings from a 1996 Conference. California Native Plant Society, Sacramento, CA.

Morey, S. 2000. Life history account for Western Spadefoot, *available at* https://www.google.com/url?sa=t&source=web&cd=40&ved=0CFYQFjAJOB4&url=http%3A%2F%2Fnrm.dfg.ca.gov%2FFileHandler.ashx%3FDocumentVersionID%3D18116&rct=j&q=%22Spea%20hammondii%22&ei=lQuXTvU6sKawApfbjOcE&usg=AFQjCNFK2A0WYYlEz54gl624jGEJgU-3Iw&cad=rja (last visited Oct. 13, 2011).

Morey, S.R. and D.A. Guinn. 1992. Activity patterns, food habits, and changing abundance in a community of vernal pool amphibians. Pp. 149–158. *In* Williams, D.F., S. Byrne and T.A. Rado (Eds.), Endangered and Sensitive Species of the San Joaquin Valley, California: Their Biology, Management, and Conservation. California Energy Commission, and The Wildlife Society, Western Section, Sacramento, California.

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, available at http://www.natureserve.org/explorer (last visited Oct. 13, 2011).

Santos-Barrera, G., G. Hammerson, and S. Morey 2004. *Spea hammondii*. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.1. <www.iucnredlist.org>. Downloaded on 13 October 2011.

Stebbins, R. C. 1985. A field guide to western reptiles and amphibians. Second edition. Houghton Mifflin Company, Boston, Massachusetts. xiv + 336 pp.

Stebbins, R. C. 2003. A Field Guide to Western Reptiles and Amphibians 3rd Ed. Houghton Mifflin Company. New York, New York. pp. 205-6.

Tinsley, R.C. 1995. Parasitic disease in amphibians: control by the regulation of worm burdens. Parasitology 111: S153–S178.

U.S. Environmental Protection Agency. 2009. Madera Irrigation District Water Supply Enhancement Project, Madera County, California. 462 pp.

U.S. Fish and Wildlife Service (FWS). 2011. Western spadefoot (Scaphiopus hammondii), *available at* http://ecos.fws.gov/speciesProfile/profile/speciesProfile.action?spcode=D02Z (last visited Oct. 13, 2011).

Whitford, W. G. 1967. Observations on territoriality and aggressive behavior in the western spadefoot toad, Scaphiopus hammondi. Herpetologica 23: 318.

Scientific Name:

Anaxyrus microscaphus

Common Name:

Arizona Toad

G Rank:

G3

IUCN Red List:

Least Concern

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

The range of the Arizona Toad is highly fragmented (Clark 2011). It occurs mainly in Arizona, but also in southeastern Nevada and southwestern Utah (Sullivan 1993). The toad has a limited distribution in the southeastern portion of California along the Colorado River, though it has likely been extirpated from the California side of the river (Sullivan 1993). Distribution in New Mexico extends from the Arizona border into the southwestern quarter of the state (Behler and King 1979, Price and Sullivan 1988, Gergus 1998, Stebbins 2003, Elliott et al. 2009).

Habitat:

The toad can be found in the loose gravelly areas of streams in the more arid portions of its range. In the less arid portions of its range, it is often seen on sandy banks of quieter waters (Behler and King 1979, NatureServe 2011). It is found in both seasonal and permanent streams in the arid lowlands and is associated with the rocky mountain streams in oak-pine forests (Stebbins 2003, Elliott et al. 2009). The toad occurs along irrigation ditches, reservoirs, and in flooded fields, as well as along streams bordered by willows and cottonwoods (Stebbins 1954, Price and Sullivan 1988, Schwaner and Sullivan 2005).

Breeding habitat includes areas along the edges of streams, side-pools, and backwashes where flows are slow. It lays eggs among gravel, leaves, or sticks, or on mud or clean sand, at bottom of flowing or shallow quiet waters of perennial or semipermanent streams or shallow ponds (Dahl et al. 2000).

Biology:

Breeding begins in early to late February in Arizona, but in early to late March to early April in Utah or at higher elevations in Arizona (Blair 1955; Sullivan 1992b, 1995). Breeding is not triggered by rain but more likely by warming nocturnal, ambient temperatures (air, 8–18 ˚C, water 12–18 ˚C; Sullivan 1992b). Spring flooding delays breeding (Sullivan 1992b). Sullivan (1992b) observed female selection of calling males in Arizona toads. Additional information on the biology of the Arizona toad summarized by Frost and Mendelson (2008) and Lannoo (2005).

Population Status:

Endangered species protections are needed for the Arizona toad because it is absent from most of its historic range due to impoundments and hybridization. Specifically, the Arizona toad is likely absent from 75 percent of its historic range (CaliforniaHerps.com 2011). It is now absent from historic localities in Arizona where the riparian corridor has been altered dramatically through the construction of impoundments (Hammerson and Schwaner 2004). They are apparently being replaced by Woodhouse's toads (*B. woodhousii*) at some localities in Arizona (Sullivan 1993) and southern Nevada (Lannoo 2005). The species was a candidate for federal protection until the FWS eliminated the C2 category, but it currently receives no federal protection under the ESA.

THREATS

Habitat alteration and destruction:

The Arizona toad is now absent from historical localities where the riparian corridor has been altered dramatically through the construction of impoundments (Sullivan 1986, 1993). Damming along the Agua Fria River has reduced lotic habitats favored by Arizona toads for breeding and provided lentic environments favored by Woodhouse's toads (Lannoo 2005), which contributes to the severe threat of hybridization, as explained below.

Overutilization:

The toads are legally collected in some states but overcollection is not documented as a threat.

Disease or predation:

Predators of Arizona toads include birds, such as killdeer (*Charadrius vociferus*) and reptiles, such as wandering gartersnakes (*Thamnophis elegans vagrans*). The parotid glands produce steroids that likely make Arizona toads unpalatable to some predators (Duellman and Trueb 1986, Clark 2011). Human subsidized predators, such as raccoons, are a likely threat in some areas, and Schwaner and Sullivan (2005) observed that raccoons (*Procyon lotor*) commonly consume Arizona toads during the breeding season.

The gastrointestinal tracts, lungs, and urinary bladders from 77 Arizona toads, 61 Woodhouse's toads, and 8 of their hybrids were examined for helminthes by Goldberg et al. (1996) and several kinds were found. But these parasites do not appear to be having a significant impact on the species.

Inadequacy of existing regulatory mechanisms:

The Arizona toad can be found on lands managed by federal agencies, including the Bureau of Indian Affairs, BLM, and FWS. It is also found on Apache-Sitgreaves, Coconino, Prescott, and Tonto National Forests, and it is listed as a U.S. Forest Service Region 3 Sensitive Species (Frost and Mendelson 2008). While lands under federal ownership are protected from development, the toads remain vulnerable on these lands to the threat of hybridization. Also, a sensitive species designation does not prevent management actions that might harm the toads but only requires analysis of impacts.

The Arizona toad is not protected as endangered or threatened in Arizona, New Mexico, Utah, or California. In Arizona, an Arizona fishing license required to take open season amphibians (Frost and Mendelson 2008). In Arizona, the toads are found on some state-owned lands, including Alamo Wildlife Area and Page Springs Fish Hatchery, Alamo Lake State Park, and Lake Pleasant County Park (Frost and Mendelson 2008). It also is found on land protected by the Nature Conservancy, namely the Hassayampa River Preserve. Again, these lands may be protected from development but the threat of hybridization remains.

Other factors:

A major threat to the Arizona toad is hybridization with Woodhouse's toad (*B. woodhousii*), possibly facilitated by dam construction. Hybridization with the Woodhouse's toad has compromised the genetic integrity of Arizona toad populations, to a point where uncontaminated Arizona toad populations no longer occur in many areas (Sullivan 1986, Schwaner and Sullivan 2005).

Woodhouse's toad prefers aquatic areas with still or standing water (Stebbins 2003). Dams and reservoirs, as well as golf course ponds, irrigation sumps and canals, and other human-altered aquatic systems have created still water favored by Woodhouse's toads and facilitated niche overlap and hybridization between Arizona toads and Woodhouse's toads (Sullivan and Lamb 1988, Stebbins 2003). Lannoo (2005) summarizes evidence that effects of hybridization are not localized and perhaps behavioral as well as ecological, and that genetic barriers to introgression have broken down, and most, if not all, hybrid zones are in areas of human disturbance (dams and golf courses along tributaries and rivers).

Hybridization occurs wherever Arizona toads occur with Woodhouse's toads. Temporal separation of the two species by different breeding times was not evident (Sullivan 1995). Interspecific hybridization appears to be unidirectional with Woodhouse's toad females mated by Arizona toad males (Malmos et al. 2001). Crosses between the Woodhouse's toad and the Arizona toad have led to hermaphrodites (Sullivan et al. 1996).

Sullivan and Lamb (1988) presented evidence that Woodhouse's toad is displacing Arizona toads in some areas in central Arizona. The Arizona toad has been replaced by Woodhouse's toad at Alamo Lake, Lake Pleasant, the Verde Valley, and Fort Mohave (lower Colorado River). Bradford et al. (2005) report the complete replacement of the Arizona toad in Las Vegas Valley with Woodhouse's toad hybrids with predominantly Woodhouse's traits. Schwaner (2003) reported a hybrid swarm at a golf course near the junction of the Beaver Dam Wash with the Virgin River and genetic introgression 60 miles upstream. Sullivan (1993) presented a hybrid-index score useful in determining degree of hybridization of intergradation.

References:

Behler, J. L., and F. W. King. 1979. The Audubon Society field guide to North American reptiles and amphibians. Alfred A. Knopf, New York. 719 pp.

Blair, A. P. 1955. Distribution, variation, and hybridization in a relict toad (Bufo microscaphus) in southwestern Utah. American Museum Novitates 1722: 1-38.

Bradford, D.F., J.R. Jaeger, and S.A. Shanahan. 2005. Distributional changes and populations status of amphibians in the Eastern Mojave Desert. Western North American Naturalist 65(4): 462.

CaliforniaHerps.com. 2011. Anaxyrus microscaphus - Arizona Toad, *available at* http://www.californiaherps.com/frogs/pages/b.microscaphus.html (last visited Oct. 13, 2011).

Clark, H.O. Jr. 2011. Arizona Toad, *Anaxyrus microscaphus* (Cope "1866" 1867). Sonoran Herpetologist 24 (2).

Dahl, A., M.P. Donovan, and T.D. Schwaner. 2000. Egg mass deposited by Arizona toads, *Bufo microscaphus*, along a narrow canyon stream. Western North American Naturalist 60: 456–458.

Degenhardt, W. G., C. W. Painter, and A. H. Price. 1996. Amphibians and reptiles of New Mexico. University of New Mexico Press, Albuquerque. xix + 431 pp.

Duellman, W. E., and L. Trueb. 1986. Biology of Amphibians. McGraw-Hill Book Company, San Francisco, CA.

Elliott, L., C. Gerhardt, and C. Davidson. 2009. The Frogs and Toads of North America. A Comprehensive Guide to their Identification, Behavior, and Calls. Houghton Mifflin Harcourt, New York.

Frost, D. R., R. W. McDiarmid, and J. R. Mendelson III. 2008. Anura: Frogs. IN B. I. Crother (ed.), Scientific and Standard English Names of Amphibians and Reptiles of North America North of Mexico, pp. 2-12 SSAR Herpetological Circular 37, *available at* https://www.google.com/url?sa=t&source=web&cd=24&ved=0CC0QFjADOBQ&url=http%3A%2F%2Fwww.azgfd.gov%2Fw_c%2Fedits%2Fdocuments%2FAnaxmicr.fi.pdf&rct=j&q=Anaxyrus%20microscaphus&ei=iz2XTs-wIeH5sQKexpDkBA&usg=AFQjCNFiDHPfIuMoqISy1MCmOesLA3bVDQ&cad=rja (last visited Oct. 13, 2011).

Gergus, E.W.A. 1998. Systematics of the *Bufo microscaphus* complex: allozyme evidence. Herpetologica 54: 317-325.

Goldberg, S. R., C. R. Bursey, K. B. Malmos, B. K. Sullivan, and H. Cheam. 1996. Helminths of the Southwestern Toad, Bufo microscaphus, Woodhouse's Toad, Bufo woodhousii (Bufonidae), and their hybrids from central Arizona. Great Basin Naturalist 56: 369-374.

Hammerson, G. and T. Schwaner 2004. *Anaxyrus microscaphus*. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.1. <www.iucnredlist.org>. Downloaded on 13 October 2011.

Hovingh, P. 1997. Amphibians in the eastern Great Basin (Nevada and Utah, USA): a geographical study with paleozoological models and conservation implications. Herpetological Natural History 5: 97–134.

Lannoo, M. (ed.). 2005. Amphibian Declines: The Conservation Status of United States Species. University of California Press.

Malmos, K. B., B. K. Sullivan, and T. Lamb. 2001. Calling behavior and directional hybridization between two toads (*Bufo microscaphus* X *B. woodhousei*) in Arizona. Evolution. 55: 626-630.

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, available at http://www.natureserve.org/explorer (last visited Oct. 13, 2011).

Nigro, E. and J. Rorabaugh. 2008. Arizona toad *Anaxyrus microscaphus*, *available at* http://www.reptilesofaz.org/Turtle-Amphibs-Subpages/h-a-microscaphus.html (last visited Oct. 13, 2011).

Price, A. H., and B. K. Sullivan. 1988. *Bufo microscaphus*. Cat. Am. Amph. Rep. 415.1-415.3.

Schwaner, T.D 2003. Hybridization with Woodhouse's toad, Bufo woodhousii, and introgression in Arizona toads, Bufo microscaphus, along the Beaver Dam Wash in southwestern Utah. Georgia Journal of Science 61: 46.

Schwaner, T. D., and B. K. Sullivan. 2005. Bufo microscaphus Cope, 1867 “1866” Arizona Toad. Pages 422-424 in: M. Lannoo, editor. Amphibian Declines: The Conservation Status of United States Species. University of California Press, Berkeley.

Stebbins, R. C. 1954. Amphibians and reptiles of western North America. McGraw-Hill Book Company, New York.

Stebbins, R. C. 1985. A field guide to western reptiles and amphibians. Second edition. Houghton Mifflin Company, Boston, Massachusetts. xiv + 336 pp.

Stebbins, R. C. 2003. A field guide to western reptiles and amphibians. Third edition. Houghton Mifflin Company, Boston.

Sullivan, B. K. 1986. Hybridization between the toads Bufo microscaphus and Bufo woodhousei in Arizona: morphological variation. Journal of Herpetology 20: 11-21.

Sullivan, B.K. 1992b. Calling behavior in the southwestern toad (*Bufo microscaphus*). Herpetologica 48: 383–389.

Sullivan, B. K. 1993. Distribution of the southwestern toad (*Bufo microscaphus*) in Arizona. Great Basin Nat. 53: 402-406.

Sullivan, B.K. 1995. Temporal stability in hybridization between *Bufo microscaphus* and *Bufo woodhousii* (Anura: Bufonidae): behavior and morphology. Journal of Evolutionary Biology 8: 233–247.

Sullivan, B. K. and T. Lamb. 1988. Hybridization between the toads *Bufo microscaphus* and *Bufo woodhousii* in Arizona: variation in release calls and allozymes. Herpetologica 44: 325-333.

Sullivan, B. K., C. R. Propper, M. J. Demlong, and L. A. Harvey. 1996. Natural hermaphroditic toad (Bufo microscaphus × Bufo woodhousii). Copeia 1996: 470-472.

FROGS

Gopher Frog (*Lithobates capito*)

Illinois Chorus Frog (*Pseudacris illinoensis*)

Foothill Yellow-legged Frog (*Rana boylii*)

Cascades Frog (*Rana cascadae*)

Scientific Name:

Lithobates capito

Common Name:

Gopher Frog (previously known as Carolina Gopher Frog)

G Rank:

G3

IUCN Red List:

Near Threatened

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

The gopher frog is found in the Coastal Plain of the southeastern United States from the southern half of North Carolina (Beaufort County) to southern Florida (Collier County on the west coast, Broward County on the east coast), west to the Tombigbee River of Alabama (Hammerson and Jensen 2004). There are isolated populations in central Alabama (with a historical record from Shelby County) and central Tennessee (Atlig and Lohoefener 1983, Bailey 1991, Conant and Collins 1991, Godley 1992, Redmond and Scott 1996, Miller and Campbell 1996). The gopher frog historically occurred throughout Florida except for the Everglades region (FFGCC 2011), and Florida represents the largest portion of the total range of the species (Jensen and Richter 2005).

Habitat:

Its primary habitat is native xeric upland habitats, particularly longleaf pine-turkey oak sand hill associations (Hammerson and Jensen 2004). It also lives in xeric to mesic longleaf pine flat woods, sand pine scrub, xeric oak hammocks, and ruderal successional stages of these habitats (Hammerson and Jensen 2004). It is absent from most coastal islands and dunes (Godley 1992). The widespread loss of the longleaf pine ecosystem is a primary threat to the frog.

Burrows of the gopher tortoise or rodents are used for shelter (Gentry and Smith 1968, Lee 1968, Franz 1986), and the gopher frog also hides under logs and under or in stumps (Wright and Wright 1949).

Breeding occurs in ephemeral to semi-permanent graminoid-dominated wetlands that lack large predatory fish (Moler and Franz 1987, Bailey 1991). Gopher frogs have also been observed breeding in ditches and borrow pits (Means 1986).

The species needs fire to maintain the open-canopied habitats that it requires. In Ocala National Forest, juvenile recruitment was higher in savanna-like sandhills than hardwood-encroached sandhill habitat (Greenberg and Tanner 2008). Newly metamorphosed gopher frogs emigrating from ponds in Ocala National Forest selected fire-maintained habitat that was associated with an open canopy, few hardwood trees, small amounts of leaf litter, and large amounts of wiregrass (*Aristida stricta*); this habitat contained higher densities of gopher tortoise and small mammal burrows used as refuges (Roznik and Johnson 2009b). Adults emigrating from breeding ponds also preferentially selected fire-maintained habitat (Roznik et al. 2009).

Biology and Taxonomy:

Throughout its range, the gopher frog is a commensal with the gopher tortoise (*Gopherus polyphemus*), hence its common name, and is almost entirely dependent on the tortoise for shelter and to some extent food (Franz 1988; see also Kent et al. 1997). It occasionally occupies a variety of other retreats including the burrows of rodents and crayfish, as well as stump holes and other crevices (Conant and Collins 1991). It is generally nocturnal and emerges to sit near the mouth of its burrow to feed on invertebrates and anurans, including toads (Godley 1992). This regular foraging activity creates a distinct resting area, also called a “platform,” outside each frog’s burrow where the soil has been cleared of vegetation and smoothed by the frog’s constant use (Stevenson and Dyer 2002).

AmphibiaWeb (2012) explains that from a few studies scattered throughout its range, it appears that the timing of breeding and larval development varies from shorter breeding and larval periods farther north to multiple breeding episodes and longer larval periods farther south (Bailey 1991, Semlitsch et al. 1995, Palis 1998, Greenberg 2001). Most studies report May through July as the peak emergence time for metamorphic juveniles (AmphibiaWeb 2012). Rainfall did not appear to trigger emigration and had a negligible influence on daily emigration rates (AmphibiaWeb 2012). Additional natural history information on the species has been summarized by Jensen and Richter (2005).

Gopher frogs in Mississippi were found distinct based on genetic studies by Young and Crother (2001) and are now considered a separate species, *Lithobates sevosa*, which are federally protected and known as dusky gopher frogs or Mississippi gopher frogs.

Population Status:

The gopher frog qualifies for endangered species status because it is experiencing widespread declines across its range in the southeastern United States due to the loss of its longleaf pine ecosystem. Specifically, the distribution and abundance of the gopher frog are much reduced from historical levels, mainly due to loss and degradation of habitat caused by silvicultural practices and fire suppression, combined with reduced gopher tortoise populations (NatureServe 2011). The species is "probably in significant decline" (Hammerson and Jensen 2004). AmphibiaWeb (2012) explains: "Surveys of herpetofauna throughout its range have listed it as uncommon, rare or endangered for at least two decades. Populations are thought to be declining from wetland habitat loss by drainage, filling, or stocking of fish, and upland habitat loss through development, fragmentation and fire suppression (Bailey, 1991)." The IUCN Red List ranks the species as Near Threatened but explains that it is close to qualifying for Vulnerable (Hammerson and Jensen 2004).

The gopher frog is extirpated at many historic locations in Alabama (Bailey 1994). It is known to be extant at a few sites on Conecuh National Forest and nearby private lands in Escambia and Covington counties (Alabama PARC 2011, Hammerson and Jensen 2004).

The gopher frog is likely extirpated in Tennessee, as it is known from just one specimen Tullahoma, in Coffee County (Redmond and Scott 1996).

Reports on the number of extant sites in South Carolina vary. NatureServe (2011) reports just two sites in South Carolina: Savannah River Ecology Lab and Santee Coastal Reserve (NatureServe 2011). Hammerson and Jensen (2004) explain that 12 sites are known from South Carolina, though only four remain extant.

Braswell (1993) found only 11 of 32 previously known breeding sites to be active in North Carolina. A few new North Carolina sites have since been documented (Beane and Hoffman 1995, 1997).

Of 23 historic breeding sites in Georgia investigated by Seyle (1994), only 12 were judged suitable, eight were considered degraded but marginally suitable, and three were judged unsuitable. Hammerson and Jenson (2004) report that the frogs are known to occur at seven sites in Georgia.

Franz and Smith (1999) concluded that gopher frog populations in Florida had declined east of the Apalachicola River in the last 20 years (1975–95), particularly in coastal counties and in South Florida where most of the human population is concentrated. Information available during the last review for NatureServe (which was in 2002) indicated that the gopher frog was extant at 79 sites east of the Apalachicola River and breeds in at least 25 sites west of the Apalachicola River. There is no quantitative information, but the population is assumed to have declined as the human population in Florida has increased and converted suitable habitat to urban, agricultural, and other land uses (FFWCC 2011).

THREATS

Habitat alteration and destruction:

The greatest threat to gopher frogs is the loss and alteration of both upland and wetland habitats resulting from commercial, residential, silvicultural, and agricultural development, as well as from fire suppression (Jensen and Richter 2005). The longleaf pine (*Pinus palustris*) community, the ecosystem primarily inhabited by this species, has been reduced to less than 5 percent of its historical range (Frost 1993, Outcalt and Sheffield 1996).

In Florida, for example, intact xerophytic upland ecosystems inhabited by gopher frogs have suffered severe losses, including longleaf pine-dominated sandhill as well as scrub habitat on the ridges of central and coastal Florida (Means and Grow 1985, Myers 1990, Kautz 1998, Enge et al. 2006). Losses have been especially severe along the highly developed coastal ridges of both southeastern and southwestern Florida, as well as the central ridges that have been mined, converted to agriculture, and more recently developed (Jackson 2004).

Loss of longleaf pine habitat occurs through maximum-yield timber management, such as the establishment of pine monocultures. Means and Means (2005) found that the number of breeding populations of gopher frogs in the Munson Sand Hills of the panhandle Florida occur at a much lower percentage on sand pine silviculture lands than in nearby native longleaf pine habitat, and that some historical breeding populations have been extirpated. They hypothesized that the principal reason is intensive soil disturbance resulting in the elimination or severe alteration of the upland habitat.

In addition, much of the remaining longleaf pine habitats are becoming degraded in the absence of fire. In the past 200 years, human settlement of the Coastal Plain has drastically altered the normal, summertime fire cycle. Not only have wildfires been actively suppressed following ignition, but roads, towns, agricultural fields, and other developments have impeded the widespread, weeks-long fires that swept the Coastal Plain regularly in pre-settlement times (Means 2011).

Without active fire management, there is an encroachment of hardwoods and shrubs in the upland habitat and a decrease in herbaceous ground cover, and the loss of gopher tortoise or pocket gopher populations that provide the primary source of upland shelters (Wolfe et al. 1988, Godley 1992, Greenberg et al. 2003, Jensen and Richter 2005, Blihovde 2006, Roznik 2007, Yager et al. 2007, Roznik et al. 2009).

Exclusion and suppression of fire from wetlands and the concomitant build-up of peat also likely threaten gopher frogs by increasing water acidity past tolerance levels (Smith and Braswell 1994). Exclusion and suppression of fire from wetlands also often leads to degradation of breeding ponds through shrub encroachment and increased evapotranspiration shortening hydroperiods (LaClaire 2001, AmphibiaWeb 2012). For example, Greenberg (2001) found that adult recruitment into ponds did not correspond with upland habitat type (fire suppressed or control burned); however, juvenile recruitment was consistently higher for ponds within the savanna-like uplands than for ponds within the fire suppressed hardwood-invaded uplands. Land managers often use fire lines to exclude prescribed fire from dry wetlands to prevent problems with smoke management or muck fires, particularly if the wetlands are associated with the wildland-urban interface (Bishop and Haas 2005).

The loss of an important microhabitat feature – burrows – is another threat to the frogs. In areas where gopher frogs rely heavily on the burrows of gopher tortoises for refuge, tortoise declines are likely to reduce gopher frog populations as well (Bailey 1991, Godley 1992, Jensen and Richter 2005). The practice of removing tree stumps ("stumping") in silvicultural areas further reduces the availability of subterranean retreats. In the past century, most of the old-growth pine stumps have been extracted to produce turpentine, rosin, and pine oil (Means 2005). Mechanical site preparation (e.g., roller chopping) also destroys burrow openings that result in entrapment of inhabitants (NatureServe 2011).

Gopher frog breeding sites are often degraded by off-road recreational vehicle (ORV) use or by sand roads that pass through or adjacent to the ponds. Hammerson and Jensen (2004) explain: "Vehicular traffic disrupts pond floor micro-topography and eliminates herbaceous vegetation (J. Palis, pers. comm.). Large tires of ORVs break the organic hardpan that lies below the pond floor. This hardpan prevents water from draining into the sand below the wetland (LaClaire and Franz 1991). Breaking the hardpan could result in a shorter hydroperiod and thus make some wetlands unsuitable for gopher frog reproduction. Loss of herbaceous vegetation from ORV use could also discourage gopher frog reproduction since egg masses are attached to stems of herbaceous vegetation (Bailey 1990; J. Palis, pers. comm.)." Use of ORVs in pond basins can also cause direct mortality of tadpoles and adults (LaClaire 2001).

The long distances sometimes traveled by gopher frogs to breed, or following metamorphosis, can make them susceptible to highway mortality. A metamorph leaving a natal pond moved 691 m before being killed, and 3 of 32 metamorphs were killed by vehicles on lightly traveled dirt roads (Roznik and Johnson 2009a) that they apparently used as migration corridors (Roznik and Johnson 2009b).

Sedimentation from road building near breeding sites is also a problem. Erosion of unpaved roads lying adjacent to breeding sites often results in sedimentation into the ponds. Introduction of sediment is exacerbated by emplacement of wing ditches that divert water from roads into ponds. Runoff from paved roads can pollute ponds with petrochemicals and other toxic substances to frogs (LaClaire 2001).

Breeding habitats of the frogs can be lost and degraded by groundwater withdrawals. The Florida Fish and Wildlife Conservation Commission (2011) explains: "The hydrology of many of Florida's depression marsh wetlands may already have been significantly influenced by anthropogenic-caused impacts related to groundwater withdrawals (R. Owen, Florida Department of Environmental Protection, pers. commun. 2010). North Florida has already undergone extreme shifts in groundwater potentiometric levels (i.e., 'groundwater contours') (Grubs and Crandall 2007). The hydrologic impact has been documented for the first time across regional hydrologic divides between the Suwannee River and St. Johns River water). Some ephemeral wetlands are independent of ground water or surface aquifer water, but other wetlands are being impacted by hydrological alterations related to groundwater withdrawal (Guzy et al. 2006). Groundwater withdrawal can shorten hydroperiods or even eliminate ephemeral wetlands."

Another source of habitat degradation is cattle grazing. Heavy grazing by cattle in summer in dried pond basins likely reduces or eliminates frog oviposition sites and/or alter pond nutrient cycling (Hammerson and Jensen 2004).

Disease or predation:

Amphibian dieoffs due to chytridiomycosis have not been detected in the southeastern U.S (FFGCC 2011). None of 18 gopher frog tadpoles examined from Florida and Georgia tested positive for chytrid (Rothermel et al. 2008). But the disease could have devastating impacts in the future, especially because chytrid has been detected in dusky gopher frogs in Mississippi (FWS 2009).

Ranaviruses may be even a greater threat to the gopher frog (Gray et al. 2009). A die-off of hundreds of ranid tadpoles, including gopher frogs, in two ponds in Withlacoochee State Forest, Hernando County, Florida, was apparently caused by an unnamed *Perkinsus*-like (or alveolate) microorganism (Davis et al. 2007, Rothermel et al. 2008). A newly identified mesomycetozoan pathogen, *Anuraperkinsus emelandra*, has been the cause of massive ranid tadpole mortalities in 10 states, including a 2003 die-off of almost all tadpoles at the only known viable breeding pond of the dusky gopher frog (FFGCC 2011).

Inadequacy of existing regulatory mechanisms:

Florida has classified the gopher frog as threatened, it is protected from take by regulation in Alabama, and it is a species of special concern in North Carolina and South Carolina (Mount 1975, Martof et al. 1980, Moler 1992a, Levell, 1997, Hammerson and Jenson 2004, FFWCC 2011b, AmphibiaWeb 2012, Alabama Dept. of Conservation and Natural Resources 2012). But these designations do not require the protection of habitat, which is the primary factor causing the observed declines. Because gopher frogs are sensitive to anthropogenic alteration of their habitat (Bailey 1991, Godley 1992), perpetuation of existing populations will require preservation of relatively undisturbed xeric longleaf pine-dominated uplands.

There are ongoing initiatives and incentives to restore longleaf pine forests within the frog's range. Restoration efforts vary from large-scale actions on federal lands to voluntary silvicultural management practices being undertaken by industrial and private timber landowners (Jones and Dorr 2004, Plum Creek 2010). But nearly all of the efforts to protect these habitats are purely voluntary and without dedicated funding, which means that it is uncertain whether these actions will continue in the future.

A number of protected areas occur on various public lands throughout the species' range (Hammerson and Jenson 2004). The largest populations are on federal lands, including Conecuh

National Forest (Alabama), Eglin Air Force Base (Florida), and Fort Benning (Georgia). These areas receive at least some protection relevant to the needs of this frog (Hammerson and Jenson 2004). Unfortunately, these federal lands protect just a fraction of the frog's range. Additionally, as explained below, fish stocking on these lands can make the habitat unsuitable for the frogs.

On public lands, prescribed burning is a significant part of many habitat management plans. However, implementation of prescribed burning has been inconsistent due to financial constraints and limitations of weather (drought, wind direction, etc.) that restrict the number of opportunities to burn (FWS 2009b).

Other factors:

The introduction of fish likely plays a role in population declines of the gopher frog (FFGCC 2011, Jensen and Richter 2005). Eastern mosquitofish (*Gambusia holbrooki*) are often introduced into isolated wetlands for mosquito control purposes, and even these small fish can have significant negative effects on gopher frog tadpoles (Gregoire and Gunzburger 2008). A far more serious threat, however, is the stocking of game fish (*Lepomis* spp. and *Micropterus* spp.) into ponds used by gopher frogs, or the introduction of predaceous fish into formerly fish-free wetlands during natural flooding events (FFGCC 2011). The introduction of bluegill (*Lepomis macrochirus*) and mosquitofish (*Gambusia affinis*) into a pond in south-central Alabama is strongly suspected for the lack of continued breeding by gopher frogs there (Jensen 1995).

Climate change is likely affecting the amount of winter precipitation in peninsular Florida (K. Enge, FWC, pers. commun. 2010, as cited in Florida Fish and Wildlife Conservation Commission 2011) and could potentially affect breeding pond hydrology (hydroperiod, timing, and water temperature) and upland habitat conditions (fire return and intensity) (C. Greenberg, U.S. Forest Service, pers. commun. 2011, as cited in Florida Fish and Wildlife Conservation Commission 2011). Long-term droughts likely already have caused some gopher frog populations to disappear because of insufficient population recruitment (Florida Fish and Wildlife Conservation Commission 2011).

References:

Alabama Dept. of Conservation and Natural Resources. 2012. Nongame Vertebrates Protected by Alabama Regulations, *available at* http://www.outdooralabama.com/watchable-wildlife/regulations/nongame.cfm (last visited June 20, 2012).

Alabama Partners in Amphibian and Reptile Conservation. 2011. Gopher Frog Initiative, *available at* http://www.alaparc.org/Initiatives/GopherFrogs.html (last visited Oct. 14, 2011).

Altig, R. and R. Lohoefener. 1983. Rana areolata. Catalogue of American Amphibians and Reptiles. 324: 1-4.

AmphibiaWeb: Information on amphibian biology and conservation. [web application]. 2012. Berkeley, California: AmphibiaWeb, *available at*: http://amphibiaweb.org (last visited June 20, 2012).

Bailey, M. A. 1990. Movement of the dusky gopher frog (*Rana areolata sevosa*) at a temporary pond in the lower coastal plain of Alabama. Pp. 27-43 in C. K. Dodd Jr., R. E. Ashton Jr., R. Franz and E. Ester (eds.). Proceedings of the Eighth Annual Meeting of the Gopher Tortoise Council. Florida Museum of Natual History, Gainesville, FL.

Bailey, M. A. 1991. The dusky gopher frog in Alabama. J. Alabama Acad. Sci. 62(1): 28-34.

Bailey, M.A. 1994. An investigation of the dusky gopher frog in Alabama. 1994. U.S. Fish and Wildlife Service, Unpublished report, Jackson, Mississippi.

Bartlett, R. D., and P. P. Bartlett. 1999. A field guide to Florida reptiles and amphibians. Gulf Publishing Company, Houston, Texas. xvi + 278 pp.

Beane, J.C. and E.L. Hoffman. 1995. Geographic Distribution. *Rana capito capito*. Herpetological Review 26: 153.

Bishop, D. C., and C. A. Haas. 2005. Burning trends and potential negative effects of suppressing wetland fires on flatwoods salamanders. Natural Areas Journal 25: 290–294.

Blihovde, W. B. 2006. Terrestrial movement and upland habitat use of gopher frogs in Central Florida. Southeastern Naturalist 5: 265–276.

Braswell, A. L. 1993. Status report on *Rana capito capito* Le Conte, the Carolina gopher frog in North Carolina. Final report to North Carolina Wildlife Commission.

Conant, R. and J. T. Collins. 1991. A field guide to reptiles and amphibians: eastern and central North America. Third edition. Houghton Mifflin Co., Boston, Massachusetts. 450 pp.

Davis A. K., M. J. Yabsley, M. K. Keel, and J. C. Maerz. 2007. Discovery of a novel alveolate pathogen affecting southern leopard frogs in Georgia: description of the disease and host effects. EcoHealth 4: 310–317.

Enge, K. M., J. E. Berish, R. Bolt, A. Dziergowski, and H. R. Mushinsky. 2006. Biological status report–gopher tortoise. Florida Fish and Wildlife Conservation Commission, Tallahassee, Florida, USA. 60pp.

Florida Fish and Wildlife Conservation Commission. 2011. Gopher Frog Biological Status Review Report, *available at* http://www.myfwc.com/media/1351559/Gopher%20Frog%20Final%20BSR.pdf (last visited Oct. 14, 2011).

Florida Fish and Wildlife Conservation Commission (FFWCC). 2011b. Florida's Endangered and Threatened Species, *available at* http://myfwc.com/media/1515251/Threatened_Endangered_Species.pdf (last visited Oct. 4, 2011).

Franz, R. 1986. The Florida gopher frog and Florida pine snake as burrow associates of the gopher tortoise in northern Florida. Pp. 16-20 in D. R. Jackson and R. J. Bryant (eds.). The Gopher Tortoise and its Community, Proceedings of the Fifth Annual Meeting of the Gopher Tortoise Council. Florida State Museum, Gainesville, FL.

Franz, R. 1988. The Florida Gopher Frog and the Florida Pine Snake as burrow associates of the Gopher Tortoise in northern Florida. Proceedings of the Annual Meeting of the Gopher and Tortoise Council. D. R. Jackson and R. J. Bryant, eds., Florida State Museum, Gainesville, 16-20.

Franz, R. 1991. Remember the drought? Florida Wildl. 45: 10-2.

Franz, R., and L. L. Smith. 1999. Distribution of the striped newt and Florida gopher frog in peninsular Florida. Florida Fish and Wildlife Conservation Commission Final Report, Tallahassee, Florida, USA. 46pp.

Frost, C.C. 1993. Four centuries of changing landscape patterns in the longleaf pine ecosystem. Pp. 17–43. *In* Hermann, S.M. (Ed.), The Longleaf Pine Ecosystem: Ecology, Restoration and Management. Proceedings of the Tall Timbers Fire Ecology Conference, Number 18, Tallahassee, Florida.

Gentry, J. B. and M. H. Smith. 1968. Food habits and burrow associates of *Peromyscus polionotus*. J. Mammalogy. 49: 562-5.

Godley, J. S. 1992. Gopher frog *Rana capito* Le Conte. Pages 15-19 in P. E. Moler, editor. Rare and endangered biota of Florida. Vol. III. Amphibians and reptiles. Univ. Press of Florida.

Goin, C. J. and M. G. Netting. 1940. A new gopher frog from the Gulf Coast, with comments upon the *Rana areolata* group. Ann. Carnegie Mus., Art. 8: 137-68.

Gray M.J., Miller D.L., Hoverman J.T. 2009. Ecology and pathology of amphibian ranaviruses. Diseases of Aquatic Organisms 87: 243–266.

Greenberg, C. H. 2001. Spatio-temporal dynamics of pond use and recruitment in Florida gopher frogs (*Rana capito aesopus*). Journal of Herpetology 35(1): 74-85.

Greenberg, C. H., and G. W. Tanner. 2008. Long-term landscape scale monitoring of amphibians at ephemeral ponds in regularly burned versus hardwood-invaded Florida longleaf pine-wiregrass uplands of the Ocala National Forest, Florida. Final report. Florida Fish and Wildlife Conservation Commission, Tallahassee, Florida, USA. 22pp.

Greenberg, C. H., A. Storfer, G. W. Tanner, and S. G. Mech. 2003. Amphibians using isolated, ephemeral ponds in Florida Longleaf Pine uplands: population dynamics and assessment of monitoring methodologies. Florida Fish and Wildlife Conservation Commission Final Report, Tallahassee, Florida, USA. 57pp.

Grubs, J. W., and C. A. Crandall. 2007. Exchanges of water between the Upper Floridan aquifer and the Lower Suwannee and Lower Santa Fe Rivers, Florida. U.S. Geological Survey Professional Paper 1656-C. 83pp.

Guzy, J. C., T. S. Campbell, and K. R. Campbell. 2006. Effects of hydrological alterations on frog and toad populations at Morris Bridge Wellfield, Hillsborough County, Florida. Florida Scientist 69: 276–287.

Hammerson, G., and J. Jensen 2004. *Lithobates capito*. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.1. <www.iucnredlist.org>. Downloaded on 14 October 2011.

Jackson, D. R. 2004. Occurrence of the gopher frog, *Rana capito*, on conservation lands in southern Florida. Florida Natural Areas Inventory, Tallahassee, Florida, USA. 22pp.

Jensen, J.B. 1995. Monitoring dusky gopher frog breeding success and management effectiveness, Conecuh National Forest, Alabama. U.S.D.A. Forest Service, Unpublished report, Andalusia, Alabama.

Jensen, J.B. and Richter, S. 2005. Gopher Frog (*Rana capito*). Status and Conservation of U.S. Amphibians. Volume 2: Species Accounts. Lannoo, M.J.,editor. University of California Press. Berkeley, California.

Jones, J.C. and B. Dorr. 2004. Habitat association of gopher tortoise burrows on industrial timberlands. Wildlife Society Bulletin 32(2): 456-464.

Kautz, R. S. 1998. Land use and land cover trends in Florida 1936–1995. Florida Scientist 61: 171–187.

Kent, D. M., M.A. Langston, and D.W. Hanf. 1997. Observations of vertebrates associated with gopher tortoise burrows in Orange County, Florida. Florida Scientist 60(3): 197-201.

LaClaire, L.V. 2001. Endangered and threatened wildlife and plants; final rule to list the Mississippi gopher frog distinct population segment of dusky gopher frog as endangered. Federal Register 66: 62993–63001.

LaClaire, L. V., and R. Franz. 1990. Importance of isolated wetlands in upland landscapes. Pages 9–15 *in* M. Kelly, editor. Proceedings of the 2nd Annual Meeting of the Florida Lake Management Society, Winter Haven, Florida, USA.

Lee, D. S. 1968. Herpetofauna associated with central Florida mammals. Herpetologica. 24: 83-4.

Levell, J.P. 1997. A Field Guide to Reptiles and the Law. Revised Second edition. Serpent's Tale, Lanesboro, Minnesota.

Martof, B. S., W. M. Palmer, J. R. Bailey, and J. R. Harrison, III. 1980. Amphibians and reptiles of the Carolinas and Virginia. University of North Carolina Press, Chapel Hill, North Carolina. 264 pp.

Means, D. B. 1986. Threatened: dusky gopher frog. Pp. 30-1 in R. H. Mount (ed.). Vertebrate Animals of Alabama in Need of Special Attention. Alabama Agricul. Exp. Sta. Auburn University, Auburn, AL.

Means, D.B. 2005. The value of dead tree bases and stump holes as habitat for wildlife. In W. E. Meshaka Jr. and K. J. Babbitt, Amphibians and reptiles: Status and conservation in Florida. Krieger Publishing Co.

Means, D.B. 2011. Diamonds in the Rough (unpublished manuscript).

Means, D. B., and G. O. Grow. 1985. The endangered longleaf pine community. ENFO (Environmental Information Center of the Florida Conservation Foundation) Report (September):1–12.

Means, D. B., and R. C. Means. 2005. Effects of sand pine silviculture on pond-breeding amphibians in the Woodville Karst Plain of North Florida. Pages 56–61 *in* W. E. Meshaka, Jr., and K. J. Babbitt, editors. Amphibians and reptiles: status and conservation in Florida. Krieger, Malabar, Florida, USA.

Miller, B. T., and D. L. Campbell. 1996. Geographic distribution: *Rana capito*. Herpetological Review 27: 86-87.

Moler, P.E. 1992a. Rare and Endangered Biota of Florida. Volume III. Amphibians and Reptiles. University Press of Florida, Gainesville, Florida.

Moler, P. E. and R. Franz. 1987. Wildlife Values of Small, Isolated Wetlands in the Southeastern Coastal Plain. Pp. 234-41 in R. R. Odum, K. A. Riddleberger, and J. C. Ozier (eds.). Proceedings of the Third Annual Nongame and Endangered Wildlife Symposium. Georgia Dept. of Nat. Res., Atlanta, GA.

Moulis, R.A. 1995a. A survey of suitable habitat in southwestern Georgia for the gopher frog (*Rana capito* ssp.), striped newt (*Notophthalmus perstriatus*), and flatwoods salamander (*Ambystoma cingulatum*). Report to the U.S. Fish and Wildlife Service, Jackson, Mississippi.

Moulis, R.A. 1995b. A survey of eastern Georgia for gopher frogs (*Rana capito* ssp.), with notes on the occurrence of *Notophthalmus perstriatus* and *Ambystoma cingulatum*. U.S. Fish and Wildlife Service, Unpublished report, Jackson, Mississippi.

Mount, R. H. 1975. The reptiles and amphibians of Alabama. Auburn University Agricultural Experiment Station, Auburn, Alabama. vii + 347 pp.

Myers, R. L. 1990. Scrub and high pine. Pages 150–193 *in* R. L. Myers and J. J. Ewel, editors. Ecosystems of Florida. University of Central Florida Press, Orlando, Florida, USA.

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, available at http://www.natureserve.org/explorer (last visited Oct. 14, 2011).

Outcalt, K.W. and R.M. Sheffield. 1996. The longleaf pine forest: trends and current conditions. U.S.D.A. Forest Service, Resource Bulletin, SRS-9, Southern Research Station, Asheville, North Carolina.

Palis, J. G. 1998. Breeding biology of the Gopher Frog, *Rana capito*, in western Florida. Journal of Herpetology 32(2): 217-223.

Plum Creek. 2010. Comments provided on 90-day finding on a petition to list the eastern population of the gopher tortoise. On file Jacksonville Ecological Services Office, Jacksonville, Florida.

Redmond, W. H., and A. F. Scott. 1996. Atlas of amphibians in Tennessee. The Center for Field Biology, Austin Peay State University, Miscellaneous Publication Number 12. v + 94 pp.

Rothermel, B. B., S. C. Walls, J. C. Mitchell, C. K. Dodd, Jr, L. K. Irwin, D. E. Green, V. M. Vazquez, J. W. Petranka, and D. J. Stevenson. 2008. Widespread occurrence of the amphibian chytrid fungus *Batrachochytrium dendrobatidis* in the southeastern USA. Diseases of Aquatic Organisms 82:3–18.

Roznik, E. A. 2007. Terrestrial ecology of juvenile and adult gopher frogs (*Rana capito*). M.S. Thesis, University of Florida, Gainesville, Florida, USA. 97pp.

Roznik, E. A., and S. A. Johnson. 2009a. Burrow use and survival of newly metamorphosed gopher frogs (*Rana capito*). Journal of Herpetology 43: 431–437.

Roznik, E. A., and S. A. Johnson. 2009b. Canopy closure and emigration by juvenile gopher frogs. Journal of Wildlife Management 73: 260–268.

Roznik, E. A., S. A. Johnson, C. H. Greenberg, and G. W. Tanner. 2009. Terrestrial movements and habitat use of gopher frogs in longleaf pine forests: a comparative study of juveniles and adults. Forest Ecology and Management 259: 187−194.

Semlitsch, R. D., J.W. Gibbons, and T.D. Tuberville. 1995. Timing of reproduction and metamorphosis in the Carolina Gopher Frog (*Rana capito capito*) in South Carolina. Journal of Herpetology 29(4): 612-614.

Seyle, C.W. 1994. Distribution and status of the gopher frog (*Rana capito*) and flatwoods salamander (*Ambystoma cingulatum*) in Georgia. U.S. Fish and Wildlife Service, Unpublished report, Jackson, Mississippi.

Smith, S. D., and A. L. Braswell. 1994. Preliminary investigation of acidity in ephemeral wetlands and the relationship to amphibian usage in North Carolina. Report to the Nongame and Endangered Wildlife Program, North Carolina Wildlife Resources Commission, Raleigh, North Carolina, USA.

Stevenson, D. J., and Dyer, K. J. 2002. *Rana capito capito* (Carolina Gopher Frog). Refugia. Herpetological Review 33(2): 128-129.

U.S. Fish and Wildlife Service (FWS). 2009. Mississippi Gopher Frog Recovery Action Plan, *available at* http://ecos.fws.gov/docs/action_plans/doc3100.pdf (last visited Oct. 14, 2011).

U.S. Fish and Wildlife Service (FWS). 2009b. Determination of Endangered Status for Reticulated Flatwoods Salamander; Designation of Critical Habitat for Frosted Flatwoods Salamander and Reticulated Flatwoods Salamander. 74 Fed. Reg. 6702 (Feb. 10, 2009).

Wolfe, S. H., J. A. Reidenauer, and D. B. Means. 1988. An ecological characterization of the Florida panhandle. U.S. Fish and Wildlife Service Biological Report 88(12). 277pp.

Wright, A. H., and A. A. Wright. 1949. Handbook of frogs and toads of the United States and Canada. Comstock Publishing Company, Ithaca, New York. xii + 640 pp.

Yager, L.Y., M.G. Hinderliter, C.D. Heise, and D.M. Epperson. 2007. Gopher tortoise response to habitat management by prescribed burning. Journal of Wildlife Management 71(2):428-434.

Young, J. E., and B. I. Crother. 2001. Allozyme evidence for the separation of Rana areolata and R. capito and for the resurrection of Rana sevosa. Copeia 2001: 382-388.

Scientific Name:

Pseudacris illinoensis or *Pseudacris streckeri illinoensis*

Common Name:

Illinois Chorus Frog

G Rank:

T3

IUCN Red List:

Subspecies not ranked

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

The Illinois chorus frog is found in disjunct populations in west-central and southwestern Illinois, southeastern Missouri, and adjacent Arkansas (Brown and Rose 1988, Conant and Collins 1991). Within Arkansas, the entire Illinois chorus frog population resides in the sandy soil of cotton and soybean fields of eastern Clay County in the historic floodplain of the St. Francis River (Trauth el at. 2004). Historically, the Illinois chorus frog occurred throughout sandy grasslands in southeastern Missouri (Missouri Dept. of Conservation 2011). The present range of the chorus frog in Missouri includes isolated populations associated with specific soil types in Mississippi, Scott, Dunklin and New Madrid counties (Missouri Dept. of Conservation 2011).

Habitat:

The Illinois chorus frog is basically terrestrial and is primarily threatened by development of its habitats for agriculture. It can be found on sand prairies and cultivated fields, open sandy areas of river lowlands (NatureServe 2011). Unlike most frogs and toads that use hindlimbs for digging, the Illinois chorus frog burrows into soil using forelimbs (Tucker et al. 1995). Its eggs and larvae develop in flooded fields, ditches, sloughs, small ponds, or other temporary bodies of water (NatureServe 2011). McCallum et al. (2006) observed that emergent vegetation in the center of temporary sand ponds is an important component of the microhabitat for the frogs.

Biology and Taxonomy:

The subspecies *Pseudacrus strecki illinoensis* was proposed as a distinct species by Collins (1991) (without supporting data). The subspecies was recognized by Phillips et al. (1999), Johnson (2000), and Crother et al. (2008). Molecular analysis by Moriarty and Cannatella (2004) did not provide support for the taxonomic elevation of the subspecies to *Pseudacris illinoensis*, but the researchers noted that further study was needed. Trauth et al. (2007) did a detailed morphological analysis. Their data provide morphological evidence of geographic (clinal) variation within a species, but do not provide support for the taxonomic elevation of the Illinois chorus frog to *Pseudacris illinoensis*, separate from *Pseudacris streckeri*. ITIS considers *Pseudacris streckeri illinoensis* to be invalid and instead recognizes *Pseudacris illinoensis*, as does Frost et al. (2011), which is the online update to Crother et al. (2008). If the FWS does not recognize the species, *Pseudacris illinoensis*, then we petition for the subspecies, *Pseudacrus strecki illinoensis*. A summary of the taxonomic issues is provided by Sparling and Dimitrie (2011).

The Illinois chorus frog is one of the earliest breeding hylid frogs in northeastern Arkansas (Trauth et al. 2004). Breeding begins as early as late January and may continue through mid-April (Butterfield 1988). Most aspects of the larval ecology of Illinois chorus frogs are chiefly unknown, although Illinois chorus frogs are capable of developing cannibalistic larval morphotypes in parts of their range (Sparling and Dimitrie 2011). A unique aspect of their natural history is the forward burrowing behavior with its forelimbs rather than, like the majority of other fossorial anuran species, burrowing backwards using the hind feet. Often, individual frogs can complete burrowing in less than two minutes (Brown et al. 1972). For a summary of life history characteristics, see Sparling and Dimitrie (2011).

Population Status:

The Illinois chorus frog qualifies for endangered status because most populations are small and suffering declines from habitat loss and degradation due largely to agricultural practices (NatureServe 2011, Sparling and Dimitrie 2011). Most populations include fewer than 20 males (Brown and Rose 1988, Herkert 1992). The species was a candidate for federal protection until the FWS eliminated the C2 category, but it currently receives no federal protection under the ESA.

Illinois chorus frogs were documented in seven counties in Illinois prior to 1980, but have not been found in Tazewell and Menard counties since that time (Phillips et al. 1999). Many historic locations in Cass, Morgan, Menard, and Scott counties likely no longer support populations (Nyboer et al. 2004). Data collected from 1993 through 2004 as part of a long-term examination of Illinois chorus frog populations in the Sand Road area of Madison County, Illinois revealed a restriction of Illinois chorus frog populations in this area (Tucker 1998, Tucker and Phillip 1993, Tucker and Phillip 1997, Tucker 2005).

The current status of the Illinois chorus frog in Missouri currently is unknown (Trauth et al. 2006), but the harmful practice of precision land-leveling is common in southeastern Missouri where these chorus frogs occur (42% of irrigated farmland; Evett et al. 2003). See Figg (1991) for a brief and dated account of survey results from southeastern Missouri.

In Arkansas, the size and number of breeding aggregates and the entire population are declining (Sparling and Dimitrie 2011). Illinois chorus frog was estimated to occur over a range of 59 km^2 in 1992, but surveys revealed a 61 percent reduction in range by 2004 and a resulting range of only 23 km^2 (Trauth et al. 2006). This range contraction continues (S. Trauth, personal communication, as cited in Sparling and Dimitrie 2011). McCallum and Trauth (2002) suggested that Illinois chorus frog could extirpated from Arkansas in less than 100 years. Even worse, four years later Trauth et al. (2006) argued that "in the absence of immediate protection and habitat modification through the reintroduction of depressions…extirpation in Arkansas may be imminent."

THREATS

Habitat alteration and destruction:

The Illinois chorus frog is primarily threatened by habitat loss due to agricultural practices and habitat degradation from water contamination and chemical spills (Beltz 1991, Sparling and Dimitrie 2011). Tucker and Phillip (1995) also found "significant reproductive failure" in ephemeral ponds that resulted from slow drainage in areas of human habitation.

Soil tilling, the use of heavy equipment over large tracts of lands, compaction of soil and the use of agricultural chemicals (i.e. insecticides, herbicides, and fertilizers) are all agricultural activities that negatively affect populations of Illinois chorus frogs (Brown and Rose 1988). Illinois chorus frogs are routinely exposed to pesticides due to the intensive application of these chemicals to control cutworms and other agronomic pests (M. McCallum and S. Trauth, unpublished data, as cited in Sparling and Dimitrie 2011). Manure pollution and soil compaction from livestock and direct predation by farm stock (e.g. pigs) also affect populations (Sparling and Dimitrie 2011).

In Illinois, habitat has been lost to drainage of breeding sites and cultivation (Brown and Rose 1988, Herkert 1992, Phillips et al. 1999). Indeed, Brown and Treadway (2001) report that nearly the entire Illinois River sand prairie has been destroyed by agricultural technology. Most of the fields in eastern Clay County were historically wetlands that were initially drained by systems of field ditches (McCallum and Trauth 2002). Specific forms of habitat modification in west-central

Illinois that negatively affect Illinois chorus frog populations include trash dumps, development adjacent to towns, chemical storage facilities, and drainage modification and tree removal via bulldozing (Brown and Rose 1988).

In Arkansas, for the last 15 years, increasing numbers of farmers have employed precision landleveling as a Best Management Practice (Trauth et al. 2006). This practice removes the depressions that serve as amphibian breeding pools as well as rearranges the top layers of soil housing chorus frog underground burrows and their resident frogs. Prior to the introduction of precision land-leveling, the natural depressions in the farm fields held water long enough for chorus frog recruitment even in years of below-average rainfall (Trauth 1992).

In Missouri, agricultural development in the Bootheel has destroyed nearly all the natural ephemeral pools where this species formerly bred, and housing development has nearly eliminated the sand prairie sites (Missouri Dept. of Conservation 2011). This frog still lives in some highly cultivated areas but is unlikely to survive continued habitat destruction (Missouri Dept. of Conservation 2011). Continued draining and clearing of bottomlands, and housing developments and other land uses, have greatly reduced the habitat in southeastern Missouri, where there also is concern over the effects of pesticides in the environment (Johnson 1987, 2000).

Direct mortality from agricultural practices is also a threat to the Illinois chorus frog. Tucker and Phillip (1995) believe that the current distribution of Illinois chorus from on the Poag sand terrace in Illinois can be best explained by assuming that agricultural practices exclude the frog through increased juvenile mortality.

Sand pits due to mining provide additional potential breed sites, but the hydroperiod of these sites is likely not long enough for larval metamorphosis to occur (Brown and Rose 1988) and such sites are likely population sinks (Sparling and Dimitrie 2011).

Disease or predation:

A related subspecies, *Pseudacris streckeri streckeri*, has been documented as hosts of tetrathyridia of *Mesocestoldes* sp. (McAllister 1987) but this is unlikely to be a threat to the Illinois chorus frog. Chytrid fungus has not been observed in Illinois chorus frogs but is a potential threat.

Inadequacy of existing regulatory mechanisms:

The FWS is considering whether to list the Illinois chorus frog and a status report was completed by Southern Illinois University (U.S. Fish and Wildlife Service 2011). It also was considered a candidate until the C2 category was eliminated, but it currently receives no federal protection under the ESA.

The Clean Water Act provides limited protection for wetland habitats of the frog, but the United States Army Corps of Engineers (USACE) still has the authority to issue general permits on a state, regional, or nationwide basis for activities in wetlands that the USACE determines will have minimal effects on wetland habitat (Mitsch and Gosselink 2000). Federal protection for the Illinois chorus frog would ensure that permits will not jeopardize the continued existence of the frog or destroy or adversely modify its critical habitat.

In 2004, the Natural Resources Conservation Service committed $164,500 to encourage farmers to set aside land in Clay County, Arkansas, to establish a chorus frog breeding preserve (R. Lemmons, NRCS field representative for Clay County, Ark., USA, personal communication, cited in Trauth et al. 2006). Negotiations were hampered, however, by the fact that the NRCS was only authorized to offer $4,273 per ha, half the current market price, and no farmers agreed to participate (*Id*.).

The Illinois chorus frog is listed as threatened in Illinois but this status does not protect the frog's habitat (Illinois Endangered Species Board 2011a). The Illinois DNR worked with the U.S. Army Corps of Engineers to develop a conservation plan for the frog that would reduce impacts of the Corps' dredging plan for the Illinois Waterway (U.S. Army Corps of Engineers undated), but no recovery plan for the frog has been developed (Illinois Endangered Species Board 2011b).

The Missouri Department of Conservation lists the Illinois chorus frog as "rare" but this status does not provide special protections (Missouri Dept. of Conservation 2011). The state agency has developed Best Management Practices for conservation of the frog (Missouri Dept. of Conservation, undated) but these are voluntary and unlikely to be widely adopted.

The Illinois chorus frog lives in very few protected areas. Even within protected areas, such as Sand Ridge State Forest in Illinois, habitat management is needed to remove trees from once open areas (Illinois Dept. of Natural Resources undated).

Other factors:

Long periods of below average rainfall and other extreme weather events could be caused by climate change. Below-normal rainfall during the breeding season (Feb–March) (National Climatic Data Center 1991–2004) likely contributed to a greatly reduced number of metamorphosing tadpoles (Trauth et al. 2006). In addition, a late winter and early spring breeding season make the Illinois chorus frog susceptible to sudden freezes, where frogs can succumb to cold weather in the form of frost injuries or death, even while in burrows (Packard et al. 1998, Tucker 2000). Flooding also poses a threat (Sparling and Dimitrie 2011). The Great Flood of 1993 likely led to the complete extirpation of Illinois chorus frogs from portions of Monroe County near Valmeyer and Fountain (Brandon and Ballard 1998).

Bullfrogs and fish are threats to breeding ponds of the Illinois chorus frog (Taubert et al. undated, Tucker and Phillips 1995). For example, during the summer of 1993, farmers deposited slash into Cahokia Creek in Madison County that created a logjam, and the resultant flooding of an Illinois chorus frog breeding pond led to the introduction of predatory fish into the pond (Sparling and Dimitrie 2011). Although some of the water and fish were removed, remnant fish remained (Brandon and Ballard 1998). In 1994, no Illinois chorus frogs used this site for breeding (Tucker 1998). Predators on Illinois chorus frog larvae have also likely contributed to other recent declines in Madison County (Tucker 2005).

References:

Beltz, E. 1991. Illinois Chorus Frog, *Pseudacris streckeri illinoensis*, 1991 survey of Cass, Menard, Morgan and Scott counties, Illinois. Unpubl. report submitted to Ill. Dept. Cons., Springfield, IL. 18pp. + maps & app.

Brandon, R. A. and S. R. Ballard. 1998. Status of Illinois chorus frogs in southern Illinois. Pages 102-112 *in* M.J. Lannoo, editor. Status and conservation of midwestern amphibians. University of Iowa Press, Iowa City, Iowa, USA.

Brown, L.E., R.C. Treadway. 2001. Death of the Sand Prairies. Bulletin of the Illinois Geographic Society XLIII: 3-29.

Brown, L.E. and G.B. Rose. 1988. Distribution, habitat, and calling season of the Illinois chorus frog (*Pseudacris streckeri illinoensis*) along the lower Illinois River. Ill. Nat. Hist. Surv. Biol. Notes No. 132.

Brown, L. E., H. O. Jackson, and J. R. Brown. 1972. Burrowing behavior of the chorus frog, *Pseudacris streckeri*. Herpetologica 28: 325-328.

Butterfield, B. P. 1988. Age structure and reproductive biology of the Illinois chorus frog (*Pseudacris streckeri illinoensis*) from northeastern Arkansas. Thesis, Department of Biological Sciences, Arkansas State University, Jonesboro, Arkansas, USA.

Collins, J. T. 1991. Viewpoint: a new taxonomic arrangement for some North American amphibians and reptiles. SSAR Herpetol. Review 22: 42-43.

Conant, R. and J. T. Collins. 1991. A field guide to reptiles and amphibians: eastern and central North America. Third edition. Houghton Mifflin Co., Boston, Massachusetts. 450 pp.

Crother, B. I. (ed.). 2008. Scientific and Standard English Names of Amphibians and Reptiles of North America North of Mexico, pp. 1–84. SSAR Herpetological Circular 37.

Evett, S., D. Carman, and D. Bucks. 2003. Expansion of irrigation in the midsouth United States: water allocation and research issues. Pages 247–260 in Proceedings of the United States Committee on Irrigation and Drainage International Conference. 12–15 May 2003, Phoenix, Arizona, USA.

Figg, D. E. 1991. Missouri Department of Conservation Annual Nongame and Endangered Species Report July 1990 - June 1991. ii + 35 pp.

