PETITION TO LIST THE NORTHERN AND SOUTHERN POPULATIONS OF WESTERN SPADEFOOT (*Spea hammondii*) AS THREATENED AND ENDANGERED, RESPECTIVELY, UNDER THE CALIFORNIA ENDANGERED SPECIES ACT

Photo credit: James Bettaso, U.S. Fish and Wildlife Service

CENTER FOR BIOLOGICAL DIVERSITY

September 2025

NOTICE OF PETITION TO THE
STATE OF CALIFORNIA FISH AND GAME COMISSION

For action pursuant to Section 670.1, Title 14, California Code of Regulations (CCR) and sections 2072 and 2073 of the Fish and Game Code relating to listing and delisting endangered and threatened species of plants and animals.

The Center for Biological Diversity submits this petition to list the northern population of western spadefoot as threatened and list the southern population of western spadefoot as endangered throughout their respective ranges in California pursuant to the California Endangered Species Act (California Fish and Game Code § 2050 et seq.). This petition demonstrates that both populations of western spadefoot clearly warrant listing based on the factors specified in the statute.

I. SPECIES BEING PETITIONED

Common name: Western spadefoot
Scientific name: *Spea hammondii*

II. RECOMMENDED ACTION:

II. RECOMMENDED ACTION: List the northern population of western spadefoot as **threatened** and list the southern population of western spadefoot as **endangered**

III. AUTHOR OF PETITION:

Sofia Prado-Irwin, PhD
Center for Biological Diversity
[redacted]
spradoirwin@biologicaldiversity.org

I hereby certify that, to the best of my knowledge, all statements made in this petition are true and complete.

Signature: [redacted] Date: September 24, 2025

The **Center *for* Biological Diversity** is a national, nonprofit conservation organization with more than 1.7 million members and online activists dedicated to the protection of endangered species and wild places.

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I. Executive Summary

The Center for Biological Diversity submits this petition to list the two distinct populations of western spadefoot (*Spea hammondii*), the northern and southern populations, as threatened and endangered, respectively, throughout their ranges in California pursuant to the California Endangered Species Act ("CESA," California Fish and Game Code § 2050 et seq.). As outlined in this petition, without the protection of CESA, both populations of western spadefoot are highly likely to become further imperiled in the near future.

The two populations of western spadefoot, northern and southern, are genetically distinct and separated by a geographical barrier, the Transverse Ranges in Southern California. Habitat destruction and fragmentation are the most severe threats to both populations, but both are also threatened by climate change, invasive species, pollutants, and disease. Habitat loss and fragmentation have been particularly severe in Southern California. Genetic data shows that the southern population has become highly fragmented and exhibits very small effective population sizes, making it extremely vulnerable to extirpation. The southern population of western spadefoot therefore meets the standard for listing as an endangered species, as it is in danger of extinction throughout all or a significant portion of its range (California Fish and Game Code § 2062). Such genetic data is not available from the northern population, but historical occurrence data and patterns of severe habitat loss indicate the northern population has also experienced significant declines and continues to be threatened by habitat destruction, fragmentation, and other factors. The northern population of western spadefoot therefore meets the standard for listing as a threatened species, as it is likely to become endangered in the foreseeable future in the absence of special protection and management efforts (California Fish and Game Code § 2067).

Western spadefoots rely on temporary wetlands connected to suitable upland grassland or shrubland habitat, most commonly vernal pool complexes. Urban development, intensive agriculture, extractive development, and roads have led to increasing destruction of vernal pools and other wetlands, reducing available habitat. An estimated 90%-95% of California's historic vernal pools have been lost, and those that remain are subject to significant development pressure. Currently at least six major development projects are likely to begin imminently or within the next few years that would significantly impact western spadefoot.

The western spadefoot currently receives no state or federal species-specific protections. The species was proposed for listing as threatened under the federal Endangered Species Act ("ESA") in 2023, but a final listing decision is unlikely to happen soon. The current federal administration has focused on limiting the application and reach of the ESA and is unlikely to increase protections for western spadefoot. Recent changes to the Clean Water Act have also weakened protections for aquatic habitats, leaving the species even more vulnerable. To halt the species' decline and give it a chance of recovery, it is imperative to list the northern and southern populations of western spadefoot as threatened and endangered under CESA.

II. Introduction

This petition summarizes the available scientific information regarding the taxonomy and natural history of the western spadefoot (*Spea hammondii*), its range, distribution, abundance, population genetics, and population trends in California. It also discusses the threats affecting its ability to survive and reproduce and the limitations of existing management measures in protecting the species. As demonstrated below, the two distinct conservation management units of the western spadefoot (northern and southern populations) meet the criteria for protection as "threatened" and "endangered," respectively, under the California Endangered Species Act (CESA) and would benefit greatly from such protection.

III. Life History (Species Description, Biology, and Ecology)

A. Species Description

The western spadefoot is a small- to moderate-sized, round anuran. Despite commonly being referred to as the "western spadefoot toad," they are not true toads, as they are not members of the family Bufonidae. To reflect this, we refer to the species as the western spadefoot throughout this petition.

