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Candidate Petition Project

REPTILES & AMBHIBIANS

PETITIONS TO LIST AS FEDERALLY ENDANGERED SPECIES

The following document contains the individual petitions for 4 reptile species and 7 amphibian species to be listed as federally endangered species under the federal Endangered Species Act.

Salado salamander	***Eurycea chisholmensis***
Austin blind salamander	***Eurycea waterlooensis***
Ozark hellbender	***Cryptobranchus alleganiensis bishopi***
Georgetown salamander	***Eurycea naufragia***
Black Warrior waterdog	***Necturus alabamensis***
Sonoyta mud turtle	***Kinosternon sonoriense longifemorale***
Cagle's map turtle	***Graptemys caglei***
Black pine snake	***Pituophis melanoleucus lodingi***
Eastern massasauga	***Sistrurus catenatus catenatus***
Columbia spotted frog (Great Basin DPS)	***Rana luteiventris***
Oregon spotted frog	***Rana pretiosa***

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PETITION TO LIST

salado salamander
(*Eurycea chisholmensis*)

AS A FEDERALLY ENDANGERED SPECIES

CANDIDATE HISTORY

CNOR 11/15/94:
CNOR 06/13/02: C

TAXONOMY

A description of the Salado salamander was published by Chippindale et al. (2000). This species was formerly included in the *Eurycea neotenes* species group.

NATURAL HISTORY

The Salado salamander is entirely aquatic and neotenic, meaning it does not metamorphose into a terrestrial adult. Adults are about 2 inches long. The Salado salamander has reduced eyes compared to other spring-dwelling *Eurycea* in north central Texas and lacks well-defined melanophores (cells containing brown or black pigments (melanin)) and iridophores (cells filled with iridescent color granules and fat soluble pigments). It has a relatively long and flat head and a blunt and rounded snout. Three pairs of reddish-brown to bright red gills are located on each side of the neck behind the jaws. The upper body is generally greyish-brown with a slight cinnamon tinge and an irregular pattern of tiny, light flecks. The underside is pale and translucent. The posterior portion of the tail generally has well-developed dorsal and ventral fins, although the dorsal tail fin may be absent (Chippindale et al. 2000).

POPULATION STATUS

Bell County, Texas, has approximately fourteen very small (0.028 to 0.28 cubic feet per second (cfs)) to large (280 to 2,800 cfs) springs (Brune 1981). The Salado salamander was known historically from two spring sites near Salado, Bell County, Texas: Big Boiling Springs (also known as Main, Salado, or Siren Springs) and Robertson Springs (Chippindale et al. 2000). These springs bubble up through faults in the northern segment of the Edwards Aquifer and associated limestones along Salado Creek (Brune 1975). Both are considered small to medium springs, depending on flow, by Brune's (1981) definition. Salado salamanders have not been located in Big Boiling Springs, the type locality, since 1991 despite over 20 additional visits that occurred between 1991 and 1998 (Chippindale et al. 2000). Robertson Springs are on private land and access to the site has not been granted. The last survey at Robertson Springs was in the early 1990s. Other spring sites may have Salado salamanders, but the U.S. Fish and Wildlife Service has no confirmed information on other springs with salamanders. Four other spring sites (Dining Room, Elm or Critchfield, Benedict, and Anderson Springs) are within a mile of Big Boiling and Robertson Springs (Brune 1981). Salamanders collected from the springs within Buttermilk Creek, which is near Salado Creek, may also be *E. chisholmensis*, but the specimens have not yet been identified (Chippindale et al. 2000).

The U.S. Fish and Wildlife Service classifies the salado salamander as a candidate for Endangered Species Act protection with a listing priority number of 2.

LISTING CRITERIA

Historical range:	Big Boiling Springs (=Salado Springs), Robertson Springs, and possibly other springs in the vicinity in Bell County, Texas.
Current range:	Apparently extirpated from Big Boiling Springs, but may persist at Robertson Springs and potentially other springs in its vicinity.
Land ownership:	Big Boiling Springs is located in a municipal park in Salado, Texas. Robertson Springs is on private property.

A. ***The present or threatened destruction, modification, or curtailment of its habitat or range.***

Primary threats to the Salado salamander include degradation of water quality and quantity due to urbanization and physical modification of its habitat. Most of the spring outlets in the City of Salado, including the type locality at Big Boiling Springs, have been modified during the past 150 years by dam construction in the mid-1800s to supply power to various mills, and a stone wall was built to keep out cattle (Brune 1981). In addition to direct habitat modification, urbanization also contributes to the threats to the Salado salamander by impairing water quality. Several groundwater contamination incidents have occurred (Price et al. 1995) within Salado salamander habitat. Big Boiling Springs is located on the south bank of Salado Creek in a municipal park, near where past contamination events have occurred (Chippindale et al. 2000; Price et al. 1995). Between 1989 and 1993, at least four incidents occurred within a quarter mile from both spring sites, including a 700 gallon and 400 gallon gasoline spill and petroleum leaks from two underground storage tanks (Price et al. 1995). Although most of Bell County is still considered rural, population projections from the Texas State Data Center (2000) estimate that Bell County will increase in population by approximately 60 percent from 2000 (population 237,974) to 2040 (population 381,839). Interstate 35 runs through the City of Salado (population 3,475; Texas State Data Center 2000) and offers the perfect expansion corridor for increasing urbanization. Because the springs are located on either side of Interstate 35 (Brune 1981) and

Big Boiling Springs is in the center of the city, increasing traffic and urbanization bring increasing risk of contamination spills and higher levels of impervious cover, with their subsequent impacts to the groundwater. Given the extremely limited known distribution of the Salado salamander, groundwater contamination is of critical concern and may have already negatively affected the species.

Urbanization can dramatically alter the normal hydrologic regime and water quality of an area. As areas are cleared of natural vegetation and replaced with impervious cover, rainfall no longer percolates through the ground but instead is rapidly converted to surface runoff (Schueler 1991).

Streamflow shifts from predominantly baseflow, which is derived from natural filtration processes and discharges from local groundwater supplies, to predominantly stormwater runoff. The amount of stormwater runoff tends to increase in direct proportion to the amount of impervious cover (Arnold and Gibbons 1996). With increasing stormflows, the amount of baseflow available to sustain water supplies during drought cycles is diminished and the frequency and severity of flooding increases. Increasing stormflows result in less water recharging the aquifer, thereby diminishing baseflow. The increased quantity and velocity of runoff increases erosion and streambank destabilization, which in turn leads to increased sediment loadings, channel widening, and detrimental changes in the morphology and aquatic ecology of the affected stream system (Schueler 1991; Arnold and Gibbons 1996).

Even at relatively low levels of impervious cover, "profound and often irreversible impacts to the hydrology, morphology, water quality, habitat, and biodiversity of streams" can occur (Schueler 1994). Both nationally and locally, consistent relationships between impervious cover and water quality degradation have been documented. The extent to which impervious cover is controlled in a watershed has been linked with indices of environmental health (City of Austin 1998; Schueler 1994).

B. *Overutilization for commercial, recreational, scientific, or educational purposes.*

None known.

C. *Disease or predation.*

None known.

D. *The inadequacy of existing regulatory mechanisms.*

No Federal, State, or local laws provide for the protection of the Salado salamander. Senate Bill 1, passed by the Texas State Legislature in 1996, charges the thirteen regional water planning regions in the State to develop long-term plans for their water needs. The Brazos (Region G) Regional Water Plan (HDR Engineering, Inc. 2000) states that Bell County is one of 30 counties which has a projected water shortage in the next 50 years in one or more of the six water use categories (livestock, irrigation, mining, municipal, steam-electric, and manufacturing). The projected shortages for Bell County are in the municipal, manufacturing, and steam-electric categories. Senate Bill 1 states that future regulatory and financing decisions of the Texas Water Development Board and the Texas Natural Resource Conservation Commission need to be consistent with the approved regional plans. The Clean Water Act only relates to state-wide water quality standards for human health and has limited application for groundwater protection. This Act primarily has regulations for point source pollution and no enforceable standards for non-point source pollution (it is all voluntary compliance).

Current Conservation Efforts: None.

E. *Other natural or manmade factors affecting its continued existence.*

The Salado salamander has a very limited distribution and appears to be highly sensitive to degradation of water quality and quantity. Although no direct data have been collected on the Salado salamander's sensitivity to water quality, based on very extensive data on other amphibians, it is likely to be highly sensitive. Research indicates that amphibians, particularly during the egg and larval stages, are sensitive to many pollutants, such as heavy metals; certain insecticides, particularly cyclodienes (endosulfan, endrin, toxaphene, and dieldrin), and certain organophosphates (parathion, malathion); nitrite; salts; and petroleum hydrocarbons (Harfenist et al. 1989). Because of their semipermeable skin, the development of their eggs and larvae in water, and their position in the food web, amphibians can be exposed to waterborne and airborne pollutants in their breeding and foraging habitats. Toxic effects to amphibians from pollutants may be either lethal or sublethal. Effects that are sublethal may nevertheless be quite severe (e.g., Hayes et al. 2002) and may include morphological and developmental aberrations, lowered reproduction and survival, and changes in behavior and certain biochemical processes. Because the Salado salamander is fully aquatic, there is no possibility for escape from contamination or other threats to its habitat. Crustaceans, particularly amphipods, on which the salamander feeds are especially sensitive to water pollution (Mayer and Ellersieck 1986; Phipps et al. 1995; Burton and Ingersoll 1994).

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PETITION TO LIST

Austin blind salamander
(*Eurycea waterlooensis*)

AS A FEDERALLY ENDANGERED SPECIES

CANDIDATE HISTORY

CNOR 6/13/02: C

TAXONOMY

A description of the Austin blind salamander was published by Hillis et al. (2001). Juvenile salamanders had been sighted occasionally in Barton Springs and thought to be a variation of the Barton Springs salamander (*Eurycea sosorum*). However, the observed juveniles more closely resembled the Texas blind salamander (*Eurycea* (*Typhlomolge*) *rathbuni*) and recently enough specimens have become available to formally describe these juveniles as a newly recognized distinct species based on morphological and genetic characteristics (Hillis et al. 2001).

NATURAL HISTORY

The Austin blind salamander is entirely aquatic and neotenic, meaning it retains the larval, gill-breathing morphology throughout its life and does not metamorphose into a terrestrial adult. Adults are approximately 2.5 inches long from snout to tail. This species lacks external eyes (no lenses are present and the dark eye spots are covered with skin), has permanent external gills, and 12 costal grooves. The head is narrower at the eye spots than at the widest point in front of the gills, and has an extended snout. Its gills and limbs are proportionately shorter than those of the Texas blind or the Robust (Blanco) (*Eurycea* (*Typhlomolge*) *robusta*) salamanders. The tail fins are not well developed and are only visibly present on the posterior half of the ventral surface. The fins are either missing or are very low on the anterior half of the dorsal surface (Hillis et al. 2001). Juveniles look similar to adults, but are less than 1 inch long (personal communication 2002 cited in U.S. Fish and Wildlife Service candidate assessment form). The skin of the Austin blind salamander appears reflective and pearly white in color with a lavender hue. Adults collected from the wild appear to be a darker lavender than either the juveniles collected from the wild or the adults raised in captivity (Hillis et al. 2001).

POPULATION STATUS

The Austin blind salamander is found in three of the four Barton Springs outlets in the City of Austin's Zilker Park, Travis County, Texas: Parthenia (Main) Spring, Eliza Spring, and Sunken Garden (Old Mill). The Main Spring forms the Barton Springs swimming pool. The Austin blind salamander has not been observed at the fourth Barton Springs outlet, known as Upper Barton Spring (Hillis et al. 2001). The only known sites have been significantly modified for human use. The area around the main spring outlet was impounded in the late 1920s to create Barton Springs Swimming Pool, and flows from Eliza and Sunken Garden springs are also retained by concrete structures, forming small pools on either side of Barton Springs Pool (U.S. Fish and Wildlife Service 1997). These springs are fed by the Barton Springs segment of the Edwards Aquifer. This segment of the Edwards Aquifer runs under portions of two counties, with flow in the Aquifer being funneled towards the Barton Springs (see section A of Listing Criteria for more information on the Aquifer).
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Because all but one of the Austin blind salamander specimens collected have been juveniles and the salamander is rarely seen at the surface, this salamander is thought to be more subterranean than the aquatic surface-dwelling Barton Springs salamander. The species is thought to live only in the Edwards Aquifer; all specimens collected are believed to have been accidentally flushed out of the aquifer (Hillis et al. 2001).

From January 1998 to February 2002, there have been only 120 documented observations of the Austin blind salamander. During this same time frame, 2,059 Barton Springs salamanders have been observed (Hillis et al. 2001). Because this species spends a large portion of its life underground, the technology to safely and reliably mark salamanders for individual recognition has not been developed, and surveying within the Edwards Aquifer cannot be done at the current time, population estimates are not possible at this time. However, when they are found, the Austin blind salamander appears to occur in relatively low numbers. Between February 2001 and February 2002, only an average of eight Austin blind salamanders were found per survey visit (City of Austin 2002a). In addition, none of the Austin blind salamander specimens seen in the wild, or brought into captivity, have developed eggs (personal communication 2002 cited in U.S. Fish and Wildlife Service candidate assessment form). At the present time, the presence of eggs is the only non-lethal way of determining the sex of the Austin blind salamanders.

The U.S. Fish and Wildlife Service classifies the Austin blind salamander as a candidate for Endangered Species Act protection with a listing priority number of 2.

LISTING CRITERIA

A. ***The present or threatened destruction, modification, or curtailment of its habitat or range.***

Historical range:	Texas.
Current range:	Barton Springs (City of Austin, Travis County, Texas).
Land ownership:	The only known location for the Austin blind salamander is operated as a

City Park by the City of Austin Parks Department. The recharge and contributing zones of the Barton Springs segment of the Edwards Aquifer are a combination of municipal and private lands.

Primary threats include degradation of water quality and quantity due to urbanization. The Austin blind salamander, like the endangered Barton Springs salamander, depends on clean water from the Barton Springs segment of the Edwards Aquifer. The Barton Springs segment covers roughly 155 square miles (401 square kilometers) from southern Travis County to northern Hays County, Texas. It has a storage capacity of over 300,000 acre-feet (U.S. Fish and Wildlife Service 1997).
Because the Edwards Aquifer is a karst aquifer, it is significantly impacted by the quality and quantity of runoff from the recharge and contributing zones (U.S. Fish and Wildlife Service 1997).

Travis County grew 2.5 percent between 1998 and 1999 (U.S. Census Bureau 2000). Based on population projections from the Texas State Data Center (2000), Travis County is expected to double in size between 1990 (population 576,407) and 2030 (population projection 1,362,538). The City of Austin (in Travis County) is one of the fastest growing cities in the U.S. and experienced a 17 percent growth rate between 1990 and 1998 (U.S. Census Bureau 1998 cited in U.S. Fish and Wildlife Service candidate assessment form but omitted from References).

Urbanization can dramatically alter the normal hydrologic regime and water quality of an area. As areas are cleared of natural vegetation and replaced with impervious cover, rainfall no longer percolates through the ground but instead is rapidly converted to surface runoff (Schueler 1991). Streamflow shifts from predominantly baseflow, which is derived from natural filtration processes and discharges from local groundwater supplies, to predominantly stormwater runoff. The amount of stormwater runoff tends to increase in direct proportion to the amount of impervious cover (Arnold and Gibbons 1996). With increasing stormflows, the amount of baseflow available to sustain water supplies during drought cycles is diminished and the frequency and severity of flooding increases. Increasing stormflows result in less water recharging the aquifer, thereby diminishing baseflow. The increased quantity and velocity of runoff increases erosion and streambank destabilization, which in turn leads to increased sediment loadings, channel widening, and detrimental changes in the morphology and aquatic ecology of the affected stream system (Schueler 1991, Arnold and Gibbons 1996). Even at relatively low levels of impervious cover, "profound and often irreversible impacts to the hydrology, morphology, water quality, habitat, and biodiversity of streams" can occur (Schueler 1994). Both nationally and locally, consistent relationships between impervious cover and water quality degradation have been documented. The extent to which impervious cover is controlled in a watershed has been linked with indices of environmental health (City of Austin 1998a, Schueler 1994).

Increases in impervious cover exceeding 10 percent are associated with measurable water quality degradation, loss of sensitive aquatic organisms, reduction in stream biodiversity, stream warming, and channel instability within a watershed (Schueler 1994). Stream aquatic life problems such as loss of species diversity, malformations, and death have been identified in watersheds having impervious cover of at least 12 percent, with severe problems in watersheds with impervious cover greater than 30 percent. Generally, stream quality impairment can be prevented if watershed imperviousness does not exceed 15 percent and for more sensitive stream ecosystems watershed imperviousness should not exceed 10 percent (Klein 1979).

The Lower Colorado River Authority (LCRA 2001) conducted a water supply study of the recharge and contributing zone areas within the Barton Springs segment of the Edwards Aquifer that looked at the amount of impervious cover within the area. The eight watersheds within the area had a range of impervious cover from 3.2 percent to 28.9 percent in 2000. The projected impervious cover limits for the same eight watersheds in 2025 ranged from 4.8 percent to 31.6 percent (LCRA 2001). The two watersheds, Williamson Creek and Sunset Valley Creek (a tributary to Williamson Creek), with the highest percentage of impervious cover are also the second closest to the Barton Springs. Therefore, any negative impacts to water quality coming from those areas will likely be less diluted when entering the Springs than water received from other, farther away, watersheds. In addition, sediments discharging from karst aquifers play a fundamental role in determining water quality (Mahler et al. 1999).

Sediments have both a direct impact on habitat quality and can act as a sink and transport mechanism for other contaminants (Menzer and Nelson 1980). Karst systems are more vulnerable to the effects of pollution because of their thin surface soils, high groundwater flow velocities, and the relatively short time water is resident within the system (Ford and Williams 1994).

Surface derived sediments have the greatest potential to concentrate and transport contaminants because of their high organic carbon content and their potential exposure to contaminants at the surface (Mahler and Lynch 1999). Because stormwater moves sediment through karst systems in a pulsed fashion, impacts to the aquifer are not limited to the relatively short duration of runoff events. Generally, stormwater pollutants attach to sediments and become part of the sediment system (Burton 1992). The term "attach" is used to describe the processes of complexation, adsorption, absorption, and secondary physical and chemical processes that incorporate pollutants into the inorganic and organic matrices of soil and sediment. Sediment is moved through the Barton Springs segment of the Edwards Aquifer in pulses caused by storm events (Mahler and Lynch 1999). Sediments (with attached pollutants) may deposit within the aquifer and be resuspended during subsequent storm events.

In an analysis performed by the City of Austin (2000), significant changes over time were reported for several chemical constituents and physical parameters in Barton Springs Pool. Conductivity, turbidity, sulfates, and total organic carbon have increased while the concentration of dissolved oxygen has decreased. The significance and presence of trends are variable depending on flow conditions (baseflow vs. stormflow, recharge vs. non-recharge) and could be attributed to impacts from watershed urbanization (City of Austin 2000). These data indicate a long-term trend of water quality degradation at Barton Springs over the past 25 years (U.S. Fish and Wildlife Service 2001).

Four pesticides (atrazine, carbaryl, diazinon, and simazine) were documented at Barton Springs

Pool and Eliza Spring in samples taken during and after a two-day storm event (USGS 2000). Atrazine, carbaryl, diazinon, and simazine at the springs were found at levels below the exhibited toxicity to aquatic animals. Although concentrations of these pesticides are below criteria set in the aquatic life protection in the Texas Surface Water Quality Standards, increases in pesticide concentrations could adversely affect aquatic organisms (U.S. Fish and Wildlife Service 2001). Ecologically realistic levels of atrazine contamination, for example, may have devastating effects on amphibian reproduction (Hayes et al. 2002). Several heavy metals, including arsenic, cadmium, copper, lead, nickel, and zinc and sediment of possible anthropogenic origin, have also been detected in Barton Springs (City of Austin 1997). Dissolved lead is very toxic in aquatic environments, and adverse effects on aquatic invertebrates and fish, including reduced survival, impaired reproduction, and reduced growth, have been reported at concentrations of 0.001 to 0.005 milligrams per liter (Eisler 1988). Sources of lead in water include industrial discharges, urban runoff, and sewage effluent (Pain 1995). Although measured concentrations for lead at Sunken Garden Spring are approximately one-half the LC50 (concentration or dose that kills 50 percent of the observed population) reported by Birge (1978), chronic and sublethal effects of lead on amphibians, such as the Austin blind salamander, could occur at much lower concentrations. However, hardness can effect the availability of dissolved lead and may provide some buffer to toxicity in a high alkalinity system like Barton Springs (U.S. Fish and Wildlife Service 2001).

The Environmental Protection Agency (1997) has proposed screening values for sediment concentrations for chemicals as evaluated in their national sediment quality survey. Sediment samples were taken from the bottom of Barton Springs and Barton Creek during normal flow periods and from storm flow of the springs and the creek during periods of heavy precipitation. Contaminated sediments were found in all pathways that lead to Barton Springs and in salamander habitat, and six heavy metals exceeded at least one sediment screening criterion on 17 occasions (U.S. Fish and Wildlife Service 2001). Adverse effects to the Barton Spring salamander, and therefore the Austin blind salamander, and its prey may be occurring by exceeding sediment criteria suggested by EPA (1997), MacDonald et al. (2000), and the Texas Natural Resource Conservation Commission (2000). These adverse effects to the salamanders may include differences in growth, weight, and behavior; morphological and developmental aberrations; and lowered reproduction (U.S. Fish and Wildlife Service 2001).

B. *Overutilization for commercial, recreational, scientific, or educational purposes.*

Because the Austin blind salamander is a newly described species and is currently unprotected through regulatory mechanisms, collectors may be interested in obtaining specimens. The City of Austin has included the Austin blind salamander in its captive breeding efforts for the Barton Springs salamander. In 2001, City of Austin Watershed Protection Division personnel in charge of the captive breeding program collected 14 Austin blind salamanders for inclusion in the captive breeding program (City of Austin 2002b), but six subsequently died of unknown causes.

C. *Disease or predation.*

None known.