Frost, Darrel R., Roy W. McDiarmid, Joseph R. Mendelson III, and David M. Green. 2011. Scientific and Standard English Names of Amphibians and Reptiles of North America North of Mexico, With Comments Regarding Confidence In Our Understanding. Edition 6.1, *available at* http://www.ssarherps.org/pages/comm_names/Anura.php (last visted June 26, 2012).

Hammerson, G. 2004. *Pseudacris streckeri*. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.2. <www.iucnredlist.org>. Downloaded on 29 November 2011.

Herkert, J. R., editor. 1992. Endangered and threatened species of Illinois: status and distribution. Vol. 2: Animals. Illinois Endangered Species Protection Board. iv + 142 pp.

Illinois Dept. of Natural Resources. Undated. Illinois Chorus Frog, *available at* www.dnr.state.il.us/orep/pdfs/frog.pdf (last visited Nov. 29, 2011).

Illinois Endangered Species Board. 2011a. Checklist of endangered and threatened animals and plants of Illinois, *available at* http://www.dnr.state.il.us/ESPB/pdf/2011_Checklist.pdf (last visited Nov. 29, 2011).

Illinois Endangered Species Board. 2011b. Illinois Endangered Species Board Recovery Plans, *available at* http://www.dnr.state.il.us/ESPB/recovery.htm (last visited Nov. 29, 2011).

ITIS Report. 2011. Pseudacris illinoensis Smith, 1951, *available at* http://www.itis.gov/servlet/SingleRpt/SingleRpt?search_topic=TSN&search_value=662726 (last visited Nov. 29, 2011).

Johnson, T. R. 1987. The amphibians and reptiles of Missouri. Missouri Department of Conservation, Jefferson City. 368 pp.

Johnson, T. R. 2000. The amphibians and reptiles of Missouri. Second edition. Missouri Department of Conservation, Jefferson City. 400 pp.

McAllister, C. T. 1987. Protozoan and metazoan parasites of Strecker's chorus frog, Pseudacris streckeri streckeri (Anura: Hylidae), from northcentral Texas. Proceedings of the Helminthological Society of Washington 54: 271-274.

McCallum, M. L., and S. E. Trauth. 2002. An investigation of the distribution and status of the Illinois chorus frog (Pseudacris streckeri illinoensis) and condition of breeding habitat in northeastern Arkansas: a needs assessment for conservation. Final report to the Arkansas Game and Fish Commission, Little Rock, USA.

McCallum, M.L., S.E. Trauth, C. McDowell, R.G. Neal, and T.L. Klotz. 2006. Calling site characteristics of the Illinois Chorus Frog (*Pseudacris streckeri illinoensis*) in northeastern Arkansas. Herpetological Natural History 9(2): 195–198.

Missouri Department of Conservation. 2011. Illinois Chorus Frog, *available at* http://mdc.mo.gov/discover-nature/field-guide/illinois-chorus-frog (last visited Nov. 29, 2011).

Missouri Department of Conservation. Undated. Best Management Practices: Illinois Chorus Frog, *available at*
http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=27&ved=0CEkQFjAGO

BQ&url=http%3A%2F%2Fmdc.mo.gov%2Flandwater-care%2Fanimal-management%2Fendangered-animal-management%2Freptiles-and-amphibians%2Fresource%2Filli-0&ei=JiDVTqDpLsOq2gWB6InNAQ&usg=AFQjCNFoYer6KJkpv8FS4_lBuPFpsFGFOw (last visited Nov. 29, 2011).

Mitsch, W. J., and J. G. Gosselink. 2000. Wetlands. Third edition. John Wiley & Sons, New York, New York, USA.

Moriarty, E. C. and D. C. Cannatella. 2004. Phylogenetic relationships of the North American chorus frogs (*Pseudacris*: Hylidae). Molecular Phylogenetics and Evolution 30: 409-420.

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, *available at* http://www.natureserve.org/explorer (Accessed: November 29, 2011).

Nyboer, R. W., J. R. Herkert, and J. E Ebinger. 2004. Endangered and threatened species of Illinois: status and distribution. Volume 2: Animals. Illinois Endangered Species Protection Board, Springfield, Illinois, USA.

Packard, G.C., J. K. Tucker, and L. D. Lohmiller. 1998. Distribution of Strecker's chorus frogs (*Pseudacris streckeri*) in relation to their tolerance for freezing. Journal of Herpetology 32: 437-440.

Phillips, C. A., R. A. Brandon, and E. O. Moll. 1999. Field guide to amphibians and reptiles of Illinois. Illinois Natural History Survey Manual 8. xv + 282 pp.

Sparling, Donald W. and David Dimitrie. 2011. A prelisting assessment of *Pseudacris streckeri illinoensis* (Illinois chorus frog). Final Report to the United States Fish and Wildlife Service – Rock Island Field Office. 78 pp.

Taubert, B. D., et. al. No date. Breeding biology and distribution of the Illinois chorus frog (*Pseudacris streckeri illinoensis*) in Illinois. A report presented to the Illinois Department of Conservation through the Illinois State Natural History Survey. 163 pp.

Trauth, S. E. 1992. 1992 field survey of the Illinois chorus frog, Pseudacris streckeri illinoensis from Clay County, Arkansas. Final report to the Missouri Department of Conservation. Jefferson City, USA.

Trauth, S. E., H. W. Robinson, and M. V. Plummer. 2004. Amphibians and reptiles of Arkansas. University of Arkansas, Fayetteville, USA.

Trauth, J.B, R.L. Johnson, S.E. Trauth. 2007. Conservation implications of a morphometric comparison between the Illinois Chorus Frog (Pseudacris streckeri illinoensis) and Strecker's Chorus Frog (P. s. streckeri) (Anura: Hylidae) from Arkansas, Illinois, Missouri, Oklahoma, and Texas. Zootaxa 1589: 23–32.

Tucker, J.K. 1998. Status of the Illinois chorus frog in Madison County, Illinois. In: M.J. Lannoo (ed.), Status and Conservation of Midwestern Amphibians, pp. 94–112. University of Iowa Press, Ames, Iowa, USA.

Tucker, J. K. 2000. Growth and survivorship in the Illinois chorus frog. Transactions of the Illinois State Academy of Science 93:63-68.

Tucker, J. K. 2005. Population status of the Illinois chorus frog in Madison County, Illinois: results of 2004 surveys. Report submitted to Illinois Department of Transportation, Springfield, Illinois, USA.

Tucker, J. K., J. B. Camerer, and J. B. Hatcher. 1995. Natural history note:Pseudacris streckeri illinoensis (Illinois Chorus Frog) Burrows. Herpetol. Rev. 26: 32-33.11

Tucker, J. K., and D. P. Philipp. 1993. Population status of the Illinoischorus frog (Pseudacris streckeri illinoensis) in Madison County,Illinois with emphasis on the New Poag Road/FAP 413 interchange and FAP413 wetland mitigation site. Illinois Nat. Hist. Surv. Aquatic Ecol.Tech. Rept. 93/17. 32 pp.

Tucker, J.K. and D.P. Phillip. 1995. Population Status of the Illinois Chorus Frog (Pseudacris streckeri illinoensis) in Madison County, Illinois: Results of the 1994 Surveys: Final Report on 1994 Results, *available at* https://www.ideals.illinois.edu/bitstream/handle/2142/10144/inhscaev01995i0000a_opt.pdf?sequence=2 (last visited Nov. 29, 2011).

Tucker, J. K. and D. P. Philipp. 1997. Population state of the Illinois chorus frog (*Pseudacris streckeri illinoensis*) in Madison County, Illinois: Results of 1997 surveys. Illinois Department of Transportation technical report, Springfield, Illinois, USA.

U.S. Army Corps of Engineers. Undated. Dredged Material Management Program: Corps' Plan Protects and Preserves Threatened Chorus Frog, *available at* http://www.mvr.usace.army.mil/publicaffairsoffice/dmmp/ChorusFrog.asp (last visited Nov. 29, 2011).

U.S. Fish and Wildlife Service. 2011. Current Status Assessment of Illinois Chorus Frog Completed by Southern Illinois University for FWS, *available at* http://www.fws.gov/FWSJournal/regmap.cfm?ARSKey=29237&emailFlag=1 (last visited Nov. 29, 2011).

Scientific Name:

Rana boylii

Common Name:

Foothill Yellow-legged Frog

G Rank:

G3

IUCN Red List:

Near Threatened

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

The range of the foothill yellow-legged frog includes Pacific drainages from the upper reaches of the Willamette River system, Oregon, south to the Upper San Gabriel River, Los Angeles County, California (NatureServe 2011). Two specimens were collected in 1965 in Baja California, Mexico (Loomis 1965) but subsequent searches have not detected the species in that area (Welsh 1988, Hollingsworth 2000, Grismer 2002, Stebbins 2003).

The species has disappeared from many portions of its historical range, especially in southern California, where it has been extirpated from Santa Barbara County to San Diego County (see Hayes and Jennings 1988, Jennings 1995), and has not been seen in or south of the Transverse Ranges since 1977 despite repeated searches (Sweet 1983, Jennings and Hayes 1994).

Habitat:

This species inhabits partially shaded, rocky perennial streams and rivers at low to moderate elevations, in areas of chaparral, open woodland, and forest, rivers in a variety of habitats including riparian, mixed conifer, and wet meadow types (Nussbaum et al. 1983, Stebbins 1985, Hayes and Jennings 1988). Feller (2005) describes the specific habitat needs as follows:

> Foothill yellow-legged frogs are primarily stream dwelling. Stebbins (1985) describes foothill yellow-legged frogs as stream or river frogs found mostly near water with rocky substrate, often found in or near riffles, and on open, sunny banks. Other authors have expanded this description, and/or offer variations (e.g. Storer 1925; Fitch 1938; Zweifel 1955; Hayes and Jennings 1988; Kupferberg 1996a; Lind et al. 1996; Van Wagner 1996). Although streams and rivers with year-round water are generally required, streams inhabited by the species in Oregon may dry to a series of potholes connected by trickles in the summer (Csuti et al. 2001). Critical habitat (i.e., habitat suitable for egg laying) is defined by Jennings and Hayes (1994a) as a stream with riffles containing cobble-sized (7.5 cm diameter) or larger rocks as substrate, which can be used as egg laying sites. These streams are generally small to mid sized with some shallow, flowing water (Jennings, 1988). Fuller and Lind (1992) observed subadults on partly shaded (20%) pebble/cobble river bars near riffles and pools.

Less typical streams lack a rocky, cobble substrate (Fitch 1938). Other types of riparian habitats include isolated pools and vegetated backwaters (Hayes and Jennings 1988, Ashton et al. 1998).

Biology and Taxonomy:

The foothill yellow-legged frog breeds from the latter part of March to the first of May (AmphibiaWeb 2012). Females oviposit eggs in shallow water toward the margin of streams, attached to sides of stones in the stream bed (AmphibiaWeb 2012). Eggs are laid in clusters (Wright and Wright 1949). Ashton et al. (1997) summarizes additional information on the natural history of the foothill yellow-legged frog. A more recent account but less detailed account is provided by Morey (2007).

Rana boylii was named after Dr. Charles Elisha Boyle, a California “49er” that collected the type specimens in 1850 (Jennings 1987). The foothill yellow legged frog was first described as a species by Baird (1854). A half-century of taxonomic uncertainty followed with several name changes (Zweifel 1968). Since 1955, the foothill yellow-legged frog has been recognized as a distinct species in the family Ranidae (Zweifel 1955, Collins 1990).

In a broad geographic survey of genetic data of extant foot hill yellow-legged frog populations, Lind (2005, pp. 89-90) found individuals from four clades that showed substantial genetic divergence from the rest of the samples. These divergent clades were all at the extremes of the north-south range of the foothill yellow-legged frog. (See Lind 2005 p. 106, Fig. 3.1 map). Lind states unequivocally that the populations in the southern portions of the foothill yellow-legged frog range “are quite divergent from the rest of the species and deserve special conservation focus" (Lind 2005, p.98.). Lind et al. (2011) conducted phylogenetic and population genetic analyses and sampled the ecological and distributional limits of the foothill yellow-legged frog to characterize mitochondrial DNA (mtDNA) variation in 77 frogs from 34 localities. Lind et al. (2011) evaluated 1525 base pairs and found several moderately supported, geographically-cohesive mtDNA clades for the foothill yellow-legged frog. Samples from localities at the edges of the foothill yellow-legged frog geographic range demonstrated substantial genetic divergence from each other and from more central populations. Foothill yellow-legged frog populations at the northern limit of the species in central Oregon and southern populations on both the Sierrian and coast range sides of the Central Valley are divergent from the rest of the species.

Population Status:

The foothill yellow-legged frog qualifies for endangered species status because it is experiencing range-wide population declines due to habitat loss from dams and other threats (U.S. Forest Service 2011). The area of occupancy, number of subpopulations, and habitat quality have also declined throughout its range (NatureServe 2011). Lind (2005) found that just under 50 percent of known localities still had foothill yellow-legged frog populations. AmphibiaWeb (2012) explains that there have been "notable declines in southern California and the west slope drainages of the Sierra Nevada and southern Cascade Mountains (Lind et. al. 1996)" and that it is threatened by construction of dams and predation by bullfrogs. The IUCN Red List ranks the species as Near Threatened but explains that it is close to qualifying for Vulnerable (Santos-Barrera et al. 2004). The species was a candidate for federal protection until the FWS eliminated the C2 category, but it currently receives no federal protection under the ESA.

Jennings and Hayes (1994) comprehensively evaluated the status in California: they reviewed all available reports, surveys, and CDFG files and data, conducted field reconnaissance from 1988-1991, and searched museum specimens and field notes of naturalists as well as relied on their 25 years of field experience for historical locations. Jennings and Hayes (1994) found that the species had disappeared from 45 percent of its historic range in California, about 66 percent of its historic range in the Sierra Nevada, and 24 percent of historical sites in the north coast (Jennings and Hayes 1994). And Fellers (2005) found that only 30 of the 213 sites in California with foothill yellow-legged frogs had populations estimated to be 20 or more adult frogs. Indeed, a large decline has occurred in southern California (Sweet 1983, Jennings and Hayes 1994). This species has probably been extirpated from the Tehachapi Mountains southward and the southern Sierra Nevada (Drost and Fellers 1996). There have also been severe declines in the central Sierra foothills (Moyle 1973, Drost and Fellers 1996). It is still present but nowhere abundant in coastal California from Monterey County southward to northwestern San Luis Obispo County (NatureServe 2011), and in the greater San Francisco Bay Area. In view of these trends, Jennings and Hayes (1994) recommended endangered status in southern and central California south of the Salinas River, Monterey County, and threatened status in the "west slope drainages of the Sierra Nevada and southern Cascade Mountains east of the Sacramento-San Joaquin River axis."

Although formerly regarded as at least locally abundant in southwestern Oregon (Fitch 1936, 1938), it is now rare or absent through the entire western half of the Oregon range (Fellers 2005). This frog has disappeared from more than 55 percent of historical locations in Oregon and is presumed extirpated from most of the northern and far eastern portions of the range in Oregon (Leonard et al. 1993, Borisenko and Hayes 1999, Csuti et al. 2001, Jones et al. 2005).

The species is most likely now extirpated from Mexico (Welsh 1988, Hollingsworth 2000, Santos-Barrera et al. 2004).

THREATS

NatureServe (2011) summarizes the threats to the foothill yellow-legged frog, which include stream scouring (negatively impacts frogs in streambed hibernation sites), stabilization of historically fluctuating stream flows as a result of dam construction, introduced incompatible aquatic animals, riverine and riparian impacts of non-selective logging practices, and other habitat degradation and disturbance caused by livestock grazing and in-stream mining. A detailed examination of the threats faced by foothill yellow-legged frogs is provided by Olson and Davis (2009).

Habitat alteration and destruction:

Dams

Lind (2005) found that former yellow-legged frog localities throughout California where frogs are now extirpated were characterized by higher numbers of all dams upstream, greater number of very large dams upstream, greater maximum height of dams upstream and closer proximity to upstream dams. On the main stem of the Trinity River, northern California, unnatural flow regimes and loss of habitat caused by dam construction are the greatest threats (Ashton et al. 1997). Potential breeding habitat was reduced by 94 percent after dam construction (Lind et al. 1996). Controlled flows allowed encroachment of riparian vegetation and retarded cobble/gravel bar formation. Since dam construction water releases have been reduced to 10-30 percent of pre-dam flows, based on both total yearly volume and magnitude of periodic high flows (Lind et al. 1996). High aseasonal flow releases from dams in late spring sometimes result in scouring of egg masses, whereas receding high flows, if poorly timed, can leave egg masses stranded "high and dry" (Lind et al. 1996). Bobzien and DiDonato (2007) concluded from frog breeding surveys in Alameda Creek in Alameda County, California, that unnatural and consistently higher discharge and irregular flows associated with dam releases appears to be a major factor in poor reproductive conditions for the frog, when compared to stream reaches with natural hydrology.

Mount et al. (2006), based on review of the literature and FERC-related reports, found foothill yellow-legged frog egg masses are negatively affected by pulsed flows (large magnitude flow fluctuations in rivers with dams) via scouring if flows occur during or after oviposition and desiccation if oviposition occurs during high flows and subsequently drops. Tadpole stranding and potential negative effects on metamorphs have been documented in multiple studies. South Fork Eel River population monitoring shows that the magnitude and timing of spring pulse flows are key factors in survival of eggs and tadpoles. While large magnitude spring pulses decrease egg survival, smaller pulses later in the spring cause even higher mortality. Fluctuations in population growth are associated with spring pulse events three years prior. Experiments suggest that during pulse flows tadpoles seek refuge from higher velocities in the substrate, but many are swept downstream. Tadpoles confined to refugia face energetic costs in terms of growth and development. Kupferberg et al. (2011) explored the effects of pulsed flows from dams on foothill yellow-legged frog tadpoles, and found that typical velocity increases in near shore habitats (provided for recreational flows for white water boating or peaking releases for hydroelectric power generation) caused tadpoles approaching metamorphosis to be displaced, and that tadpoles exposed to repeated sub-critical velocity stress grew significantly less and experienced greater predation than tadpoles reared at ambient velocities.

Dams not only eliminate habitat and cause local extirpations, and they also interfere with normal dispersal and movements, which can impede recolonization after local extirpations (Fellers 2005, Peek 2010). Kupferberg et al. (2009b) found that water control management that avoids aseasonal flow fluctuations would benefit foothill yellow-legged frogs, and other taxa, whose lifecycles are synchronous with the natural timing of runoff in California's rivers. Most recently, Kupferberg et al. (2012) found that the foothill yellow-legged fro is more likely to be absent downstream of large dams than in free-flowing rivers, and breeding populations are on average 5 times smaller in regulated rivers than in unregulated rivers.

Logging

Timber harvest decreases populations of aquatic amphibians like the foothill yellow-legged frog by increasing water temperatures to lethal levels and by causing siltation of streambeds (Corn and Bury 1989). High levels of silt inhibit the attachment of frog egg masses to the substrate (Applegarth 1994, Ashton et al. 1997), and excessive accumulation of silt on the egg masses likely has adverse effects on embryo development (Jennings and Hayes 1994). Silt also reduces the interstitial spaces available for use by tadpoles, reduces algal growth on which the tadpoles feed (Power 1990), and can have a significant negative impact on adult frog food resources (e.g., aquatic macro-invertebrates; Petts 1984). Sediment impacts likely adversely affect preferred foothill yellow-legged frog habitat through bed aggradation, surface texture fining or changes in hydraulic geometry (Yarnell 2000).

Livestock Grazing

Livestock grazing likely results in bank erosion, degrading shorelines and increasing stream sedimentation (Davis and Olson 2009). These effects could directly impact instream habitats for frogs. The Sierra Nevada Ecosystem Project, an assessment of the Sierra Nevada ecoregion, concluded that more open vegetation resulting from overgrazing can expose amphibians to predation and desiccation, and direct trampling by livestock is likely an important cause of amphibian mortality (SNEP 1996). Borisenko and Hayes (1999) found locations with frogs had significantly less grazing than locations without frogs. They reported grazing or agricultural concerns for the Coos, Hooskanadan, Pistol and Rogue Rivers. Masters (1997b) described the negative impacts of cattle grazing on habitat used by foothill yellow-legged frogs in Jackson Creek, in the Umpqua National Forest, Oregon:

> Direct impacts of cattle in riparian areas include crushing eggs and tadpoles of foothill yellow-legged frogs, as well as juveniles and adults…Indirect impacts include alteration and/or elimination of vegetation, alteration of the microhabitat

> conditions, degradation of water quality, alteration of the structure and composition of the vegetation, and introduction of non-native vegetative species…Increased sedimentation covers up the cobble-sized rocks that the foothill yellow-legged frog requires for breeding, tadpole development, and juvenile and adult habitat. The cowpies and urine degrade the water quality…sedimentation, resulting from cattle grazing…reduces the interstital spaces available for use by tadpoles and it may inhibit attachment of egg masses.

<u>Mining</u>

Ashton et al. (1997) explained that mining can have deleterious effects on egg masses and tadpoles, as well as disturbing postmetamorphic behavior patterns. In southwestern Oregon, suction-dredging/placer-mining is an extensive historic in-stream activity, allowed by the 1872 Mining Act (Olson and Davis 2009). In Josephine County, Oregon, there are 1600 mining permits on U.S. Forest Service land (D. Clayton, pers. commun., as cited in Olson and Davis 2009). Yet the actual extent of mining across the frog's range in Oregon is unknown, and much is uncontrolled (Olson and Davis 2009).

Gravel extractions are another type of mining to be considered. Stream substrates are removed, processed and relocated during the mining procedures, and all life history stages of foothill yellow-legged frogs would be at risk of direct mortality if such mining occurred at occupied sites (Olson and Davis 2009). The tailings of abandoned mines often have contaminants, such as mercury used to historically extract gold as would settling ponds (Olson and Davis 2009). Mining activities likely contributed to the extirpation of the yellow-legged frog population from Baja (Welsh 1988).

<u>Roads and Urbanization</u>

Roads and urbanization are logical potential threats to this frog (Davis and Olson 2009). The human population continues to increase within its range and this results in continued expansion of urban and agricultural areas and construction of new roads. Road construction crossing streams likely adversely affects frogs due to sedimentation during road building, maintenance or failures. As explained above, sediments can embed stream substrates and removes interstitial spaces used by these frogs. The use of culverts that do not easily pass frogs also impacts population connectivity. Proximity to cities and increasing road density were negatively associated with frog occurrence in the initial threat assessment for Oregon conducted by Olson and Davis (2009). Lind (2006) similarly found that foothill yellow-legged frog presence was associated with less urban development nearby, using data from both Oregon and California.

<u>Recreation</u>

There are potential threats related to recreation (Olson and Davis 2009). Jet boats create waves that could potentially result in dislodgement and loss of egg masses, stranding of tadpoles, disruption of adult basking behavior, and erosion of shorelines (Borisenko and Hayes 1999). Borisenko and Hayes (1999) reported jet boats passing every five minutes with wakes up to a meter high breaking on shore in the lower Rogue River, and no frogs in that area. They also reported recreation concerns for the Chetco River. Vehicles driven along stream gravel bars and recreationists fishing, swimming, walking or camping along shores likely adversely affects frogs, including disruption of frog basking opportunities (Borisenko and Hayes 1999). Damage to montane stream habitat from off-road vehicles is credited as a partial cause of the extirpation of the foothill yellow-legged frog from some southern California coastal streams (Sweet 1983). Off-road vehicle activity also likely eliminated a frog population from Corral Hollow in San Joaquin County. M.R. Jennings documented motorcycle use in riparian zones that crushed juvenile and adult foothill yellow-legged frogs (SNEP 1996).

Disease or predation:

Chytrid fungus has been found in this species, but its population effects are unknown (Fellers 2005). Chytrid fungus was found in foothill yellow-legged frogs and Pacific treefrogs in 10 of 12 sites sampled in the Diablo Mountains, San Benito County, and western San Joaquin foothills, Fresno County, California, in 2006 (Lowe 2007). In laboratory experiments, Davidson et al. (2007) found that chytrid infection reduced growth of newly metamorphosed foothill yellow-legged frogs by approximately one-half and that exposure to the pesticide carbaryl likely increases susceptibility to chytrid infection.

In the main stem of the Trinity River, there is evidence of fungal infections of amphibian egg masses, possibly *Saprolegnia* sp. (Blaustein et al. 1994, Kiesecker and Blaustein 1997). Fungal infection has been observed on foothill yellow-legged frog egg masses (Ashton et al. 1997).

Known from related species are the bacterial disease "red leg" (*Aeromonas hydrophila*) (e.g., *Rana muscosa*, Bradford 1991) and iridoviruses (*Ranavirus* species), which are a complex of viruses found in frogs and fish (Mao et al. 1999).

Inadequacy of existing regulatory mechanisms:

The foothill yellow-legged frog is considered "vulnerable" in Oregon (Olson and Davis 2009), and it is a California Species of Special Concern. But the frog is not state protected in either state and therefore receives no formal protection.

The frog is a U.S. Forest Service sensitive species on national forests in Oregon and California and on BLM land in Oregon (Olson and Davis 2009). But sensitive species designations afford little protection, requiring only that the impacts be considered but not preventing actions that would harm the boreal toad. Thus, the Forest Service or the BLM can conclude in a Biological Evaluation that individuals or populations will be harmed or destroyed by an action, but still carry out this action.

Some populations of this species occur in national forests in California and Oregon. Specifically, Since 1990, foothill yellow-legged frogs have been observed at 24 localities (= populations) on the three Southern Sierra Nevada National Forests: 21 on the Stanislaus, one on the Sierra, and two on the Sequoia (Lind 2003). It also occurs in a few national, regional and state parks, and on properties owned by The Nature Conservancy. But these protected lands do not provide adequate protection from threats such as pesticides or nonnative predators.

Conservation of foothill yellow-legged frogs may be enhanced by maintaining or restoring channels with shapes that provide stable breeding sites over a range of river stages (Kupferberg 1996, Yarnell 2005). New breeding habitat can be created; populations have responded to "bank feathering" restoration projects within one year of construction (Lind et al. 1996). Reintroduction at unoccupied historic sites should also be considered (Lind and Shaffer 2005). But without a federal recovery plan or other mandatory efforts to restore habitat, such methods are unlikely to be utilized.

Other factors:

Climate Change and UV-Radiation

Climate change and UV-B radiation appear to be contributing factors in the decline of this species (Fellers 2005, Olson and Davis 2009). Davidson et al. (2002) examined the spatial patterns of declining frogs in California and hypotheses of spatial patterns of ultraviolet radiation effects and climate change. For foothill yellow-legged frogs, they found a north-to-south gradient of increasing frog losses, consistent with climate change hypotheses (more losses at drier sites to the south), but increasing frog declines at lower elevations, which was at odds with the UV-B hypothesis. Lind (2005) considered climate change as a potential threat to foothill yellow-legged frog, due to precipitation being associated with frog presence.

Kupferberg et al. (2009a) presented data supporting a link between periods of unusually warm summer water temperatures during 2006 and 2008 in a northern California river, outbreaks of the parasitic copepod *Lernaea cyprinacea*, and malformations in tadpoles and young of the year foothill yellow-legged frogs.

Pollution

According to Fellers (2005), in the Sierra Nevada foothills of California, air-borne pesticides (that move east on the prevailing winds blowing across the highly agriculturalized Central Valley) are likely to be the primary threat to foothill yellow-legged frogs (LeNoir et al. 1999, Sparling et al. 2001, Hayes et al. 2002b, Sparling and Fellers 2007, Sparling and Fellers 2008). It is unknown whether pesticides are contributing to the decline of foothill yellow-legged frogs in Oregon (especially east of the agricultural parts of the Willamette Valley), but it should be examined (Fellers 2005). The populations of foothill yellow-legged frogs in greatest decline are all downwind of highly impacted (mostly agriculturalized) areas, while the largest, most robust frog populations are along the Pacific coast (Fellers 2005).

Davidson et al. (2002) found evidence that airborne agrochemicals have played a significant role in the decline of the species. Davidson (2004) examined the association between the spatial patterns of declines for five California amphibian species and historical patterns of pesticide use in California from 1974 to 1991, and found that historical pesticide use was a strong, significant variable in population declines for the foothill yellow-legged frog, especially so for organophosphates and carbamates. In particular, they found that sublethal exposure to the pesticide carbaryl likely inhibits the innate immune defense of foothill yellow-legged frogs and increase susceptibility to disease. Sparling and Fellers (2007) found that environmental concentrations of the pesticides chlorpyrifos, malathion and diazinon and their oxons can be harmful to populations of the frog. Sparling and Fellers (2009) established the chronic toxicity of chlorpyrifos and endosulfan, two of the insecticides most commonly used in the Central Valley and found in the mountains, which likely contributes to observed declines in the frog. Kerby (2007) examined the sublethal effects of four pesticides on foothill yellow-legged frogs and found significant alteration of behavior and development.

Ashton et al. (1997) mentioned the potential for spills of toxic materials into streams along roads along the Trinity River in northern California. Bury (1972) found that spilled diesel fuel had negative impacts on foothill yellow-legged frog tadpoles and partially transformed individuals but apparently little impact on adults.

Mercury contamination is another threat to the frog. Hothem et al. (2010) found mercury concentrations in the foothill yellow-legged frog that were high enough to pose a potential hazard to human or wildlife consumption, with the total Hg concentration exceeding the FDA criterion (1.0 μg/g) for regulation of commercial fish in at least one sample at 24 percent of the yellow-legged frog sites, with 13 of the sites (62 percent) exceeding the EPA Hg criterion (0.3 μg/g) for issuance of health advisories for fish consumption. Research shows that mercury likely adversely affects amphibian development and can decrease survival through metamorphosis (Unrine et al. 2004). Other effects can include impaired reproduction, growth inhibition, behavioral modification, and various sublethal effects (Zillioux et al. 1993).

Exotic Species

A host of vertebrates and perhaps some aquatic invertebrates feed on foothill yellow-legged frogs (Fellers 2005), but it is the nonnative predators that are threatening the species. It is well documented that adults, larvae, and/or eggs are vulnerable to an array of non-native predators such as predatory fishes, bullfrogs, and crayfish (Moyle 1973, Lind et al. 1996, Kupferberg 1996, Ashton et al. 1997, Lind et al. 2003, Fellers 2005, Paoletti 2009, Paoletti et al. 2011). Rombough et al. (2005) found that foothill yellow-legged frog abundance and production was inversely related to abundance of smallmouth bass (*Micropterus dolomieu*) and American bullfrogs (*R. catesbeiana*). Predation by feral pigs is a concern in some locations (Ely 1993, 1994).

Dam-controlled flows and lack of winter flooding likely results in stable pool areas with established aquatic vegetation (Lind et al. 1996, Kupferberg 1996), and this can increase suitable habitat for exotic species such as bullfrogs (Ashton et al. 1997). Decreased flows can force frogs into permanent pools where they are more susceptible to predation (Hayes and Jennings 1988).

Kupferberg (1997) found that bullfrog larvae perturbed aquatic community structure and exerted detrimental effects on foothill yellow-legged frog populations in northern California. Interspecific matings between male foothill yellow-legged frog and female bullfrogs have been observed; these interactions with non-native bullfrogs might reduce the reproductive output of foothill yellow-legged frogs (Lind et al. 2003).

References:

Airola, D. A. 1980. California wildlife habitat relationships program – northeast interior zone. Volume II – reptiles and amphibians.

AmphibiaWeb: Information on amphibian biology and conservation. [web application]. 2012. Berkeley, California: AmphibiaWeb, *available at*: http://amphibiaweb.org/ (last visted June 20, 2012).

Applegarth, J. S. 1994. Special status amphibians and reptiles of the Eugene District: A guide for their conservation. Division of Resources, Eugene District Office, U. S. Bureau of Land Management, Eugene, Oregon.

Ashton, D. T., A. J. Lind, and K. E. Schlick. 1998. *Rana boylii* – foothill yellow-legged frog. Natural history review. USDA Forest Service, Pacific Southwest Research Station, Redwood Sciences Laboratory, Arcata, California.

Baird, S. F. 1854. Descriptions of new genera and species of North American frogs. Proceedings of the Academy of Natural Sciences of Philadelphia 7(2): 59-62.

Blaustein, A. R., D. G. Hokit, R. K. O'Hara, and R. A. Holt. 1994b. Pathogenic fungus contributes to amphibian losses in the Pacific Northwest. Biological Conservation 67(3): 251-254.

Bobzien, S. and J. E. DiDonato. 2007. The Status of the California Tiger Salamander (*Ambystoma californiense*), California Red-Legged Frog (*Rana draytonii*), Foothill Yellow-Legged Frog (*Rana boylii*), and Other Aquatic Herpetofauna in the East Bay Regional Park District, California. East Bay Regional Park District.

Borisenko A.N. and M.P. Hayes. 1999. Status of the foothill yellow-legged frog (Rana boylii) in Oregon. Final Report to The Nature Conservancy under contract to the US Fish and Wildlife Service with assistance from the Oregon Department of Fish and Wildlife, the US Army Corps of Engineers, and the United States Geological Survey. 39 p. Available from: Oregon State Office of US Fish and Wildlife Service, 2600 SE 98th Avenue, Portland, OR 97266.

Bury, R. Bruce. 1972. The effects of diesel fuel on a stream fauna. California Department of Fish and Game Bull. 58: 291-295.

Bury, R. B. 1997. Biogeography of the herpetofauna in the Siskiyou Mountains region (Oregon-California border). *In*: Beigel, J. K., E. S. Jules, and B. Snitkin (editors), Proceedings of the First Conference on Siskiyou Ecology, May 30-June 1, 1997. Siskiyou Regional Education Project.

California Department of Fish and Game (CDFG). 1984. Foothill yellow-legged frog. California's wildlife, Vol. I, amphibians and reptiles.

Cockran, C. C. and C. Thoms. 1996. Amphibians of Oregon, Washington, and British Columbia: A field identification guide. Lone Pine Publishing. Vancouver, British Columbia, Canada. 175 pp.

Collins, J. T. 1990. Standard common and current scientific names for North American amphibians and reptiles. Third Edition. Society for the Study of Amphibians and Reptiles, Herpetological Circular No. 19. 41 pp.

Corn, P. S. and R. B. Bury. 1989. Logging in western Oregon: responses of headwater habitats and stream amphibians. Forest Ecology and Management 29: 1-19.

Corum, S. 2003. [North Coast] Effects of Sacramento Pikeminnow on Foothill Yellow-Legged Frogs in Coastal Streams. California State University, Humboldt. Masters Thesis, pp. 1-42.

Csuti, B., T. A. O'Neil, M. M. Shaughnessy, E. P. Gaines, and J. C. Hak. 2001. Atlas of Oregon wildlife: Distribution, habitat, and natural history. Second edition. Oregon State University Press, Corvallis, Oregon.

Davidson, C. 1995. Frog and toad calls of the Pacific Coast. Library of Natural Sounds, Cornell Laboratory of Ornithology. Ithaca, New York.

Davidson, C. 2004. Declining Downwind: Amphibian Population Declines in California and Historical Pesticide Use. *Ecological Applications* 14(6): 1892–1902.

Davidson, C., H.B. Shaffer, and M.R. Jennings. 2002. Spatial tests of the pesticide drift, habitat destruction, uv-b, and climate-change hypotheses for California amphibian declines. Conservation Biology 16(6): 1588–1601.

Davidson, C., M.F. Bernard, H.B. Shaffer, J.M. Parker, C. O'Leary, J.M. Conlon, and L.A. Rollins-Smith. 2007. Effects of chytrid and carbaryl exposure on survival, growth and skin peptide defenses in foothill yellow-legged frogs. Environmental Science and Technology 41: 1771-1776.

Drost, C. A., and G. M. Fellers. 1996. Collapse of a regional frog fauna in the Yosemite area of the California Sierra Nevada, USA. Conservation Biology 10: 414-425.

Duellman, W. E. and L. Trueb. 1986. Biology of the amphibians. McGraw-Hill Publishing Company. New York, NY. 670 pp.

Ely, E. 1992. Federal candidate herpetological survey. Submitted to Hollister Resource Area Bureau of Land Management, Hollister, California.

Ely, E. 1993. Current status of *Rana aurora*, *Rana boylii*, and *Bufo boreas* at Pinnacles National Monument, California. Unpublished report provided by K. Hutchinson, Monument Superintendent.

Ely, E. 1994. Report and findings 1994 amphibian surveys, Pinnacles National Monument, California. Unpublished report prepared for the National Park Service, Point Reyes, California.

Emlen, S. T. 1968. Territoriality in the bullfrog, Rana catesbeiana. Copeia 1968: 240-243.

Evenden, F. G., Jr. 1948. Food habitats of *Triturus granulosus* in western Oregon. Copeia 1948(3): 219-220.

Fellers, G. M. 2005. *Rana boylii* Baird, 1854(b). Pages 534-536 in M. Lannoo, editor. Amphibian declines: the conservation status of United States species. University of California Press, Berkeley.

Fitch, H.S. 1936. Amphibians and reptiles of the Rogue River Basin. American Midland Naturalist 17: 634-652

Fitch, H.S. 1938. *Rana boylii* in Oregon. Copeia 1938: 148.

Fuller, D.D. and A.J. Lind. 1992. Implications of fish habitat improvement structures for other stream vertebrates. Pp. 96–104. *In* Harris, R. and D. Erman (Eds.), Proceedings of the Symposium on Biodiversity of Northwestern California, Santa Rosa, California.

Gosner, K. L. 1960. A simplified table for staging anuran embryos and larvae with notes on identification. Herpetologica 16: 183-190.

Grismer, L. L. 2002. Amphibians and reptiles of Baja California including its Pacific islands and islands in the Sea of Cortes. University of California Press, Berkeley. xiii + 399 pp.

Hayes, M. P. 1985. *Rana catesbeiana* (Bullfrog). Food. Herpetological Review 16(4): 109.

Hayes, M. P. and M. R. Jennings. 1986. Decline of ranid frog species in western North America: are bullfrogs (*Rana catesbeiana*) responsible? Journal of Herpetology 20(4): 490-509.

Hayes, M. P., and M. R. Jennings. 1988. Habitat correlates of distribution of the California red-legged frog (RANA AURORA) and the foothill yellow-legged frog (*Rana boylii*): implications for management. Pages 144-158 in Szaro, R.C., et al., technical coordinators. Management of amphibians, reptiles, and small mammals in North America. USDA For. Serv., Gen. Tech. Rep. RM-166.

Hayes, T., K. Haston, M. Tsui, A. Hoang, C. Haeffle and A. Vonk. 2002. Atrazine-induced hermaphroditism at 0.1 ppb in American leopard frogs (*Rana pipiens*): laboratory and field evidence. Environmental Health Perspectives .

Hollingsworth, B. 2000. Amphibians of Baja California and nearby islands in the Gulf of California and Pacific Ocean. San Diego Natural History Museum.

Jennings, M. R. 1987. A biography of Dr. Charles Elisha Boyle, with notes on his 19th century natural history collection from California. Wassman J. Biol. 45: 59-68.

Jennings, M.R. 1988. Natural history and decline of native ranid frogs in California. Pp. 61–72. *In* De Lisle, H.F., P.R. Brown, B. Kaufman and B.M. McGurty (Eds.), Proceedings of the Conference on California Herpetology. Southwestern Herpetologists Society, Special Publication, Number 4, Van Nuys, California.

Jennings, M. R., and M. P. Hayes. 1994. Amphibian and reptile species of special concern in California. Final Report submitted to the California Department of Fish and Game, Inland Fisheries Division. Contract No. 8023. 255 pp.

Jones, L.L.C., W. P. Leonard, and D. H. Olson, editors. 2005. Amphibians of the Pacific Northwest. Seattle Audubon Society, Seattle, Washington. xii + 227 pp.

Kerby, J.L. 2007. Pesticide effects on amphibians: A community ecology perspective; DAI-B 68(02): 130.

Kiesecker, J. M. and A. R. Blaustein. 1997. Influences of egg laying behavior on pathogenic infection of amphibian eggs. Conservation Biology 11(1): 214-220.

Kupferberg, S. J. 1996a. Hydrologic and geomorphic factors affecting conservation of a river-breeding frog (*Rana boylii*). Ecological Applications 6(4): 1322-1344.

Kupferberg, S. 1996b. The ecology of native tadpoles (*Rana boylii* and *Hyla regilla*) and the impacts of invading bullfrogs (*Rana catesbeiana*) in a northern California river. Dissertation. University of California, Berkeley, California, USA.

Kupferberg, S.J., 1997. Bullfrog (Rana catesbeiana) invasion of a California river: The role of larval competition. Ecology 78: 1736–1751.

Kupferberg, S., Catenazzi, A., Lunde, K., Lind, A., and Palen, W. 2009a. [North Coast] Parasitic copepod (*Lernaea cyprinacea*) outbreaks in Foothill Yellow-legged Frogs (*Rana boylii*) linked to unusually warm summers and amphibian malformations in Northern California. Copeia 2009(3): 529-537.

Kupferberg, S., Lind, A., Mount, J., and Yarnell, S. 2009b. [Sierra Nevada] Pulsed Flow Effects on the Foothill Yellow-Legged Frog (*Rana boylii*): Integration of Empirical, Experimental, and Hydrodynamic Modeling Approaches. Final Report. California Energy Commission, PIER.

Kupferberg, S. J., A. J. Lind, V. Thill, and S. M. Yarnell. 2011. Water Velocity Tolerance in Tadpoles of the Foothill Yellow-legged Frog (*Rana boylii*): Swimming Performance, Growth, and Survival. Copeia 2011 (1): 141–152.

Kupferberg, S.J., W.J. Palen, A.J. Lind, S. Bobzien, A. Catenazzi, J. Drennan, and M.E. Power. 2012. Effects of Flow Regimes Altered by Dams on Survival, Population Declines, and Range-Wide Losses of California River-Breeding Frogs. Conservation Biology 26(3): 513–524.

LeNoir, J.S., L.L. McConnell, G.M. Fellers, T.M. Cahill and James N. Seiber. 1999. Summertime transport of current-use pesticides from California's Central Valley to the Sierra Nevada Mountain Range, USA. Environmental Toxicology and Chemistry 18: 2715–2722.

Leonard, W.P., H.A. Brown, L.L.C. Jones, K.R. McAllister and R.M. Storm. 1993. Amphibians of Washington and Oregon. Seattle Audubon Society, Trailside Series. Seattle Audubon Society, Seattle, Washington.

Lind, A. 2003. [Sierra Nevada] The Distribution and Habitat of Foothill Yellow-legged Frogs (*Rana boylii*) on National Forests in Southern Sierra Nevada Mountains of California. Report to the FHR Program of Region 5 of the USDA Forest Service. pp. 1-31.

Lind, A.J. 2005. Reintroduction of a declining amphibian: determining an ecologically feasible approach for the foothill yellow-legged frog (Rana boylii) through analysis of decline factors, genetic structure, and habitat associations. Ph.D. Dissertation, University of California, Davis. (March) 169 pp.

Lind, A. J. and H. H. Welsh, Jr. 1994. Ontogenetic changes in foraging behaviour and habitat use by the Oregon garter snake, *Thamnophis atratus hydrophilus*. Animal Behaviour 48: 1261-1273.

Lind, A.J., and B.H. Sheffer. 2005. Reintroduction of a declining amphibian: Determining an ecologically feasible approach for the foothill yellow-legged frog (Rana boylii) through analysis of decline factors, genetic structure, and habitat associations. Dissertation Abstracts International. Section B: Physical Sciences and Engineering: 66(4) 1853.

Lind, A.J., H.H. Welsh Jr., and R.A. Wilson. 1996. [North Coast] The effects of a dam on breeding habitat and egg survival of the foothill yellow-legged frog (*Rana boylii*) in Northwestern California. Herpetological Review 27 (2).

Lind, A.J., J.B. Bettaso and S.M. Yarnell. 2003. Natural history notes: *Rana boylii* (foothill yellow-legged frog) and *Rana catesbeiana* (bullfrog). Reproductive behavior. Herpetological Review 34: 234–235.

Lind, A. J., P. Q. Spinks, G. M. Fellers, and H. B. Shaffer. 2011. Rangewide Phylogeography and Landscape Genetics of the Western U.S. Endemic Frog *Rana Boylii* (Ranidae): Implications for the Conservation of Frogs and Rivers. Conserv. Genet. 12:269–284.

Loomis, R. B. 1965. The yellow-legged frog, *Rana boylei*, from the Sierra San Pedro Martir, Baja California Norte, Mexico. Herpetologica 21: 78-80.

Lowe, J. 2007. Chytrid Distribution in Foothill Yellow-legged Frogs and Pacific Tree Frogs in the Diablo Mountains and San Joaquin Valley Foothills of Central Coastal California. Presented at California-Nevada Amphibian Populations Task Force Meeting, January 18-20, 2007, Las Vegas, Nevada. Bureau of Land Management, Hollister Field Office, CA.

MacTague, L. and P. T. Northern. 1993. Underwater vocalization by the foothill yellow-legged frog (*Rana boylii*). Transactions of the Western Section of the Wildlife Society 29: 1-7.

Mao, J., D.E. Green, G.M. Fellers and V.G. Chincar. 1999. Molecular characterization of iridoviruses isolated from sympatric amphibians and fish. Virus Research 63: 45–52.

Martof, B. S. 1953. Territoriality in the green frog, *Rana clamitans*. Ecology 34: 165-174.

Masters, C. 1997. Amphibian and reptile habitat monitoring in relation to cattle grazing on the Tiller Ranger District of the Umpqua National Forest. Unpublished report provided by D. Otsby, Forest Supervisor.

Milne, L. and M. Milne. 1980. The Audubon Society field guide to North American insects and spiders. Alfred A. Knopf, Inc. New York. 990 pp.

Morey, S. 2007. Life history account for Foothill Yellow-legged Frog, *available at* http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=79&ved=0CFkQFjAIOEY&url=http%3A%2F%2Fnrm.dfg.ca.gov%2FFileHandler.ashx%3FDocumentVersionID%3D17711&ei=sFTVTuudJ83CsQLiscT7Dg&usg=AFQjCNED6S9_fXT5GBfiq_p6WtTsVcMZ_A (last visited Nov. 29, 2011).

Mount, J., S. Yarnell, S. Kupferberg, and A. Lind. 2006. Pulsed Flow Effects on the Foothill Yellow-Legged Frog (*Rana boylii*): Integration of Empirical, Experimental and Hydrodynamic Modeling Approaches. Prepared by Center for Watershed Sciences, U.C. Davis, for California Energy Commission Public Interest Energy Research (PIER) Program.

Moyle, P. 1973. [Sierra Nevada] Effects of Introduced Bullfrogs, *Rana catesbeiana*, on the Native Frogs of the San Joaquin Valley, California. Copeia. 1973(1): 18-22.

Moyle, P. B. and L. R. Brown. 1997. Invading Species in the Eel River, California: Successes, Failures, and Relationships With Resident Species. Environmental Biology of Fishes 49: 271-291.

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia. Available http://www.natureserve.org/explorer. (Accessed: November 29, 2011).

Nussbaum, R.A., E.D. Brodie, Jr., and R.M. Storm. 1983. Amphibians and Reptiles of the Pacific Northwest. University Press of Idaho, Moscow, Idaho. 332 pp.

Olson, D.H. and R.J. Davis. 2009. Conservation Assessment for the Foothill Yellow-legged Frog (Rana boylii) in Oregon, *available at* http://www.fs.fed.us/r6/sfpnw/issssp/documents/planning-docs/ca-ha-aneides-flavipunctatus-2008-10-07.doc (last visited Nov. 30, 2011).

Palen, W. J., S. J. Kupferberg, and M.E. Power. 2007. Evaluating the accuracy of amphibian monitoring designs in the context of hydroelectric dam relicensing: A retrospective analysis of a 16-year *Rana boylii* egg mass census.

Paoletti, D. P. 2009. Responses of Foothill Yellow-legged Frog (Rana boylii) larvae to an introduced predator. Unpubl. Master's thesis, Oregon State University, Corvallis, Oregon.

Paoletti, D.J., Deanna H. Olson, and Andrew R. Blaustein. 2011. Responses of Foothill Yellow-legged Frog (Rana boylii) Larvae to an Introduced Predator. Copeia 2011 (1): 161–168, *available at* http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=29&ved=0CFwQFjAIOBQ&url=http%3A%2F%2Fwww.fs.fed.us%2Fpnw%2Flwm%2Faem%2Fdocs%2Folson%2F2011_paoletti_olson_blaustein_copeia.pdf&ei=MlPVTvDkCfSPsAKQ9vmWDw&usg=AFQjCNFMDJ0Hr_E53NxTcB7Q54yRwI2evA (last visited Nov. 29, 2011).

Peek, R. 2010. [Sierra Nevada] Landscape Genetics of Foothill Yellow-Legged Frogs (*Rana boylii*) in regulated and unregulated rivers: Assessing connectivity and genetic fragmentation. University of San Francisco. Masters Thesis, pp. 1-69.

Petts, G. E. 1984. Impounded rivers: perspectives for ecological management. John-Wiley and Sons. Chichester, England. 285 pp.

Power, M. E. 1990. Resource enhancement by indirect effects of grazers: Armored catfish, algae, and sediment. Ecology 71(3): 897-904.

Rombough, Chris, Marc P. Hayes, and Nancy L. Duncan. 2005. Foothill yellow-legged frog abundance in Cow Creek. Northwestern Naturalist 86(2): 114-16.

Santos-Barrera, G., G. Hammerson, G. Fellers 2004. *Rana boylii*. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.2. <www.iucnredlist.org>. Downloaded on 29 November 2011.

Sierra Nevada Ecosystem Project (SNEP). 1996. Sierra Nevada Ecosystem Project Report. University of California, Davis, Centers for Water and Wildland Resources.

Sparling, D.W., and G.M. Fellers. 2007. Comparative toxicity of chlorpyrifos, diazinon, malathion and their oxon derivatives to Rana boylii. Environmental Pollution 147: 535-539.

Sparling, Donald W. and G.M. Fellers. 2009. Toxicity of two insecticides to California, USA, Anurans and its relevance to declining amphibian populations. Environmental Toxicology & Chemistry 28(8): 1696.

Sparling, D.W., G.M. Fellers and L.L. McConnell. 2001. Pesticides and amphibian declines in California, USA. Environmental Toxicology and Chemistry 20: 1591–1595.

Stebbins, R. C. 1985. A field guide to western reptiles and amphibians. Second edition. Houghton Mifflin Company, Boston, Massachusetts. xiv + 336 pp.

Stebbins, R. C. 2003. A field guide to western reptiles and amphibians. Third edition. Houghton Mifflin Company, Boston.

Storer, T. I. 1925. A synopsis of the amphibia of California. University of California Publication Zoology 27: 1-342.

Sweet, S. 1983. Mechanics of a natural extinction event: Rana boylii in southern California. Abstract of paper present at 26th annual meeting of SSAR and 31st annual meeting of HL, Salt Lake City, Utah, 7-12 August 1983.

Twitty, V. C., D. Grant, and O. Anderson. 1967. Amphibian orientation: An unexpected observation. Science 155(3760): 352-353.

Unrine, J. M., C.H. Jagoe, W.A. Hopkins, and H.A. Brant. Adverse effects of ecologically relevant dietary mercury exposure in southern leopard frog (*Rana sphenocephala*) larvae. Environmental Toxicology and Chemistry 23: 2964–2970.

U.S. Forest Service. 2011. Foothill Yellow-legged Frog (*Rana boylii*) Ecology, River Regulation, and Conservation, *available at* http://www.fs.fed.us/psw/topics/wildlife/herp/rana_boylii/ (last visited Nov. 29, 2011).

Van Wagner, T. J. 1996. Selected life-history and ecological aspects of a population of foothill yellow-legged frogs (*Rana boylii*) from Clear Creek, Nevada County, California. Master's Thesis, Department of Biological Sciences, California State University, Chico. 143 pp.

Welsh, H. H. 1988. An ecogeographic analysis of the herpetofauna of the Sierra San Pedro Martir region, Baja California, with a contribution to the biogeography of the Baja California herpetofauna. Proceedings of the California Academy of Sciences 46(1): 1-72.

Werschkul, D. F. and M. T. Christensen. 1977. Differential predation by *Lepomis macrochirus* on the eggs and tadpoles of *Rana*. Herpetologica 33: 237-241.

Wiseman, K. 2004. [Sierra Nevada] Foothill Yellow-Legged Frog (*Rana boylii*) Predation by the Introduced Signal Crayfish (*Pacifasticus leniusculus*) in the North Fork Feather River, CA. Oral presentation at California/Nevada Declining Amphibians Population Task Force Annual Meeting. Reno, NV.

Wright, A. H. and A. A. Wright. 1949. Handbook of frogs and toads of the United States and Canada. Comstock Publishing Company, Inc., Ithaca, New York.

Yarnell, S.M. 2000. The Influence of Sediment Supply and Transport Capacity On Foothill Yellow-Legged Frog Habitat, South Yuba River, California. Master's Thesis, Geology Department, University of California, Davis.

Zeiner, D. C., W. F. Laudenslayer, Jr., and K. E. Mayer, eds. 1988. California's wildlife, volume 1, amphibians and reptiles. California Department of Fish and Game. 272 pp.

Zillioux, E. J., D.B. Porcella, and J.M. Benoit. 1993. Mercury cycling and effects in freshwater wetland ecosystems. Environmental Toxicology and Chemistry 12: 2245–2264.

Zweifel, R. G. 1955. Ecology, distribution, and systematics of frogs of the *Rana boylii* group. Univ. of California Publ. Zool. 54: 207-292.

Zweifel, R. G. 1968. *Rana boylii* Baird, Foothill yellow-legged frog. Catalogue of American Amphibians and Reptiles. Pp. 71.1-71.2.


Photo courtesy of Walter Siegmund

Scientific Name:

Rana cascadae

Common Name:

Cascades Frog

G Rank:

G3

IUCN Red List:

Near Threatened

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

Pearl and Adams (2005) explain that Cascades frogs historically occupied moderate and high elevation (about 400–2,500 m) lentic habitats throughout the Cascade Range, from the very northern edge of California's Sierra Nevada to within 25 km of the British Columbia border (Dunlap and Storm 1951, Dunlap 1955, Dumas 1966, Bury 1973a, Hayes and Cliff 1982, Nussbaum et al. 1983, Fellers and Drost 1993, Jennings and Hayes 1994a). In Washington, Cascades frogs occur in the Pacific Coast, North Cascades, West Cascades and East Cascades ecoregions (Hallock and McAllister 2009).

Severe range contractions have been documented in the southern end of their range (Fellers and Drost 1993, Jennings and Hayes 1994a). Jennings and Hayes (1994a) and Fellers and Drost (1993) estimate that Cascades frogs are extirpated from about 99 percent of their southernmost population clusters (Mt. Lassen and surroundings), and 50 percent of their total historical distribution in California. Since that time, further range contractions have occurred (Fellers et al. 2007). Its historic range might have included much lower altitudes (Leonard et al. 1993).

Habitat:

According to NatureServe (2011), Cascades frogs inhabit wet mountain meadows, sphagnum bogs, ponds, lakes, and streams, in open or patchy coniferous forests. Generally they are closely associated with water, but they sometimes move from one drainage to another by crossing over high mountain ridges. These frogs hibernate in mud at the bottom of ponds and in spring-water saturated ground up to at least 75 meters from a pond (Briggs 1987). Breeding sites are quiet ponds, where eggs are laid in open shallow water or among submerged vegetation. Adults and breeding can occur in anthropogenic wetland habitats such as pump chances (Quinn et al. 2001). The frogs habitats are being widely degraded by introduced fishes.

Biology:

The Cascades frog calls from above or below water's surface (Stebbins 1985). It is diurnal (active during the day) and breeds from March to mid-August, soon after pond ice begins to thaw (Stebbins 1985). Details on the natural history and biology of the Cascades frog are summarized by Pearl and Adams (2005) and Garwood and Welsch (2007).

Population Status:

The Cascades frog qualifies for endangered species status because it is "probably in significant decline" (Hammeron and Pearl 2004) due to threats such as introduced salmonids. The IUCN Red List ranks the species as Near Threatened but explains that it is close to qualifying to Vulnerable (Hammerson and Pearl 2004). The species was a candidate for federal protection until the FWS eliminated the C2 category, but it currently receives no federal protection under the ESA.

In California, surveys suggest that the Cascades frog is rare to nonexistent in most Californian portions of the historical range (G. Fellers, H. Welsh, personal communications, cited by Pearl and Adams 2005). Historic accounts and museum records indicate that the frog was previously abundant in the Mount Lassen area, California (Fellers et al. 2007). But this species has declined greatly and is now very rare (Fellers et al. 2007). A 1991 survey located no Cascades frogs at 16 historic localities, and found that the frog occupied only 2 percent of the suitable sites surveyed (1 of 50 sites) (Fellers and Drost 1993). Since 1991, four large-scale surveys have been conducted to evaluate the occurrence of aquatic-breeding amphibians throughout the Lassen region (Fellers 1998, Koo et al. 2004, Welsch and Pope 2004, Stead et al. 2005). These data were analyzed by Fellers et al. (2007) and show that the situation has worsened significantly.

From 1993 to 2007, Fellers et al. (2007) conducted 1,873 amphibian surveys at 856 sites within Lassen Volcanic National Park and Lassen National Forest, California. These surveys encompassed all Cascades frog habitats: ponds, lakes, meadows, and streams on those lands. They found frogs at only six sites during 14 years of surveys, and obtained one report of a single frog at one additional locality. These occupied sites represented less than one percent of the historically suitable habitat within the Lassen region. They found no evidence of reproduction in most of the populations, and reproduction at all but one of the other sites remained lower than the annual reproductive output of one breeding pair for greater than 12 years.

Declines have also occurred in Oregon (Nussbaum et al. 1983, Blaustein and Wake 1990, Fite et al. 1998, Olson 2001). AmphibiaWeb (2012): explains that the frog has "declined extremely in Oregon." Although abundant there in the early 1970's, 80 percent of 30 Oregon populations that A. Blaustein has monitored since the mid 1970's have disappeared (Blaustein and Wake 1990).

With extensive declines in California and Oregon, it is apparent that the frogs are declining from the south to the north. The species qualifies for endangered status despite the fact that it appears to be widespread across its historical habitat in Washington (Hallock and McAllister 2009) because the southern areas in which the frogs are declining constitute a significant portion of range.

THREATS

As explained below, potential causative factors in the decline of Cascades frogs include the introduction of fish into historically fishless habitats (e.g., Knapp and Matthews 2000, Knapp 2005, Welsh et al. 2006), disease (e.g., Fellers et al. 2001, Briggs et al. 2005), the downwind drift of airborne pesticides from agricultural areas (e.g., Davidson 2004, Fellers et al. 2004), and synergy among these and other factors (e.g., Blaustein et al. 2003).

Habitat alteration and destruction:

The Cascades frog is suffering from habitat loss and fragmentation. Declines in the Lassen Volcanic National Park are due in part to gradual loss of open meadows and associated aquatic habitats, and loss of breeding habitat due to drought (Fellers and Drost 1993). In this region, fire suppression and cessation of cattle grazing have increased the natural invasion of shrubs and trees into open meadows; former open breeding sites are now clogged with vegetation (Fellers and Drost 1993).

Disease or predation:

A troubling recent finding is that over 50 percent of sampled specimens were infected by chytrid fungus at a montane site in Washington (Gaulke et al. 2011). And that chytrid was detected at 64 percent of sites surveyed in the Klamath Mountains of California and that Cascades frogs were often infected (Piovia-Scott et al. 2011). While the frogs have experienced increased mortality from exposure to the fungus in the laboratory (Piovia-Scott et al. 2011, Garcia et al. 2006), the current impact on wild frogs is unclear as many infected frogs appear asymptomatic (Gaulke et al. 2011) and many extant populations appear to be coexisting with the pathogen (Piovia-Scott et al. 2011).

Field experiments suggest that the oomycete fungus, *Saprolegnia ferax*, is related to embryonic mortality in Cascades frogs and are likely enhanced by other stressors such as ultraviolet radiation (Kiesecker and Blaustein 1995, 1997b). Romansic et al. (2007) found that juvenile

Cascades frogs exposed to *Saprolegnia* had significantly greater rates of mortality than unexposed controls.

Inadequacy of existing regulatory mechanisms:

Cascades frogs are considered a Species of Special Concern in California (California Department of Fish and Game 2011), and Sensitive-Vulnerable in Oregon (Oregon Natural Heritage Program 2008). The Cascades frog is also a Washington State Monitor species. These statuses reflect the fact that the species is suffering population declines but does not afford any legal protection.

According to Hammerson and Pearl (2004), some populations are within protected national park and wilderness areas in Oregon (such as Crater Lake National Park and the Three Sisters wilderness area), Washington (Olympic and Mount Rainier National Parks), and California (Mount Lassen and Trinity Alps). However, factors such as pesticide drift, UV radiation, and fish introductions are prominent threats even in montane protected areas.

Management agencies have not completed management plans that address the Cascades frog (Fellers et al. 2007). In California, the Department of Fish and Game has initiated a conservation strategy for protecting and enhancing native amphibian species while attempting to optimize recreational trout fishing opportunities (Garwood and Welch 2007). DFG has been implementing this conservation strategy in the Sierra Nevada Mountains through watershed-based management plans (http://www.dfg.ca.gov/habcon/conproj/big_pine.html, Milliron 2005). But these plans are focused on mountain (and Sierra) yellow-legged frogs (Garwood and Welsch 2007). Important differences between the ecology of Cascades frogs and mountain yellow-frogs make these plans inadequate to protect Cascades frogs (Garwood and Welsch 2007). In addition, there is no guarantee that this voluntary conservation plan will be fully implemented.

Other factors:

Introduced Species

Introduced salmonids are now widespread in high lakes throughout the range of Cascades frogs and represent a common predator of larvae and small adults, which is limiting its distribution in montane areas (Hayes and Jennings 1986, Fellers and Drost 1993, Jennings and Hayes 1994a, Simons 1998, Adams et al. 2001).