<u>*Adults*</u>

The western spadefoot is a small- to moderate-sized, round anuran (**Figure 1**). Adult snout-to-vent length ranges from 3.8-6.3 cm (1.5-2.5 in) (Stebbins, 2003). The average snout-to-vent lengths of metamorphs is about 2.5 cm (Alvarez & Kerss, 2023). Adult and juvenile dorsal coloration varies between greenish, grayish, or brownish with irregular dark and light stripes or markings and tubercles with dark orange or reddish tips. Ventral coloring is solid cream or light gray. Western spadefoots have large eyes with pale gold irises and vertical pupils in bright light and big round pupils in the dark. They have teeth on their upper jaw, short and stout limbs, and a wedge-shaped, keratinized black spade on each hind foot, which they use for digging. The male mating call has been described as "hoarse" and "snore-like," lasting on average 0.5 to 1 second with a mean pulse rate of 29.4 to 44.5 pulses per second (Brown, 1976).

<u>*Tadpoles*</u>

Tadpoles have a large, round body and a thin, vertically flattened tail (**Figure 1**). Total length has been recorded up to 7.5 cm (3 in) (Stebbins & McGinnis, 2012). Dorsal coloration can be brownish, gray, or greenish with dark mottling. Ventral coloration is pale and iridescent. Their eyes are set relatively close together when viewed from above, and they have a beaked upper mandible, a notched lower mandible, and oral papillae that encircle the mouth (Stebbins, 2003). Some populations develop predaceous and cannibalistic tadpoles that have a beaked upper jaw, a notch in the lower jaw, and enlarged jaw muscles (Stebbins, 2003).

<u>*Eggs*</u>

Western spadefoot eggs are light olive green or sooty above and whitish below (Stebbins & McGinnis, 2012). They are enclosed in two jelly envelopes and form irregular cylindrical clusters of about 10 to 42 eggs (average 24) with a diameter of 3.2 to 5.7mm (0.1 to 0.2 in). Eggs are attached to underwater plant stems or other submerged objects in temporary and permanent ponds and quiet parts of streams (Stebbins, 2003; Stebbins & McGinnis, 2012).

Figure 1. *Western spadefoot adult (left) and tadpole (right). Photos by Chris Brown/USGS (left) and Joanna Gilkeson/USFWS (right).*

B. Taxonomy and Population Genetics

The western spadefoot's scientific name is *Spea hammondii*. It is in the family Scaphiopodidae, which consists of two genera of North American spadefoots: *Scaphiopus* and *Spea* (Blackburn & Wake, 2011). Species in these genera were formerly in the family Pelobatidae. However, based on phylogenetic analyses identifying divergences in mitochondrial DNA, this has since been revised (García-París et al., 2003). Scaphiopodidae now consists of the North American spadefoots while Pelobatidae consists of spadefoots in Europe, central and western Asia, and northwestern Africa (García-París et al., 2003).

Spea was considered a subgenus in the genus *Scaphiopus* until phylogenomic analyses demonstrated that these two genera were distinct (Tanner, 1989; Wiens & Titus, 1991). Although the recognition of *Spea* as a full genus is refuted by some (*e.g.*, Hall, 1998), this nomenclature is generally accepted (Crother et al., 2017; Tanner, 1989; Wiens & Titus, 1991).

Two closely-related species now known as the Great Basin spadefoot (*Spea intermontana*) and the Mexican spadefoot (*Spea multiplicata*) were previously considered to be conspecifics (*i.e.*, subspecies) of *Spea hammondii*, but differences in morphology, breeding behavior, and reproductive biology indicate that they are reproductively isolated and constitute distinct species (Brown, 1976; Hall, 1998).

Spadefoots west of the Sierra Nevada Mountains and in Baja California are currently considered one species: western spadefoot (*Spea hammondii*). However, genetic analyses indicate that there are two genetically distinct populations, or management units—a northern and a southern population--divided by the Transverse Ranges (**Figure 2**) (García-París et al., 2003; Neal et al., 2018).