D. ***The inadequacy of existing regulatory mechanisms.***

The conspecific Barton Springs salamander is federally and State listed as endangered, and the City of Austin is covered for incidental take of the Barton Springs salamander from its swimming pool maintenance activities though an Endangered Species Act section 10(a)(1)(B) permit and the associated Habitat Conservation Plan (City of Austin 1998b). The Austin blind salamanders that exit the aquifer and enter the pool benefit from protection measures for the Barton Springs salamander. Controls of nonpoint source pollution in the watershed consist of a variety of local ordinances, which range from relatively strict controls by the City of Austin in its Extraterritorial Jurisdiction to lesser controls in outlying areas, and adoption of the "Edwards Rules" (water quality protection measures for the recharge and contributing zones of the Edwards Aquifer) by the Texas Natural Resources Conservation Commission (TNRCC) in 1995 and 1997. However, the U.S. Fish and Wildlife Service believes that these improvements have not provided adequate protection for the Austin blind salamander because the Texas state legislature "grandfathered" existing projects in the watershed in 1999, which further weakened existing water quality protection (U.S. Fish and Wildlife Service 2001).

The villages of Bee Cave and Dripping Springs also have regulations in place that offer some water quality protection. These protections include riparian buffers and impervious cover limitations of up to 55 percent (U.S. Fish and Wildlife Service 2001). In recent months, there has been an unexplained die-off of Barton Springs salamanders in Sunken Garden, where the Austin blind salamander is also found, and Upper Barton Springs. The U.S. Fish and Wildlife Service, City of Austin, U.S. Geological Survey, and U.S. Environmental Protection Agency are still awaiting lab results on water, sediment, and tissue samples. At this point it is unclear whether gas bubble disease (which has never been seen before in amphibians in the wild); gas bubble disease in conjunction with some other water quality factor, or some other agent is responsible for the deaths of 13 Barton Springs salamanders (City of Austin 2002a; personal communication 2002 cited in U.S. Fish and Wildlife Service candidate assessment form).

Current Conservation Efforts: The City of Austin has included the Austin blind salamander in its captive breeding program along with the Barton Springs salamander.

E. ***Other natural or manmade factors affecting its continued existence.***

The Austin blind salamander has a very limited distribution. Amphibians, particularly during egg and larval stages, are sensitive to many pollutants, such as heavy metals; certain insecticides, particularly cyclodienes (endosulfan, endrin, toxaphene, and dieldrin), and certain organophosphates (parathion, malathion); nitrite; salts; and petroleum hydrocarbons (Harfenist et al. 1989). Because of their semipermeable skin, the development of their eggs and larvae in water, and their position in the food web, amphibians can be exposed to waterborne and airborne pollutants in their breeding and foraging habitats. Toxic effects to amphibians from pollutants may be either lethal or sublethal. Effects that are sublethal may nevertheless be quite severe (e.g., Hayes et al. 2002)and may include morphological and developmental aberrations, lowered reproduction and survival, and changes in behavior and certain biochemical processes. Because the Austin blind salamander is fully aquatic, it cannot escape from contamination or other threats to its habitat. Crustaceans, particularly amphipods, on which the salamander feeds are especially sensitive to water pollution (Mayer and Ellersieck 1986; Phipps et al. 1995; Burton and Ingersoll 1994).

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PETITION TO LIST

Ozark hellbender
(*Cryptobranchus alleganiensis bishopi*)

AS A FEDERALLY ENDANGERED SPECIES

CANDIDATE HISTORY

CNOR 10/30/01: C
CNOR 6/13/02: C

TAXONOMY

The Ozark hellbender (*Cryptobranchus alleganiensis bishopi*) (Cryptobranchidae) was originally described as *C. bishopi* by Grobman (1943) from a specimen collected from the Current River in Carter County, Missouri. Schmidt (1953) listed the Ozark hellbender as a subspecies of the eastern hellbender, *C. alleganiensis*, a view supported by Dundee and Dundee (1965) and not inconsistent with the results of early genetic studies by Merkle et al. (1977) and Shaffer and Breden (1989). This subspecific rank persisted until Collins (1991) revived *C. bishopi* based on the lack of intergradation between the eastern and Ozark hellbenders. Although Ozark hellbenders have been shown to be distinct both phenotypically (Grobman 1943, Dundee and Dundee 1965, Dundee 1971) and genetically (Routman 1993, Wagner et al. 1999) from eastern hellbenders, the name *C. a. bishopi* is used here as it is the name currently recognized by the Center for North American Herpetology (Collins and Taggart 2002).

NATURAL HISTORY

Morphology
The Ozark hellbender is a large, strictly aquatic salamander endemic to streams of the Ozark plateau in southern Missouri and northern Arkansas. Its dorsoventrally flattened body form helps it remain immobile in the fast flowing streams it inhabits (Wagner et al. 1999). Hellbenders have a large, keeled tail and tiny eyes. Adult Ozark hellbenders may attain total lengths of 29 - 57 cm (Dundee and Dundee 1965, Johnson 1987). Numerous fleshy folds along the sides of the body provide surface area for respiration (Nickerson and Mays 1973a) and obscure poorly developed costal grooves (Dundee 1971). Ozark hellbenders are distinguishable from eastern hellbenders by their smaller body size, dorsal blotches, increased skin mottling, heavily pigmented lower lips, smooth surfaced lateral line system, and reduced spiracular openings (Grobman 1943,

Dundee 1971, Peterson et al. 1983, LaClaire 1993).

Behavior

Adult hellbenders are nocturnal, remaining beneath cover during the day and emerging to forage primarily on crayfish at night, although they are not entirely nocturnal (Nickerson and Mays 1973a, Noeske and Nickerson 1979, Collins 1997). Ozark hellbenders are territorial and will defend occupied cover from conspecifics (Nickerson and Mays 1973a). This species migrates little, with one tagging study revealing that 70 percent of marked individuals moved less than 30 meters from the site of original capture (Nickerson and Mays 1973b). Home ranges average 28 square meters for females and 81 square meters for males (Peterson and Wilkinson 1996).

Typically, Ozark hellbender populations are dominated by older, large adults (Nickerson and Mays 1973a, Peterson et al. 1983, LaClaire 1993). Juveniles reach sexual maturity between 5 and 8 years, with males maturing at a smaller size and younger age than females. Ozark hellbenders may live 25 - 30 years in the wild (Peterson et al. 1983).

Breeding generally occurs between September and November, with Spring River, Arkansas, populations breeding in January (Peterson et al. 1983). Ozark hellbenders mate via external fertilization, and males will guard the fertilized eggs from predation by conspecifics (Nickerson and Mays 1973a). Clutch sizes vary from 138 to 450 eggs per nest (Dundee and Dundee 1965, Zug 1993), and eggs hatch after approximately 80 days (Zug 1993). Hatchlings and larvae are collected rarely during surveys, probably due to low capture efficiency and high mortality of young. Larvae and small individuals often live beneath small stones in gravel beds or shallow water habitats (Nickerson and Mays 1973a, LaClaire 1993).

Habitat

Eastern and Ozark hellbenders are very similar in habitat selection, movement, and reproductive biology (Nickerson and Mays 1973a). Published works on the eastern hellbender may provide insights into Ozark hellbender ecology. Adult Ozark hellbenders are frequently found beneath large rocks in moderately deep (< 1m), rocky, fast-flowing streams in the Ozark plateau (Johnson 1987, Fobes and Wilkinson 1995, Wagner et al. 1999). In spring-fed streams, Ozark hellbenders typically concentrate just downstream of the area where there is no water temperature change throughout the year (Dundee and Dundee 1965).

Distribution

Ozark hellbenders are endemic to the Black and White River drainages in Arkansas and Missouri (Johnson 1987) in portions of the Spring, White, Eleven Point, and Current Rivers and their tributaries (LaClaire 1993). This taxon is believed to be declining throughout its range, and no populations appear to be stable. Declines have been evident throughout the range of the eastern hellbender, as well, which has state protected status in many eastern states. A description of what is known about Ozark hellbender populations follows.

White River System

White River- There is only one Ozark hellbender record from the main stem of the White River, coming from Baxter County, Arkansas, in 1997 (personal communication cited in U.S. Fish and

Wildlife Service candidate assessment form). It is not known whether a viable population exists at this site or if the individual captured is a member of a relict population that was separated from the North Fork White River population by Norfork reservoir.

North Fork White River - The North Fork White River historically contained a considerable Ozark hellbender population. In 1973, results of a mark-recapture study indicated approximately 1,150 hellbenders within a 2.67 km reach of river in Ozark County, Missouri, with a density of one individual per eight to ten m^2(1/8-10 m^2) (Nickerson and Mays 1973b). Ten years later, hellbender density in a 4.6 km section of the North Fork White River in the same county remained rather high, with densities between 1/6-7 m^2 and 1/13-16 m^2 (Peterson et al. 1983). Individuals caught in this study also represented a range of lengths (172 - 551 mm), indicating that reproduction was occurring in this population, and most individuals were sized at between 250 - 449 mm.

Subsequently, in a 1992 qualitative study also in Ozark County, Missouri, 122 hellbenders were caught during 49 man-hours of searching (Ziehmer and Johnson 1992). These individuals ranged from 254 - 457 mm, and no average size was included in this publication. Up to the 1992 study, the North Fork White River population appeared to be fairly healthy. However, in a 1998 study of the same reach of river censused in 1983 (Peterson et al. 1983) and using the same collection methods, only 50 hellbenders were captured (Wheeler et al. 1999). These individuals ranged in length from 200 - 507 mm, with most being between 400 - 500 mm, and were on average significantly longer than those collected twenty years earlier (Wheeler 1999). This shift in length distribution was not a result of an increase in maximum length of individuals; instead, there were fewer individuals collected in the smaller size classes.

In order to compare results between these qualitative and quantitative studies, Wheeler et al. (1999) converted historical hellbender collections (Peterson et al. 1983) to numbers of individuals caught per day. In addition, the other studies that were not included in that conversion (Peterson 1983, Peterson 1988, Ziehmer and Johnson 1992) have been converted here. For comparison purposes, one search day is defined as 8 hours of searching by 3 people (i.e., 24 person-hours). Although this search day may be an underestimate of actual effort, a conservative estimate of effort will result in a conservative estimate of hellbender population declines. Therefore, in 1983, approximately 51 hellbenders were caught per sampling day (Peterson et al. 1983). In 1992, 60 hellbenders/day were caught (Ziehmer and Johnson 1992), and, in 1998, 16 hellbenders/day were caught (Wheeler 1999). Based on these comparisons, a decline in the North Fork White River is evident.

The North Fork White River had been considered the stronghold of the species, and the populations inhabiting this river were deemed stable (Ziehmer and Johnson 1992, LaClaire 1993). However, these populations now appear to be experiencing declines similar to those in other streams. The collection of young individuals has become rare, indicating little recruitment. In species such as the Ozark hellbender, which are long lived and mature at a relatively late age, detecting declines related to recruitment can take many years, as recruitment under healthy population conditions is typically low (Nickerson and Mays 1973a). A gradual, long-term decline appears to be occurring in the North Fork White River, although quantitative studies are needed to determine the likely effects of this decline on the population.

Bryant Creek - Bryant Creek is a tributary of the North Fork White River in Ozark County, Missouri, which flows into Norfork Reservoir. Ziehmer and Johnson (1992) expected to find Ozark hellbenders in this stream during an initial survey, but none were captured or observed after 22 man-hours. This apparent lack of the species conflicted with reports from Missouri Department of Conservation (MDC) personnel and fisherman who reported observations of fairly high numbers of hellbenders in Bryant Creek during winter months (Ziehmer and Johnson 1992). A subsequent survey of the creek resulted in the capture of 6 hellbenders (Wheeler et al. 1999), confirming the existence of a population in this tributary. However, this population is isolated from the other North Fork White River populations by Norfork reservoir, which could contribute to this population's apparent small size.

Black River System
Black River- There are no documented records of Ozark hellbenders in the Black River, although it has not been extensively surveyed. Portions of the Black River in Missouri were surveyed in 1999 by researchers at Arkansas State University, but no Ozark hellbenders were observed (Wheeler et al. 1999). The Black River is presumed to be part of the historic range of the species, due to the presence of hellbenders in several of its tributaries, including the Spring, Current, and Eleven Point Rivers (Firschein 1951, Trauth et al. 1992).

Spring River**-** The Spring River, a tributary of the Black River, flows from Oregon County, Missouri, south into Arkansas. Ozark hellbender populations have been found in the Spring River near Mammoth Spring, Fulton County, Arkansas (LaClaire 1993). In the early 1980's, 370 individuals were captured during a mark-recapture study along 7 km of stream south of Mammoth Spring (Peterson et al. 1988). Hellbender density at each of the two surveyed sites was fairly high (approximately 1/23 m^2 and 1/111 m^2).

These individuals were considerably larger than hellbenders captured from other streams during the same time period, with 74 percent of Spring River Ozark hellbenders measuring over 450 mm total length (maximum 600 mm) (Peterson et al. 1988). This may indicate that Spring River populations are somewhat distinct genetically from other Ozark hellbender populations. This conclusion was upheld by a genetic study of the Spring, Current, and Eleven Point River populations (Wagner et al. 1999). In 1991, a longer reach (26 km) was surveyed for Ozark hellbenders, and only 20 were observed during 41 search hours over a 6 month period, at many of the same sites sampled by Peterson et al. (1983) (Trauth et al. 1992). No length information is available, although the large sizes of the 1988 captures may be indicative of a population experiencing little recruitment. Although the recent surveys were less intensive than the previous studies, it is apparent that hellbenders have declined in this stream.

Eleven Point River- The Eleven Point River, a tributary of the Black River, has been surveyed several times since the 1970's. Historical data provided by Peterson was analyzed by Wheeler (1999). In 1978, 87 hellbenders were captured in Oregon County, Missouri, over 3 days, yielding 29 hellbenders/day. Later, in 9 collection days from 1980 - 1982 in the same area, 314 hellbenders were captured, yielding 35 hellbenders/day. Lengths over this period ranged from

119 - 451 mm. Six years later, Peterson et al. (1988) captured 211 hellbenders from the Eleven Point River and estimated hellbender density to be approximately 1/20 m^2. Total lengths of these individuals ranged from 120 - 450 mm, with most between 250 - 350 mm. Although this statistic was not presented, it can be estimated that roughly 40 hellbenders were caught per day during this study.

Approximately 10 years later, Wheeler (1999) captured 36 hellbenders over 4 days from Peterson et al.'s (1988) sites, for an average of 9 hellbenders/day. These hellbenders were larger than those captured previously, with total lengths of 324 - 457 mm, and there were significantly fewer individuals in the smaller size classes. In summary, the population appeared stable until 1988 (captures of 29, 35, and roughly 40 hellbenders/day), and then dropped in 10 years to 9 hellbenders/day, and these individuals were considerably larger than those caught previously. Therefore, in the Eleven Point River, similar declines and lack of recruitment are evident as in other streams.

<u>Current River</u>- The Current River had not been surveyed extensively until the 1990's. Nickerson and Mays (1973a) reported a large population in this stream, but no numbers were presented. In 1992, Ziehmer and Johnson (1992) found 12 Ozark hellbenders in 60 man-hours in Shannon County, Missouri, or approximately 5 hellbenders/day, using the same search day conversion as presented above. These individuals ranged in length from 115 mm to over 380 mm (maximum length was not reported), with most between 330 mm and 380 mm. Seven years later, 14 hellbenders were collected over 3 collection days (approximately 5 hellbenders/day), also in Shannon County, Missouri, and the individuals ranged from 375 - 515 mm, with most between 450 - 499 mm (Wheeler 1999). It appears that this population is small, and may not be declining. However, the average size of individual has increased by nearly 100 mm, and this population shows a lack of recruitment.

<u>Jacks Fork</u>- Jacks Fork, a tributary of the Current River, was surveyed for the first time in 1992 for Ozark hellbenders (Ziehmer and Johnson 1992). Four hellbenders were collected over 66 man-hours, roughly 2 hellbenders/day. The individuals were large, ranging from 330 - 430 mm. There have been no subsequent investigations of Jacks Fork, so no conclusions may be drawn about population trends in this stream.

POPULATION STATUS

Much of the hellbender habitat was destroyed by the series of dams constructed in the 1940s and 1950s on the upper White River, including Beaver, Table Rock, Bull Shoals, and Norfork dams. Hellbenders may be collected with a permit from the Arkansas Game and Fish Commission. Collecting specimens for the pet trade and biological laboratories has impacted local amphibian populations. Trauth et al. (1992) suspected that collection of hellbenders in the Spring River, Arkansas, contributed to observed population declines. Agricultural runoff and the acidic runoff from large scale mining operations threaten much of the hellbenders habitat (Danch 1996).

The U.S. Fish and Wildlife Service classifies the Ozark hellbender as a candidate for Endangered Species Act protection with a listing priority number of 6. The Arkansas and Missouri Natural Heritage Programs rank the Ozark Hellbender as Critically Imperiled.

LISTING CRITERIA

A. ***The present or threatened destruction, modification, or curtailment of its habitat or range.***

Historical range: Streams of the Ozark plateau in southern Missouri and northern Arkansas.

Current range: The Ozark hellbender is found only in the Black River and White River systems of Missouri and Arkansas (Danch 1996). This includes portions of the Spring, White, Eleven Point, and Current Rivers and their tributaries.

Land ownership: Approximately 80 percent of the land within the range of the Ozark hellbender is in private ownership, with the remaining 20 percent federally owned and managed by the U.S. Forest Service (Mark Twain National Forest).

The decline of the Ozark hellbender in the White and Black River systems in Missouri and Arkansas is likely the result of habitat degradation in the form of impoundments, ore and gravel mining, silt and nutrient runoff, and den site disturbance due to recreational uses of the rivers it inhabits (Williams et al. 1981, LaClaire 1993). Although the precise causes of hellbender declines are likely complex interrelationships among threats and the species' life history characteristics, habitat degradation is the most frequent cause of lotic faunal declines (Allan and Flecker 1993). Hellbenders are habitat specialists that depend on constant levels of dissolved oxygen, temperature, and flow (Williams et al. 1981). Therefore, even minor alterations to stream habitat are likely detrimental to hellbender populations.

Impoundments impact stream habitat in many ways. When a dam is built on a free-flowing stream, riffle and run habitats in the area impounded by the dam are converted to open water. As a result, water temperatures tend to increase and dissolved oxygen levels tend to decrease, due to the lotic conditions of the water (Allen 1995). Because hellbenders are habitat specialists, they cannot tolerate a wide range of habitat conditions. Hellbenders depend upon highly vascularized lateral skin folds for respiration; therefore, lakes and reservoirs are unsuitable habitats for Ozark hellbenders, as these areas have lower oxygen levels and higher water temperatures (Williams et al. 1981, LaClaire 1993) than their fast flowing, cool water, highly oxygenated stream habitat. In addition, impoundments on inhabited streams create unsuitable habitat for hellbenders and, therefore, are impediments to movement between populations. In the upper White River, construction of Beaver, Table Rock, Bull Shoals, and Norfork dams in the 1940's and 1950's has destroyed much of the historic hellbender habitat that occurred there and has effectively isolated hellbender populations.

Norfork dam was constructed on the North Fork White River in 1944 and has isolated Ozark hellbender populations in Bryant Creek and the White River from those in the North Fork White River. Additionally, populations downstream of Beaver, Table Rock, Bull Shoals, and Norfork dams were extirpated due to hypolimnetic releases from the reservoir. These releases are much cooler than normal stream temperatures, and the water in such releases is typically depleted of oxygen. In addition, the tailwater zones below dams experience extreme water level fluctuations and scouring for many miles downstream which impact hellbender populations by washing out the gravel and chert used by juveniles and creating unpredictable habitat conditions that fluctuate outside the Ozark hellbender's range of tolerance.

Gravel mining has occurred in many southeastern streams, including a number of streams within the historic range of the Ozark hellbender, which has contributed to Ozark hellbender habitat alteration and loss. Dredging results in stream instability both up and downstream of the dredged portion (Neves et al. 1997, Box and Mossa 1999). Head cutting, in which the increase in transport capacity of a dredged stream causes severe erosion and degradation upstream, results in extensive bank erosion, sloughing, and increased turbidity levels (Allan 1995).

Reaches downstream of the dredged stream reach often experience aggradation as the sediment transport capacity of the stream is reduced (Box and Mossa 1999). These activities disturb hellbender den sites in dredged areas, and associated silt plumes can cover downstream den sites. In addition, these effects reduce crayfish populations, which are the primary prey species for Ozark hellbenders. Gravel dredging is widespread in the White and Black River systems in southernMissouri and northern Arkansas (LaClaire 1993). Modifications of stream channels associated with gravel mining, as well as the removal of small stones and chert that are important microhabitat for larvae and subadults, contribute to the decline of Ozark hellbenders in these systems.

Portions of the Ozark plateau have a history of being major producers of lead and zinc, and some mining activity still occurs in the southeastern Ozarks, though at less than historic levels. Results of a recent USGS water quality study in the Ozark plateau revealed that concentrations of lead and zinc in bed sediment and fish tissue were substantially higher at sites with historical or active mining activity and that these concentrations were high enough to suggest adverse biological effects, such as reduced enzyme activity or death of aquatic organisms. Although mining for lead and zinc no longer occurs within the range of the Ozark hellbender, elevated concentrations are still present in the streams where mining occurred historically (Petersen et al. 1998).

Despite the claim by some that many Ozark streams outwardly appear to exist in pristine conditions, Harvey (1980) clearly demonstrated that various sources of pollution exist in the ground water in the Springfield-Salem Plateaus of southern Missouri. In comparing ground-water quality of sites within the Ozark Plateaus (including Arkansas and Missouri) with other National Water-Quality Assessment Program (NAWQA) sites, Petersen et al. (1998) documented that: 1) nitrate concentrations in parts of the Springfield Plateau aquifer were higher than in most other NAWQA drinking-water aquifers, and 2) volatile organic compounds were detected more frequently in drinking-water aquifers within the Ozark Plateaus than in most other drinking-water aquifers. These studies overlap well with the current distribution of Ozark hellbender in Arkansas and Missouri.

Silt and sediment runoff from land use activities in the area have contributed to habitat degradation. Hellbenders are intolerant of siltation and turbidity (Nickerson and Mays 1973a) and can be impacted by these in several ways. First, sediment deposition in densites will cover and suffocate eggs. Second, sediment will fill in interstitial spaces in gravel/chert areas, reducing suitable habitat for larvae and subadults (FISRWG 1998). Third, suspended sediment loads can also cause water temperatures to increase, as there are more particles to absorb heat, thereby reducing dissolved oxygen levels (Allen 1995). Because the Ozark hellbender requires cool temperatures and high levels of dissolved oxygen, perturbations to environmental conditions can be detrimental to hellbender populations. Fourth, the Ozark hellbender's highly permeable skin causes them to be negatively affected by sedimentation. Various chemicals, such as pesticides, bind to silt particles and become suspended in the water column when flushed into a stream. The hellbender's permeable skin provides little barrier to these chemicals, which can be toxic (Blaustein and Wake 1990, Wheeler et al. 1999).