Non-native trout, including brook trout (*Salvelinus fontinalis*), brown trout (*Salmo trutta*), and rainbow trout (*Oncorhynchus mykiss*), have been introduced throughout the range of the Cascades frog. These introductions occurred in formerly fishless lakes and streams where the frogs were once abundant. In the Klamath-Siskiyou region of northwestern California, Welsh et al. (2006) found that Cascades frog distribution negatively correlates with fish distribution, and the larvae occurred 3.7 times more frequently in lakes without trout. And Garwood and Welsch (2007) found summer Cascades frog densities to be 6.3 times higher in a stream lacking trout than at a similar stream with high densities of brook trout. Pope (2008) found that within three years of fish removals from three lakes, Cascades frog densities increased by a factor of 13.6. In addition, the survival of young adult frogs increased from 59 to 94 percent, and realized population growth and recruitment rates at the fish-removal lakes were more than twice as high as the rates for fish-free reference lakes and lakes that contained fish (Pope 2008).

Effects from introduced fish can range from direct predation on frogs (Simons 1998), competition for food (Finlay and Vredenburg 2007), and indirectly through a shared predator (Zavaleta et al. 2001, Pope et al. in review, cited in Garwood and Welsch 2007). Joseph et al. (2010) suggest that reductions in the availability of emerging aquatic insects cause Cascades frogs to consume more terrestrial prey where trout are present. Thus, introduced trout influences native amphibians directly through predation and, indirectly, through pre-emptive resource competition.

Ultraviolet Radiation

Cowger (1988) explains that many researchers suspect that UV-B radiation is a likely cause of the Cascades frog's reduction in numbers (Fellers and Drost 1993, Blaustein et al. 1994, Blaustein and Wake 1995, Blaustein et al. 1995, Kiesecker and Blaustein 1995). Ozone depletion during the last century has allowed higher levels of UV-B to enter our atmosphere (Blaustein et al. 1994). Since ambient UV-B also increases with altitude, populations of organisms living at higher elevations are more affected by UV rays (Blaustein et al. 1995). Since the Cascades frog is only found at high altitudes and needs to thermoregulate to keep warm, it is exposed to a large amount of UV radiation.

Many organisms, including the Cascades frog, contain the enzyme photolyase which helps to repair DNA damaged by light (Blaustein et al. 1994, Blaustein and Wake 1995). However, the Cascades frog has relatively low levels of the photolyase enzyme (Blaustein et al. 1994). UV-B rays weaken the Cascades frog's immune system causing it to be more prone to bacterial and viral infections (Kiesecker and Blaustein 1995).

UV-B rays have been directly implicated as a cause of increasing bacterial *Saprolegina* infections in the Cascades frog which lead to mass population declines of the Cascades frog in some areas (Kiesecker and Blaustein 1995). Increased solar radiation also is likely damaging frog retinas (Fite et al. 1998). In addition, Romansic et al. (2009) found that UVB-exposed Cascades frog larvae displayed decreased growth, increased prevalence of deformities, and increased susceptibility to predation.

Pollution

Agrochemicals are a threat in some areas (Davidson et al. 2002). Fertilizers such as urea likely pose a threat; in laboratory studies, juveniles were unable to sense and avoid toxic levels (Hatch et al. 2001). Nitrites can affect behavior and metamorphosis of larvae (Marco and Blaustein 1999). Paulk and Wagner (2004) found that glyphosate and malathion significantly affect Cascades frog larvae mortality and development at levels below EPA-recommended maximum levels for surface water.

Small and isolated populations

Monson and Blouin (2004) found that the Cascasdes frog exhibits extreme isolation by distance with reduced gene flow at distances greater than 10 km. As such, populations that go extinct are unlikely to be re-colonized quickly, especially if they are greater than 10 km from the nearest population. Consistent with this conclusion is the observation that recolonization of one historic Cascades frog site was reported to have taken 12 years despite the presence of a population within 2 km (Blaustein et al. 1994). This species spends over half the year in hibernation and given the limited amount of time that they are active, combined with their ephemeral habitat, it is not surprising long distance gene flow is rare in this species (Monson and Blouin 2004).

Additionally, Young and Clarke (2000) observed that the small size of, and lack of connectivity between, the current populations of the Cascades frog in the Lassen area greatly reduces their longterm viability, potentially leading to a genetic bottleneck.

References:

Adams, M. A., D. E. Schindler, and R. B. Bury. 2001. Association of amphibians with attenuation of ultraviolet-b radiation in montane ponds. Oecologia 128: 519-525.

AmphibiaWeb: Information on amphibian biology and conservation. [web application]. 2012. Berkeley, California: AmphibiaWeb, available at: http://amphibiaweb.org (last visited June 20, 2012).

Blaustein, A. R., and D. B. Wake. 1990. Declining amphibian populations: a global phenomenon? Trends in Ecology and Evolution 5: 203-204.

Blaustein, A.R., and D.B. Wake, 1995. The puzzle of declining amphibian populations. Scientific American: 91: 52-57.

Blaustein, A.R., Hoffman, P.D., Hokit, D.G., Kiesecker, J.M., Walls, S.C., and J.B. Hays, 1994. UV repair and resistance to solar UV-B in amphibian eggs: a link to population declines? Proc Natl. Acad. Sci. USA 91: 1791-1795.

Blaustein, A.R., J.M. Romansic, J.M. Kiesecker, and A.C. Hatch. 2003. Ultraviolet radiation, toxic chemicals and amphibian population declines. Diversity and Distributions 9: 123-140.

Briggs, J. L., Sr. 1987. Breeding biology of the Cascade frog, *Rana cascadae*, with comparisons to *R. aurora* and *R. pretiosa*. Copeia 1987: 241-245.

Briggs, C.J., V.T. Vredenburg, R.A. Knapp, and L.J. Rachowicz. 2005. Investigating the population-level effects of chytridiomycosis: An emerging infectiousdisease of amphibians. Ecology 86: 3149-3159.

Bury, R.B. 1973. The cascade frog, Rana cascadae, in the north Coast Range of California. Northwest Science. 47: 228-229.

California Dept. of Fish and Game. 2011. Special Animals List, *available at* http://www.dfg.ca.gov/biogeodata/cnddb/pdfs/SPAnimals.pdf (last visited Dec. 2, 2011).

Cowger, L. 1998. Cascades Frog, Rana cascadae, *available at* http://biology.uoregon.edu/reference/herpetology/documents/cowger.html (last visited Dec. 2, 2011).

Davidson, C. 2004. Declining downwind: Amphibian population declines in California and historical pesticide use. Ecological Applications 14: 1892-1902.

Davidson, C., H. B. Shaffer, and M. R. Jennings. 2002. Spatial tests of the pesticide drift, habitat destruction, UV-B, and climate-change hypotheses for California amphibian declines. Conservation Biology 16: 1588-1601.

Dumas, P.C. 1966. Studies of the *Rana* species complex in the Pacific Northwest. Copeia 1966: 60–74.

Dunlap, D.G. 1955. Inter- and intraspecific variation in Oregon frogs of the genus *Rana*. American Midland Naturalist 54: 314–331.

Dunlap, D.G. and R.M. Storm. 1951. The Cascade frog in Oregon. Copeia 1951: 81.

Fellers, G.M. and C.A. Drost. 1993. Disappearance of the Cascades frog RANA CASCADAE at the southern end of its range, California, USA. Biological Conservation 65: 177-181.

Fellers, G.M., D.E. Green, and J.E. Longcore. 2001. Oral chytridiomycosis in the Mountain Yellow-legged Frog (Rana muscosa). Copeia 2001: 945-953.

Fellers, G.M., L.L. McConnell, D. Pratt, and S. Datta. 2004. Pesticides in Mountain Yellow-legged Frogs (Rana muscosa) from the Sierra Nevada Mountains of California, USA. Environmental Toxicology and Chemistry 23: 2170-2177.

Fellers, G.M.; Karen L. Pope; Jonathan E. Stead; Michelle S. Koo; and Hartwell H. Welsh, Jr. 2007. Turning population trend monitoring into active conservation: Can we save the Cascades Frog (*Rana cascadae*) in the Lassen region of California? Herpetological Conservation and Biology 3(1):28-39.

Finlay, J., and Vredenburg V.T. 2007. Introduced trout sever trophic connections between lakes and watersheds: consequences for a declining montane frog. Ecology 88: 2187–2198.

Fite, K. V., A. R. Blaustein, L. Bengston, and H. E. Hewitt. 1998. Evidence of retinal light damage in *Rana cascadae*: a declining amphibian species. Copeia 1998: 906-914.

Garcia, T.S., J.M. Romansic, and A.R. Blaustein. 2006. Survival of three species of anuran metamorphs exposed to UV-B radiation and the pathogenic fungus Batrachochytrium dendrobatidis. Diseases of Aquatic Organisms; 72(2): 163-169.

Garwood, J.M. and H.H. Welsh, Jr. 2007. Ecology of the Cascades frog (Rana cascadae) and interactions with garter snakes and non-native trout in the Trinity Alps Wilderness, California. Final report prepared for the California Department of Fish and Game and the National Fish and Wildlife Foundation. Arcata, California. Biology 3(1): 28-39.

Gaulke, C.A., J.T. Irwin and R.S. Wagner. 2011. Prevalence and Distribution of *Batrachochytrium dendrobatidis* at Montane Sites in Central Washington State, USA. Herpetol. Rev. 42(2): 209-211.

Hammerson, G. and Christopher Pearl 2004. *Rana cascadae*. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.2. <www.iucnredlist.org>. Downloaded on 30 November 2011.

Hallock, L.A. 2009. Surveys for Oregon Spotted Frog (Rana pretiosa) and Cascades Frog (Rana cascadae) at select wetlands in the Trout Lake Creek Watershed, Gifford Pinchot National Forest, Mt. Adams Ranger District, *available at* http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=97&ved=0CEoQFjAGOFo&url=http%3A%2F%2Fwww.fs.fed.us%2Fr6%2Fsfpnw%2Fissssp%2Fdocuments%2Finventories%2Finv-rpt-ha-rapr-gip-surveys 2009.pdf&ei=pfDYTu7DGcTq2AXD_oy2Dg&usg=AFQjCNGp-9zoB3FKEjw_XwTiiL_LDYUk5g (last visited Dec. 2, 2011).

Hallock, L.A., and K.R. McAllister. 2009. Cascades Frog. Washington Herp Atlas. http://www1.dnr.wa.gov/nhp/refdesk/herp

Hatch, A.C., L.K. Belden, E. Scheessele and A.R. Blaustein. 2001. Juvenile amphibians do not avoid potentially lethal levels of urea on soil substrate. Environmental Toxicology and Chemistry 20: 2328–2335.

Hayes, M.P. and F.S. Cliff. 1982. A checklist of the herpetofauna of Butte County, the Butte Sink, and Sutter Buttes, California. Herpetological Review 13: 85–87.

Joseph, M.B., J. Piovia-Scott, S.P. Lawler, and K.L. Pope. 2010. Indirect effects of introduced trout on Cascades frogs (Rana cascadae) via shared aquatic prey. Freshwater Biology (2010), *available at* http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=2&ved=0CCgQFjAB&url=http%3A%2F%2Fwww.fs.fed.us%2Fpsw%2Fpublications%2Fpope%2Fpsw_2010_pope(joseph)003.pdf&ei=Zk7ZTtmxIIny2gXOys3GDw&usg=AFQjCNEe7lhrUc3UH2Uy6S_1lsFoipbPWg (Dec. 2, 2011).

Jennings, M. R., and M. P. Hayes. 1994. Amphibian and reptile species of special concern in California. Final Report submitted to the California Department of Fish and Game, Inland Fisheries Division. Contract No. 8023. 255 pp.

Kiesecker, J.M. and A.R. Blaustein, 1995. Synergism between UV-B radiation and a pathogen magnifies amphibian embryo mortality in nature. Proc. Natl. Acad. Sci. USA 92: 11049-11052.

Kiesecker, J. M., and A. R. Blaustein. 1997. Influences of egg laying behavior on pathogenic infection of amphibian eggs. Conservation Biology 11: 214-220.

Koo, M.S., J.V. Vindum, and M. McFarland. 2004. Results of 02- CS-11050650-029, The 2003 California Academy of Sciences Survey: Amphibians and reptiles of the Lassen National Forest. California Academy of Sciences. 175 p.

Knapp, R.A., and K.R. Matthews. 2000. Non-native fish introductions and the decline of the Mountain Yellowlegged Frog from within protected areas. Conservation Biology 14: 428-438.

Knapp, R.A. 2005. Effects of nonnative fish and habitat characteristics on lentic herpetofauna in Yosemite National Park, USA. Biological Conservation 121: 265-279.

Leonard, W. P., H. A. Brown, L. L. C. Jones, K. R. McAllister, and R. M. Storm. 1993. Amphibians of Washington and Oregon. Seattle Audubon Society, Seattle, Washington. viii + 168 pp.

Marco, A. and A.R. Blaustein. 1999. The effects of nitrite on behavior and metamorphosis in Cascades frogs (*Rana cascadae*). Environmental Toxicology and Chemistry 18: 946–949.

Monsen, K.J., and M.S. Blouin. 2004. Extreme isolation by distance in a montane frog Rana cascadae. Conservation Genetics 5: 827–835.

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia. Available http://www.natureserve.org/explorer. (Accessed: November 30, 2011).

Nussbaum, R.A., E.D. Brodie, Jr., and R.M. Storm. 1983. Amphibians and Reptiles of the Pacific Northwest. University Press of Idaho, Moscow, Idaho. 332 pp.

Olson, D. H. 2001. Ecology and management of montane amphibians of the U.S. Pacific Northwest. Biota 2: 51-74.

Oregon Dept. of Natural Resources. 2008. Sensitive Species, *available at* http://www.dfw.state.or.us/wildlife/diversity/species/docs/SSL_by_taxon.pdf (last visited Dec. 2, 2011).

Palen, W.J., D.E. Schindler, M.J. Adams, C.A. Pearl, R.B. Bury and S.A. Diamond. 2002. Optical characteristics of natural waters protect amphibians from UV-B in the U.S. Pacific Northwest. Ecology 83: 2951–2957.

Paulk, Nicole K., and R.S. Wagner. 2004. Interaction of glyphosate and malathion on mortality and development in Cascades frogs (Rana cascadae). Northwestern Naturalist 85(2): 24.

Pearl, C. and M.J. Adams. 2005. Rana cascadae. In M. Lannoo, editor. Amphibian declines: the conservation status of United States species. University of California Press, Berkeley.

Piovia-Scott, J., K. L. Pope, S. P. Lawler, E. M. Cole, and J. E. Foley. 2011. Factors related to the distribution and prevalence of the fungal pathogen *Batrachochytrium dendrobatidis* in *Rana cascadae* and other amphibians in the Klamath Mountains. Biological Conservation 144: 2913-2921

Pope, K. 2008. Assessing Changes in Amphibian Population Dynamics Following Experimental Manipulations of Introduced Fish. Conservation Biology 22(6): 1572.

Quinn, T., J. Gallie and D.P. Volen. 2001. Amphibian occurrences in artificial and natural wetlands of the Teanaway and Lower Sauk River Drainages of Kittitas County, Washington. Northwest Science 75: 84–89.

Romansic, J.M., E.M. Higashi, K.A. Diez, A.R. Blaustein. 2007. Susceptibility of newly-metamorphosed frogs to a pathogenic water mould (Saprolegnia sp.). Herpetological Journal 17(3): 161.

Romansic, J., A. Waggener, B. Bancroft, and A. Blaustein. 2009. Influence of ultraviolet-B radiation on growth, prevalence of deformities, and susceptibility to predation in Cascades frog (*Rana cascadae*) larvae. Hydrobiologia 624(1): 219.

Simons, L.H. 1998. Natural history notes: *Rana cascadae* (Cascades frog). Predation. Herpetological Review 29: 232.

Stead, J.E., H.H. Welsh, and K.L. Pope. 2005. Survey of amphibians and fishes at all lentic habitats in Lassen Volcanic National Park: A report to the National Park Service. LVNP Study Number: LAVO-00717 contracted with Southern Oregon University and US Forest Service, Redwood Sciences Laboratory.

Stebbins, R. C. 1985. A Field Guide to Western Reptiles and Amphibians. Houghton Mifflin, Boston.

Welsh, H.W., and K.L. Pope. 2004. Impacts of introduced fishes on the native amphibians of northern California wilderness areas: Final Report to the California Department of Fish and Game, contract number P0010025 AM#1 with US Forest Service, Redwood Sciences Laboratory.

Welsh, H.H., K.L. Pope, and D. Boiano. 2006. Subalpine amphibian distributions related to species palatability to non-native salmonids in the Klamath Mountains of northern California. Diversity and Distributions 12: 298-309.

Young, A.G., and G.M. Clarke (Eds.). 2000. Genetics, Demography and Viability of Fragmented Populations. Cambridge University Press, Cambridge, UK.

Zavaleta, E. S., R. J. Hobbs, and H. A. Mooney. 2001. Viewing invasive species removal in a whole- ecosystem context. Trends in Ecology and Evolution 16: 454-459.

LIZARDS

Cedar Key Mole Skink (*Plestiodon egregius insularis*)

Colorado Checkered Whiptail (*Aspidoscelis neotesselata*)

Colorado Desert Fringe-toed Lizard (*Uma notata*)

Florida Scrub Lizard (*Sceloporus woodi*)

Panamint Alligator Lizard (*Elgaria panamintina*)

Reticulate Collared Lizard (*Crotaphytus reticulatus*)

Sandstone Night Lizard (*Xantusia gracilis*)

Arizona Night Lizard (*Xantusia arizonae*)

Bezy's Night Lizard (*Xantusia bezyi*)

Yuman Desert Fringe-toed Lizard (*Uma rufopunctata*)


USGS Photo of *Plestiodon egregius*

Scientific Name:

Plestiodon egregius insularis

Common Name:

Cedar Key Mole Skink

G Rank:

T1

IUCN Red List:

Subspecies not ranked

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

The Cedar Key mole skink (*Eumeces egregius insularis*) is confined to islands in the vicinity of Cedar Key, Levy County, Florida (NatureServe 2011). It has been recorded on Seahorse Key, Cedar Key, and Airstrip Island, and it may occur on other larger islands near Cedar Key (Christman 1992). With such a restricted range – found only on Florida's imperiled coastlands, the species is vulnerable to extinction from disturbance of its sparse remaining undisturbed habitat for shoreline development and other activities.

Habitat:

The mole skink can be found under driftwood and tidal wrack along the shore, as well as farther inland in loose sand (Christman 1992).

Biology and Taxonomy:

Subspecies *insularis* was proposed as a distinct species by Collins (1991), but no supporting data were presented, and other authors have not adopted this proposal.

Nothing is known concerning the life history of the Cedar Key mole skink but it is probably similar to that described by Mount (1963) for the mainland race (Christman 1992). It likely lays from 3 to 7 eggs once a year under debris, usually in a nest cavity constructed in the sand. Females remain with the eggs until hatching, which probably offers protection from small predators. The skink likely eats mainly marine arthropods, which can be found under tidal wrack and driftwood.

Population Status:

The Cedar Key mole skink is endangered because of its rarity across its narrow range and ongoing threats of development and overcollection. NatureServe (2011) considers the Cedar Key mole skink to be critically imperiled. Storm tides and intensive collecting likely have reduced numbers in parts of its range (Christman 1992), but the lizard has not been well monitored (Millsap et al. 1990). The species was a candidate for federal protection until the FWS eliminated the C2 category, and it currently receives no federal protection under the ESA.

THREATS

Habitat alteration and destruction:

The Cedar Key mole skink has been impacted heavily by habitat destruction (Christman 1992, NatureServe 2011). Because it is found along highly valued coastland habitats (Christman 1992), its habitats are vulnerable to degradation and destruction from development, recreation, and other human actions.

Development is a consequence of the human population increases that Florida has experienced. Florida's population increased to 16 million people in 2000, and 80 percent of that growth occurred along the coasts (Paul 1996). More than 75 percent of Florida's human population resides in coastal counties, and up to 49 million people visit Florida annually (Bureau of Economic and Business Research 1999). By 1975, nearly 20 percent of Florida's barrier islands were developed and since then the number has grown dramatically (Lins 1980).

Beachfront development, especially in the form of high-rise buildings on the foredunes, is intense (Doyle et al. 1984, Pilkey et al. 1984). Even if the foredunes are left intact, development of coastal property is a major threat to the skink because of the resulting increased human activity on the beach (see Enge et al. 2003). Wind and water erosion of beach dunes occur where pedestrian or ORV traffic damages vegetation, creating a gap or blowout (Enge et al. 2003). Coastal habitat is also lost to dredging and excavation, spoil disposal, impounding, and sediment diversions (Senner and Howe 1984).

Overutilization:

The Cedar Key mole skink has beautiful coloration and is threatened by overcollection by herp enthusiasts (NatureServe 2011). Christman (1992) explains: "Intense collecting by amateur retile fanciers has drastically reduced its numbers [on Airstrip Island near Cedar Key] over the past 15 years."

Inadequacy of existing regulatory mechanisms:

The Cedar Key mole skink is not state listed but recognized as a "Species of Greatest Conservation Need" because it is rare and biologically vulnerable (Florida Fish and Wildlife Commission, undated). But that status provides no legal protection for the skinks. As such, the skinks remained threatened by overcollection, even on protected areas, such as the Cedar Key National Wildlife Refuge (Christman 1992).

The skink is also threatened because its habitats are inadequately protected. Cox and Kautz (2000) conclude that "The habitat available on public lands and proposed [Strategic Habitat Conservation Area]s falls short of that needed to ensure long-term security. In addition to the low total acreage of habitat with [Managed Areas], habitat patches on [Managed Areas] are closely clustered and provide little independence, so additional habitat might be sought to help expand the geographic range and number of protected populations." Federal protection and designation of critical habitat would help ensure that essential habitats are protected.

Other factors:

Branch et al. (2003) studied three Florida lizards and found that haplotype diversity was lowest for the Cedar Key mole skink, which also has the most restricted distribution. The authors explained that this low diversity may result from loss of haplotypes through genetic drift in small populations, or from founder effects in populations that were colonized by only a few individuals. Indeed, small, isolated populations are more susceptible to extirpations due to stochastic events, human impacts, and environmental factors (Soulé 1987, Begone et al. 1990). Lack of gene flow can cause loss of genetic variability due to random genetic drift (Wright 1931) and inbreeding depression is likely to occur (Franklin 1980). Once these populations are extirpated, their isolation precludes genetic interchange and recolonization of habitat.

The Cedar Keys mole skink is also likely threatened by sea-level rise from climate change as it is restricted to shoreline habitats in the Florida keys. In addition, climate change will likely increase the frequency of major storm events and storm tides have already reduced some habitat (Christman 1992). Most of Florida's coast has been eroding landward at an average of 0.3–0.6 m/yr (Doyle et al. 1984, Pilkey et al. 1984).

References:

Begone, M., J. L. Harper, and C. R. Townsend. 1990. Ecology: Individuals, populations, and communities. Blackwell Scientific Publications.

Branch, L.C., A.M. Clark, P.E.Moler, and B.W. Bowen. 2003. Fragmented landscapes, habitat specificity, and conservation genetics of three lizards in Florida scrub. Conservation Genetics 4: 199–212.

Bureau of Economic and Business Research. 1999. Florida statistical abstract 1999. University of Florida, Gainesville, Florida, USA, *available at* http://ufdc.ufl.edu/UF00080556/00006/thumbs (last visited June 25, 2012).

Christman, S.P. 1992. Cedar Key Mole Skink. In: Rare and Endangered Biota of Florida. Vol. III Amphibians and Reptiles (ed. Moler PE), pp. 223-26. University of Florida Press, Gainesville, Florida.

Collins, J. T. 1991. Viewpoint: a new taxonomic arrangement for some North American amphibians and reptiles. SSAR Herpetol. Review 22: 42-43.

Cox, J., and R. Kautz. 2000. Habitat conservation needs of rare and imperiled wildlife in Florida. Office of Environmental Services, Florida Fish and Wildlife Conservation Commission, Tallahassee, Florida, *available at* http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=34&ved=0CDMQFjADOB4&url=http%3A%2F%2Fwww.nwfwmd.state.fl.us%2Fpermits%2Ferp%2Ferp_downloads%2FHab%2520Con%2520Needs%2520Of%2520R%2520and%2520I%2520Wild%2520In%2520Fl%25202000.pdf&ei=7i_dTsC3Bce42QWJ1933Dw&usg=AFQjCNFdBVsiHdhMkvVnvzq44auSkvRljw (last visited Dec. 5, 2011).

Doyle, L. J., D. C. Sharma, A. C. Hine, O. H. Pilkey, Jr., W. J. Neal, O. H. Pilkey, Sr., D. Martin, and D. F. Belknap. 1984. Living with the West Florida shore. Duke University Press, Durham, North Carolina, USA.

Enge, Kevin M., Brian A. Millsap, Terry J. Doonan, Jeffery A. Gore, Nancy J. Douglass, and Gary L. Sprandel. 2003. Conservation Plans for Biotic Regions in Florida Containing Multiple Rare or Declining Wildlife Taxa. Final Report. Florida Fish and Wildlife Conservation Commission.

Florida Fish and Wildlife Commission. Undated. Scrub Species of Greatest Conservation Need, *available at* http://www.myfwc.com/conservation/special-initiatives/fwli/taking-action/scrub/scrub-sgcn/ (last visited Dec. 5, 2011).

Franklin, I. R. 1980. Evolutionary changes in small populations. In Conservation Biology: An evolutionary-ecological perspective (ed. a. B. A. W. M. E. Soule). pp. 135-149.

Hammerson, G.A. 2007. Plestiodon egregius. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.2. <www.iucnredlist.org>. Downloaded on 05 December 2011.

Lins, H. F., Jr. 1980. Patterns and trends of land use and land cover on Atlantic and Gulf coast barrier islands. U.S. Geological Survey Professional Paper No. 1156.

Millsap, B., Jeffery A. Gore, Douglas E. Runde, Susan I. Cerulean. 1990. Setting Priorities for the Conservation of Fish and Wildlife Species in Florida. Wildlife Monographs: 111: 3-57.

Mount, R.H. 1963. The natural history of the Red-tailed Skink, *Eumeces egregious* Baird. Amer. Mid. Nat. 70(2): 356-85.

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, *available at* http://www.natureserve.org/explorer (Accessed: December 5, 2011).

Paul, R. T. 1996. Reddish egret, Egretta rufescens. Pages 281–294 in J. A. Rodgers, Jr., H. W. Kale II, and H. T. Smith, editors. Rare and endangered biota of Florida. Volume V. Birds. University Press of Florida, Gainesville, Florida, USA.

Pilkey, O. H., Jr., D. C. Sharma, H. R. Wanless, L. J. Doyle, O. H. Pilkey, Sr., W. J. Neal, and B. L. Gruver. 1984. Living with the East Florida shore. Duke University Press, Durham, North Carolina, USA.

Senner, S. E., and M. A. Howe. 1984. Conservation of Nearctic shorebirds. Pages 379–420 in J. Burger and B. L. Olla, editors. Shorebirds: breeding behavior and populations. Behaviour of marine animals. Volume 5. Plenum, New York, New York, USA.

Soulé, M. 1987. Viable populations for conservation. Cambridge University Press, Great Britain. 189 pp.

Wright, S. 1931. Evolution in Mendelian populations. Genetics 16: 97-159.

© 2011 Timothy Burkhardt

Scientific Name:

Aspidoscelis neotesselata (formerly, *Cnemidophorus neotesselatus*)

Common Name:

Colorado Checkered Whiptail (also known as Southeastern Colorado Whiptail)

G Rank:

G2

IUCN Red List:

Near Threatened

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

As a species with a highly restricted range where it faces serious threats, the Colorado checkered whiptail is imperiled and at risk of extinction. The known range of the Colorado checkered whiptail includes only southeastern Colorado, at elevations below 2,135 meters (7,000 feet) (Walker et al. 1997, Hammerson 1999, Stebbins 2003).

Habitat:

This whiptail occurs in valleys, arroyos, canyons, and on hillsides, in areas domiated by plains grassland or juniper woodland, including areas such as parks with minor habitat disturbance (Walker et al. 1997).

Taxonomy and Biology:

Cnemidophorus neotesselata was described in 1997 (Walker et al. 1997). Formerly it was included in *C. tesselatus* (*Aspidoscelis tesselata*). Crother et al. (2003) recognizes *neotesselata* and *tesselata* as distinct species. Collins and Taggart (2009) use the common name "Southeastern Colorado Whiptail" for *A. neotesselata* and "Colorado Checkered Whiptail" for *A. tesselata*. de Quieroz and Reeder (2011) uses the common name "Colorado Checkered Whiptail" for *A. neotesselata*.

The Colorado checkered whiptail is a triploid, parthenogenetic lizard that originated from a hybridization event involving normally parthenogenetic *Aspidoscelis tesselata* and bisexual *Aspidoscelis sexlineata* (Taylor et al. 2006). In parthenogenetic species, reproduction is asexual, and egg cells develop without having been fertilized by male gametes. Consequently, offspring are genetically identical with their mother.

Population Status:

The Colorado checkered whiptail is extremely rare throughout its narrow range in southeastern Colorado, and many historic localities have been lost due to ongoing habitat destruction. As such, it qualifies for endangered status. Extent of occurrence, area of occupancy, number of subpopulations, and population size have declined in recent decades (NatureServe 2011).

When originally described as a distinct species, this lizard was known from about 15-20 localities (Walker et al. 1997). NatureServe (2011) reports that the whiptail has been extirpated or declined in many areas as a result of urbanization or agricultural development (citing Walker et al. 1996, 1997). The IUCN Red List ranks the species as Near Threatened but explains that it almost qualifies for Vulnerable status.

<u>THREATS</u>

Habitat alteration and destruction:

This species has been extirpated from, or has greatly declined in, some areas as a result of urbanization or conversion of habitat to agricultural uses (Walker et al. 1996, 1997). While some populations can persist in areas with minor disturbance (e.g., in parks and at rural landfills) and in nearby undisturbed habitats, habitat loss threatens to severely fragment remaining populations (Hammerson 2007).

Overutilization:

The Colorado checkered whiptail suffers from collection for the pet trade (see, e.g., Repticzone.com) but the extent of the threat is unknown.

Inadequacy of existing regulatory mechanisms:

There are several documented occurrences of this species on the Timpas Unit of the Comanche National Grassland, where suitable habitat occurs (U.S. Forest Service 2008). While these lands are protected from urbanization, other threats such as collection continue to occur on these lands.

It is a species of special concern in Colorado (Colorado DNR 2010) but this status does not afford any legal protection for the lizards or their habitats.

References:

Colorado Dept. of Natural Resources (DNR). 2010. Threatened and Endangered List, *available at* http://wildlife.state.co.us/WildlifeSpecies/SpeciesOfConcern/ThreatenedEndangeredList/Pages/ListOfThreatenedAndEndangeredSpecies.aspx (last visited Dec. 6, 2011).

Collins, J. T., and T. W. Taggart. 2009. Standard common and current scientific names for North American amphibians, turtles, reptiles, and crocodilians. Sixth edition. The Center for North American Herpetology, Lawrance, Kansas. iv + 44 pp.

Crother, B.I., J. Boundy, J.A. Campbell, K. de Quieroz, D. Frost, D.M. Green, R. Highton, J.B. Iverson, R.W. McDiarmid, P.A. Meylan, T.W. Reeder, M.E. Seidel, J.W. Sites, Jr., S.G. Tilley, and D.B. Wake. 2003. Scientific and standard English names of amphibians and reptiles of North America north of Mexico: update. Herpetological Review 34: 198-203.

de Quieroz, K. and T.W. Reeder. 2011. Scientific and Standard English Names of Amphibians and Reptiles of North America North of Mexico, With Comments Regarding Confidence In Our Understanding. Edition 6.1, *available at* http://www.ssarherps.org/pages/comm_names/Lizards.php (last visited July 8, 2012).

Hammerson, G.A. 2007. Aspidoscelis neotesselata. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.2. <www.iucnredlist.org>. Downloaded on 05 December 2011.

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, *available at* http://www.natureserve.org/explorer. (Accessed: December 5, 2011).

Stebbins, R. C. 2003. A field guide to western reptiles and amphibians. Third edition. Houghton Mifflin Company, Boston.

Taylor, H.L., R.J. Rondeau, and J. Sovell. 2006. Alternative ontogenetic pathways to color pattern class b in a newly discovered population of parthenogenetic *Aspidoscelis neotesselata* (Squamata: Teiidae). Herpetological Review 37(1): 40-44.

U.S. Forest Service. 2008. Cimarron and Comanche National Grasslands Land Management Plan – Draft, *available at* http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=83&ved=0CCsQFjACOFA&url=http%3A%2F%2Fwww.fs.fed.us%2Fouternet%2Fr2%2Fpsicc%2Fprojects%2Fforest_revision%2Fprv2%2F080929_prv2_appendix-d.pdf&ei=qEzeTrmhKbCDsAL_quTYBg&usg=AFQjCNED7Qf3vlTkq3AnP3nHdhCM3_AYZA (last visited Dec. 6, 2011).

Walker, J. M., J. E. Cordes, and H. L. Taylor. 1996. Extirpation of the parthenogenetic lizard *Cnemidophorus tesselatus* from historically significant sites in Pueblo County, Colorado. Herpetological Review 27: 16-17.

Walker, James M., James E. Cordes, and Harry L. Taylor. 1997. Parthenogenetic Cnemidophorus tesselatus complex (Sauria: Teiidae): A neotype for diploid *C. tesselatus* (Say, 1823), redescription of the taxon, and description of a new triploid species. Herpetologica 53(2): 233-259.

Walker, J. M., H. L. Taylor, J. E. Cordes, and M. A. Paulissen. 1997. Distributional relationships and community assemblages of three members of the parthenogenetic *Cnemidophorus tesselatus* complex and *C. sexlineatus* (Squamata: Teiidae) at Higbee, Otero County, Colorado. Herpetological Natural History 5: 69-78.

Scientific Name:

Uma notata

Common Name:

Colorado Desert Fringe-toed Lizard

G Rank:

G3

IUCN Red List:

Near Threatened

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

The Colorado Desert fringe-toed lizard has a narrow and fragmented distribution that includes southeastern California and northeastern Baja. Hammerson and Hollingsworth (2007) describe the range as follows:

> The range includes southeastern California in the United States, and northeastern Baja California in Mexico (Pough 1977, Jennings and Hayes 1994, Trepanier and Murphy 2001, Grismer 2002, Stebbins 2003). Its populations are discontinuously distributed across disjunct patches of suitable habitat. In Mexico it ranges south from the United States border to the western edge of the Sierra los Cucapás. There is a disjunct subpopulation further south on the northern edge of the Sierra las Pintas (and it might occur further south in this range, but surveys have not been carried out). The species might also occur on the eastern slopes of the Sierra Juárez. Its elevational range extends from below sea level at the edge of the

Salton Sea to around 180 m (600 feet) northeast of Borrego Springs in San Diego County, California (Jennings and Hayes 1994, Stebbins 2003).

Habitat:

This lizard is restricted to sparsely vegetated windblown sand of dunes, flats, riverbanks, and washes (Stebbins 2003). It also uses areas with extensive sand hummocks around the bases of plants (Grismer 2002). It requires fine, loose sand for burrowing; vegetation is usually scant, consisting of creosote bush or other scrubby growth (Stebbins 2003). Eggs are laid in subsurface burrows in sand (NatureServe 2011).

Biology and Taxonomy:

Trepanier and Murphy (2001) concluded that for the northern species of *Uma* existing data support either a two-species arrangement (*Uma scoparia*, *Uma notata*) or five-species classification (*Uma scoparia*, *Uma notata*, *Uma inornata*, and *Uma rufopunctata*, plus an undescribed species from Mohawk Dunes, Arizona). Trepanier and Murphy prefered the latter arrangement, and Crother et al. (2003) adopted the taxonomy preferred by Trepanier and Murphy. ITIS (2011) and Collins and Taggart (2009) also recognize *Uma notata* as a valid species.

Surface activities of the Colorado Desert fringe-toed lizard may be in response to fluctuating temperatures (Cowles 1941). During early spring and fall, they are active mid-day (Pamero 2000). From May to September, lizards are active in the morning and late afternoon, retreating underground when temperatures are high (Palmero 2000). Hibernation occurs from November to February (Mayhew 1964). Juveniles may not become completely torpid during the winter (Palmero 2000). Breeding occurs from May to August, and reproductive activity of males may be delayed during dry winters, probably reflecting food shortages (Palmero 2000). Females contain oviductal eggs from May to mid-August, regardless of winter rainfall (Palmero 2000). Clutch size varies from one to five eggs (mean two eggs), and two to three successive clutches per year may be produced (Palmero 2000). Most young reach sexual maturity in their second summer after hatching, at sizes of 80 mm (3.2 in) and 70 mm (2.8 in), for males and females, respectively (Mayhew 1966, 1967). See Mayhew (1966) for additional information on the reproductive biology of the species. Life history is described by Palmero (2000). Burrowing ecology is described by Pough (1970).

Population Status:

The Colorado Desert fringe-toed lizard is declining across its narrow and fragmented range and has been extirpated from several localities. Specifically, Jennings and Hayes (1994) mapped approximately 34 extant locations in California and documented that this lizard has been extirpated from at least four locations. There have been observed declines in some areas and the species faces many threats. The IUCN Red List ranks the species as Near Threatened but explains that it is close to qualifying for Vulnerable. The species was a candidate for federal protection until the FWS eliminated the C2 category, and it currently receives no federal protection under the ESA.

THREATS

Habitat alteration and destruction:

The Colorado Desert fringe-toed lizard is a habitat specialist (NatureServe 2011). The extent and quality of its habitat are declining (Hammerson and Hollingsworth 2007). The lizard is susceptible to soil compaction due to foot or vehicle activity because it requires fine, loose sand for burrowing (NatureServe 2011). To be sure, populations have been negatively impacted by off-road vehicle damage to sand dune habitat (Jennings and Hayes 1994). Luckenbach and Bury (1983) found significantly fewer Colorado Desert fringe-toed lizards on ORV-impacted plots than control plots.

Other threats to habitat include commercial and residential development and the placement of windbreaks and other structures that stabilize sand (NatureServe 2011). Some populations have also been impacted by the spread of intensive agriculture, for example around Mexicali (Hammerson and Hollingsworth 2007).

Overutilization:

State regulations allow collection of the lizard but overcollection has not been documented as a threat.

Inadequacy of existing regulatory mechanisms:

There is a need to protect sites from intrusion by vehicles and from initiatives to prevent the movement of wind-blown sand (Turner et al. 1984, Hammerson and Hollingsworth 2007). Federal protection would help ensure that essential habitats are protected from such disturbances.

In California, the species has state special concern status (California Dept. of Fish and Game 2011) and is a BLM sensitive species in California (BLM undated). These statuses reflect the rarity of the species but do not afford protection to the species. Personal collection of two animals per day is allowed. CCR § 5.60(b). Federal protection would ban take and result in important habitat protections.

References:

BLM. Undated. Colorado Desert fringe-toed lizard, *available at* http://www.blm.gov/ca/forms/wildlife/details.php?metode=serial_number&search=2520&detaillabelc=Colorado%20Desert%20fringe-toed%20lizard&detaillabels=Uma%20notata (last visited Dec. 6, 2011).

California Dept. of Fish and Game. 2011. Special Animals List, *available at* http://www.dfg.ca.gov/biogeodata/cnddb/pdfs/SPAnimals.pdf (last visited Dec. 6, 2011).

Collins, J. T., and T. W. Taggart. 2009. Standard common and current scientific names for North American amphibians, turtles, reptiles, and crocodilians. Sixth edition. The Center for North American Herpetology, Lawrance, Kansas. iv + 44 pp.

Cowles, R. B. 1941. Observations on the winter activities of desert reptiles. Ecology 22: 125-140.

Crother, B.I., J. Boundy, J.A. Campbell, K. de Quieroz, D. Frost, D.M. Green, R. Highton, J.B. Iverson, R.W. McDiarmid, P.A. Meylan, T.W. Reeder, M.E. Seidel, J.W. Sites, Jr., S.G. Tilley, and D.B. Wake. 2003. Scientific and standard English names of amphibians and reptiles of North America north of Mexico: update. Herpetological Review 34: 198-203.

Grismer, L. L. 2002. Amphibians and reptiles of Baja California including its Pacific islands and islands in the Sea of Cortes. University of California Press, Berkeley. xiii + 399 pp.
Hammerson, G.A. and Hollingsworth, B. 2007. Uma notata. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.2. <www.iucnredlist.org>. Downloaded on 06 December 2011.

ITIS. 2011. Uma notata Baird, 1859, *available at* http://www.itis.gov/servlet/SingleRpt/SingleRpt (last visited Dec. 6, 2011).

Jennings, M. R., and M. P. Hayes. 1994. Amphibian and reptile species of special concern in California. Final Report submitted to the California Department of Fish and Game, Inland Fisheries Division. Contract No. 8023. 255 pp.

Luckenbach, R.A. and R. Bruce Bury. 1983. Effects of Off-Road Vehicles on the Biota of the Algodones Dunes, Imperial County, California. Journal of Applied Ecology 20(1): 265-286.

Mayhew, W.W. 1964. Photoperiodic responses in three species of the lizard genus Uma. Herpetologica 20: 95-113.

Mayhew, W.W. 1966. Reproduction in the Arenicolous Lizard *Uma notata*. Ecology 47(1): 9-18.

Mayhew, W. W. 1967. Comparative reproduction in three species of the genus *Uma*. Pages 45-61 in W. W. Milstead, ed. Lizard Ecology: A Symposium. Univ. Missouri Press, Columbia. 300pp.

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, *available at* http://www.natureserve.org/explorer (Accessed: December 6, 2011).

Palermo, L. 2000. California Wildlife Habitat Relationships System: Colorado Desert fringe-toed lizard, *available at*
http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=65&ved=0CEYQFjAEODw&url=http%3A%2F%2Fnrm.dfg.ca.gov%2FFileHandler.ashx%3FDocumentVersionID%3D1

7643&ei=5VzeTrDpHaHU2AW8waX2BA&usg=AFQjCNGnlkjbkqwlXZh477AvRNyb9Co3wA (last visited Dec. 6, 2011).

Pough, F. H. 1970. The burrowing ecology of the sand lizard, *Uma notata*. Copeia 1970(1): 145-58.

Pough, F. H. 1977. *Uma notata*. Catalogue of American Amphibians and Reptiles 197: 1-2.

Stebbins, R. C. 2003. A field guide to western reptiles and amphibians. Third edition. Houghton Mifflin Company, Boston.

Trepanier, T. L., and R. W. Murphy. 2001. The Coachella Valley fringe-toed lizard (*Uma inornata*): genetic diversity and phylogenetic relationships of an endangered species. Molecular Phylogenetics and Evolution 18: 327-334.

Turner, F.B., Donald C. Weaver, James C. Rorabaugh. 1984. Effects of Reduction in Windblown Sand on the Abundance of the Fringe-Toed Lizard (*Uma inornata*) in the Coachella Valley, California. Copeia 1984(2): 370-378.

Wilgenbusch, J., and K. de Queiroz. 2000. Phylogenetic relationships among the phrynosomatid sand lizards inferred from mitochondrial DNA sequences generated by heterogeneous evolutionary processes. Systematic Biology 49: 592-612.

Scientific Name:

Sceloporus woodi

Common Name:

Florida Scrub Lizard

G Rank:

G3

IUCN Red List:

Near Threatened

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

The Florida scrub lizard occurs in four disjunct areas in central and southern Florida: vicinity of Ocala National Forest in northern peninsular Florida, inland sites in Polk and Highlands counties, Atlantic coast scrubs in central and southern Florida (Brevard County to Broward County), and Gulf Coast scrubs in Lee and Collier counties (DeMarco 1992, NatureServe 2011). Severe range contractions have been observed in Collier County on the southwest coast and in the scrub along the sandy southeastern coastline (Bartlett and Bartlett 1999).

The lizard is suffering from habitat fragmentation throughout its range. Scrub patches within the northern and southern parts of the Lake Wales Ridge (Arbuckle and Carter Creek, Archbold and Venus) once formed part of a very large, highly connected scrub ridge bisected by Josephine Creek (Lohrer and Commings 1993). During the last century, more than 85 percent of the xeric uplands of the Lake Wales Ridge has been converted to human use resulting in large-scale habitat fragmentation (McDonald and Hamrick 1996).

Habitat:

Hammerson (2007) describes the habitat as follows:

> It is largely restricted to evergreen oak scrub and young sand pine scrub with ample open space; it is less common in the ecotone between scrub and sandhills, sandhills surrounded by scrub, scrubby flatwoods, and citrus groves. It prefers sites with open sandy areas (for nesting, basking, and foraging) in close proximity to mature trees (*Pinus* or *Quercus*) that can provide shade and perch sites. Development of a closed canopy (e.g., in the absence of fire) results in increasingly unsuitable habitat. It never occurs in nonxeric sites. The species is mostly terrestrial but commonly perches low on tree trunks. See De Marco (1992) for further information. Eggs are laid in soil (e.g., *Geomys* and tortoise mounds) (Ashton and Ashton 1985).

In addition, the Florida scrub lizard prefers recently burned areas (Tiebout and Anderson 1997, 2001) and tends to occur in higher densities with shorter time-since-fire (unpublished data cited in Schrey et al. 2011).

Biology and Taxonomy:

Hammerson (2007) explains that in the past, some authors questioned whether *S. woodi* is sufficiently differentiated from *S. undulatus* to be considered a distinct species. More recently, *S. woodi* has consistently recognized as a distinct species (see, e.g., Collins and Taggart (2009)).

Mating of the Florida scrub lizard occurs from the end of March through late June; two to seven eggs are laid three to four weeks later and hatch in about 75 days (Iverson 1974, Fogarty 1978, Ashton and Ashton 1985). The Florida scrub lizard is a sit-and-wait predator (Schoener 1971) that opportunistically feeds mainly on ground-dwelling arthropods. Individuals spend most of their time foraging on the ground or from low perches on tree trunks, logs, or pine cones (Jackson 1973b). For additional natural history information, see Enge et al. (1986). Research on locomotion of the species is provided by McElroy et al. (2012). Foraging behavior and diet of the Florida scrub lizard is described by Williams and McBrayer (2011).

Population Status:

The Florida scrub lizard qualifies as endangered because the species has experienced population declines and range contractions due to the widespread and ongoing loss of scrub habitats. It was historically far more widespread and numerous on Florida's now largely developed sandy ridges (Bartlett and Bartlett 1999). The area of occupancy, number of subpopulations, and population size are undoubtedly declining (NatureServe 2011). The Florida Committee on Rare and Endangered Plants and Animals, a group of experts on the flora and fauna of Florida, has classified this species as threatened. The IUCN Red List ranks the species as near threatened but explains that it is close to qualifying as vulnerable. The species was a candidate for federal protection until the FWS eliminated the C2 category, and it currently receives no federal protection under the ESA.

THREATS

Habitat alteration and destruction:

The Florida scrub lizard is a habitat specialist and the primary threat to the species is the loss of the scrub habitat upon which it depends. The Florida sand pine scrub habitat is one of the most endangered ecosystems in the southeastern United States (Millsap et al. 1990, Peters and Noss 1995, Noss et al. 1995, Branch and Hokitt 2000, Enge et al. 2003, Tucker et al. 2011). Nearly all remaining Florida scrub now exists as small managed preserves (Turner et al. 2006). During the last two decades, large areas of scrub have been converted by agriculture (especially citrus, Mosesso 1996), urbanization, forestry, and mining (e.g. Myers 1990, 1991; McCoy and Mushinsky 1992; NatureServe 2011; Hammerson 2007). Only about 1.2 percent of Florida is now covered by scrub habitat, which has declined precipitously since European settlement (Kautz et al. 1993, Enge et al. 2003).

A primary driver of the loss has been citrus groves and housing developments (Enge et al. 1986). For example, by 1981 on the southern Lake Wales Ridge in Highlands County, there had been a 64 percent loss of xeric vegetative types with 44 percent of the loss due to cultivation, 14 percent to housing development and 6 percent to improved pasture (Peroni and Abrahamson 1985). An additional 10 percent of xeric vegetation had been moderately disturbed, primarily by road construction for future housing subdivisions. Florida's citrus production doubled from 1960 to 1978 (Fernald 1981) and has accelerated following severe freezes in the 1980s (Enge et al. 1986). Most historic scrub lizard localities in Lake (except those in the national forest) and northern Polk counties no longer contain scrub habitat (Enge et al. 1986). In southern Polk and Highlands counties, large-scale conversion of scrub habitat to citrus production has fragmented parts of its range (Enge et al. 1986). Most scrub lizard habitat is presently being converted to other uses or will be converted in the foreseeable future (Enge et al. 1986).

Along the Atlantic Coast, scrub habitat has been lost to citrus production in the past but the greatest ongoing threats are residential, commercial and recreational developments because most remaining habitat is owned by developers (Enge et al. 1986). Most historic scrub lizard localities in Dade, Broward, and southern Palm Beach counties have been lost or will be lost in the near future (Enge et al. 1986). Historic locations along the southwestern Gulf Coast are either already cleared or on property owned by a developer (Enge et al. 1986).

In particular, population pressure along the coasts has resulted in the loss of coastal scrub habitat, especially in southern counties (Enge et al. 1986). For example, from 1960 to 1980, the population of Florida nearly doubled from 4.95 to 9.75 million (Terhune 1982). Given the population pressure and because most remaining scrub habitat is owned by developers, there will be a greater loss of scrub in the future (Enge et al. 1986).

Even on areas protected from citrus protection and development, habitat is being lost through commercial logging or fire suppression. The largest remaining contiguous tract of Florida scrub is within Ocala National Forest. Natural and anthropogenic fires are suppressed in the Ocala National Forest, and the landscape is managed primarily for pulp wood production (Tiebout and Anderson 2001). Forest management practices that allow the build up of excessive course woody debris harms the lizard because it requires open sandy spaces.

Moreover, because the Florida scrub lizard is an early successional species (Campbell and Christman 1982, Mushinsky 1985), it is apparently able to colonize clearcuts only within about the first 10 years after harvest. As young sand pines grow and the canopy closes, the local scrub lizard population plummets (Greenberg et al. 1994, Tiebout and Anderson 1997). As a young stand ages, bare sand becomes covered with a complex and irregular layer of pine needle and leaf litter, lichens, and course woody debris (Anderson and Tiebout 1993). This new substratum is likely to provide unfavorable thermal environments, reduce prey availability, or interfere with lizard movements (Miles 1994), foraging, visual vigilance for predators, or other behaviors. To eliminate this surface cover, severe burns must be prescribed (Tiebout and Anderson 2001)

As such, suppression of fires, which are a natural component of the scrub ecosystem, also has resulted in habitat loss for scrub lizards (University of Florida 2009). Aggressive fire control programs in the Ocala National Forest have nearly eliminated the landscape effects of wildfire (Whelan 1995). Fire suppression is also likely impacting the genetic composition of the species, as Schrey et al. (2011) found that genetic diversity of the Florida scrub lizard is increased by fire.

Because the Florida scrub lizard has a high habitat specificity and limited vagility, it is difficult for the lizards to disperse to a new site when the habitat becomes unsuitable (Hokitt et al. 1999, Tucker et al. 2011). McCoy et al. (2004) documented a population decline over three years in a small habitat fragment. The decline was associated with a decline in the survivorship of reproductive females, possibly due to increased risk of predation. Hokit and Branch (2003) found that production of hatchlings, recruitment, survivorship, and lizard abundance were positively associated with habitat patch size (at patch sizes between 2 and 6 ha). Many remaining habitat patches are too small for the species, and ongoing development will further reduce the size of remaining fragments.

The majority of scrub lizard sites in open sand pine, oak, and rosemary scrubs exhibit some degree of OHV use (Enge et al. 1986). While light use by recreational vehicles may create sandy habitat preferred by the lizards, areas of heavy use are avoided likely due to impairment in running ability or reduction in prey abundance (Enge et al. 1986). As such, intense recreational vehicle use is another threat to the species.

Overutilization:

The Florida scrub lizard is a unique and beautiful Florida endemic and therefore is of interest to both amateur and scientific collectors (Enge et al. 1986), who may legally collect them in unlimited numbers (Nanjappa and Conrad 2012).

Disease and predation:

A variety of parasites have been documented to afflict the Florida scrub lizard (Telford and Bursey 2003) but are unlikely to threaten the species.

Inadequacy of existing regulatory mechanisms:

The scrub lizard is not listed as a threatened or endangered species at the state or federal level (University of Florida 2009). The Florida Scrub Lizard is listed as a species of greatest conservation need in Florida (FFWCC 2005), and the Florida Committee on Rare and Endangered Plants and Animals, a group of experts on the flora and fauna of Florida, has classified this species as threatened. But these statuses only reflect the lizard's imperilment and do not create any legal protections. Current law allows unlimited collection of the lizards (Nanjappa and Conrad 2012).

Some habitat is protected on state and federal land. Hammerson (2007) explains, "Populations occur in several sites on the Avon Park Bombing Range and at least one population occurs on each of the following areas: Archbold Biological Station, Highlands Hammock State Park, Johnathan Dickinson State Park, Saddle Blanket Lakes Preserve. There are multiple occupied sites on the Lake Arbuckle State Forest/State Park, Ocala National Forest, and Tiger Creek Preserve." Yet even on these protected lands, the lizard experiences the threat of habitat degradation because it requires active management with fire to maintain suitability. Moreover, these lands represent just a fraction of the lizard's historic range and are not sufficient to ensure viability. Protection of scrub is not addressed by local regulations (Enge et al. 1986).

The Florida scrub ecosystem is the focus of some state and national conservation initiatives (The Nature Conservancy 1991, USFWS 1991, Branch and Hokitt 2000). But these efforts are solely voluntary and unlikely to preserve adequate habitat for the lizard. Protection of the Florida scrub lizard as a federally endangered species would lead to management practices specific to the lizard and offer much needed protection of its habitats. To be sure, long-term survival of the Florida scrub lizard is dependent upon preservation of sufficient scrub habitat (University of Florida 2009).

Other factors:

Isolation is another threat to the lizard. Florida scrub lizards are poor dispersers that are not likely to move among distant scrub patches without intermediate scrub patches (Greenberg et al. 1994, Tiebout and Anderson 1997, Hokit et al. 2010). Researchers have demonstrated that Florida scrub lizard populations are effectively isolated by broad zones of unsuitable habitat and by anthropogenic factors (Clark et al. 1999, Hokit et al. 1999). Loss of habitat has caused a decline in scrub lizard populations and increased isolation of remaining populations and is likely resulting in decreased genetic diversity (University of Florida 2009, Branch et al. 2003). Field studies and landscape modelling have demonstrated that scrub lizards are not likely to recolonize distant patches following local extinctions, particularly if intervening scrub patches have been removed (Tiebout and Anderson 1997, Hokit et al. 1999). To be sure, small, isolated populations are more susceptible to extirpations due to stochastic events, human impacts, and environmental factors (Soulé 1987, Begone et al. 1990). Lack of gene flow may cause loss of genetic variability due to random genetic drift (Wright 1931) and inbreeding depression is likely to occur (Franklin 1980). Once these populations are extirpated, their isolation precludes genetic interchange and recolonization of habitat.

In addition, Florida scrub lizards occupying habitat near human habitation may be easily preyed upon by house cats because of their conspicuous use of open areas on the ground (Enge et al. 1986).

Finally, the application of herbicides and pesticides poses a possible threat to both populations of scrub lizards and their prey (Enge et al. 1986).

References:

Anderson, R.A. and H. M. Tiebout III. 1993. Final Report, The Nature Conservancy, Washington, DC.

Ashton, R. E., Jr., and P. S. Ashton. 1985. Handbook of reptiles and amphibians of Florida. Part two. Lizards, turtles & crocodilians. Windward Pub., Inc., Miami. 191 pp.

Bartlett, R. D., and P. P. Bartlett. 1999. A field guide to Florida reptiles and amphibians. Gulf Publishing Company, Houston, Texas. xvi + 278 pp.

Begone, M., J. L. Harper, and C. R. Townsend. 1990. Ecology: Individuals, populations, and communities. Blackwell Scientific Publications.

Branch, L.C., D.G. Hokitt. 2000. A comparison of scrub herpetofauna on two Central Florida sand ridges. Florida Scientist 63(2): 108.

Branch, L.C., A.M. Clark, P.E. Moler, and B.W. Bowen. 2003. Fragmented landscapes, habitat specificity and conservation genetics of three lizards in Florida scrub. Conservation Genetics 4(2): 199-212.

Campbell, H.W., and S. P. Christman. 1982. The herpetological components of the Florida sandhill and pine scrub associations. In N. J. Scott (ed.), Herpetological Communities, pp. 163-171. USFWS Wildlands Res. Rep. 13, Washington, DC.

Clark, A.M., B.W. Bowen, and L.C. Branch. 1999. Effects of natural habitat fragmentation on an endemic scrub lizard (*Sceloporus woodi*): An historical perspective based on a mitochondrial DNA gene genealogy. Molecular Ecology 8: 1093–1104.

Collins, J. T., and T. W. Taggart. 2009. Standard common and current scientific names for North American amphibians, turtles, reptiles, and crocodilians. Sixth edition. The Center for North American Herpetology, Lawrance, Kansas. iv + 44 pp.

DeMarco, V. 1992. Florida scrub lizard *Sceloporus woodi* Stejneger. Pages 141-145 in P. E. Moler, editor. Rare and endangered biota of Florida. Vol. III. Amphibians and reptiles. Univ. Press of Florida.

Enge, K. M., M. M. Bentzien, and F. Percival. 1986. Florida scrub lizard status survey. Tech. Rept. No. 26 to U.S. Fish Wildl. Serv., Jacksonville End. Sp. Office, Florida, *available at* http://ufdc.ufl.edu/UF00067421/00001 (last visited June 26, 2012).

Enge, K. M., B. A. Millsap, T. J. Doonan, J. A. Gore, N. J. Douglass, and G. L. Sprandel. 2003. Conservation plans for biotic regions in Florida containing multiple rare or declining wildlife taxa. Final Report. Florida Fish and Wildlife Conservation Commission, Tallahassee, Florida, USA.

Fernald, E.A. 1981. Atlas of Florida. Fla. State Univ. Found. Inc. Tallahassee. 276 pp.

Florida Fish and Wildlife Conservation Commission (FFWCC). 2005. Florida's wildlife legacy initiative. Florida's comprehensive wildlife strategy. Florida Fish and Wildlife Conservation Commission, Tallahassee (FL).

Fogarty, M.. 1978. Florida Scrub Lizard. Pages 56-57 in R.W. McDiarmid editor. Rare and endangered biota of Florida. Vol. III. Amphibians and reptiles. Univ. Press of Florida.

Franklin, I. R. 1980. Evolutionary changes in small populations. In Conservation Biology: An evolutionary-ecological perspective (ed. a. B. A. W. M. E. Soule). pp. 135-149.

Greenberg, C.H., D.G. Neary, and L.D. Harris. 1994. Effect of high-intensity wildfire and silvicultural treatments on reptile communities in sand-pine scrub. Conserv Biol. 8: 1047–1057.

Hammerson, G.A. 2007. Sceloporus woodi. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.2. <www.iucnredlist.org>. Downloaded on 06 December 2011.

Hokit, D. G., and L. C. Branch. 2003. Habitat patch size affects demographics of the Florida scrub lizard (*Sceloporus woodi*). Journal of Herpetology 37: 257-265.

Hokit, D. G., B. M. Smith, and L. C. Branch. 1999. Effects of landscape structure in Florida scrub: a population perspective. Ecological Applications 9: 124-134.

Hokit, D.G., M. Ascunce, J. Ernst, L.C. Branch, and A.M. Clark. 2010. Ecological metrics predict connectivity better than geographic distance. Conserv. Genet. 11: 149–159.

Iverson, J.B. 1974. Eggs and hatchlings of the Florida Scrub Lizard. Florida Scientist: 37: 169-72.

Jackson, J.F. 1973. The phenetics and ecology of a narrow hybrid zone. Evolution 27: 58.

Lohrer, F., and D. Commings. 1993. Fiftieth Anniversary Biennial Report 1991–92. Archbold Biological Station, Lake Placid, Florida.

McCoy, E.D., and H.R. Mushinsky. 1992. Rarity of organisms in the sand pine scrub habitat of Florida. Conservation Biology 6: 537-548.

McCoy, E. D., P. P. Hartmann, and H. R. Mushinsky. 2004. Population biology of the rare Florida scrub lizard in fragmented habitat. Herpetologica 60: 54-61.

McDonald, D.B., and J.L. Hamrick. 1996. Genetic variation in some plants of Florida scrub. American Journal of Botany 83: 21-27.

McElroy, E.J., K.L. Archambeau, and L.D. McBrayer. 2012. The correlation between locomotor performance and hindlimb kinematics during burst locomotion in the Florida scrub lizard, Sceloporus woodi. Journal of Experimental Biology 215(3): 442-54.

Miles, D. B. 1994. Covariation between morphology and locomotor performance in Sceloporine lizards. In L. J. Vitt and E. R. Pianka (eds.), Lizard Ecology: Historical and Experimental Perspectives, pp. 207- 235. Princeton Univ. Press, Princeton, NJ.

Millsap, B.A., J.A. Gore, D.E. Runde, and S.I. Cerulean. 1990. Setting priorities for the conservation of fish and wildlife species in Florida. Wildl. Monogr. 11: 1-57.

Mushinsky, H. R. 1985. Fire and the Florida sandhill herpetofaunal community: with special attention to responses of Cnemidophorus sexlineatus. Herpetologica 41: 333-342.

Myers, R.L. 1990. Scrub and high pine. In R. L. Myers and J. J. Ewel (eds.), Ecosystems of Florida, pp. 150-193. Univ. of Central Florida Press, Orlando.

Myers, R.L. 1991. Condominiums, trailer parks and high intensity fires: the future of sandpine scrub reserves in Florida. Proc. Tall Timbers Fire Ecol. Conf. 17: 301.