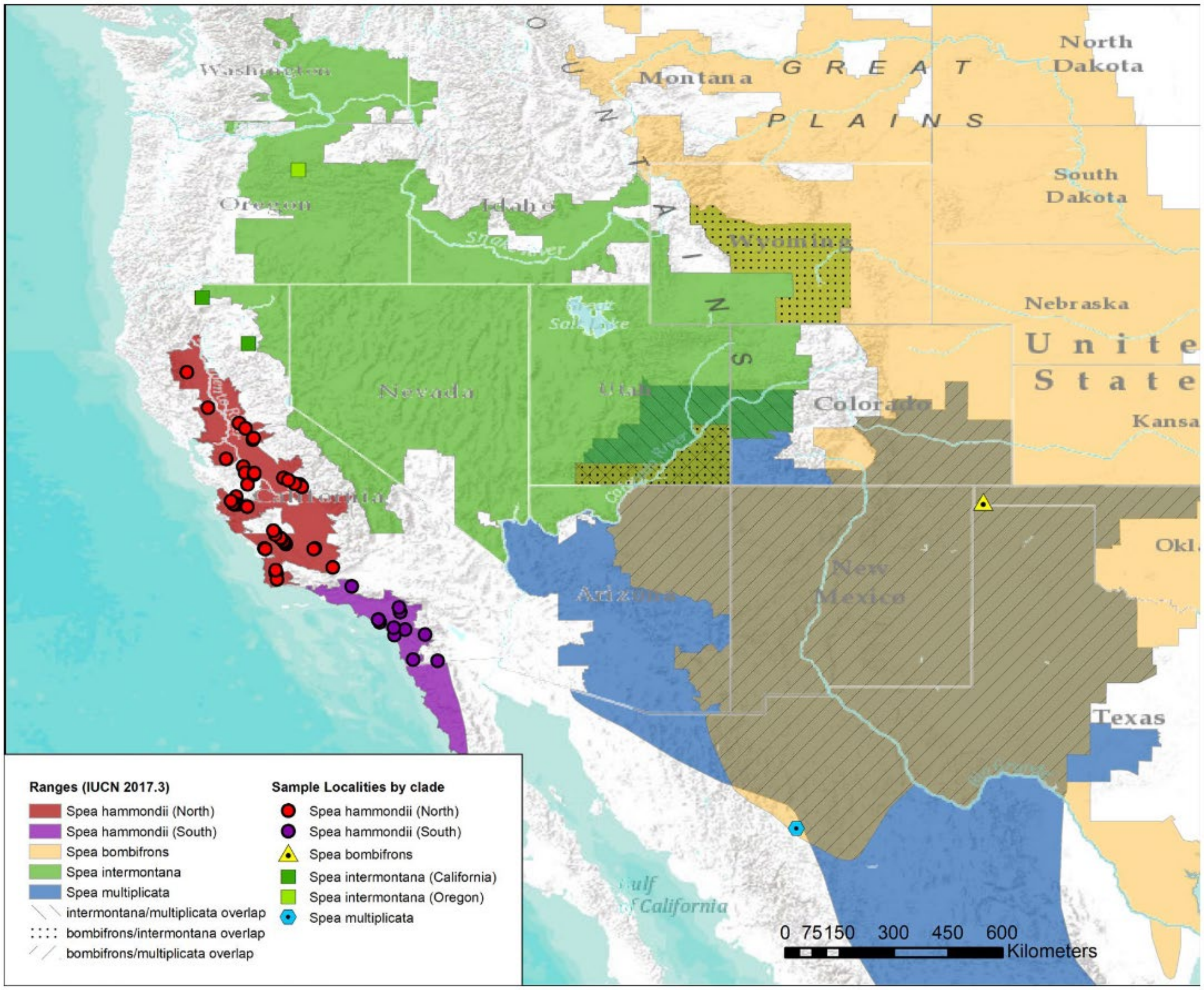

Figure 2. *North (red) and South (purple) distinct genetic populations of western spadefoot (Spea hammondii) in California. Other Spea species ranges include the Plains spadefoot (Spea bombifrons, yellow); Great Basin spadefoot (Spea intermontana, green); and Mexican spadefoot (Spea multiplicata, blue). Source: Neal et al., (2018).*

In a study analyzing two mitochondrial genes (cytochrome *b* and 16S RNA), researchers found that western spadefoots from Alameda County, CA and San Diego County, CA do not make up a monophyletic clade (García-París et al., 2003). Western spadefoots sampled from San Diego County were found to share a common ancestor with the Plains spadefoot (*Spea bombifrons*) while western spadefoots from Alameda were found to share a common ancestor with a clade formed by Plains spadefoot (*Spea bombifrons*), Great Basin spadefoot (*Spea intermontana*), and the San Diego County western spadefoots (**Figure 3**) (García-París et al., 2003). This suggests that the northern population of western spadefoot is genetically distinct from the southern population.