Timber harvesting is prominent in many areas within the range of the Ozark hellbender, and roads probably introduce the bulk of suspended sediment through erosion from road construction and the sediment-transporting ability of constructed roads. Roads can also cause marginally stable slopes to fail, and they capture surface runoff and channel it directly into streams (Allan 1995). In addition, erosion from roads contributes more sediment than the land harvested for timber (Box and Mossa 1999). Peak stream flows often rise in watersheds with timber harvesting activities, due in part to compacted soils resulting from roads, landings, and vegetation removal (Allan 1995, Box and Mossa 1999). The cumulative effects of timber harvest on sedimentation rates last for many years, even after harvest practices have ceased in the area (Frissell 1997).

Nitrogen and phosphorus are essential plant nutrients that are found naturally in streams. However, elevated concentrations of these nutrients causes excessive growth of aquatic algae and plants in many streams and has detrimental effects upon water quality.

Contamination of water in the Ozark plateau by nutrients has occurred from runoff of poultry and cattle wastes, human wastes, and fertilizers. National Water Quality Assessment (NAWQA) data collected in the Ozarks in 1993-1995 from wells and springs indicated that nitrate concentrations were strongly associated with the percentage of agricultural land near the wells or springs. In addition, fecal coliform levels have been elevated in these areas (Petersen et al. 1998). Livestock wading in streams, poor agricultural practices that lead to the degradation of riparian buffer zones, and faulty septic and sewage treatment systems have resulted in these elevated levels, which cause more algae to grow on streambed rocks. This growth affects aquatic species composition and causes benthic-feeding organisms to thrive (Petersen et al. 1998). Agriculture comprises approximately 30 percent of the land use within the range of the Ozark hellbender, which is intolerant of nutrient pollution (Nickerson and Mays 1973a).

Habitat disturbances may also be affecting hellbender success in several rivers. Canoeing and fishing are common in many of the rivers inhabited by the Ozark hellbender, including the Spring, Current, and North Fork White Rivers. Although no data are available that support this assertion, it has been speculated that the disturbance of den sites by contact with canoes may lead to the abandonment of those sites. In addition, some larger rocks have been removed in order to prevent canoe damage (Nickerson and Mays 1973a, Wheeler et al. 1999). The areas under such large rocks are used as hellbender den sites, so removal of these rocks reduces the number of available den sites.

B. ***Overutilization for commercial, recreational, scientific, or educational purposes.***

Anecdotal reports indicate that Ozark hellbenders have been collected for both commercial and scientific purposes (Trauth et al. 1992). Commercial collections are currently illegal in both Missouri and Arkansas, but in Arkansas hellbenders may be collected with a permit from the Arkansas Game and Fish Commission. Missouri imposed a moratorium on hellbender collecting from 1991 to 1996 and has since only allowed limited numbers of collecting permits (personal communication cited in U.S. Fish and Wildlife Service candidate assessment form). Nonetheless, illegal collecting for the pet trade has been documented, with one report of over 100 hellbenders illegally collected nearly 18 years ago (personal communication cited in U.S. Fish and Wildlife Service candidate assessment form), and likely remains a threat. In addition, there are unpublished reports of hellbenders accidently killed by frog giggers, who may gig a hellbender inadvertently. When considered cumulatively, collection and illegal or unintentional harvest is a threat to many of the declining hellbender populations. Because the species is long lived and does not reproduce until approximately age 7, the removal of even a few individuals from a population that is experiencing declines can impact the recruitment potential of that population. Presently, collecting levels appear reduced (LaClaire 1993), but collecting could become more of a threat if populations continue to decline.

C. ***Disease or predation.***

The occurrence of disease is virtually unknown in Ozark hellbender populations and has been studied little. Although young hellbenders are occasionally preyed upon by large fish, turtles, and water snakes, this is rare due to their noxious skin secretions and likely does not occur after hellbenders reach 380 mm (Nickerson and Mays 1973a, Peterson et al. 1983). It is unlikely that an otherwise healthy population would be threatened by natural levels of predation. No evidence has been presented that would indicate that disease or predation are serious threats.

D. ***The inadequacy of existing regulatory mechanisms.***

The states of Arkansas and Missouri prohibit the taking of Ozark hellbenders for any purpose without a state scientific collecting permit. However, enforcement of this permit requirement is difficult. Additionally, state regulations do not protect hellbenders from other threats. Existing authorities available to protect riverine ecosystems, such as the Clean Water Act (CWA), administered by the Environmental Protection Agency (EPA) and the U.S. Army Corps of Engineers, may not have been fully exercised in an effort to prevent in-stream activities and the resulting habitat degradation. This may have contributed to the general habitat degradation apparent in riverine ecosystems and decline of both eastern and Ozark hellbender populations throughout their ranges. Although the Ozark hellbender coexists with other federally listed species throughout parts of its range, listing under the Endangered Species Act would provide additional protection, as the threats to hellbenders and the other endangered species are not identical.

Currently, there are no regulations governing best management practices (BMPs) of timber harvesting, which would reduce impacts on water quality. Existing BMPs are established by the Arkansas Forestry Commission and Missouri Department of Conservation and lack mandatory requirements for implementing methods to reduce aquatic resource impacts associated with timber harvests. Many timber harvests involve clear-cutting to the streambank, which promotes bank erosion.

Current Conservation Efforts: No conservation agreements have been developed for the Ozark hellbender. However, the states of Arkansas and Missouri have recognized the need for conservation of this species. Although the species is not state listed, Missouri has provided extra protection for the Ozark hellbender in the Wildlife Code of Missouri, outlawing collection of hellbenders. Outreach has been considerable in both states, which have erected signs throughout the range of the Ozark hellbender alerting recreationists to their presence. Additionally, numerous stream surveys have been conducted by both states. The Missouri Department of Conservation is considering the inclusion of the Ozark hellbender in their five-year threatened and endangered species plan, with tributary surveys and life history studies included (U.S. Fish and Wildlife Service candidate assessment form). Presently, the U.S. Fish and Wildlife Service, U.S. Geological Survey, and Arkansas Game and Fish Commission have funded surveys to fill in unsurveyed gaps in the distribution of the species in Arkansas and Missouri, and work is being done at Mammoth Springs National Fish Hatchery to examine potential refugia as well as life history characteristics (U.S. Fish and Wildlife Service candidate assessment form).

E. ***Other natural or manmade factors affecting its continued existence.***

Certain population characteristics of Ozark hellbenders cause the species to be fairly vulnerable to local extirpations and extinction. The Ozark hellbender, having specialized habitat requirements, is extremely vulnerable to environmental perturbations. When populations are small, they are less likely to rebound following these perturbations. In addition, Ozark hellbenders exhibit very low genetic diversity (Merkle et al. 1977, Wagner et al. 1999). This genetic uniformity is consistent with habitat specialization (Nevo 1978, Wagner et al. 1999). Ozark hellbenders have adapted to a relatively constant environment and a variety of structural, behavioral, and physiological specializations have resulted (Williams et al. 1981). These specializations, in combination with the stable environment, seems to have resulted in very low levels of genetic diversity (Wagner et al. 1999). Fragmentation of populations by impoundments, habitat degradation, and other impediments to dispersal may exacerbate this situation.

Without the level of interchange the hellbender experienced historically, many small, isolated populations do not receive the influx of new genetic material that once occurred. As the populations decrease in size, genetic diversity is lost and inbreeding can occur, which may result in decreased fitness, and the loss of genetic heterozygosity can result in a significantly increased risk of extinction in localized natural populations (Saccheri et al. 1998). This is illustrated by Routman's (1983) study, in which hellbender populations from different rivers showed very little within-population variability, and relatively high between-population variability. Due to this population fragmentation, local extinctions cannot be repopulated.

Ozark hellbenders do not reproduce until approximately 7 years of age. Declines being observed presently may be the result of activities that occurred years earlier. Because juvenile hellbenders are rarely observed, it takes many years to detect population trends. The lack of recruitment in most Ozark hellbender populations is a significant sign that little reproduction has occurred in these populations for several years. Delayed reproduction, when paired with a long life span, can disguise declines until they become severe.

The present distribution and status of Ozark hellbender populations in the White and Black River systems in Arkansas and Missouri may be demonstrating the characteristics mentioned above. Genetic studies have repeatedly demonstrated very low genetic diversity in hellbender populations, which may be a factor in the decline of the species. The current combination of population fragmentation and habitat degradation may prohibit this species from recovering without the intervention of conservation measures carefully designed to facilitate hellbender recovery.

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PETITION TO LIST

Georgetown salamander
(*Eurycea naufragia*)

AS A FEDERALLY ENDANGERED SPECIES

CANDIDATE HISTORY

CNOR 11/15/94:	
CNOR 10/30/01:	C
CNOR 6/13/02:	C

TAXONOMY

The Georgetown salamander (*Eurycea naufragia*) (Plethodontidae) was recognized as a distinct species by Chippindale et al. (2000). This species was formerly included in the *Eurycea neotenes* species group. To maintain consistency with the U.S. Fish and Wildlife Service candidate species list, we use here the common name Georgetown salamander, although the common name recognized by the Center for North American Herpetology is San Gabriel Springs salamander (Collins and Taggart 2002).

NATURAL HISTORY

Adults are about 2 inches long. Georgetown salamanders are characterized by a broad, relatively short head with three pairs of bright-red gills on each side behind the jaws, a rounded and short snout, and large eyes with a gold iris. The upper body color is generally greyish with varying patterns of melanophores and iridophores, while the underside is pale and translucent. The tail tends to be long with poorly-developed dorsal and ventral fins that are golden-yellow at the base, cream-colored to translucent toward the outer margin, and mottled with melanophores and iridophores. Unlike the Jollyville Plateau salamander, the Georgetown salamander has a distinct dark border along the lateral margins of the tail fin (Chippindale et al. 2000).

The Georgetown salamander is entirely aquatic and neotenic, meaning it does not metamorphose into a terrestrial adult.

The Georgetown salamander is known from springs along 5 tributaries (South, Middle, and North forks; Cowan Creek; and Berry Creek) to the San Gabriel River and one cave in the City of Georgetown, Williamson County, Texas.

POPULATION STATUS

Populations within the City of Georgetown proper probably are on the brink of extinction (Chippindale et al. 2000). Development of retirement and leisure communities (Sun City Georgetown), and quarrying (Middle Fork San Gabriel River), are occurring near some salamander populations, but currently these do not appear to jeopardize salamander habitat (Chippindale et al. 2000). Because this species spends a portion of its life underground, and the technology to safely and reliably mark salamanders for individual recognition has not been developed, population estimates are not possible at this time. However, anecdotal information suggests population declines and individual deformities following water quality degradation.

The U.S. Fish and Wildlife Service classifies the Georgetown salamander as a candidate for Endangered Species Act protection with a listing priority number of 2. The Texas Natural Heritage Program ranks the Georgetown salamander as Critically Imperiled.

LISTING CRITERIA

A. *The present or threatened destruction, modification, or curtailment of its habitat or range.*

Historical range:	Texas, Williamson County.
Current range:	Texas, Williamson County. Springs and possibly one cave associated with drainages of the south, middle, and north forks of the San Gabriel River.
Land ownership:	Based on the known range, it appears that all of the Georgetown salamander locations are under private ownership.

Primary threats include degradation of water quality and quantity due to urbanization. The Georgetown salamander occurs in an area that is undergoing rapid urban expansion. Williamson County grew 7.7% between 1998 and 1999 (U.S. Census Bureau 2000). Based on population projections from the Texas State Data Center (2000), the population of Williamson County in 2030 is projected to be 7 times the size of the 1990 population (projected increase from 139,551 to 989,139). Georgetown is the fastest growing city in Williamson County, and Williamson County is the second fastest growing non-urban county in the United States (Georgetown Chamber of Commerce 2000).

Urbanization can dramatically alter the normal hydrologic regime and water quality of an area. As areas are cleared of natural vegetation and replaced with impervious cover, rainfall no longer percolates through the ground but instead is rapidly converted to surface runoff (Schueler 1991). Streamflow shifts from predominantly baseflow, which is derived from natural filtration processes and discharges from local groundwater supplies, to predominantly stormwater runoff.

The amount of stormwater runoff tends to increase in direct proportion to the amount of impervious cover (Arnold and Gibbons 1996). With increasing stormflows, the amount of baseflow available to sustain water supplies during drought cycles is diminished, and the frequency and severity of flooding increases. Increasing stormflows result in less water recharging the aquifer, thereby diminishing baseflow. The increased quantity and velocity of runoff increases erosion and streambank destabilization, which in turn leads to increased sediment loadings, channel widening, and detrimental changes in the morphology and aquatic ecology of the affected stream system (Schueler 1991, Arnold and Gibbons 1996).

Even at relatively low levels of impervious cover, profound and often irreversible impacts to the hydrology, morphology, water quality, habitat, and biodiversity of streams can occur (Schueler 1994). Both nationally and locally, consistent relationships between impervious cover and water quality degradation have been documented. The extent to which impervious cover is controlled in a watershed has been linked with indices of environmental health (Schueler 1994; City of Austin 1998).

Research suggests that increases in impervious cover exceeding 10 percent are associated with measurable water quality degradation, loss of sensitive aquatic organisms, reduction in stream biodiversity, stream warming, and channel instability within a watershed (Schueler 1994). Stream aquatic life problems have been identified with watersheds having impervious cover of at least 12 percent, with severe problems in watersheds with impervious cover greater than 30 percent. Generally, stream quality impairment can be prevented if watershed imperviousness does not exceed 15 percent and for more sensitive stream ecosystems watershed imperviousness should not exceed 10 percent (Klein 1979).

Chippindale et al. (2000) state that populations of Georgetown salamanders in the City of Georgetown are on the brink of extinction. Populations along Cowan Creek lie within the Sun City Georgetown retirement community, designed to accommodate 9,000 homes. Salamander sites along the Middle Fork of the San Gabriel River are near and downstream from a large quarry (Chippindale et al. 2000). Many of the springflows have been reduced and the San Gabriel River raised to the point that the direction of flow is often reversed, with river water flowing into the springs (Price et al. 1995).

B. *Overutilization for commercial, recreational, scientific, or educational purposes.*

None known.

C. *Disease or predation.*

None known.

D. *The inadequacy of existing regulatory mechanisms.*

No Federal, State, or local laws provide for the protection of the Georgetown salamander.

Current Conservation Efforts: None.

E. *Other natural or manmade factors affecting its continued existence.*

The Georgetown salamander has a very limited distribution and appears to be highly sensitive to water quality and quantity degradation. Research indicates that amphibians, particularly their eggs and larvae, are sensitive to many pollutants, such as heavy metals; certain insecticides, particularly cyclodienes (endosulfan, endrin, toxaphene, and dieldrin) and certain organophosphates (parathion, malathion); nitrite; salts; and petroleum hydrocarbons (Harfenist et al. 1989).

Because of their semipermeable skin, the development of their eggs and larvae in water, and their position in the food web, amphibians can be exposed to waterborne and airborne pollutants in their breeding and foraging habitats. Toxic effects to amphibians from pollutants may be either lethal or sublethal, including morphological and developmental aberrations, lowered reproduction and survival, and changes in behavior and certain biochemical processes. Since the salamander is fully aquatic, there is no possibility for escape from contamination or other threats to its habitat. Crustaceans, particularly amphipods, on which the salamander may feed are especially sensitive to water pollution (Mayer and Ellersieck 1986; Phipps et al. 1995; Burton and Ingersoll 1994).

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PETITION TO LIST

Black Warrior waterdog
(*Necturus alabamensis*)

AS A FEDERALLY ENDANGERED SPECIES

CANDIDATE HISTORY

CNOR 11/21/91:	
CNOR 11/15/94:	
CNOR 10/25/99:	C
CNOR 10/30/01:	C
CNOR 6/13/02:	C

TAXONOMY

The Black Warrior waterdog, *Necturus alabamensis* (Proteidae), was first described more than half a century ago from the upper Black Warrior River in Alabama (Viosca 1937). In subsequent years, the name *N. alabamensis* was mistakenly applied to a more common species of waterdog that occurs on the Gulf coastal plain from Lake Pontchartrain through the Ochlockonee River drainage (taxonomic history reviewed in Bart et al. 1997; also see Petranka 1998 and Collins and Taggart 2002). Clarifying the proper name for this more common and more broadly distributed *Necturus* will require further study. For now, Bart et al. (1997) suggest that this latter form be referred to as *Necturus* sp. cf. *beyeri*, with the name *Necturus alabamensis* properly applied to the distinct and endangered species that is limited to the upper Black Warrior River.

NATURAL HISTORY

Morphology

The Black Warrior waterdog is a large, gilled aquatic salamander with a maximum recorded length of 248 millimeters (9.8 inches) (Bailey 1995). The Black Warrior waterdog has a flattened body and distinctive pigmentation**.** bushy external gills, two gill slits, a laterally compressed tail, and four toes on front and hind feet. The dorsum is reddish brown to nearly black. Some populations have spots on the dorsum. The venter usually lacks spots. Tips of the toes are light colored. The body and head are flattened. Sexually mature males can be distinguished by the swollen cloaca and pair of enlarged cloacal papillae that project posteriorly. Hatchlings are mottled dorsally with a few light spots. In some populations, juveniles have light stripes on the head and back, similar to juveniles of *N. maculosus* (Neill, 1963; Bart et al., 1997; Petranka, 1998).

Behavior
The Black Warrior waterdog spends virtually all of its life at the bottom of streams. Eggs are laid in spring, attached to the undersides of objects in water, and hatch in 4-6 weeks. This species is paedomorphic. Food items include crayfish, worms, snails, small fishes, and other small aquatic animals. It is nocturnal (occasionally active during the day) and apparently active all year.

Information on the Black Warrior waterdog is limited. It received little attention between the time it was described in 1937 and the mid-1980's when it was found during surveys in the Tenn-TomWaterway (Ashton and Peavy 1985). During this time, reference to the species, beyond field guides and summary descriptions, could be found in only three scientific publications and one unpublished Ph.D. dissertation (Hecht 1958, Neil 1963, Gunter and Brode 1964, Brode 1969).

Habitat
The Black Warrior waterdog inhabits streams above the Fall Line within the Black Warrior River Basin (Basin) in Alabama, including parts of the North River, Locust Fork, Mulberry Fork, and Sipsey Fork drainages and their tributaries. Rocks, submerged ledges, and other cover probably play an important role in determining habitat suitability (Ashton and Peavy 1986). Semi-permanent leaf beds (where they exist) are likely visited frequently (Ashton and Peavy 1986). Guyer (1997) analyzed habitat to distinguish sites with waterdogs from those lacking the species. He found that Black Warrior waterdogs were associated with clay substrates lacking silt; wide and/or shallow stream morphology; increased snail and *Desmognathus* (dusky salamanders) abundance; and decreased *Corbicula* (Asiatic clam) occurrence.

Guyer and Durflinger (1999) conducted a demographic study at the best Black Warrior waterdog population. At this locality, they sampled an area of approximately 840 meters^2 (m^2) (2,756 feet (ft^2)). Within this area, all the captured waterdogs occurred in a 40 m^2 (131 ft^2) area in leaves accumulated at the base of a large dead tree that had fallen into the river. This demonstrates the importance of leaf packs for cover. All larval waterdogs captured over the years have been found exclusively in leaf packs.

Distribution
Records of the historical distribution of the Black Warrior waterdog are few. This species can be expected to potentially inhabit the same streams as the threatened flattened musk turtle (*Sternotherus depressus*), which is also restricted to permanent streams above the Fall Line in the Basin (Mount 1975). The Black Warrior waterdog is thought to have occurred in large streams (10 m (33 ft) wide or greater), with moderate flows and alternating pools and rapids, throughout the Basin (Ashton and Peavy 1986, Bailey 1992). One hundred and twenty sites have been sampled for waterdogs since 1990 (Guyer 1997).

The species has been reported recently from only ten sites (8 percent success rate) in Blount, Tuscaloosa, Walker, and Winston Counties, Alabama, despite surveys in 1990, 1991, 1992, 1994, 1996, 1997, and 1998 (Bailey 1995, Guyer 1997, 1998). Survey sites included all stream localities within the range of the species that approached or intersected roads and had appropriate habitat. Guyer (1997) did a statistical analysis of all waterdog field survey data. He concluded that waterdogs were unlikely to have been missed if they were present, especially at sites visited more than once. The data suggested that 200 additional surveys would be needed to discover a single new locality for the species. Bailey (2000) conducted a habitat assessment of all sites (12) which have historical records for the Black Warrior waterdog. Two sites were subsequently combined because of their proximity to each other (separated by less than a mile of stream). This adjusted the total to 11 historical sites. Assessments were based on subjective impressions of habitat suitability using parameters such as stream width and depth, water quality, substrate, structure (crevices, logs, etc.), and invertebrate fauna. Sites were stratified into four categories: good to excellent, moderate, poor to unsuitable, and impounded.

Bailey (2000) concluded that two (18%) of the sites were good to excellent, 3 (27%) were moderate in quality, two (18%) were poor to unsuitable, and four (36%) were in impoundments. Two of the impounded sites were based on historical collections made prior to the impoundments. The other two records of the species from impoundments were based on the capture of one animal at each site.

POPULATION STATUS

Fewer than 1,000 individual waterdogs exist in fewer than 2,000 acres, in fewer than 10 miles of stream length. The species is rare with sporadic occurrences within the presumed geographic range (Guyer 1997). Remaining Black Warrior waterdog habitat is currently degraded by acid mine drainage, feedlot and agricultural runoff, pesticides, sedimentation from roads, and urban development and logging operations. Habitat degradation is the primary factor that has reduced the distribution of viable flattened musk turtle populations to an estimated 15 percent of their historical distribution in the upper Black Warrior system (U.S. Fish and Wildlife Service 1990). Black Warrior waterdogs have probably experienced similar declines.

The U.S. Fish and Wildlife Service classifies the Black Warrior waterdog as a candidate for Endangered Species Act protection with a listing priority number of 5. The Alabama Natural Heritage Program ranks the Black Warrior waterdog as Imperiled.

LISTING CRITERIA

A. ***The present or threatened destruction, modification, or curtailment of its habitat or range.***

Historical range: Black Warrior River drainage, Alabama. Above the Fall Line within the Black Warrior River Basin (Basin) including parts of the North River, Locust Fork, Mulberry Fork, and Sipsey Fork drainages and their tributaries.