Nanjappa, P. and Paulette M. Conrad 2012. State of the Union: Legal Authority Over the Use of Native Amphibians and Reptiles in the United States, *available at* http://www.fishwildlife.org/files/SOU_FULL-lo-res.pdf (last visited April 30, 2012).

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, *available at* http://www.natureserve.org/explorer. (Accessed: December 6, 2011).

Noss, R.E, E.T. Laroe III, and J.M. Scott. 1995. Endangered ecosystems of the United States: a preliminary assessment of loss and degradation. U.S. Department of the Interior, National Biological Service, Biological Rep. no. 28, Washington, DC.

Peters, R. L., and R. F Noss. 1995. America's endangered ecosystems. Defenders 70: 16-27.

Schrey, A.W., K.G. Ashton, S. Heath, E.D. McCoy, and H.R. Mushinsky. 2011. Fire Alters Patterns of Genetic Diversity Among 3 Lizard Species in Florida Scrub Habitat. Journal of Heredity 102(4): 399-409.

Schoner, T.W. 1971. Theory of feeding strategies. Annual Review of Ecology and Systematics 2: 369-404.

Sites, J. W., Jr., J.W. Archie, C.J. Cole and O. Flores-Villela. 1992. A review of phylogenetic hypotheses for lizards of the genus *Sceleporus* (Phrynosomatidae): implications for ecological and evolutionary studies. Bull. Am. Mus. Nat. Hist. (213): 1-110.

Soulé, M. 1987. Viable populations for conservation. Cambridge University Press, Great Britain. 189 pp.

Stout, I.J., D. R. Richardson, and R. E. Roberts. 1988. USDA Forest Service, General Technical Report RM-166, Fort Collins, CO, 1988:98-108, unpubl.

Telford, Sam R., Jr., and C.R. Bursey. 2003. Comparative parasitology of squamate reptiles endemic to scrub and sandhills communities of North-central Florida, U.S.A. Comparative Parasitology 70(2): 172-181.

Terhune, F.W. (editor). 1982. Florida statistical abstract. Univ. Presses Fla.

The Nature Conservancy. 1991. Lake Wales/Highland Ridge Ecosystem Carl Project Proposal. Unpublished report. Archbold Biological Station, Florida Natural Areas Inventory.
Tiebout, H.M., and R.A. Anderson. 1997. A comparison of corridors and intrinsic connectivity to promote dispersal in transient successional landscapes. Conserv Biol. 11: 620–627.

Tucker, D.B., J.S. Harrison, and L.C. McBrayer. 2011. Genetic isolation of *Sceloporus woodi* due to mature forests and major roadways. Society for Integrative and Comparative Biology – Meeting Abstracts, *available at* http://www.sicb.org/meetings/2011/schedule/abstractdetails.php3?id=1047 (last visited Dec. 6, 2011).

Tiebout, H.M., III and R.A. Anderson. 2001. Mesocosm Experiments on Habitat Choice by an Endemic Lizard: Implications for TimberManagement. Journal of Herpetology 35(2): 173-185.

Turner ,W.R., D.S. Wilcove, H.M. Swain. 2006. State of the scrub, *available at*: http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=1&ved=0CE8QFjAA&url=http%3A%2F%2Fwww.archbold-station.org%2Fstation%2Fdocuments%2Fpublicationspdf%2FTurner_etal-2006-StateotScrub.pdf&ei=xijFT-qAGoas9ASCncSnCw&usg=AFQjCNHhMU8vEf1PmObaoQlS2XJFaGGt-A&sig2=PTe9iqA4MwCq_NR-Xcf1DA (last visited May 29, 2012).

University of Florida - Institute of Food and Agricultural Sciences Extension. 2009. Florida scrub lizard (*Sceloporus woodi*), *available at* http://edis.ifas.ufl.edu/uw133 (last visited Dec. 6, 2011).

U.S. Fish and Wildlife Service. 1991. Preliminary project proposal: Proposed establishment of Lake Wales Ridge National Wildlife Refuge. USFWS, Atlanta, GA.

Whelan, R. J. 1995. The Ecology of Fire. Cambridge Univ. Press, Cambridge.

Williams, S.C., and L. McBrayer. 2011. Attack-based indices, not movement patterns, reveal intraspecific variation in foraging behavior. Behavioral Ecology 22(5): 993-1003.

Wright, S. 1931. Evolution in Mendelian populations. Genetics 16: 97-159.

Scientific Name:

Elgaria panamintina

Common Name:

Panamint Alligator Lizard

G Rank:

G2

IUCN Red List:

Vulnerable

<u>NATURAL HISTORY, BIOLOGY, AND STATUS</u>

Range:

The small range of the panamint alligator lizard is severely fragmented (Hammerson 2007). The species is known only from desert mountains of Inyo and Mono counties, east-central California (Banta et al. 1996, Stebbins 2003). Elevational range is about 2,500-7,500 feet (760-2,300 meters) (NatureServe 2011).

Habitat:

NatureServe (2011) reports:

> This lizard occurs in regions dominated by scrub desert, Joshua-tree woodland, and the lower edge of the pinyon-juniper belt (Stebbins 1985). Most known locations are in canyon riparian zones below permanent springs; but individuals may range into talus slopes some distance from the immediate riparian zone (Good 1988, Jennings and Hayes 1994, Banta et al. 1996). Individuals have been

> found under willow thickets along watercourses, under shrubs in drier areas, and in rock slides. When inactive, the lizards hide underground, under stones or wood, or in crevices. Habitats in order of decreasing favorability: (1) along creeks with riparian vegetation, (2) along small springs with riparian vegetation, (3) near small springs in pinyon-juniper or Joshua tree woodland, (4) pinyon juniper and Joshua tree woodland in canyons or washes. Occupied riparian zones are typically only a few meters wide and less than 3 km long (Jennings and Hayes 1994).

Marlow (2000) explains: "All specimens have been taken near permanent water in canyons or in talus near dense vegetation. This species appears to occupy a habitat that is relictual from former wetter times. As the desert has dried out it has been restricted to a few moist mountain localities."

Biology:

Comprehensive studies and field data on the natural history of the panamint alligator lizard are lacking, but various aspects of the natural history are based on incidental field observations (see Jennings and Hayes 1994, Banta et al. 1996) and infrequent monitoring of pitfall traps (Banta 1962, 1963). The species appears active from April-October, with peak activity periods in June and a decrease in activity and aestivation occurring during the months of July and August (Banta 1963). Although typically active during the day, the lizards are nocturnal during the hot summer months (Dixon 1975). The panamint alligator lizard spends much of its time foraging in thick brush and along talus slopes. An adult female captured on 1 May 1959 contained 12 eggs (2.4-4.4 mm in diameter; Banta 1963). Mahrdt and Beaman (undated), Marlow (2000), and Banta et al. (1996) summarize additional life history information on the species.

Population Status:

The panamint alligator lizard is extremely rare, as only 24 museum specimens from 16 localities and an additional 11 sight records have accumulated since the panamint alligator lizard was described in 1958 (Banta et al. 1996). Banta et al. (1996) mapped 23 locations, including some based on sight records. In addition, most populations have experienced habitat degradation as a result of human activities (NatureServe 2011). The lizard qualifies for endangered status because its viability is threatened by its small range, rarity, and ongoing threat of habitat degradation. The species was a candidate for federal protection until the FWS eliminated the C2 category, and it currently receives no federal protection under the ESA.

<u>THREATS</u>

Habitat alteration and destruction:

Decline of the panamint alligator lizard is attributed to the direct loss of riparian habitat (Mahrdt and Beaman undated). As explained below, riparian habitat is lost due to expanded mining operations, off-highway vehicle (OHV) activity, grazing (domestic and feral), and introduction of non-native invasive plant species (e.g., tamarisk) (Jennings and Hayes 1994, Mahrdt and Beaman undated).

Construction or improvement of access roads in concert with increased vehicular traffic through or adjacent to "islands" of riparian habitat threatens lizard populations (Mahrdt and Beaman undated). Proposed mining operations that alter or modify springs, seeps, and stream flow are also a direct threat to the hydrology within the species' habitat (NatureServe 2011, Mahrdt and Beaman undated).

The hydrophilic salt cedar (*Tamarix ramosissima*) is a non-native invasive species that has naturalized along the western slopes of the White-Inyo Range (up to 6000 ft/1829 m) (Mahrdt and Beaman undated). Tamarisk has also become widespread in mesic desert habitat and is a serious threat to springs and seeps in the region (DeDecker 1991). Because the lizard requires moist riparian habitats, introduction of tamarisk degrades its habitats and is a threat.

Off-highway vehicle activity has increased significantly and many OHV enthusiasts wench up steep canyon walls and washes to explore the Panamint Mountains (Mahrdt and Beaman undated). These vehicles can damage the lizard's riparian habitat, harming its prey base and causing direct morality.

Overutilization:

California regulations prohibit collection of the lizard but illegal collection using pitfall traps and coverboards also likely threatens populations (Mahrdt and Beaman undated).

Inadequacy of existing regulatory mechanisms:

According to Jennings and Hayes (1994), all but a few of the known populations occur on private lands and are currently at risk from mining, feral and domestic livestock grazing, and increasing off-road vehicle activity. Federal protection would allow action to protect desert springs and stream courses from alterations, such as spring capping, water diversion, mining, and logging for firewood.

The panamint alligator lizard is listed as a Species of Special Concern by the state of California (Jennings and Hayes 1994). It is a BLM sensitive species in California, and a California species of special concern (Mahrdt and Beaman undated). These statuses reflect its rarity but do not offer legal protection.

The species has been documented on Death Valley National Park (Persons and Nowak 2006). Yet the Park does not include enough habitat to ensure population viability, and even within the Park the lizard faces threats such as introduced species.

Other factors:

Because of its isolated, relictual distribution in mesic habitats, the panamint alligator lizard is likely vulnerable to climate-driven changes in habitats (Persons and Nowak 2006), as well as the harmful effects associated with isolation populations, such as inbreeding depression and increased vulnerability to stochastic events.

References:

Banta, B.H. 1962. A preliminary account of the herpetofauna of the Saline Valley hydrographic basin, Inyo County, California. Wasmann J. Biol. 20: 161-251.

Banta, B.H. 1963. Remarks upon the natural history of *Gerrhonotus panamintinus* Stebbins. Occ. Pap. California Acad. Sci. (36): 1-12.

Banta, B. H., C. R. Mahrdt, and K. R. Beaman. 1996. *Elgaria panamintina*. Catalogue of American Amphibians and Reptiles 629.1-629.4.

DeDecker, M. 1991. Shrubs and flowering plants. pp. 108-241. *In:* C.A. Hall, Jr. (ed.), Natural History of the White-Inyo Range, Eastern California. California Nat. Hist. Guides (55), Univ. California Press, Berkeley, California.

Dixon, J.R. 1975. Geographic distribution: *Gerrhonotus panamintinus*. Herpetol. Rev. 6: 45.

Jennings, M. R., and M. P. Hayes. 1994. Amphibian and reptile species of special concern in California. Final Report submitted to the California Department of Fish and Game, Inland Fisheries Division. Contract No. 8023. 255 pp.

Mahrdt, C.R. and Kent R. Beaman. Undated. Panamint Alligator Lizard, *available at* http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=4&ved=0CDwQFjAD&url=http%3A%2F%2Fwww.blm.gov%2Fpgdata%2Fetc%2Fmedialib%2Fblm%2Fca%2Fpdf%2Fpdfs%2Fcdd_pdfs.Par.20e65744.File.pdf%2Fpanamint1.PDF&ei=QIXeTqS-O-L7sQKk5pXKBg&usg=AFQjCNFJafQYphL_ABfc_LPJc9iXpqynIQ (last visited Dec. 6, 2011).

Marlow, R. 2000. California Wildlife Habitat Relationships System: Panamint Alligator Lizard, *available at* http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=25&ved=0CDgQFjAEOBQ&url=http%3A%2F%2Fnrm.dfg.ca.gov%2FFileHandler.ashx%3FDocumentVersionID%3D17917&ei=_YfeTqDIG7P02wWD4NyIBQ&usg=AFQjCNHvYTmWuaztJt6NfYwuQq34Ta4zTw (last visited Dec. 6, 2011).

Hammerson, G.A. 2007. *Elgaria panamintina*. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.2. <www.iucnredlist.org>. Downloaded on 06 December 2011.

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, *available at* http://www.natureserve.org/explorer. (Accessed: December 6, 2011).

Persons, T.B. and E.M. Nowak. 2006. Inventory of Amphibians and Reptiles at Death Valley National Park, *available at* http://balance64.cefns.nau.edu/Orgs/CPCESU/current/documents/NAU6388133DeathValleyHerpsFINALREPORTMay2006.pdf (last visited Dec. 6, 2011).

Stebbins, R.C. 2003. A field guide to western reptiles and amphibians. Third edition. Houghton Mifflin Company, Boston.

Scientific Name:

Crotaphytus reticulatus

Common Name:

Reticulate Collared Lizard

G Rank:

G3 (N2 in U.S.)

IUCN Red List:

Vulnerable

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

The reticulate collared lizard's viability is threatened by its small range and spotty distribution (Axtell 1989, NatureServe 2011). NatureServe (2011) describes the range as follows:

> This lizard occurs in a relatively small area[] in southern Texas, and Coahuila, Nuevo Leon, and Tamaulipas, Mexico (McGuire 1996). The range extends from Eagle Pass, Texas, on the north to Mission, Texas, on the southeast (Conant and Collins 1991, see Judd 1985). The Balcones escarpment of the Edwards Plateau in Texas is the northern limit of distribution, and the western distribution limit coincides with the eastern slopes of the Sierra Madre Oriental in Mexico (see Judd 1985).

Dixon (2000) indicated the presence of this lizard in 11 counties in Texas.

Habitat:

The reticulate collared lizard depends on open spaces for running and foraging and is threatened by nonnative grasses that form dense mats (Scott 1996). NatureServe (2011) describes the habitat as follows:

> This lizard inhabits thorn-scrub vegetation, usually on well-drained rolling terrain of shallow gravel, caliche, or sandy soils. It often occurs on scattered flat rocks below escarpments or isolated rock outcrops among scattered clumps of prickly-pear and mesquite, but it also commonly ranges into mesquite flats far from the nearest rocky habitat, and it is absent on some rocky outcroppings along the margins of bluffs where other *Crotaphytus* species might be expected to occur. Fence posts or the branches of mesquite trees may be used a basking perches. As in *Gambelia*, when approached, this collared lizard often runs to the base of a shrub and remains motionless. Eggs are laid probably under large rocks or underground. Summarized mainly from McGuire (1996).

Biology:

Little is known about the biology of the reticulate collared lizard (Smith 1946, Husack and Ackland 2003). The lizard is known to be active by day, and Montanucci (1971) anecdotally reported diel and seasonal activity patterns. The lizard eats mostly arthropods; also sometimes small reptiles, mice, and plant matter (McGuire 1996). An account of foraging behavior is described by Husack and Ackland (2003).

Population Status:

The reticulate collared lizard is threatened with extinction because it has a small range with a spotty distribution and low density (NatureServe 2011, Axtell 1989). This is the rarest of the collared lizards (*Crotaphytus*) in the United States (NatureServe 2011). Abundance is expected to decline as a result of ongoing habitat degradation (Hammerson et al. 2007). The species was a candidate for federal protection until the FWS eliminated the C2 category, but it currently receives no federal protection under the ESA.

<u>THREATS</u>

Habitat alteration and destruction:

The greatest threat to the reticulate collared lizard is habitat alteration, particularly land clearing practices, the conversion of native grazing lands to farms and improved pastures, and the planting of exotic mat-forming grasses, including buffelgrass (*Cenchrus ciliaris*), for livestock grazing (Judd 1985, Axtell 1989, NatureServe 2011).

The exotic grasses degrade or destroy the preferred habitat. Buffelgrass is planted for cattle forage in the lower Rio Grande Valley of southern Texas and northeastern Mexico, and in the Sonoran Desert (Nabham 1994, Warren 1994). The grass is very successful at colonizing open ground between mesquite and ironwood (*Olneya tesota*) trees. The exotic grass has escaped from pastures and is common along road corridors (Judd 1985). The lizard depends on open spaces and running for foraging and escape, and buffelgrass clogs their habitat, rendering it unsuitable (Scott 1996).

Other potential threats include residential development and pit mining (NatureServe 2011).

Overutilization:

Even though the lizard is state listed as threatened, some take is permitted (Nanjappa and Conrad 2012), and collection is considered a threat (Dixon 2000).

Inadequacy of existing regulatory mechanisms:

The reticulate collared lizard is listed as threatened in Texas (Texas Parks and Wildlife 2011), but this status does not protect the lizard's habitat, which is its greatest threat. In addition, Texas regulations allow some permitted take of the species, which is considered a threat. Federal protection would prohibit take of this rare species.

References:

Axtell, R.W. 1989. Interpretive atlas of Texas lizards: *Crotaphytus reticulatus*. No. 9. Published by the author. 8 pp.

Conant, R. and J. T. Collins. 1991. A field guide to reptiles and amphibians: eastern and central North America. Third edition. Houghton Mifflin Co., Boston, Massachusetts. 450 pp.

Dixon, J. R. 2000. Amphibians and reptiles of Texas. Texas A&M University Press, College Station.

Hammerson, G.A., P. Lavin, and F. Mendoza Quijano. 2007. *Crotaphytus reticulatus*. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.2. <www.iucnredlist.org>. Downloaded on 07 December 2011.

Husak, J.F. and E.N. Ackland. 2003. Foraging mode of the reticulate collared lizard, *Crotaphytus reticulatus*. The Southwestern Naturalist 48(2): 282.

Judd, F. W. 1985. Status of *Siren intermedia texana*, *Notophthalmus meridionalis*, and *Crotaphytus reticulatus*, final report. Submitted to Office of Endangered Species, United States Fish and Wildlife Service. Coop. Agreement No. 14-16-0002-81-923. Modification No. 1. 60 pp.

Klein, T., Jr. 1951. Notes on the feeding habits of *Crotaphytus reticulatus*. Herpetologica 7: 200.

McGuire, J. A. 1996. Phylogenetic systematics of crotaphytid lizards (Reptilia: Iguania: Crotaphytidae). Bulletin of Carnegie Museum of Natural History (32): 1-143.

Montanucci, R. R. 1971. Ecological and distributional data on *Crotaphytus reticulatus* (Sauria: Iguanidae). Herpetologica 27: 183–197.

Nabham, G.P. 1994. Proximate and ultimate threats to endangered species. Conservation Biology 8: 928-929.

Nanjappa, P. and Paulette M. Conrad 2012. State of the Union: Legal Authority Over the Use of Native Amphibians and Reptiles in the United States, *available at* http://www.fishwildlife.org/files/SOU_FULL-lo-res.pdf (last visited April 30, 2012).

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, *available at* http://www.natureserve.org/explorer (Accessed: December 7, 2011).

Scott, N. J., Jr. 1996. Evolution and management of the North American grassland herpetofauna. Pages 40-53 in D. M. Finch editor. Ecosystem disturbance and wildlife conservation in western grasslands: a symposium proceedings. General Technical Report RM-GTR-285. U.S. Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado, *available at* http://www.fs.fed.us/rm/pubs_rm/rm_gtr285/rm_gtr285_040_053.pdf (last visited Dec. 7, 2011).

Smith, H.M. 1946. Handbook of lizards. Lizards of the United States and Canada. Comstock Publishing Associates, Ithaca, New York.

Texas Parks and Wildlife. 2011. Endangered and Threatened Reptiles and Amphibians in Texas and the United States, *available at* http://www.tpwd.state.tx.us/huntwild/wild/species/endang/animals/reptiles_amphibians/ (last visited Dec. 7, 2011).

Warren, P. 1994. Drawing management conclusions in conservation biology. Conservation Biology 8: 928.

Scientific Name:

Xantusia gracilis

Common Name:

Sandstone Night Lizard

G Rank:

G1

IUCN Red List:

Vulnerable

<u>NATURAL HISTORY, BIOLOGY, AND STATUS</u>

Range:

The sandstone night lizard is vulnerable to extinction because of its extremely restricted range where it is facing significant threats. NatureServe (2011) explains that the known range is confined to the Truckhaven Rocks area (an area of about 3 km by 1.3 km), Anza-Borrego Desert State Park, southeastern flank of the Santa Rosa Mountains, San Diego County, California; elevation 240-305 meters (787-1,000 feet) (Grismer and Galvan 1986, Lovich 2001, Lovich and Grismer 2003).

Habitat:

The lizard's habitat consists of sandstone crevices and burrows primarily in and along the canyons that cut through the Truckhaven Rocks (Lovich and Grismer 2003). Some of the lizards "were unearthed in what appeared to be rodent burrows atop piles of hardened siltstone" (Grismer and Galvan 1986).

Biology and Taxonomy:

The sandstone night lizard was recognized as a species (rather than subspecies) by Lovich (2001) and Vicario et al. (2003). This conclusion is supported by previously published allozyme data (showing fixed allelic differences) and a compatible historical biogeographical scenario (NatureServe 2011). ITIS (2011) and Collins and Taggart (2009) recognize the species status as valid.

Little is known about the life history of the sandstone night lizard. Its diet includes invertebrates and reptile eggs (Grismer and Galvan 1986). It is likely exceptionally sedentary (NatureServe 2011).

Population Status:

The sandstone night lizard qualifies for Endangered Species Act protection because of its extremely small range and ongoing threats from isolation and road mortality. It is known only from a small area a few kilometers across in a state park in southern California (NatureServe 2011). The species was a candidate for federal protection until the FWS eliminated the C2 category, and it currently receives no federal protection under the ESA.

THREATS

Habitat alteration and destruction:

The habitat is offered protection from development because it is entirely within the state park. But habitat fragmentation by roads occurs within the park and road mortality is a threat (California Roadkill Observation System 2011).

Inadequacy of existing regulatory mechanisms:

The species has special concern status but is not listed as threatened or endangered (California Dept. of Fish and Game 2011b). The habitat is offered protection within the state park and the lizard is protected from collection by California law. But without federal or state protected status, the species does not benefit from the attention given to listed species, and it is without a recovery plan and unmonitored. While it is illegal to collect the sandstone night lizard (California Dept. of Fish and Game 2011a), some collection is likely to still occur. Federal protection would increase penalties and enforcement and deter illegal collection.

Other factors:

Geographic isolation and an extremely limited distribution places the species at a high risk of extinction (California Dept. of Fish and Game 2005) and makes it inherently vulnerable to even minor disturbances (ARSSC 2009). To be sure, small, isolated populations are more susceptible to extirpations due to stochastic events, human impacts, and environmental factors (Soulé 1987; Begone et al. 1990). Lack of gene flow may cause loss of genetic variability due to random genetic drift (Wright 1931) and inbreeding depression is likely to occur (Franklin 1980). Once these populations are extirpated, their isolation precludes genetic interchange and recolonization of habitat.

References:

ARSSC. 2009. California Amphibian and Reptile Species of Special Concern, *available at* http://arssc.ucdavis.edu/species/species/lizards.html (last visited Dec. 7, 2011).

Begone, M., J. L. Harper, and C. R. Townsend. 1990. Ecology: Individuals, populations, and communities. Blackwell Scientific Publications.

California Department of Fish and Game. 2005. California Wildlife Habitat Relationships System: Sandstone Night Lizard, *available at* http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=23&ved=0CCkQFjACOBQ&url=http%3A%2F%2Fnrm.dfg.ca.gov%2FFileHandler.ashx%3FDocumentVersionID%3D18000&ei=IJTfTrKaO8Lu2gWssIj0BA&usg=AFQjCNGSZQFfctMvBcl6IRZxlQ4CpajH5A (last visited Dec. 7, 2011).

California Dept. of Fish and Game. 2011a. California State Rules and Regulations Regarding Hunting and Collecting California Reptiles and Amphibians for 2011, *available at* http://www.californiaherps.com/info/herpinglaws.html (last visited Dec. 7, 2011).

California Dept. of Fish and Game. 2011b. Special Animals, *available at* http://www.dfg.ca.gov/wildlife/nongame/list.html (last visited Dec. 7, 2011).

California Roadkill Observation System. 2011. List of California Animals, *available at* http://www.wildlifecrossing.net/california/taxa?tid=7 (last visited Dec. 7, 2011).

Collins, J. T., and T. W. Taggart. 2009. Standard common and current scientific names for North American amphibians, turtles, reptiles, and crocodilians. Sixth edition. The Center for North American Herpetology, Lawrance, Kansas. iv + 44 pp.

Franklin, I. R. 1980. Evolutionary changes in small populations. In Conservation Biology: An evolutionary-ecological perspective (ed. a. B. A. W. M. E. Soule). pp. 135-149.

Grismer, L. L. 2002. Amphibians and reptiles of Baja California including its Pacific islands and islands in the Sea of Cortes. University of California Press, Berkeley. xiii + 399 pp.

Grismer, L. L., and M. A. Galvan. 1986. A new night lizard (*Xantusia henshawi*) from a sandstone habitat in San Diego County, California. Trans. San Diego Soc. Nat. Hist. 21(10): 155-165.

Hammerson, G.A. 2007. Xantusia gracilis. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.2. <www.iucnredlist.org>. Downloaded on 07 December 2011.

ITIS. 2011. *Xantusia gracilis* Grismer and Galvan, 1986, *available at* http://www.itis.gov/servlet/SingleRpt/SingleRpt?search_topic=TSN&search_value=683049 (last visited Dec. 7, 2011).

Lovich, R. 2001. Phylogeography of the night lizard, *Xantusia henshawi*, in southern California: evolution across fault zones. Herpetologica 57: 470-487.

Lovich, R. E., and L. L. Grismer. 2003. *Xantusia gracilis*. Catalogue of American Amphibians and Reptiles 772:1-2.

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, *available at* http://www.natureserve.org/explorer (Accessed: December 7, 2011).

Soulé, M. 1987. Viable populations for conservation. Cambridge University Press, Great Britain. 189 pp.

Vicario, S., Adalgisa Caccone and Jacques Gauthier 2003. Xantusiid "night" lizards: a puzzling phylogenetic problem revisited using likelihood-based Bayesian methods on mtDNA sequences. Molecular Phylogenetics and Evolution 26 (2): 243-261.

Wright, S. 1931. Evolution in Mendelian populations. Genetics 16: 97-159.


© 2011 Timothy Burkhardt

Scientific Name:

Xantusia arizonae (or in the alternative, *Xantusia vigilis arizonae*)

Common Name:

Arizona Night Lizard

G Rank:

G1

IUCN Red List:

Least concern

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

The Arizona night lizard has an extremely limited range and is thus highly vulnerable to extinction from any threat within this small range. As redefined by Sinclair et al. (2004), the Arizona night lizard is known only from Yavapai County, Arizona, where it has been found from the Weaver Mountains to the McCloud Mountains and associated ranges (Bezy 2005). It extends from the Saguaro-Paloverde association of the Arizona Upland Sonoran Desertscrub at 760 m (2500 ft) elevation to Interior Chaparral at 1524 m (5000 ft) (Brown 1994).

Habitat:

The Arizona night lizard occurs primarily in rock-crevice habitat (Pappenfuss et al. 2001), but it also has been found in *Neotoma* (pack rat) nests and in decaying *Yucca baccata* (Bezy 2005).

Biology and Taxonomy:

Papenfuss et al. (2001) examined genetic and morphological variation of *Xantusia* and reviewed allozyme data. They concluded that three species are represented in Arizona: *Xantusia vigilis*, a yucca-dwelling species; *X. arizonae*, a granite-adapted species; and *X. bezyi* (newly described), another granite-associated species. Sinclair et al. (2004) examined phylogeographic patterns in *Xantusia* and concluded that *X. bezyi* is a valid species (but more widely distributed than previously known) and that *X. arizonae* has a smaller range than previously understood. The results of Leavitt et al. (2007) confirm the results of Sinclair et al. (2004) and also call for the recognition of *X. arizonae* as a separate species, not subspecies. Crother et al. (2008) (in a change from the previous edition) elevated the Arizona night lizard to species status. ITIS does not recognize *Xantusia arizonae* as a valid species but considers the Arizona night lizard as a subspecies, *Xantusia vigilis arizonae*. We hereby petition for the Arizona night lizard as either a full species or in the alternative, a subspecies.

Little is known about the life history of the Arizona night lizard but it feeds on invertebrates (NatureServe 2011).

Population Status:

Federal protection for the Arizona night lizard is needed because it is very rare and is found only in a small set of localities that are subject to the threat of overcollection. Sinclair et al. (2004) mapped six collection sites, and Leavitt et al. (2007) listed 10 collection localities.

THREATS

Habitat alteration and destruction:

The rocky habitat of the Arizona night lizard is not readily convertible to destructive human uses, but habitat degradation by collectors likely is significant in some localities (NatureServe 2011).

Overutilization:

Removal of lizards by collectors is likely significant in some localities (NatureServe 2011).

Inadequacy of existing regulatory mechanisms:

The Arizona night lizard lacks protected status in Arizona (Arizona Game and Fish Dept., undated), but it is a U.S. Forest Service Region 3 Sensitive Species. Regulations of Arizona Game and Fish Department prohibit the use of manual or powered jacking or prying devices to take reptiles or amphibians (Bezy 2005, see AGFD R12-4-303-C). Nevertheless, because the species lacks state or federal protection and collection remains legal, habitat degradation by collectors is likely significant in some localities (NatureServe 2011). Federal protection of the lizard would result in steep fines for collection and deter these destructive activities.

Other factors:

The Arizona night lizard is known only from Yavapai County, Arizona (Bezy 2005). Small, isolated populations like that of the lizard are more susceptible to extirpations due to stochastic events, human impacts, and environmental factors (Soulé 1987, Begone et al. 1990). Lack of gene flow may cause loss of genetic variability due to random genetic drift (Wright 1931) and inbreeding depression is likely to occur (Franklin 1980). Once these populations are extirpated, their isolation precludes genetic interchange and recolonization of habitat.

References:

Arizona Game and Fish Department. Undated. Arizona Game and Fish Department Heritage Data Management System: Arizona Night Lizard, *available at* http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=13&ved=0CCkQFjACOAo&url=http%3A%2F%2Fwww.azgfd.gov%2Fw_c%2Fedits%2Fdocuments%2FXantariz.d_000.pdf&ei=KAHdToDiJrC_2QWOu-zIDg&usg=AFQjCNG1IW2yFdokuoxJPyERAPM-tozBMQ (last visited Dec. 5, 2011).

Begone, M., J. L. Harper, and C. R. Townsend. 1990. Ecology: Individuals, populations, and communities. Blackwell Scientific Publications.

Bezy, R.L. 2005. The Night Lizards (*Xantusia*) of Arizona. Sonoran Herpetologist. 18(2): 14-19.

Crother, B. I. (ed.). 2008. Scientific and Standard English Names of Amphibians and Reptiles of North America North of Mexico, pp. 1-84.

Franklin, I. R. 1980. Evolutionary changes in small populations. In Conservation Biology: An evolutionary-ecological perspective (ed. a. B. A. W. M. E. Soule). pp. 135-149.

Hammerson, G.A. 2007. *Xantusia arizonae*. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.2. <www.iucnredlist.org>. Downloaded on 05 December 2011.

ITIS. 2011. *Xantusia vigilis arizonae* Klauber, 1931, *available at* http://www.itis.gov/servlet/SingleRpt/SingleRpt?search_topic=TSN&search_value=208991 (last visited Dec. 5, 2011).

Klauber, Laurence M. 1931. A new species of *Xantusia* from Arizona, with a synopsis of the genus. Transactions of the San Diego Society of Natural History 7(1): 1-16.

Leavitt, D. H., R. L. Bezy, K. A. Crandall, and J. W. Sites. Jr. 2007. Multi-locus DNA sequence data reveal a history of deep cryptic vicariance and habitat-driven convergence in the desert night lizard Xantusia vigilis species complex (Squamata: Xantusiidae). Molecular Ecology 16: 4455-4481.

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, *available at* http://www.natureserve.org/explorer. (Accessed: December 5, 2011).

Papenfuss, T. J., J. R. Macey, and J. A. Schulte, II. 2001. A new lizard species in the genus Xantusia from Arizona. Natural History Museum, The University of Kansas, Scientific Papers (23):1-9.

Sinclair, E. A., R. L. Bezy, K. Bolles, J. L. Camarillo R., K. A. Crandall, and J. W. Sites, Jr. 2004. Testing species boundaries in an ancient species complex with deep phylogeographic history: genus Xantusia (Squamata: Xantusiidae). American Naturalist 164: 396-414.

Soulé, M. 1987. Viable populations for conservation. Cambridge University Press, Great Britain. 189 pp.

Wright, S. 1931. Evolution in Mendelian populations. Genetics 16: 97-159.

Scientific Name:

Xantusia bezyi

Common Name:

Bezy's Night Lizard

G Rank:

G2

IUCN Red List:

Least concern

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

The Bezy's night lizard has a narrow range that leaves the species vulnerable to extinction. The lizard is confined to central Arizona (Sinclair et al. 2004, Leavitt et al. 2007). Bezy (2005) explains that the lizard ranges from the Mazatzal Mountains (Maricopa County) south to the Galiuros, and that it is found from the Saguaro-Paloverde Association of the Arizona Upland Sonoran Desertscrub at 730 m (2400 ft) elevation, through Semi-desert Grassland to Interior Chaparral at 1770 m (5800 ft).

Habitat:

This species is found under exfoliating rock in granite outcrops (Papenfuss et al. 2001).

Taxonomy:

Papenfuss et al. (2001) examined genetic and morphological variation of *Xantusia* and reviewed allozyme data. They concluded that three species are represented in Arizona: *Xantusia vigilis*, a yucca-dwelling species; *X. arizonae*, a granite-adapted species; and *X. bezyi* (newly described), another granite-associated species. Sinclair et al. (2004) examined phylogeographic patterns in *Xantusia* and concluded that *X. bezyi* is a valid species (but more widely distributed than previously known) and that *X. arizonae* has a smaller range than previously understood). Leavitt et al. (2007) confirmed the distinctiveness and distributions of *X. arizonae* and *X. bezyi*. Crother et al. (2008) and ITIS also recognize *X. bezyi* as a valid species.

Bezy (2005) reports that nothing has been published about the biology of the species.

Population Status:

The Bezy's night lizard qualifies for endangered status because it is very rare and has been found in just a small set of localities where it is threatened by overcollection. Sinclair et al. (2004) mapped eight localities that they assigned to this species. Leavitt et al. (2007) listed nine collection localities.

<u>THREATS</u>

Habitat alteration and destruction:

The rocky habitat of the Arizona night lizard is not readily convertible to destructive human uses, but habitat degradation by collectors using devices to pry on its rocky habitats is likely significant in some localities (NatureServe 2011).

Overutilization:

Removal of lizards by collectors is likely significant in some localities (NatureServe 2011).

Inadequacy of existing regulatory mechanisms:

The Bezy's night lizard lacks protected status in Arizona (Arizona Game and Fish Dept, undated). Because the species lacks state or federal protection and collection remains legal, habitat degradation by collectors is likely significant in some localities (NatureServe 2011), despite the fact that regulations of Arizona Game and Fish Department prohibit the use of manual or powered jacking or prying devices to take reptiles or amphibians (Bezy 2005; *see* AGFD R12-4-303-C). Federal protection of the lizard would result in steep fines for collection and deter these destructive activities.

A small portion of the lizard's range is protected as Tonto National Forest but it is not recognized as a Forest Service sensitive species (Bezy 2005). As such, impacts of forest activities on the lizard are not analyzed, nor are steps taken to actively monitor or manage the species.

Other factors:

The Bezy's night lizard is confined to central Arizona. Small, isolated populations, such as that of the lizard, are more susceptible to extirpations due to stochastic events, human impacts, and environmental factors (Soulé 1987, Begone et al. 1990). Lack of gene flow may cause loss of genetic variability due to random genetic drift (Wright 1931) and inbreeding depression is likely occur (Franklin 1980). Once these populations are extirpated, their isolation precludes genetic interchange and recolonization of habitat.

References:

Begone, M., J. L. Harper, and C. R. Townsend. 1990. Ecology: Individuals, populations, and communities. Blackwell Scientific Publications.

Bezy, R.L. 2005. The Night Lizards (*Xantusia*) of Arizona. Sonoran Herpetologist. 18(2): 14-19.

Crother, B. I. (ed.). 2008. Scientific and Standard English Names of Amphibians and Reptiles of North America North of Mexico, pp. 1-84.

Franklin, I. R. 1980. Evolutionary changes in small populations. In Conservation Biology: An evolutionary-ecological perspective (ed. a. B. A. W. M. E. Soule). pp. 135-149.

Hammerson, G.A. 2007. *Xantusia bezyi*. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.2. <www.iucnredlist.org>. Downloaded on 05 December 2011.

ITIS. 2011. *Xantusia bezyi* Papenfuss, Macey and Schulte, 2001, *available at* http://www.itis.gov/servlet/SingleRpt/SingleRpt (last visited Dec. 5, 2011).

Leavitt, D. H., R. L. Bezy, K. A. Crandall, and J. W. Sites. Jr. 2007. Multi-locus DNA sequence data reveal a history of deep cryptic vicariance and habitat-driven convergence in the desert night lizard Xantusia vigilis species complex (Squamata: Xantusiidae). Molecular Ecology 16: 4455-4481.

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, *available at* http://www.natureserve.org/explorer. (Accessed: December 5, 2011).

Papenfuss, T. J., J. R. Macey, and J. A. Schulte, II. 2001. A new lizard species in the genus Xantusia from Arizona. Natural History Museum, The University of Kansas, Scientific Papers (23):1-9.

Sinclair, E. A., R. L. Bezy, K. Bolles, J. L. Camarillo R., K. A. Crandall, and J. W. Sites, Jr. 2004. Testing species boundaries in an ancient species complex with deep phylogeographic history: genus Xantusia (Squamata: Xantusiidae). American Naturalist 164: 396-414.

Soulé, M. 1987. Viable populations for conservation. Cambridge University Press, Great Britain. 189 pp.

Wright, S. 1931. Evolution in Mendelian populations. Genetics 16: 97-159.

Scientific Name:

Uma rufopunctata

Common Name:

Yuman Desert Fringe-toed Lizard or Sonoran Fringe-toed Lizard

G Rank:

G3

IUCN Red List:

Near Threatened

<u>NATURAL HISTORY, BIOLOGY, AND STATUS</u>

Range:

According to NatureServe (2011), the small range of the Yuman Desert fringe-toed lizard includes scattered areas of suitable habitat in southwestern Arizona (south of the Gila River; mainly in the Mohawk and Yuma dune systems, Yuma County, and the Pinta Sands, Pima County) and northwestern Sonora south to Tepoca Bay (Pough 1977, Trepanier and Murphy 2001, Stebbins 2003). In Arizona, the known elevational range is around 49-275 meters (160-900 feet). The lizard is threatened with extinction because even within this small range it is confined to scattered areas of suitable habitat that face ongoing threats of degradation and development.

Habitat:

This species is restricted to sparsely-vegetated windblown sand dunes and sandy flats (NatureServe 2011). It requires fine, loose sand for burrowing, and vegetation is usually scant, consisting of creosote bush or other scrubby growth (Stebbins 2003).

Biology and Taxonomy:

Trepanier and Murphy (2001) concluded that for the northern species of *Uma* existing data support either a two-species arrangement (*Uma scoparia*, *Uma notata*) or five-species classification (*Uma scoparia*, *Uma notata*, *Uma inornata*, and *Uma rufopunctata*, plus an undescribed species from Mohawk Dunes, Arizona). Trepanier and Murphy prefered the latter arrangement. Crother et al. (2003), Collins and Taggart (2009), and ITIS (2011) adopted the taxonomy preferred by Trepanier and Murphy and accept *Uma rufopunctata* as a separate and valid species (for a summary of taxonomic studies see NatureServe 2011).

Very little is known about the life history of the Yuman Desert fringe-toed lizard (NatureServe 2011). It eats chiefly insects, but occasionally other lizards and buds, leaves, and flowers (Stebbins 1985). It is inactive in cold temperatures or extreme heat (NatureServe 2011).

Population Status:

Recent data is unavailable but the Yuman Desert fringe-toed lizard was known from just four sites in Arizona and ten in Sonora (Pough 1977, S. Schwartz, pers. comm., 1997, cited in NatureServe 2011), and some populations are now likely extirpated due to observed habitat destruction. It is vulnerable to extinction because of its rarity across its small range and ongoing threats to its habitat. The IUCN Red List considers the species as Near Threatened but explains that it is close to qualifying as Vulnerable.

THREATS

Habitat alteration and destruction:

The extent and quality of the lizard's habitat is declining (Hammerson et al. 2007). This lizard is specialized for fragile aeolian sand habitats that are frequently damaged by off-road vehicle use (NatureServe 2011). Other threats to habitat include commercial, residential, and agricultural development (e.g., in the Mohawk Valley of Arizona) (Arizona Game and Fish Department undated).

Overutilization:

Collection of this beautiful lizard is allowed under state law but is not documented as a threat.

Disease or predation:

Blood-feeding midges (*Leptoconops californiensis*) may attack the lizards (Turner and Olson 2005) but they are unlikely to pose a threat.

Inadequacy of existing regulatory mechanisms:

In Arizona, the species occurs on lands administered by BLM (Yuma Field Office), Bureau of Reclamation (Yuma Area), Department of Defense (Barry M. Goldwater Air Force Range), FWS (Cabeza Prieta National Wildlife Refuge), and the Arizona State Land Department (Hammerson et al. 2007). These lands are protected from development but habitat degradation from off-road vehicle use still threatens lizards on these lands. Land within the Barry M. Goldwater Air Force Range is subject to monitoring to help protect the desert ecosystem (Villarreal et al. 2011), but any efforts are purely voluntary and therefore inadequate. Without a federal listing, habitat for lizards that occurs on private lands is unprotected from development (Hammerson et al. 2007).

The lizard is a BLM sensitive species in Arizona (Arizona Dept. of Fish and Game undated), and a state sensitive species (Arizona Dept. of Fish and Game 2010). These designations reflect the rarity of this lizard but offer no legal protection for the lizards or their habitat. The lizard is threatened in Mexico (Rorabaugh 2008). Personal collection of the lizard is allowed with a hunting license (Arizona Game and Fish Department 2011). Given the rarity of this lizard, these state regulations are inadequate; with federal protection all take of this rare lizard would be protected.

Other factors:

A potential threat is posed by non-native annual mustard (*Brassica*), which recently invaded southwestern Arizona. The mustard forms thick carpets and degrades habitat (Arizona Game and Fish Department, as cited in NatureServe 2011).

References:

Adest, G. A. 1977. Genetic relationships in the genus UMA (Iguanidae). Copeia 1977: 47-52.

Arizona Game and Fish Department. 2011. Amphibian and Reptile Regulations 2011 & 2012, *available at* http://www.azgfd.gov/pdfs/h_f/regulations/ReptileAmphibian.pdf (last visited May 30, 2012).

Arizona Game and Fish Dept. Heritage Data Management System. 2010. Arizona Bureau of Land Management Sensitive Species List – December 2010, *available at* http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=87&ved=0CFEQFjAGOFA&url=http%3A%2F%2Fwww.blm.gov%2Fpgdata%2Fetc%2Fmedialib%2Fblm%2Faz%2Fpdfs.Par.64804.File.dat%2Fsensitive-species.pdf&ei=5cTfTsm4K6Sl2AXko5yIBQ&usg=AFQjCNHQU9EWz_UBTFfz71UPIeEdTxWhXw (last visited Dec. 7, 2011).

Arizona Game and Fish Department Heritage Data Management System. Undated. Uma rufopunctata, *available at* http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=11&ved=0CCYQFjAAOAo&url=http%3A%2F%2Fwww.azgfd.gov%2Fw_c%2Fedits%2Fdocuments%2FUmarufo.d_004.pdf&ei=BL_fTuGTAofq2AWSyoH6BA&usg=AFQjCNG619Rb6VwE9hxGJKKZ5Uf7A-XovA (last visited Dec. 7, 2011).

Collins, J. T., and T. W. Taggart. 2009. Standard common and current scientific names for North American amphibians, turtles, reptiles, and crocodilians. Sixth edition. The Center for North American Herpetology, Lawrance, Kansas. iv + 44 pp.

Crother, B. I., J. Boundy, J. A. Campbell, K. de Quieroz, D. Frost, D. M. Green, R. Highton, J. B. Iverson, R. W. McDiarmid, P. A. Meylan, T. W. Reeder, M. E. Seidel, J. W. Sites, Jr., S. G. Tilley, and D. B. Wake. 2003. Scientific and standard English names of amphibians and reptiles of North America north of Mexico: update. Herpetological Review 34:198-203.

Hammerson, G.A., Frost, D.R. & Gadsden, H. 2007. Uma rufopunctata. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.2. <www.iucnredlist.org>. Downloaded on 07 December 2011.

ITIS. 2011. Uma rufopunctata Cope, 1895, *available at* http://www.itis.gov/servlet/SingleRpt/SingleRpt (last visited Dec. 7, 2011).

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, *available at* http://www.natureserve.org/explorer. (Accessed: December 7, 2011).

Pough, F. H. 1977. *Uma notata*. Catalogue of American Amphibians and Reptiles 197: 1-2.

Rorabaugh, R.C. 2008. An Introduction to the Herpetofauna of Mainland Sonora, México, with Comments on Conservation and Management. Journal of the Arizona-Nevada Academy of Science 40(1): 20-65.

Stebbins, R. C. 1985. A field guide to western reptiles and amphibians. Second edition. Houghton Mifflin Company, Boston, Massachusetts. xiv + 336 pp.

Stebbins, R. C. 2003. A field guide to western reptiles and amphibians. Third edition. Houghton Mifflin Company, Boston.

Trepanier, T. L., and R. W. Murphy. 2001. The Coachella Valley fringe-toed lizard (*Uma inornata*): genetic diversity and phylogenetic relationships of an endangered species. Molecular Phylogenetics and Evolution 18: 327-334.

Turner, D. and C. Olson. 2005. Blood-feeding Midges in Arizona: New Lizard Hosts and Range Extension for Leptoconops californiensis. Sonoran Herpetologist 18 (5): 54, *available at* http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=1&ved=0CD8QFjAA&u

rl=http%3A%2F%2Ftucsonherpsociety.org%2FTurner%2520and%2520Olson%25202005_18(5)54.pdf&ei=gIfGT8r_MI2E8ASgu9C2Bg&usg=AFQjCNG1jCKlxJgGH6Rgqm_cjHe866C-ng&sig2=Vl4zL7k0KYH09OQnmZnBgA.

Villarreal, M.L.,Charles van Riper III, Robert E. Lovich, Robert L. Palmer, Travis Nauman Sarah E. Studd, Sam Drake, Abigail S. Rosenberg, Jim Malusa, and Ronald L. Pearce. 2011. An Inventory and Monitoring Plan for a Sonoran Desert Ecosystem: Barry M. Goldwater Range–West, *available at* http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=71&ved=0CCYQFjAAOEY&url=http%3A%2F%2Fpubs.usgs.gov%2Fof%2F2011%2F1232%2Fof2011-1232.pdf&ei=pcPfTqIRiqraBbaL6asF&usg=AFQjCNGFrpTGf_qPC-6JHrB2BABhbo9xag (last visited Dec. 7, 2011).

Wilgenbusch, J., and K. de Queiroz. 2000. Phylogenetic relationships among the phrynosomatid sand lizards inferred from mitochondrial DNA sequences generated by heterogeneous evolutionary processes. Systematic Biology 49: 592-612.

SALAMANDERS

Inyo Mountains Salamander (*Batrachoseps campi*)
Lesser Slender Salamander (*Batrachoseps minor*)
Kings River Slender Salamander (*Batrachoseps regius*)
Relictual Slender Salamander (*Batrachoseps relictus*)
Kern Plateau Salamander (*Batrachoseps robustus*)
Kern Canyon Slender Salamander (*Batrachoseps simatus*)
Oregon Slender Salamander (*Batrachoseps wrighti*)
California Giant Salamander (*Dicamptodon ensatus*)
Cascade Caverns Salamander (*Eurycea latitans*)
Limestone Salamander (*Hydromantes brunus*)
Shasta Salamander (*Hydromantes shastae*)
Blue Ridge Gray-cheeked Salamander (*Plethodon amplus*)
Caddo Mountain Salamander (*Plethodon caddoensis*)
Cheoah Bald Salamander (*Plethodon cheoah*)
Fourche Mountain Salamander (*Plethodon fourchensis*)
Peaks of Otter Salamander (*Plethodon hubrichti*)
South Mountain Gray-cheeked Salamander (*Plethodon meridianus*)
Pigeon Mountain Salamander (*Plethodon petraeus*)
White-spotted Salamander (*Plethodon punctatus*)
Weller's Salamander (*Plethodon welleri*)
Green Salamander (*Aneides aeneus*)
Cascade Torrent Salamander (*Rhyacotriton cascadae*)
Columbia Torrent Salamander (*Rhyacotriton kezeri*)
Olympic Torrent Salamander (*Rhyacotriton olympicus*)

Scientific Name:

Batrachoseps campi

Common Name:

Inyo Mountains Salamander

G Rank:

G2

IUCN Red List:

Endangered

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

The Inyo Mountains salamander is vulnerable to extinction because of its extremely restricted range in the Inyo Mountains, Inyo County, California, where the salamanders face the ongoing threat of habitat destruction. Specifically, occupied habitat for Inyo Mountains salamanders totals less than 20 ha (Giuliani 1977, Papenfuss and Macey 1986), and it is known from less than 15 localities (AmphibiaWeb 2012; see also Jennings and Hayes 1994, Jockusch 2001). Its elevational range is 1,800-8,036 ft (550-2450 m) (Jockusch 2001).

Habitat:

The salamander is found along small permanent desert springs and seeps with riparian vegetation; generally under stones, wood, or in holes or crevices in moist soil near spring seepages and pools (NatureServe 2011). Vegetation along water courses consists of willows and wild rose (NatureServe 2011). Surrounding slopes are arid, grown to sagebrush, buckwheat, rabbitbrush, and cactus (Stebbins 1985). All inhabited sites are highly localized and isolated with small areas of suitable habitat bordered by large expanses of inhospitable desert or semi-desert terrain (Hanson and Wake 2005). Jenning and Hayes (1994) explains: "Much of its known habitat is associated with springs that can attract significant human (*Homo sapiens*), horse (*Equus caballus*), and burro (*E. asinus*) activity that is likely to imperil its survival."

Biology:

The Inyo Mountains salamander appears to be nocturnal, taking shelter under moist rocks or in damp crevices during the daytime (Macey and Papenfuss 1991a). The species likely has direct development similar to other members of the genus *Batrachoseps* where the reproductive pattern is known (Jenning and Hayes 1994). Nesting sites are likely to be moist subterranean localities within the talus slopes or fissures of the habitat where this species has been found (Jenning and Hayes 1994). No data are available on the movement ecology or physiology of this species or on the potential differential use of habitat by various life stages (Jenning and Hayes 1994). While little is known about the species, available information on the biology is summarized by Hansen and Wake (2005).

Population Status:

The Inyo Mountains salamander is endangered because of its extremely restricted range with few known populations that are vulnerable from threats like mining and water diversions. It is known from fewer than 15 localities in the Inyo Mountains, California (NatureServe 2011). In addition, there has been a continuing decline in the extent and quality of its habitat, and the number of mature individuals (Hammerson 2004). The species was a candidate for federal protection until the FWS eliminated the C2 category, but it currently receives no federal protection under the ESA.

THREATS

Habitat alteration and destruction:

The Inyo Mountains salamander is extremely vulnerable to habitat destruction. AmphibiaWeb (2012) explains: "Because of its restricted habitat, isolation, and probably small population size, any habitat alteration should be considered a threat. The stream-side habitat of the species is severely limited in extent and the climate is exceptionally harsh (intensely hot and dry) for a terrestrial salamanders. Any habitat destruction in its limited range could threaten local populations."

Papenfuss and Macey (1986) carefully detailed specific threats to each of the then-known populations of Inyo Mountains salamanders. Among these were mining activities, damage from livestock and feral burros, and water diversions. Similarly, Jennings and Hayes (1994) explain: "A combination of water diversion from springs, disturbance of the substrate through mining, and damage to potentially sheltering riparian plants by feral burros and domestic cattle (*Bos taurus*) currently pose some degree of threat to every one of the 16 localities where this species is known to occur."

Mining activities destroy and degrade habitat for the salamanders. More than 360 mining claims are located near Inyo Mountains salamander populations (Papenfuss and Macey 1986). Even when mining lands are reclaimed, habitat is unlikely to be restored to suitable conditions and recolonization is unlikely to occur given that remaining localities for the salamander are highly isolated (see Hanson and Wake 2005).

Water diversion is another documented threat. Because the salamanders require moist soil near spring seepages and pools, diversion of water makes the habitat unsuitable and causes extirpations. Water diversion is an especially severe threat for Barrel Spring (1,950 m), where the original 0.8-km- (0.5-mi-) long corridor of riparian habitat has been reduced 90 percent by water diversion and road construction (Giuliani 1990).

Grazing pressure from livestock and feral burros is another threat. These animals trample the riparian habitats required by the salamander. Degraded habitat from burros has been documented at some sites (Giuliani 1977, Papenfuss and Macey 1986).

Inadequacy of existing regulatory mechanisms:

Nearly all populations of the salamander occur on federal lands managed by the U.S. Bureau of Land Management, U.S. Forest Service, or the National Park Service (Death Valley National Park, NPS 2012), but the level of protection is questionable (Hammerson 2004). Specifically, it is listed as a Forest Service and BLM Sensitive Species (California Dept. of Fish and Game 2011), but sensitive species designations do not prevent harmful activities in salamander habitat but only require analysis of impacts. As a result, activities harmful to the salamander, such as mining and grazing is not prohibited.

State regulations prohibit collection of slender salamanders (*Batrachoseps* spp.) from Inyo County (CCR § 5.05, Nanjappa and Conrad 2012). In addition, this species is registered as a Species of Special Concern by the California Department of Fish and Game (California Dept. of Fish and Game 2011). These regulations and status are insufficient, however, because they do not prevent destruction of the salamander’s habitats.

Because of the species’ highly restricted range, there is a need for immediate effective protection of remaining habitat from any alteration (Hammerson 2004). Protection of all populations is necessary to maintain diversity, since each population is genetically isolated and unique (Hammerson 2004).

Concerted efforts have been made by federal agencies to reduce the number of feral burros in the Inyo Mountains region but they remain a threat (Hansen and Wake 2005). As explained by Jennings and Hayes (1994): “Existing populations of *B. campi* would be better protected if the areas associated with the springs in which they occur were closed to vehicles and mining (see Marlow et al. 1979). Federal protection would ensure that essential habitats are protected from adverse modification.

Other factors:

At Long John Canyon, flash flooding in 1985 caused a scouring of the canyon bottom, resulting in complete loss of riparian vegetation (Giuliani 1990); the population appeared to be recovering slowly by 1995 (Giuliani 1996). Considerable loss of riparian vegetation occurred at French Spring during flash flooding in 2001 (Hansen and Wake 2005). Extreme weather events are a predicted impact of climate change so the salamanders face the threat of an increased frequency of flooding events in the future.

The species is also threatened by isolation due to the fact that all remaining occupied sites are localized and isolated with small areas of suitable habitat bordered by large expanses of inhospitable desert or semi-desert terrain (Hanson and Wake 2005). Small, isolated populations are more susceptible to extirpations due to stochastic events, human impacts, and environmental factors (Soulé 1987, Begone et al. 1990). Lack of gene flow may cause loss of genetic variability due to random genetic drift (Wright 1931) and inbreeding depression may occur (Franklin 1980). Once these populations are extirpated, their isolation precludes genetic interchange and recolonization of habitat.

References:

AmphibiaWeb: Information on amphibian biology and conservation. [web application]. 2012. Berkeley, California: AmphibiaWeb, *available at* http://amphibiaweb.org (last visited June 19, 2012).

Begone, M., J. L. Harper, and C. R. Townsend. 1990. Ecology: Individuals, populations, and communities. Blackwell Scientific Publications.

California Dept. of Fish and Game. 2011. Special Animals List - January 2011, *available at* http://www.dfg.ca.gov/biogeodata/cnddb/pdfs/SPAnimals.pdf (last visited Jan. 11, 2012).

Franklin, I. R. 1980. Evolutionary changes in small populations. In Conservation Biology: An evolutionary-ecological perspective (ed. a. B. A. W. M. E. Soule). pp. 135-149.

Giuliani, D. 1977. Inventory of habitat and potential habitat for *Batrachoseps* sp. Contract Number CA-101-PH6-805, U.S.D.I. Bureau of Land Management, Bakersfield District Office, Bakersfield, California.

Giuliani, D. 1990. New salamander populations from the eastern Sierra Nevada and Owens Valley region of California with notes on previously known sites. California Department of Fish and Game, Contracts FG7533 and FG8450, Sacramento, California.

Giuliani, D. 1996. Resurvey of eastern Sierra Nevada salamanders. California Department of Fish and Game, Unpublished report, Sacramento, California.

Hammerson, G. 2004. *Batrachoseps campi*. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.2. <www.iucnredlist.org>. Downloaded on 11 January 2012.

Hansen, R.W. and D. Wake. 2005. Batrachoseps campi - Inyo Mountains Salamander. In: M. Lannoo, editor. Amphibian Declines: The Conservation Status of United States Species. University of California Press, Berkeley.

Jennings, M. R., and M. P. Hayes. 1994. Amphibian and reptile species of special concern in California. Final Report submitted to the California Department of Fish and Game, Inland Fisheries Division. Contract No. 8023. 255 pp.

Jockusch, E.L. 2001. Batrachoseps campi. Catalogue of American Amphibians and Reptiles. 722: 1-2.

Marlow, R. W., J. M. Brode, and D. B. Wake. 1979. A new salamander, genus *Batrachoseps*, from the Inyo Mountains of California, with a discussion of relationships in the genus. Contributions in Science of the Natural History Museum of Los Angeles County (308): 1-17.

Nanjappa, P. and Paulette M. Conrad 2012. State of the Union: Legal Authority Over the Use of Native Amphibians and Reptiles in the United States, *available at* http://www.fishwildlife.org/files/SOU_FULL-lo-res.pdf (last visited April 30, 2012).

National Park Service (NPS). 2011. Amphibians, *available at* http://www.nps.gov/deva/naturescience/amphibians.htm (last visited on Jan. 11, 2012).

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, *available at* http://www.natureserve.org/explorer (Accessed: January 11, 2012).

Papenfuss, T.J. and J.R. Macey. 1986. A review of the population status of the Inyo Mountains salamander. U.S. Fish and Wildlife Service, Order Number 10188-5671-5, Endangered Species Office, Sacramento, California.

Soulé, M. 1987. Viable populations for conservation. Cambridge University Press, Great Britain. 189 pp.

Stebbins, R. C. 1985. A field guide to western reptiles and amphibians. Second edition. Houghton Mifflin Company, Boston, Massachusetts. xiv + 336 pp.

Wright, S. 1931. Evolution in Mendelian populations. Genetics 16: 97-159.

Scientific Name:

Batrachoseps minor

Common Name:

Lesser Slender Salamander

G Rank:

G1

IUCN Red List:

Data deficient

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

The viability of Lesser slender salamanders is directly limited by its extremely restricted distribution in the southern Santa Lucia Range of north-central San Louis Obispo County in central coastal California. In the north, they occur immediately north of Black Mountain and range south and east into the drainages of Paso Robles and Santa Rita Creeks (Stebbins 2003, Hansen and Wake 2005). Populations found farther south and west of Atascadero and in the Cuesta Ridge Botanical Area have not been examined for DNA sequences or allozymes but are tentatively assigned to this species (Jockusch et al. 2001). The elevational range is generally 400–640 m (Hansen and Wake 2005).

Habitat:

Lesser slender salamanders appear to be restricted to areas that are either higher in elevation or more mesic than surrounding areas (Jockusch et al. 2001). The type locality is a mesic canyon surrounded by relatively more xeric habitats (Hansen and Wake 2005). Here salamanders were collected from a deeply shaded slope with deep leaf litter. Canopy trees included tanbark oak, coast live oak, sycamore, and laurel; dense shrubs, predominantly poison oak, were also present (Hansen and Wake 2005). A second site, at York Mountain, is characterized by blue oaks and coast live oaks and generally is less heavily shaded (Jockusch et al. 2001).

Biology:

Pacific slender salamanders, such as the lesser slender salamanders, are active underground from April or May until November or December (Kucera 2006). After the first winter rains, when moisture and temperature conditions are favorable, they increase surface activities (Stebbins 1954). They are normally active at night, and return to cover during daylight (Kucera 2006). During periods of extended rainfall, they may remain on the surface during the day to feed (Hendrickson 1954). The salamanders are highly sedentary (Yanev 1978). The salamanders lay eggs during late fall and winter, and the number of eggs per set ranged from 13 to 20 (Kucera 2006). Hatchlings emerge during winter and early spring. It is not known if adults tend their young (Stebbins 1954). Additional information on life history characteristics is provided by Kucera (2006).

Population Status:

Lesser slender salamanders qualify for endangered status because they have become extremely rare within their restricted range, where they are suffering from the threat of roadbuilding and other development. Lesser slender salamanders were common locally about 20 years ago, but they are now difficult to find anywhere (Jockusch et al. 2001). Fewer than five of these salamanders have been seen in the past decade despite many attempts to find them. A single specimen was found in the spring of 2000 following diligent searches by several experienced herpetologists over the course of several years (Encyclopedia of Life 2012). Wake (personal communication, cited in CaliforniaHerps.com 2012) characterizes the species as "genuinely rare." AmphibiaWeb (2012) explains: "This species was common locally about 20 years ago but in the past 10 years it has been almost impossible to find."

<u>THREATS</u>

Habitat alteration and destruction:

The factors in its decline are not well understood, but in the immediate vicinity of the York Mountain Vinyards and Winery, where it was once abundant, modernization and expansion has eliminated some habitat (Hansen and Wake 2005, Encyclopedia of Life 2012). In addition, road mortality has been documented (California Roadkill Observation System 2012).