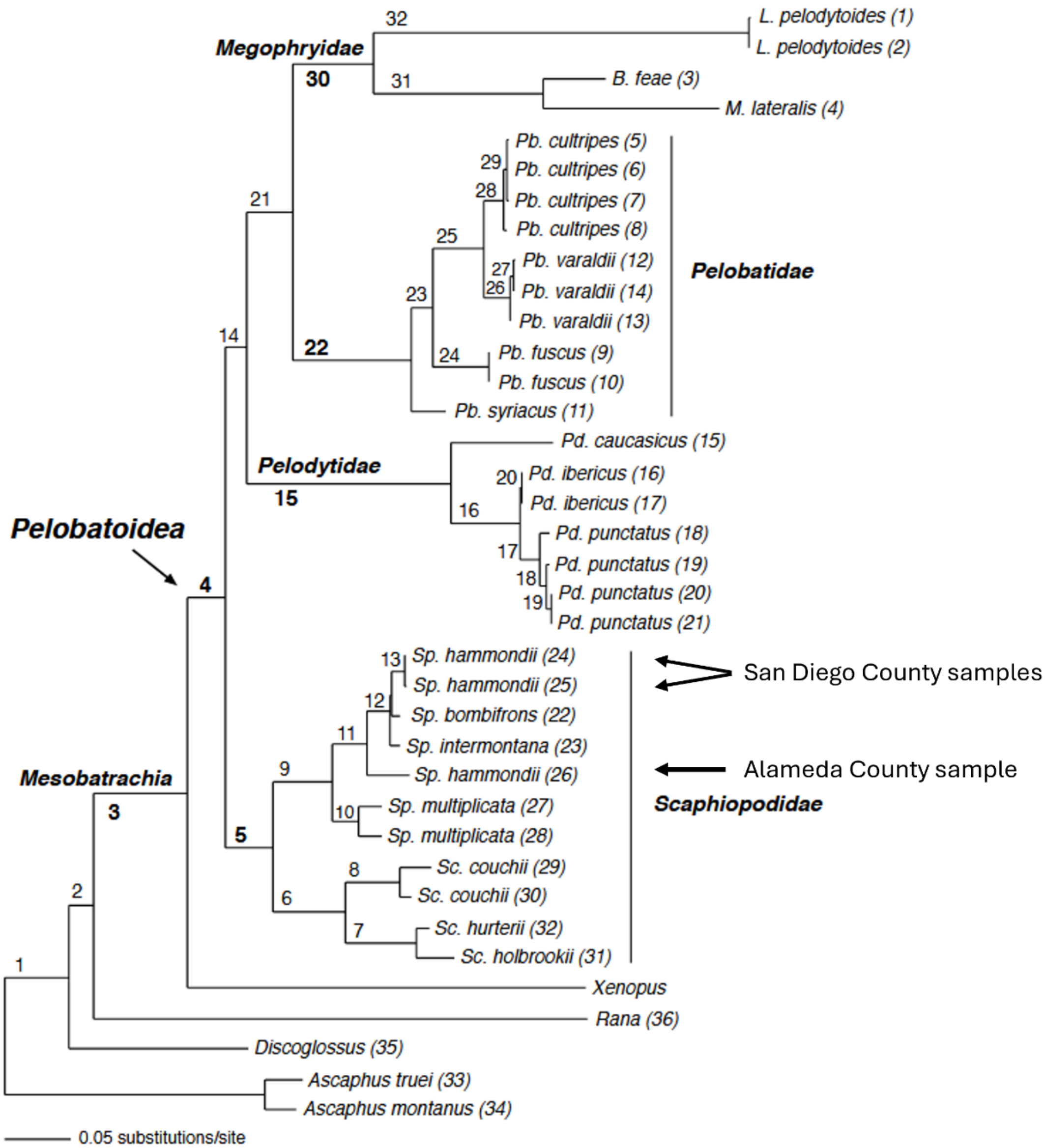

Figure 3. *Phylogenetic tree indicating that western spadefoots from San Diego County (24 and 25) may be genetically distinct from western spadefoots from Alameda County (26). Source: García-Paris et al. (2003).*

Neal et al. (2018) provide further evidence that northern and southern populations of western spadefoot are discrete populations. Investigating five nuclear protein-coding genes (AKAP9, NTF3, RAG1, Rhod1, and SIA) and one mitochondrial gene (ND2), the researchers demonstrated that the species consists of two genetically distinct clusters separated by the Transverse Ranges (Neal et al., 2018). Furthermore, the authors found that the two genetic clusters are ecologically distinct, with niche models resulting in unexpected differences in habitat suitability between the two groups (Neal et al., 2018). Results from an ecological niche modeling study further support the conclusion that the northern and southern populations are distinct, as they had separate glacial refugia and distinct dispersal corridors during and after the last glacial maximum (21,000 years ago), therefore the two populations likely have not met for many thousands of years (Gherghel & Martin, 2020). The authors argue that "[t]he strength of the genetic isolation suggests the two clusters may in fact represent distinct species" and "recommend that each cluster be considered its own conservation unit with potentially unique management needs" (Neal et al., 2018).

In Southern California, the genetic health of western spadefoot populations is deteriorating due to loss of habitat and connectivity between breeding pools. Neal et al. (2020) identified at least two genetic clusters of spadefoots in Orange County: inland and coastal populations. The analyses indicate long-term evolutionary divergence between the coastal and inland populations, potentially reflecting the strong philopatry and limited movement observed by Baumberger et al. (2019). However, Neal et al. (2020) also note that the genetic isolation found in their study and the limited movement documented by Baumberger et al. (2019) may be a result of spadefoots being relegated to marginal habitat in hilly terrain that may prevent them from moving longer distances and increasing gene flow. The researchers hypothesize that prior to intense urbanization and development, spadefoots in Southern California may have been able to move further distances across well-connected pond networks in the flat Los Angeles Basin (Neal et al., 2020).

Neal et al. (2020) also found genetic differentiation among adjacent ponds and estimated very low effective population sizes (N_e of 1.2 to 12.2) and numbers of effective breeders (N_{eb} of 1.4 to 19.8) at breeding ponds. These estimates are as low or lower than the estimated effective population sizes for the federally and state endangered California red-legged frogs (*Rana draytonii*) (Richmond et al., 2014) and the federally threatened California tiger salamanders (*Ambyostma californiense*) (Wang et al., 2011).