Current range: The species has been reported recently from only ten sites (8 percent success rate) in Blount, Tuscaloosa, Walker, and Winston Counties, Alabama, despite surveys in 1990, 1991, 1992, 1994, 1996, 1997, and 1998 (Bailey 1995, Guyer 1997, 1998).

Land ownership: Federal ownership 10 percent (Bankhead National Forest); private ownership 90 percent.

Water quality degradation is the biggest threat to the continued existence of the Black Warrior waterdog. Bailey (1995) considered water quality degradation to be the primary reason for the extirpation of this species over much of its historic range in the upper Black Warrior River system. Most streams surveyed for the Black Warrior waterdog showed evidence of water quality degradation and many appeared biologically depauperate (Bailey 1992, 1995, Guyer 1997).

Sources of point and nonpoint pollution in the Black Warrior Basin have been numerous and widespread. Water quality, and the resident aquatic fauna, have declined as a result. Pollution is generated from inadequately treated effluent from industrial plants, sanitary landfills, sewage treatment plants, and drain fields from individual private homes (U.S. Fish and Wildlife Service 1998). Poultry and cattle feedlots are other major contributors of pollution to the drainage (Deutsch et al. 1990).

The large population centers of Birmingham, Tuscaloosa, and Jasper contribute substantial runoff to the Basin. The watershed occupied by these three cities contains more industrial and residential land area than any other river basin in the State. Streams draining these areas have a history of serious water quality problems. Species of fishes, mussels, and snails (Mettee et al.1989, Hartfield 1990), and populations of the flattened musk turtle (U.S. Fish and Wildlife Service 1990), have been extirpated from large areas of the watershed due primarily to water quality degradation.

Mettee et al. (1989) noted the absence of at least nine fish species from streams draining the Birmingham metropolitan area where they were previously common. These species were otherwise abundant and easily collected in the lower Sipsey, Mulberry, and Locust Forks. Hartfield (1990) documented the extirpation of most species of mussels from tributaries of the Black Warrior River. He conducted extensive surveys of sites where mussels had been collected previously. Although historically the Black Warrior River Basin supported at least 45 species, only five species of live or fresh dead mussels were found on the Locust Fork, six species on the Mulberry Fork and its tributaries, and six species on the Sipsey Fork. Locust Fork tributaries had little evidence of an extant unionid fauna. This was reflected in the lack of mussel shell in muskrat middens (refuse heaps), which were composed entirely of *Corbicula*.

Surface mining represents another threat to the biological integrity of streams in the Black Warrior River system and has undoubtedly affected the distribution of the Black Warrior waterdog (Bailey 1995). Strip mining for coal results in hydrologic problems (e.g., erosion, sedimentation, decline in groundwater levels, and general degradation of water quality) that affect many aquatic organisms (U.S. Fish and Wildlife Service 1998).

Runoff from coal surface mining generates pollution through acidification, increased mineralization, and sediment loading. Impacts are generally associated with past activities and abandoned mines, since presently operating mines are required to employ environmental safeguards established by the Federal Surface Mining Control and Reclamation Act of 1977 and the Clean Water Act of 1972 (U.S. Fish and Wildlife Service 1998). Old, abandoned mines will continue to contribute pollutants to streams for the foreseeable future. At present levels of manpower and funding, it will take 166 years to reclaim known mines in the Basin (personal communication 1999 cited in U.S. Fish and Wildlife Service candidate assessment form).

Forestry operations and highway construction are also sources of nonpoint pollution when Best Management Practices (BMPs) are not followed to protect streamside management zones (Hartfield 1990, U.S. Fish and Wildlife Service 1998). Logging can cause erosion, siltation, and stream bed structural changes from the introduction of tree slash. Highway construction and bridge replacements can also result in increased sedimentation, and runoff may introduce toxic chemicals into streams. In addition, highway construction may reroute streams or change their shape.

Dodd et al. (1986) concluded that sedimentation in the upper Black Warrior River system negatively affected the flattened musk turtle by: (1) reduction of mollusks and other invertebrates used as food; (2) physical alteration of rocky habitats where the animals forage and take cover, and (3) accumulation of substrate in which chemicals toxic to animals and their prey persist. Habitat degradation is the primary factor that has reduced the distribution of viable flattened musk turtle populations to an estimated 15 percent of their historical distribution in the upper Black Warrior system (U.S. Fish and Wildlife Service 1990). Black Warrior waterdogs have probably experienced similar declines. They are vulnerable to sedimentation since they spend virtually all of their lives at the stream bottom. Therefore, they are in almost constant contact with any toxic sediments that may be present (Bailey 1995).

Creation of large impoundments within the Black Warrior Basin has flooded thousands of square hectares of habitat previously considered appropriate for the Black Warrior waterdog. Impoundments do not have the shallow, flowing water preferred by the species. As a result, they are likely marginal or unsuitable habitat for the salamander.

The abundance of predatory fish in impoundments further renders these lakes unsuitable for the Black Warrior waterdog. Impoundments have been trapped for waterdogs and flyers have been circulated (offering a reward for the species) to 187 bait shops, marinas, conservation officers, and other individuals throughout the target area (Bailey 1995, Guyer 1997). As a result of these efforts, only three Black Warrior waterdogs have been reported from impoundments (Bailey 2000). All three specimens were captured by fishermen fishing off a bank or near streams that empty into the reservoirs. The question remains whether impoundments represent suitable habitat or are habitat sinks. Given the habitat requirements of the species, it seems unlikely that a viable population of Black Warrior waterdogs could be sustained in an impoundment.

Hartfield (1990) summarized the number of miles of streams affected by these impoundments. He found that the entire main channel of the Black Warrior River, over 272 kilometers (km) (170 miles (mi)), has been affected. At least 32 km (20 mi) of the lower reach of the Locust Fork, 64 km (40 mi) of the lower Mulberry Fork, 48 km (30 mi) of the North River, and 48 km (30 mi) of the Sipsey Fork (and at least as many kilometers of its tributaries) have been impounded or are affected by impoundments. The Sipsey Fork is the best remaining locality for the Black Warrior waterdog (Guyer 1998). Bailey and Guyer (1998) recently completed a study of the flattened musk turtle at this site. They found that the turtle population was declining and suggested that habitat quality is deteriorating at this site.

B. *Overutilization for commercial, recreational, scientific, or educational purposes.*

Direct take of Black Warrior waterdogs for commercial, recreational, scientific, or educational purposes is not currently considered to be a threat.

C. *Disease or predation.*

Disease and predation are not known to be factors in the decline of the Black Warrior waterdog.

D. *The inadequacy of existing regulatory mechanisms.*

The State of Alabama provides no protection for the Black Warrior waterdog (personal communication 1999 cited in U.S. Fish and Wildlife Service candidate assessment form). The Federal Surface Mining Control and Reclamation Act of 1977 and the Clean Water Act of 1972 have been ineffective in preventing the continued decline of species in the Black Warrior Basin (Dodd et al. 1986, Mettee et al.1989, Hartfield 1990, Bailey and Guyer 1998, U.S. Fish and Wildlife Service 1998).

Current Conservation Efforts: None.

E. *Other natural or manmade factors affecting its continued existence.*

The remaining Black Warrior waterdog populations are isolated from each other by unsuitable habitat created by impoundments, pollution, or other factors. The fragmentation of habitat renders populations vulnerable to catastrophic events such as flood, drought, or chemical spills. In addition, even if stream quality improves within areas of the Basin, impoundments and polluted reaches will act as barriers to reestablishment of Black Warrior water dog populations.

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PETITION TO LIST

Sonoyta mud turtle
(*Kinosternon sonoriense longifemorale*)

AS A FEDERALLY ENDANGERED SPECIES

CANDIDATE HISTORY

CNOR 9/19/97:	C
CNOR 10/25/99:	C
CNOR 10/30/01:	C
CNOR 6/13/02:	C

TAXONOMY

The Sonoyta mud turtle (*Kinosternon sonoriense longifemorale*) is one of two recognized subspecies of the Sonoran mud turtle (Iverson 1981; Ernst et al. 1994).

NATURAL HISTORY

The Sonoyta mud turtle lives in a variety of aquatic habitats, such as springs, creeks, and ponds. It feeds on insects, crustaceans, snails, fish, frogs, and plants. Females lay a clutch of two to nine eggs, buried in soil on land, between May and September.

POPULATION STATUS

The Sonoyta mud turtle occurs only in one pond and limited stream habitat area at Quitobaquito Springs in Organ Pipe Cactus National Monument, Arizona, and in the nearby Rio Sonoyta, Sonora, Mexico (Ernst et al. 1994). The subspecies was once abundant at Quitobaquito, but the population declined from probably several hundred in the 1950s to less than 100 in the late 1980s. Juvenile survivorship has increased in recent years, and the population in 1995 was estimated at about 130 individuals (Rosen and Lowe 1996a). In the Rio Sonoyta, the subspecies is known historically from the Rio Sonoyta at and near the highway bridge in Sonoyta, Sonora, and from a perennial reach south of Quitobaquito, but the turtles may have been extirpated from the area near the highway bridge in 1987-1989 (Rosen and Lowe 1996b).

The Sonoyta mud turtle possibly occurs or occurred in other perennial reaches, which are distributed patchily and in short reaches over approximately 40 kilometers (25 miles) of the streamcourse (McMahan and Miller 1982).

The U.S. Fish and Wildlife Service classifies the Sonoyta mud turtle as a candidate for Endangered Species Act protection with a listing priority number of 3.

LISTING CRITERIA

A. ***The present or threatened destruction, modification, or curtailment of its habitat or range.***

Historical range:	Arizona and Sonora, Mexico (extent of former range uncertain)
Current range:	A single pond and limited stream habitat area at Quitobaquito Springs in Organ Pipe Cactus National Monument, Arizona, and in the nearby Rio Sonoyta, Sonora, Mexico (Ernst et al. 1994).
Land ownership:	In the United States, 100 percent of the turtle's habitat is owned by the National Park Service.

Quitobaquito is a dredged and impounded pond fed by springs and seeps in nearby granite outcrops. Flow from springs may have been connected to the Rio Sonoyta via surface flows in recent times, but is now separated by approximately 1.5 km of Sonoran desert and Mexico Highway 2. The effects of the original dredging and impoundment on the Sonoyta mud turtle are unknown. However, the imperilled status of the turtle was apparently unknown to Park Service personnel for many years. The pond at Quitobaquito was drained twice to eliminate nonnative fish and enhance habitat for the endangered desert pupfish. During these drying episodes many turtles were collected and given away as pets (Rosen 1986). The pond at Quitobaquito could silt in over time, or the dam could fail in a storm, with devastating consequences for the Sonoyta mud turtle.

Rio Sonoyta is a disjunct stream of the Colorado River system that was likely isolated in the Pinacate Region during a volcanic activity period in the Pleistocene (Ives 1936, Hubbs and Miller 1948). Aquatic habitat in the Rio Sonoyta is being lost and degraded due to groundwater pumping, livestock grazing, and pesticide application (McMahon and Miller 1982, Hendrickson and Varela-Romero 1989, Rutman 1997). The introduction of non-native bullfrogs (*Rana catesbeiana*), which may prey on turtles, is a possible threat, and similar potential threats would be posed by the introduction of non-native fish, such as largemouth bass (*Micropterus salmoides*).

B. ***Overutilization for commercial, recreational, scientific, or educational purposes.***

The subspecies has been illegally collected at Quitobaquito (Rosen and Lowe 1996b), but the extent of this activity is unknown. Collecting pressure in the Rio Sonoyta is unknown, but turtles may be taken by children and killed by stray dogs. Because of low population sizes and limited reproductive potential, any collecting, particularly of adult female turtles, could be critically harmful to population viability.

C. *Disease or predation.*

No non-native predators capable of consuming mud turtles or their eggs are known from Quitobaquito or the Rio Sonoyta, with the exception of feral and domestic cats and dogs in and near Sonoyta. Introduction of non-native bullfrogs is a potential threat. Bullfrogs are known to prey on turtles and may be capable of impacting populations of mud turtles (Schwalbe and Rosen 1988). Concern has also been expressed over possible non-native fish introduction into Quitobaquito. Some non-native species, such as largemouth bass, are capable of preying on mud turtles. However, as yet largemouth bass are not known from Quitobaquito or the Rio Sonoyta. As recently as 1993, a new introduced species, the black bullhead (*Amieurus melas*) was collected in Quitobaquit0.

A study of turtles found dead between 1989 and 1993 and pond sediments from Quitobaquito Springs was conducted. Mud turtles from Quitobaquito exhibited relatively low body lipid (fat) reserves, indicating a possible dietary deficiency. Relatively high levels of boron, chromium, selenium, strontium, and zinc in mud turtle tissues, combined with low availability of protein rich foods may be limiting turtle survival (King et al. 1996). Low lipid reserves may also result in reduced egg production. Pesticide use in agricultural lands along the Rio Sonoyta may contaminate habitats of the turtle: low levels of DDE metabolites and Dacthal, an herbicide, were found in mud turtles from Quitobaquito since 1981 (Rosen and Lowe 1996a). The effects of such pesticides on this species are unknown.

D. *Inadequacy of existing regulatory mechanisms.*

Collection of mud turtles from Organ Pipe Cactus National Monument is illegal except by special permit. However, law enforcement coverage is limited and some illegal collection occurs. Arizona State law does not prohibit collection of the Sonoyta mud turtle; the bag limit is four per year, live or dead.

Current Conservation Efforts: The U.S. Fish and Wildlife Service has begun discussions with Organ Pipe Cactus National Monument about the status of and potential conservation measures for this subspecies. The Phoenix Zoo has expressed interest in propagating Sonoyta mud turtles and perhaps establishing a captive population on the zoo grounds. A mailing list has been prepared for the pre-notification status summary status and information letter. A monitoring study was scheduled to begin in 2001.

E. *Other natural or manmade factors affecting its continued existence.*

Aquatic habitat in the Rio Sonoyta is extremely dynamic due to climatic extremes (Ives 1936, Hendrickson and Varela-Romero 1989). Mud turtle populations are likely reduced due to the dynamic nature of their habitat. Because turtle populations have a low intrinsic population growth rate, they are incapable of expanding rapidly to take advantage of temporary habitats created by periods of high precipitation, but populations can decline rapidly during drought years. Also, populations of mud turtles are relatively small. Small populations are vulnerable to environmental and demographic random events, which increase the probability of extinction (Shafer 1990).

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PETITION TO LIST

Cagle's map turtle
(*Graptemys caglei*)

AS A FEDERALLY ENDANGERED SPECIES

CANDIDATE HISTORY

CNOR 12/30/82:	
CNOR 9/18/85:	
CNOR 1/06/89:	
CNOR 11/15/94:	
CNOR 2/28/96:	C
CNOR 9/19/97:	C
CNOR 10/25/99:	C
CNOR 10/30/01:	C
CNOR 6/13/02:	C

TAXONOMY

Graptemys caglei is the accepted taxon for Cagle's map turtle.

NATURAL HISTORY

Morphology

Highly aquatic river turtle. a greenish color with the typical map turtle pattern on the shell. Males reach 4.5 inches while females can be as large as seven inches. G. caglei is a narrow-head species with high carapacial keels.

Behavior

Lays eggs in late spring or early summer. Hatchlings have been found September-November. Exhibits temperature-dependent sex determination, as do all GRAPTEMYS (Wibbels et al. 1991). As many as three clutches may be laid yearly containing one to six eggs (Vermersch 1992). Eggs are deposited in nests located near the water in cavities approximately 15 cm deep. Males spend much of their time in gravel bar riffles and transition areas between pools and riffles. Adult males feed primarily on insects (e.g., caddisfly larvae and immature stages of other aquatic insects; adult females feed mainly on mollusks (e.g., Asiatic clam); juveniles eat aquatic insects, snails, and clams (USFWS 1993). Similar to other map turtles, this species is wary and difficult to approach. With the exception of nesting, this species rarely comes onto land (Vermersch 1992).

Habitat

The species lives in riverine habitat with river bed mostly silt and gravel, and with gravel bars connecting long pool areas with a shallow average depth and a muddy moderate flow; optimal habitat appears to include both riffles and pools (riffles may be an important producers of insect prey); basking sites include fallen trees and shrubs, logs, rocks, and cypress knees (see USFWS 1993). Some nests are found on sand bars, but much of the habitat lacks sand bars.

Distribution

This highly aquatic river turtle is confined to the Guadalupe River system of Texas and optimal habitat appears to include both riffles and pools (Haynes and McKown 1974, Killebrew 1991, Killebrew 1992). Gravel bar riffles and transition areas between riffles and pools are considered to be important since these areas are considered to be highly productive of insect prey items (Killebrew 1991). Historical populations were reported from the Guadalupe-San Antonio River System (Haynes and McKown, 1974), including voucher specimens from Kerr, Hays, Gonzales, and DeWitt counties and sight records from the San Antonio and Medina rivers. Collectively Dixon (1987), Killebrew (1991) and Porter (1992) determined that the Cagle’s map turtle occurs in scattered sites in seven counties (Kerr, Kendall, Comal, Guadalupe, Gonzales, Dewitt, and Victoria) on the Guadalupe, San Marcos, and Blanco rivers.

Historical population numbers are unknown. Current population sizes have been estimated by marking and recapturing over 1,000 turtles. Preliminary analyses (Killebrew, pers. comm., 1994) indicate that the small population on the upper Guadalupe River (above Canyon Lake) contains no more than 400 individuals. A population model based upon ten years of data collected from a 36 kilometer (22 mile) stretch of the Guadalupe River near Cuero yielded a population estimate of between 1,354 and 2,184 individuals (Killebrew and Babitzke 1996). Below Canyon Dam, the large population on the middle Guadalupe and lower San Marcos rivers contains an estimated 11,300 individuals (Killebrew, pers. comm., 1994) including a 200-km (124-mi) core segment of the Guadalupe River between Seguin and Cuero (60-70% of total population), decreasing in abundance downstream to Victoria (83 km or 51 mi), and including a few turtles on the San Marcos River from its mouth upstream to Ottine (37 km or 23 miles)(Porter 1992).

POPULATION STATUS

Much of the species' habitat loss is due to reservoir construction, water diversions, water quality degradation, and human depredation (collection for pet trade and intentional shooting). Over 50% of the suitable habitat would be eliminated by construction of the Cuervo Reservoir. Five other reservoirs are proposed along tributaries to the Guadalupe River. When dams are present, the water quality in the river is altered. Silt covers rocks and decreases the number of riffles in the ecosystem, thus reducing the foodstuff for male turtles.

The U.S. Fish and Wildlife Service classifies the Cagle's map turtle as a candidate for Endangered Species Act protection with a listing priority number of 5. The Texas Parks and Wildlife Department (TPWD) listed the Cagle's map turtle as threatened, effective November 16, 2000 (Dorinda Scott, TPWD, pers. comm. 2002). The International Union for the Conservation of Nature classifies the species as "vulnerable."

LISTING CRITERIA

A. ***The present or threatened destruction, modification, or curtailment of its habitat or range.***

Historical range: Small range in the Guadalupe River system, Texas

Current Range: Small range in the Guadalupe River system, Texas

Land Ownership: Private

Loss and degradation of riverine habitat from large and small impoundments (dams or reservoirs)
is the primary threat to Cagle's map turtle. Cagle's map turtle is absent from deep water/non-riverine habitat in its range (Killebrew 1991).

Cagle's map turtles occur where the Guadalupe River empties into Canyon Lake (a 3335 hectare (8,240 acre) reservoir) and above the reservoir, but not in the lake proper (Killebrew 1991). The water released from the deeper and cooler portion of Canyon Lake may decrease the suitability of
riverine habitat for Cagle's map turtle below Canyon Dam. Cagle's map turtle has been observed in only one small, warm pool between Canyon Lake and New Braunfels (Killebrew 1991).
One effect of impoundment is the loss of riffle and riffle/pool transition areas used by males for foraging. Depending on its size, a dam itself may be a partial or complete barrier to Cagle's map turtle movement and could fragment a population. Construction of smaller impoundments and human activities on the river have likely eliminated or reduced foraging and basking habitats.

Senate Bill 1, comprehensive water legislation for Texas, was enacted in June 1997. With the expected growth in Texas, the 75th Legislature put in place a water planning process designed to ensure future water needs for Texas. Texas was divided into 16 regions that will individually provide options for future water needs. Cagle's map turtle is located within Region L. As of August 2000, Region L proposed 79 possible scenarios for meeting their future water needs. Several of the proposals have the potential to affect Cagle's map turtle by altering flow and physical habitat of the Guadalupe River and existing Cagle's map turtle habitats, and inhibiting the potential for species recovery in tributaries of the Guadalupe River. These options include off-channel storage, diversion of flood water, dams, and well fields.

West Texas A&M University is conducting a study on the habitat requirements and instream flow needs of the Cagle's map turtle on the Guadalupe River. This study is being funded by the Edwards Aquifer Authority (EAA 2001), and should be completed soon. "Preliminary results suggest that the areas in which the turtle is now found are more limited than the range identified in previous studies" (WTAMU 2001).

B. *Overutilization for commercial, recreational, scientific, or educational purposes.*
The Cagle's map turtle is of interest to collectors because it is a recently described Texas endemic. The species is vulnerable to over-collecting for the pet trade, zoos, museums, and scientific studies (Killebrew 1991, 1992). Pet-trade dealers reportedly are selling Cagle's map turtles to wholesalers and have offered $50 per hatchling and $400 per breeding pair to collectors (Killebrew, pers. comm., 1991). Kingsnake.com, International Reptile, and Turtles-Turtles-Turtles, internet advertisers, were selling Cagle's map turtles for $50 to $100 each in August 2000. International Reptile was still selling Cagle's map turtles for $100 each in February 2002. Between 1995 and 1998 over 140,000 live *Graptemys* were exported according to the CITES Law Enforcement Division. Comments from collectors, scientists, and local residents indicate that a substantial effort is underway to collect Cagle's map turtle before it is listed (Killebrew, pers. comm., 1991, 1994, 2000).

The species also is vulnerable to target shooting (Killebrew 1992). About 5 percent of Cagle's map turtles handled in the field have shell deformities indicative of shootings (Killebrew, pers. comm., 1992).

Turtles may be incapable of sustaining historic populations under even modest levels of harvest (Warwick et al. 1990). Late maturation and erratic reproductive success are important considerations. Considering the rarity of Cagle's map turtle and its significant loss of habitat, collection of live specimens (especially from small populations) could result in loss of a significant portion of the surviving individuals. For small populations, this loss may not be recoverable by natural reproduction.