Overutilization:

Limited collection of the salamanders is allowed under California regulation (CCR § 5.05) but the species does not appear to be a major object of trade.

Inadequacy of existing regulatory mechanisms:

Habitat of the lesser slender salamanders is wholly unprotected because the three known localities where specimen allocation to species has been confirmed by examination of DNA sequences occur on privately owned land (Hansen and Wake 2005). It is not protected at the state or federal level (California Dept. of Fish and Game 2011). Federal protection would prohibit take and ensure that its remaining habitat is adequately protected.

Other factors:

The species is considered highly sensitive to climate change. Further warming, drying or wildfires could conceivably destroy remaining mesic habitat (ARSSC 2009). Pesticide drift from the adjacent winery is also a likely threat.

References:

AmphibiaWeb: Information on amphibian biology and conservation. [web application]. 2012. Berkeley, California: AmphibiaWeb, *available at* http://amphibiaweb.org (last visited June 19, 2012).

ARSSC. 2009. California Amphibian and Reptile Species of Special Concern, *available at* http://arssc.ucdavis.edu/species/species/salamanders.html (last visited Jan. 12, 2012).

California Dept. of Fish and Game. 2011. Special Animals List - January 2011, *available at* http://www.dfg.ca.gov/biogeodata/cnddb/pdfs/SPAnimals.pdf (last visited Jan. 11, 2012).

California Roadkill Observation System. 2012. Amphibians, *available at* http://www.wildlifecrossing.net/california/taxa (last visited Jan. 11, 2012).

CaliforniaHerps.com. 2012. *Batrachoseps minor* - Lesser Slender Salamander, *available at* http://www.californiaherps.com/salamanders/pages/b.minor.html (last visited June 21, 2012).

Encylopedia of Life. 2012. *Batrachoseps minor* - Lesser Slender Salamander, *available at* http://eol.org/pages/332468/overview (last visited Jan. 12, 2012).

Hammerson, G. 2004. *Batrachoseps minor*. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.2. <www.iucnredlist.org>. Downloaded on 12 January 2012.

Hansen, R.W. and D. Wake. 2005. *Batrachoseps minor* - Lesser Slender Salamander. In: M. Lannoo, editor. Amphibian Declines: The Conservation Status of United States Species. University of California Press, Berkeley.

Hendrickson, J. R. 1954. Ecology and systematics of salamanders of the genus Batrachoseps. Univ. Calif. Publ. Zool. 54: 1-46.

Jockusch, E. L., K. P. Yanev, and D. B. Wake. 2001. Molecular phylogenetic analysis of slender salamanders, genus *Batrachoseps* (Amphibia: Plethodontidae), from central coastal California with descriptions of four new species. Herpetological Monographs 15: 54-99.

Kucera, T. 2006. Life history account for Lesser Slender Salamander, *available at* http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=46&ved=0CDsQFjAFOCg&url=http%3A%2F%2Fnrm.dfg.ca.gov%2FFileHandler.ashx%3FDocumentVersionID%3D17817&ei=PfUOT4SjDYH3ggfM84CABA&usg=AFQjCNE-fXLtKbByypK_vvRDyst98anTIw (last visited Jan. 12, 12).

Nanjappa, P. and Paulette M. Conrad. 2012. State of the Union: Legal Authority Over the Use of Native Amphibians and Reptiles in the United States, *available at* http://www.fishwildlife.org/files/SOU_FULL-lo-res.pdf (last visited April 30, 2012).

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, *available at* http://www.natureserve.org/explorer (Accessed: January 12, 2012).

Stebbins, R. C. 1954. Amphibians and reptiles of western North America. McGraw-Hill, New York. 536pp.

Stebbins, R. C. 2003. A Field Guide to Western Reptiles and Amphibians 3rd Ed. Houghton Mifflin Company. New York, New York. pp. 205-6.

Yanev, K. P. 1978. Evolutionary studies of the plethodontid salamander Genus Batrachoseps. Ph. D. Diss. Univ. Calif., Berkeley. 251pp.

Scientific Name:

Batrachoseps regius

Common Name:

Kings River Slender Salamander

G Rank:

G1

IUCN Red List:

Vulnerable

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

Kings River slender salamanders (*Batrachoseps regius*) were previously known from only the drainage of the Kings River (Fresno County) on the western slope of the Sierra Nevada of California (Hansen and Wake 2005). The type locality and nearby sites are located on the south and east sides of the North Fork of the Kings River, at elevations of 335–440 m (Jockusch et al. 1998). In 2002, salamanders discovered in the Middle Fork Kaweah River drainage (610 m elevation, Sequoia National Park, Tulare County) were referred to this species (Jockusch and Wake 2002); this new find extends the range approximately 29 km south from the nearest Kings River drainage population.

Another population, provisionally assigned to this species, is known from Summit Meadow, within the South Fork Kings River drainage, at an elevation of 2,470 m and about 37 kilometers east-southeast (direct distance) of the lower elevation sites (Hansen and Wake 2005).

The intervening areas within the Kings River drainage (a region of difficult terrain and few roads) have not been surveyed adequately for *Batrachoseps*, and it seems likely that additional populations of Kings River slender salamanders could be found (Hansen and Wake 2005).

Even with the additional provisional populations, the viability of the Kings River slender salamander is threatened because of its restricted range.

Habitat:

Individuals occur along streams and in moist wooded canyons in valley foothill riparian habitats, blue oak woodlands, and Sierra mixed conifer woodlands (Yanev 1978). It is known from a well-shaded, north-facing slope in an area of mixed chaparral with *Aesculus*, *Umbellularia*, and *Quercus wislizenii* and scattered *Pinus sabiniana*, *P. ponderosa*, and *Q. douglasii*; found under rocks in areas of talus near the roadside (Jockusch et al. 1998).

Biology:

Little data are available on the life-history of this species, but it is probably similar to other *Batrachoseps* species (AmphibiaWeb 2012). They are completely terrestrial, laying eggs usually under logs or lear litter (AmphibiaWeb 2012). The eggs hatch directly into small terrestrial salamanders, there is no aquatic larval stage (AmphibiaWeb 2012). Available information on life history is summarized by Hansen and Wake (2005).

Population Status:

The Kings River slender salamander qualifies by endangered status because it is extremely rare within its restricted range and faces the threat of habitat alternation. NatureServe (2011) ranks the salamander as critically imperilled. Salamanders have been found in the lower elevations sites intermittently for the last 25 years (Hammerson 2004), but these sites occupy localized habitat immediately adjacent to roads, and thus should be considered vulnerable to alteration (Hansen and Wake 2005). A total of seven specimens have been found at the single high-elevation site on two occasions over a 45-year period (Hansen and Wake 2005). It is considered Vulnerable because it is confirmed from only two locations (Hammerson 2004).

<u>THREATS</u>

Habitat alteration and destruction:

The lower Kings River sites are located immediately adjacent to a road and likely to be affected by road construction (Hansen and Wake 2005), and the road likely has already destroyed some habitat (NatureServe 2012). Also, road mortality has been documented and should be considered a threat (California Roadkill Observation System 2012).

Overutilization:

Limited collection of the salamanders is allowed under California regulation (CCR § 5.05) but the species does not appear to be a major object of trade.

Inadequacy of existing regulatory mechanisms:

The Kings River slender salamander receives no state or federal protection (California Dept. of Fish and Game 2011). All localities for Kings River slender salamanders occur on public lands administered by the U.S. Forest Service or National Park Service (Hansen and Wake 2005). These lands offer inadequate protection, however, because without federal protection, habitat alteration (like road construction) of its few occurrences could cause extirpation. Federal protection would lead to increased attention and funding for the species and necessary recovery actions.

Other factors:

Kings River slender salamanders occupy a geographically small range and are confirmed from only three areas within a single river drainage. Each of these populations appears to be quite localized (Jockusch and Wake 2002). Small, isolated populations, such as those of the Kings River slender salamander, are more susceptible to extirpations due to stochastic events, human impacts, and environmental factors (Soulé 1987, Begone et al. 1990). Lack of gene flow is likely to cause loss of genetic variability due to random genetic drift (Wright 1931) and inbreeding depression may occur (Franklin 1980). Once these populations are extirpated, their isolation precludes genetic interchange and recolonization of habitat.

In addition, as a high altitude species that depends on moist habitats, the Kings River slender salamander is vulnerable to the effects of climate change. The species' moist habitats could dry and become unsuitable with global warming.

References:

AmphibiaWeb: Information on amphibian biology and conservation. [web application]. 2012. Berkeley, California: AmphibiaWeb, *available at* http://amphibiaweb.org (last visited June 19, 2012).

Begone, M., J. L. Harper, and C. R. Townsend. 1990. Ecology: Individuals, populations, and communities. Blackwell Scientific Publications.

California Dept. of Fish and Game. 2011. Special Animals List - January 2011, *available at* http://www.dfg.ca.gov/biogeodata/cnddb/pdfs/SPAnimals.pdf (last visited Jan. 11, 2012).

California Roadkill Observation System. 2012. Amphibians, *available at* http://www.wildlifecrossing.net/california/taxa (last visited Jan. 11, 2012).

Franklin, I. R. 1980. Evolutionary changes in small populations. In Conservation Biology: An evolutionary-ecological perspective (ed. a. B. A. W. M. E. Soule). pp. 135-149.

Hansen, R.W. and D. Wake. 2005. Batrachoseps regius - Kings River Slender Salamander. In: M. Lannoo, editor. Amphibian Declines: The Conservation Status of United States Species. University of California Press, Berkeley.

Hammerson, G. 2004. Batrachoseps regius. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.2. <www.iucnredlist.org>. Downloaded on 12 January 2012.

Jockusch, E.L. and D.B. Wake. 2002. Falling apart and merging: diversification of slender salamanders (Plethodontidae: *Batrachoseps*) in the American West. Biological Journal of the Linnean Society 76: 361–391.

Jockusch, E. L., D. B. Wake, and K. P. Yanev. 1998. New species of slender salamanders, BATRACHOSEPS (Amphibia: Plethodontidae), from the Sierra Nevada of California. Natural History Museum of Los Angeles County Contributions in Science 472: 1-17.

Kucera, T. 2006. Life history account for Kings River Slender Salamander, *available at* http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=17&ved=0CEcQFjAGOAo&url=http%3A%2F%2Fnrm.dfg.ca.gov%2FFileHandler.ashx%3FDocumentVersionID%3D17798&ei=QgEPT8bGBYe62wXNvKnPAg&usg=AFQjCNFayw5Abfd8QeVhHutVxliiLvm2AA (last visited Jan. 12, 2012).

Nanjappa, P. and Paulette M. Conrad. 2012. State of the Union: Legal Authority Over the Use of Native Amphibians and Reptiles in the United States, *available at* http://www.fishwildlife.org/files/SOU_FULL-lo-res.pdf (last visited April 30, 2012).

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, *available at* http://www.natureserve.org/explorer (Accessed: January 12, 2012).

Soulé, M. 1987. Viable populations for conservation. Cambridge University Press, Great Britain. 189 pp.

Wright, S. 1931. Evolution in Mendelian populations. Genetics 16: 97-159.

Yanev, K.P. 1978. Evolutionary studies of the plethodontid salamander genus *Batrachoseps*. Ph.D. dissertation. University of California, Berkeley, California.

Scientific Name:

Batrachoseps relictus

Common Name:

Relictual Slender Salamander

G Rank:

G2

IUCN Red List:

Data deficient

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

Relictual slender salamanders are vulnerable to extinction because they have an extremely narrow range in the southern Sierra Nevada of California that was even further narrowed in a recent study by Jockusch et al. (2012).

Prior to the findings of Jockusch et al. (2012), it was believed that the relictual slender salamanders ranged from the lower Kern River Canyon (Kern County) to highlands drained by the Tule and Kern rivers (Tulare County) (Hansen and Wake 2005), with two principal distributional units: (1) lower Kern River Canyon, the type locality where relictual slender salamanders were found at six sites at elevations of 485–730 m (Brame and Murray 1968); and (2) higher elevations north of the Kern River in the Greenhorn Mountains north to the Tule River drainage, at elevations of 1,125–2,440 m (Hansen and Wake 2005).

Jockusch et al. (2012) placed relictual slender salamanders from the higher elevation unit north of the Kern River in a newly described species, *Batrachoseps altasierrae.* Jockusch et al. (2012)

also assigned a previously unidentified population south of the Kern River on Breckenridge Mountain to the relictual slender salamander.

With the description of *B. altasierrae* by Jockusch et al. (2012), the only populations included in the relictual slender salamander as described by Brame and Murray (1968) that are still recognized as the relictual slender salamander are those from the type locality and vicinity in the Lower Kern River Canyon region of California. However, despite repeated and careful searches, the relictual slender salamander has not been found in the Canyon since 1971 and is presumed extirpated from that locality (Jennings and Hayes 1994a, Hansen 1997, Jockusch et al. 1998, Jockusch et al. 2012). The construction of State Route 178 along the lower, southern slopes of the Canyon severely impacted seepages and springs that once harbored relictual slender salamanders (Hansen and Wake 2005). Although specimens were collected at these sites for a number of years following highway construction, this loss of habitat likely initiated local population declines that finally reached non-sustainable levels (Hansen 1988).

Thus, the only known population of relictual slender salamander is the population recently identified by Jockusch et al. (2012) on Breckenridge Mountain. The relictual slender salamander is now considered a Breckenridge Mountain endemic, with an elevational range from 480 m along the Kern River to at least 2000 m. Despite this large elevational range, the total area occupied by the species is very small, with a maximum distance of 15 km separating historic populations, and less than 5 km separating the only two known extant populations, giving it the smallest known range for a described species of *Batrachoseps* (Jockusch et al. 2012).

Habitat:

Brame and Murray (1968) described relictual slender salamanders from the Kern River Canyon as "semiaquatic" and Hilton (1948) noted that specimens were "in a slightly unusual situation for them [*Batrachoseps*]; in a very wet place, in and out of the water of a spring." As explained above, salamanders from this locality are likely extirpated.

All recent sightings of the relictual slender salamander (from the Breckenridge Mountain) have been from two small high elevation sites in pine-fir forest in close association with water, reminiscent of the "semiaquatic" habitat reported for the salamander from the Lower Kern River Canyon (Jockusch et al. 2012). At one of its extant localities (east of Squirrel Meadow, 2000 m elev.), Jockusch et al. (2012) typically found relictual slender salamanders directly associated with a small seep. Salamanders have been found beneath rocks with water underneath, typically on a sandy-gravel substrate (Jockusch et al. 2012). Rarely, salamanders have been found beyond surface water; Jockusch et al. (2012) found two adults within a moist log about 45 m upslope from the seep area. At the second Breckenridge Mountain site (headwater seepage of Lucas Creek, 1665–1700 m elev.), all specimens were found under cover objects directly beside the stream over a distance of about 750 m (Jockusch et al. 2012).

Biology:

Information on the natural history of the relictual slender salamander is largely unavailable because virtually all published reports (Hanson and Wake 2005, Kucera 1997) are from populations that Jockusch et al. (2012) has now placed in *B. altasierrae*. As with other members of the genus, direct development is presumed (Jennings and Hayes 1994). Feeding behavior is not well known, but other *Batrachoseps* species are sit-and-wait predators that use a projectile tongue to catch small invertebrate prey (CaliforniaHerps.com 2012).

Population Status:

The relictual slender salamander qualifies for endangered status because the lower Kern River unit is extirpated and the Breckenridge Mountain unit has an extremely limited range with a small population size that has declined due to timber harvest and road construction. Jockusch et al. (2012) recommends formal protection for the relictual slender salamander "[o]n the basis of its limited range, small number of known populations, apparent small population sizes, and apparent extirpation at lower elevations." Both the previously unassigned population on Breckenridge Mountain and the relictual slender salamander were considered candidates for federal protection until the FWS eliminated the C2 category.

THREATS

Habitat alteration and destruction:

Disappearance of populations in the lower Kern River Canyon likely resulted from habitat changes caused during construction of State Route 178 (Hansen and Wake 2005). Road margin habitat has been severely degraded by road maintenance and related construction activities.

The salamanders recently identified at Breckenridge Mountain have suffered from habitat loss and degradation. The population east of Squirrel Meadow was severely degraded by the construction of a logging road through the seepage area subsequent to its discovery in 1979 (Jockusch et al. 2012). Black oaks which bordered the habitat were cut down, altering the structure and hydrology of the habitat (CaliforniaHerps. com 2012). A subsequent fire and timber harvest further compromised this site, and salamanders were not found here despite multiple searches over the next 22 years (Jockusch et al. 2012). Recent visits suggest that the population has rebounded somewhat, but prime seep habitat is quite small (Jockusch et al. 2012).

Overutilization:

Limited collection of the salamanders is allowed under California regulation (CCR § 5.05) but the species does not appear to be a major object of trade.

Inadequacy of existing regulatory mechanisms:

Relictual slender salamanders are a Species of Special Concern (California Department of Fish and Game 2011) and a Forest Service Sensitive Species (California Dept. of Fish and Game 2011). Sensitive species designations do not bar activities that threaten the species but only require analysis of those impacts.

The salamander occurs on public lands administered by the U.S. Forest Service. Some forest plan guidance and use of best management practices aimed at retaining large woody debris likely benefits the salamanders. But this guidance is largely voluntary and has not protected the salamander from habitat degradation and destruction from threats such as roadbuilding and logging (Hansen and Wake 2005, Jockusch et al. 2012).

Federal protection would lead to habitat protections vital for preservation of the relictual slender salamander and other endemics to the Kern River Canyon. The Kern River Canyon at the southern end of the Sierra Nevada has both a high level of endemism and of diversity (Lapointe and Rissler 2005). As a result of the endemism for both species and subspecific lineages, this area was identified as "irreplaceable" in an analysis of California biodiversity (Rissler et al. 2006).

Other Factors:

Relictual slender salamanders have an extrememly restricted range with only two small, isolated populations remaining. Small, isolated populations are more susceptible to extirpations due to stochastic events, human impacts, and environmental factors (Soulé 1987, Begone et al. 1990). Lack of gene flow is likely to cause loss of genetic variability due to random genetic drift (Wright 1931) and inbreeding depression may occur (Franklin 1980). Once these populations are extirpated, their isolation precludes genetic interchange and recolonization of habitat.

References:

AmphibiaWeb: Information on amphibian biology and conservation. [web application]. 2012. Berkeley, California: AmphibiaWeb, *available at* http://amphibiaweb.org (last visited June 19, 2012).

Begone, M., J. L. Harper, and C. R. Townsend. 1990. Ecology: Individuals, populations, and communities. Blackwell Scientific Publications.

Brame, A.H., Jr. and K.F. Murray. 1968. Three new slender salamanders (*Batrachoseps*) with a discussion of relationships and speciation within the genus. Science Bulletin of the Natural History Museum of Los Angeles County, Number 4, Los Angeles, California.

CaliforniaHerps.com. 2012. Batrachoseps relictus - Relictual Slender Salamander, *available at* http://www.californiaherps.com/salamanders/pages/b.relictus.html (last visited June 21, 2012).

California Dept. of Fish and Game. 2011. Special Animals List - January 2011, *available at* http://www.dfg.ca.gov/biogeodata/cnddb/pdfs/SPAnimals.pdf (last visited Jan. 11, 2012).

California Roadkill Observation System. 2012. Amphibians, *available at* http://www.wildlifecrossing.net/california/taxa (last visited Jan. 11, 2012).

Franklin, I. R. 1980. Evolutionary changes in small populations. In Conservation Biology: An evolutionary-ecological perspective (ed. a. B. A. W. M. E. Soule). pp. 135-149.

Hammerson, G. 2004. Batrachoseps relictus. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.2. <www.iucnredlist.org>. Downloaded on 12 January 2012.

Hansen, R.W. 1980. The Kern Plateau herpetofauna: a preliminary survey. U.S.D.A. Forest Service, Unpublished report, Sequoia National Forest, Porterville, California.

Hansen, R.W. 1997. Salamander diversity in the Kern Valley region. Kern River Research Center Fieldnotes 6: 1-3.

Hansen, R.W. and D. Wake. 2005. Batrachoseps relictus - Relictual Slender Salamander. In: M. Lannoo, editor. Amphibian Declines: The Conservation Status of United States Species. University of California Press, Berkeley.

Jennings, M. R., and M. P. Hayes. 1994. Amphibian and reptile species of special concern in California. Rancho Cordova: California Department of Fish and Game, Inland Fisheries Division.

Jockusch, E. L., D. B. Wake, and K. P. Yanev. 1998. New species of slender salamanders, BATRACHOSEPS (Amphibia: Plethodontidae), from the Sierra Nevada of California. Natural History Museum of Los Angeles County Contributions in Science 472: 1-17.

Jockusch, E., I. Martinez-Solano, R.W. Hanson, and D.B. Wake. 2012. Morphological and molecular diversification of slender salamanders (Caudata: Plethodontidae: Batrachoseps) in the southern Sierra Nevada of California with descriptions of two new species. Zootaxa 3190: 1–30.

Kucera, T. 1997. Life history account for Relictual Slender Salamander, *available at* http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=43&ved=0CCkQFjACOCg&url=http%3A%2F%2Fnrm.dfg.ca.gov%2FFileHandler.ashx%3FDocumentVersionID%3D17962&ei=_hUPT8TPGsKU2QW4s4DBAg&usg=AFQjCNFjxg7CLGtU6UoD6AS6g0G3tXZETA

Lapointe, F.J., and L.J. Rissler. 2005. Congruence, consensus, and the comparative phylogeography of codistributed species in California. American Naturalist 166: 290–299.

Nanjappa, P. and Paulette M. Conrad. 2012. State of the Union: Legal Authority Over the Use of Native Amphibians and Reptiles in the United States, *available at* http://www.fishwildlife.org/files/SOU_FULL-lo-res.pdf (last visited April 30, 2012).

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, *available at* http://www.natureserve.org/explorer (Accessed: January 12, 2012).

Rissler, L.J., R.J. Hijmans, C.H. Graham, C. Moritz, and D.B. Wake. 2006. Phylogeographic lineages and species comparisons in conservation analyses: a case study of California herpetofauna. American Naturalist 167: 655–666.

Soulé, M. 1987. Viable populations for conservation. Cambridge University Press, Great Britain. 189 pp.

Stebbins, R. C. 1972. California Amphibians and Reptiles. University of California Press, Berkeley, California.

Steinhart, P. 1990. California's wild heritage: Threatened and endangered animals in the Golden State. San Francisco: California Department of Fish and Game, California Academy of Sciences, and Sierra Club Books.

Wright, S. 1931. Evolution in Mendelian populations. Genetics 16: 97-159.

Scientific Name:

Batrachoseps robustus

Common Name:

Kern Plateau Salamander

G Rank:

G2

IUCN Red List:

Near Threatened

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

Kern Plateau salamanders are endemic to the southeastern Sierra Nevada of California and have only recently been described (Wake et al. 2002). Their range consists of three principal units: the Kern Plateau (Tulare County), where they have been recorded from a number of sites; the eastern slopes of the Sierra Nevada draining into Owens and Indian Wells valleys (Inyo County), where they are restricted to steep, east-facing canyons; and the Scodie Mountains (Kern County), south of the main body of the species' range (Hansen and Wake 2005). The elevational range is from 1,700–2,800 m on the Kern Plateau (Tulare County); 1,430–2,440 m on the Sierran east slope (Inyo County); and 1,980–2,025 m in the Scodie Mountains (Kern County) (Hansen and Wake 2005).

Habitat:

The salamander occurs in moist areas among a variety of montane conifer, hardwood and shrub species, including Jeffrey pine and red fir in the northern and more humid parts of its range, and lodgepole pine, pinyon pine, black oak canyon oak, big sagebrush and rabbitbrush in the drier areas (Wake et al. 2002, Stebbins 2003). The habitat is of limited extent, especially the springs of the Kern Plateau and Scodie Mountains (NatureServe 2011). The habitats are vulnerable to degradation through capping of springs by humans or other alterations.

Biology:

Wake et. al (2002) hypothesize that surface activity for salamanders present at elevations below 2000 m is restricted to late winter and early spring, before surfaces heat up and lose their moisture. At high elevations, their activity is restricted to between the months of May or June to October, when temperatures are warmer and snow levels have dropped enough to provide conditions for courtship and egg-laying (AmphibiaWeb 2012). It breeds terrestrially and clutch size, at least at higher elevations, is small and egg and hatchling size relatively large (AmphibiaWeb 2012). Development of eggs in one experiment was shown to occur between 96 and 103 days (AmphibiaWeb 2012). Juvenille salamanders were more frequently seen above ground in the late season than adults, probably due to their greater need to build up fat deposits for winter (AmphibiaWeb 2012). When handled, salamanders try to crawl away, coil, or thrash, and sometimes produce a sticky skin secretion (AmphibiaWeb 2012). Life history is summarized by Hansen and Wake (2005) and CWHR Program Staff (2005).

Population Status:

The Kern Plateau salamander is endangered because it has a restricted range with habitat that is vulnerable to alteration by humans and flash floods (NatureServe 2011). The IUCN Red List ranks the species as Near Threatened but explains that it is close to qualifying for Vulnerable (Hammerson 2004).

THREATS

Habitat alteration and destruction:

The Kern Plateau salamander is vulnerable to habitat degradation through capping of springs by humans and other alterations of spring water or habitat (NatureServe 2011).

The salamander is threatened by renewable energy projects, specifically, plans to develop wind energy near its habitat (Evelyn 2010). It would be adversely affected by the hilltop grading and soil displacement associated with wind mill installation sites and access roads (Evelyn 2010).

This species is susceptible to road mortality (California Roadkill Observation System 2012).

Overutilization:

Limited collection of the salamanders is allowed under California regulation (CCR § 5.05) but the species does not appear to be a major object of trade. No slender salamanders (*Batrachoseps* spp.) may be taken from Inyo County (Nanjappa and Conrad 2012).

Inadequacy of existing regulatory mechanisms:

These salamanders are found on public and private land (Hammerson 2004). The Kern Plateau and Scodie Mountain populations are on U.S. Forest Service land. Although this land is protected from development, it is vulnerable to habitat degradation from threats such as road building (see Hammerson 2004). It is also a Forest Service Sensitive Species (California Dept. of Fish and Game 2011), but this designation does not prevent activities that destroy habitat. Without federal protection, essential habitats are likely to be destroyed for wind mill installation sites and access roads.

References:

AmphibiaWeb: Information on amphibian biology and conservation. [web application]. 2012. Berkeley, California: AmphibiaWeb, *available at* http://amphibiaweb.org (last visited June 19, 2012).

California Dept. of Fish and Game. 2011. Special Animals List - January 2011, *available at* http://www.dfg.ca.gov/biogeodata/cnddb/pdfs/SPAnimals.pdf (last visited Jan. 11, 2012).

California Roadkill Observation System. 2012. Amphibians, *available at* http://www.wildlifecrossing.net/california/taxa (last visited Jan. 11, 2012).

CWHR Program Staff. 2005. Life History Account of Kern Plateau Salamander, *available at* http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=53&ved=0CCwQFjACODI&url=http%3A%2F%2Fnrm.dfg.ca.gov%2FFileHandler.ashx%3FDocumentVersionID%3D17796&ei=Ny0PT6mfNKny2QWH9tz_Ag&usg=AFQjCNHImpo68PAyV1k_ZI8aBNBF5vF8qw (last visited Jan. 12, 2012).

Evelyn, C. 2010. Comments on Draft of the Desert Renewable Energy Conservation Plan, *available at* http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=27&ved=0CE8QFjAGOBQ&url=http%3A%2F%2Fwww.drecp.org%2Fdocuments%2Fcomments_independent_science%2FUC_Santa_Barbara_Chris_Evelyn.pdf&ei=1isPT6rWGefq2QX8h6DWAg&usg=AFQjCNFtmIb1kiY9D_47T2wgrHEuqmKOVw (last visited Jan. 12, 2012).

Hammerson, G. 2004. *Batrachoseps robustus*. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.2. <www.iucnredlist.org>. Downloaded on 12 January 2012.

Hansen, R.W. and D. Wake. 2005. *Batrachoseps robustus* - Kern Plateau Salamander. In: M. Lannoo, editor. Amphibian Declines: The Conservation Status of United States Species. University of California Press, Berkeley.

Nanjappa, P. and Paulette M. Conrad. 2012. State of the Union: Legal Authority Over the Use of Native Amphibians and Reptiles in the United States, *available at* http://www.fishwildlife.org/files/SOU_FULL-lo-res.pdf (last visited April 30, 2012).

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, *available at* http://www.natureserve.org/explorer (Accessed: January 12, 2012).

Stebbins, R. C. 2003. A Field Guide to Western Reptiles and Amphibians 3rd Ed. Houghton Mifflin Company. New York, New York. pp. 205-6.

Wake, D. B., K. P. Yanev, and R. W. Hansen. 2002. New species of slender salamander, genus *Batrachoseps*, from the southern Sierra Nevada of California. Copeia 2002: 1016-1028.

Scientific Name:

Batrachoseps simatus

Common Name:

Kern Canyon Slender Salamander

G Rank:

G2

IUCN Red List:

Vulnerable

<u>NATURAL HISTORY, BIOLOGY, AND STATUS</u>

Range:

The Kern Canyon slender salamander is vulnerable to extinction because of its extremely restricted range and ongoing threats of habitat destruction and degradation. It is known from only the lower Kern River Canyon, Kern County, California (Stebbins 1985, California Department of Fish and Game 1990). The Kern River serves as a biogeographic break with the Kern Canyon slender salamander known only from the south side of the river (Jockusch et al. 2012). Some researchers had thought that populations from Fairview, Tulare County might be Kern Canyon slender salamanders but Jockusck et al. (2012) assigned those salamanders to a new species, *B. bramei*, Fairview slender salamander.

Habitat:

The salamander favors north-facing slopes and small wooded tributary canyons (NatureServe 2011). It is found in oak-pine communities on slopes; willow and cottonwood communities along streams; may range into grassland adjacent to woods (NatureServe 2011). Biosystems

Analysis (1989) also mentioned chaparral as a habitat. Often found in crevices in talus slopes or under rocks and logs (NatureServe 2011).

Biology:

Little is known about the life history. There is one report from Breckenridge Mountain of a female laying eggs in mid-June (Stebbins 2003). Available life history information is summarized by Morey and Basey (2008).

Population Status:

The Kern Canyon slender salamander needs federal protection because it is uncommon and found in just a few localities within its extremely restricted range. The IUCN Red List ranks the species as vulnerable because it is known from fewer than five locations (Hammerson 2004). It is uncommon within its restricted range (Morey and Basey 2008). The species was a candidate for federal protection until the FWS eliminated the C2 category, but it currently receives no federal protection under the ESA.

THREATS

Habitat alteration and destruction:

Threats include habitat destruction/degradation caused by cattle grazing, logging, mining, highway construction, small hydro development, and fire wood collecting (Hammerson 2004, NatureServe 2011). Proposed development of water storage facilities within the Kern River Canyon is another threat (Hammerson 2004).

Construction of State Highway 178 through Kern Canyon diminished the salamander population, and further development of this highway could cause additional population declines (Ellis 1987).

Overutilization:

Limited collection of the salamanders is allowed under California regulation (CCR § 5.05) but the species does not appear to be a major object of trade.

Inadequacy of existing regulatory mechanisms:

Nearly all the known populations occur on public lands administered by the Sequoia National Forest. While these lands are protected from development, they remain vulnerable to degradation from the multiple uses that occur on national forests, such as logging, grazing, and mining (Hammerson 2004, NatureServe 2011). It is a U.S. Forest Service Sensitive Species, but that designation does not prevent activities that could harm the species and instead only requires analysis of impacts. Indeed, Ellis (1987) found that the Forest Service “planned land use activities which are not compatible with protection of this species.” For example, the Forest allows firewood collection, which conflicts with the salamander’s need for downed wood and surface debris.

Kern Canyon slender salamanders are listed as Threatened by the State of California (California Dept. of Fish and Game 2011). While the state listing prohibits take and provides some protection against habitat destruction, federal ESA protection would add additional protections on federal lands through section 7 consultation and critical habitat designation and would allow for use of section 6 funds to recover the species.

Other factors:

The Kern Canyon slender salamander occurs in isolated colonies (NatureServe 2011). Small, isolated populations are more susceptible to extirpations due to stochastic events, human impacts, and environmental factors (Soulé 1987; Begone et al. 1990). Lack of gene flow is likely to cause loss of genetic variability due to random genetic drift (Wright 1931) and inbreeding depression may occur (Franklin 1980). Once these populations are extirpated, their isolation precludes genetic interchange and recolonization of habitat.

References:

AmphibiaWeb: Information on amphibian biology and conservation. [web application]. 2012. Berkeley, California: AmphibiaWeb, *available at* http://amphibiaweb.org (last visited June 19, 2012).

Begone, M., J. L. Harper, and C. R. Townsend. 1990. Ecology: Individuals, populations, and communities. Blackwell Scientific Publications.

Biosystems Analysis, Inc. 1989. Endangered Species Alert Program Manual: Species Accounts and Procedures. Southern California Edison Environmental Affairs Division.

California Dept. of Fish and Game. 2011. Special Animals List - January 2011, *available at* http://www.dfg.ca.gov/biogeodata/cnddb/pdfs/SPAnimals.pdf (last visited Jan. 11, 2012).

Ellis, S.R. 1987. Five Year Status review for Kern Canyon slender salamander, *available at* http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=22&ved=0CCUQFjABOBQ&url=http%3A%2F%2Fnrm.dfg.ca.gov%2FFileHandler.ashx%3FDocumentVersionID%3D55769&ei=elgPT8vPB8KKgwftx7HaAw&usg=AFQjCNF4lyZLNqamJgj3ie8T4VxUXowcnw (last visited Jan. 12, 2012).

Franklin, I. R. 1980. Evolutionary changes in small populations. In Conservation Biology: An evolutionary-ecological perspective (ed. a. B. A. W. M. E. Soule). pp. 135-149.

Hammerson, G. 2010. *Batrachoseps simatus*. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.2. <www.iucnredlist.org>. Downloaded on 12 January 2012.

Morey, S. and H. Basey. 2008. Life history account for Kern Canyon slender salamander, *available at* https://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=1&ved=0CDEQFjAA&

url=https%3A%2F%2Fnrmsecure.dfg.ca.gov%2FFileHandler.ashx%3FDocumentVersionID%3D17795&ei=0iLjT_2bG8fq2AXRu7XYCw&usg=AFQjCNF8i8y3VShRJQbqtHdIasR7_HiTrg&sig2=T-tUQzr2oOhrxFaWlzxnlA&cad=rja (last visited Jan. 12, 2012).

Nanjappa, P. and Paulette M. Conrad. 2012. State of the Union: Legal Authority Over the Use of Native Amphibians and Reptiles in the United States, *available at* http://www.fishwildlife.org/files/SOU_FULL-lo-res.pdf (last visited April 30, 2012).

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, *available at* http://www.natureserve.org/explorer (Accessed: January 12, 2012).

Soulé, M. 1987. Viable populations for conservation. Cambridge University Press, Great Britain. 189 pp.

Stebbins, R. C. 1985. A field guide to western reptiles and amphibians. Second edition. Houghton Mifflin Company, Boston, Massachusetts. xiv + 336 pp.

Stebbins, R. C. 2003. A field guide to western reptiles and amphibians. Third edition. Houghton Mifflin Company, Boston.

Wright, S. 1931. Evolution in Mendelian populations. Genetics 16: 97-159.

Scientific Name:

Batrachoseps wrighti (previously *B. wrightorum*)

Common Name:

Oregon Slender Salamander

G Rank:

G2

IUCN Red List:

Vulnerable

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

The range of the Oregon slender salamander includes western Oregon from the Columbia River Gorge in Multnomah and Hood River counties southward in the Cascade Mountains to southern Lane County (NatureServe 2011). Most of the range is on the western slopes of the Cascades, but several sites are on the eastern slope in Hood River and Wasco counties (Kirk 1991, Stebbins 2003, Jones et al. 2005; see also Kirk and Forbes 1991). Animals are found from about 15 m elevation (in the Columbia Gorge) to around 1400 m (Nussbaum et al. 1983, Leonard et al. 1993, Petranka 1998, Stebbins 2003). See Nussbaum et al. (1983) and Kirk (1991) for spot maps.

Habitat:

The Oregon slender salamander is an old-growth forest associated species that is dependent on moist microhabitat in Douglas-fir, mixed maple, hemlock, and red-cedar forests. It is strongly correlated with high amounts of decaying large-diameter down woody debris on the forest floor. The salamander is significantly positively associated with high canopy closure, snags, logs in the 50 to 75 cm (20 to 30 in) diameter class, and east and west aspects. Downed logs that are too small or in low or intermediate levels of decay do not provide adequate microhabitat. The salamander is negatively correlated with small logs (10 to 25 cm, 4 to 10 in) logs and logs in intermediate levels of decay (classes 2 and 3) (Clayton and Olson 2009).

Numerous studies have shown that the salamander is most abundant in late-successional forest and cannot persist in young clearcuts (Bury and Corn 1988, Gilbert and Allwine 1991, Vesely 1999, Clayton and Olson 2009, Walcott 2010). The Oregon slender salamander has very limited mobility and clearcuts result in population isolation and increased extinction risk. The Oregon slender has been detected in forests of all seral stages, but only when a sufficient amount of large-diameter woody debris in advanced decay classes is present. Logging standards on private and state timber lands and on Matrix lands under the Northwest Forest Plan do not provide sufficient amounts of downed woody debris to provide for the long-term persistence of the Oregon slender salamander. Oregon slenders may persist in remnant downed wood in logged areas for a few generations, but isolated populations will ultimately be extirpated when woody debris disintegrates and is not replaced.

The Oregon slender salamander also occurs under moss-covered rocks in the Columbia River Gorge and in stabilized talus and lava flows elsewhere (Nussbaum et al. 1983, Jones et al. 2005). Individuals can also be found in the ground and in termite burrows.

Biology:

Many aspects of Oregon slender salamander life history make them susceptible to habitat fragmentation (Miller et al. 2005). These traits include low reproductive rate (delayed onset of female oviposition until 4–5 years of age, clutch size averages 6.3 eggs; Tanner 1953) and low rates of dispersal. Although there have been few studies of dispersal, movement, and home range size in slender salamanders (Genus *Batrachoseps*, summarized in Stebbins and Cohen 1995), home range size of the California slender salamander (*B. attenuates*) was observed to have a diameter of only 1.7 m (Hendrickson 1954). Thus, home range size and dispersal are also thought to be limited in the Oregon slender salamander (Miller et al. 2005). Genetic analyses corroborate that dispersal is likely limited in this species. Mitochondrial DNA analysis shows evidence of two major lineages, a northern and southern population, and random amplified polymorphic DNA (RAPD) analysis shows a pattern of isolation by distance (Wagner 2000).

Population Status:

The Oregon slender salamander qualifies for endangered status because it is facing declines over a restricted range due to widespread logging and other threats. The area of occupancy, number of subpopulations, population size, and habitat quality have declined over the long term. As of the early 1990s, at least 65 percent of Oregon's old-growth forest had already been logged (Bolsinger and Waddell 1992). Because the Oregon slender salamander is a habitat specialist, it can be assumed that habitat loss reflects population loss. Further, many isolated remnant populations will be extirpated as downed wood decays and is not replaced. The IUCN Red List considers the Oregon slender vulnerable to extinction because its small range is severely fragmented and because there is continuing decline in the extent and quality of its forest habitat in Oregon (Hammerson and Bury 2004).

THREATS

Habitat alteration and destruction:

Logging of late-successional forest is the primary threat to the Oregon slender salamander. Logging removes canopy closure, disturbs substrates, and alters microhabitat refuges and microclimates. Tree-felling and ground-based logging systems mechanically disturb the substrate and ground cover which can result in both substrate compaction and loss of the integrity of existing down wood. These actions can result in loss of interstices used by salamanders as refuges and for their movements, and a drying out of the ground surface. Loss of standing live trees reduces the future potential for down wood recruitment, and as new trees regenerate in harvested stands, their smaller sizes likely do not provide the same functions for salamanders for several decades to centuries (Clayton and Olson 2009).

The Oregon slender has declined as a result of widespread logging practices that have eliminated or reduced its necessary microhabitat (Nussbaum et al. 1983, Bury and Corn 1988, Gilbert and Allwine 1991, Jones et al. 2005, Clayton and Olson 2009). Large, decayed logs used by slender salamanders for nesting are rare in clearcuts and plantations, and so forests intensively managed on short harvest rotations are population sinks (Vesely 1999). Marshall et al. (1992) conclude that forest management practices are likely to lead to local extirpation and negatively affect overall species viability.

In addition, dispersal patterns of northwest mesic forest-associated organisms like the Oregon slender salamander are influenced by historical biogeographic boundaries as well as current processes of forest fragmentation (i.e., timber harvests and rural development; U.S. Forest Service and U.S. Bureau of Land Management 1994). Subsequently, mature forest-associated species with limited dispersal capabilities like the salamander are severely impacted by increasing fragmentation (Miller et al. 2005). The entire range of the Oregon slender is highly fragmented by past timber harvest, and logging poses an ongoing threat to the species' survival.

Inadequacy of existing regulatory mechanisms:

No existing regulatory mechanisms adequately protect the Oregon slender salamander. Oregon classifies the salamander as a sensitive species of vulnerable status (Oregon Dept. of Fish and Wildlife 2008), which requires a permit prior to collection but does not protect the salamander from habitat loss, the primary threat to its survival. No regulatory mechanisms protect the salamander from logging on state or private lands.

The salamander is ranked as a special status species by federal land management agencies: U.S. Forest Service, Region 6 - Sensitive; Bureau of Land Management, Oregon – Sensitive. There are 740 known sites for the Oregon slender salamander, which collapse to 407 sites when locations within 200 m of each other are combined (Clayton and Olson 2009). The vast majority of known sites for the salamander are on federal land (687 of 740 records; 93 percent). Federal sites are managed under the Northwest Forest Plan. Known sites are located on the Salem and Eugene BLM Districts, and the Mount Hood and Willamette National Forests. Management of the species follows Forest Service 2670 Manual policy and BLM 6840 Manual direction.

The entire range of the Oregon slender salamander has been fragmented by logging, and the majority of the known sites where it occurs are on lands where logging is an ongoing threat. Eighty percent (595 of 740) of salamander localities occur on land allocations in which timber harvest activities may occur (nonfederal lands, federal Matrix and Adaptive Management Areas). Looking at federal lands only, 542 of 687 sites (79 percent) occur on land with programmed timber harvest (Clayton and Olson 2009).

Only 19 percent of known salamander localities are on federally reserved lands under the Northwest Forest Plan. Using the minimum convex polygon estimate of occurrence across the landscape, 31 percent of the salamander's range is potentially in federal reserves, not including Riparian Reserves. (Riparian Reserves are likely of little benefit to the salamander because they are too narrow to provide adequate habitat.) Under current timber harvest practices on non-reserve lands, the salamander will ultimately be extirpated as surviving generations in remnant patches of decaying downed wood blink out and are unable to disperse or be rescued due to limited dispersal capabilities and inhospitable habitat patches in the fragmented landscape. The low percentage of the salamander's range that is protected combined with genetic isolation make the Oregon slender salamander at high risk of extinction.

The threat to the salamander's survival from logging is further exacerbated by current legislative and administrative proposals to increase logging on federal lands in western Oregon. Among these proposals are: 1) H.R. 4019, the "Federal Forests County Revenue, Schools, and Jobs Act of 2012," 2) a "Legislative Counsel Draft" of the ''O&C Trust, Conservation, and Jobs Act,'' 3) "Oregon Forestry Plan" Revisions for western Oregon BLM, and 4) a proposed increase in "Active Management" including clearcutting in northern spotted owl critical habitat. Current regulatory mechanisms on federal lands are not adequate to ensure the salamander's long-term persistence, and the multiple proposals to increase logging on federal lands in western Oregon place the Oregon slender salamander at even higher risk of extinction.

Other factors:

As explained below, the Oregon slender salamander is threatened by several other factors including reduced gene flow, pesticides, and global climate change.

Wagner and Haig (2000) found reduced amount of gene flow between populations of the Oregon slender salamander. Small, isolated populations are more susceptible to extirpations due to stochastic events, human impacts, and environmental factors (Soulé 1987; Begone et al. 1990). Lack of gene flow is likely to cause loss of genetic variability due to random genetic drift (Wright 1931) and inbreeding depression may occur (Franklin 1980). Once salamander populations are extirpated, their isolation precludes genetic interchange and recolonization of habitat. The ongoing logging of the salamander's habitat is causing increasing population isolation and magnified extinction risk.

Chemicals such as herbicides, pesticides, fungicides, fertilizers and fire retardants may have a direct impact on all woodland salamanders (Clayton and Olson 2009). These chemicals are widely applied following logging operations in the Pacific Northwest.

The range of the Oregon slender salamander includes habitats that are particularly vulnerable to predicted patterns of global climate change (Clayton and Olson 2009). In particular, a change in storm patterns that alters annual snow accumulation or seasonal pattern of rainfall and snowfall could negatively impact the Oregon slender salamander. Warming trends will further restrict the salamander's already restricted distribution. Altered fire regimes, altered vegetation conditions, and changes in predator and prey communities pose additional threats from climate change.

References:

Begone, M., J. L. Harper, and C. R. Townsend. 1990. Ecology: Individuals, populations, and communities. Blackwell Scientific Publications.

Bolsinger, C.L. and K.L. Waddell. 1993. Area of old-growth forests in California, Oregon, and Washington. U.S. Department of Agriculture Pacific Northwest Research Station Resource Bulletin PNW-R8-197. 32 pp., *available at*: http://www.fs.fed.us/r5/rsl/publications/oldgrowth/old-growth-ca-or-wa.pdf (last visited June 26, 2012).

Bury, B. 2005. *Batrachoseps wrighti* - Oregon Slender Salamander. In: M. Lannoo, editor. Amphibian Declines: The Conservation Status of United States Species. University of California Press, Berkeley.

Bury, R. B., and P. S. Corn. 1988. Douglas-fir forests in the Oregon and Washington Cascades: relation of the herpetofauna to stand age and moisture. Pages 11-22 in Management of amphibians, reptiles and small mammals in North America. USDA Forest Service Rocky Mountain Research Station, Ft. Collins, Colorado. Gen. Tech. Rep. RM-166.

Clayton, D. R., and D. H. Olson. 2007. Conservation assessment for the Oregon slender salamander (*Batrachoseps wrighti*). Version 1.0.

Clayton, D.R. and D.H. Olson. 2009. Conservation assessment for the Oregon slender salamander (*Batrachoseps wrighti*). Version 2.0. U.S.D.A. Forest Service Region 6 and U.S.D.I. Bureau of Land Management. 76 pp., *available at:* www.fs.fed.us/r6/sfpnw/issssp/.../ca-ha-bawr-ver2-2009-01.doc (last visited June 26, 2012).

Doerr, J. 2008. 2008 ISSSSP Surveys for Oregon Slender Salamander (*Batrachoseps wrightorum*) on federal lands managed by the Eugene Bureau of Land Management and the Willamette National Forest, *available at* http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=22&ved=0CCcQFjABOBQ&url=http%3A%2F%2Fwww.fs.fed.us%2Fr6%2Fsfpnw%2Fissssp%2Fdocuments%2Finventories%2Finv-rpt-ha-bawr-wil-eug-surveys-2008-

10.pdf&ei=9kQQT7mbFpHpggebrNDKAw&usg=AFQjCNH2h3UiQmM6QuqumwRfMTPX8j31fg (last visited Jan. 13, 2012).

Dowlan, S. 2006. Conservation Assessment and Management Recommendations for Oregon slender salamander, *Batrachoseps wrightorum* (*wrighti*). Cascades Resource Area, Salem District, Bureau of Land Management. Unpublished.

Franklin, I. R. 1980. Evolutionary changes in small populations. In Conservation Biology: An evolutionary-ecological perspective (ed. a. B. A. W. M. E. Soule). pp. 135-149.

Gilbert, F. F., and R. Allwine. 1991. Terrestrial amphibian communities in the Oregon Cascade Range. Pages 318-324 in L. F. Ruggiero, K. B. Aubry, A. B. Carey, and M. H. Huff, technical coordinators. Wildlife and vegetation of unmanaged Douglas-fir forests. USDA Forest Service, Pacific Northwest Research Station, General Technical Report PNW-GTR-285, Portland, OR.

Hammerson, G. and B. Bury 2004. *Batrachoseps wrighti*. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.2. <www.iucnredlist.org>. Downloaded on 13 January 2012.

Hendrickson, J.R. 1954. Ecology and systematics of salamanders of the genus Batrachoseps. Univ. Calif. Publ. Zool. 54: 1–46.

Jones, L.L.C., W. P. Leonard, and D. H. Olson, editors. 2005. Amphibians of the Pacific Northwest. Seattle Audubon Society, Seattle, Washington. xii + 227 pp.

Kirk, J.J. 1991. Batrachoseps wrighti. Catalogue of American Amphibians and Reptiles. 506: 1-3.

Kirk, J.J. and R.B. Forbes. 1991. Geographic distribution: *Batrachoseps wrighti* (Oregon slender salamander). Herpetological Review 22: 22–23.

Leonard, W. P., H. A. Brown, L. L. C. Jones, K. R. McAllister, and R. M. Storm. 1993. Amphibians of Washington and Oregon. Seattle Audubon Society, Seattle, Washington. viii + 168 pp.

Marshall, D.B., M. Chilcote, H. Weeks. 1992. Sensitive Vertebrates of Oregon. Oregon Department of Fish and Wildlife. Portland, Oregon.

Miller, M.P. S.M. Haig, and R.S. Wagner. 2005. Conflicting patterns of genetic structure produced by nuclear and mitochondrial markers in the Oregon slender salamander (*Batrachoseps wrighti*): Implications for conservation efforts and species management. Conservation Genetics 6: 275–287.

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, *available at* http://www.natureserve.org/explorer (Accessed: January 13, 2012).

Nussbaum, R.A., E.D. Brodie, Jr., and R.M. Storm. 1983. Amphibians and Reptiles of the Pacific Northwest. University Press of Idaho, Moscow, Idaho. 332 pp.

Oregon Dept. of Fish and Wildlife. 2008. Sensitive Species List, *available at* http://www.dfw.state.or.us/wildlife/diversity/species/docs/SSL_by_taxon.pdf (last visited Jan. 13, 2012).

Petranka, J. W. 1998. Salamanders of the United States and Canada. Smithsonian Institution Press, Washington, D.C.

Soulé, M. 1987. Viable populations for conservation. Cambridge University Press, Great Britain. 189 pp.

Stebbins, R. C. 2003. A field guide to western reptiles and amphibians. Third edition. Houghton Mifflin Company, Boston.

Stebbins R.C., and N.W. Cohen. 1995. A Natural History of Amphibians. Princeton University Press, Princeton, NJ.

Tanner, W. 1953. Notes on the life history of *Plethopsis wrightii* Bishop. Herpetologica 9: 139–140.

U.S. Forest Service and U.S. Bureau of Land Management. 1994. Final Supplemental Environmental Impact Statement on Management of Habitat for Inter-successional and Old Growth Forest Related Species Within the Range of the Northern Spotted Owl. Portland, Oregon.

Vesely, D. G. 1999. Habitat selection by Oregon slender salmanders (*Batrachoseps wrighti*) in the western Oregon Cascades. Final report to Oregon Department of Fish and Wildlife.

Wagner, R.S, and S.M. Haig. 2000. Phylogeographic variation, genetic structure and conservation unit designation in forest-associated terrestrial salamanders: the Oregon slender salamander and the Larch Mountain salamander. Northwestern Naturalist 81(2): 90.

Walcott, P. 2010. Inventory for Oregon Slender Salamander, Batrachoseps wrightorum, on the Mt. Hood National Forest Report to the Interagency Special Status Sensitive Species Program (FY 2009), *available at* http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=16&ved=0CEIQFjAFOAo&url=http%3A%2F%2Fwww.fs.fed.us%2Fr6%2Fsfpnw%2Fissssp%2Fdocuments%2Finventories%2Finv-rpt-ha-bawr-mth-2009.pdf&ei=dUIQT8uwDoHagAf_jIWEBA&usg=AFQjCNGxsTJ0yVeHz0j1Qg227L5hZ-bBnQ (last visited Jan. 13, 2012).

Wright, S. 1931. Evolution in Mendelian populations. Genetics 16: 97-159.

Scientific Name:

Dicamptodon ensatus

Common Name:

California Giant Salamander

G Rank:

G3

IUCN Red List:

Near Threatened

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

California giant salamanders occur in the Pacific Coastal region around San Francisco Bay north to Mendocino, Sonoma, and Napa counties, and south to Santa Cruz County (Good 1989, Bury 2005). A disjunct population exists farther to the south in Monterey County (Anderson 1969, Nussbaum 1976, Petranka 1998). The overall current distribution is similar to the historic distribution, but numerous populations have been lost with the intense urbanization and habitat alteration characteristic of many parts of their range (Bury 2005).

Habitat:

Larvae of this species usually inhabit clear, cold streams, but are also found in mountain lakes and ponds (Hammerson and Bury 2004). Adults are found in humid forests under rocks and logs, for example, near mountain streams or rocky shores of mountain lakes (Stebbins 1985b). Aggregations of adults have been observed in wet areas underneath degraded culverts (Fellers et al. 2008). Eggs are usually laid in the headwaters of mountain streams, and breeding typically occurs in water-filled nest chambers under logs and rocks or in rock crevices (Hammerson and Bury 2004).

Biology:

Little is known about the biology of the terrestrial adult (AmphibiaWeb 2012). Nests of the California giant salamander have been found below cover objects (rocks and longs) submerged in running water (AmphibiaWeb 2012). Clutch size ranged from 70 - 100 eggs (Petranka 1998). California giant salamanders are probably similar to other species of *Dicamptodon* in many features (AmphibiaWeb 2012). Larval diet has not been studied, but presumably includes aquatic invertebrates and some aquatic vertebrates, as seen in *D. copei* (Nussbaum et al. 1983). Juveniles and adults forage above ground on rainy nights and can sometimes be found on rural roads (AmphibiaWeb 2012). Adult California giant salamanders have been reported to eat smaller California giant salamanders (AmphibiaWeb 2012). Birds and shrews may also prey on California giant salamanders, but they have to contend with a strong defensive bite (Petranka 1998). California giant salamanders are known to vocalize (Stebbins 1951). Life history characteristics are summarized by Bury (2005).

Population Status:

California giant salamanders qualify for endangered status because their small range is limited to an area with abundant human activity where populations are facing extirpation and significant declines due to threats such as urbanization and agriculture (see Bury 2005). The IUCN Red List ranks the species as Near Threatened but explains that it is close to qualifying for Vulnerable.

THREATS

Habitat alteration and destruction:

California giant salamanders have a small range that exists within an area of intense human activity, and as such, they are at threat from siltation of their stream habitats and urbanization (Petranka 1998). Habitat fragmentation and degradation due to land use changes, including agriculture and timbering also threatens the salamanders (NatureServe 2011). This species is susceptible to road mortality (California Roadkill Observation System 2012).

In the related Pacific giant salamander (*D. tenebrosus*), larvae is likely to be reduced in numbers where there has been clearcut logging (Corn and Bury 1989) or siltation from roads (Welsh and Ollivier 1998). Pogue (2008) found that streams impacted by sediment exhibited fewer surviving age classes and also significantly less biomass per square meter of pool bottom. Similarly, Kroll et al. (2008) found that giant salamander occupancy was positively associated with stand age, which suggests that intensive management of timber is not compatible with the species. And Johnston and Frid (2002) found that clearcut logging restricts the movements of terrestrial Pacific giant salamanders. The same likely holds true for the California giant salamander.

Overutilization:

Limited collection of the salamanders is allowed under California regulation (CCR § 5.05) but the species does not appear to be a major object of trade.

Inadequacy of existing regulatory mechanisms:

The California giant salamander lacks protected status in California (California Dept. of Fish and Game 2011). As such, the species is not protected from take and its habitat is vulnerable to destruction. The species occurs in some protected areas (Hammerson and Bury 2004), which protects the species from urbanization in those areas but does not prevent habitat destruction and degradation from threats such as logging and roadbuilding.

Other Factors:

The proximity of California giant salamanders to agricultural areas likely subjects them to toxic pesticides, which should be considered a threat.

References:

AmphibiaWeb: Information on amphibian biology and conservation. [web application]. 2012. Berkeley, California: AmphibiaWeb, *available at* http://amphibiaweb.org (last visited June 19, 2012).

Anderson, J.D. 1968. *Dicamptodon, D. ensatus*. Catalogue of American Amphibians and Reptiles. 76:1-2.

Bury, B. 2005. *Dicamptodon ensatus* - California Giant Salamander. In: M. Lannoo, editor. Amphibian Declines: The Conservation Status of United States Species. University of California Press, Berkeley.

California Dept. of Fish and Game. 2011. Special Animals List - January 2011, *available at* http://www.dfg.ca.gov/biogeodata/cnddb/pdfs/SPAnimals.pdf (last visited Jan. 11, 2012).

California Roadkill Observation System. 2012. Amphibians, *available at* http://www.wildlifecrossing.net/california/taxa (last visited Jan. 11, 2012).

Corn, P. S., and R. B. Bury. 1989. Logging in western Oregon: repsonses of headwater habitats and stream amphibians. Forest Ecology and Management 29:39-57.

Fellers, G.M., L.L. Wood, S. Carlisle, and D. Pratt. 2008. Ununual subterranean aggregations of the California giant salamander, Dicamptodon ensatus. Herpetological Conservation and Biology 5(1):149-154.

Good, D. A. 1989. Hybridization and cryptic species in *Dicamptodon* (Caudata: Dicamptodontidae). Evolution 43: 728-744.

Hammerson, G. and B. Bury 2004. Dicamptodon ensatus. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.2. <www.iucnredlist.org>. Downloaded on 13 January 2012.

Johnston, B. and L. Frid. 2002. Clearcut logging restricts the movements of terrestrial Pacific giant salamanders (*Dicamptodon tenebrosus* Good). Canadian Journal of Zoology, 2002, 80:(12) 2170-2177, 10.1139/z02-213.

Kroll, A.J., K. Risenhoover, T. McBride, E. Beach, B. J. Kernohan, J. Light, J. Bach. 2008 Factors influencing stream occupancy and detection probability parameters of stream-associated amphibians in commercial forests of Oregon and Washington, USA. Forest Ecology and Management 255: 3726–3735.

Kucera, T. 1997. Life History Account of California Giant Salamander, http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=58&ved=0CFYQFjAHODI&url=http%3A%2F%2Fnrm.dfg.ca.gov%2FFileHandler.ashx%3FDocumentVersionID%3D17590&ei=XGAQT_2FPMGLgwegm-3YCQ&usg=AFQjCNFsRI7aLK7x6LbjwtR1ynjjSlad9g (last visited Jan. 13, 2012).

Nanjappa, P. and Paulette M. Conrad 2012. State of the Union: Legal Authority Over the Use of Native Amphibians and Reptiles in the United States, *available at* http://www.fishwildlife.org/files/SOU_FULL-lo-res.pdf (last visited April 30, 2012).

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia. Available http://www.natureserve.org/explorer (Accessed: January 13, 2012).

Nussbaum, R.A. 1976. Geographic variation and systematics of salamanders of the genus *Dicamptodon strauch* (Ambysomatidae). Misc. Publ. Mus. Zool. Univ. Michigan 149:1-94.

Nussbaum, R. A., E.D. Brodie, Jr., and R.M. Storm. 1983. Amphibians and Reptiles of the Pacific Northwest. University of Idaho Press, Moscow, Idaho.

Petranka, J. W. 1998. Salamanders of the United States and Canada. Smithsonian Institution Press, Washington, D.C.

Pogue, S. 2008. Measuring the effects of increasing loads of fine sediment from timber harvest and road building on aquatic populations of Dicamptodon Tenebrosus (Pacific gian salamander) in California's redwoods. Master's Thesis, *available at* http://www.fire.ca.gov/CDFBOFDB/PDFS/Pogue2008_HSU_MS_Thesis_Final.pdf (last visited Jan. 13, 2012).

Stebbins, R.C. 1951. Amphibians of Western North America. University of California Press, Berkeley.

Stebbins, R. C. 1985. A field guide to western reptiles and amphibians. Second edition. Houghton Mifflin Company, Boston, Massachusetts. xiv + 336 pp.

Welsh, H. H., Jr., and L. M. Ollivier. 1998. Stream amphibians as indicators of ecosystem stress: a case study from California's redwoods. Ecological Applications 8: 1118-1132.

© 2007 Danté B Fenolio

Scientific Name:

Eurycea latitans

Common Name:

Cascade Caverns Salamander

G Rank:

G3

IUCN Red List:

Vulnerable

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

Chippindale et al. (2000) included in the *E. latitans* complex populations from Cascade Caverns, Bear Creek Spring, Cibolo Creek Spring, Kneedeep Cave Spring, Honey Creek Cave Spring, Less Ranch Spring, Cherry Creek Spring, Cloud Hollow Springs, and Rebecca Creek Spring, in Comal, Kerr, Kendall, and Hays counties, Texas. Chippindale et al. (1994) noted that further research may reveal that these populations comprise multiple species.

Habitat:

It can be found in springs and caves containing water in limestone karst (Hammerson and Chippendale 2004).

Biology and Taxonomy:

Chippindale (2000) and Chippindale et al. (2000) provisionally recognized the *E. latitans* complex, in which they included the population at the type locality, plus many others from the Edwards Plateau in Comal, Kendall, and eastern Kerr counties. If the species complex is not recognized by the FWS, then we petition for the species that is recognized at the type locality (e.g. Collins and Taggart 2009).

The Cascade Caverns salamander is completely aquatic and does not metamorphose (Hammerson and Chippendale 2004). Available information about life history characteristics is summarized by Chippendale (2005).