Although minimum viable effective population size has been found to vary depending on the species (Frankham, 1995; Frankham et al., 2014; Traill et al., 2010), general conservation management practice over the past few decades has followed a 50/500 rule, under which an effective population size of 50 is assumed sufficient to prevent inbreeding depression in the short term (over the duration of five generations) and an effective population size of 500 is assumed sufficient to retain evolutionary potential in perpetuity (Clarke et al., 2024; Frankham et al., 2014; Traill et al., 2010). However, the 50/500 rule was based on limited data from animal breeders and domestic and laboratory animals, and Frankham et al. (2014) recommended revising it to a 100/1000 rule to more accurately reflect the needs of wild populations and facilitate more effective conservation management.

The Orange County western spadefoot populations are small and isolated, and their effective population sizes are dangerously low, well below these conservation thresholds (Neal et al., 2020). This makes them extremely vulnerable to extirpation; they have a high genetic risk of severe inbreeding as well as a high demographic risk of being wiped out by extended drought or landslides due to flooding after wildfire. To prevent local extinction of these populations, Neal et al. (2020) "strongly recommend a combined approach of 1) assisted migration to counteract the negative effects of inbreeding depression (Frankham et al. 2019) among both natural and artificial ponds within Inland and Coast regions, and 2) construction of additional, artificial breeding sites with sufficiently long predicted hydroperiods (Pyke 2004) to enhance the ability of spadefoots to naturally establish local metapopulations as effective strategies to maintain this declining vernal pool specialist on the Southern California landscape."

There is no population genetic information available for the northern population. However, as detailed below (**Habitat Loss and Fragmentation**), much of the historical habitat north of the Transverse ranges—including the vast majority of grassland vernal pool complexes that once existed throughout the Central Valley—has been lost, and remaining habitat is highly fragmented. Given this habitat loss and fragmentation, it is likely their populations are on a similar trajectory of genetic isolation, low genetic diversity, and risk of extirpation. Genomic studies of western spadefoot populations in central and northern California would be extremely valuable in further understanding the status of the northern population.

C. Life Cycle

Western spadefoots are a cryptic species with a biphasic life cycle that requires connected aquatic and terrestrial habitats. Adults are almost entirely fossorial except when they emerge aboveground and migrate to vernal pools during seasonal rains to breed in large aggregations and lay their eggs. Eggs and tadpoles are aquatic until juveniles metamorphose and exit the pools.

<u>*Terrestrial Adult Ecology and Behavior*</u>

Little is known about the terrestrial activity of western spadefoots. They spend most of their lives in self-made underground burrows, though sometimes they temporarily take refuge in burrows constructed by small mammals, like gophers, squirrels, or kangaroo rats (Stebbins, 2003; US Fish and Wildlife Service, 2005). They aestivate, or go into long-term torpor or dormancy, during the dry season. One study in Southern California found that western spadefoots spent 125 to 220 days in their aestivation burrows, though the authors acknowledge the study was limited due to small sample size (n=15), male bias (80% were male), and a drought year (Baumberger et al., 2019). Therefore, this information may not fully encompass the species' aestivation patterns or preferences, but it provides some insight regarding how long individuals may stay underground.

Adults are nocturnal and emerge from their summer burrows to forage and breed after rains in the late fall through late spring. The factors that trigger emergence are not well understood, though the sounds and vibrations from rain striking the ground seem to trigger the emergence of other North American spadefoot species (Couch's spadefoot [*Scaphiopus couchii*] and the Mexican spadefoot [*Spea multiplicata*]) (Dimmitt & Ruibal, 1980). In addition, flooding or wetting the soil where spadefoots are burrowed causes them to emerge (Dimmitt & Ruibal, 1980; Ruibal et al., 1969). Ruibal et al. (1969) also found that some Mexican spadefoots were active in their burrows and moved towards the surface prior to heavy rains, which suggests that the sounds and vibrations of rain may not be the only cues that trigger spadefoots to move to the surface. Recent research on western spadefoots suggests they may be more active outside of their burrows than was previously thought, with emergences throughout the year rather than during a specific seasonal window (Alvarez & Woodall, 2024). Additional research on western spadefoot movement would be extremely useful in understanding their habitat use and needs throughout the wet and dry seasons.

Breeding Season Ecology and Behavior

Western spadefoots are generally active on the surface from October to May, with breeding occurring from January to May. However, they are opportunistic breeders, and the timing of surface activity can vary depending on rainfall and region. For example, western spadefoot breeding was observed between January and March in San Luis Obispo and Riverside counties after warm heavy rains (Morey & Reznick, 2004). And in San Diego County they bred after substantial rains in August as well as from October to December (Cass, 2007; Ervin et al., 2005). Western spadefoot breeding vocalizations and larvae were also documented from May to August in Kern County (Groff et al., 2012). In addition, Goldberg (2023) found that western spadefoots exhibit reproductive readiness throughout the year, further supporting the notion that they are opportunistic breeders that react to environmental conditions.