C. *Disease or predation.*

Disease and parasites are not known to be significant threats to Cagle's map turtle. To date, specimens have contained minor infections of coccidial parasites (McAllister et al. 1991;

Killebrew, pers. comm., 1994). Whereas minor parasitic infections usually cause little harm to their host, severe infections can cause disease or stress that directly or indirectly affect host mortality. Predation likely is insignificant for hard-shelled adult turtles, but may be more significant for their soft-shelled eggs and young (Killebrew 1992). Although the magnitude of predation on Cagle's map turtle is unknown, many predatory birds, mammals, fish, and snakes eat turtle eggs and hatchlings (Harless and Morlock 1979). Additionally, small and medium-sized turtles may be more vulnerable to predation than larger turtles (Warwick et al. 1990). When compounded by other mortality factors, predation and parasites may further reduce Cagle's map turtle populations.

D. ***The inadequacy of existing regulatory mechanisms.***

The Texas Parks and Wildlife Department (TPWD) listed the Cagle's map turtle as threatened, effective November 16, 2000 (Dorinda Scott, TPWD, pers. comm. 2002) TPWD regulations prohibit the taking, possession, transportation, or sale of any of the animal species designated by state law as endangered or threatened without the issuance of a permit.

On January 26, 2000, the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES), an international treaty, which regulates international trade in certain animals and plants, proposed Cagle's map turtle for listing under Appendix III. Appendix III includes species that any party country identifies as being subject to regulation within its jurisdiction for purposes of preventing or restricting exploitation, and for which it needs the cooperation of other parties to control trade. The Washington Office of International Affairs, Branch of CITES, hopes to finalize the Appendix III listing soon (Bruce Weissgold, USFWS, pers. comm. 2002).

Currently exploitation is not regulated at the Federal level and is minimal at the State level. Commercial exportation requires only a declaration to the Service at Ports of Entry. Between 1995 and 1998 more than 140,000 live *Graptemys* were exported. Only two are noted as Cagle's map turtle. However, if inspectors are unable to identify the *Graptemys* species' from each other, they can easily be misidentified and lumped together as one species (Weissgold, U.S. Fish and Wildlife Service, in litt., 1999).

Previously the State only required a hunting license to hunt, collect or trade Cagle's map turtles. In 1999 the State implemented new regulations requiring anyone collecting animals from the wild or captive-breeding them for commercial purposes (sale or trade of dead or alive animals) to obtain a Non-game Collection Permit. In addition, anyone selling the animals is required to obtain a Dealer's Permit. Both permits require reporting; however, there is currently no method of tracking for accuracy of reported data. Enforcement in the field is hindered by the distribution of the species in water primarily surrounded by private lands. Access to these lands is often difficult.

Current Conservation Efforts: Exploitation is not regulated at the Federal level and there are no habitat protections. The U.S. Fish and Wildlife Service produced a paper documenting the populations and nesting behavior of the Cagle's map turtle. The outcome of this project was a paper entitled "Population Analysis and Nesting Study of Cagle's Map Turtle" (Killebrew and Babitzke 1996).

E. *Other natural or manmade factors affecting its continued existence.*

Cagle's map turtle has a naturally limited distribution and thus is more vulnerable to extinction than wider ranging species. Alterations to a single river system could change the location and suitability of nesting areas, thus affecting hatch rates and sex ratios (Wibbels et al. 1991).
Dams and large areas of unsuitable habitat may be partial or complete barriers to Cagle's map turtle movements, preventing repopulation of decimated areas. The prevalence of limestone beds and banks (unsuitable habitat) in headwaters of the Guadalupe and Blanco rivers may naturally limit populations as well. Operation of Canyon Lake for flood control could accentuate the problem by: 1) scouring the channel free of loose substrates needed for feeding and nesting immediately below the dam (i.e., exposing bedrock); 2) reducing the magnitude and frequency of historic flood flows needed for formation and maintenance of downstream habitat, and; 3) releasing cold water that affects nest temperatures and sex ratios (Wibbels et al. 1991) and inhibits turtle metabolism and growth.
Erosion of river banks and water pollution may have negative impacts to the Cagle's map turtle. For example, turtle nests may not survive flooding events that overtop low elevation sandbars and erode unstable banks (Killebrew, pers. comm., 1994). In the case of water pollution, the cities of New Braunfels and Seguin are major point-sources of treated municipal wastewater on the Guadalupe River, permitted for a combined discharge of 10.2 million gallons per day. Nonpoint sources of pollution within the Guadalupe River watershed (fertilizers, herbicides, insecticides) also could deplete the prey base. In addition, dumping and littering, especially on the upper Guadalupe River, result in heavy accumulations of non-biodegradable debris (Albright 1994, Killebrew 1991). The capability of the Guadalupe River system to assimilate this and other nutrient loading depends on adequate stream flow.

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PETITION TO LIST

black pine snake
(*Pituophis melanoleucus lodingi*)

AS A FEDERALLY ENDANGERED SPECIES

CANDIDATE HISTORY

CNOR 12/30/82:	
CNOR 9/18/85:	
CNOR 1/6/89:	
CNOR 11/21/91:	
CNOR 11/15/94:	
CNOR 10/25/99:	C
CNOR 10/30/01:	C
CNOR 6/13/02:	C

TAXONOMY

The black pine snake (*Pituophis melanoleucus lodingi*) is one of 15 recognized subspecies (10 of which occur in the United States) of *P. melanoleucus* (pine, bull, and gopher snakes) (Behler 1979; Smith and Brodie 1982; Sweet and Parker 1990). The black pine snake is geographically isolated from all other pine snakes. However, there is some indication that the black pine snake may have been in contact with other pine snakes in the past. A form intermediate between the black pine snake and the Florida pine snake (*P. m. mugitus*) occurs in Baldwin and Escambia counties in Alabama and Escambia County in Florida. These snakes are separated from populations of the "true" black pine snake by the Mobile River Delta and the Alabama River (Duran 1998a).

NATURAL HISTORY

Black pine snakes are endemic to the upland longleaf pine forests that once covered the southeastern United States. Habitat for these snakes consists of sandy, well-drained soils with an overstory of longleaf pine, a fire suppressed mid-story, and dense herbaceous ground cover (Duran 1998a). Duran (1998b) conducted a radio-telemetry study of the black pine snake that provided data on habitat use. Snakes in this study were usually located on well-drained, sandy-loam soils on hilltops, ridges, and toward the tops of slopes. They were rarely found in riparian areas, hardwood forests, or closed canopy conditions. More than half of the time, black pine snakes were located underground, usually in the trunks or root channels of rotting pine stumps.

POPULATION STATUS

There are historical records for the black pine snake from one parish in Louisiana, 14 counties in Mississippi, and 3 counties in Alabama west of the Mobile River Delta. Duran (1998a) recently completed a status survey for the species. He concluded that black pine snakes have been extirpated from Louisiana (Washington Parish) and from two counties (Lauderdale and Walthall) in Mississippi. They have not been reported west of the Pearl River in either Mississippi or Louisiana in 24 years (Duran 1998a). There are no recent (post-1979) records for three additional Mississippi counties (Greene, Jackson, and Lamar) where they once occurred. Surveys indicated that black pine snakes remain in 3 out of 3 counties in Alabama (Clarke, Mobile, and Washington) and 9 out of 14 counties in Mississippi (Forrest, George, Harrison, Jones, Marion, Pearl River, Perry, Stone, and Wayne). However, the distribution of populations within these counties has become highly restricted due to the fragmentation of the remaining longleaf pine habitat. In seven of the nine occupied Mississippi counties, populations of black pine snakes are concentrated in the DeSoto National Forest (68% of all known records). In the remaining occupied Mississippi counties, a single population is known from the Marion County Wildlife Management Area and a single one occurs on private land. Most of the remaining populations in Alabama are found on private, non-industrial timberland where they face an uncertain future. All black pine snake populations outside of the DeSoto National Forest appear to be small and isolated on islands of suitable longleaf pine habitat (Duran 1998a).

Duran (2000) reported the initial results of a habitat assessment of all known black pine snake records. Habitat suitability of the sites was based on how the habitat compared to that selected by black pine snakes in a recently completed telemetry study (Duran 1998a). A probability of occurrence rating was derived for each locality using a combination of the habitat suitability rating and data on how recently and/or frequently black pine snakes had been recorded at the site. Of the 157 known records, it was determined that black pine snakes probably no longer occurred at 53 sites (34% of total). Comparing individual records gives equal weight to the many occurrences that have been recently recorded in areas of pine snake abundance and the sparse records from areas where pine snakes have been extirpated. This greatly underestimates population losses. Removing the more recent records from 1990 to the present eliminates significant bias because during this period a concerted effort was made to locate black pine snakes, especially in areas of quality habitat. Subtracting these records would leave a total of 83 sites which could be considered "historical" records. Of these, black pine snakes probably no longer occur at 42 (51% of historical records).

The U.S. Fish and Wildlife Service classifies the black pine snake as a candidate for Endangered Species Act protection with a listing priority number of 6.

LISTING CRITERIA

A. ***The present or threatened destruction, modification, or curtailment of its habitat or range.***

Historical range:	Alabama (3 counties, see above), Louisiana (1 parish), and Mississippi (14 counties, see above).
Current range:	Alabama (3 counties, see above), Mississippi (9 counties, see above) .
Land ownership:	Of extant populations, 60 percent are on Federal (DeSoto National Forest), 39 percent are on private, and l percent are on State-managed (Marion County Wildlife Management Area) lands.

The historical distribution of the black pine snake is highly correlated with the historical range of the longleaf pine ecosystem in extreme southeastern Louisiana, southern Mississippi, and extreme southwestern Alabama (Duran 1998a). Today, the remaining longleaf pine forest in the southeast has been reduced to less than 5 percent of its original extent (Frost 1993, Outcalt and Sheffield 1996). In the range of the black pine snake, longleaf pine is now largely confined to isolated patches on private land and the DeSoto National Forest (DNF) in Mississippi. Black pine snake habitat has been eliminated through land use conversions, primarily urban development and conversion to agriculture and pine plantations. Most of the remaining patches of longleaf pine on private land are fragmented, degraded, second-growth forests. Conversion of longleaf pine forest to pine plantation often reduces the quality and suitability of a site for black pine snakes. Duran (1998b) found that black pine snakes prefer open canopies, reduced mid-stories, and dense herbaceous understories. He also found that these snakes are frequently underground in rotting pine stumps. Forest management strategies such as fire suppression (see Factor E), increased stocking densities, and removal of downed trees and stumps all contribute to degradation of habitat attributes preferred by black pine snakes.

Fragmentation and degradation of longleaf pine habitat is ongoing. The coastal counties of southern Mississippi and Mobile County, Alabama, are being developed at a rapid rate due to increases in the human population. Urbanization appears to have reduced historical black pine snake populations in Mobile County by approximately 50 percent (Duran 1998a). Much of this reduction has occurred in the last 15 to 20 years. For example, Jennings and Fritts (1983) reported that, in the 1980's, the black pine snake was one of the most frequently encountered snakes on the grounds of the Environmental Studies Center (Center) in Mobile County. Urban development has now engulfed lands adjacent to the Center and black pine snakes have not been seen on the property in the last 16 years (personal communication cited in U.S. Fish and Wildlife Service candidate assessment form). Black pine snakes were occasionally seen in the 1970's on the campus of the University of South Alabama in western Mobile (Duran 1998a). They have not been observed there in over a decade (personal communication cited in U.S. Fish and Wildlife Service candidate assessment form). There is no extensive public ownership of longleaf pine habitat in Mobile County and the black pine snake continues to survive only on parcels of unprotected private land.

B. ***Overutilization for commercial, recreational, scientific, or educational purposes.***

Direct take of black pine snakes for recreational, scientific, or educational purposes is not currently considered to be a threat. However, there is some indication that collecting for the pet trade may be a problem (Duran 1998a).

C. ***Disease or predation.***

Disease and predation are not presently considered to be threats to the black pine snake.

D. ***Inadequacy of existing regulatory mechanisms.***

In Mississippi, the black pine snake is classified as endangered by the Mississippi Department of Wildlife, Fisheries and Parks (Mississippi Natural Heritage Program 2002). In Alabama, it is protected as a non-game animal. Both Mississippi and Alabama regulations restrict collecting of the species. However, they do nothing to alleviate the loss of habitat which has caused the decline of this snake. The best remaining habitat for the black pine snake is on the DNF in Mississippi Forestry management programs, which protect gopher tortoises and red-cockaded woodpeckers or reestablish longleaf pine on the DNF, are of benefit to the snakes. Nevertheless, the DNF has no management program in place specific to the black pine snake. There are no restrictions on activities such as stump removal, which may have been detrimental to black pine snakes in the past (Duran 1998a). Multiple use priorities, such as timber production, and military and recreational use, do not put protection of the black pine snake at the forefront.

Current Conservation Efforts: There have been some preliminary conversations with the U.S. Forest Service concerning development of a Memorandum of Understanding for the black pine snake. Such an agreement would identify management needed to protect the snake on the DeSoto National Forest. However, it is not clear, even in the event that such a document is actually developed, that it would provide the urgently needed protection required to reverse the decline of the black pine snake.

E. *Other natural or manmade factors affecting its continued existence.*

Fire is needed to maintain the longleaf pine ecosystem. Fire suppression has been considered the primary reason for the degradation of the remaining longleaf pine forest. It is a contributing factorin reducing the quality and quantity of available habitat for the black pine snake. Lowered fire frequencies and reductions in average area burned per fire event (strategies often used in management of pine plantations) produce sites with thick mid-stories. These areas are avoided by black pine snakes (Duran 1998b).

Habitat fragmentation within the longleaf pine ecosystem threatens the continued existence of all the black pine snake populations on private lands. This is frequently the result of urban development, conversion of longleaf pine sites to pine plantations, and the associated increases in number of roads. When patches of available habitat become separated beyond the dispersal range of a species, populations are more sensitive to genetic, demographic, and environmental variability and extinction becomes possible. This is likely the cause for the extirpation of the black pine snake in Louisiana and the loss of populations in two (and possibly a total of five) counties in Mississippi (personal communication 1999 cited in U.S. Fish and Wildlife Service candidate assessment form).

Roads surrounding and traversing the remaining habitat pose a threat to the black pine snake. Lalo (1987) estimated that one million individual vertebrates are killed per day on roads in the United States. Black pine snakes frequent the sandy hilltops and ridges where roads are most frequently sited. During Duran’s (1998b) study, 17 percent of the black pine snakes with transmitters were killed while attempting to cross a road. In many parts of Louisiana, Mississippi, and Alabama, there is a lack of understanding of the importance of snakes to a healthy ecosystem. Snakes are often killed intentionally when they are observed. During his study, Duran (1998b) found a dead black pine snake that had been shot. In another instance, the tracks of a 4-wheel drive vehicle could be seen swerving to the wrong side of the road and into a ditch where a flattened dead black pine snake was later found. As development pressures increase on the remaining black pine snake’s habitat, especially in Mobile County, Alabama, human/snake interactions will increase and frequently result in the death of the snake.

Duran (1998b) suggested that reproductive rates of wild black pine snakes may be low. Thus, the loss of mature adults, through road mortality or direct killing, increases in significance. As existing occupied habitat becomes reduced in quantity and quality, low reproductive rates threaten population viability.

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PETITION TO LIST

eastern massasauga rattlesnake
(*Sistrurus catenatus catenatus*)

AS A FEDERALLY ENDANGERED SPECIES

CANDIDATE HISTORY

CNOR 12/30/82:	
CNOR 01/06/89:	
CNOR 11/21/91:	
CNOR 11/15/94:	
CNOR 10/25/99:	C
CNOR 10/30/01:	C
CNOR 06/13/02:	C

TAXONOMY

The taxonomic status of the eastern massasauga rattlenake (*Sistrurus catenatus catenatus*) as a valid subspecies is uncontroversial (e.g., Behler 1979; Smith and Brodie 1982; Conant and Collins 1991).

NATURAL HISTORY

The eastern massassauga lives in shallow wetlands and adjacent upland habitat. Suitable wetland habitat includes peatlands, marshes, sedge meadows, and swamp forest; typical upland habitat includes open savannas, prairies, and old fields. Seasonal use of these habitats varies across the range of the subspecies. Food includes small rodents, frogs, and other snakes (Behler 1979; Conant and Collins 1991).

POPULATION STATUS

Additional information, including the U.S. Fish and Wildlife Service 1998 Status Assessment is available on the Web at the USFWS Region 3 website: http://midwest.fws.gov/endangered/lists/candidat.html)

Complete demographic information is not available across the range of the subspecies; however, information regarding the historical and current number of populations, recruitment potential, distribution and proximity of subpopulations, and quantity and quality of habitat permits an assessment of the subspecies' long-term viability. Each state and Canadian province across the range of the eastern massasauga has lost more than 30 percent (and for the majority more than 50 percent) of their historical populations. Furthermore, fewer than 35 percent of the remaining populations are considered secure.

The U.S. Fish and Wildlife Service classifies the eastern massasauga as a candidate for Endangered Species Act protection with a listing priority number of 9.

LISTING CRITERIA

See 1998 Status Assessment for further information

A. ***The present or threatened destruction, modification, or curtailment of its habitat or range.***

Historical range:	Illinois, Indiana, Iowa, Michigan, Minnesota, Missouri, New York, Ohio, Pennsylvania, Wisconsin, and Ontario.
Current range:	Illinois, Indiana, Iowa, Michigan, Minnesota, Missouri, New York, Ohio, Pennsylvania, Wisconsin, and Ontario.
Land ownership:	Throughout the range of the subspecies, the eastern massasauga is found on both public and private land (~59% of the populations occur wholly or in part on public land). The majority of public land is State managed, although populations also occur on county and U.S. Army Corps of Engineers lands. Squaw Creek NWR, Swan Lake NWR, Trempealeau NWR, and possibly the LaCrosse District of the Upper Mississippi National Wildlife and Fish Refuge support massasauga populations. Necedah NWR is conducting a study of reintroduction techniques.

Although at a gross scale the current range of the eastern massassauga resembles the subspecies' historical range, the geographic distribution has in fact been dramatically restricted by the loss of the subspecies from much of the area within the boundaries of that range. Approximately 40 percent of the counties that were historically occupied by the eastern massassauga no longer support it. The eastern massassauga is currently considered imperiled in every state and province it occupies. Recent information indicates that the range of the eastern massassauga extends throughout all of Missouri and probably Iowa as well. This suggests that previously published accounts of the subspecies' range that identified an intergradation zone in Missouri and Iowa are not accurate.

Habitat loss is an important factor in the decline of the eastern massassauga. The effects of widespread wetland loss continue to impact populations. Development and agricultural practices continue to result in habitat loss. Habitat loss increases the distance between populations and can isolate seasonally used habitats within individual populations. Consequently, eastern massassauga populations become more susceptible to road mortality, predation, and persecution as snakes disperse from populations or make their seasonal movements between habitat types.

Destruction or modification of habitat is affecting at least 50 populations rangewide. A few examples are as follows. In Illinois, the Des Plaines River Valley population continues to be fragmented into smaller subpopulations isolated by development or otherwise unsuitable habitat.(Mierzwa 1993 cited in Szymanski 1998). In Michigan, a major residential development at the Green/Union Lakes site in Oakland County, Michigan, recently eliminated much of the existing habitat and severely degraded the remaining habitat (Legge 1996 cited in Szymanski 1998). At Wixom, Michigan, both wetland and upland habitat were recently degraded by agricultural practices and highway construction (Legge 1996 cited in Szymanski 1998). Similarly, in Bremer County, Iowa, a golf course is encroaching upon massasauga habitat (Christiansen 1993 cited in Szymanski 1998). In Wisconsin, cranberry operations are potential threats to massasauga populations (Cathy Carnes, U.S. Fish and Wildlife Service, *in litt.* 1997 cited in U.S. Fish and Wildlife Service candidate assessment form). In Pennsylvania, four companies within the last year have applied for sand and gravel mining permits in areas supporting massasauga populations (*in litt.* 1997 cited in U.S. Fish and Wildlife Service candidate assessment form). One of Ohio’s largest population (Killdeer Plains) was bulldozed and plowed under in 1994.

In addition, urban encroachment has disrupted the natural disturbance processes (such as hydrological cycles and fire frequency), resulting in changes in habitat structure and plant composition. For example, in Pennsylvania increasing woody vegetation was cited as a threat at 75 percent of the massasauga sites surveyed (Reinert and Bushar 1993 cited in Szymanski 1998).

B. *Overutilization for commercial, recreational, scientific, or educational purposes.*

The overharvesting of massasaugas is well documented, and the pernicious effects of past anti-rattlesnake campaigns are still apparent today. Several populations have been harvested beyond a recoverable threshold, and are thus functionally extinct. Intentional killing and illegal collection continue. Recent law enforcement actions involving individuals from several states revealed the immediacy and magnitude of this threat. An Indiana Department of Natural Resources law enforcement investigation in 1998 uncovered a well-organized, multi-state effort to launder State-protected reptile species (including eastern massasauga). The investigation concluded with the indictment of 40 defendants.

C. *Disease or predation.*

Predation under natural conditions is not a notable threat for the eastern massassauga. However, due to habitat loss as described under Factor A, eastern massassauga populations are extremely vulnerable to predators and as a result they experience abnormally high predation rates. Furthermore, the thermoregulatory needs of gravid female massasaugas make them most vulnerable to collection and predation, which exacerbates their impact. Eastern massasauga populations occurring at low densities are particularly sensitive to collection or predation because predation/collection of just a few individuals could greatly diminish the population's reproductive potential. Similarly, a Population Viability Analysis (PVA) indicated that eastern massasauga populations are most sensitive to adult mortality. Given the species' low biological replacement rate, even small increases in adult mortality can precipitate irreversible declines. These biological traits and the threat factors identified above interact synergistically, which exacerbates the effect of individual factors and can lead to an extinction vortex for those populations affected by one or more factors.

D. ***Inadequacy of existing regulatory mechanisms.***

The eastern massassauga is listed as endangered in Illinois, Indiana, Iowa, Minnesota, Missouri, New York, Ohio, Pennsylvania, and Wisconsin; as threatened in Ontario; and as special concern in Michigan. Although the species is afforded some level of state protection across the range of the subspecies, protection of its habitat is nearly nonexistent. Given the significance and pervasiveness of habitat loss, the decline of the eastern massassauga will continue unabated without additional protections.