Population Status:

ESA protection for the Cascade Caverns salamander is warranted because it is highly vulnerable due to its restricted distribution and would probably be exterminated if the underground waterways in which it lives were flooded or polluted (Bury et al. 1980).

The *E. latitans* complex is known from about 10 locations (Chippindale et al. 2000). As with many cave-dwelling populations of Texas *Eurycea*, it is difficult to assess population sizes (Chippendale 2005). Its distribution appears to be limited and patchy (Chippendale 2005) and populations are likely to be declining (NatureServe 2011). It is considered vulnerable because of its small range and severely fragmented distribution, and there is a continuing decline in the quality of its cave spring habitat (Hammerson and Chippendale 2004).

<u>THREATS</u>

Habitat alteration and destruction:

Because this salamander respires via external gills and through its skin, it requires clean, clear flowing water, with a high dissolved oxygen content, for survival (NBII 2012). As such, threats to the salamander include declining water quantity and quality (NatureServe 2011).

A primary threat to the salamander is loss of springflows. Use of groundwater in the region decreases flow of water to the springs upon which the salamander depends. Texas is one of the fastest-growing states in the nation (Combs 2007). The Texas Data Center and the Office of the State Demographer project that the state's population will increase by 71.5 percent between 2000 and 2040, from 20.9 million to 35.8 million (Combs 2007). The 2040 projected population of 35.8 million is a 151 percent increase from the 1980 population of 14.2 million (Combs 2007). Human population increases will cause increasing demand on the groundwater. This situation would be compounded during drought (FWS 1996).

In addition, the salamander is threatened by declining water quality. Lower aquifer levels and springflows may also decrease water quality because of a decreased dilution ability (i.e., less water to dilute any pollutants in the system, resulting in higher pollutant concentrations) (FWS 1996). Lower water levels also increase temperature fluctuations. Due to its wide ranging influence on many different biotic and chemical factors (Armour 1991), water temperature is an important consideration (FWS 1996).

Moreover, the salamander is found in Hays and Comal counties, which are among the top five counties in central Texas for confirmed underground storage tank leaks. In 1988, Hays County had between 6 and 10 leaks, and Comal County had between 2 and 5 leaks (TWC 1989). The Texas Water Commission estimates that, on average, every leaking underground storage tank will leak about 500 gallons per year of contaminants before the leak is detected. These tanks are considered one of the most significant sources of groundwater contamination in the state (TWC 1989).

Surface runoff, particularly in urban areas, may impact the springs inhabited by the Cascade Caverns salamander. Stormwater runoff may include such things as pesticides and herbicides, fertilizers, soil eroded from construction activities, silt, suspended solids, garbage, hydrocarbon and inorganic/metal compounds from vehicles and machinery, household solvents and paints, and other urban runoff from point and non-point pollution sources (Urban Drainage and Flood Control District 1992). Other species, such as invertebrate prey species on which they feed, could also be affected by runoff of herbicides, pesticides, and other non-point source pollutants (FWS 1996).

A report produced by the Edward's Underground Water District (EUWD 1993) summarizes information on increasing development in the Edwards Aquifer recharge zone and the effects of these activities in Comal and Hays (and other) counties. The report concluded that the cumulative impact of pollution resulting from urbanization over the Edwards recharge zone was not being adequately addressed, and that degradation of Edwards Aquifer water could be imminent.

Such threats to the salamander's aquatic habitats are of great concern. Because of its specialized habitat and restricted distribution, the salamander would be extirpated if the underground waterway in which it lives were flooded or polluted (Bury et al. 1980).

Inadequacy of existing regulatory mechanisms:

The salamander is listed as Threatened by the state of Texas (Chippendale 2005). This state status prohibits take but does nothing to prevent water quantity loss and quality degradation, which are the biggest threats to the salamander.

The range of the Cascades Cavern salamander includes Guadeloupe River State Park. Yet the salamander cannot be adequately protected unless activities throughout the watersheds it inhabits are regulated. Federal listing would require analysis of many activities that threatened water quality, such as pesticide use.

References:

Armour, C.L. 1991. Guidance forevaluating and recommending temperature regimes to protect fish. Instream Flow Information Paper. USFWS Biological Report. 90(22) l.3pp.

Bury, R. B., C. K. Dodd, Jr., and G. M. Fellers. 1980. Conservation of the Amphibia of the United States: a review. U.S. Fish and Wildlife Service, Washington, D.C., Resource Publication 134. 34 pp.

Chippindale, P.T. 2000. Species boundaries and species diversity in the central Texas hemidactyliine plethodontid salamanders, genus Eurycea. The Biology of Plethodontid Salamanders. Bruce, R., Houck, L. and Jaeger, R.,editor. 149-160. Kluwer Academic/Plenum Publishing. New York.

Chippendale, P. 2005. *Eurycea latitans* - Cascade Caverns Salamander. In: M. Lannoo, editor. Amphibian Declines: The Conservation Status of United States Species. University of California Press, Berkeley.

Chippindale, P. T., D. M. Hillis, and A. H. Price. 1994. Relationships, status, and distribution of central Texas hemidactyliine plethodontid salamanders (*Eurycea* and *Typhlomolge*). Final Section 6 Report, July 1994. 21 pp. + 1 Fig.

Chippindale, P. T., A. H. Price, J. J. Wiens, and D. M. Hillis. 2000. Phylogenetic relationships and systematic revision of central Texas hemidactyliine plethodontid salamanders. Herpetological Monographs 14: 1-80.

Collins, J. T., and T. W. Taggart. 2009. Standard common and current scientific names for North American amphibians, turtles, reptiles, and crocodilians. Sixth edition. The Center for North American Herpetology, Lawrance, Kansas. iv + 44 pp.

Combs, S. 2007. Texas in Focus: A Statewide View of Opportunities, *available at* http://www.window.state.tx.us/specialrpt/tif/population.html (last visited June 21, 2012).

Edwards Underground Water District. 1993. Urban Development on the Edwards Aquifer Recharge Zone. Report 93-09. 40 pp.

Hammerson, G. and P. Chippindale. 2004. *Eurycea latitans*. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.2. <www.iucnredlist.org>. Downloaded on 13 January 2012.

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, *available at* http://www.natureserve.org/explorer (Accessed: January 13, 2012).

National Biological Information Infrastructure (NBII). 2012. Threatened Species Spotlight: Cascade Caverns Salamander.

Texas Water Commission (TWC). 1989. Ground-water quality of Texas-an overview of natural and man-affected conditions. Austin, Texas. 197 pp.

Urban Drainage and Flood Control District. 1992. Urban Storm Drainage Criteria Manual. Vol. 3. Best Management Practices, Stormwater Quality, Urban Drainage and Flood Control District, Denver, Colorado. U.S. Army Corps of Engineers.

U.S. Fish and Wildlife Service. 1996. San Marcos and Comal Spring and Associated Aquatic Ecosystems (Revised) Recovery Plan, *available at*

http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=1&ved=0CFAQFjAA&url=http%3A%2F%2Fwww.fws.gov%2Fecos%2Fajax%2Fdocs%2Frecovery_plan%2F960214.pdf&ei=-DTjT4K0Aurc0QHf3ojNAw&usg=AFQjCNGAfv-gY8jVl7Ih6JmTliCMSKvW2Q&sig2=vE3RLRWXt-AhGS39tDef5w (last visited June 21, 2012).

Scientific Name:

Hydromantes brunus

Common Name:

Limestone Salamander

G Rank:

G1

IUCN Red List:

Vulnerable

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

The limestone salamander has an extremely restricted range, which directly limits its viability. The species occurs only along a short section (about 16–17 km) of the Merced River, from the vicinity of the type locality on the main highway to Yosemite National Park west to a region known as Hell Hollow, and a short distance up the North Fork of the Merced River, between elevations of 365–760 m (Wake and Pappenfuss 2005).

Habitat:

Wake and Pappenfuss (2005) describe its habitat as follows:

> There is a general association with limestone, but salamanders have been found on the surface under both slate slabs and irregularly shaped pieces of limestone. They have been found in small areas of moss-covered or barren talus (Tordoff, 1980), as well as in rock crevices and even in abandoned mine tunnels. The

vegetation in the region where salamanders are found is mainly chaparral, with a scattering of xeric-adapted trees such as gray pine (*Pinus sabiniana*), and with California laurel (*Umbellularia californica*) in more mesic sites. In general, the habitat appears marginal for salamander occupancy, and yet four species are sympatric in this seemingly hostile environment. The species has been found on relatively level ground, but it is more typically encountered on steep slopes, where individuals use their tail to assist in locomotion (Gorman, 1954). Temperatures at which this species has been taken range from 10.0 ˚C–14.0 ˚C (mean 11.4 ˚C; Feder et al., 1982).

Biology:

Egg deposition has not been observed, nor has development, but its close relatives undergo direct development (AmphibiaWeb 2012). Ovarian eggs were enlarged (4.6 mm in diameter) in the holotype, collected in late February (Gorman 1954). Eggs are probably laid in late spring and develop over the summer (AmphibiaWeb 2012). For additional life history information, see Wake and Pappenfuss (2005) and Basey and Morey (2000).

Population Status:

Federal protection is warranted for the limestone salamander because it is known from only a few scattered localities in one county in California (NatureServe 2011), where its remaining habitat is under siege from threats like mining and roadbuilding. There have been no significant new discoveries of populations for many years (NatureServe 2011). The species was a candidate for federal protection until the FWS eliminated the C2 category, but it currently receives no federal protection under the ESA.

<u>THREATS</u>

Habitat alteration and destruction:

A proposed gold mine operation in Hell Hollow poses the most serious threat (NatureServe 2011). Other potential threats include highway construction and quarrying for limestone (California Department of Fish and Game 1990). It disappears from cleared areas (NatureServe 2011), and is therefore susceptible to any activities that clear the land (see, e.g., Steinhart 1990, Jennings and Hayes 1994). The type locality of this species is along the main access route to Yosemite National Park, and any widening of the road would destroy habitat, which is already highly restricted (Basey and Morey 2000, Wake and Pappenfuss 2005). Another significant threat is construction of dams, which has caused some local flooding (AmphibiaWeb 2012).

Inadequacy of existing regulatory mechanisms:

Limestone Salamander Ecological Reserve (LSER) protects 120 acres of habitat. The Bureau of Land Management has designated an additional 1,600 acres as the Limestone Salamander Area of Critical Environmental Concern (LSACEC) (encompasses both confirmed and potential habitat) (California Dept. of Fish and Game 1990). Yet these areas protect just a fraction of the species occupied habitat. With such a highly restricted species, all occupied habitat must be protected.

The species is listed as Threatened by the California Department of Fish and Game, under the State of California Endangered Species Act. Such status offers protection from collection and some protection to habitat but does not prevent all activities that threaten to degrade or destroy habitat. Federal ESA protection would add additional protections on federal lands through section 7 consultation and critical habitat designation and would allow for use of section 6 funds to recover the species.

References:

AmphibiaWeb: Information on amphibian biology and conservation. [web application]. 2012. Berkeley, California: AmphibiaWeb, *available at* http://amphibiaweb.org (last visited June 19, 2012).

Basey, H. and S. Morey. 2000. Life History Account for Limestone Salamander, *available at* http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=45&ved=0CDwQFjAEOCg&url=http%3A%2F%2Fnrm.dfg.ca.gov%2FFileHandler.ashx%3FDocumentVersionID%3D17820&ei=0pUQT7_yLqaFsgKBlPj1Aw&usg=AFQjCNFoO4VF3BPYi1tMThANVZTuM9WmJg (last visited Jan. 13, 2012).

California Department of Fish and Game (CDF&G). 1990. 1989 annual report on the status of California's state listed threatened and endangered plants and animals. 188 pp.

Feder, M.E., J.F. Lynch, H.B. Shaffer, and D.B. Wake. 1982. Field body temperatures of tropical and temperate zone salamanders. Smithsonian Herpetological Information Service, Number 52, Smithsonian Institution, Washington, D.C.

Gorman, J. 1954. A new species of salamander from central California. Herpetologica 10: 153–158.

Hammerson, G. and D. Wake 2004. Hydromantes brunus. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.2. <www.iucnredlist.org>. Downloaded on 13 January 2012.

Jennings, M. R., and M. P. Hayes. 1994. Amphibian and reptile species of special concern in California. Rancho Cordova: California Department of Fish and Game, Inland Fisheries Division.

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, *available at* http://www.natureserve.org/explorer (Accessed: January 13, 2012).

Steinhart, P. 1990. California's wild heritage: Threatened and endangered animals in the Golden State. San Francisco: California Department of Fish and Game, California Academy of Sciences, and Sierra Club Books.

Tordoff, W., III. 1980. Report of study of limestone salamander on the Merced River. U.S. Bureau of Land Management, Folsom District Office, Contract Number CA-040-CTO-09, Folsom, California.

Wake, D. and T.J. Pappenfuss. 2005. Hydromantes brunus - Limestone Salamander. In: M. Lannoo, editor. Amphibian Declines: The Conservation Status of United States Species. University of California Press, Berkeley.

Scientific Name:

Hydromantes shastae

Common Name:

Shasta Salamander

G Rank:

G1

IUCN Red List:

Vulnerable

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

The Shasta salamander is vulnerable to extinction because of its small range, where its habitat faces numerous threats. NatureServe 2011 describes the range as follows:

> This range is restricted to a small area in northern California, in the headwaters of the Shasta Reservoir drainage, Shasta County, California; occurrences exist northward to the Dutch Creek and Centipede Creek drainages and are distributed primarily along the Pit River, Squaw Creek, and McCloud River arms of upper Shasta Reservoir, in areas west of the southwest part of the reservoir, and in the area south of the reservoir east of Interstate Highway 5; known sites are east and west of the Sacramento River and both north and south of the Pit River arm of the reservoir (Bury et al. 1980, Lindstrand 2000, Stebbins 2003, Nauman and Olson 2004, Wake and Papenfuss 2005). The entire range is less than 35 kilometers in

greatest dimension (Wake and Papenfuss 2005). Elevational range is approximately 1,000-3,200 feet (300-975 meters).

There are presently 64 known occurrence sites within the Shasta salamander's highly restricted range (California Department of Fish and Game 2010).

Habitat:

The Shasta salamander is known from three of four major limestone belts in Shasta County (Kennet Formation, McCloud Limestone, and Hosselkus Limestone (Lewendal 1995). Although the Shasta salamander occurs only in a limestone-dominated region, it may not be as strongly tied to limestone substrates as previously thought Lindstrand 1999, 2000, 2002; Nauman and Olson 2002). It has been found in association with a volcanic rock outcrop (Papenfuss and Cross 1980) and in areas with scattered rocks and colluviums pockets but no extensive talus slopes or other rock outcrops (Lindstrand 2000)

Indeed, some detections have been several miles from the nearest significant limestone formations where cover objects included logs, leaf litter, and other types of layered rock (e.g. Nauman et al. 2003). Some of these discoveries occurred in a variety of forested habitats, on all aspects, with a range in total canopy cover from 0 to 75 percent (Lindstrand 2002). The salamanders are also found in small to large caves (Wake and Pappenfuss 2005).

The salamanders are mostly found in oak-sabine (digger) pine woodland, also in Douglas-fir woodland and ponderosa/Jeffrey pine-oak at higher elevations (Bury et al. 1980, Stebbins 2003). Eggs have been found attached to rock surfaces in crevices in damp limestone caves (NatureServe 2011).

Biology:

The Shasta salamanders are excellent climbers, their webbed toes allow them to climb sheer, slippery rock surfaces (Gorman and Camp 1953). On steep slopes, to aid climbing, they curl the tail tip forward and place it on the ground as the hind foot is lifted, a behavior unique to North America's western salamanders (AmphibiaWeb 2012). The salamander lays and broods eggs in moist caves during summer (Stebbins 1985), and it crawls out in open at night during rains of fall, winter, and spring (AmphibiaWeb 2012). Egg clutch size is 9-12 (Hansen and Papenfuss 1994). Hatchlings reach 15-17 mm in snout-vent length (AmphibiaWeb 2012). Juveniles are 22-24 mm in snout-vent length and resemble adults (Gorman 1956). Like all *Hydromantes* species, the Shasta salamander can shoot its tongue out its mouth 1/3 the length of its body to catch prey (Stebbins 1985). For additional information on life history, see Wake and Pappenfuss (2005), Olson (2005), and Mooney (2010).

Population Status:

The Shasta salamander qualifies for endangered status because it is extremely rare across its very limited range and it is threatened by proposed destruction of its habitat by raising of Shasta Dam. A survey of 40 sites in the Shasta Reservoir area yielded only three individuals in 360 personhours (compared to 352 *Aneides flavipunctatus* and 348 *Ensatina*) (Nauman and Olson 2004; see also Nauman et al. 2003). The species was a candidate for federal protection until the FWS eliminated the C2 category, but it currently receives no federal protection under the ESA.

THREATS

Habitat alteration and destruction:

The construction of Shasta Dam created Shasta Lake, which submerged key salamander habitats and caused population declines (Hansen and Papenfuss 1994, Wake and Papenfuss 2005, NatureServe 2011). Lake Shasta is used for hydropower and recreation and plans are in place to raise its levels by up to 200 feet (Bingham 2011). The raising of Lake Shasta is one of the biggest potential threats to the salamander (California Department of Fish and Game 1990). A rise in water level would flood hundreds of acres of salamander habitat and thereby threaten the genetic diversity and overall survival of the species (California Dept. of Fish and Game 1987, Hansen and Papenfuss 1994, NatureServe 2011). Specifically, the Backbone Ridge habitat would be greatly reduced in size with additional habitat loss at Dekkas Ranch and Hirz Mountain area (California Dept. of Fish and Game 1987).

Because some populations are small and isolated, they are vulnerable to destruction by relatively small amounts of habitat disturbance (Wake and Papenfuss 2005). Road building and quarrying for limestome have negatively impacted the species in the past (Hansen and Papenfuss 1994, Wake and Papenfuss 2005) and remain as potential threats (California Department of Fish and Game 1990, Hansen and Papenfuss 1994). Indeed, there are several old limestone quarries within the range of the salamander that would threaten the salamander if mining resumed (California Dept. of Fish and Game 1987).

Timber management and human recreational activities are additional threats (Hansen and Papenfuss 1994, NatureServe 2011). For example, fuels-reduction projects can harm the salamanders and have been planned in their habitat (Clayton and Morey 2003). The threat of recreational activity is expected to grow with the rising human population in the Redding area (California Dept. of Fish and Game 1987).

Inadequacy of existing regulatory mechanisms:

The Shasta salamander is listed as Threatened under the State of California Endangered Species Act (California Dept. of Fish and Game 2011). With this state listing, take is prohibited but some illegal collection likely occurs. The state has developed guidelines for addressing the needs of Shasta salamanders during consultation (California Dept. of Fish and Game 2008) but this does not prevent all actions that destroy salamander habitat. Federal ESA protection would add additional protections on federal lands through section 7 consultation and critical habitat designation and would allow for use of section 6 funds to recover the species.

Many of the known populations occur on U.S. Forest Service and Bureau of Land Management land, and it is a U.S. Forest Service and Bureau of Land Management Sensitive Species

(California Dept. of Fish and Game 2011). This status does not prevent activities that could harm the salamander as it only requires analysis of impacts.

The Shasta-Trinity National Forest developed a management plan for the salamander in 1979 (U.S. Forest Service 1979, California Dept. of Fish and Game 1987). This plan provides that rocky outcrops, caves, and ravines with limestone deposits must be protected from disturbance and recommends that vegetative ground cover, downed logs, and surface rocks be left undisturbed within 300 feet of limestone deposits (*id.*). This management plan has some benefits for the salamander but only for those populations within the national forest that are associated with limestone deposits. Other populations remain vulnerable to habitat disturbance, and with a species as vulnerable as the Shasta salamander, all populations must be protected.

Some of the known populations are on private land (California Dept. of Fish and Game 1987). Human activities could easily lead to their destruction by relatively small amounts of habitat disturbance (Wake and Pappenfuss 2005).

Other factors:

The distribution of the Shasta salamander is discontinuous because the limestone areas it inhabits are separated and the salamander apparently does not migrate between localities (AmphibiaWeb 2012). As such, as with all isolated populations, the Shasta salamander is susceptible to extirpations due to stochastic events, human impacts, and environmental factors (Soulé 1987, Begone et al. 1990). Lack of gene flow may cause loss of genetic variability due to random genetic drift (Wright 1931) and inbreeding depression may occur (Franklin 1980). Once these populations are extirpated, their isolation precludes genetic interchange and recolonization of habitat.

References:

AmphibiaWeb: Information on amphibian biology and conservation. [web application]. 2012. Berkeley, California: AmphibiaWeb, *available at* http://amphibiaweb.org (last visited June 19, 2012).

Begone, M., J. L. Harper, and C. R. Townsend. 1990. Ecology: Individuals, populations, and communities. Blackwell Scientific Publications.

Bingham, R. 2011. Phylogeography of Hydromantes shastae: Implications for Management, *available at* http://www.youtube.com/watch?v=uD3dXSJjw18 (last visited Jan. 16, 2012).

Bury, R.B., C.K. Dodd Jr., and G.M. Fellers. 1980. Conservation of the Amphibia of the United States. U.S. Fish and Wildlife Service Resource Publ. 134. 34 pp.

California Dept. of Fish and Game. 1987. Five Year Status Report, *available at* http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=2&ved=0CFAQFjAB&url=http%3A%2F%2Fnrm.dfg.ca.gov%2FFileHandler.ashx%3FDocumentVersionID%3D79529

&ei=42q2T5rtJY6y8ASExv28Cg&usg=AFQjCNGPyALGXz2U8NP7zPSAQTmSxaTE5w&sig2=sKYCkVLcs4SutUqqhJVvgA (last visited May 18, 2012).

California Department of Fish and Game (CDF&G). 1990. 1989 annual report on the status of California's state listed threatened and endangered plants and animals. 188 pp.

California Department of Fish and Game. 2008. Shasta Salamander Web Site. Interior Timberland Planning, Habitat Conservation Program, Northern Region. Redding, California, *available at* https://r1.dfg.ca.gov/portal/itp/Fisheries/ShastaSalamander/tabid/671/Default.aspx

California Department of Fish and Game. 2010. California Natural Diversity Database [Data file]. Retrieved from http://www.dfg.ca.gov/biogeodata/cnddb/

California Dept. of Fish and Game. 2011. Special Animals List - January 2011, *available at* http://www.dfg.ca.gov/biogeodata/cnddb/pdfs/SPAnimals.pdf (last visited Jan. 11, 2012).

California Natural Diversity Database. 2008. Full condensed report for selected elements – *Hydromantes shastae*. 34 occurrences.

Clayton, D. and S. Morey. 2003. The national fire plan and amphibians: proposed management recommendations for two terrestrial salamanders when planning for fuels reduction projects. Northwestern Naturalist 84(2): 97.

Franklin, I. R. 1980. Evolutionary changes in small populations. In Conservation Biology: An evolutionary-ecological perspective (ed. a. B. A. W. M. E. Soule). pp. 135-149.

Gorman, J. 1956. 'Reproduction in plethodontid salamanders of the genus Hydromantes. Herpetologica 12: 249-259.

Gorman, J. and C.L. Camp. 1953. A new cave species of salamander of the genus Hydromantes from California, with notes on habits and habitats.' Copeia 1953: 39-43.

Hammerson, G., D. Wake, and A. Herman 2004. Hydromantes shastae. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.2. <www.iucnredlist.org>. Downloaded on 16 January 2012.

Hansen, R. W., and T. J. Papenfuss. 1994. Shasta Salamander. Pages 256-257 in C. G. Thelander, editor. Life on the Edge: A Guide to California's Endangered Natural Resources. Volume I: Wildlife. Biosystems Books, Santa Cruz, California.

Inventory of Survey and Manage Species. 2002. Report for selected species – Hydromantes shastae. 88 records.

Lewendal, P. 1995. Habitat-use by the Shasta salamander, *Hydromantes shastae*. Report to the California Department of Fish and Game, Region 1, Redding, CA. Shasta Land Management Consultants, Redding, CA. 32 pp.

Lindstrand III, L. 2000. Discovery of Shasta salamanders in atypical habitat. California Fish and Game 86(4): 259-261.

Lindstrand III, L. 2002. Green Mountain Shasta Salamander Purposive Surveys, North State Resources, Inc. Shasta Trinity National Forest, Redding, California, 13 pages.

Lindstrand III, L. 2008. Shape file of known locations for selected species – *Hydromantes shastae*. 56 records.

Mooney, T.L. 2010. Predicting hydromantes shastae occurrences in Shasta County, California University of Southern California Master's Thesis. 45 pp.

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, *available at* http://www.natureserve.org/explorer (Accessed: January 16, 2012).

Nauman, R.H., and D.H. Olson. 2002. Surveys for Shasta salamander and other amphibians in Shasta County, California.

Nauman, R., L. Lindstrand, and D. Olson. 2003. Recent discoveries of Shasta salamanders (Hydromantes shastae) in unusual habitats: relative rarity and conservation implications. Northwestern Naturalist 84(2): 108.

Olson, D.H. 2005. Shasta salamander, Hydromantes shastae Gorman and Camp. In: Jones, L.L.C.; Leonard, W.P.; Olson, D.H., eds. Amphibians of the Pacific Northwest. Seattle, WA: Seattle Audubon Society.

Olson, D.H. and P.C. Lewendall. 1999. Survey Protocol for the Shasta Salamander, in, Survey Protocols for Amphibians under the Survey and Manage Provision of the Northwest Forest Plan, Version 3.0, pages 79-124.

Papenfuss, T. and P. Brouha. 1979. The status of the Shasta Salamander (Hydromantes shastae). Shasta-Trinity National Forest Comprehensive Species Management Plan and a Species Status Report: 1-36.

Pappenfuss, T. and D. Cross. 1980. The status of the Shasta salamander Hydromantes shastae in the Mountain Gate and Cedar Creek areas of Shasta County. Unpublished report to the USDA Forest Service. 9 p. Available from- US Forest Service, Shasta-Trinity National Fcorest, 2400 Washington Avenue, Redding, CA 96001.

Soulé, M. 1987. Viable populations for conservation. Cambridge University Press, Great Britain. 189 pp.

Stebbins, R. C. 1985. A Field Guide to Western Reptiles and Amphibians. Houghton Mifflin, Boston.

Stebbins, R. C. 2003. A field guide to western reptiles and amphibians. Third edition. Houghton Mifflin Company, Boston.

UC Berkeley Museum specimens, 2002, Query of selected species – *Hydromantes shastae*. 35 records.

U.S. Forest Service. 1979. Comprehensive Species Management Plan, Shasta Salamander, *Hydromantes shastae*.

Wake, D. and T.J. Pappenfuss. 2005. *Hydromantes shastae* - Shasta Salamander. In: M. Lannoo, editor. Amphibian Declines: The Conservation Status of United States Species. University of California Press, Berkeley.

Wright, S. 1931. Evolution in Mendelian populations. Genetics 16: 97-159.

Scientific Name:

Plethodon amplus

Common Name:

Blue Ridge Gray-cheeked Salamander

G Rank:

G1

IUCN Red List:

Vulnerable

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

The Blue Ridge gray-cheeked salamander is vulnerable to extinction because of its very small range, which only includes the Blue Ridge Mountains in Buncombe, Rutherford, and Henderson counties, North Carolina. Its elevational range is at least 1,109-1,116 m (Highton and Peabody 2000).

Habitat:

Blue Ridge gray-cheeked salamanders are reported from crevices in metamorphic rock and from the forest floor (Adler and Dennis 1962, Rubin 1969). It can be found in mesic forest, often under leaf-litter, logs, or mossy rocks (Hammerson and Beamer 2004).

Biology and Taxonomy:

This species was recently separated from *Plethodon jordani* (Highton and Peabody 2000, Weisrock and Larson 2006). Little is known about the salamander's life history. It is nocturnal and secretive, and all members of the genus *Plethodon* produce noxious skin secretions (Brodie, 1977). For more information on life history characteristics, see Beamer and Lannoo (2005).

Population Status:

The Blue Ridge gray-cheeked salamander deserves federal protection because it is rare within its small range where the species faces the ongoing threat of habitat destruction. It is known from fewer than five locations (Hammerson and Beamer 2004). Some local extirpations apparently have occurred (Beamer and Lannoo 2005).

<u>THREATS</u>

Habitat alteration and destruction:

Clearcutting has been reported as strongly depleting local populations of the *P. jordani* complex (Petranka et al. 1993, 1999). Beamer and Lannoo (2005) stated that habitat destruction and modification have caused local extirpations of the Blue Ridge gray-cheeked salamander.

Overutilization:

The salamanders can be legally taken and up to 24 can be possessed without a license or permit (Nanjappa and Conrad 2012), but the species does not appear to be a major object of trade.

Inadequacy of existing regulatory mechanisms:

Blue Ridge gray-cheeked salamanders are not protected in North Carolina, the only state within their range (Beamer and Lannoo 2005). Among members of the *P. jordani* complex, Blue Ridge gray-cheeked salamanders have one of the smallest distributions. Within this range there are few federal and state properties that contain suitable habitat for these salamanders. It probably receives some protection on lands in the Pisgah National Forest where the forest cover is maintained (Hammerson and Beamer 2004) but is harmed by logging, especially clearcutting. Conservation activities that promote mature closed-canopy forests should benefit this species (Beamer and Lannoo 2005). Given its small range, federal protection is needed to ensure that all remaining localities are protected from habitat destruction.

Other factors:

Small, isolated populations, such as those of the Blue Ridge gray-cheeked salamander, are more susceptible to extirpations due to stochastic events, human impacts, and environmental factors (Soulé 1987, Begone et al. 1990). Lack of gene flow is likely to cause loss of genetic variability due to random genetic drift (Wright 1931) and inbreeding depression may occur (Franklin 1980).

Once these populations are extirpated, their isolation precludes genetic interchange and recolonization of habitat.

References:

Adler, K.K. and D.M. Dennis. 1962. *Plethodon longicrus*, a new salamander (Amphibia: Plethodontidae) from North Carolina. Special Publications of the Ohio Herpetological Society 4:1–14.

Beamer, D. A., and M. J. Lannoo. *Plethodon amplus* Highton and Peabody. 2000. Blue Ridge gray-cheeked salamander. Pages 789-790 in M. Lannoo, editor. Amphibian declines: the conservation status of United States species. University of California Press, Berkeley.

Begone, M., J. L. Harper, and C. R. Townsend. 1990. Ecology: Individuals, populations, and communities. Blackwell Scientific Publications.

Franklin, I. R. 1980. Evolutionary changes in small populations. In Conservation Biology: An evolutionary-ecological perspective (ed. a. B. A. W. M. E. Soule). pp. 135-149.

Hammerson, G. and David Beamer 2004. Plethodon amplus. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.2. <www.iucnredlist.org>. Downloaded on 31 January 2012.

Highton, R., and R. B. Peabody. 2000. Geographic protein variation and speciation in salamanders of the *Plethodon jordani* and *Plethodon glutinosus* complexes in the southern Appalachian Mountains with the description of four new species. Pages 31-93 in R. C. Bruce, R. G. Jaeger, and L. D. Houck, editors. The biology of plethodontid salamanders. Kluwer Academic/Plenum Publishers, New York. xiii + 485 pp.

Nanjappa, P. and Paulette M. Conrad 2012. State of the Union: Legal Authority Over the Use of Native Amphibians and Reptiles in the United States, *available at* http://www.fishwildlife.org/files/SOU_FULL-lo-res.pdf (last visited April 30, 2012).

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia. Available http://www.natureserve.org/explorer (Accessed: January 31, 2012).

Petranka, J. W. 1999. Recovery of salamanders after clearcutting in the southern Appalachians: a critique of Ash's estimates. Conservation Biology 13: 203-205.

Petranka, J. W., M. E. Eldridge, and K. E. Haley. 1993. Effects of timber harvesting on southern Appalachian salamanders. Conservation Biology 7(2): 363-370.

Rubin, D. 1969. Food habits of *Plethodon longicrus* Adler and Dennis. Herpetologica 25:102–105.

Soulé, M. 1987. Viable populations for conservation. Cambridge University Press, Great Britain. 189 pp.

Weisrock, D.W. and A. Larson. 2006. Testing hypotheses of speciation in the Plethodon jordani species complex with allozymes and mitochondrial DNA sequences. Biological Journal of the Linnean Society. Linnean Society of London 89(1): 25-51.

Wright, S. 1931. Evolution in Mendelian populations. Genetics 16: 97-159.

Photo by Michael Graziano

Scientific Name:

Plethodon caddoensis

Common Name:

Caddo Mountain Salamander

G Rank:

G2

IUCN Red List:

Near Threatened

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

The Caddo Mountain salamander is limited solely to the Caddo Mountains, Ouachita Mountains region, southwestern Arkansas (Conant and Collins 1991, Petranka 1998), at elevations of 900-2150 ft (274-655 m). The species inhabits a geologically distinct area of the Ouachita Mountains called the Arkansas Novaculite uplift (Pope and Pope 1951, Trauth and Wilhide 1999).

Habitat:

Caddo Mountain salamanders are found at higher elevations of mixed deciduous, north-facing wooded slopes (Pope and Pope 1951, Plummer 1982, Petranka 1998). Rocks, logs, and other forest debris are common cover objects. Moisture conditions at the surface appear to influence activity greatly (Plummer 1982), with salamanders retreating to lower levels of talus to escape hot and dry conditions (Spotila 1972). Caves and abandoned mines are also utilized (Saugey et al. 1985, Heath et al. 1986).

Biology:

Females of the species have been observed attending egg clutches (Anthony 2005). Eggs have been observed as early as 9 June and hatching as late as 5 November (Anthony 2005). Adult Caddo Mountain salamanders recognize and respond to odors of conspecifics (Anthony 1993), and they defend areas in laboratory chambers (Thurow 1976, Anthony 1995). See Anthony (2005) for a summary of available natural history information.

Population Status:

The Caddo Mountain salamander qualifies for endangered status because of its very small range where the salamanders's few remaining occurrences face threats from harmful logging and mining. As of 2004, the Arkansas Natural Heritage Commission had recorded about 20 occurrences, but most of the occurrences did not have recent information (Anthony 2005). The IUCN Red List ranks the species as Near Threatened but explains that it is close to qualifying for Vulnerable (Hammerson 2004). The species was a candidate for federal protection until the FWS eliminated the C2 category, but it currently receives no federal protection under the ESA.

THREATS

Habitat alteration and destruction:

The Caddo Mountain salamander is threatened by habitat destruction and fragmentation resulting from silviculture and mining (NatureServe 2011). Timber management activities and conversion of land to pine plantations most likely reduced suitable habitat for this species (Trauth and Wilhide 1999, Warriner 2002). Habitat degradation and loss continue to threaten the species, including within the Ouachita National Forest because of logging.

Overutilization:

Collection of the salamanders is allowed, and it is vulnerable to scientific overcollection (Huston 2009).

Disease or predation:

The trombiculid mite, *Hannemania dunni*, is known to infest 12 salamander species across a wide phylogenetic and geographic range (Dunn and Heinze 1933, Rankin 1937, Duncan and Highton 1979). Salamanders endemic to the Ouachita Mountains of Oklahoma and Arkansas, including the Caddo Mountain salamander, show high prevalence of chiggers relative to other salamander species (Rankin 1937, Pope and Pope 1951, Winter et al. 1986). Damage to nasolabial grooves by these parasites may reduce the ability of males to forage efficiently, to function as territory holders, and to find mates (Anthony et al. 1994).

Inadequacy of existing regulatory mechanisms:

Most populations are in the Ouachita National Forest and the salamander is a Region 8 sensitive species (U.S. Forest Service 2000). On the National Forest, the species is vulnerable to habitat destruction due to silvicultural practices, as the sensitive species designation only requires analysis of, but does not prevent, harmful activities. Federal protection would lead to additional conservation measures aimed at preservation of habitat in its limited range. For example, hardwood buffers left around the margins of talus would help maintain viable populations by providing leaf litter and shading (Petranka 1998).

Caddo Mountain salamanders are considered a Species of Special Concern in Arkansas (Arkansas Game and Fish Commission 2004). Such status reflects the rarity of the species but does nothing to provide habitat protection. Collection is legal and up to six can be personally possessed per day (Nanjappa and Conrad 2012).

A Memorandum of Understanding on Caddo Mountain Salamander Conservation was developed between the U.S. Forest Service and FWS after a status review was conducted by the FWS (Huston 2009). As a result, gates were placed at sensitive mining sites (USDA Forest Service 1988). Such measures are beneficial in protecting habitats at these discrete locations but additional measures are needed to protect the species' forested habitats.

Other factors:

Climatic changes that result in decreased relative humidity (e.g., increased temperature and/or decreased precipitation) could be severe for Caddo Mountain salamander given its small body size and thus high rate of dehydration (Shepard and Burbrink 2011).

References:

Anthony, C.D. 1993. Recognition of conspecific odors by *Plethodon caddoensis* and *P. ouachitae*. Copeia 1993:1028–1033.

Anthony, C.D. 2005. *Plethodon caddoensis* - Caddo Mountain Salamander. In M. Lannoo, editor. Amphibian declines: the conservation status of United States species. University of California Press, Berkeley.

Anthony, C.D., Joseph R. Mendelson III, and Richard R. Simons. 1994. Differential Parasitism by Sex on Plethodontid Salamanders and Histological Evidence forStructural Damage to the Nasolabial Groove. American Midland Naturalist 132(2): 302-307.

Arkansas Game and Fish Commission. 2004. Arkansas Endangered, Threatened, And Species of Special Concern, *available at* http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=85&ved=0CD4QFjAEOFA&url=http%3A%2F%2Fwww.agfc.com%2Flicenses%2FDocuments%2F1-9-04%2520Special%2520Concern%2520Species.pdf&ei=Ek0oT5LsBs6EtgeG8sHnBA&usg=AFQjCNEPh4Cy7Hg2hzNXeXY6U3_PF2QB4g (last visited Jan. 31, 2012).

Conant, R. and J. T. Collins. 1991. A field guide to reptiles and amphibians: eastern and central North America. Third edition. Houghton Mifflin Co., Boston, Massachusetts. 450 pp.

Duncan, R. and R. Highton. 1979. Genetic relationships of the eastern large Plethodon of the Ouachi- ta Mountains. Copeia 1979: 96-110.

Dunn, E. R. and A. A. Heinze. 1933. A new salamander from the Ouachita Mountains. Copeia, 1933: 121-122.

Flawn, P.T., A. Goldstein Jr., P.B. King, and C.E. Weaver. 1961. The Ouachita System. Bureau of Economic Geology, University of Texas, Austin.

Foti, T.L., and G.A. Bukenhofer. 1998. A description of the sections and subsections of the Interior Highlands of Arkansas and Oklahoma. J. Arkansas Acad. Sci. 52: 53–62.

Hammerson, G. 2004. Plethodon caddoensis. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.2. <www.iucnredlist.org>. Downloaded on 31 January 2012.

Hatcher Jr., R.D., W.A. Thomas, and G.W. Viele. 1989. The Appalachian–Ouachita Orogen in the United States. The Geology of North America, vol. F-2. Geological Society of America, Boulder, Colorado.

Heath, D.R., D.A. Saugey and G.A. Heidt. 1986. Abandoned mine fauna of the Ouachita Mountains, Arkansas: vertebrate taxa. Proceedings of the Arkansas Academy of Science 40: 33–36.

Huston, R. 2009. Biological Evaluation for Two Mile Watershed Project, *available at* http://www.fs.fed.us/outernet/r8/ouachita/projects/two_mile/Two_Mile_Biological_Evaluation.doc (last visited Jan. 31, 2012).

Nanjappa, P. and Paulette M. Conrad 2012. State of the Union: Legal Authority Over the Use of Native Amphibians and Reptiles in the United States, *available at* http://www.fishwildlife.org/files/SOU_FULL-lo-res.pdf (last visited April 30, 2012).

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, *available at* http://www.natureserve.org/explorer. (Accessed: January 31, 2012).

Petranka, J. W. 1998. Salamanders of the United States and Canada. Smithsonian Institution Press, Washington, D.C.

Plummer, M.V. 1982. The status of the Caddo Mountain and Fourche Mountain salamanders (*Plethodon caddoensis*, *P. fourchensis*) in Arkansas. Report to the Arkansas Natural Heritage Commission, Little Rock, Arkansas.

Pope, C. H. and S. H. Pope. 1951. A study of the salamander Plethodon ouachitae and the description of an allied form. Bull. Chic. Acad. Sci. 9: 129-152.

Rankin, J. S. 1937. An ecological study of parasites of some North American salamanders. Ecol. Monogr. 7: 169-269.

Saugey, D.A., G.A. Heidt and D.R. Heath. 1985. Summer use of abandoned mines by the Caddo Mountain salamander, *Plethodon caddoensis* (Plethodontidae) in Arkansas. Southwestern Naturalist 30: 318–319.

Shepard, D.B. and F. T. Burbrink. 2011. Local-scale environmental variation generates highly divergent lineages associated with stream drainages in a terrestrial salamander, Plethodon caddoensis. Molecular Phylogenetics and Evolution 59: 399–411.

Spotila, J.R. 1972. Role of temperature and water in the ecology of lungless salamanders. Ecological Monographs 42: 95–125.

Thurow, G.R. 1976. Aggression and competition in eastern Plethodon (Amphibia, Urodela, Plethodontidae). Journal of Herpetology 10: 277–291.

Trauth, Stanley E., and J.D. Wilhide. 1999. Status of three plethodontid salamanders (genus Plethodon) from the Ouachita National Forest of southwestern Arkansas. J. Arkansas Acad. Sci. 53: 125-137.

U.S. Forest Service. 1988. Management of Amphibians, Reptiles, and Small Mammals in North America Symposium. General Technical Report RM-166. Fort Collins, CO. 65pp.

U.S. Forest Service. 2000. List of Sensitive Species, *available at* www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsm8_036237.pdf (last visited Jan.31, 2012).

Trauth, S.E., and J.D. Wilhide. 1999. Status of three plethodontid salamanders (Genus Plethodon) from the Ouachita National Forest of southwestern Arkansas. J. Arkansas Acad. Sci. 53: 125–137.

Warriner, M. D. 2002. Arkansas Natural Heritage Commission, Rare Amphibian Fact Sheet, Caddo Mountain Salamander - *Plethodon caddoensis.*

Winter, D. A., W. M. Zawada and A. A. Johnson. 1986. Comparison of the symbiotic fauna of the family Plethodontidae in the Ouachita Mountains of western Arkansas. Proc. Ark. Acad. Sci. 40: 82-85.

Scientific Name:

Plethodon cheoah

Common Name:

Cheoah Bald Salamander

G Rank:

G2

IUCN Red List:

Vulnerable

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

One of the biggest threats to the Cheoah Bald salamander is its very small range where the salamander continues to lose habitat due to logging. This salamander is known from an area of approximately 15 square kilometers in Cheoah Bald, Graham and Swain counties, North Carolina; the elevational range is at least 975-1,524 meters (Highton and Peabody 2000, Hammerson and Beamer 2004).

Habitat:

It is found in mesic forest, often under leaf-litter, logs, or mossy rocks. It is a terrestrial breeder. Most of the species range includes second growth forest (Hammerson and Beamer 2004).

Biology:

Very little is known about the life history of these salamanders. They likely make vertical migrations, moving from the forest floor to underground sites with the onset of seasonally related cold or dry conditions, then back up to the forest floor with the return of favorable surface conditions (Beamer and Lannoo 2005). It is likely that the females brood, as has been observed in other *Plethodon* species (Beamer and Lannoo 2005). Available information is summarized by Beamer and Lannoo (2005).

Population Status:

The Cheoah Bald salamander is endangered because it is experiencing declines from habitat destruction across its very small range. Beamer and Lannoo (2005) explain that declines have been observed.

<u>THREATS</u>

Habitat alteration and destruction:

Clearcutting is likely to strongly deplete local populations of the *P. jordani* complex (Petranka et al. 1993), such as the Cheoah Bald salamander. The time required for recovery of salamander populations impacted by clearcutting is debatable but is at least a few decades (Ash 1997, Petranka 1999, Ash and Pollock 1999).

Inadequacy of existing regulatory mechanisms:

Part of the range of this species is within the Nantahala Game Lands, which offer some measure of protection because the forest is typically left intact (Hammerson and Beamer 2004). But these protected lands comprise only a fraction of the range. Given that the salamander's range is so restricted, all occupied habitat needs to be protected to ensure species viability. Conservation activities that promote mature closed-canopy forests should benefit this species (Beamer and Lannoo 2005).

Cheoah Bald salamanders are not protected in North Carolina, the only state within their range (Beamer and Lannoo 2005). The North Carolina Natural Heritage Program considers the species "significantly rare" (North Carolina Natural Heritage Program 2008). Collection is legal and up to 24 can be collected without a permit or license (Nanjappa and Conrad 2012). Federal protection would prohibit take and ensure that all essential habitats are adequately protected.

Other factors:

This salamander is found in an area of approximately 15 square kilometers in two counties in North Carolina (Hammerson and Beamer 2004). Small, isolated populations like this are more susceptible to extirpations due to stochastic events, human impacts, and environmental factors (Soulé 1987, Begone et al. 1990). Lack of gene flow is likely to cause loss of genetic variability due to random genetic drift (Wright 1931) and inbreeding depression may occur (Franklin 1980).

Once these populations are extirpated, their isolation precludes genetic interchange and recolonization of habitat.

References:

Ash, A.N. 1997. Disappearance and return of plethodontid salamanders to clearcut plots in the southern Blue Ridge Mountains. Conservation Biology: 983-989.

Ash, A.N., and Pollock, K.H. 1999. Clearcuts, salamanders, and field studies. Conservation Biology: 206-208.

Beamer, D. A., and M. J. Lannoo. *Plethodon cheoah* - Cheoah Bald Salamander. In M. Lannoo, editor. Amphibian declines: the conservation status of United States species. University of California Press, Berkeley.

Begone, M., J. L. Harper, and C. R. Townsend. 1990. Ecology: Individuals, populations, and communities. Blackwell Scientific Publications.

Franklin, I. R. 1980. Evolutionary changes in small populations. In Conservation Biology: An evolutionary-ecological perspective (ed. a. B. A. W. M. E. Soule). pp. 135-149.

Hammerson, G. and D. Beamer 2004. Plethodon cheoah. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.2. <www.iucnredlist.org>. Downloaded on 31 January 2012.

Highton, R. and Peabody, R.B. 2000. Geographic protein variation and speciation in salamanders of the *Plethodon jordani* and *Plethodon glutinosus* complexes in the southern Appalachian Mountains with the description of four new species. In: Bruce, R.C., Jaeger, R.G. and Houck, L.D. (eds), The Biology of Plethodontid Salamanders, pp. 31-93. Kluwer Academic / Plenum Publishers, New York.

Nanjappa, P. and Paulette M. Conrad 2012. State of the Union: Legal Authority Over the Use of Native Amphibians and Reptiles in the United States, *available at* http://www.fishwildlife.org/files/SOU_FULL-lo-res.pdf (last visited April 30, 2012).

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, *available at* http://www.natureserve.org/explorer. (Accessed: January 31, 2012).

North Carolina Natural Heritage Program. 2008. Natural Heritage Program List of the Rare Animal Species of North Carolina, *available at* http://www.ncnhp.org/Images/2008-animal-book-complete.pdf (last visited Jan. 31, 2012).

Petranka, J.W. 1999. Recovery of salamanders after clearcutting in the southern Appalachians: a critique of Ash's estimates. Conservation Biology: 203-205.

Soulé, M. 1987. Viable populations for conservation. Cambridge University Press, Great Britain. 189 pp.

Wright, S. 1931. Evolution in Mendelian populations. Genetics 16: 97-159.

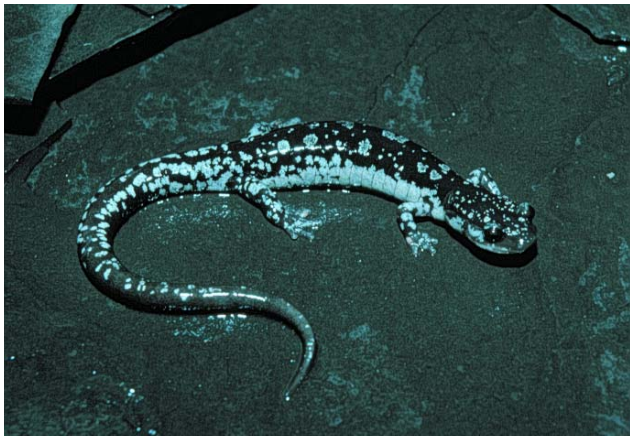


Photo by Stanley Trauth

Scientific Name:

Plethodon fourchensis

Common Name:

Fourche Mountain Salamander

G Rank:

G2

IUCN Red List:

Vulnerable

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

The small range of the Fourche Mountain salamander includes Fourche Mountain and Irons Fork Mountian, Polk and Scott counties, west-central Arkansas, at elevations of 1,560-2,400 feet (503-730 meters) (Conant and Collins 1991, Trauth et al. 2004, Anthony 2005).

Habitat:

The Fourche Mountain salamander inhabits moist, shady, hardwood and mixed deciduous pine forests, and it lives under logs, forest litter, and rocks. It is a terrestrial breeder with direct development (Hammerson 2004).

Biology and Taxonomy:

Fourche Mountain salamanders were previously considered a variant of Rich Mountain salamanders (*P. ouachitae*; the Buck Knob variant; Blair and Lindsay, 1965), and originally described by Duncan and Highton (1979). Recent molecular phylogenetic studies, however, have supported the recognition of all three currently recognized species in the *P. ouachitae* complex (Kozak et al. 2006, 2009; Shepard and Burbrink 2008, 2009, 2011; Wiens et al. 2006). Shepard et al. (2011) found that recognition of Fourche Mountain salamander as suggested by genetic studies is was well supported by their morphological studies. They further suggested that lineages within Fourche Mountain salamander likely warrant recognition as species (Wiens and Penkrot 2002). The species is recognized by Crother et al. (2008) and Collins and Taggart (2009). If any species within the Fourche Mountain salamander are identified as separate species, we hereby petition for those species.

Very little information on the life history characteristics of the Fourche Mountain salamanders is available (Anthony 2005). Individuals move underground in late May but may return to the surface during periods of rainfall and/or cool weather (Anthony 2005). By mid September, adults can again be found under cover objects at the surface (Anthony 2005). And they probably hibernate from mid November to late March (Anthony 2005). For available information on life history characteristics, see Anthony (2005).

Population Status:

Fourche Mountain salamanders qualify for endangered status because of documented declines across their restricted range where only a handful of occurrences exist and remain threatened by poor forest management practices. The salamanders are listed as Vulnerable because they are known from fewer than five locations (Hammerson 2004). Lohoefener and Jones (1991) noted a reduction in numbers at several localities. The species was a candidate for federal protection until the FWS eliminated the C2 category, but it currently receives no federal protection under the ESA.

<u>THREATS</u>

Habitat alteration and destruction:

Habitat loss and degradation are threats. All known locations of this salamander are within the Ouachita National Forest, which affords some level of protection (Warriner 2002). However, the Fourche Mountain salamander has probably been impacted by deforestation in the past, and local populations are presumably still impacted by deforestation and poor timber management practices (Hammerson 2004). Its restricted distribution renders it particularly susceptible to any new threats (Hammerson 2004).

Disease or predation:

Winter et al. (1986) found trematodes, cestodes, nematodes, protozoans, and mites on six adults. Intradermal mites of the genus *Hannemania* are common on the feet and toes and appear as raised red pustules. Toe loss and damage have been attributed to these mites (Conant and Collins 1991). Duncan and Highton (1979) found that 89–92 percent of individuals from three populations had mite infestations. It is unclear whether mite infestations is contributing to observed population declines.

Inadequacy of existing regulatory mechanisms:

Its entire known range is within the Ouachita National Forest (Warriner 2002), and it is considered a sensitive species for the Forest (U.S. Forest Service 2002). Sensitive species designation does not prevent harmful actions but only requires analysis of their impacts. As such, some harmful actions on the national forest continue to occur, and forest management practices (e.g., timber harvesting, controlled burning, etc.) likely need to be modified to ensure its longterm conservation (Trauth and Wilhide 1999, Shepard et al. 2011).

Other factors:

The Fourche Mountain salamander is found in just two counties in west-central Arkansas (Conant and Collins 1991, Trauth et al. 2004, Anthony 2005). Small, isolated populations such as these are more susceptible to extirpations due to stochastic events, human impacts, and environmental factors (Soulé 1987, Begone et al. 1990). Lack of gene flow is likely to cause loss of genetic variability due to random genetic drift (Wright 1931) and inbreeding depression may occur (Franklin 1980). Once these populations are extirpated, their isolation precludes genetic interchange and recolonization of habitat.

In addition, climate change could impact this species, which is restricted to high-elevation, mesic forest on five montane isolates in the Ouachita Mountains and therefore unable to shift its elevational range upslope as the climate warms. Scientists have found evidence *Plethodon* mountaintop endemics are likely range-restricted due to climactic factors (Kozak and Wiens 2006, Arif et al. 2007).

References:

Anthony, C. 2005. *Plethodon fourchensis* - Fourche Mountain Salamander. In M. Lannoo, editor. Amphibian declines: the conservation status of United States species. University of California Press, Berkeley.

Arif, S., Dean C. Adams and Jill A. Wicknick. 2007. Bioclimatic modelling, morphology, and behavior reveal alternative mechanisms regulating the distributions of two parapatric salamander species. Evolutionary Ecology Research 9: 843–854.

Begone, M., J. L. Harper, and C. R. Townsend. 1990. Ecology: Individuals, populations, and communities. Blackwell Scientific Publications.

Collins, J. T., and T. W. Taggart. 2009. Standard common and current scientific names for North American amphibians, turtles, reptiles, and crocodilians. Sixth edition. The Center for North American Herpetology, Lawrance, Kansas. iv + 44 pp.

Conant, R. and J. T. Collins. 1991. A field guide to reptiles and amphibians: eastern and central North America. Third edition. Houghton Mifflin Co., Boston, Massachusetts. 450 pp.

Crother, B. I. (editor). 2008. Scientific and standard English names of amphibians and reptiles of North America north of Mexico, with comments regarding confidence in our understanding. Sixth edition. Society for the Study of Amphibians and Reptiles Herpetological Circular 37:1-84.

Duncan, R., and R. Highton. 1979. Genetic relationships of the eastern large *Plethodon* of the Ouachita Mountains. Copeia 1979: 95-110.

Franklin, I. R. 1980. Evolutionary changes in small populations. In Conservation Biology: An evolutionary-ecological perspective (ed. a. B. A. W. M. E. Soule). pp. 135-149.

Hammerson, G. 2004. Plethodon fourchensis. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.2. <www.iucnredlist.org>. Downloaded on 01 February 2012.

Kozak, K. H., and J. J. Wiens. 2006. Does niche conservatism drive speciation? A case study in North American salamanders. Evolution 60: 2604–2621.

Kozak, K. H., D. W. Weisrock, and A. Larson. 2006. Rapid lineage accumulation in a nonadaptive radiation: Phylogenetic analysis of diversification rates in eastern North American woodland salamanders (Plethodontidae: Plethodon). Proceedings of the Royal Society BBiological Sciences 273: 539–546.

Kozak, K. H., R. W. Mendyk, and J. J. Wiens 2009. Can parallel diversification occur in sympatry? Repeated patterns of body-size evolution in coexisting clades of North American salamanders. Evolution 63: 1769–1784.

Lohoefener, R., and R. L. Jones. 1991. Status survey of *Plethodon fourchensis* and *P. caddoensis*. U.S. Fish and Wildlife Service, Endangered Species Office, Status Survey Report, Jackson, Mississippi.

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, *available at* http://www.natureserve.org/explorer (Accessed: February 1, 2012).

Shepard, D. B., and F. T. Burbrink. 2008. Lineage diversification and historical demography of a sky island salamander, Plethodon ouachitae, from the Interior Highlands. Molecular Ecology 17: 5315–5335.

Shepard, D. B., and F. T. Burbrink. 2009. Phylogeographic and demographic effects of Pleistocene climatic fluctuations in a montane salamander, Plethodon fourchensis. Molecular Ecology 18: 2243–2262.

Shepard, D. B., and F. T. Burbrink. 2011. Local-scale environmental variation generates highly divergent lineages associated with stream drainages in a terrestrial salamander, Plethodon caddoensis. Molecular Phylogenetics and Evolution 59: 399–411.

Shepard, D.B., K.J. Irwin, and F.T. Burbrink. 2011. Morphological Differentiation in Ouachita Mountain Endemic Salamanders. Herpetologica 67(4): 355-368.

Soulé, M. 1987. Viable populations for conservation. Cambridge University Press, Great Britain. 189 pp.

Trauth, Stanley E., and J.D. Wilhide. 1999. Status of three plethodontid salamanders (genus Plethodon) from the Ouachita National Forest of southwestern Arkansas. J. Arkansas Acad. Sci. 53: 125-137.

Trauth, S. E., H. W. Robison, and M. V. Plummer. 2004. The amphibians and reptiles of Arkansas. University of Arkansas Press.

U.S. Forest Service. 2002. Amended Biological Evaluation for Timber Harvest, *available at* www.fs.fed.us/outernet/oonf/plan/nepa/robck_ea_app4.html (last visited Feb. 1, 2012).

Warriner, M. D. 2002. Arkansas Natural Heritage Commission, Rare Amphibian Fact Sheet, Fourche Mountain Salamander - *Plethodon fourchensis.*

Wiends, J. J., and T. A. Penkrot. 2002. Delimiting species using DNA and morphological variation and discordant species limits in spiny lizards (Sceloporus). Systematic Biology 51: 69–91.

Wiens, J. J., T. N. Engstrom, and P. T. Chippindale. 2006. Rapid diversification, incomplete isolation, and the ‘‘speciation clock’’ in North American salamanders (Genus: Plethodon): Testing the hybrid swarm hypothesis of rapid radiation. Evolution 60:2585–2603.

Winter, D.A., W.M. Zawada and A.A. Johnson. 1986. Comparison of the symbiotic fauna of the family Plethodontidae in the Ouachita Mountains of western Arkansas. Proceedings of the Arkansas Academy of Science 40:82–85.

Wright, S. 1931. Evolution in Mendelian populations. Genetics 16: 97-159.

© 2004 Henk Wallays

Scientific Name:

Plethodon hubrichti

Common Name:

Peaks of Otter Salamander

G Rank:

G2

IUCN Red List:

Vulnerable

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

Peaks of Otter salamanders are endemic to the Blue Ridge Mountains, restricted to an approximately 19-km length of the mountain chain in Bedford and Botetourt counties, Virginia (Highton 1972, 1986; Mitchell and Reay 1999). This species has one of the most restricted ranges of any salamander in the United States (VI Dept. of Game and Inland Fisheries 2012). With such a restricted range, the salamander is vulnerable to extinction from ongoing threats to its habitat.

Habitat:

Peaks of Otter salamanders are found in mature hardwood forest, mainly on north-facing slopes and in caves in the Peaks of Otter region of west-central Virginia (Hammerson and Mitchell 2004). They are also found in shaded ravines and rhododendron thickets. They are generally found above 845 m (Petranka 1998). They can be found primarily under downed logs and rocks, and among wet leaf litter, in middle to late successional stages of oak-maple woodland (Bury et al. 1980, Mitchell 1991). They often climb into vegetation, especially ferns, in June-September (Kramer et al. 1993).

Biology:

Mating occurs in September and October when pairs of males and females may be found under the same log or rock (Wicknick 1995). Females lay eggs in June (Pague 1989), and hatching occurs in August and September (Mitchell and Wicknick 2005). Mitchell and Wicknick (2005) summarize available life history information.

Population Status:

The Peaks of Otter salamander qualifies for endangered species protection because of its extremely restricted range, which makes it vulnerable to extinction due to disturbance of its habitats from logging and other threats. The species declined due to agricultural practices in the area in the 1800s and early 1900s (Mitchell and Wicknick 2005). Current populations are harmed by timber operations in parts of its range in the Jefferson National Forest (Mitchell and Wicknick 2005, Kramer et al. 1993). The species was a candidate for federal protection until the FWS eliminated the C2 category, but it currently receives no federal protection under the ESA.

THREATS

Habitat alteration and destruction:

Any activities that disrupt the forest canopy and associated understory vegetation likely have an adverse impact on local populations of Peaks of Otter salamanders (Mitchell 1991, Kramer et al. 1993). These activities range from timber harvesting practices (Mitchell 1996), such as clear cutting, to impending forest defoliation due to the gypsy moth (Kramer et al. 1993), as well as recreational development (Mitchell 1991). Firewood collection is also a threat in some areas (NatureServe 2011).

The Peaks of Otter salamander is highly susceptible to the threat of vegetation disruption from timber harvest because it is endemic to a small portion of the Blue Ridge Mountains in Virginia, where much of its range lies within a high timber producing area owned by the U.S. Forest Service (Mitchell et al. 1996). Because of very low dispersal rates, intensive timbering and habitat fragmentation is likely highly detrimental to this species (Kramer et al. 1993, Petranka 1998). As explained below, it is well documented that timber harvest eliminates the Peaks of Otter salamander from some sites and causes population declines and fragmentation. The species could be driven to extinction if habitat destruction continues to cause population declines and fragmentation across its small range.

Mitchell et al. (1996) determined that clearcut sites consistently supported fewer individuals of the Peaks of Otter salamander than mature forest or stands that had been thinned (shelterwood sites). Both types of timbering reduced diet quality for the salamander (Mitchell et al. 1996). The researchers explain that individuals not directly impacted by the immediate logging operation are probably subjected to stresses associated with reduced or altered prey resources and changes in the physical characteristics of the soil/litter ecosystem (Mitchell et al. 1996). Peaks of Otter salamander populations are sometimes completely eliminated from a site by timber operations of clearcutting and shelterwood cutting, and population declines are well documented. They are reduced 45-47 percent by clearcutting and 10-66 percent by shelterwood cutting, as compared to populations in adjacent mature sites (Mitchell et al. 1996).