Western spadefoots form large (> 1,000 individuals), highly vocal breeding aggregations mostly in vernal pools (Jennings & Hayes, 1994), though they may also breed in intermittent streams, reservoirs, irrigation ditches, and even road ruts (Baumberger et al., 2019; Stebbins, 2003). Multiple bouts of breeding may occur in one season, though later aggregations include fewer individuals (Morey, 2005). Amplexus is pelvic. Females lay 300-500 eggs in irregular cylindrical clusters of 10-42 eggs attached to underwater plant material or the tops of submerged rocks (Morey, 2005; Stebbins, 2003).

Western spadefoots appear to exhibit strong site fidelity to natal pools and breeding sites, at least in coastal and inland Southern California populations (Baumberger et al., 2019; Neal et al., 2020). However, researchers suggest that this may be due to extensive habitat loss and the lack of flat, connected pond networks resulting from decades of intense urbanization throughout the region, which limits their movement (Neal et al., 2020). Limited information is available regarding site fidelity and movement patterns of the northern western spadefoot populations.

Eggs, Tadpoles, and Juvenile Ecology and Behavior

Eggs develop at temperatures of 9 to 30°C (Brown, 1967). In nature, they usually hatch in 3-4 days (Morey, 2005). Larval development time varies depending on resource availability, water temperature, and water volume (Denver et al., 1998; Jennings & Hayes, 1994). In San Luis Obispo and Riverside counties, larval development lasted an average of 58 days with a range from 30 to 79 days (Morey, 1998; Morey & Reznick, 2004). However, reduced water volume has been found to accelerate metamorphosis (Denver et al., 1998), and in laboratory experiments larvae metamorphosed in as few as 14 days (Morey, 1998; Morey & Reznick, 2004). Earlier metamorphs that had less time to develop were smaller compared to metamorphs that emerged later from pools that held water for longer, which could affect their chances for survival (Morey & Reznick, 2004).

Tadpoles stop eating and exhibit reduced movement prior to metamorphosis (Denver et al., 1998). Once they have developed forelimbs, they take short terrestrial excursions away from natal ponds at night and take refuge in moist cracks around the edges of the drying pool (Morey, 2005). At complete metamorphosis, when the tail is completely resorbed, tadpoles lose 30% or more of their body mass and weigh an average 3.7 g with a range of 1.5 to 10.4 g (Morey, 1998). The average snout-vent length of metamorphs is about 2.5 cm (Alvarez & Kerss, 2023).

Individuals reach sexual maturity when their snout-vent length is about 4 to 4.5 cm (Storer, 1925; Thomson et al., 2016). Age at maturity is not well understood, though it likely depends on environmental conditions and food availability. In lab studies, males reached sexual maturity at one to two years after metamorphosis while females took at least two years to reach sexual maturity (Morey & Reznick, 2001).

D. Diet, Foraging Ecology, and Predators

There is limited information regarding western spadefoot diet. Adults and juveniles appear to be generalist predators, hunting at night during the rainy season and preying on various invertebrates including terrestrial arthropods, beetles, moths, crickets, true bugs, flies, and earthworms (Morey & Guinn, 1992).

No information is available regarding larval diet of western spadefoots, though Mexican spadefoots (*Spea multiplicata*) have been found to have both carnivore and omnivore morphs, feeding either exclusively on fairy shrimp or on detritus and algae, respectively (Pfennig, 1990). Pfennig (1990) found that most carnivorous tadpoles were in pools that had high fairy shrimp densities and fast drying times, and carnivores developed much more quickly than omnivores (Pfennig, 1990). He hypothesized that tadpoles are by default omnivores unless they ingest a critical number of fairy shrimp that triggers them to be strictly carnivorous (Pfennig, 1990). It is possible that larval western spadefoots have similar diets.

Reported predators include California tiger salamander larvae, adult American bullfrogs, garter snakes, and raccoons (Morey, 2005). Birds and mammals likely prey on large larvae, especially in pools where the water is clear or where larvae density is increasing as the pools dry (Morey, 2005). Adult western spadefoots produce unpalatable skin secretions to ward off predators, and according to Morey (2005), "To the taste, the sticky skin secretions of an injured western spadefoot toad are strongly suggestive of a pharmacologically active substance; in the eyes or nose, the secretions cause a burning sensation."

E. Burrowing Behavior

Numerous reports state that western spadefoots can be found in burrows up to one meter deep (e.g., (Jennings & Hayes, 1994; US Fish and Wildlife Service, 2005)). However, the reports rely on a study titled "The terrestrial ecology of the spadefoot toad *Scaphiopus hammondii*," (Ruibal et al., 1969), which was conducted in southeastern Arizona and likely refers to the Mexican spadefoot (*Spea multiplicata*) before it was recognized as a distinct species from the western spadefoot. Therefore, while burrow depths may be similar, it is unclear how deep western spadefoot burrows can be.