Current Conservation Efforts: Management and monitoring guidelines for the eastern massassauga were developed under USFWS Region 3 guidance (The Eastern Massasauga: Handbook for Land Managers 2000). This handbook was broadly distributed and is being used by public land managers to develop conservation agreements for massasauga. As population data are limited at most sites, these conservation efforts are still in the initial stages of information gathering. In Wisconsin, for example, limited resources were dedicated to completing exhaustive surveys at one site. Continued survey efforts are planned at this site and others. The U.S. Fish and Wildlife Service continues to collect status information at several priority sites rangewide and efforts will focus on developing conservation agreements for these populations, but reversal of the large-scale decline of the eastern massasauga will likely require greater protection and recovery efforts.

E. ***Other natural or manmade factors affecting its continued existence.***

No other specific natural or anthropogenic factors threatening this snake have been identified.

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Also see literature cited within Szymanski 1998.

PETITION TO LIST

Columbia spotted frog-Great Basin
(Rana luteiventris)

AS A FEDERALLY ENDANGERED SPECIES

CANDIDATE HISTORY

CNOR 11/21/91:	
CNOR 11/15/94	
CNOR 2/28/96L:	C
CNOR 9/19/97:	C
CNOR 10/25/99:	C
CNOR 10/30/01:	C
CNOR 6/13/02:	C

TAXONOMY

Rana luteiventris is the accepted taxon for the Columbia spotted frog-Great Basin. Since nearly the time of its original description in 1853, the systematics of the "Western Spotted Frog" group has been a source of some confusion and debate. In 1996, however, a team led by David M. Green published the results of a study on the genetics of Spotted Frogs and concluded that the group actually contained two "sibling" species—the Oregon Spotted Frog and the Columbia Spotted Frog (Green et al. 1996, 1997). The decision to "split" the species was based upon the results of laboratory studies that indicated significant genetic differences, despite a lack of reliable morphological differences. Because the two species have allopatric ranges, they may be reliably identified based upon the location where a frog is encountered.

NATURAL HISTORY

Morphology

The Columbia Spotted Frog may be brown, tan, or gray with irregular-shaped black spots with light-centers. The undersides are cream colored with an orange or salmon-colored pigment usually present on the hind legs and lower abdomen. In some Nevada populations the hind legs and abdomen of frogs are yellow. The hind legs are relatively short relative to body length and there is extensive webbing between the toes on the hind feet. The eyes are upturned. Females may grow to approximately 100 mm (4 inches) snout-to-vent length, while males may reach approximately 75 mm (3 inches) snout-vent length (Nussbaum et al. 1983; Stebbins 1985; Leonard et al. 1993).

Behavior

The Columbia Spotted Frog is a highly aquatic species and nearly always is found in close proximity to water. Breeding habitats include a variety of relatively exposed, shallow-water (<60 cm), emergent wetlands such as sedge fens, riverine over-bank pools, beaver ponds, and the wetland fringes of ponds and small lakes. Vegetation in the breeding pools generally is dominated by herbaceous species such as grasses, sedges (Carex spp.) and rushes (Juncus spp.). After breeding is completed, adults often disperse into adjacent wetland, riverine and lacustrine habitats.

Adults exhibit a strong fidelity to breeding sites, with oviposition typically occurring in the same areas in successive years. Males arrive first, congregating around breeding sites, periodically vocalizing "advertisement calls" in a rapid series of 3 to 12 "tapping" notes that have little carrying power (Davidson 1995; Leonard et al. 1996). It is unknown to what extent the weak calls serve to attract females, but they may serve to distribute males at the breeding sites thus minimizing male–male encounters. As a female enters the breeding area, she is approached by and subsequently pairs with a male in a nuptial embrace referred to as amplexus. From several hours to possibly days later, the female releases her complement of eggs into the water while the male, still clinging to the female, releases sperm upon the ova.

Females may lay only one egg mass per year; yearly fluctuations in the sizes of egg masses are extreme (Utah Division of Wildlife Resources 1998). Successful egg production and the viability and metamorphosis of spotted frogs are susceptible to habitat variables such as temperature, depth, and pH of water, cover, and the presence/absence of predators (e.g., fishes and bullfrogs) (Morris and Tanner1969; Munger et al. 1996; Reaser 1996b).

After a few weeks, thousands of small tadpoles emerge and cling to the remains of the gelatinous egg masses. After several days the small hatchling tadpoles begin swimming and feeding upon algae, detritus, and in some cases, bacteria, using their minute brush-like mouthparts. In the Columbia Basin tadpoles may grow to 100 mm (4 inches) total length prior to metamorphosing into froglets in their first summer or fall. At high-elevation montane sites, however, tadpoles barely reach 45 mm in total length prior to the onset of metamorphosis in late fall.

Mortality of eggs, tadpoles, and newly metamorphosed frogs is high, with approximately 5% surviving the first winter (David Pilliod, personal communication). At low-elevation sites sexual maturity is probably attained in two to three years, while three or four years may be required at high-elevation sites (Turner 1960; Licht 1975).

Food includes arthropods (e.g., spiders, insects), earthworms and other invertebrate prey (Whitaker et al. 1982). In turn, Columbia Spotted Frogs may be preyed upon by mink, river otter, raccoon, herons, bitterns, corvids, and garter snakes, while larvae may be consumed by larvae of dragonflies, predacious diving beetles, fish, garter snakes, and wading birds.

Habitat

Columbia spotted frogs are found closely associated with clear, slow-moving or ponded surface waters, with little shade (Reaser 1997). Reproducing populations have been found in habitats characterized by springs, floating vegetation, and larger bodies of pooled water (e.g., oxbows, lakes, stock ponds, beaver-created ponds, seeps in wet meadows, backwaters) (Idaho Department of Fish and Game (IDFG) et al. 1995; Reaser 1997). A deep silt or muck substrate may be required for hibernation and torpor (Morris and Tanner 1969). In colder portions of their range, Columbia spotted frogs will use areas where water does not freeze, such as spring heads and undercut streambanks with overhanging vegetation (IDFG et al. 1995).

Distribution

Nevada

Columbia spotted frogs in Nevada are found in the central (Nye County) and northeastern (Elko and Eureka Counties) parts of the state, usually at elevations between 1,700 and 2,650 meters (5,600 and 8,700 feet), although they have been recorded historically in a broader range including

Lander County in central Nevada and Humboldt County in northwest Nevada (Reaser 2000). The Great Basin population of Columbia spotted frogs in Nevada is geographically separated into

three distinct subpopulations; the Jarbidge-Independence Range, Ruby Mountains, and Toiyabe Mountains subpopulations.

The largest of Nevada's three subpopulation areas is the Jarbidge-Independence Range in Elko and Eureka counties. This subpopulation area is formed by the headwaters of streams in two major hydrographic basins. The South Fork Owyhee, Owyhee, Bruneau, and Salmon Falls drainages flow north into the Snake River basin. Marys River, North Fork of the Humboldt, and Maggie Creek drain into the interior Humboldt River basin. The Jarbidge-Independence Range subpopulation is considered to be genetically and geographically most closely associated with Columbia spotted frogs in southern Idaho (Reaser 1997).

Columbia spotted frogs occur in the Ruby Mountains in the areas of Green Mountain, Smith, and Rattlesnake creeks on lands in Elko County managed by the U.S. Forest Service (Forest Service).

Although geographically, Ruby Mountains spotted frogs are close to the Jarbidge-Independence Range subpopulation, preliminary allozyme evidence suggests they are genotypically different (J.

Reaser, pers. comm., 1998). The Ruby Mountains subpopulation is considered discrete because of this difference (J. Reaser, pers. comm., 1998) and because it is geographically isolated from the Jarbidge-Independence Range subpopulation area to the north by an undetermined barrier (e.g., lack of suitable habitat, connectivity, and/or predators), and from the Toiyabe Mountains subpopulation area to the southwest by a large gap in suitable Humboldt River drainage habitat.

In the Toiyabe Range, spotted frogs are found in seven drainages in Nye County, Nevada--the Reese River (Upper and Lower), Cow and Ledbetter Canyons, and Cloverdale, Stewart, Illinois, and Indian Valley Creeks. Although historically they also occurred in Lander County, preliminary surveys have found them absent from this area (J. Tull, Forest Service, pers. comm., 1998). Toiyabe Range spotted frogs are geographically isolated from the Ruby Mountains and Jarbidge-Independence Range subpopulations by a large gap in suitable habitat and they represent R. luteiventris in the southern-most extremity of its range. Genetic analyses of Great Basin Columbia spotted frogs from the Toiyabe Range suggest that these frogs are distinctive in comparison to frogs from the Ruby Mountains and Jarbidge-Independence Range subpopulation areas (Green et al. 1996, 1997; J. Reaser, pers. comm., 1998). Genetic (mtDNA) differences between the Toiyabe Range frogs and the Ruby Mountains frogs are less than those between the Toiyabe Range frogs and the Jarbidge-Independence Range frogs, but this may be because of similar temporal and spatial isolation (J. Reaser, pers. comm., 1998).

Distribution- Idaho and Oregon

Historically, the range of the Columbia spotted frog in Idaho included the Raft River and Goose Creek drainages in Minidoka County and the Owyhee Mountains in Owyhee County in southern Idaho. In eastern Oregon, the historic range of spotted frogs included the Blue and Wallowa Mountains in Wallowa County and the Owyhee Mountains in Malheur County. Surveys conducted in the Raft River and Goose Creek drainages in Idaho failed to relocate spotted frogs (Reaser 1997; Shipman and Anderson 1997; Turner 1962). In 1994 and 1995, the Bureau of Land Management (BLM) conducted surveys in the Jarbidge and Snake River Resource Areas in Twin Falls County, Idaho. These efforts were also unsuccessful in locating spotted frogs (McDonald 1996). Only six historical sites were known in the Owyhee Mountain range in Idaho, and only 11 sites were known in southeastern Oregon in Malheur County prior to 1995 (Munger et al. 1996).

Currently, Columbia spotted frogs appear to be widely distributed throughout southwestern Idaho (mainly in Owyhee County) and eastern Oregon, but local populations within this general area appear to be isolated from each other by either natural or human induced habitat disruptions. The largest local population of spotted frogs in Idaho occurs in Owyhee County in the Rock Creek drainage. The largest local population of spotted frogs in Oregon occurs in Malheur County in the Dry Creek drainage.

POPULATION STATUS

Spotted frog habitat degradation and fragmentation is probably a combined result of past and current influences of heavy livestock grazing, spring development, agricultural development, urbanization, and mining activities. These activities eliminate vegetation necessary to protect frogs from predators and UV-B radiation; reduce soil moisture; create undesirable changes in water temperature, chemistry and water availability; and can cause restructuring of habitat zones through trampling, rechanneling, or degradation which in turn can negatively affect the available invertebrate food source (IDFG et al. 1995; Munger et al. 1997; Reaser 1997; Engle and Munger

2000; Engle 2002). Spotted frog habitat occurs in the same areas where these activities are likely to take place or where these activities occurred in the past and resulting habitat degradation has not improved over time. Natural fluctuations in environmental conditions tend to magnify the detrimental effects of these activities, just as the activities may also magnify the detrimental effects of natural environmental events.

The U.S. Fish and Wildlife Service classifies the Columbia spotted frog as a candidate for Endangered Species Act protection with a listing priority number of 3. The Nevada Division of Wildlife classifies the spotted frog as a protected species, but they are not afforded official protection and populations are not monitored. The species is included on the Forest Service sensitive species list; as such, its management must be considered during forest planning processes. However, little habitat restoration, monitoring or surveying has occurred on Forest Service lands. The frog is on the sensitive species list for the State of Idaho, but is not given any special protection by the State.

Population status Nevada. Declines of Columbia spotted frog populations in Nevada have been recorded since 1962 when it was observed that in many Elko County localities where spotted frogs were once numerous, the species was nearly extirpated (Turner 1962). Extensive loss of habitat was found to have occurred from conversion of wetland habitats to irrigated pasture and spring and stream dewatering by mining and irrigation practices. In addition, there was evidence of extensive impacts on riparian habitats due to intensive livestock grazing. Recent work by researchers in Nevada have documented the loss of historically known sites, reduced numbers of individuals within local populations, and declines in the reproduction of those individuals (Hovingh 1990; Reaser 1996a, 1996b, 1997). Surveys in Nevada between 1994 and 1996 indicated that 54 percent of surveyed sites known to have frogs before 1993 no longer supported individuals (Reaser 1997).

Little historical or recent data are available for the largest subpopulation area in Nevada, the Jarbidge-Independence Range. Presence/absence surveys have been conducted by Stanford University researchers and the Forest Service, but dependable information on numbers of breeding adults and trends is unavailable. Between 1993 and 1998, 976 sites were surveyed for the presence of spotted frogs in northeastern Nevada, including the Ruby Mountains subpopulation area (Shipman and Anderson 1997; Reaser 2000). Of these, 746 sites (76 percent) that were believed to have characteristics suitable for frogs were unoccupied. For these particular sites there is no information on historical presence of spotted frogs. Of 212 sites that were known to support frogs before 1992, 107 (50 percent) no longer had frogs, while 105 sites did support frogs. At the occupied sites, surveyors observed more than 10 adults at only 13 sites (12 percent). Frogs in this area appear widely distributed (Reaser 1997). No monitoring or surveying has taken place in northeastern Nevada since 1998.

Between 1993 and 1998, 339 sites were surveyed for the presence of Columbia spotted frogs in the Toiyabe Range. Surveyors visited 118 sites (35 percent) with suitable habitat characteristics where no frogs were present. Ten historical frog sites no longer had frogs when surveyed by

Reaser between 1993 and 1996 (Reaser 1997). However, at 211 other historical sites, frogs were still present during this survey period. Of these 211 sites, surveyors reported greater than 10 adult frogs at 133 sites (63 percent) (Reaser 1997). In 2000, frog mark-recapture surveys of the Toiyabe Range subpopulation were conducted by the University of Nevada, Reno. Preliminary estimates of frog numbers in the Indian Valley Creek drainage were around 5,000 breeding individuals, which is greater than previously believed (K. Hatch, pers. comm., 2001). However, during the 2000-2001 winter, Hatch (2002) noted a large population decrease, ranging between 66 and 86.5 percent at several sites. Lack of standardized or extensive monitoring and routine surveying has prevented dependable determinations of frog population numbers or trends in Nevada.

Population status Oregon and Idaho. Extensive surveys since 1996 throughout southern Idaho and eastern Oregon, have led to increases in the number of known spotted frog sites. Although efforts to survey for spotted frogs have increased the available information regarding known species locations, most of these data suggest the sites support small numbers of frogs. Of the 49 known local populations in southern Idaho, 61 percent had 10 or fewer adult frogs and 37 percent had 100 or fewer adult frogs (Engle 2000; Idaho Conservation Data Center (IDCDC) 2000). The largest known local population of spotted frogs occurs in the Rock Creek drainage of Owyhee County and supports under 250 adult frogs (Engle 2000). Extensive monitoring at 10 of the 46 occupied sites since 1997 indicates a general decline in the number of adult spotted frogs encountered (Engle 2000; Engle and Munger 2000; Engle 2002). All known local populations in southern Idaho appear to be functionally isolated (Engle 2000; Engle and Munger 2000).

Of the16 sites that are known to support Columbia spotted frogs in eastern Oregon, 81 percent of these sites appear to support fewer than 10 adult spotted frogs. In southeastern Oregon, surveys conducted in 1997 found a single population of spotted frogs in the Dry Creek drainage of Malheur County. Population estimates for this site are under 300 adult frogs (Munger et al. 1996). Monitoring (since 1998) of spotted frogs in northeastern Oregon in Wallowa County indicates relatively stable, small local populations (less than five adults encountered) (Pearl 2000). All of the known local populations of spotted frogs in eastern Oregon appear to be functionally isolated.

LISTING CRITERIA

A. ***The present or threatened destruction, modification, or curtailment of its habitat or range.***

Historical range:	Nevada; Columbia spotted frogs in Nevada are found in the central (Nye County) and northeastern (Elko and Eureka Counties) parts of the state, usually at elevations between 1,700 and 2,650 meters (5,600 and 8,700 feet), although they have been recorded historically in a broader range including Lander County in central Nevada and Humboldt County in northwest Nevada (Reaser 2000). Idaho, and Oregon; Historically, the range of the Columbia spotted frog in Idaho included the Raft River and

Goose Creek drainages in Minidoka County and the Owyhee Mountains in Owyhee County in southern Idaho. In eastern Oregon, the historic range of spotted frogs included the Blue and Wallowa Mountains in Wallowa County and the Owyhee Mountains in Malheur County.

Current Range: Nevada; Columbia spotted frogs in Nevada are found in the central (Nye County) and northeastern (Elko and Eureka Counties) parts of the state. Idaho, and Oregon; Columbia spotted frogs appear to be widely distributed throughout southwestern Idaho (mainly in Owyhee County) and eastern Oregon, but local populations within this general area appear to be isolated from each other by either natural or human induced habitat disruptions. The largest local population of spotted frogs in Idaho occurs in Owyhee County in the Rock Creek drainage. The largest local population of spotted frogs in Oregon occurs in Malheur County in the Dry Creek drainage.

Land Ownership: An estimated 90 percent of all known habitat for spotted frog occurs on lands managed by the Forest Service and the BLM. The remainder of known or suspected sites occur on private, tribal, State lands.

Spotted frog habitat degradation and fragmentation is probably a combined result of past and current influences of heavy livestock grazing, spring development, agricultural development, urbanization, and mining activities. These activities eliminate vegetation necessary to protect frogs from predators and UV-B radiation; reduce soil moisture; create undesirable changes in water temperature, chemistry and water availability; and can cause restructuring of habitat zones through trampling, rechanneling, or degradation which in turn can negatively affect the available invertebrate food source (IDFG et al. 1995; Munger et al. 1997; Reaser 1997; Engle and Munger 2000; Engle 2002).

Springs provide a stable, permanent source of water for frog breeding, feeding, and winter refugia (IDFG et al. 1995). Springs provide deep, protected areas which serve as hibernacula for spotted frogs in cold climates. Springs also provide protection from predation through underground openings (IDFG et al. 1995; Patla and Peterson 1996). Most spring developments result in the installation of a pipe or box to fully capture the water source and direct water to another location such as a livestock watering trough. Loss of this permanent source of water in desert ecosystems can also lead to the loss of associated riparian habitats and wetlands used by spotted frogs. Developed spring pools could be functioning as attractive nuisances for frogs, concentrating them into isolated groups, increasing the risk of disease and predation (Engle 2001). Many of the springs in southern Idaho, eastern Oregon, and Nevada have been developed.

The reduction of beaver populations has been noted as an important feature in the reduction of suitable habitat for spotted frogs. Beaver are important in the creation of small pools with slowmoving water that function as habitat for frog reproduction and create wet meadows that

provideforaging habitat and protective vegetation cover, especially in the dry interior western United States (St. John 1994). Beaver trapping is still common in Idaho and harvest is unregulated in most areas (IDFG et al. 1995). In some areas, beavers are removed because of a perceived threat to water for agriculture or horticultural plantings. As indicated above, permanent ponded waters are important in maintaining spotted frog habitats during severe drought or winter periods.Removal of a beaver dam in Stoneman Creek in Idaho is believed to be directly related to the decline of a spotted frog subpopulation there. Intensive surveying of the historical site where
frogs were known to have occurred has documented only one adult spotted frog (Engle 2000). Fragmentation of habitat may be one of the most significant barriers to spotted frog recovery and population persistence.

Studies in Idaho indicate that spotted frogs exhibit breeding site fidelity (Patla and Peterson 1996; Engle 2000; Munger and Engle 2000; J. Engle, IDFG, pers. comm., 2001). Movement of frogs from hibernation ponds to breeding ponds may be impeded by zones of unsuitable habitat. As movement corridors become more fragmented due to loss of flows within riparian or meadow habitats, local populations will become more isolated (Engle 2000; Engle 2001). Vegetation and surface water along movement corridors provide relief from high temperatures and arid environmental conditions, as well as protection from predators. Loss of vegetation and/or lowering of the water table as a result of the above mentioned activities can pose a significant threat to frogs moving from one area to another. Likewise, fragmentation and loss of habitat can prevent frogs from colonizing suitable sites elsewhere.

Though direct correlation between spotted frog declines and livestock grazing has not been studied, the effects of heavy grazing on riparian areas are well documented (Kauffman et al. 1982; Kauffman and Kreuger 1984; Skovlin 1984; Kauffman et al. 1985; Schulz and Leininger 1990). Heavy grazing in riparian areas on state and private lands is a chronic problem throughout the Great Basin. Efforts to protect spotted frog habitat on state lands in Idaho have been largely unsuccessful because of lack of cooperation from the State. In northeast Nevada, the Forest Service has completed three riparian area protection projects in areas where spotted frogs occur. These projects include altering stocking rates or changing the grazing season in two allotments known to have frogs and constructing riparian fencing on one allotment. However, these three sites have not been monitored to determine whether efforts to protect riparian habitat and spotted frogs have been successful. In the Toiyabe Range, a proposal to fence 3.2 kilometers (km) (2 miles (mi)) of damaged riparian area along Cloverdale Creek to protect it from grazing is scheduled to occur in the summer of 2002. In addition to the riparian exclosure, BLM biologists located a diversion dam in 1998 on Cloverdale Creek which was completely de-watering approximately 1.6 km (1 mi) of stream. During the summer of 2000, this area was reclaimed and water was put back into the stream. This area of the stream is not currently occupied by spotted frogs but it is historical habitat.

The effects of mining on Great Basin Columbia spotted frogs, specifically, have not been studied, but the adverse effects of mining activities on water quality and quantity, other wildlife species, and amphibians in particular have been addressed in professional scientific forums (Chang et al. 1974; Birge et al. 1975; Greenhouse 1976; Khangarot et al. 1985).

B. ***Overutilization for commercial, recreational, scientific, or educational purposes.***
This is not known to be a threat to Great Basin Columbia spotted frogs at this time.

C. ***Disease or predation.***
Predation by fishes is likely an important threat to spotted frogs. The introduction of nonnative salmonid and bass species for recreational fishing may have negatively affected frog species throughout the United States. The negative effects of predation of this kind are difficult to document, particularly in stream systems. However, significant negative effects of predation on frog populations in lacustrine systems have been documented (Hayes and Jennings 1986; Pilliod et al. 1996, Knapp and Matthews 2000). One historic site in southern Idaho no longer supports spotted frog although suitable habitat is available. This may be related to the presence of introduced bass in the Owyhee River (IDCDC 2000). The stocking of nonnative fishes is common throughout waters of the Great Basin. The Nevada Division of Wildlife (NDOW) has committed to conducting stomach sampling of stocked nonnative and native species to determine the effects of predation on spotted frogs. However, this commitment will not be fulfilled until the spotted frog conservation agreements are signed. To date, NDOW has not altered fish stocking rates or locations in order to benefit spotted frogs.