In another study, numbers of salamanders were stable over a three-year period in mature hardwood and shelterwood stands, but declined substantially after clearcutting (Sattler and Reichenbach 1998). In the clearcut site, 30 percent of the initial population remained after treatment; many adults and juveniles either likely emigrated or died. Similarly, Knapp et al. (2003) found that all treatments with canopy removal had significantly fewer salamanders than the control treatment, but salamander abundances on alternative treatments with canopy removal did not differ significantly from salamander abundances on the clearcuts.

More recently, Reichenbach and Sattler (2007) found that Peaks of Otter salamander populations in timbered areas are usually lower, and sometimes absent, when compared to untimbered areas. The means at clearcut sites declined 41 percent during the first year posttimbering and then declined over the next three years to a low of 75 percent below pretimbering means (Reichenbach and Sattler 2007). The means stabilized at 45 percent below pre-timbering means for the remainder of the study. Immediately after timbering 41 percent of the salamanders moved from transects established at the edge of clearcuts to reference transects that were 3–9 m away. Clearcuts had less canopy closure and dead leaf cover than reference and shelterwood cuts, which likely degraded habitat for salamanders (Reichenbach and Sattler 2007).

To be sure, impacts of forest harvest on terrestrial salamanders have been well documented. Across 16 research projects, control stands had about 4.3 times more captures of salamanders than clearcut stands (deMaynadier and Hunter 1995). Timbering, especially clearcuts, increases temperature and decreases moisture on the forest floor (Covington 1981, Ash 1995).

In clearcut stands, recovery of salamanders has been estimated from 20 to 70 years in the Appalachians based on chronosequence and observational studies (Petranka et al. 1993, Ash 1997). More than 80 years is likely necessary for the negative effects of logging roads on terrestrial salamanders to dissipate, indicating that forest disturbances have long-lasting effects on salamanders (Semlitsch et al. 2007). As explained below, logging continues within the range of the salamander.

Overutilization:

The species faces high collection pressure at some sites (NatureServe 2011), although collection is prohibited on most federal lands within its range (Nanjappa and Conrad 2012).

Inadequacy of existing regulatory mechanisms:

The Forest Service lists them as a Sensitive Species (Mitchell and Wicknick 2005). Sensitive species designation does not prevent activities that are likely to harm the salamander – it only requires analysis of the impacts.

Mitchell and Wicknick (2005) explain that the U.S. Forest Service and the U.S. National Park Service have a conservation plan for the species (George Washington and Jefferson National Forests 1997):

> [The plan] specifies the entire range of the Peaks of Otter salamander as a Special Biological Area with a primary conservation area that allows no logging activities and a secondary conservation area that allows logging to take place under certain restrictions. These restrictions include the use of the shelterwood technique only, a 50 ft^2 basal area minimum, no cutting of remaining hardwood trees for 15 yr, location of the timber sale area away from previous sale areas, and seeding of the logged area with downed woody debris. In addition, cutting cannot occur during peak salamander surface activity time and no more than 100 ac may be harvested per year.

While this conservation plan provides important benefits to the species, Hammerson and Mitchell (2004) explain that "the management of these lands [Jefferson National Forest and Blue Ridge Parkway] is unlikely compatible with the conservation of this species." Indeed, given the extremely small range of this species, any forest management activity that removes canopy cover is a threat (Mitchell et al. 1996). While the conservation plan restricts harmful management activities in some areas, it fails restrict all harvest across the entire range and is therefore inadequate. In addition, the conservation plan cannot protect the species from the threat posed by climate change, which is discussed below. Federal protection would result in the designation of critical habitat and development of a recovery plan, which would be the best way to address the threats and ensure the salamander's viability.

Peaks of Otter salamanders are recognized as a Species of Concern by the Virginia Department of Game and Inland Fisheries. This status reflects its restricted range but does not provide any habitat protection. Salamanders are protected from commercial collection in Virginia but no permit is required to collect native amphibians and reptiles for personal use with a daily bag limit of five animals (Nanjappa and Conrad 2012).

Other factors:

Isolation

Timbering operations that occur within the salamander's range eliminates animals from many of the cut areas (Pough et al. 1987, Ash 1988) and likely also creates barriers between populations (Kramer et al. 1993). Recolonization after forest recovery, and the reestablishment of contact between separated populations, is likely a slow process due to limited movements of this salamander (Kramer et al. 1993).

In addition, the Blue Ridge Parkway, logging roads, and other alterations to the forest habitat likely produce similar impassible barriers, thus causing fragmentation of the population. Effectively, salamander populations become fragmented and genetically isolated. Loss of genetic variability associated with genetic drift could then occur thereby decreasing the long-term probability of survival for this species (Soule 1983). As such, isolation is a threat to the species.

Climate Change

The long-term persistence of the species is likely threatened by climate change and incursion by the more common red-backed salamanders (Marsh undated). Climate change is a threat because Peaks of Otter salamanders are confined to a single ridgetop and are therefore unable to shift their elevational range upslope as the climate warms. Scientists have found evidence that *Plethodon* mountaintop endemics are likely range-restricted due to climactic factors (Kozak and Wiens 2006, Arif et al. 2007). Red-backed salamanders are successful at lower elevations and are very common below the current elevational range of the Peaks of Otter salamander (Marsh undated). When Peaks of Otter salamanders move out of an area, even temporarily, red-backed salamanders are likely to move in and compete with Peaks of Otter salamanders for food and shelter sites (Wicknick 1995).

References:

Arif, S., Dean C. Adams and Jill A. Wicknick. 2007. Bioclimatic modelling, morphology, and behavior reveal alternative mechanisms regulating the distributions of two parapatric salamander species. Evolutionary Ecology Research 9: 843–854.

Ash, A. 1988. Disappearance of salamanders from clearcut plots. Journal of the Elisha Mitchell Scientific Society 104: 116–122.

Ash, A.N. 1995. Effects of clear-cutting on litter parameters in the Southern Blue Ridge Mountains. Castanea 60: 89-97.

Ash, A.N. 1997. Disappearance and return of plethodontid salamanders to clearcut plots in the southern Blue Ridge Mountains. Conservation Biology 11: 983–989.

Buhlmann, K.A., C.A. Pague, J.C. Mitchell and R.B. Glasgow. 1988. Forestry operations and terrestrial salamanders: techniques in a study of the Cow Knob salamander, *Plethodon punctatus*. Pp. 38–49. *In* Szaro, R.C., K.E. Severson and D.R. Patton (Eds.), Management of Amphibians, Reptiles, and Mammals in North America. U.S.D.A. Forest Service, General Technical Report, RM-166, Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado.

Bury, R.B., C.K. Dodd, Jr., and G.M. Fellers. 1980. Conservation of the Amphibia of the United States: a review. Resource Publication: 1-34.

Covington, W.W. 1981. Changes in forest floor organic matter and nutrient content following clear cutting in northern hardwoods. Ecology 62: 41-48.

de Maynadier, P.G., Hunter Jr., M.L., 1995. The relationship between forest management and amphibian ecology: a review of the North American literature. Environmental Reviews 3: 230–261.

George Washington and Jefferson National Forests. 1997. Habitat conservation assessment for the Peaks of Otter salamander (*Plethodon hubrichti*). U.S. Fish and Wildlife Service, Unpublished report. 28 p.

Hammerson, G., and J. Mitchell 2004. *Plethodon hubrichti*. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.2. <www.iucnredlist.org>. Downloaded on 01 February 2012.

Highton, R. 1972. Distributional interactions among eastern North American salamanders of the genus *Plethodon*. Pp. 139–188. *In* Holt, P.C. (Ed.), The Distributional History of the Biota of the Southern Appalachians. Part III: Vertebrates. Research Division Monograph, Virginia Polytechnic Institute and State University, Blacksburg, Virginia.

Highton, R. 1986. *Plethodon hubrichti*. Catalogue of American Amphibians and Reptiles: 1-2.

Knapp, R.A., Preisler, H.K., Matthews, K.R., Jellison, R., 2003. Developing probabilistic models to predict amphibian site occupancy in a patchy landscape. Ecological Applications 13: 1069–1082.

Kozak, K. H., and J. J. Wiens. 2006. Does niche conservatism drive speciation? A case study in North American salamanders. Evolution 60: 2604–2621.

Kramer, P., Reichenbach, N., Hayslett, M. and Sattler, P. 1993. Population dynamics and conservation of the Peaks of Otter salamander, *Plethodon hubrichti*. Journal of Herpetology: 431-435.

Marsh, D. Undated. Field Herpetology goes Environmental: Proposal for an ACS Environmental Curriculum Development Award, *available at* www.colleges.org/enviro/proposals/WLU_Marsh_proposal.pdf (last visted July 2, 2012).

Mitchell, J.C. 1991. Amphibians and reptiles. In: Terwilliger, K. (ed.), Virginia's Endangered Species: Proceedings of a Symposium, pp. 411-476. McDonald and Woodward Publishing Company, Blacksburg, Virginia.

Mitchell, Joseph C. 1996. Effects of timber harvesting on peaks of otter salamander (*Plethodon hubrichti*) populations. Virginia Journal of Science 47(2): 153.

Mitchell, J.C. and J.A. Wicknick. 2005. *Plethodon hubrichti* - Peaks of Otter Salamander. In M. Lannoo, editor. Amphibian declines: the conservation status of United States species. University of California Press, Berkeley.

Mitchell, J. C., J. A. Wicknick, and C. D. Anthony. 1996. Effects of timber harvesting practices on Peaks of Otter salamander (*Plethodon hubrichti*) populations. Amphibian and Reptile Conservation 1(1): 15-19.

Mitchell, J.C. and R.K. Reay. 1999. Atlas of amphibians and reptiles in Virginia. Special Publication Number 1, Wildlife Diversity Division, Virginia Department of Game and Inland Fisheries, Richmond, Virginia.

Nanjappa, P. and Paulette M. Conrad 2012. State of the Union: Legal Authority Over the Use of Native Amphibians and Reptiles in the United States, *available at* http://www.fishwildlife.org/files/SOU_FULL-lo-res.pdf (last visited April 30, 2012).

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia. Available http://www.natureserve.org/explorer. (Accessed: February 1, 2012).

Pague, C.A. 1989. King of the mountain. Virginia Wildlife 50: 8–13.

Petranka, J.W. 1998. Salamanders of the United States and Canada. Smithsonian Institution Press, Washington, D.C.

Petranka, J.W., M.E. Eldridge and K.E. Haley. 1993. Effects of timber harvesting on southern Appalachian salamanders. Conservation Biology 7: 363–370.

Pough, H.F., E.M. Smith, D.H. Rhodes and A. Collazo. 1987. The abundance of salamanders in forest stands with different histories of disturbance. Forest Ecology and Management 20: 1–9.

Reichenbach, N. and P. Sattler. 2007. The Effects of Timbering on Plethodon hubrichti over 12 Years. Journal of Herpetology, *available at* http://works.bepress.com/norman_reichenbach/12 (last visited Feb. 2, 2012).

Reichenbach, Norman and Sattler, P. 1998. The Effects of Timbering on *Plethodon hubrichti*: Short Term Effects. Faculty Publications and Presentations. Paper 21, *available at* http://digitalcommons.liberty.edu/bio_chem_fac_pubs/21

Sattler, P., and N. Reichenbach. 1998. The effects of timbering on *Plethodon hubrichti*: short-term effects. Journal of Herpetology 32: 399-404.

Semlitsch, R.D., T.J. Ryan, K. Hamed, M. Chatfield, B. Drehman, N. Pekarek, M. Spath, and A. Watland. 2007. Salamander abundance along road edges and within abandoned logging roads in Appalachian forests. Conservation Biology 21: 159–167.

Soulé, M. 1987. Viable populations for conservation. Cambridge University Press, Great Britain. 189 pp.

Virginia Dept. of Game and Inland Fisheries. 2012. Peaks of Otter salamander (Plethodon hubrichti), *available at* http://www.dgif.virginia.gov/wildlife/information/?s=020039 (last visited Feb. 2, 2012).

Wicknick, J.A. 1995. Interspecific competition and territoriality between a widespread species of salamander and a species with limited range. Ph.D. dissertation. University of Southwestern Louisiana, Lafayette, Louisiana.


© 2010 Dr. Stephen G. Tilley

Scientific Name:

Plethodon meridianus

Common Name:

South Mountain Gray-cheeked Salamander

G Rank:

G3

IUCN Red List:

Vulnerable

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

The viability of the South Mountain gray-cheeked salamander is limited by its very small distribution, where it faces ongoing threats of habitat destruction. This species can be found in the South Mountains, in Burke, Cleveland, and Rutherford counties, in the Piedmont Province of North Carolina, USA; its elevational range is at least 543-823 m above sea level (Highton and Peabody 2000).

Habitat:

The south mountain gray-cheeked salamander is found in mesic forest, often under leaf-litter, logs, or mossy rocks (Hammerson and Beamer 2004). Much of its range occurs in second growth forest (Hammerson and Beamer 2004).

Biology and Taxonomy:

This species was recently separated from *Plethodon jordani* (Highton and Peabody 2000).

Very little is known about the life history of the South Mountain gray-cheeked salamander. As with other species of *Plethodon*, animals likely feed at night, with activity proportional to moisture levels (Beamer and Lannoo 2005). Prey items include small invertebrates, especially insects, that inhabit or are associated with the forest floor (Beamer and Lannoo 2005). For a summary of available information on life history characteristics, see Beamer and Lannoo (2005).

Population Status:

The South Mountain gray-cheeked salamander qualifies for endangered species status because of its small range and few occurrences, as well as the fact that declines have been observed, likely due to habitat destruction. Highton (2003, as cited in Beamer and Lannoo 2005) sampled two counties (Burke and Cleveland counties, North Carolina) in the decade between 1967 and 1977 and again in 1997, and found numbers to be reduced compared to historical numbers. It is considered vulnerable because it is known from fewer than five locations (Hammerson and Beamer 2004).

THREATS

Habitat alteration and destruction:

Clearcutting has been reported to strongly deplete local populations of the *P. jordani* complex (Petranka et al. 1993), such as the South Mountain gray-cheeked salamander. The time required for recovery from clearcutting is likely at least a few decades (Ash 1997, Petranka 1999, Ash and Pollock 1999). Some areas in the South Mountains are being used for residential development, so habitat destruction for development is also a threat (Hammerson and Beamer 2004).

Inadequacy of existing regulatory mechanisms:

South Mountain gray-cheeked salamanders have one of the smallest distributions of salamanders in the *P. jordani* complex, and within their range, there are only a few state properties that contain suitable habitat for these salamanders. Specifically, one portion of the species' range is protected in South Mountain State Park. The species also occurs in South Mountain Game Lands but recently much of this land has been converted to housing (Beamer and Lannoo 2005). Moreover, protected lands comprise only a fraction of the salamander's range, and all occupied habitats of this highly restricted endemic must be protected to ensure viability.

The North Carolina Natural Heritage Program considers the species "significantly rare" (North Carolina Natural Heritage Program 2008). Nevertheless, South Mountain gray-cheeked salamanders are not protected by North Carolina, the only state within their range. As such, the salamanders can be legally taken and up to 24 can be possessed without a license or permit (Nanjappa and Conrad 2012). Federal protection would prohibit take and ensure that all essential habitats are adequately protected.

Other factors:

Because South Mountain gray-cheeked salamanders have one of the smallest distributions of salamanders in the *P. jordani* complex, its small, isolated populations are susceptible to extirpations due to stochastic events, human impacts, and environmental factors (Soulé 1987, Begone et al. 1990). Lack of gene flow is likely to cause loss of genetic variability due to random genetic drift (Wright 1931) and inbreeding depression may occur (Franklin 1980). Once these populations are extirpated, their isolation precludes genetic interchange and recolonization of habitat.

References:

Ash, A.N. 1997. Disappearance and return of plethodontid salamanders to clearcut plots in the southern Blue Ridge Mountains. Conservation Biology 11: 983–989.

Ash, A. N., and K. H. Pollock. 1999. Clearcuts, salamanders, and field studies. Conservation Biology 13: 206-208.

Beamer, D.A. and M. J. Lannoo. 2005. Plethodon meridianus - Southern Gray-cheeked Salamander, South Mountain Graycheek Salamander. In M. Lannoo, editor. Amphibian declines: the conservation status of United States species. University of California Press, Berkeley.

Begone, M., J. L. Harper, and C. R. Townsend. 1990. Ecology: Individuals, populations, and communities. Blackwell Scientific Publications.

Franklin, I. R. 1980. Evolutionary changes in small populations. In Conservation Biology: An evolutionary-ecological perspective (ed. a. B. A. W. M. E. Soule). pp. 135-149.

Hammerson, G., and D. Beamer 2004. *Plethodon meridianus*. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.2. <www.iucnredlist.org>. Downloaded on 02 February 2012.

Highton, R., and R. B. Peabody. 2000. Geographic protein variation and speciation in salamanders of the *Plethodon jordani* and *Plethodon glutinosus* complexes in the southern Appalachian Mountains with the description of four new species. Pages 31-93 in R. C. Bruce, R. G. Jaeger, and L. D. Houck, editors. The biology of plethodontid salamanders. Kluwer Academic/Plenum Publishers, New York. xiii + 485 pp.

Nanjappa, P. and Paulette M. Conrad 2012. State of the Union: Legal Authority Over the Use of Native Amphibians and Reptiles in the United States, *available at* http://www.fishwildlife.org/files/SOU_FULL-lo-res.pdf (last visited April 30, 2012).

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, *available at* http://www.natureserve.org/explorer (Accessed: February 2, 2012).

North Carolina Natural Heritage Program. 2008. Natural Heritage Program List of the Rare Animal Species of North Carolina, *available at* http://www.ncnhp.org/Images/2008-animal-book-complete.pdf (last visited Jan. 31, 2012).

Petranka, J.W. 1999. Recovery of salamanders after clearcutting in the southern Appalachians: a critique of Ash's estimates. Conservation Biology: 203-205.

Petranka, J. W., M. E. Eldridge, and K. E. Haley. 1993. Effects of timber harvesting on southern Appalachian salamanders. Conservation Biology 7(2): 363-370.

Soulé, M. 1987. Viable populations for conservation. Cambridge University Press, Great Britain. 189 pp.

Wright, S. 1931. Evolution in Mendelian populations. Genetics 16: 97-159.

Scientific Name:

Plethodon petraeus

Common Name:

Pigeon Mountain Salamander

G Rank:

G2

IUCN Red List:

Vulnerable

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

The Pigeon Mountain salamander has a small range within which timber harvesting is degrading remaining habitats. It is limited to the Cumberland Plateau of extreme north-western Georgia (Hammerson 2004). All known populations occur on the eastern slope of Pigeon Mountain in Walker and Chattooga counties (Wynn et al. 1988, Jensen 1999, Buhlmann 2001, Jensen et al. 2002, Jensen and Camp 2004). Sites occur at altitudes ranging from 220-570 m above sea level (Wynn et al. 1988).

Habitat:

Pigeon Mountain Salamanders are associated with limestone outcroppings, boulder fields, and caves (Wynn et al. 1988, Jensen et al. 2002). Those found in caves are rarely deeper than the twilight zone, and individuals are most often found in and around cracks and crevices within rocks. These microhabitats are embedded within mesic deciduous forests consisting of an overstory comprised primarily of oak and hickory (Jensen 1999).

Biology:

General accounts are in Conant and Collins (1991), Petranka (1998), and Jensen (1999). The diets of the Pigeon Mountain salamander and a sympatric species were compared in Jensen and Whiles (2000). Aspects of life history were discussed in Jensen et al. (2002). Morphology, as it relates to climbing abilities, was described by Wynn et al. (1988). Habitat features were presented in Wynn et al. (1988), Jensen (1999), and Jensen et al. (2002). Genetic relationships, as determined by analysis of allozymes, with other members of the *P. glutinosus* species group and other members of the genus Plethodon were presented by Wynn et al. (1988) and Highton (1995).

Population Status:

The Pigeon Mountain salamander qualifies for federal protection because it is known from just one occurrence and that occurrence is facing declines and ongoing threats such as habitat destruction. Specifically, Jensen and Camp (2004) mapped seven tightly clustered localities that could be regarded as a single elongate occurrence. Its small area of habitat is vulnerable to habitat degradation and other threats (NatureServe 2011). Pigeon Mountain salamanders have become uncommon at one locality, possibly due to disturbance created by increased cave visitation or scientific overcollecting (Jensen 1999).

<u>THREATS</u>

Habitat alteration and destruction:

The salamander has been impacted by timber cutting in the past, namely near Harrisburg Cave between 1986 and 1992 (NatureServe 2011). Timber harvest remains a threat, especially when there is loss or reduction of moisture-trapping canopy cover as a result of timber removal (Jensen and Camp 2005).

Overutilization:

The salamander faces some collection pressure, especially at Pettijohn Cave, which is a very popular site for recreational caving (NatureServe 2011). As such, overutilization should be considered a threat to the species.

Inadequacy of existing regulatory mechanisms:

Most of the species' potential habitat is in the Crockford-Pigeon Mountain Wildlife Management Area. Some activities that harm the salamanders can still occur on this land, including timber management.

The Pigeon Mountain salamander is listed as Rare and thus protected from take by the State of Georgia (Jensen and Camp 2005). But such status does not provide protection for the salamander's habitat and is therefore inadequate.

Other factors:

The Pigeon Mountain salamander is known from seven tightly clustered localities that operate as a single occurrence (Jensen and Camp 2004), and as such, as with any small, isolated population, it is more susceptible to extirpations due to stochastic events, human impacts, and environmental factors (Soulé 1987, Begone et al. 1990). Lack of gene flow is likely to cause loss of genetic variability due to random genetic drift (Wright 1931) and inbreeding depression may occur (Franklin 1980). Once these populations are extirpated, their isolation precludes genetic interchange and recolonization of habitat.

Disturbance from recreational spelunkers also likely threatens populations (Jensen and Camp 2005).

References:

Begone, M., J. L. Harper, and C. R. Townsend. 1990. Ecology: Individuals, populations, and communities. Blackwell Scientific Publications.

Buhlmann, K.A. 2001. A biological inventory of eight caves in northwestern Georgia with conservation implications. J. Cave Karst Stud.

Conant, R. and J.T. Collins. 1991. A Field Guide to Reptiles and Amphibians. Eastern and Central North America. 3rd ed. Houghton Mifflin Co., Boston, Massachusetts.

Franklin, I. R. 1980. Evolutionary changes in small populations. In Conservation Biology: An evolutionary-ecological perspective (ed. a. B. A. W. M. E. Soule). pp. 135-149.

Hammerson, G. 2004. Plethodon petraeus. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.2. <www.iucnredlist.org>. Downloaded on 02 February 2012.

Highton, R. 1989. Biochemical evolution in the Slimy Salamanders of the Plethodon glutinosus complex in the eastern United States. Part I. Geographic protein variation. Illinois Biol. Monogr. 57:1–78.

Highton, R. 1995. Speciation in eastern North American salamanders of the genus Plethodon. Ann. Rev. Ecol. Syst. 26:579–600.

Jensen, J.B. 1999. Pigeon Mountain Salamander: Plethodon petraeus, p. 104–105. In T.W. Johnson, J.C. Ozier, J.L. Bohannon, J.B. Jensen, and C. Skelton (eds.), Protected Animals of Georgia. Georgia Department of Natural Resources, Wildlife Resources Division, Social Circle, Georgia.

Jensen, J. B., and C. D. Camp. 2004. *Plethodon petraeus*. Catalogue of American Amphibians and Reptiles 783:1-2.

Jensen, J. and C. Camp. 2005. Plethodon petraeus - Pigeon Mountain Salamander. In M. Lannoo, editor. Amphibian declines: the conservation status of United States species. University of California Press, Berkeley.

Jensen, J.B. and M.R. Whiles. 2000. Diets of sympatric *Plethodon petraeus* and *Plethodon glutinosus*. J. Elisha Mitchell Sci. Soc. 116: 245–250.

Jensen, J.B., C.D. Camp, and J.L. Marshall. 2002. Ecology and life history of the Pigeon Mountain Salamander. Southeast. Nat. 1: 3–16.

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia. Available http://www.natureserve.org/explorer (Accessed: February 2, 2012).

Petranka, J.W. 1998. Salamanders of the United States and Canada. Smithsonian Institution Press, Washington, DC.

Powell, R., J.T. Collins, and E.D. Hooper, Jr. 1998. A Key to Amphibians & Reptiles of the Continental United States and Canada. Univ. Press. Kansas, Lawrence.

Soulé, M. 1987. Viable populations for conservation. Cambridge University Press, Great Britain. 189 pp.

Wright, S. 1931. Evolution in Mendelian populations. Genetics 16: 97-159.

Wynn, A.H., R. Highton, and J.F. Jacobs. 1988. A new species of rockcrevice dwelling Plethodon from Pigeon Mountain, Georgia. Herpetologica 44: 135–143.

Scientific Name:

Plethodon punctatus

Common Name:

Cow Knob Salamander or White-spotted Salamander

G Rank:

G3

IUCN Red List:

Near Threatened

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

Hammerson and Mitchell (2004) report that this species has a narrow range in the Shenandoah, North, and Great North mountains, in George Washington National Forest, Virginia (Augusta, Rockingham, and Shenandoah Counties) and West Virginia (Green and Pauley 1987, Conant and Collins 1991, Petranka 1998), from 735-1,200m above sea level (but mainly from 900-1,200m above sea level). Graham (2007) examined Nathaniel Mountain, West Virginia, which revealed new records on Elliot Knob, extending the known range several miles south.

Habitat:

This species occurs in ridge and valley areas in mixed deciduous forest interspersed with Virginia pine and hemlock and numerous rock outcrops (Green and Pauley 1987). It is found in old-growth forests with many downed logs and in areas with an abundance of surface rocks (Mitchell 1991), including talus, especially on north-facing slopes (Hammerson and Mitchell

2004). During the day, it is found under rocks and logs or in burrows (Hammerson and Mitchell 2004).

Biology:

Cow Knob salamanders are opportunistic carnivores on and above the forest floor during wet conditions (Mitchell et al. 2005). Most active foraging apparently occurs at night, and individuals have been observed to climb on tree trunks and rocks at night (Buhlmann et al. 1988). Adults and juveniles prey on a wide variety of invertebrates, including ants, collembolans, beetles, dipterans, coleopterans, orthopterans, insect larvae, millipedes, centipedes, spiders, and mites (Fraser 1976b, Tucker 1998). The size of the prey item is positively correlated with the size of the salamander (Mitchell et al. 2005). For additional life history information, see Tucker (1998), Mitchell et al. (2005), and Virginia Fish and Wildlife (2012).

Population Status:

Federal protection for the Cow Knob salamander is needed because its populations are declining and its occupied range is shrinking, mostly due to habitat loss across its narrow range. According to NatureServe (2011), deforestation has likely reduced distribution and abundance compared to historical levels.

THREATS

Habitat alteration and destruction:

A principal threat to this species has been deforestation through logging because it prefers old-growth forests with many downed logs (Mitchell 1991). As such, forest management practices such as short rotation forestry are a threat to the species. Firewood collection is also a potential threat (Hammerson and Mitchell 2004). Mortality can be minimized if logging occurs outside the seasonal activity period but any intensive logging is detrimental (Buhlmann et al. 1988).

It is also threatened by defoliation by gypsy moths (*Lymantria dispar*) (Mitchell 1991). Spraying pesticides to control gypsy moths migh also impact the salamanders but the effect is unknown (Mitchell 1991).

Overutilization:

There is a Candidate Conservation Agreement between the U.S. Forest Service (USFS) and the U.S. Fish and Wildlife Service (USFWS) that prohibits the collection of the Cow Knob salamander on the national forest (Nanjappa and Conrad 2012). Nevertheless, overcollection of individuals is a potential threat (Hammerson and Mitchell 2004).

Inadequacy of existing regulatory mechanisms:

The George Washington National Forest includes the Cow Knob salamander as a Management Indicator Species in the draft forest plan (U.S. Forest Service 2011). As such, 47,000 acres are managed for the salamander with a forest plan species diversity objective (OBJ SPD-9) to "[m]aintain a stable and/or increasing population trend for the Cow Knob salamander over the planning period through protection and maintenance of the Cow Knob Salamander Habitat Conservation Area. (See Management Prescription Area 8E7)" (U.S. Forest Service 2011). While this objective provides important benefits for the salamander, it only applies to those populations within the management area of the George Washington National Forest.

In addition, a formal Conservation Agreement exists, via a Memorandum of Understanding between the George Washington National Forest and the U.S. Fish and Wildlife Service, which affords the habitat of this species on public lands some protection from logging and other potentially damaging operations (Mitchell 1994). But this agreement is only voluntary and cannot ensure that its limited habitat will be adequately protected.

Timber management with cutting, temporary road-building, and oil and gas leasing with Controlled Surface Occupancy is planned within the salamander's range (U.S. Forest Service 2011). The Forest Service also continues to run off-highway vehicle routes through the salamanders' home territory (Virginia Forest Watch undated).

Because the rarity of this species was realized after some preliminary studies (Highton 1972, Fraser 1976, Buhlmann et al. 1988), the Cow Knob salamander was recognized as a species of special concern by the Virginia Department of Game and Inland Fisheries and by the West Virginia Division of Natural Resources (Flint and Harris 2005). But such designation provides no habitat protection, which is needed to ensure species viability.

Other factors:

Cow Knob salamanders are also likely threatened by the loss of hemlock trees by the introduced hemlock woolly adelgid (*Adelges tsugae*) (Mitchell et al. 2005).

References:

Buhlmann, K.A., C. Pague, J. Mitchell, and R. Glasglow. 1988. Forestry operations and terrestrial salamanders: techniques in a study of the Cow Knob salamander, *Plethodon punctatus*. In: Szaro, R., Severson, K. and Patton, D. (eds), Management of Amphibians, Reptiles and Small Mammals in North America, pp. 38-44. USDA For. Serv. Gen. Tech. Rep. RM-166.

Conant, R. and J.T. Collins. 1991. A Field Guide to Reptiles and Amphibians: Eastern and Central North America. Third Edition. Houghton Mifflin Company, Boston, Massachusetts.

Flint, W.D. and R. N. Harris. 2005. The Efficacy of Visual Encounter Surveys for Population Monitoring of *Plethodon punctatus* (Caudata: Plethodontidae). Journal of Herpetology 39(4): 578-584.

Fraser, D.F. 1976. Coexistance of salamanders in the genus Plethodon: a variation of the Santa Rosalia theme. Ecology 57: 238-251.

Graham, Matthew R. 2007. Distribution and Conservation Genetics of the Cow Knob Salamander, Plethodon punctatus Highton (Caudata: Plethodontidae). Masters degree thesis, Marshall University, *available at* http://www.marshall.edu/etd/masters/graham-matthew-2007-ma.pdf (last visited Feb. 3, 2012).

Green, N.B. and T.K. Pauley. 1987. Amphibians and Reptiles in West Virginia. University of Pittsburgh Press, Pittsburgh, PA, USA.

Hammerson, G., and J. Mitchell 2004. Plethodon punctatus. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.2. <www.iucnredlist.org>. Downloaded on 03 February 2012.

Highton, R. 1972. Distributional interactions among Eastern North American salamanders of the genus Plethodon. In C. Holt (ed.), The Distributional History of the Biota of the Southern Appalachians. Part III. Vertebrates, pp. 139-188. Research Division Monograph, Virginia Polytechnic Institute and State Univ., Blacksburg.

Nanjappa, P. and Paulette M. Conrad 2012. State of the Union: Legal Authority Over the Use of Native Amphibians and Reptiles in the United States, *available at* http://www.fishwildlife.org/files/SOU_FULL-lo-res.pdf (last visited April 30, 2012).

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, *available at* http://www.natureserve.org/explorer (Accessed: February 3, 2012).

Mitchell, J.C. 1991. Amphibians and reptiles. In: Terwilliger, K. (ed.), Virginia's Endangered Species: Proceedings of a Symposium, pp. 411-476. McDonald and Woodward Publishing Company, Blacksburg, Virginia.

Mitchell, J.C. 1994. Habitat conservation assessment for the cow knob salamander (Plethodon punctatus) in the George Washington National Forest.

Mitchell, J.C., T.K. Pauley, and Allegheny Institute of Natural History. 2005. *Plethodon punctatus* - Cow Knob Salamander, White-spotted Salamander. In M. Lannoo, editor. Amphibian declines: the conservation status of United States species. University of California Press, Berkeley.

Petranka, J.W. 1998. Salamanders of the United States and Canada. Smithsonian Institution Press, Washington, D.C.

Tucker, R.B. 1998. Ecology and natural history of the cow knob salamander, *Plethodon punctatus* Highton, in West Virginia. M.S. Thesis. Marshall University.

U.S. Forest Service. 2011. Draft Revised Land and Resource Management Plan - George Washington National Forest, *available at* http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=14&ved=0CEEQFjADOAo&url=http%3A%2F%2Fdocuments.foodandwaterwatch.org%2Fdoc%2FFrack_Actions_Geor

geWashingtonNationalForestVA.pdf&ei=e87xT-GvGYfW2AXvvJypBg&usg=AFQjCNFeg3606wGB3KL7lNktSf17wzJRXQ&sig2=k5z0PQpsbPw0FWTxGsyzWA (last visited July 2, 2012).

Virginia Fish and Wildlife. 2012. Taxonomy chapter for Salamander, Cow Knob (020027), *available at* http://www.vafwis.org/fwis/booklet.html?Menu=_.All+Chapters&bova=020027&version=15248 (last visited Feb. 3, 2012).

Virginia Forest Watch. Undated. Our Land, Our Water, Our Home: Ensuring a Healthy Future For Our George Washington National Forest, *available at* www.virginiaforestwatch.org/docs/ourland.pdf (last visited July 2, 2012).

Scientific Name:

Plethodon welleri

Common Name:

Weller's Salamander

G Rank:

G3

IUCN Red List:

Endangered

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

Weller's salamander has a highly fragmented distribution and is found in Whitetop, Mount Rogers, and Pine Mountain, Virginia, south-westward to Yancy County, North Carolina, extreme eastern Tennessee, and eastward to Grandfather Mountain, Caldwell County, North Carolina, USA (Pague 1991). Populations are scattered and isolated (Pague 1991) and appear to be relictual from a previously more widespread distribution (Thurow 1956). It occurs at elevations of 760 m above sea level or more, chiefly in spruce forests above 1,500m above sea level (Conant and Collins 1991).

Habitat:

The Weller's salamander inhabits spruce-fir, birch-hemlock, and primarily deciduous forests, and is also found in grassy spots and boulder fields (Hammerson and Beamer 2004). It is usually found under rocks or logs, or in leaf-litter, during the day, and tends to be associated with rocky substrates (Hammerson and Beamer 2004). Breeding and non-breeding habitats are the same (Hammerson and Beamer 2004). The eggs are laid in small cavities in rotting conifer logs or beneath moss mats. The species does best in forests with much cover of rocks and downed logs (Mitchell 1991). It is apparently not tolerant of habitat disturbance (Hammerson and Beamer 2004).

Biology:

Apparently breeding of the Weller's salamander takes place in spring and fall. Males had vasa deferentia swollen with sperm in April and September and secondary sex characters show greatest development at this time (Thurow 1963). Organ (1960) observed courtship of captive individuals in October. Available life history information is summarized by Organ (1960) and Beamer and Lannoo (2005).

Population Status:

ESA protection for the Weller's salamander is needed because it is experiencing significant declines across its highly fragmented distribution and remaining populations are subject to ongoing habitat loss. Highton (2005) sampled Weller's salamanders from a population on the North Carolina–Tennessee border (Mitchell and Unicoi counties, respectively) and from a population in Virginia (Grayson-Smyth County) prior to 1988 and again after 1993 and found precipitous declines in both populations. Hammerson and Beamer (2004) rank the Weller's salamander as endangered because of its small range and severely fragmented distribution, and there is continuing decline in the extent and quality of its habitat in Virginia, North Carolina, and Tennessee.

THREATS

Habitat alteration and destruction:

Loss of habitat is the primary threat to the Weller's salamander, as it appears very sensitive to habitat change and cannot tolerate disturbance (Braswell 1989, Petranka 1998). At Whitetop Mountain, where there are anthropogenic disturbances adjacent to the spruce-fir forest, the species does not appear in the disturbed areas (NatureServe 2011). Specifically, habitat loss due to development and logging is a major threat (Braswell 1989).

In Petranka's (1998) view, the die-off of spruce-fir forests in the southern Appalachians – from acid rain or spruce budworm – constitutes a major threat to populations of Weller's salamanders. Other forest losses, such as from catastrophic fires, are also a threat (Pague 1991).

Inadequacy of existing regulatory mechanisms:

The Weller's salamander occurs on state or federal property and therefore enjoys some degree of protection in these areas. For example, in Virginia, much of the range occurs in the Mount Rogers National Recreation Area, primarily in zones designated as protected, and the Grandfather Mountain population appears protected by the current landowner (Hammerson and

Beamer 2004). Yet these modest protections cover only a fraction of the salamander's range and have not been sufficient to prevent the ongoing and significant declines observed for the species.

It is listed as a species of special concern in North Carolina and Virginia and as Wildlife in Need of Management in Tennessee (Redmond and Scott 1996). These statuses reflect the rarity of the Weller's salamander but do not protect its habitat.

Collection is prohibited in Tennessee (Nanjappa and Conrad 2012). In North Carolina, the salamanders can be legally taken and up to 24 can be possessed without a license or permit (Nanjappa and Conrad. 2012). Salamanders are protected from commercial collection in Virginia but no permit required to collect native amphibians and reptiles for personal use with a daily bag limit of five animals (Nanjappa and Conrad 2012). Federal protection is necessary so that take of these rare animals would be prohibited across its range.

It is also considered a Regional Forester's Sensitive species (U.S. Forest Service undated). Sensitive species status requires analysis of impacts on the species but does not prevent harmful activities, such as intensive timber management.

Other factors:

As a montane-restricted species, the Weller's salamander is especially vulnerable to climate change because it is unable to shift its elevational range upslope as the climate warms. Scientists have found evidence *Plethodon* mountaintop endemics are likely range-restricted due to climactic factors (Kozak and Wiens 2006, Arif et al. 2007).

In addition, due to the generally small and isolated populations of this species, all populations are vulnerable to accidents or policy changes in land management (Braswell 1989). To be sure, small, isolated populations are more susceptible to extirpations due to stochastic events, human impacts, and environmental factors (Soulé 1987, Begone et al. 1990). Lack of gene flow is likely to cause loss of genetic variability due to random genetic drift (Wright 1931) and inbreeding depression may occur (Franklin 1980). Once these populations are extirpated, their isolation precludes genetic interchange and recolonization of habitat.

References:

Beamer, D. and M. Lannoo. 2005. In M. Lannoo, editor. Amphibian declines: the conservation status of United States species. University of California Press, Berkeley.

Begone, M., J. L. Harper, and C. R. Townsend. 1990. Ecology: Individuals, populations, and communities. Blackwell Scientific Publications.

Bishop, S.C. 1943. A Handbook of Salamanders. The Salamanders of the United States, of Canada, and of Lower California. Comstock Publishing Company, Ithaca, New York.

Braswell, A. L. 1989. Scientific council report on the conservation status of North Carolina amphibians and reptiles. Submitted to : Nongame Advisory Committee, North Carolina Wildlife Resources Commission.

Conant, R. and J. T. Collins. 1991. A field guide to reptiles and amphibians: eastern and central North America. Third edition. Houghton Mifflin Co., Boston, Massachusetts. 450 pp.

Franklin, I. R. 1980. Evolutionary changes in small populations. In Conservation Biology: An evolutionary-ecological perspective (ed. a. B. A. W. M. E. Soule). pp. 135-149.

Hammerson, G., and D. Beamer. 2004. Plethodon welleri. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.2. <www.iucnredlist.org>. Downloaded on 03 February 2012.

Highton, R. 2005. Declines of eastern North American woodland salamanders (*Plethodon*). Pages 34-46 in M. Lannoo, editor. Amphibian declines: the conservation status of United States species. University of California Press, Berkeley.

Mitchell, J.C. 1991. Amphibians and reptiles. In: Terwilliger, K. (ed.), *Virginia's Endangered Species: Proceedings of a Symposium*, pp. 411-476. McDonald and Woodward Publishing Company, Blacksburg, Virginia.

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, *available at* http://www.natureserve.org/explorer (Accessed: February 3, 2012).

Organ, J.A. 1960. Studies on the Life History of the Salamander, Plethodon welleri. Copeia 1960: 4.

Pague, C. A. 1991. Amphibians and reptiles. Pages 411-76 in K. Terwilliger (coordinator). Virginia’s Endangered Species: Proceedings of a Symposium. McDonald and Woodward Publishing Company, Blacksburg, Virginia.

Petranka, J. W. 1998. Salamanders of the United States and Canada. Smithsonian Institution Press, Washington, D.C.

Redmond, W. H., and A. F. Scott. 1996. Atlas of Amphibians in Tennessee. Miscellaneous Publication No. 12. Center for Field Biology, Austin Peay State University, Clarkville, Tennessee.

Soulé, M. 1987. Viable populations for conservation. Cambridge University Press, Great Britain. 189 pp.

Thurow, G.R. 1956. A new subspecies of *Plethodon welleri*, with notes on other members of the genus. American Midland Naturalist 55: 343–356.

Thurow, G.R. 1963. Taxonomic and ecological notes on the salamander, *Plethodon welleri*. University of Kansas Science Bulletin 44: 87–108.

U.S. Forest Service. Undated. Biological Evaluation for the Environmental Assessment of the Northside Project, *available at* http://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5188474.pdf (last visited Feb. 3, 2012).

Wright, S. 1931. Evolution in Mendelian populations. Genetics 16: 97-159.

Scientific Name:

Aneides aeneus

Common Name:

Green Salamander

G Rank:

G3

IUCN Red List:

Near Threatened

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

While the range of the green salamander encompasses the Appalacian region, the salamander exists in just fragments of remaining habitat within that range with populations facing extirpations and significant declines. Specifically, NatureServe (2011) explains that the range extends discontinuously from extreme southwestern Pennsylvania, extreme western Maryland, and southern Ohio to extreme northeastern Mississippi, northern Alabama, northern Georgia, western North Carolina, and western South Carolina (Pauley and Watson 2005), with a widely disjunct occurrence in Crawford County, southern Indiana (Madej 1994). The North Carolina/South Carolina/northeastern Georgia distribution is disjunct from the main portion of the range (NatureServe 2011). The Great Smoky Mountain population has not been located since its discovery in the 1930s (Redmond and Scott 1996).

The elevational range is around 140-1350 meters, with the highest known occurrences in Kentucky and North Carolina (NatureServe 2011).

Habitat:

The green salamander occupies damp (but not wet) crevices in shaded rock outcrops and ledges (Petranka 1998, NatureServe 2011). Occasionally, green salamanders are found on dry rock outcrops (Pauley 1993b, Waldron et al. 2000). Rock types include sandstone, limestone, dolomite, granite, and quartzite. Type of rock may be less important than crevice size and moisture (Gordon and Smith 1949).

It can also be found beneath loose bark and in cracks of standing or fallen trees (e.g., in cove hardwoods) (NatureServe 2011), and sometimes in or under logs on ground (e.g., Wilson 2003). Previously, arboreal habitat was deemed secondary to rock outcrops as preferred habitat and in situations where more suitable habitat was not available (Gordon and Smith 1949, Gordon 1952, Woods 1968, Barbour 1971, Mount 1975). Recent studies (Wilson 2003, Waldron and Humphries 2005) indicate that woody and arboreal habitats play a much larger role in the life-history of green salamanders than generally thought.

Eggs are laid in rock crevices (e.g., Davis 2004), rotting stumps, or similar dark, damp places (NatureServe 2011). During the spring and summer, breeding females require cool, clean and moist horizontal crevices or narrow chambers in which to suspend their eggs from an overhead substrate. Such habitat provides a specific micro-climate necessary for successful embryonic development. In fall, individuals of all age classes congregate near deep rock crevices for use during winter hibernation. Largely due to these unique habitat requirements, the green salamander is patchily distributed and uncommon throughout its range (NatureServe 2011).

Taxonomy and Biology:

Phyogenetic studies have suggested that there may be more than one species currently assigned to *Aneides aeneus* (Brodman 2004). If additional species within the green salamander are described while this petition is pending, we hereby petition for those species.

Green salamanders are one of the most unique salamander species in the eastern United States, as they represent the only member of the "climbing family" of salamanders (genus *Aneides*) east of the Rocky Mountains (Wilson 2001). Waldron and Pauley (2007) found that green salamanders reach reproductive maturity at a greater age than has been reported for most plethodontid salamanders. The researchers suggest that this, along with its narrow habitat associations, might offer insight into why the species is sensitive to population declines. Life history of the green salamander is summarized by Pauley and Watson (2005) and Brodman (2004).

Population Status:

The green salamander qualifies as endangered because the area of occupancy, number of subpopulations, and especially abundance have declined significantly over the past several decades (NatureServe 2011). The IUCN Red List considers it Near Threatened because this species is in significant decline because of habitat loss, over-harvest, disease, and drought, thus making the species close to qualifying for Vulnerable (Hammerson 2004).

In 2005, Waldron and Humphries remarked that the green salamander is "listed as critically imperiled, imperiled, or vulnerable in 10 of the 13 states in which it occurs." More recently, NatureServe (2011) provides that it is critically imperiled, imperiled, or vulnerable in all states in its range. Much of the range includes isolated and peripheral populations that are vulnerable to statewide extinction (Brodman 2004). Recent reports of local extirpations come from throughout the species' distribution (Cline 2008). In addition, the species is experiencing habitat destruction from logging and mining across its remaining habitat.

Nine counties in North Carolina, South Carolina, and Georgia make up the disjunct Blue Ridge portion of the green salamanders' range. These counties historically contained 37 populations of green salamanders, but this species has apparently disappeared from 78 present of its known localities (U.S. Fish and Wildl. Serv. 1987), and more declines have likely occurred since that study. Corser (2001) monitored seven historical populations of green salamanders throughout the 1990s on the Blue Ridge Escarpment in northeastern Georgia, northwestern South Carolina, and southwestern North Carolina and found a 98 percent decline in relative abundance compared to numbers observed in the 1970s. He attributed this sharp decline to habitat loss, overcollecting, epidemic disease, and climate changes.

Green salamanders declined for some unknown reason in the 1970s in North Carolina (Mitchell et al. 1999). Snyder (1991) discussed several potential reasons for this decline but concluded that the probable cause was that unusually prolonged cold periods may have frozen many hibernating green salamanders in a torpid condition, which prevented them from moving into deeper and safer crevices.

THREATS

Habitat alteration and destruction:

Habitat loss and alteration is a primary threat to the green salamander (Mitchell et al. 1999, Corser 2001). As explained below, habitat destruction and degradation can occur as a result of logging, mining, road construction, water impoundments, and chemical contamination.

Logging

Logging has direct and indirect effects on green salamanders. Logging destroys the arboreal habitats of the salamanders (Waldron and Humphries 2005). The ideal woody crevice habitat would consist of large, thick slabs of exfoliating bark or hollowed logs (Wilson 2001). Such features are characteristic of old growth forests but the second- and third-growth forests that currently predominate the landscape contain very little large-woody-debris (Wilson 2001). As such, loss of old-growth forests and arboreal foraging sites due to logging (and chestnut blight) have likely primarily restricted the green salamanders to crevices of rock outcrops in the last half century (Gordon 1952, Corser 1991, Petranka 1998, Corser 2001, Wilson 2003).

Logging that removes trees that shade rock outcrops also results in greater insolation that ultimately dries nesting and foraging crevices, making them unsuitable (Pauley and Watson 2005, Cline 2008, Brodman 2004). Such logging can also alter the prey base for the salamanders (Cline 2008).

Road Construction

Road cuts and other larger corridors adjacent to emergent rocks and outcrops can result in an increase in airflow and greater insolation, thus increasing temperatures and decreasing moisture (Pauley and Watson 2005). Plus, removal of emergent rocks in forests for roads eliminates this species in such areas (Pauley and Watson 2005). Road mortality is also a concern, as green salamanders have been seen crossing roads (Williams and Gordon 1961).

Mountaintop Removal and Other Mining

Mountaintop removal negatively impacts many salamander species including high elevation endemics like the green salamander (Gatwicke 2008). As mountain topography is altered, high-elevation habitat is entirely lost. The flattened topography alters the soil structure and hydrology and removes microhabitats essential for the salamanders. Valley-filling activities also result in flooding, sedimentation, and aquatic pollution and alter essential ecological elements for salamanders including organic matter, cover, and the prey base (Gatwicke 2008).

Gatwicke (2008) states: "Of all the mining practices in the Appalachians, mountain top removal is the most visible and destructive for salamanders. For example, in 2007, West Virginia alone had 300,000 acres of active mountain-top removal mining permits amounting to 2% of the State's total land area (OSMRE 2007). Salamanders are the vertebrates hardest hit by these activities and their populations can take as long as 70 years to recover to pre-disturbance levels (Williams 2003)."

In addition to mountaintop removal, strip mining also negatively affects salamanders. Strip mining results in girded mountain islands that permanently fragment and isolate mountaintop habitat, and hydrology is altered leading to the drying out of forest fragments and severe edge effects (EPA 2005, Gatwicke 2008).

Extensive strip mining for coal is conducted in West Virginia, Kentucky, Virginia, Tennessee, and Alabama (Dodd 1997). As of 2004, more than 1.1 million acres of land in Appalachia were undergoing active mining operations (Loveland et al. 2003). The EPA projects that from 1992-2013, 761,000 acres of Appalachian forest will be lost to surface coal mining (Pomponio 2009). This figure does not include the forest lost prior to 1992. Nearly seven percent of the forest that still existed in 1992 will be lost to coal mining by 2013 (Pomponio 2009). In some West Virginia watersheds, more than 25 percent of the area has been surface mined for coal (Palmer and Bernhardt 2009). Much of former habitat in Alabama has been strip-mined (Cline 2008).

Sand mining is also a threat to the green salamander (Pennsylvania Natural History Fund undated), including a population that historically occurred at one site in southern Fayette County, Pennsylvania, the northernmost known locality in their limited Appalachian range.

Water Impoundments

River impoundments have inundated bluffs that previously supported populations (Cline 2008). A large swath of habitat was lost in 1973 when the lower gorges were impounded by Jocasse Lake (Corser 2001).

Overutilization:

Overcollecting is likely a threat to the salamanders in some areas (Mitchell et al. 1999, Corser 2001, Wilson 2001, Cline 2008). With its bright green color and tolerance to dry conditions, the green salamander is coveted by collectors (Wilson 2001, Brodman 2004).

Disease or predation:

Thus far, no diseases are documented in green salamanders but as a montane habitat specialist with low fecundity the green salamander is vulnerable to chytridiomycosis (Daszak et al. 1999, Corser 2001, Hammerson 2004). The parasitic nematode *Batacholandros magnavulgaris* has been reported from green salamanders (Baker 1987).

Inadequacy of existing regulatory mechanisms:

The green salamander is protected from collection in states that have listed the species as endangered or threatened: Indiana, Ohio, Maryland, Pennsylvania, North Carolina, Mississippi, West Virginia (Indiana DNR undated, Ohio DNR 2010, Maryland DNR 2011, Pennsylvania Game Commission 2011, Mississippi Wildlife Fisheries and Parks 2002, West Virgina Dept. of Natural Resources 2007, North Carolina Wildlife Resources Commission 2008). The animal is listed as Rare by the Georgia and South Carolina Departments of Natural Resources (SC DNR 2010, GA DNR 2009).

The green salamander lacks special status in Alabama, Virginia and Kentucky (Alabama Dept. of Conservation and Natural Resources 2008, Virginia Department of Game and Inland Fisheries 2011, Kentucky Fish and Wildlife Resources 2001, Tennesee Wildlife Resources Agency 2011). Although not protecting the salamanders as a state listed species, some states, such as Georgia, protect the salamanders from collection as protected nongame wildlife. *See, e.g.*, O.C.G.A. § 27-1-28.

Even those states with designations that offer some protection to the salamanders are inadequate, however, because they do nothing to prevent habitat destruction, which is the primary threat the salamanders face.

The green salamander is designated as a Regional Forester Sensitive Species in the Eastern Region of the Forest Service (Brodman 2004). Occurrence of the green salamander is documented but not designated as sensitive on the Hoosier and Monongahela National Forests (Brodman 2004), as well as Chattahoochee National Forest within the Ellicott Rock Wilderness Area (Georgia DNR 2009). Sensitive species designations do not prevent actions that might harm the salamanders, requiring only that the effects of the actions be analyzed.

To maintain the shading of rock outcrops, the U. S. Forest Service has a policy of leaving a 100 foot buffer around potential habitats when logging (Wilson 2001). For example, the Chattahoochee National Forest requires that mature forest cover is maintained within 100 feet slope distance from the top of cliffs (U.S. Forest Service 2004). However, as explained below, the long-term survival of these small colonies likely suffers due to a larger problem of depressed populations, limited dispersal, and stochastic events (Wilson 2001). The rock outcrop "patches" are essentially islands among a sea of non-suitable habitat (Wilson 2001). Logging beyond the 100 feet buffer also destroys arboreal habitats.

The green salamander is also found on some state lands throughout its range. For example, in Georgia, the salamander is found on Cloudland Canyon and Tallulah Gorge State Parks, Zahnd Natural Area, and Crockford-Pigeon Mountain Wildlife Management Area (Georgia DNR 2009). And in North Carolina, the salamander can be found on Dupont State Forest (National Wildlife 2007). While these salamander habitats are protected from development, they are not necessarily protected from roadbuilding, logging, and other threats to the salamander.

As explained above, the green salamander is threatened by coal mining. The Surface Mining Control and Reclamation Act of 1977 ("SMCRA") is intended to prevent the degradation of habitats from coal mining activities. But due to increased demand for coal, lax enforcement of environmental laws, and deference to economic development over species' protection, SMCRA is not adequately protecting the green salamander. FWS has acknowledged that mining activities continue to be permitted even when imperiled species are placed at risk: "[I]t has been the Service's experience, after dealing with hundreds of mining projects, that in nearly all cases where there is a conflict between endangered species and a mining project, the project is permitted with only minor modifications" (FWS 1997, p. 1651). In addition, reclamation required under SMCRA is not rigorously enforced (Ward 2009). Even when reclamation is conducted, it has not resulted in the restoration of pre-mining hydrologic characteristics or ecological functions (Townsend et al. 2009, Palmer et al. 2010).

Other factors:

Corser (2001) explains that climate change is likely linked to green salamander declines. Climate change is especially problematical for relictual forest-obligate species like the green salamander, which occupies disjunct areas on the periphery of their range (Homan and Blows 1993). In addition, extreme weather events triggered by climate change could cause dieoffs. Snyder (1991) attributed a decline in Blue Ridge populations to mortality associated with prolonged cold periods in winter, and severe drought might exacerbate other threats or cause temporary declines (Hammerson 2004).

The total range of the green salamander is fragmented into several disjunct populations, and such isolated populations are always at risk of extinction (Pauley and Watson 2005). Small, semi-isolated plethodontid populations (Larson et al. 1984) face demographic and genetic and challenges as a result of a fragmented metapopulation structure. Johnson (2002) showed that gene flow can occur within a one kilometer radius, but is reduced or absent at three kilometers.

Human-induced chemical threats to the green salamander include acid precipitation, heavy metals, herbicides, and pesticides (Cline 2008). The latter three are likely to negatively impact species through exposure via the food chain or through contamination of groundwater.

References:

Alabama Dept. of Conservation and Natural Resources. 2008. Endangered, Threatened or Protected Species, *available at* http://www.outdooralabama.com/education/generalinfo/endangered-species.cfm (last visited Dec. 8, 2011).

Barbour, R.W. 1971. Amphibians and reptiles of Kentucky. Univ. Press of Kentucky, Lexington. x + 334 pp.

Brodman, R. 2004. R9 Species Conservation Assessment for the Green Salamander, Aneides aeneus (Cope and Packard), *available at* http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=77&ved=0CE0QFjAGOEY&url=http%3A%2F%2Fwww.fs.fed.us%2Fr9%2Fwildlife%2Ftes%2Fca-overview%2Fdocs%2FAmphibians%2520and%2520Reptiles%2FGreen%2520salamander%2520assessment.pdf&ei=7uDfTrzdFqm62gXh2KD8BA&usg=AFQjCNEMyYRlarv4GByzsxF2-P-02GgpRQ (last visited Dec. 7, 2011).

Cline, G.R. 2008. Green Salamander, *available at* http://www.dcnr.state.al.us/watchable-wildlife/what/Amphibians/Salamanders/gs.cfm (last visited Dec. 7, 2011).

Corser, J.D. 2001. Decline of disjunct green salamander (*Aneides aeneus*) populations in the southern Appalachians. Biological Conservation 97: 119-126.

Daszak, P., Berger, L., Cunningham, C.C., Hyatt, A.D., Green, D.E., Speare, R., 1999. Emerging infectious diseases and amphibian population declines. Emerging Infectious Diseases 5: 735±748.

Davis, A. 2004. *Aneides aeneus* and *Plethodon glutinosus*. Nesting observations. Herpetological Review 35: 51-52.

Dodd, C.K. Jr. 1997. Imperiled Amphibians: A Historical Perspective. p. 165-200 In: Benz, G.W. and D.E. Collins (editors). 1997. Aquatic Fauna in Peril: The Southeastern Perspective. Southeast Aquatic Research Institute Special Publication 1, Lenz Design and Communications, Decatur, GA. 553 pp.

Georgia Dept. of Natural Resources. 2009. Green salamander species account, *available at* http://www.georgiawildlife.com/sites/default/files/uploads/wildlife/nongame/pdf/accounts/amphibians/aneides_aeneus.pdf (last visited Dec. 8, 2011).

Gordon, R.E. 1952. A contribution to the life history and ecology of the plethodontid salamander *Aneides aeneus* (Cope and Packard). American Midland Naturalist 47: 666–701.

Gordon, R.E. 1967. Aneides aeneus. Catalogue of American Amphibians and Reptiles 30: 1-2.

Gordon, R. E., and R. L. Smith. 1949. Notes on the life history of the salamander Aneides aeneus. Copeia 1949: 173–175.

Gratwicke, B (ed). 2008. Proceedings of the Appalachian Salamander Conservation Workshop. IUCN/SSC Conservation Breeding Specialist Group: Apple Valley, MN.

Hammerson, G. 2004. *Aneides aeneus*. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.2. <www.iucnredlist.org>. Downloaded on 07 December 2011.

Homan, A.A., and M.W. Blows. 1993. Evolutionary genetics and climate change: will animals adapt to global warming?. In: Karieva, P.M., Kingsolver, J.G., Huey, R.B. (Eds.), Biotic Interactions and Global Climate Change. Sinauer Associates, Sunderland, MA, pp. 165±178.

Indiana Dept. of Natural Resources. Undated. Indiana's Species of Greatest Conservation Need, available at http://www.in.gov/dnr/fishwild/2356.htm (last visited Dec. 8, 2011).

Johnson, A.N. 2002. Determining the Genetic Distances between sub-populations of Aneides aeneus in the Westvāco Wildlife and Ecological Research Forest, *available at* http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=90&ved=0CGIQFjAJOFA&url=http%3A%2F%2Fwww.marshall.edu%2Fetd%2Fmasters%2Fjohnson-andrew-2002-ms.pdf&ei=jujfTufaB6Sg2AXJ58mIBQ&usg=AFQjCNEGwWr7jZyYO6AV4hSuYSGGkXJcYw (last visited Dec. 7, 2011).

Kentucky Fish and Wildlife Resources. 2011. Species Information, *available at* http://fw.ky.gov/kfwis/speciesInfo/speciesList.asp?strTaxonomic=ScientificName&strKey=Aneides%20aeneus&strSort1=CommonName&strSort2=ScientificName&strSort3=Class (last visited Dec. 7, 2011).

Kentucky Fish and Wildlife Resources. 2001. Kentucky's Threatened and Endangered Species, *available at* http://fw.ky.gov/telst.asp (last visited Dec. 8, 2011).

Larson, A., D.B. Wake, K.P. Yanev. 1984. Measuring gene flow among populations having high levels of genetic fragmentation. Genetics 106: 293±308.

Madej, R. F. 1994. *Aneides aeneus* (green salamander). Herpetological Review 25: 31.

Maryland Dept. of Natural Resources. 2011. Rare, Threatened, and Endangered Animals of Maryland, *available at* http://www.dnr.state.md.us/wildlife/Plants_Wildlife/rte/pdfs/rte_Animal_List.pdf (last visited Dec. 8, 2011).

Mitchell, J. C., T. K. Pauley, D. I. Withers, S. M. Roble, B. T. Miller, A. L. Braswell, P. V. Cupp, Jr,. and C. S. Hobson. 1999. Conservation status of the southern Appalachian herpetofauna. Virginia Journal of Science 50: 13-35.

Mississippi Wildlife Fisheries and Parks. 2002. Mississippi Natural Heritage Program: ENDANGERED SPECIES OF MISSISSIPPI, *available at* http://www.google.com/url?q=http://www.mdwfp.com/museum/downloads/tandelist.pdf&sa=U&ei=st7gTqHmBcb2gAfngsneBQ&ved=0CAgQFjAC&client=internal-uds-cse&usg=AFQjCNFpRpgxFj8LEXzvfsrmlQ0nujzKwA (last visited Dec. 8, 2011).

Mount, R. H. 1975. The reptiles and amphibians of Alabama. Auburn University Agricultural Experiment Station, Auburn, Alabama. vii + 347 pp.

National Wildlife (World Edition). Making Plans to Save Wildlife 45(4): 52.

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, *available at* http://www.natureserve.org/explorer (Accessed: December 7, 2011).

North Carolina Wildlife Resources Commission. 2008. North Carolina Endangered Species, *available at* http://www.ncwildlife.org/Conserving/Species.aspx (last visited Dec. 8, 2011).