A more recent study of western spadefoots in Southern California observed the depths of adult burrows to range between 1-18 cm (Baumberger et al., 2019), though the authors acknowledge the study was limited because it only included coastal populations, had a small sample size (n=15) with male bias (80% were male), and was conducted during a drought year. The data likely do not fully encompass the variation within the studied populations or the species that occurs throughout the state in coastal and inland populations. Juveniles have been documented digging burrows 10 to 20 cm deep (Morey & Reznick, 2001).

Burrow locations and depths may vary depending on the season. Mexican spadefoots have been found to have shallower burrows near breeding pools during the rainy season and deeper burrows away from breeding pools during the dry season (Ruibal et al., 1969). This may be similar for western spadefoots given that greater depth to bedrock increases the likelihood of western spadefoot occurrence (Neal et al., 2020).

F. Movement

Studies conducted in Southern California populations indicate that western spadefoot movement patterns may be site-specific and dependent on weather conditions.

In coastal populations, rainfall and relative humidity are significant drivers of movement and distance traveled (Baumberger et al., 2019; Halstead et al., 2021), though individual western spadefoots have been observed to move during the breeding season when no rain was present (Baumberger et al., 2019). Adults were found to disperse rapidly after breeding and travel up to 601 m from breeding pools during a relatively wet year (Halstead et al., 2021). This was substantially further than the 82-m maximum distance recorded at the same site in drier years (Baumberger et al., 2019).

Meanwhile, at inland breeding sites, the maximum distance from a breeding pond during a relatively wet year was 145 m (Halstead et al., 2021). Although the spatial, temporal, and seasonal variation of western spadefoot movement requires more investigation, it is clear that western spadefoots may travel long distances from breeding pools.

Little is known about when juveniles leave the breeding pool area or how far they travel, though most juvenile movements likely occur on calm, humid nights in April to June (Morey, 2005). A study conducted at Carnegie State Vehicular Recreation Area near Tracy, CA suggests that juveniles do not immediately leave dried pools after metamorphosis. Biologists documented newly metamorphosed juveniles on the soil surface immediately adjacent to aquatic breeding habitat, feeding and seeking refuge about 5 to 7 cm from the soil surface in deep cracks in dried pool bottoms for at least two weeks after metamorphosis was completed (Alvarez & Kerss, 2023). As juveniles outgrew the cracks, they would partially bury themselves in a moist soil layer under the thin surface crust of the dried pool for about 4 to 6 weeks (Alvarez & Kerss, 2023). During that time, the juveniles would explore up to 7 m from the refuge sites at night (Alvarez & Kerss, 2023). It is not known how far juveniles disperse.

According to herpetologist Dr. Steven Morey at the U.S. Fish and Wildlife Service ("USFWS"), western spadefoots are capable of traveling distances of at least 1 km and perhaps much more from breeding sites (Laabs et al., 2001). Other North American spadefoot species have been found to travel long distances. Timm et al. (2014) found adult eastern spadefoots (*Scaphiopus holbrookii*) migrating up to 449 m from breeding pools, and Richardson & Oaten, (2013) (as cited by Baumberger et al., 2019) showed that Great Basin spadefoots (*Spea. intermontana*) move up to 2350 m from breeding pools. Therefore, it is possible that western spadefoots may be capable of traveling further than 601 m, depending on available flat habitat and connected vernal pool complexes as well as weather conditions.

It is important to note however that western spadefoots may be more movement-limited than other spadefoot species. A genetic analysis of western spadefoots in Orange County revealed evidence of significant differentiation among populations, even among adjacent ponds, indicating limited dispersal between populations (Neal et al., 2020). Similarly, a telemetry study in the same region showed that the mean maximum distance western spadefoots travelled from breeding pools was only 69m, and the observed movement distances were not far enough for individuals to move between breeding locations, though the authors acknowledge the study was limited due to small sample size (n=15), male bias (80% were male), and a drought year (Baumberger et al., 2019).

G. Survivorship and Mortality

Limited information is available regarding the survivorship and mortality of western spadefoots. However, Halstead et al. (2021) estimated the annual probability of survival for adult western spadefoots in Southern California to be 51%. They found that adults had a higher mortality risk during the active breeding season compared to when they were in aestivation (Halstead et al., 2021). However, their observations and estimates do not include human-caused mortalities, like wildlife-vehicle collisions. Western spadefoots were found to be susceptible to road mortality and fragmentation and were ranked as having high road risk in an assessment of 166 reptile and amphibian species in California (Brehme et al., 2018).

Desiccation and predation are major threats to larval survival. Even though larvae have been documented to speed up their metamorphosis when pools start drying out, larvae are frequently at risk of desiccation if their pools dry out before they complete metamorphosis. Drying pools can also concentrate the larvae within the water, making it easier for predators, like birds and small mammals, to prey upon the larvae.