The bull frog (Rana catesbeiana), a nonnative ranid species, occurs within the range of the spotted frog in the Great Basin. Bullfrogs are known to prey on other frogs (Hayes and Jennings 1986). They are rarely found to co-occur with spotted frogs, but whether this is an artifact of competitive exclusion is unknown at this time.

Although a diversity of microbial species is naturally associated with amphibians, it is generally accepted that they are rarely pathogenic to amphibians except under stressful environmental conditions. Chytridiomycosis (chytrid) is an emerging panzootic fungal disease in the United States (Fellers et al. 2001). Clinical signs of amphibian chytrid include abnormal posture, lethargy, and loss of righting reflex. Gross lesions, which are usually not apparent, consist of abnormal epidermal sloughing and ulceration; hemorrhages in the skin, muscle, or eye; hyperemia of digital and ventrum skin, and congestion of viscera. Diagnosis is by identification of characteristic intracellular flask-shaped sporangia and septate thalli within the epidermis.

Chytrid can be identified in some species of frogs by examining the oral discs of tadpoles which may be abnormally formed or lacking pigment (Fellers et al. 2001). Chytrid was confirmed in the Circle Pond site, Idaho, where long term monitoring since 1998 has indicated a general decline in the population (Engle 2002). It is unclear whether the presence of this disease will eventually result in the loss of this subpopulation. Two additional sites may have chytrid, but this has yet to be determined (J. Engle, pers. comm., 2001). Protocols to prevent further spread of the disease by researchers were instituted in 2001. Chytrid has also been found in the Wasatch Columbia spotted frog distinct population segment (K. Wilson, pers comm., 2002). Chytrid has not been found in Nevada populations of spotted frogs.

D. ***The inadequacy of existing regulatory mechanisms.***
Spotted frog occurrence sites and potential habitats occur on both public and private lands. This species is included on the Forest Service sensitive species list; as such, its management must be considered during forest planning processes. However, little habitat restoration, monitoring or surveying has occurred on Forest Service lands.

In the fall of 2000, 250 head of cattle were allowed to graze for 45 days on one pasture in the Indian Valley Creek drainage of the Humboldt-Toiyabe National Forest in central Nevada for the first time in 6 years (M. Croxen, pers. comm., 2002). Grazing was not allowed in this allotment in 2001. Recent mark-recapture data indicated that this drainage supports more frogs than previously presumed, potentially around 5,000 individuals (K. Hatch, pers. comm., 2000). Perceived improvements in the status of frog populations in the Indian Valley Creek area may be a result of past removal of livestock grazing. The reintroduction of grazing disturbance into this relatively dense area of frogs may negatively impact the population.

BLM policies direct management to consider candidate species on public lands under their jurisdiction. To date, BLM efforts to conserve spotted frogs and their habitat in Idaho, Oregon, and Nevada have not been adequate to address threats.
The southernmost known population of spotted frogs can be found on the BLM's San Antone Allotment south of Indian Valley Creek in the Toiyabe Range. Grazing is allowed in this area from November until June (L. Brown, pers. comm., 2002). The season of use is a very sensitive portion of the spotted frog annual life cycle which includes migration from winter hibernacula to breeding ponds, breeding, egg laying and hatching, and metamorphosing of young. Additionally, the riparian Standards and Guidelines were not met in 1996, the last time the allotment was evaluated.

The status of local populations of spotted frogs on Yomba-Shoshone or Duck Valley Tribal lands is unknown. Tribal governments do not have regulatory or protective mechanisms in place to protect spotted frogs.

The Nevada Division of Wildlife classifies the spotted frog as a protected species, but they are not afforded official protection and populations are not monitored. Though the spotted frog is on the sensitive species list for the State of Idaho, this species is not given any special protection by the State. Columbia spotted frogs are not on the sensitive species list for the State of Oregon. Protection of wetland habitat from loss of water to irrigation or spring development is difficult because most water in the Great Basin has been allocated to water rights applicants based on historical use and spring development has already occurred within much of the known habitat of spotted frogs. Federal lands may have water rights that are approved for wildlife use, but these rights are often superceded by historic rights upstream or downstream that do not provide for minimum flows. Also, most public lands are managed for multiple use and are subject to livestock grazing, silvicultural activities, and recreation uses that may be incompatible with spotted frog conservation without adequate mitigation measures.

Current Conservation Efforts: Efforts to create conservation agreements among Federal, State, and tribal entities for the three spotted frog subpopulations in Nevada began in 1997. Though conservation agreements have been drafted for both the northeastern and central Nevada subpopulations of the Great Basin Columbia spotted frog, neither of these agreements has been signed. Recent setbacks in finalizing these agreements are a result of changes in team members responsible for creating the documents and reaching consensus. Despite the fact that neither of the documents have been signed, some of the parties have been fulfilling some of the commitments outlined in the agreements since 1998 and may continue implementation regardless of signing.

The Snake River Basin Office in Boise, Idaho has been working with the BLM, Boise State University, the State of Idaho, and private landowners to complete surveys for spotted frogs. Extensive monitoring funded by the BLM, and completed by Boise State University has raised concern for populations of frogs in southwestern Idaho where frogs appear to be declining. Attempts to conserve isolated local populations on State of Idaho and BLM lands in Idaho have been unsuccessful to date. Conservation efforts in eastern Oregon include continued inventory and monitoring programs and implementation of riparian protection measures at select pond sites in the Wallowa-Whitman National Forest. The Vale District BLM has implemented long-term monitoring at Dry Creek and Castro Springs. The species is included on the Forest Service sensitive species list; as such, its management must be considered during forest planning processes. However, little habitat restoration, monitoring or surveying has occurred on Forest Service lands.

E. *Other natural or manmade factors affecting its continued existence.*
Multiple consecutive years of less than average precipitation may result in a reduction in the number of suitable sites available to spotted frogs. Local extirpations eliminate source populations from habitats that in normal years are available as frog habitat (Lande and Barrowclough 1987; Schaffer 1987; Gotelli 1995). These climate events are likely to exacerbate the effects of other threats, thus increasing the possibility of stochastic extinction of subpopulations by reducing their size and connectedness to other subpopulations (see Factor A for additional information). As movement corridors become more fragmented, due to loss of flows within riparian or meadow habitats, local populations will become more isolated (Engle 2000). Increased fragmentation of the habitat can lead to greater loss of populations due to demographic and/or environmental stochasticity.

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PETITION TO LIST

Oregon spotted frog
(Rana pretiosa)

AS A FEDERALLY ENDANGERED SPECIES

CANDIDATE HISTORY

CNOR 11/21/91:	
CNOR 11/15/94:	
CNOR 2/28/96:	C
CNOR 9/19/97:	C
CNOR 10/25/99:	C
CNOR 10/30/01:	C
CNOR 6/13/02:	C

TAXONOMY

Oregon spotted frogs have recently been classified as a species separate from the Columbia spotted frog, now Rana luteiventris. Researchers at McGill University in Canada split the species into Rana luteiventris and Rana pretiosa in 1996 (Green el al., 1996, Leonard). The researchers found that while the two species are nearly identical morphologically, they differ genetically and occupy different ranges.

NATURAL HISTORY

Morphology
The Oregon spotted frog is a medium-sized frog with light-centered black spots on the head and back. Adult frogs are green, brown or reddish brown as adults, while juveniles are brown or olive green. Two dorsolateral folds, which are usually lighter in color than the frog's body, appear as stripes part way along the back. The eyes are set so that when you look straight down on an Oregon spotted frog, it gives the appearance of looking straight back up at you. Adults can grow to a length of 2 to 4 in. from the snout to the rump.

When viewed from a distance, Oregon spotted frogs have a distinct posture on land - they crouch to the ground, rather than sitting up straight as red-legged frogs do. Oregon spotted frogs spend much of their time in the water, and when disturbed will dive to the bottom and stay there for quite some time.

Behavior

Male Oregon spotted frogs are not territorial and may gather in groups of 10 to more than 25 individuals at specific locations (Leonard et al. 1993; M. Hayes, pers. comm., 2002). Breeding occurs in February or March at lower elevations and as late as May or early June at higher elevations (Leonard et al. 1993; M. Hayes, pers. comm., 2002). Egg-laying occurs at the same general location at a site in successive years (M. Hayes, pers. comm., 2002). Egg masses are generally laid communally in groups of a few to several hundred in shallow, often temporary, pools of water that are easily warmed by the sun, which hastens egg development (Licht 1971; Nussbuam et al. 1993; Cook 1984; Hayes et al. 1997; McAllister and Leonard 1997; Engler and Friesz 1998). Tadpoles metamorphose into froglets during their first summer (Leonard et al. 1993). Adults begin to breed by between one and three years of age, depending on elevation and latitude. Males may breed at one year at lower elevations and latitudes, but generally require a second year to reach maturity at other sites. Females breed by two or three years of age, depending on elevation and latitude. Longevity of the species is not known; however, skeletochronology studies indicate the species is not long-lived. Individuals four years of age or older are rare, and most only reach two or three years of age (McAllister and Leonard 1997; M. Hayes, pers. comm., 2002).

Adult Oregon spotted frogs eat mostly invertebrates such as beetles, flies, spiders, and water striders. They are "sit and wait" predators, remaining motionless in the water or on the shore, until prey approaches. The frogs then lunge toward the prey and capture it with a sticky tongue. They have also been reported to eat juvenile frogs of other species.

Oregon spotted frog tadpoles are grazers, eating algae, decaying plant matter, and detritus.

Habitat

Warm water microhabitat in different types of marsh and marsh-like habitat appear to be preferred by Oregon spotted frogs. These habitats have been found at elevations from sea level to 1,676 meters (m) (5,500 feet (ft)) in a north-south gradient (Dunlap 1955; Hayes 1997; McAllister and Leonard 1997). The highest elevation known site, however, is not at the most southern end of the species' range; Oregon spotted frogs could occur at higher elevations farther south in the range (M. Hayes, pers. comm., 2002).

Fairly large marshes (approximate minimum size of 4 hectares (ha) (9 acres (ac)) that reach suitably warm temperatures in active-season (summer) microhabitats can likely support populations large enough to persist despite high predation rates (Hayes 1994). Oregon spotted frogs have different microhabitat preferences or requirements in the breeding season, the active season, and for overwintering. In the active season, this species inhabits emergent wetland habitats in forested landscapes, although it is not typically found under forest canopy. It is almost always found in or near a perennial body of water, such as a spring, pond, lake, or sluggish stream. These habitats usually include zones of shallow water and abundant emergent

or floating aquatic plants, which are used for basking and escape cover from predators (Leonard et al. 1993; Corkran and Thoms 1996; McAllister and Leonard 1997; Joe Engler, Service, pers. comm., 1999). Breeding microhabitat consists of shallow, marginal shelves associated with the active-season habitat that typically do not retain water year-round. Overwintering habitat is aquatic and appears to be selected on the basis of sufficient dissolved oxygen and sheltering from
freezing (M. Hayes, pers. comm., 2002).

Distribution
Historically, the Oregon spotted frog ranged from British Columbia, Canada, to the Pit River drainage in northeastern California (Hayes 1997; McAllister and Leonard 1997). Currently, the Oregon spotted frog is found from extreme southwestern British Columbia south through the eastern side of the Puget/Willamette Valley Trough, and in the Cascades Range from south-central Washington at least to the Klamath Basin in Oregon. Only 15 of 59 historic localities, where the species' previous existence can be verified (e.g., museum specimens, photographs, reliable published records), are occupied (Hayes 1997; McAllister and Leonard 1997). Currently, 35 Oregon spotted frog locations are known in Washington (1 historic, 5 new) and Oregon (12 historic, 17 new). Oregon spotted frogs have not been documented in recent surveys in California. In British Columbia, Oregon spotted frogs have been rediscovered at the historic site at South Langley, and found at three new sites in 1996 and 1997 (Hayes 1997; Hayes et al. 1997; McAllister and Leonard 1997; Mark Hayes, Portland State University, pers. comm., 1999; Kelly McAllister, Washington Department of Fish and Wildlife, pers. comm., 1999).

POPULATION STATUS

Threats to the species' habitat include development (loss of habitat), siltation and vegetation removal caused by livestock grazing, introduction of exotic plant (reed canary grass) and animal species (bass and bullfrogs), plant successional changes, changes in hydrology due to construction of dams and alterations to seasonal flooding, poor water quality, and water contamination (run-off from farm fields (pesticides, fertilizers)), acid rain.

Oregon spotted frogs are far more aquatic than other native frogs - they leave the water for very short periods when foraging, and never move between ponds except by connecting waterways. This makes the frogs especially vulnerable to fragmentation of their habitat.

The U.S. Fish and Wildlife Service classifies Oregon spotted frogs as a candidate for Endangered Species Act protection with a listing priority number of 2. Oregon spotted frogs were declared an Endangered species in Canada by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) in 1999. The species is also Red-listed in B.C. The Oregon spotted frog is the only organism that has received an "emergency listing" as an endangered species in Canada. The Oregon spotted frog was listed as a state endangered species in Washington in August 1997 (Watson et al. 1998; WAC 232-12-014).

LISTING CRITERIA

A. ***The present or threatened destruction, modification, or curtailment of its habitat or range.***

Historical range:	Washington, Oregon, California, British Columbia (Canada). Historically, the Oregon spotted frog ranged from British Columbia, Canada, to the Pit River drainage in northeastern California (Hayes 1997; McAllister and Leonard 1997).
Current Range:	Washington, Oregon, British Columbia (Canada). Currently, the Oregon spotted frog is found from extreme southwestern British Columbia south through the eastern side of the Puget/Willamette Valley Trough, and in the Cascades Range from south-central Washington at least to the Klamath Basin in Oregon.
Land Ownership:	In Washington, two Thurston County Oregon spotted frog populations occur on private land, and one population occurs on National Wildlife Refuge land (Black River Unit of the Nisqually National Wildlife Refuge). The two Trout Lake sites are on both private and public land, including the WDNR's Trout Lake Natural Area Preserve and Gifford Pinchot National Forest. The Conboy Lake population occurs predominately within the Conboy Lake National Wildlife Refuge, with the remaining portion on privately owned land. In Oregon, 89 percent of Oregon spotted frog populations are at least partially in public ownership (Forest Service, BLM, and the U.S. Fish and Wildlife Service). Only two sites in the Deschutes drainage, La Pine and Little Deschutes River, are primarily under private ownership. Small portions of the Little Deschutes River locality are also managed by the BLM. Fourteen of the remaining sites are within the Deschutes National Forest. One site is managed by the Mount Hood National Forest, with a small portion of it on privately owned land. All localities in the Willamette drainage are under the management of the Willamette National Forest. These localities include Gold Lake Bog (a Research Natural Area) and several sites within the Three Sisters Wilderness Area. The five sites in the Klamath Basin are under both Federal and private management. The Klamath Marsh National Wildlife Refuge is managed by the U.S Fish and Wildlife Service, but portions of that population also occur on private lands. The Wood River Wetlands locality includes land managed by BLM and private land. The Fourmile Creek and Buck Lake localities include private, BLM, and Winema National Forest lands. The Jack Creek

population is on the Winema National Forest and privately owned land. Five more recently discovered sites include three on Forest Service land and two that are partly on BLM land and partly on private land.

Threats to the species' habitat include development, livestock grazing, introduction of exotic plant species, plant successional changes, changes in hydrology due to construction of dams and alterations to seasonal flooding, poor water quality, and water contamination.

Habitat losses and alterations can affect amphibian species in a variety of ways, including eliminating immigration through losses of adjacent populations (see Factor E) and effects on critical aspects of the habitat (Hayes and Jennings 1986). These critical aspects may include suitable egg-laying and nursery sites, refuges from predation or unfavorable environmental conditions, and temperature maximums and minimums necessary for egg-laying, growth, and development (Hayes and Jennings 1986).

Several aspects of the Oregon spotted frog's life history make it particularly vulnerable to habitat alterations: (1) communal egg-laying at sites used year after year restricts the number of reproductive sites; (2) the species' warmwater microhabitat requirement results in habitat overlap with introduced warmwater fish species and other warmwater fauna (e.g., bullfrogs (Rana catesbeiana)); (3) the active-season warmwater requirement limits suitable habitat in the cool climate of the Pacific Northwest; (4) the species is vulnerable to the potential loss or alteration of springs used for overwintering; and (5) the site complexity (e.g., spatial structure) for overwintering, active season, and breeding habitats is more complex than for other frog species (Hayes et al. 1996; M. Hayes, pers. comm., 2002). Breeding habitat is probably the single most important habitat component for many aquatic-breeding amphibians because amphibian embryos and larvae depend on aquatic habitats for survival (Leonard 1997).

Loss of Wetlands: Conservative estimates indicate that over 33 percent of wetlands in Washington were drained, diked, and filled between presettlement times and the 1980s (Canning and Stevens 1990; Mcallister and Leonard 1997). Losses of Oregon spotted frog habitat have been greater because of the high degree of development in the low elevations of the Puget Trough. Similar losses of wetlands have occurred in Oregon (estimated 95 percent in the Willamette hydrographic basin and 98 percent in the Klamath Basin) (Hayes 1997; McAllister and Leonard 1997). Based on surveys of historic sites, the Oregon spotted frog is now absent from at least 76 percent of its former range. The species may be absent from as much as 90 percent of its former range because the collections of historic specimens do not adequately reflect
its actual geographic and elevational range (Hayes 1997; McAllister and Leonard 1997). This species is now found in the most suitable habitat remaining in its historic range at sites having the least-altered hydrology and the fewest introduced predators (Hayes et al. 1997).

Hydrological Changes: Most of the currently occupied Oregon spotted frog sites are threatened by changes in hydrology. Twenty-one of 28 (75 percent) sites surveyed have had some human-related hydrological alterations, ranging from minor changes (e.g., local ditching around springs) to substantial changes, including major modifications of historic flow patterns (Hayes 1997; Hayes et al. 1997; Pearl 1999). Dams in the upper watersheds of the Willamette Valley, the Deschutes drainage, and the Puget Trough have significantly reduced the amount of shallow overflow wetland habitat historically created by natural flooding and used by this species (Hayes 1997; Hayes et al. 1997; Pearl 1999). Inundation of large marsh complexes and habitat fragmentation due to the construction of reservoirs in the Cascades also have eliminated and degraded this species' habitat. Relatively small areas of suitable habitat (25 ha (63 ac) or less) remaining at 23 of 28 (82 percent) sites surveyed indicate a number of these sites may be at risk because so little suitable habitat is available (Hayes 1997; Hayes et al. 1997; Pearl 1999). More recently discovered sites have not changed this basic pattern (M. Hayes, pers. comm., 2002). Changing water levels at critical periods in the Oregon spotted frog's life cycle, whether natural or human-induced, can negatively affect the species. Lowered water levels expose individuals to predation by reducing cover and confining them to smaller areas where they are more vulnerable to predators (see Factor C). Water level reduction during the breeding season can result in the loss of the entire reproductive effort for the year due to drying out of the egg masses (see Factor E). Drought periods can result in reduced recruitment (addition of young individuals to the adult population) regionally (Hayes 1997; Pearl 1999). Several seasons of low water can eliminate populations of Oregon spotted frogs, particularly where a small population occupies a limited marsh habitat that has a high abundance of aquatic predators (Pearl 1999). Excessive seasonal flooding at critical periods can result in the loss of shallow wetlands needed for egg-laying and development.

Water Quality and Contamination: Water acidity (low pH) can inhibit fertilization and embryonic development in amphibians, reduce their growth and survival through physiological alterations, and produce developmental anomalies (Hayes and Jennings 1986; Boyer and Grue 1995). A low pH may enhance the effects of other factors, such as activating heavy metals in sediments. An elevated pH, acting singly or in combination with other factors such as low dissolved oxygen, high water temperatures, and elevated un-ionized ammonia levels, may have detrimental effects on developing frog embryos (Boyer and Grue 1995).

Studies comparing responses of amphibians to other aquatic species have demonstrated that amphibians are as sensitive, and often more sensitive, than other species when exposed to aquatic
contaminants (Boyer and Grue 1995). Immature amphibians absorb contaminants during respiration through the skin and gills. They may also ingest contaminated prey. Pesticides, herbicides, heavy metals, nitrates, and other contaminants introduced into the aquatic environment from urban and agricultural areas are known to negatively affect various life stages of a wide range of amphibian species, including ranid frogs (Hayes and Jennings 1996; Boyer and Grue 1995; Hecnar 1995; Environment Canada 1998; Materna et al. 1995; Northern Prairie Wildlife Research Center 1998).

For example, the use of synthetic pyrethroids for insect pest control, including use in agricultural

and aquatic systems, has increased. Although pyrethroids are relatively non-toxic to birds and mammals, they are extremely toxic to aquatic organisms, including fish and invertebrates. Their effects on amphibians, however, are less well-known. Materna et al. (1995) demonstrated negative effects (inactivity, convulsive actions, death) of one widely used synthetic pyrethroid pesticide, esfenvalerate, on leopard frog (Rana spp.) tadpoles in laboratory and field experiments.
Methoprene, another chemical widely applied to wetlands for mosquito control, has been linked to abnormalities in southern leopard frogs (Rana utriculata), including completely or partially missing hind limbs, discoloration, and missing eyes. Missing eyes and delayed development in northern cricket frogs (Acris crepitans) have also been linked to methoprene (Donald W. Sparling, Patuxent Wildlife Research Center, pers. comm., 1999).

Poor water quality and water contamination have probably played a role in the decline of Oregon spotted frogs, although data specific to this species is limited. Eutrophic (nutrient-rich) conditions, characterized by blooms of algae that can produce a high pH and low dissolved oxygen, have increased in Upper Klamath Lake and may have contributed to the absence of Oregon spotted frogs there. Kirk (1988) documented spotted frog mortality due to forest spraying of DDT in 1974. Marco (1997) demonstrated the strong sensitivity of Oregon spotted frog tadpoles to nitrate and nitrite ions and suggested that nitrogen-based chemical fertilizers may have contributed to the species' decline in the lowland areas of its distribution. Recommended levels of nitrates and nitrites in drinking water are moderately to highly toxic for Oregon spotted frogs, indicating EPA water quality standards do not protect sensitive amphibian species (Marco et al. 1999).

Although the effects on amphibians of rotenone, used to remove undesirable fish from lakes, are poorly understood, mortality likely occurs at treatment levels used on fish. The role of rotenone treatments in the disappearance of Oregon spotted frogs from historic sites, however, is unknown (Hayes 1997).