Office of Surface Mining Reclamation and Enforcement (OSMRE). 2007. Annual evaluation summary report for the regulatory and abandoned mine land reclamation programs administered by the state of West Virginia. Page 46. Charleston Field Office, West Virginia. http://www.osmre.gov/EvaluationInfo/2007.htm

Ohio Dept. of Natural Resources. 2010. Wildlife that are considered to be endangered, threatened, species of concern, special interest, extirpated, or extinct in Ohio, *available at* http://www.dnr.state.oh.us/Home/ExperienceWildlifeSubHomePage/Endangeredthreatenedspeciesplaceholder/resourcesmgtplansspecieslist/tabid/5664/Default.aspx (last visited Dec. 8, 2011).

Palmer, M.A., and E.S. Bernhardt. 2009. Mountaintop Mining Valley Fills and Aquatic Ecosystems: A Scientific Primer on Impacts and Mitigation Approaches, *available at*: wvgazette.com/static/mtrwhitepaper.pdf (last visited July 9 , 2012).

Palmer, M.A., E.S. Bernhardt, W.H. Schlesinger, K.N. Eshleman, E. Foufoula-Georgiou, M.S. Hendryx, A.D. Lemly, G.E. Likens, O.L. Loucks, M.E. Power, P.S. White, and P.R. Wilcock. 2010. Mountaintop mining consequences. Science 327: 148-149.

Pauley, T.K. 1993b. Report of Upland Vertebrates in the New River Gorge National River. Volumes I–III. Report to the United States Department of the Interior-National Park Service, Glen Jean, West Virginia.

Pauley, T.K. and M.B. Watson. 2005. *Aneides aeneus* - Green Salamander. In: M. Lannoo, editor. Amphibian Declines: The Conservation Status of United States Species. University of California Press, Berkeley.

Pennsylvania Game Commission. Undated. Pennsylvania's Comprehensive Wildlife Conservation Strategy*, available at* http://www.portal.state.pa.us/portal/server.pt/document/841548/section_x_%E2%80%93_cwcs-priority_species_%282%29?qid=09476744&rank=2 (last visited Dec. 8, 2011).

Pennsylvania Natural Heritage Program. Undated, *available at* http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=14&ved=0CDYQFjADOAo&url=http%3A%2F%2Fwww.naturalheritage.state.pa.us%2Ffactsheets%2F10831.pdf&ei=G9vfTrzzHqbe2AXDvIX4BA&usg=AFQjCNHEDqguYBxlc_wLq0X9IL0rLUXjgA (last visited May 16, 2012).

Petranka, J. W. 1998. Salamanders of the United States and Canada. Smithsonian Institution Press, Washington, D.C.

Pomponio, J.R. 2009. Testimony of John "Randy" Pomponio, Director, Environmental Assessment and Innovation Division, U.S. Environmental Protection Agency Mid-Atlantic Region, before the Committee On Environment And Public Works Subcommittee on Water And Wildlife, U.S. Senate. June 25, 2009.

Redmond, W. H., and A. F. Scott. 1996. Atlas of amphibians in Tennessee. The Center for Field Biology, Austin Peay State University, Miscellaneous Publication Number 12. v + 94 pp.

Sessions, S.K. and J. Kezer. 1987. Cytogenetic evolution in the plethodontid salamander genus *Aneides*. Chromosoma 95: 17–30.

South Carolina Dept. of Natural Resources. 2010. Rare, Threatened, and Endangered Species and Communities Known to Occur in South Carolina, *available at* http://www.dnr.sc.gov/species/pdf/SC_state_wide.pdf (last visited Dec. 8, 2011).

Snyder, D. H. 1991. The green salamander (*Aneides aeneus*) in Tennessee and Kentucky, with comments on the Carolinas' Blue Ridge populations. Journal of the Tennessee Academy of Science 66: 165-169.

Starnes, L., and D. C. Gasper. 1995. Effects of surface mining on aquatic resources in North America. Fisheries Management and Ecology 20: 20-26.

Tennessee Wildlife Resources Agency. 2011. Endangered Species, *available at* http://www.tn.gov/twra/pdfs/endangered.pdf (last visited Dec. 8, 2011).

Townsend, P.A., D.P. Helmers, C.C. Kingdon, B.E. McNeil, K.M de Beurs, and K.N. Eshleman. 2009. Changes in the extent of surface mining and reclamation in the Central Appalachians detected using a 1976–2006 Landsat time series. Remote Sensing of Environment 113: 62–72.

U.S. Fish and Wildlife Service. 1987. Notice of status review—Blue Ridge Mountain population of the green salamander (Aneides aeneus) (Cope and Packard). U.S. Dep. Int., Fish and Wildl. Serv. Announcement. Endangered Species Field Off., Asheville, N.C. 6pp.

U.S. Fish and Wildlife Service (USFWS). 1997. Endangered and Threatened Wildlife and Plants; Determination of Endangered Status for the Cumberland Elktoe, Oyster Mussel, Cumberlandian Combshell, Purple Bean, and Rough Rabbitsfoot, Final Rule. 62 FR 1647. January 10, 1997.

U.S. Forest Service. 2004. Chattahoochee-Oconee National Forests Land and Resource Management Plan, *available at* http://www.fs.usda.gov/detail/conf/landmanagement/planning/?cid=fsm9_029115 (last visited Dec. 8, 2011).

U.S. Environmental Protection Agency (EPA). 2005. Mountaintop Mining/Valley Fills in Appalachia Final Programmatic Environmental Impact Statement. October 2005. EPA 9-03-R-05002.

Virginia Department of Game and Inland Fisheries. 2011. Special Legal Status Faunal Species in Virginia, *available at* http://www.dgif.virginia.gov/wildlife/virginiatescspecies.pdf (last visited Dec. 7, 2011).

Waldron, J.L. and W. Jeffrey Humphries. 2005. Arboreal Habitat Use by the Green Salamander, Aneides aeneus, in South Carolina. Journal of Herpetology 39(3): 486-492.

Waldron, J.L. and Thomas K. Pauley. 2007. Green Salamander (*Aneides Aeneus*) Growth and Age at Reproductive Maturity. Journal of Herpetology 41(4): 638-644.

Ward, K. Jr. 2009. Mining: West Virginia mountains go unrestored—OSM report. Charleston Gazette July 27, 2009.

West Virginia Dept. of Natural Resources. 2007. Rare, threatened, and endangered animals, *available at* http://www.wvdnr.gov/Wildlife/documents/Animals2007.pdf (last visited Dec. 8, 2011).

Williams, J. M. 2003. Impacts on Terrestrial and Streamside Herpetofauna by Mountaintop Removal Mining in Southern West Virginia. Davis College of Agriculture, Forestry, and Consumer Sciences. West Virginia University.

Williams, K.L. and R.E. Gordon. 1961. Natural dispersal of the salamander *Aneides aeneus*. Copeia 1961:353.

Wilson, C. 2001. Green Salamander (Aneides aeneus). Chatooga Quarterly, *available at* http://www.chattoogariver.org/Articles/2001SS/GreenSalamander.htm (last visited Dec. 7, 2011).

Wilson, C.R. 2003. Woody and arboreal habitats of the Green Salamander (*Aneides aeneus*) in the Blue Ridge Mountains, Contempory Herpetology 2, *available at* http://www.contemporaryherpetology.org/ch/2003/2/ (last visited Dec. 7, 2011).

Wood, D. 2009. Memorandum to J. Bailey, J. Wirts, and P. Campbell. Evidence for Secretary Randy Huffman. June 30, 2009. West Virginia Department of Environmental Protection, Charleston, WV.

Scientific Name:

Rhyacotriton cascadae

Common Name:

Cascade Torrent Salamander

G Rank:

G3

IUCN Red List:

Near Threatened

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

The Cascade torrent salamander is found on the west slope of the Cascade Mountains from just north of Mount Saint Helens, Skamania County, Washington, south to northeastern Lane County, Oregon (Good and Wake 1992). Within this area, the species is patchily distributed (Howell and Maggiulli 2011). The patchy distribution and small range directly limit the viability of the Cascade torrent salamander.

Habitat:

These salamanders generally are found in coniferous forests in small, cold mountain streams and spring seepages (Hammerson 2004). Physical features of stream habitats appear to have an important influence on the distribution and abundance of torrent salamanders at multiple spatial scales (Russell et al. 2005).

Larvae often occur under stones in shaded streams (Hammerson 2004). Adults also inhabit these streams or streamsides in saturated moss-covered talus, or under rocks in splash zone (Hammerson 2004). Two *Rhyacotriton* nests were found in deep, narrow rock crevices; eggs were lying in cold, slow-moving water (Nussbaum et al. 1983).

Rhyacotriton species occur primarily in older forest sites because required microclimatic and microhabitat conditions generally exist only in older forests (Welsh 1990). *Rhyacotriton* are among the most, if not the most, desiccation intolerant salamander genus known (Ray 1958). Desiccation intolerance is likely linked to high dependence on skin surfaces for oxygen uptake (average = 74 percent) because their lungs are highly reduced (Whitford and Hutchison 1966).

Biology and Taxonomy:

Courtship has not been observed for any species of *Rhyacotriton* (Petranka 1998). But based on the presence of spermatophores in the cloacae of females, it appears that courtship and mating occur over most of the year, concentrated in the fall and spring months (AmphibiaWeb 2012). Females may oviposit at any time of the year, but tend to lay in late spring (Nussbaum and Tait 1977). Average clutch size is eight (Nussbaum and Tait 1977, Good and Wake 1992), and there is apparently no attendance of the developing eggs (Nussbaum 1969, Nussbaum and Tait 1977). Clutch frequency is once per year (Nussbaum and Tait 1977, Nussbaum et al. 1983). Eggs laid in spring hatch five to six months later (Nussbaum 1969).

Information on diet of the Cascade torrent salamander is not available, but adults and juveniles likely feed on a mixture of aquatic and semiaquatic invertebrates (AmphibiaWeb 2012). Likewise, natural predators have not been documented but larval and adult Pacific giant salamanders *(Dicamptodon)*, as well as garter snakes *(Thamnophis)* are probably important (AmphibiaWeb 2012). *Rhyacotriton* respond to attacks by coiling the body and raising and undulating the tail, which contains poison glands (Nussbaum et al. 1983). See Nijhuis and Kaplan (1998), Hayes (2005), and Howell and Maggiulli (2011) for additional life history information.

The genus *Rhyacotriton* previously contained a single species with two subspecies, *R. o. olympicus* and *R. o. variegatus.* Genetic studies revealed substantial variation and subdivision throughout the range and the single species was split into four species: *R. olympicus, R. variegatus, R. cascadae,* and *R. kezeri* (Good et al. 1987, Good and Wake 1992).

Population Status:

The Cascade torrent salamander qualifies for endangered species protection because its populations are declining and experiencing well documented ongoing habitat loss from logging and road building. Effects of timber harvest on this species have been a concern for many years (Kroll 2009). Specifically, there is concern that the rapid rate of conversion of mature and old-growth forests to young stands as a function of timber harvest are limiting habitat quality through increased microhabitat temperatures and sedimentation, and that local extirpation is resulting from these changes (Corn and Bury 1989, Howell and Maggiulli 2011). The IUCN Red List ranks the species as Near Threatened but explains that it is close to qualifying for Vulnerable.

NatureServe (2011) suggests that the salamanders may be relatively stable in extent of occurrence but are experiencing declines in population size, area of occurrence, and number/condition of occurrences.

THREATS

Habitat alteration and destruction:

The main threats to the Cascade torrent salamander are forest management activities that increase water temperature, turbidity, peak flow or debris flow events, and habitat degradation and fragmentation (Crisafulli et al. 2005, Howell and Maggiulli 2011).

Timber management

The Cascade torrent salamander is found on on mixture of state, private, and federal lands where intensive timber production occurs. Loss of mature forests – upon which the salamander depends – is occurring across the range of the species, especially in northwest Oregon and southwest Washington, where state and private lands predominate. To be sure, in the western Cascades, old-growth forests have been reduced from 40-70 percent of the landscape in presettlement times, to just 13-18 percent today (National Research Council 2000).

Timber harvest is a threat because timber harvest negatively affects *Rhyacotriton* salamanders more than it does many other amphibians (Bury and Corn 1988, Corn and Bury 1989, Welsh and Lind 1996). Sedimentation of streams can lead to asphyxiation of embryos and larvae as well as a degradation of overwintering habitat that likely results in local extinctions (McAllister 1992). In addition, forest management practices that alter stream temperatures are harmful to the salamanders because the delivery of oxygen to their tissues appears limited at higher temperatures (Mullen et al. 2006). Logging also eliminates suitable habitat (Bury 1983, Welsh and Lind 1991, Good and Wake 1992, Petranka 1998).

In the Cascades of southern Washington, the Cascade torrent salamander was more often found in forests older than 25 years, and despite the presence of assumed suitable stream conditions, populations were found at low levels in streams surrounded by young forests (Steele et al. 2003). In addition, Pollett (2005) found that Cascade torrent salamander density was positively correlated with the percent of 80-year-old forest cover per drainage and number of riparian trees greater than 64 cm dbh.

Moreover, MacCracken et al. (2006) found that captures of Cascade torrent salamanders were lowest in 0 to 24-y-old stands (see also Kroll et al. 2008). In addition, captures were greatest in streams with temperatures less than 9 degrees Celcius (and stand age was not related to water temperature). Captures were greatest in streams in unmanaged stands and lowest in streams in clearcuts without buffers (MacCracken et al. 2006). Pollett and others (2011) also found Cascade torrent salamander densities were 7-fold lower in streams in managed forests than in streams in unharvested forest. In addition, the species was less abundant in unbuffered streams than streams with buffers or in second-growth forest. Cascade torrent salamanders were nearly absent from streams where temperatures exceeded 14 degrees Celcius for more than 35 consecutive hours (Pollett et al. 2011).

<u>Culverts and Roads</u>

Culverts and roads can have multiple effects on the Cascade torrent salamander (Howell and Maggiulli 2011). First, they are likely sources of erosion that result in stream sedimentation (Howell and Maggiulli 2011). Second, roads likely pose barriers to amphibian movement, and in the context of roads crossing streams, their culverts likely pose barriers (Howell and Maggiulli 2011). An inability to disperse puts populations at risk because it limits gene flow and the ability to recolonize after disturbance (Jackson 2003). Specifically, perched culverts are problematic due to loss of substrate continuity and increased velocity of water above a surface that does not present any natural characteristics, such as instream structures, substrate, or quiet pools, which would facilitate animal movement (Howell and Maggiulli 2011). Unfortunately, inventories to assess passage problems still focus on fish-bearing streams (USDI 2004) and likely does not include headwater segments, which are more likely to have Cascade torrent salamander present (Howell and Maggiulli 2011).

<u>Recreational Activities</u>

It is likely that trails made by humans, pollutants from vehicles (at stream/road crossings), and inputs of salt and sand from ski areas has negative effects to the species (Howell and Maggiulli 2011). None of these have been studied in relation to any of the torrent salamanders however.

Disease or predation:

One species of monogenoidean fluke, to date unique to *Rhyacotriton* (Kristsky et al. 1993), has been recorded from Cascade torrent salamanders, but it is unlikely to be a threat.

Inadequacy of existing regulatory mechanisms:

The Cascade torrent salamander occurs on a mix of private and government lands: state lands (39 percent), National Forests (37 percent), private lands (industrial and non-industrial, 14 percent), and BLM lands (10 percent) (Howell and Maggiulli 2011). Without federal protection, habitat on private lands is likely to be lost due to timber management activities or development. And even within those areas under federal or state ownership, the salamanders are likely subject to harmful timber management activities as this species avoids young forests.

Cascade torrent salamanders have state Sensitive status in Oregon (Oregon Dept. of Fish and Wildlife 2008). The salamanders are also BLM and U.S. Forest Service sensitive species in Washington, but not Oregon (Howell and Maggiulli (2011). In Washington, the salamander is a species of special concern (Washington Dept. of Fish and Wildlife 2012). Hayes (2005) explains that a sensitive species designation does not provide any legal protections, but it is applied to species for which serious concerns exist related to habitat loss to increase awareness among resource protection agencies.

Across ownerships, there is concern also that headwater streams, seeps, and springs (all habitats presumed to be occupied) lack adequate protection (Hayes 2005). Hayes (2005) explains that some efforts are being made to study the impacts of timber management on the salamanders:

> In 2000, scientists representing private timber companies, Native American tribes, and state and federal resource agencies assessed the risk of forest management activities to stream-associated vertebrate species. This work, done in preparation to the Washington Forest and Fish Agreement (FFA), concluded that seven species of amphibians (including all three species of torrent salamanders in Washington State) were at high risk of local extirpation from forest management. One outcome of identifying species at risk and including them in the list of species protected under new forest practice rules was that those species would be studied as part of an innovative, ambitious adaptive management program. The goal of FFA adaptive management is to determine whether new riparian buffer prescriptions designed for headwater streams are effective in protecting resources to which they are linked, including local populations of *Rhyacotriton*. To date, this research has identified the most appropriate sampling methods for landscape detection and relative abundance assessment of Cascade torrent salamanders and the other stream-associated amphibians. These methods will be essential for manipulative studies to test the effectiveness of forest practice rules in protecting *Rhyacotriton*.

These efforts are likely to lead to important research on the conservation needs of the salamanders but no commitments have been made to ensure that the recommendations from these studies are actually implemented. The FFA is wholly voluntary and is therefore inadequate to preserve the salamanders.

Since 1994, on National Forest System and Bureau of Land Management lands within the range of this species, the Northwest Forest Plan (NWFP; USDA/USDI 1994) has guided the delineation of Riparian Reserves, which provide different widths of management for five categories of streams or water bodies (Howell and Maggiulli 2011). These numbers do not represent areas where no timber harvest will occur; they are, however, areas where activities are regulated so that the needs of riparian-dependent resources receive primary emphasis But over half of the salamander's range is found on state and private lands, where the NWFP does not apply.

Efforts to protect the salamanders on state and private lands are largely absent or inadequate. Vesely and McComb (2003) explain that riparian buffer strip widths currently required by state forest practices regulations are likely insufficient to ensure that amphibian communities in managed stands remain as diverse as in unlogged forests. Pollette (2003) explains that most buffers are as narrow as 50 feet and that the effectiveness of buffers this size is largely untested on perennial non-fishbearing streams. To be sure, there is a general lack of information on the most effective buffer widths to protect *Rhyacotriton* spp. (Petranka 1998, Pollett 2003, Bury 2008, Pollett et al. 2011). Federal protection would likely provide additional resources to address these research needs and result in implementation of more effective buffer regimes.

Other factors:

Isolation

Some populations are isolated by intervening areas of unsuitable habitat, and these are then vulnerable to extirpation through natural processes exacerbated by timber harvest (especially of old growth stands on north-facing slopes) (Hammerson 2004). Small, isolated populations are more susceptible to extirpations due to stochastic events, human impacts, and environmental factors (Soulé 1987, Begone et al. 1990). Lack of gene flow is likely to cause loss of genetic variability due to random genetic drift (Wright 1931) and inbreeding depression may occur (Franklin 1980). Once these populations are extirpated, their isolation precludes genetic interchange and recolonization of habitat.

Climate change

Concerns for water availability from snow melt and headwater stream flows, especially in the central and southern Cascade Range have been suggested (Suzuki and Olson, submitted, as cited in Howell and Maggiulli 2011). It is likely that shifts in the timing of runoff will result in increased summer drought and a decrease in suitable headwater habitat (Halofsky et al., in press, as cited in Howell and Maggiulli 2011). Torrent salamanders are very tied to aquatic environments due to their intolerance to desiccation (Ray 1958) and for this reason are likely quite vulnerable to changing water regimes from climate change.

Chemical applications

It is possible that broad-scale herbicide treatments applied to suppress the shrub layer on forest lands after harvest could negatively impact Cascade torrent salamanders (Howell and Maggiulli 2011). In addition, episodic release of chemicals trapped in snowmelt is likely to have some direct and indirect effects on the salamanders, particularly where they accumulate in the foothills of the Cascade Range (Howell and Maggiulli 2011). On federal lands, herbicides used for work such as eradicating and minimizing the spread of invasive plant species would be the chemicals most likely to impact amphibians (Howell and Maggiulli 2011).

References:

AmphibiaWeb: Information on amphibian biology and conservation. [web application]. 2012. Berkeley, California: AmphibiaWeb, *available at*: http://amphibiaweb.org (last accessed June 21, 2012).

Begone, M., J. L. Harper, and C. R. Townsend. 1990. Ecology: Individuals, populations, and communities. Blackwell Scientific Publications.

Bury, R. B. 1983. Differences in amphibian populations in logged and old growth redwood forest. Northwest Science 57: 167-178.

Bury, R.B. 2008. Low thermal tolerances of stream amphibians in the Pacific Northwest: Implications for riparian and forest management. Applied Herpetology 5: 63-74.

Bury, R.B. and P.S. Corn. 1988. Responses of aquatic and streamside amphibians to timber harvest: a review. In: Raedaeke, K. (ed.), Streamside management: riparian wildlife and forestry interactions, pp. 165-181. Univ. Washington.

Corn, P.S. and R.B. Bury. 1989. Logging in western Oregon: responses of headwater habitats and stream amphibians. Forest Ecology and Management: 39-57.

Crisafulli, C.M., L.S. Trippe, C.P. Hawkins, and J.A. MacMahon. 2005. Amphibian Responses to the 1980 Eruption of Mount St. Helens, Chapter 13 in Ecological Responses to the 1980 Eruption of Mt. St. Helens, V.H. Dale, F.J. Swanson, and C.M. Crisafulli, Eds., Springer Science+Business Media, Inc., New York, NY.

Franklin, I. R. 1980. Evolutionary changes in small populations. In Conservation Biology: An evolutionary-ecological perspective (ed. a. B. A. W. M. E. Soule). pp. 135-149.

Good, D. A., and D. B. Wake. 1992. Geographic variation and speciation in the torrent salamanders of the genus *Rhyacotriton* (Caudata: Rhyacotritonidae). Univ. California Publication Zoology 126:i-xii, 1-91.

Good, D.A., G.Z. Wurst and D.B. Wake. 1987. Patterns of geographic variation in allozymes of the Olympic salamander, *Rhyacotriton olympicus* (Caudata: Dicamptodontidae). Fieldiana, Zoology, New Series, Number 32, Field Museum of Natural History, Chicago, Illinois.

Hammerson, G. 2004. *Rhyacotriton cascadae*. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.2. <www.iucnredlist.org>. Downloaded on 06 February 2012.

Hayes, M.P. 2005. In M. Lannoo, editor. Amphibian declines: the conservation status of United States species. University of California Press, Berkeley.

Howell, B. and N. Maggiulli. 2011. Conservation Assessment for the Cascade Torrent Salamander (*Rhyacotriton cascadae*), *available at* www.fs.fed.us/r6/sfpnw/issssp/documents/inventories/ca-ha-rhyacotriton-cascadae-2011-02.docx (last visited Feb. 6, 2012).

Jackson, S.D. 2003. Ecological considerations in the design of river and stream crossings. Pages 20–29. In: C. L. Irwin, P. Garrett, and K. P. McDermott. Eds. Proceedings of the international conference on ecology and transportation. North Carolina State University, Center for Transportation and the Environment, Raleigh, NC.

Jones, L.L.C., W. P. Leonard, and D. H. Olson, editors. 2005. Amphibians of the Pacific Northwest. Seattle Audubon Society, Seattle, Washington. xii + 227 pp.

Kritsky, D.C., E.P. Hoberg, and K.B. Aubry. 1993. Lagarocotyle salamandrae N. Gen., N. Sp. (Monogenoidea, Polyonchoinea, Lagarocotylidea N. Ord.) from the cloaca of Rhyacotriton cascadae Good and Wake (Caudata, Rhyacotritonidae) in Washington state. Journal of Parasitology 79(3): 322-330.

Kroll, A.J. 2009. Sources of uncertainty in stream-associated amphibian ecology and responses to forest management in the Pacific Northwest USA: a review. Forest Ecology and Management 257: 1188–1199.

Kroll, A.J., K. Risenhoover, T. McBride, E. Beach, B.J. Kernohan, J. Light, and J. Bach. 2008. Factors influencing stream occupancy and detection probability parameters of stream-associated amphibians in commercial forests of Oregon and Washington, USA. Forest Ecology and Management 255: 3726-3735.

MacCracken, James G., Graig A. Steele, and Kathleen M. Pollett. 2006. Effects of forest practices on cascade torrent salamanders in Washington. Northwestern Naturalist 87(2): 178.

McAllister, 1992. Amphibian management standards. Department of Wildlife, State of Washington.

Mullen, L., J. Irwin, and S.R. Wagner. 2006. Respiration constraints of pacific northwest forest-associated salamanders: implications for stream management. Northwestern Naturalist 87(2): 180.

National Research Council, Committee on Environmental Issues in Pacific Northwest Forest Management. 2000. Environmental Issues in Pacific Northwest Forest Management. National Academy Press, Washington, DC. 259 pp., *available at* http://www.nap.edu/openbook.php?record_id=4983&page=7 (last visited July 2, 2012).

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia. Available http://www.natureserve.org/explorer (Accessed: February 6, 2012).

Nijhuis, Michelle J., and Robert H. Kaplan. 1998. Movement patterns and life history characteristics in a population of the Cascade torrent salamander (Rhyacotriton cascadae) in the Columbia River Gorge, Oregon. Journal of Herpetology 32(2): 301.

Nussbaum, R. 1969. A nest site of the Olympic Salamander, *Rhyacotriton olympicus* (Gaige). Herpetologica 25: 277-278.

Nussbaum, R., and C.K. Tait. 1977. Aspects of the life history and ecology of the Olympic Salamander, Rhyacotriton olympicus. American Midland Naturalist 98: 176-199.

Nussbaum, R.A., E.D. Brodie, Jr., and R.M. Storm. 1983. Amphibians and Reptiles of the Pacific Northwest. University Press of Idaho, Moscow, Idaho. 332 pp.

Oregon Dept. of Fish and Wildlife. 2008. Oregon Department of Fish and Wildlife Sensitive Species: Frequently Asked Questions and Sensitive Species List, *available at* http://www.dfw.state.or.us/wildlife/diversity/species/docs/SSL_by_taxon.pdf (Feb. 8, 2012).

Petranka, J.W. 1998. Salamanders of the United States and Canada. Smithsonian Institution Press, Washington, D.C.

Pollett, K.L. 2003. Responses of stream breeding amphibians to riparian buffers in headwater streams: ameliorating the effects of regeneration harvest. Northwestern Naturalist 84(2): 111.

Pollett, K.L. 2005. Responses of stream-breeding amphibians to riparian buffers in headwater streams: ameliorating the effects of regeneration harvest. Master's Thesis, Utah State University.

Pollett, K.L., J.G. MacCrackenb, and J.A. MacMahona. 2011. Stream buffers ameliorate the effects of timber harvest on amphibians in the Cascade Range of Southern Washington, USA. Forest Ecology & Management 260(6): 1083-88.

Ray, C. 1958. Vital limits and rates of desiccation in salamanders. Ecology 39:75-83.

Russell, Kevin R., T.J. Mabee, M. Cole, and M. Rochelle. 2005. Evaluating biotic and abiotic influences on torrent salamanders in managed forests of western Oregon. Wildlife Society Bulletin 33(4): 1413-1424.

Soulé, M. 1987. Viable populations for conservation. Cambridge University Press, Great Britain. 189 pp.

Steele, C.A., E.D. Brodie, and J.G. MacCracken. 2003. Relationships between abundance of Cascade torrent salamanders and forest age. Journal of Wildlife Management 67(2): 447-453.

U.S. Department of the Interior. 2004. Biological Opinion for USDA Forest Service Fish Passage Restoration Activities in Eastern Oregon and Washington 2004-2008. Portland, OR: Oregon Fish and Wildlife Office (USFWS) and Lacey WA: Western Washington Fish and Wildlife Office.

U.S. Department of Agriculture and U.S. Department of the Interior (USDA and USDI). 1994. Record of decision for amendments to Forest Service and Bureau of Land Management planning documents within the range of the northern spotted owl: Standards and guidelines for management of habitat for late-successional and old-growth forest related species within the range of the northern spotted owl. Portland, OR: Interagency SEIS Team, 1994.

Washington Dept. of Fish and Wildlife. 2012. Washington State Species of Concern Lists, *available at* http://wdfw.wa.gov/conservation/endangered/list/Amphibian/ (last visited Feb. 6, 2012).

Welsh, H. H., Jr. 1990. Relictual amphibians and old-growth forests. Conservation Biology 4:309-19.

Welsh, H.H., Jr. and A.J. Lind. 1996. Habitat correlates of the southern torrent salamander, *Rhyacotriton variegatus* (Caudata: Rhyacotritonidae), in northwestern California. Journal of Herpetology: 385-398.

Whitford, W.G. and V.H. Hutchison. 1966. Cutaneous and pulmonary gas exchange in ambystomatid salamanders. Copeia 1966: 573–577.

Wright, S. 1931. Evolution in Mendelian populations. Genetics 16: 97-159.

Scientific Name:

Rhyacotriton kezeri

Common Name:

Columbia Torrent Salamander

G Rank:

G3

IUCN Red List:

Near Threatened

<u>NATURAL HISTORY, BIOLOGY, AND STATUS</u>

Range:

The Columbia torrent salamanders are restricted to coastal and near-coastal regions of northwestern Oregon and southwestern Washington, from the Little Nestucca River system (Tillamook County, Oregon) in the south to the Chehalis River (Gray's Harbor County, Washington) in the north (Good and Wake 1992, McAllister 1995). Much of this range is private and state lands under industrial forest management.

Habitat:

It can be found in coastal coniferous forests in small, cold mountain streams and spring seepages (Hammerson 2004). Larvae often occur under stones in shaded streams; adults also inhabit these streams or streamsides in saturated moss-covered talus, or under rocks in the splash zone (Hammerson 2004). Two *Rhyacotriton* nests were found in deep, narrow rock crevices, and the eggs were lying in cold, slow-moving water (Nussbaum et al. 1983). Russell and others (2002) describe three additional nests.

This species is found primarily in older forest sites since the required microclimatic and microhabitat conditions generally exist only in older forests (Welsh 1990, see also Wallas 2004). However, Russell et al. (2004) found that at the landscape level, occupancy and relative abundance of Columbia torrent salamanders were not related to age or composition of riparian vegetation but rather to abiotic landform features (see also Russell 2001). They found that variation in physical features of stream habitats may have an important influence on distribution and abundance of Columbia torrent salamanders at multiple spatial scales.

Similarly, Wilkins and Peterson (2000) found that the likelihood of habitat occupancy by torrent salamanders increased as channel gradient increased and basin area decreased. When adjusted for basin area, torrent salamander abundance increased as the proportion of the active channel with flowing water decreased, and at more northerly aspects. In addition, Hayes et al. (2003) found that Columbia torrent salamander density was significantly greater in waterfalls than other reach types.

Through the generation of a coarser-grained substrate structure, balsalt substrates are thought to provide more suitable habitat than sedimentary substrates (Wilkins and Peterson, 2000), but competence of the substrate, rather than the formation per se, may be the important factor in habitat quality, all else being equal (see Dupuis et al. 2000).

Biology and Taxonomy:

Courtship has not been observed for any species of *Rhyacotriton* (Petranka 1998). Females may oviposit at any time of the year, but tend to lay in late spring (Nussbaum and Tait 1977, Welsh and Lind 1992). Egg deposition sites are in seeps at the heads of springs (Nussbaum 1969). The Columbia torrent salamander is the only species of *Rhyacotriton* for which eggs have been found in nature (AmphibiaWeb 2012). Thirty-two eggs were found in the nest, and based on estimates of the average clutch size, this was likely the reproductive effort of at least three females (Nussbaum 1969). Females apparently do not attend the developing eggs (AmphibiaWeb 2012). Clutch frequency is once per year (Nussbaum and Tait 1977, Nussbaum et al. 1983). Eggs laid in spring hatch 5-6 months later (Nussbaum 1969).

Generally, less is known about the biology of Columbia torrent salamander than of other species of *Rhyacotriton.* Species of *Rhyacotriton* are likely to be similar in many ways, for example with respect to diet and potential predators. Predation on the eggs of Columbia torrent salamander by giant salamanders *(Dicamptodon)* has been documented (Nussbaum 1969). For additional life history information, see Hayes and Quinn (2005), Hicks and others (2008).

Until recently the genus *Rhyacotriton* contained a single species with two subspecies, *R. o. olympicus* and *R. o. variegatus.* Genetic studies revealed substantial variation and subdivision throughout the range and the single species was split into four species: *R. olympicus, R. variegatus, R. kezeri,* and *R.cascadae* (Good et al. 1987, Good and Wake 1992).

Population Status:

The Columbia torrent salamander qualifies for endangered status because it is at great risk of habitat loss from timber harvest across its small range and is suffering from population declines. NatureServe (2011) explains that it is likely relatively stable in extent of occurrence but is declining in population size, area of occurrence, and number/condition of occurrences. The species is considered Near Threatened by the IUCN because of its small range (Hammerson 2004).

THREATS

Habitat alteration and destruction:

Over 95 percent of the known distribution of Columbia torrent salamanders lies within private and state lands subject to intensive timber harvest (Good and Wake 1992, McAllister 1995). Loss of mature forests – upon which the salamander depends – is occurring across the range of the species. To be sure, in the western Cascades, old-growth forests have been reduced from 40-70 percent of the landscape in presettlement times, to just 13-18 percent today (National Research Council 2000).

Torrent salamanders (*Rhyacotriton* spp.) are among Pacific Northwest stream amphibians reported to be most at risk from timber harvest (Bury 1983, Bury and Corn 1988, Corn and Bury 1989, Good and Wake 1992, Welsh and Lind 1991, Welsh and Lind 1996). Timber harvest is thought to extirpate torrent salamanders from streams for decades by: (1) depositing sediments that degrade microhabitats and (2) removing canopy cover resulting in elevated stream temperatures (Bury and Corn 1988, Corn and Bury 1989, Welsh and Lind 1996, Dupuis et al. 2000). Sedimentation of streams can lead to asphyxiation of embryos and larvae as well as a degradation of overwintering habitat that is likely to result in local extinctions (McAllister 1992). In addition, forest management practices that alter stream temperatures is harmful to the salamanders because the delivery of oxygen to their tissues appears limited at higher temperatures (Mullen et al. 2006).

In a study of the effects of timber harvest on terrestrial salamanders in southwest Washington, Columbia torrent salamanders were captured only in 45 to 60 year-old forested areas and not in adjacent areas clearcut two to five years previously (Grialou et al. 2000). Kroll et al. (2008) found similar results for *Rhyacotriton* spp. on managed lands in Oregon and Washington with occupancy being lowest in the youngest and oldest sampled stands.

Russell et al. (2004) explains that, in areas where abiotic features have much greater influence on habitat than human-induced changes in vegetation, existing practices designed to conserve headwater amphibian populations (e.g., riparian buffers) may be well-intentioned but ineffective (Sutherland and Bunnell 2001). Further, they suggest that low-gradient, marine sediment streams could represent less favorable habitats for torrent salamanders regardless of management regime. If this is the case, torrent salamanders are not likely to benefit from protective measures implemented in low-gradient marine sediment streams (Sutherland and Bunnell 2001).

Hayes and Quinn (2005) explains that some efforts are being made to study the impacts of timber management on the salamanders:

> In 2000, scientists representing private timber companies, Native American tribes, and state and federal resource agencies assessed the risk of forest management activities to stream-associated vertebrate species. This work, done in preparation to the Washington Forest and Fish Agreement (FFA), concluded that seven species of amphibians (including all three species of torrent salamanders in Washington State) were at high risk of local extirpation from forest management. One outcome of identifying species at risk and including them in the list of species protected under new forest practice rules was that those species would be studied as part of an innovative, ambitious adaptive management program. The goal of FFA adaptive management is to determine whether new riparian buffer prescriptions designed for headwater streams are effective in protecting resources to which they are linked, including local populations of *Rhyacotriton*. To date, this research has identified the most appropriate sampling methods for landscape detection and relative abundance assessment of Cascade torrent salamanders and the other stream-associated amphibians. These methods will be essential for manipulative studies to test the effectiveness of forest practice rules in protecting *Rhyacotriton*.

These efforts are likely to lead to important research on the conservation needs of the salamanders but no commitments have been made to ensure that the recommendations from these studies are actually implemented. The FFA is wholly voluntary and is therefore inadequate to preserve the salamanders.

Inadequacy of existing regulatory mechanisms:

The species is a U.S. Fish and Wildlife Service Species of Concern (USFWS 2007). In Washington, the salamander is a species of special concern (Washington Dept. of Fish and Wildlife 2012). Columbia torrent salamanders also have state Sensitive status in Oregon (Oregon Dept. of Fish and Wildlife 2008). Hayes and Quinn (2005) explains that a sensitive species designation does not afford legal protections, but it is applied to species for which serious concerns exist related to habitat loss in order to increase awareness among resource protection agencies.

Although other torrent salamander species have significant ranges in federal lands with long-term management objectives for old-growth forest conditions (Tuchmann et al. 1996), over 95 percent of the known distribution of Columbia torrent salamanders lies within private and state lands subject to intensive timber harvest (Good and Wake 1992, McAllister 1995). With the vast majority of its range on private lands, the Columbia torrent salamanders are at great risk of extirpation without federal protection.

Riparian buffer strip widths currently required by state forest practices regulations are likely insufficient to ensure that amphibian communities in managed stands remain as diverse as in unlogged forests (Vesely and McComb 2002). Across ownerships, there is concern also that headwater streams, seeps, and springs (all habitats presumed to be occupied) lack adequate protection (Hayes and Quinn 2005).

Other factors:

Isolation

Most populations of torrent salamanders are isolated by intervening areas of unsuitable habitat, and these are then vulnerable to extirpation through natural processes exacerbated by timber harvest (especially of old growth stands on north-facing slopes) (Hammerson 2004). Recolonization of streams is expected to be rare because torrent salamanders are thought to have limited dispersal abilities and small home ranges (Nussbaum et al. 1983). As such, isolation is a threat.

Climate change

Torrent salamanders are very tied into aquatic environments due to their intolerance to desiccation (Ray 1958) and for this reason are likely quite vulnerable to changing water regimes from climate change.

References:

AmphibiaWeb: Information on amphibian biology and conservation. [web application]. 2012. Berkeley, California: AmphibiaWeb, *available at*: http://amphibiaweb.org (last accessed June 21, 2012).

Bury, R. B. 1983. Differences in amphibian populations in logged and old growth redwood forest. Northwest Science 57: 167-178.

Bury, R. B., and P. S. Corn. 1988. Responses of aquatic and streamside amphibians to timber harvest: a review. Pages 165-181 in Raedaeke, K., editor. Streamside management: riparian wildlife and forestry interactions. Univ. Washington.

Corn, P. S., and R. B. Bury. 1989. Logging in western Oregon: repsonses of headwater habitats and stream amphibians. Forest Ecology and Management 29: 39-57.

Dupuis, L. A., F. L. Bunnell, and P. A. Friele. 2000. Determinants of the tailed frog's range in British Columbia, Canada. Northwest Science 74: 109-115.

Good, D. A., and D. B. Wake. 1992. Geographic variation and speciation in the torrent salamanders of the genus *Rhyacotriton* (Caudata: Rhyacotritonidae). Univ. California Publication Zoology 126:i-xii, 1-91.

Good, D. A., G. Z. Wurst, and D. B. Wake. 1987. Patterns of geographic variation in allozymes of the Olympic salamander, *Rhyacotriton olympicus* (Caudata: Dicamptodontidae). Fieldiana Zoolgy 32 (1374). 15 pp.

Grialou, Julie A., Stephen D. West and R. Neal Wilkins. 2000. The Effects of Forest Clearcut Harvesting and Thinning on Terrestrial Salamanders. The Journal of Wildlife Management 64(1): 105-113.

Hammerson, G. 2004. *Rhyacotriton kezeri*. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.2. <www.iucnredlist.org>. Downloaded on 08 February 2012.

Hayes, Marc P., D.J. Dugger, T.L. Hicks, and T. Quinn. 2003. Headwater habitat variation: its relationship to stream amphibian distribution. Northwestern Naturalist 84(2): 100.

Hayes, M.P. and T. Quinn. 2005. Rhyacotriton kezeri - Columbia torrent salamander. In M. Lannoo, editor. Amphibian declines: the conservation status of United States species. University of California Press, Berkeley.

Hicks, T. L., D. E. Mangan, A. P. McIntyre, and M. P. Hayes. 2008. *Rhyacotriton kezeri* larval diet. Herpetological Review 39(4): 456-457.

Kroll, A.J., K. Risenhoover, T. McBride, E. Beach, B.J. Kernohan, J. Light, and J. Bach. 2008. Factors influencing stream occupancy and detection probability parameters of stream-associated amphibians in commercial forests of Oregon and Washington, USA. Forest Ecology and Management 255: 3726-3735.

McAllister, 1992. Amphibian management standards. Department of Wildlife, State of Washington.

McAllister, K.R. 1995. Distribution of amphibians and reptiles in Washington State. Northwest Fauna 3:81-112.

Mullen, L., J. Irwin, and S.R. Wagner. 2006. Respiration constraints of pacific northwest forest-associated salamanders: implications for stream management. Northwestern Naturalist 87(2): 180.

National Research Council, Committee on Environmental Issues in Pacific Northwest Forest Management. 2000. Environmental Issues in Pacific Northwest Forest Management. National Academy Press, Washington, DC. 259 pp., *available at* http://www.nap.edu/openbook.php?record_id=4983&page=7 (last visited July 2, 2012).

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia, *available at* http://www.natureserve.org/explorer (Accessed: February 8, 2012).

Nussbaum, R. 1969. A nest site of the Olympic Salamander, *Rhyacotriton olympicus* (Gaige). Herpetologica 25: 277-278.

Nussbaum, R., and C.K. Tait. 1977. Aspects of the life history and ecology of the Olympic Salamander, Rhyacotriton olympicus. American Midland Naturalist 98: 176-199.

Nussbaum, R.A., E.D. Brodie, Jr., and R.M. Storm. 1983. Amphibians and Reptiles of the Pacific Northwest. University Press of Idaho, Moscow, Idaho. 332 pp.

Oregon Dept. of Fish and Wildlife. 2008. Oregon Department of Fish and Wildlife Sensitive Species: Frequently Asked Questions and Sensitive Species List, *available at* http://www.dfw.state.or.us/wildlife/diversity/species/docs/SSL_by_taxon.pdf (Feb. 8, 2012).

Petranka, J. W. 1998. Salamanders of the United States and Canada. Smithsonian Institution Press, Washington and London.

Ray, C. 1958. Vital limits and rates of desiccation in salamanders. Ecology 39:75-83.

Russell, Kevin R. 2001. Distribution and habitat of Columbia torrent salamanders on young growth, managed forests of north coastal Oregon. Northwestern Naturalist 82(2): 80.

Russell, Kevin R., Alex A. Gonyaw, Justin D. Strom, Katherina E. Diemer and Kevin C. Murk. 2002. Three New Nests of the Columbia Torrent Salamander, Rhyacotriton kezeri, in Oregon with Observations of Nesting Behavior. Northwestern Naturalist 83(1): 19-22.

Russell, K.R., T.J. Mabee, M.B. Cole. 2004. Distribution and Habitat of Columbia Torrent Salamanders at Multiple Spatial Scales in Managed Forests of Northwestern Oregon. The Journal of Wildlife Management 68(2): 405-417.

Russell, Kevin R., Todd J. Mabee, Michael B. Cole, and Michael J. Rochelle. 2005. Evaluating biotic and abiotic influences on torrent salamanders in managed forests of western Oregon. Wildlife Society Bulletin 33(4): 1413-1424.

Stoddard, Margo A., and John P. Hayes. 2005. The influence of forest management on headwater stream amphibians at multiple spatial scales. Ecological Applications 15: 811–823.

Sutherland, D. and F . L. Bunnell. 2001. Cross-scale classification trees for assessing risks of forest practices to headwater stream amphibians. Pages 550-555 in D. H. Johnson and T. A. O'Neill, managing directors. Wildlife-habitat relationships in Oregon and Washington. Oregon State University Press, Corvallis, Oregon, USA.

Tuchmann, E. T., K. P. Connaughton, L. E. Freedman, and C . B. Moriwak. 1996. The Northwest Forest Plan: a report to the President and Congress. U.S. Office of Forestry and Economic Assistance, Portland, Oregon, USA.

U.S. Fish and Wildlife Service. 2007. Listed and proposed endangered and threatened species and critical habitat; candidate specis; and species of concern, *available at* http://www.fws.gov/wafwo/speciesmap/GRAYS%20HARBOR.html (last visited Feb. 8, 2012).

Vesely, D.G. and W.C. McComb. 2002. Salamander abundance and amphibian species richness in riparian buffer strips in the Oregon Coast Range. Forest Science 48: 291–297.

Wallas, H. 2004. Observations on Torrent Salamanders (Rhyacotriton) in Oregon and California, *available at* http://www.caudata.org/cc/articles/Rhyacotriton.shtml (last visited Feb. 8, 2012).

Washington Dept. of Fish and Wildlife. 2012. Washington State Species of Concern Lists, *available at* http://wdfw.wa.gov/conservation/endangered/list/Amphibian/ (last visited Feb. 6, 2012).

Welsh, H. H., Jr. and A.J. Lind. 1991. The structure of the herpetofaunal assemblage in the Douglas-fir/hardwood forests of northwestern California and southwestern Oregon. Wildlife and Vegetation of Unmanaged Douglas-fir Forests, USDA Forest Service, General Technical Report PNW-GTR-285. USDA Forest Service, 394-413.

Welsh, H. H., Jr., and A. J. Lind. 1996. Habitat correlates of the southern torrent salamander, *Rhyacotriton variegatus* (Caudata: Rhyacotritonidae) in northwestern California. Journal of Herpetology 30: 385-398.

Wilkins, R.N. and N.P. Peterson. 2000. Factors related to amphibian occurrence and abundance in headwater streams draining second-growth douglas-fir forests in southwestern Washington. Forest ecology and management 139(1-3): 79-91.

Scientific Name:

Rhyacotriton olympicus

Common Name:

Olympic Torrent Salamander

G Rank:

G3

IUCN Red List:

Vulnerable

NATURAL HISTORY, BIOLOGY, AND STATUS

Range:

This species can be found in the Olympic Peninsula in Clallam, Grays Harbor, Jefferson, and Mason counties, Washington, United States (Good and Wake 1992).

Habitat:

The Olympic torrent salamander can be found in coastal coniferous forests in small, cold mountain streams and spring seepages (Hammerson 2004). Larvae often occur under stones in shaded streams; adults also inhabit these streams or streamsides in saturated moss-covered talus, or under rocks in splash zone (Hammerson 2004). It is primarily found in older forest sites because required microclimatic and microhabitat conditions generally exist only in older forests (Welsh 1990).

Two *Rhyacotriton* were found in deep, narrow rock crevices; eggs were lying in cold, slow-moving water (Nussbaum et al. 1983). Leonard et al. (1993) state that "*R. olympicus* are nearly always seen in or very near cold, clear streams, seepages, or waterfalls. Their typical haunt is the splash zone, where a thin film of water runs between or under rocks." Adams and Bury (2002) found that unconsolidated surface geology can provide suitable habitat for stream amphibians, including Olympia torrent salamanders. A survey of Olympic National Park (Bury and Adams 2000) revealed that Olympic torrent salamanders were more abundant in streams with northerly aspects and steep gradients.

Biology and Taxonomy:

Most breeding of Olympic torrent salamanders occurs in spring and early summer (AmphibiaWeb 2012). The sexually active male may perform a tail-wagging display towards the female prior to spermatophore deposition, curling the tail and arching the back (Arnold 1977). Females lay single, unpigmented eggs in clusters of 2-16 (Stebbins and Lowe 1951, Stebbins 1985), but larger clusters may be found due to communal egg laying (Nussbaum 1969, Stebbins 1985). There is apparently no attendance of the developing eggs (Nussbaum 1969, Nussbaum and Tait 1977). Clutch frequency is once per year (Nussbaum et al. 1983). Larvae may take over three years to metamorphose, and metamorphosis usually occurs between 30 and 40cm SVL (Stebbins 1985, Nussbaum et al. 1983).

The larval diet may include a variety of aquatic invertebrates, and varies with availability and location (Nussbaum et al. 1983). The diet of metamorphosed *Rhyacotriton* includes aquatic and semi-aquatic invertebrates, as well as larval and adult beetles, flies, earthworms, snails and other invertebrates (Nussbaum et al. 1983).

Predators of *Rhyacotriton* probably include the giant salamander, *Dicamptodon,* and garter snakes, but have not been reported (Nussbaum et al. 1983). Large larvae and adults may exhibit a defensive behavior consisting of coiling the body and elevating and undulating the tail, exposing the bright yellow underside (Nussbaum et al. 1983). For additional information on life history, see Nussbaum and Tait (1977), Hayes and Jones (2005), and Howell and Roberts (2008).

Until recently the genus *Rhyacotriton* contained a single species with two subspecies, *R. o. olympicus* and *R. o. variegatus.* Genetic studies revealed substantial variation and subdivision throughout the range and the single species was split into four species: *R. olympicus, R. variegatus, R. kezeri,* and *R.cascadae* (Good et al. 1987, Good and Wake 1992).

Population Status:

The Olympic torrent salamander qualifies for federal protection because it is at great risk of habitat loss from timber harvest across its small and fragmented range, where it is experiencing continuing population declines. NatureServe (2011) noted declines in population size, area of occurrence, and number/condition of occurrences (see also Howell and Roberts 2008). The IUCN Red List lists it as Vulnerable because of its small range, severely fragmented distribution and continuing decline in the extent and quality of its forest habitat in Washington State (Hammerson 2004).

THREATS

Habitat alteration and destruction:

Timber harvest

Torrent salamanders (*Rhyacotriton* spp.) are among Pacific Northwest stream amphibians reported to be most at risk from timber harvest (Bury and Corn 1988, Corn and Bury 1989, Welsh and Lind 1996). Timber harvest is thought to extirpate torrent salamanders from streams for decades by: (1) depositing sediments that degrade microhabitats and (2) removing canopy cover resulting in elevated stream temperatures (Bury and Corn 1988, Corn and Bury 1989, Welsh and Lind 1996, Dupuis et al. 2000). Sedimentation of streams can lead to asphyxiation of embryos and larvae as well as a degradation of overwintering habitat that is likely to result in local extinctions (McAllister 1992). In addition, forest management practices that alter stream temperatures is harmful to the salamanders because the delivery of oxygen to their tissues appears limited at higher temperatures (Mullen et al. 2006). AmphibiaWeb (2012) explains that the Olympic torrent salamander is "associated with old-growth forest, and is virtually absent from recently logged areas. (Good and Wake 1992; Stebbins and Lowe 1952)."

Using surveys conducted on 20 x 40 m plots, Vesely and McComb (2002) showed that torrent salamanders were sensitive to forest practices in riparian areas, and that riparian buffer strips approximately 43 m wide would support a total salamander abundance (including torrent salamanders) similar to that in unlogged forests. Stoddard and Hayes (2005) studied the influence of forest management on torrent salamanders and found that when analyzed at an intermediate scale, presence of torrent salamanders was positively associated with presence of a 46-m band of forested habitat on each side of the stream, but no relationship was detected at the drainage scale.

Russell et al. (2004) explains that, in areas where abiotic features have much greater influence on habitat than human-induced changes in vegetation, existing practices designed to conserve headwater amphibian populations (e.g., riparian buffers) may be well-intentioned but ineffective (Sutherland and Bunnell 2001). Further, they suggest that low-gradient, marine sediment streams could represent less favorable habitats for torrent salamanders regardless of management regime. If this is the case, torrent salamanders are not likely to benefit from protective measures implemented in low-gradient marine sediment streams (Sutherland and Bunnell 2001).

Hayes and Jones (2005) explain that some efforts are being made to study the impacts of timber management on the salamanders:

> In 2000, scientists representing private timber companies, Native American tribes, and state and federal resource agencies assessed the risk of forest management activities to stream-associated vertebrate species. This work, done in preparation to the Washington Forest and Fish Agreement (FFA), concluded that seven species of amphibians (including all three species of torrent salamanders in Washington State) were at high risk of local extirpation from forest management. One outcome of identifying species at risk and including them in the list of

> species protected under new forest practice rules was that those species would be studied as part of an innovative, ambitious adaptive management program. The goal of FFA adaptive management is to determine whether new riparian buffer prescriptions designed for headwater streams are effective in protecting resources to which they are linked, including local populations of *Rhyacotriton*. To date, this research has identified the most appropriate sampling methods for landscape detection and relative abundance assessment of Cascade torrent salamanders and the other stream-associated amphibians. These methods will be essential for manipulative studies to test the effectiveness of forest practice rules in protecting *Rhyacotriton*.

These efforts are likely to lead to important research on the conservation needs of the salamanders but no commitments have been made to ensure that the recommendations from these studies are actually implemented. The FFA is wholly voluntary and is therefore inadequate to preserve the salamanders.

<u>Culverts and roads:</u>

Culverts and roads can pose barriers to amphibian movement and an inability to disperse puts populations at risk because it limits gene flow and the ability to recolonize after disturbance (Jackson 2003). As Howell and Roberts (2008) explain:

> [C]ulverts present barriers at outflow pipes where there are significant drops and where they have encouraged increased velocity of water above a surface that does not present any natural characteristics, such as instream structures or quiet pools, which would facilitate animal movement. Additionally, *Rhyacotriton olympicus*, given its close association to the stream channel and adjacent, saturated ground, may not likely move any significant distance upland to navigate around such barriers. These types of culverts have long been recognized as problems for fish and have only recently become more of a topic of concern for amphibians. Unfortunately, inventories to assess passage problems still focus on fish-bearing streams (USDI 2004), and may not include headwater segments, which are more likely to have *R. olympicus* present.

Inadequacy of existing regulatory mechanisms:

Approximately 39 percent of the range of *Rhyacotriton olympicus* is within federal ownership (1,556,328 acres) in the Olympic National Park and Olympic National Forest (with a minor amount, less than 1,500 acres on BLM lands). Within the acres on federal lands, approximately 65 percent are in congressional reserves (Olympic National Park and the five wildernesses on the Olympic National Forest) (Howell and Roberts 2008). As such, the majority of its range falls outside of these lands and is subjected to harmful timber management practices (see Howell and Roberts 2008). Conservation needs include retention of old-growth buffers around headwater streams (Petranka 1998).

Olympic torrent salamanders are state monitored in Washington (Washington Dept. of Fish and Wildlife 2012). This status reflects that the species is of conservation concern but provides no legal protections.

Other factors:

Isolation

Some populations of torrent salamanders are isolated by intervening areas of unsuitable habitat, and these are then vulnerable to extirpation through natural processes exacerbated by timber harvest (especially of old growth stands on north-facing slopes) (Hammerson 2004). Recolonization of streams is expected to be rare because torrent salamanders are thought to have limited dispersal abilities and small home ranges (Nussbaum et al. 1983). As such, isolation is a threat.

Climate change

Climate change is a threat because torrent salamanders are very tied to aquatic environments due to their intolerance to desiccation (Ray 1958) and for this reason are likely quite vulnerable to changing water regimes from climate change.

Pollutants

Salt and sand, as components of ski area management, however, may potentially enter the stream channel and affect the species (Howell and Roberts 2008).

References:

Adams, M.J. and R.B. Bury. 2002. The endemic headwater stream amphibians of the American Northwest: associations with environmental gradients in a large forested preserve. Global Ecology & Biogeography 11: 169–178

AmphibiaWeb: Information on amphibian biology and conservation. [web application]. 2012. Berkeley, California: AmphibiaWeb, *available at*: http://amphibiaweb.org (last accessed June 21, 2012).

Arnold, S.J. 1977. The evolution of courtship behavior in New World salamanders with some comments on Old World salamanders. The Reproductive Biology of Amphibians. D.H.Taylor and S.I. Guttman, eds., Plenum Press, New York, 141-183.

Bury, R.B. and M.J. Adams. 2000. Inventory and monitoring of amphibians in North Cascades and Olympic National Parks, 1995–1998. U.S.G.S. Forest and Rangeland Ecosystem Science Center in cooperation with Olympic National Park, Corvallis, Oregon.

Bury, R. B., and P. S. Corn. 1988. Responses of aquatic and streamside amphibians to timber harvest: a review. Pages 165-181 in Raedaeke, K., editor. Streamside management: riparian wildlife and forestry interactions. Univ. Washington.

Corn, P. S., and R. B. Bury. 1989. Logging in western Oregon: repsonses of headwater habitats and stream amphibians. Forest Ecology and Management 29: 39-57.

Dupuis, L. A., F. L. Bunnell, and P. A. Friele. 2000. Determinants of the tailed frog's range in British Columbia, Canada. Northwest Science 74: 109-115.

Good, D. A., and D. B. Wake. 1992. Geographic variation and speciation in the torrent salamanders of the genus *Rhyacotriton* (Caudata: Rhyacotritonidae). Univ. California Publication Zoology 126:i-xii, 1-91.

Good, D. A., G. Z. Wurst, and D. B. Wake. 1987. Patterns of geographic variation in allozymes of the Olympic salamander, *Rhyacotriton olympicus* (Caudata: Dicamptodontidae). Fieldiana Zoolgy No. 32 (1374). 15 pp.

Hammerson, G. 2004. Rhyacotriton olympicus. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.2. <www.iucnredlist.org>. Downloaded on 09 February 2012.

Hayes, M.P. and L.L.C. Jones. 2005. *Rhyacotriton olympicus* -- Olympic Torrent Salamander. In M. Lannoo, editor. Amphibian declines: the conservation status of United States species. University of California Press, Berkeley.

Howell, B.L. and C.R. Roberts. 2008. A Conservation Assessment for the Olympic Torrent Salamander (*Rhyacotriton olympicus*), *available at* http://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=1&ved=0CFIQFjAA&url=http%3A%2F%2Fwww.fs.fed.us%2Fr6%2Fsfpnw%2Fissssp%2Fdocuments%2Fplanning-docs%2Fca-ha-rhyacotriton-olympicus-2008-12.doc&ei=Ou7xT_fMDYW4rQGbgPGOAg&usg=AFQjCNFjuv530UNyBmGWn0Br1mbKcjxgeg&sig2=KVIcgdptzzpxHVR9TBEkOw (last visited July 2, 2012).

Kroll, A.J., K. Risenhoover, T. McBride, E. Beach, B.J. Kernohan, J. Light, and J. Bach. 2008. Factors influencing stream occupancy and detection probability parameters of stream-associated amphibians in commercial forests of Oregon and Washington, USA. Forest Ecology and Management 255: 3726-3735.

Leonard, W. P., H. A. Brown, L. L. C. Jones, K. R. McAllister, and R. M. Storm. 1993. Amphibians of Washington and Oregon. Seattle Audubon Society, Seattle, Washington. viii + 168 pp.

McAllister, 1992. Amphibian management standards. Department of Wildlife, State of Washington.

Mullen, L., J. Irwin, and S.R. Wagner. 2006. Respiration constraints of pacific northwest forest-associated salamanders: implications for stream management. Northwestern Naturalist 87(2): 180.

Nussbaum, R. 1969. A nest site of the Olympic Salamander, *Rhyacotriton olympicus* (Gaige). Herpetologica 25: 277-278.

Nussbaum, R. A., and C. K. Tait. 1977. Aspects of the life history and ecology of the Olympic salamander, Rhyacoitriton olympicus (Gaige). American Midland Naturalist 98(1): 176-199.

Nussbaum, R.A., E.D. Brodie, Jr., and R.M. Storm. 1983. Amphibians and Reptiles of the Pacific Northwest. University Press of Idaho, Moscow, Idaho. 332 pp.

Petranka, J.W. 1998. Salamanders of the United States and Canada. Smithsonian Institution Press, Washington, D.C.

Ray, C. 1958. Vital limits and rates of desiccation in salamanders. Ecology 39: 75-83.

Russell, Kevin R., Tood J. Mabee, Michael B. Cole. 2004. Distribution and Habitat of Columbia Torrent Salamanders at Multiple Spatial Scales in Managed Forests of Northwestern Oregon. The Journal of Wildlife Management 68 (2): 405-417.

Russell, Kevin R., Todd J. Mabee, Michael B. Cole, and Michael J. Rochelle. 2005. Evaluating biotic and abiotic influences on torrent salamanders in managed forests of western Oregon. Wildlife Society Bulletin 33(4): 1413-1424.

Stebbins, R. C. 1985. A Field Guide to Western Reptiles and Amphibians. Houghton Mifflin, Boston.

Stebbins, R.C., and C.H. Lowe, Jr. 1951. Subspecific differentiation in the Olympic salamander Rhyacotriton olympicus. University of California Publications Zoology 50: 465-484.

Stoddard, Margo A., and John P. Hayes. 2005. The influence of forest management on headwater stream amphibians at multiple spatial scales. Ecological Applications 15: 811–823.

Sutherland, D. and F.L. Bunnell. 2001. Cross-scale classification trees for assessing risks of forest practices to headwater stream amphibians. Pages 550-555 in D. H. Johnson and T. A. O'Neill, managing directors. Wildlife-habitat relationships in Oregon and Washington. Oregon State University Press, Corvallis, Oregon, USA.

Vesely, D.G. and W.C. McComb. 2002. Salamander abundance and amphibian species richness in riparian buffer strips in the Oregon Coast Range. Forest Science 48: 291–297.

Washington Dept. of Fish and Wildlife. 2012. Washington State Species of Concern Lists, *available at* http://wdfw.wa.gov/conservation/endangered/list/Amphibian/ (last visited Feb. 6, 2012).

Welsh, H. H., Jr. 1990. Relictual amphibians and old-growth forests. Conservation Biology 4:309-19.

Welsh, H. H., Jr., and A. J. Lind. 1996. Habitat correlates of the southern torrent salamander, *Rhyacotriton variegatus* (Caudata: Rhyacotritonidae) in northwestern California. Journal of Herpetology 30: 385-398.

INDEX OF PETITIONED SPECIES BY SCIENTIFIC NAME

INDEX OF PETITIONED SPECIES BY TAXA AND COMMON NAME

Turtles

Snakes

Toads

Frogs

Lizards

Salamanders