In 1999, Four Rivers Vector Control planned to apply pyrethroids, methoprene, and other pesticides in wetlands and other bodies of water within the range of the Oregon spotted frog. This company is funded primarily by homeowners, homeowner associations, and businesses in the Sunriver area of Oregon to control mosquitos. Due to the concerns about the use of methoprene, an informal meeting of biologists from the Deschutes National Forest, Oregon Department of Fish and Wildlife, the Service, and Four Rivers Vector Control addressed the possible effects of the mosquito abatement program on the Oregon spotted frog. To reduce impacts to the species, the company is not permitted to use the chemical on the Deschutes National Forest and is voluntarily restricting its use to a few sites. Multiagency surveys were initiated in 1999 to further determine the species' distribution in this area. Additional recommendations on the use of methoprene in the Sunriver area may be issued in the future (Carol Morehead, Deschutes National Forest, pers. comm., 1999; Dede Steele, Service, pers. comm., 1999).

Development threatens Oregon spotted frog habitat at several sites. The uplands surrounding the an Oregon spotted frog site on Dempsey Creek have considerable potential for residential development. Potential development at the newly discovered Beaver Creek site in Washington includes a gravel extraction operation, golf course, and housing development ((McAllister and Leonard 1997; K. McAllister, pers. comm., 1999). Development at these sites would likely result in habitat loss and hydrological changes, as well as changes in water quality and introduction of contaminants into the aquatic environment. Although the Washington Department of Natural Resources (WDNR) established the Trout Lake Natural Area Preserve, and The Nature Conservancy also purchased some land at Trout Lake, the remaining land is not secure and is vulnerable to subdivision. In Oregon, the LaPine Creek site landowner has expressed a desire to develop the property. Future widening of U.S. Highway 97 may remove a substantial portion of a breeding pond located in an Oregon Department of Transportation right-of-way.

Livestock Grazing: The effects of livestock on the species vary with the site, livestock numbers, and the intensity of grazing. Livestock graze and trample emergent and riparian vegetation, compact soil in riparian and upland areas, and introduce urine and feces to water sources (Hayes 1997, 1998a; 61 FR 25813). The resulting increases in temperature and sediment production, alterations to stream morphology, effects on prey organisms, and changes in water quality have negatively affected Oregon spotted frogs.

Fourteen of 28 (50 percent) sites surveyed were directly or indirectly influenced by livestock grazing (Hayes 1997; Hayes et al. 1997; Pearl 1999). Severe habitat modification has been caused by too many cattle at several Oregon spotted frog localities in Oregon. Large numbers of cattle at a site may negatively affect Oregon spotted frog habitat, particularly at springs that possibly are used as overwintering sites (Hayes 1997). Preliminary results from exclosure studies at two sites in Oregon show significant improvement in vegetation where cattle are excluded (M. Hayes, pers. comm., 1999). Fencing to exclude livestock to protect the riparian corridor at Jack Creek in Oregon also excludes native grazers, such as elk, and may be resulting in the loss of Oregon spotted frog habitat to succession (changes in plant communities) (Hayes 1998a).

Changes in Vegetation: Exotic plant invasions, such as reed canary grass (Calamagrostis spp.), may completely change the structure of wetland environments and can create dense areas of vegetation that are unsuitable as Oregon spotted frog habitat (McAllister and Leonard 1997). Exotic vegetation was found at 20 of 28 (71 percent) sites surveyed. Reed canary grass dominates large areas at lower elevations and is apparently continuing to broaden its range to higher elevations (Hayes 1997; Hayes et al. 1997; Pearl 1999).

Plant succession may be a factor at almost all Oregon spotted frog sites, particularly where marsh-to-meadow changes are occurring (Hayes 1997). Pearl (1999) suggested that, in lake basins with a variety of aquatic habitats available, reproductive sites only exist within a narrow successional window, although a broader range of habitat types is used by adults in the nonbreeding season. As marsh size decreases due to plant succession, shallow warm water sites required by this species are lost to increased shading by woody vegetation (Pearl 1999). Recent succession-related losses of Oregon spotted frog habitat apparently have been considerably greater than succession-related habitat gains (Hayes 1997; Hayes et al. 1997). Such succession related losses may be accelerated by human activities, livestock grazing, altered hydrology, and development.

B. ***Overutilization for commercial, recreational, scientific, or educational purposes.***

Intentional collection of Oregon spotted frogs and vandalism of their habitat are not presently known to be a problem. Simply listing a species as endangered or threatened publicizes the species' rarity, however, and can precipitate both legal and illegal commercial or scientific interest. The species can be threatened by unauthorized and uncontrolled collection for commercial and scientific purposes, by researchers or by curiosity seekers.

C. ***Disease or predation.***

Most Oregon spotted frog populations are small, and small populations that are already stressed by other factors, such as drought or low food availability, are more vulnerable to random, naturally occurring events. Amphibians are affected by a variety of diseases, and some diseases are known to negatively affect declining amphibian species.

Disease: Little information exists on the specific effects of disease and parasitism on Oregon spotted frogs. Red-leg syndrome has been identified in several declining amphibian species (Berger 1999). However, this syndrome is not known to be a significant problem for the Oregon spotted frog (Andrew Blaustein, Oregon State University, pers. comm., 1999). The fungus Saprolegnia has been suggested as one of the causes of amphibian declines in the Pacific Northwest and is probably a much more significant threat to the Oregon spotted frog. McAllister and Leonard (1997) reported destruction of developing Oregon spotted frog egg masses by this fungus.

Amphibians exposed to ultraviolet-B radiation (UV-B), a type of solar radiation that causes damage to plants and animals, may be more susceptible to pathogens and parasites that can interfere with normal development and increase mortality. Kiesecker and Blaustein (1997) found increased mortality associated with the fungus Saprolegnia ferax in amphibian embryos exposed to UV-B. This suggests the possibility that mortality is increased by the combined effects (synergism) of the fungus and UV-B. Field experiments conducted in the Oregon Cascade Mountains determined that ambient levels of UV-B from the sun can cause high rates of mortality and deformities in embryos of some amphibian species (Blaustein et al. 1997). Amphibian species, such as the Oregon spotted frog, that lay their eggs in areas with little vegetative cover will experience greater exposure to UV-B. Oregon spotted frog hatching success, however, was not affected in one study of the effects of ambient levels of UV-B. Additional experimental tests at various life stages are warranted as changing atmospheric conditions and fluctuating UV-B levels may decrease hatching success at the study sites at different times or in other regions (Blaustein et al. 1997).

The North American Reporting Center for Amphibian Malformations (Northern Prairie Wildlife Research Center 1997) documents amphibian malformations throughout the United States. Malformations of several Rana species, including the Cascades frog (Rana cascadae), red-legged frog (R. aurora), foothill yellow-legged frog (R. boylii), and bullfrog, have been reported within the current and historic range of the Oregon spotted frog in Washington, Oregon, and California. There is one report from Thurston County, Washington, of an Oregon spotted frog with an extra forelimb (Northern Prairie Wildlife Research Center 1997).

Predation: The warm water microhabitat requirement of the Oregon spotted frog, unique among native ranids of the Pacific Northwest, exposes it to a number of introduced fish species (Hayes 1994). Introduced fish species within the historic range of the Oregon spotted frog may have contributed to losses of populations. These species include smallmouth bass (Micropterus dolomieu), largemouth bass (Micropterus salmoides), pumpkinseed (Pomoxis gibbosus), yellow perch (Perca flavescens), bluegill (Lepomis macrochirus), brown bullhead (Ameiurus nebulosus), black crappie (Pomoxis nigromaculatus), warmouth (Lepomis gulosus), brook trout (Salvelinus fontinalis), rainbow trout (Oncorhynchus mykiss), and fathead minnow (Pimephales promelas) (Hayes and Jennings 1986; Hayes 1997; Hayes et al. 1997; McAllister and Leonard 1997; J. Engler, pers. comm., 1999). Oregon spotted frogs, which are palatable to fish, did not evolve with these introduced species and may not have the mechanisms to avoid predatory fish that prey on the tadpoles of native amphibians.

Surveys from 1993 to 1997 in British Columbia, Washington, and Oregon documented at least one introduced predator in 20 of 24 sites (Hayes et al. 1997). Brook trout, occurring at 18 sites, was the most frequently recorded introduced predator. Although differences in temperature requirements between the two species may limit their interactions, brook trout apparently occur with the Oregon spotted frog at cold water springs where the latter species probably overwinters and where cooler water is favorable to brook trout (Hayes et al. 1997). Brook trout predation may have affected Oregon spotted frog populations during the 1992 and 1994 droughts (Hayes et al. 1997). Brook trout are likely to prey on Oregon spotted frog larval stages under drought conditions. Dropping water levels cause overlap in habitat use between these two species by reducing refuges and concentrating vulnerable life stages of the Oregon spotted frog (Hayes et al. 1997; Hayes 1998c).

Demographic data suggest introduced fish have a negative effect on Oregon spotted frogs because sites with a disproportionate ratio of older spotted frogs to juvenile frogs (i.e., poor recruitment) also have significant numbers of brook trout and/or fathead minnow (Hayes 1997, 1998a). Field experiments are needed to accurately determine the role of predation by introduced fish on Oregon spotted frogs. There are, however, relevant studies of the relationship between introduced fish and closely related frog species. A study of the impacts of introduced trout on Columbia spotted frog populations in Idaho revealed that, although fish and adult frogs coexisted at many of the stocked lakes, most stocked lakes contained fewer than 10 adult frogs and no egg masses or tadpoles (Pilliod and Peterson 1997). Other factors probably complicate the apparent cause and effect relationship between introduced fish and the Oregon spotted frog. Field experiments have demonstrated that smallmouth bass in combination with introduced bullfrogs negatively affect red-legged frogs by influencing their micro-habitat use, growth, and development (Kiesecker and Blaustein 1998). Pearl (1999) concluded that brook trout are probably the most significant threat to one population in Oregon and, when combined with low water conditions, can lower recruitment in drought years. Although, there are no experimental data, observations and evidence from other amphibian species strongly suggest introduced fish represent a significant threat to Oregon spotted frogs (Pearl 1999).

Bullfrogs have been introduced into the Pacific Northwest from eastern North America. Bullfrogs will eat native frogs and can out-compete or displace them from their habitat or optimal conditions (Kiesecker and Blaustein 1998). They are able to out-compete native frogs because: (1) bullfrogs have evolved with many of the introduced fish species and developed defenses against these predators; (2) bullfrog tadpoles are not palatable to fish or birds (Kruse and Francis 1997; McAllister and Leonard 1997); (3) bullfrog tadpoles may displace tadpoles of other frog species from warmer water where conditions are optimal for to cooler water, which slows development (Hayes 1994; Kiesecker and Blaustein 1998); and (4) bullfrog tadpoles are more resistant to the effects of pesticides and heavy metals than other ranid frogs (Hayes and Jennings 1986).

Bullfrogs share similar habitat and temperature requirements with the Oregon spotted frog, and overlap in time and space between the two species is probably extensive. The introduction of bullfrogs may have played a role in the disappearance of Oregon spotted frogs from the Willamette Valley and the Puget Sound area in Washington. The digestive tracts of a sample of 25 adult bullfrogs from Conboy Lake contained nine Oregon spotted frogs, including seven adults. A later examination of the stomachs of two large bullfrogs revealed two adult or subadult Oregon spotted frogs in one stomach and four in the second.

Bullfrogs, however, have probably coexisted with Oregon spotted frogs for nearly 50 years in the Glenwood Valley, which includes Conboy Lake National Wildlife Refuge (Engler and Hayes 1998). The coexistence of these two species at this site may be related to differences in seasonal and permanent wetland use. Some female spotted frogs reach a larger size at Conboy Lake than anywhere within the species' range and do not appear to be vulnerable to bullfrog predation. Bullfrogs, however, tend to be smaller at Conboy Lake than elsewhere in their range. Winterkill may be a factor in controlling the bullfrog population at Conboy Lake (Engler and Hayes 1998).

D. *The inadequacy of existing regulatory mechanisms.*

The Oregon spotted frog was listed as a state endangered species in Washington in August 1997

(Watson et al. 1998; WAC 232-12-014). Although there is no state Endangered Species Act in Washington, the Washington Fish and Wildlife Commission has the authority to list species (RCW 77.12.020). State listed species are protected from direct take, but the designation does not provide protection for their habitat (RCW 77.15.120). Under the State Forest Practices Act, however, the Washington State Forest Practices Board has the authority to designate critical wildlife habitat for state-listed animal species affected by forest practices (WAC 222-16-050; WAC 222-16-080). Critical wildlife habitat has not been designated by the Forest Practices Board for the Oregon spotted frog.

Oregon has a state Endangered Species Act, but the Oregon spotted frog is not state listed. Although this species is on the Oregon sensitive species list and is considered critically sensitive, this designation provides little protection (Oregon Department of Fish and Wildlife 1996; OAR 635-100-0040). Once an Oregon “native wildlife” species is federally listed as threatened or endangered, it is included as a state listed species and receives some protection and management, primarily on State-owned or managed lands (OAR 635-100-0100 tp OAR 635-100-0180; ORS 496.171 to ORS 496.192).

The Environmental Protection Agency (EPA) recently approved new water quality standards for temperature, dissolved oxygen, and pH that Oregon proposed in 1996. The EPA examined the effects of implementing these standards on the Oregon spotted frog, as well as other candidate and proposed species. The EPA concluded, however, that there is too little available information to make a determination of the effects on this species. These water quality standards may change again within the next two years to improve habitat conditions for salmonids and warmwater fish. It is uncertain, however, if the Oregon spotted frog will be considered as part of the review and approval process for these new standards (Elizabeth Materna, Service, pers. comm., 1998, 1999). However, they are not required to do so. Species that have been proposed for listing are covered by the conference provision under section 7(a)(4) of the Act. For example, the Oregon spotted frog is not considered a sensitive species by the Bureau of Land Management (BLM) or U.S. Forest Service (Forest Service). Listing the Oregon spotted frog as an endangered species would provide protection for this species under sections 7, 9, and 10 of the Act.

Section 404 of the Clean Water Act is the primary Federal law that could provide protection for the Oregon spotted frog’s aquatic habitat. Through a permit process under section 404, the U.S. Army Corps of Engineers (Corps) regulates the discharge of all fill into waters of the United States, including navigable waters and wetlands. In Washington and Oregon, current section 404 regulations allow the issuance of nationwide permits for projects involving the permanent loss of less than 1.2 ha (3 ac) of headwaters or isolated waters, including wetlands, unless a listed species may be jeopardized. Projects under a nationwide permit receive minimal public and agency review. Individual permits, which are subject to a more rigorous review, could be required for projects that have more than minimal impacts. The Corps, however, rarely requires an individual permit when a project qualifies under a nationwide permit, unless a threatened or endangered species or other resources are significantly and adversely affected by the project. Oregon spotted frog habitat could be affected by a project requiring only a nationwide permit from the Corps. Habitat can also be affected by agricultural practices that are exempt from regulation under section 404 of the statute, such as maintenance of existing agricultural drainage systems and other activities associated with an ongoing farming operation in existing cropped wetlands.

Current Conservation Efforts: In July 2000, the U.S. Fish and Wildlife Service entered into a Conservation Agreement with the Forest Service and the Oregon Department of Fish and Wildlife. The objective of the Conservation Agreement was the protection and conservation of the two Oregon spotted frog populations in the Mink Lake Basin in the Three Sisters Wilderness Area of the Willamette National Forest. Survey, monitoring, management, and education activities will occur during this 10-year agreement to address the threats that include site size, introduced fish (i.e., brook trout), effects of drought, habitat succession, and isolation of these populations.

Surveys were conducted in the Sunriver area near Crescent, Oregon, in 1999. These surveys were part of a multi-agency effort to learn more about the Oregon spotted frog's distribution and make a determination on whether the chemical methoprene should continue to be used for mosquito control in this area (see Factor A).

The Big Marsh site in Oregon, which has the largest population of Oregon spotted frogs, is currently undergoing modification to restore historic habitat conditions. Big Marsh is included in the Oregon Cascades Recreation Area. Restoring wetland values and providing for semi-primitive recreation are goals for this area. Wetland restoration efforts have included removing a diversion structure and breaching berms in an effort to put more water back in the marsh. The effect of this restoration effort on Oregon spotted frogs is unclear and is being monitored.
In 1995, Ridgefield National Wildlife Refuge initiated a series of distributional surveys for a variety of species, including the Oregon spotted frog, at Conboy Lake National Wildlife Refuge. Subsequent research at Conboy Lake, in cooperation with Dr. Marc P. Hayes, has included demographic studies, egg mass surveys, and a bullfrog diet study to assess the impacts of bullfrog
predation on Oregon spotted frogs. Continued research into the breeding requirements and productivity of the Oregon spotted frog and habitat utilization patterns of the Oregon spotted frog
and the bullfrog at Conboy Lake National Wildlife Refuge has been funded by a Cooperative Agreement between the Service and Dr. Hayes. The final report on the oviposition aspects of the study was completed in 2000. The bullfrog habitat partitioning study is continuing. Information from this research will be used in making management decisions to ensure the long-term survival of the Oregon spotted frog on the refuge and will be useful for making management recommendations for other sites as well.

In 1997, Port Blakely Tree Farms, L.P., Washington Department of Fish and Wildlife, and the Service initiated a cooperative study in response to the interest of private landowners to better manage and protect property for the Oregon spotted frog at the Dempsey Creek site. The goals of this study were to examine this species' habitat use patterns, especially as they relate to hydrology and cattle grazing, and to estimate the size of this population, develop an index to monitor population trends, determine seasonal movements, and identify sexual differences in movement patterns. A final report on the ecology of this remnant population was completed in 2000. Research and monitoring continues at this site.

In 1997 and 1998, the Oregon spotted frog population at Trout Lake Natural Area Preserve and the Trout Lake beaver dam wetlands were surveyed in a study to determine breeding size and the relative size of this population. Research and monitoring continues at this site.

In Spring 2001, personnel from Nisqually National Wildlife Refuge and Washington Department of Fish and Wildlife observed Oregon spotted frogs on a parcel of land purchased by the refuge for the Black River Unit of the Nisqually National Wildlife Refuge. Oregon spotted frogs were observed in an emergent wetland dominated by spikerush (Eleocharis spp.). However, much of the surrounding wetlands are dominated by reed canarygrass. A wetland restoration and enhancement project for this site has been developed by the Refuge.

Several habitat-use studies were conducted in 2000 and 2001 to establish movements of Oregon spotted frogs between active-season habitat and overwintering habitat. Uniquely marked individuals were followed at Dempsey Creek, Trout Lake, and Conboy Lake National Wildlife from September 2000 to February 2001.

Graduate thesis research, initiated in 2000 at the Beaver Creek site in Washington, includes a habitat manipulation study. Studies of the Penn Lake population in Oregon by the U.S. Fish and Wildlife Service and the U. S. Geological Survey was expanded in 2000 to include data collection on Oregon spotted frog movement patterns at montane sites using tagged individuals. Two Oregon spotted frog projects funded in 2000 by the Species-at-Risk Program of the Biological Resource Division of the U.S. Geological Survey included a genetics study and a study of a population's status, effects of introduced fish, and habitat associations.

In Oregon, a Conservation Agreement has been developed between the U.S. Fish and Wildlife Service, Oregon Department of Fish and Wildlife, the Sunriver Nature Center, and the communities of Sunriver, Crosswater, and Vandevert Acres. The purpose of the conservation agreement is to implement conservation measures for the Oregon spotted frog. The conservation area includes 188 ha (465 ac) within the 1,214 ha (3,000 ac) ownership area. In 1999, survey results suggested a total Oregon spotted frog population within the conservation area in excess of 4,000 adult frogs. Conservation measures to be implemented include providing information to local residents and property owners, using Integrated Pest Management strategies for wildlife pest and weed control, limiting application or release of chemicals to Oregon spotted frog water bodies to those approved by the Service, preventing or reducing bullfrog colonization, and monitoring Oregon spotted frog populations and water quality. Designation of conservation units assists the Service and other agencies in identifying priority areas for conservation planning under the consultation (section 7) and recovery (section 4) programs. The U.S. Fish and Wildlife

Service identified eight conservation units within the historic range of the Oregon spotted frog that are considered essential to the survival and recovery of the species.
These conservation units are:
(1) Puget Trough
(2) Willamette Valley below 500 m (1,500 ft)
(3) Southwest Washington Cascades
(4) West Oregon Cascades, 500 m (1,500 ft) to the crest
(5) East Oregon Cascades, (i.e., Deschutes Hydrographic Basin)
(6) Klamath Hydrographic Basin
(7) Closed Interior basins of Oregon and northern California
(8) Pit River drainage

E. *Other natural or manmade factors affecting its continued existence.*

Most species' populations are cyclic in nature, responding to such natural factors as weather events, disease, and predation. These factors, however, have less impact on a species with a wide and continuous distribution. Populations that are small, fragmented, or isolated by habitat loss, water development, water diversion, and other human-related factors are more vulnerable to extirpation by natural randomly occurring events and cumulative effects. The small sizes and isolation of the majority of Oregon spotted frog populations makes them even more vulnerable to drought, disease, and predation. Natural recolonization is unlikely in 23 of 28 (82 percent) Oregon spotted frog sites due to their high degree of isolation (Hayes 1997; Hayes et al. 1997; Pearl 1999).

Changes in water levels due to drought can cause seasonal loss of habitat and degradation of essential shoreline vegetation. Hayes (1997) assessed 9 of 24 (38 percent) Oregon spotted frog sites as having a moderate to high risk from drought. Drought risk was based on the potential for a drop in water level that could reduce or eliminate the species' habitat. Sites with the greatest risk included those depending on surface flow rather than flows from springs and sites having low precipitation levels. Sites with the greatest risk from drought are in the Klamath and Deschutes Basins of Oregon (Hayes 1997; Hayes et al. 1997). The impact of a drought on an Oregon spotted frog population depends on the amount of complex marsh habitat at a site, the availability of alternative breeding and rearing areas, and the abundance of aquatic predators (Pearl 1999).

Hybridization between Oregon spotted frogs and closely related frog species is unlikely to affect the survival of the Oregon spotted frog. Hybridization between Oregon spotted frogs and Cascade frogs has been demonstrated experimentally and verified in nature (Haertel and Storm 1970; Green 1985). The offspring are infertile, however, and the two species seldom occur together. No Oregon spotted frog and Columbia spotted frog populations are known to occur together.

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