In Fresno County, Feaver (1971) reported that 73% of examined vernal pools dried out, causing 100% larval mortalities in pools that dried within four weeks. Combined with predation by concurrent California tiger salamander larvae, garter snakes, and predators like great blue heron, spotted skunks, and American bullfrogs, the spadefoot population had an 81.27% larval mortality rate (Feaver, 1971). This varies considerably with a study conducted in San Luis Obispo and Riverside counties, which reported that 15% of examined vernal pools dried out (Morey & Reznick, 2004). Survival and mortality rates likely fluctuate depending on weather conditions, food availability, and predator presence.

The survival rate of metamorphs is unknown and not reported in the scientific literature. However, it is speculated that the age and size of metamorphs may affect their fitness and survival (Morey & Reznick, 2001). Younger tadpoles that metamorphose earlier due to pools drying have smaller body size compared to individuals that metamorphose at an older age (Denver et al., 1998; Morey & Reznick, 2001), which could make them more vulnerable to drying out or cause them to spend more time at the surface foraging, which would increase their risk of mortality due to predation (Morey & Reznick, 2001). Other mortality risk includes being smashed by off-road vehicles (Goldberg, 2023) or being hit by vehicles on roads (Brehme et al., 2018).

H. Population Dynamics

Western spadefoots are demographically unstable, meaning their population recruitment varies from year to year depending on environmental conditions, especially rainfall (R. N. Fisher & Shaffer, 1996). The species therefore experiences large annual fluctuations in population sizes, as breeding and recruitment can boom during exceptionally wet years and be impossible in dry/drought years.

As is the case with many pond-breeding amphibians, due to their dependence on ephemeral aquatic breeding sites connected to suitable upland habitat, western spadefoots are particularly sensitive to habitat disruption. Increasingly fragmented habitats can create isolated subpopulations that have higher risks of local extinction (Neal et al., 2020). Although it is unclear if western spadefoot populations throughout California meet the strict definition of a metapopulation dynamic (i.e. subpopulations that experience an exchange of individuals leading to increased genetic diversity and the recolonization of suitable habitats following local extinction events (Baumberger et al., 2019; M. A. Smith & Green, 2005)), it is evident that the persistence of the remaining fragmented subpopulations of western spadefoots is fragile (Halstead et al., 2021; Neal, 2019; Neal et al., 2020). Species with such high demographic instability are sensitive to habitat alterations that may interfere with recolonization and reestablishment after unsuccessful recruitment years (R. N. Fisher & Shaffer, 1996), and are therefore more susceptible to local and regional extinctions.

IV. Habitat Necessary for Survival

Because of their biphasic life cycle, western spadefoots require aquatic breeding habitat connected to terrestrial over-summering habitat. Vernal pool complexes are ideal, with pools adequately spread out so individuals can travel between pools. Western spadefoots primarily occur below 365 m (1000 ft) elevation (Morey, 2005), though the species has been recorded as high as 1410 m (4626 ft) in San Diego County ((Lemm, 2006), as cited by Goldberg, 2023).

Western spadefoots are most often associated with grasslands, but they have also been found in coastal sage scrub, chaparral, oak woodlands, washes, floodplains of rivers, alluvial fans, playas, lowlands, and foothills (**Figure 4**) (Stebbins, 2003; Stebbins & McGinnis, 2012). Such habitats must be connected to suitable breeding habitat, which includes seasonal water bodies like vernal pools and intermittent streams, though they have also been found to breed in reservoirs, irrigation ditches, stock ponds, and even road ruts (Morey, 2005; Stebbins, 2003). They have also been found to breed in human-made mitigation ponds, with researchers reporting a 33% breeding success rate at such ponds in Orange County after 10 years (Baumberger et al., 2020).

Figure 4. *Western spadefoot breeding pools in Limestone Canyon, California (top) and Crystal Cove State Park, California (bottom). USGS photos. From Halstead et al. (2021).*

Hydroperiod length of breeding pools is important. Generally, the temporary nature of vernal pools allows species like western spadefoots to find refuge from predators that require permanent waterbodies, like invasive fish and American bullfrogs. In San Luis Obispo and Riverside counties, breeding pools persisted an average of 81 days (range 26-127 days) and complete larval development took an average of 58 days (range 30-79 days) (Morey & Reznick, 2004).

Although studies are limited, available information suggests that, like their movement patterns, habitat use by spadefoots seems to vary geographically, temporally, and seasonally. Recent studies indicate that individual home ranges vary depending on location, rainfall, relative humidity, temperature, and potential resource availability (Baumberger et al., 2019; Halstead et al., 2021). In Southern California, the mean 95% home range area was 0.52 ha (range 0.0067 to 6.1 ha), with coastal populations having a mean home range area 3.6 times larger than the inland populations (Halstead et al., 2021). It is unclear what is driving this difference, but it is important to account for larger and connected areas to accommodate home range sizes that change depending on the environmental conditions (Baumberger et al., 2019; Halstead et al., 2021).

It is crucial that suitable core terrestrial habitat is adjacent to breeding pools (Searcy et al., 2013; Semlitsch & Bodie, 2003). In Southern California, Halstead et al. (2021) predicted the 95th percentile of the population distribution to be within 486 m of breeding pools in coastal populations, which encompassed all but one individual's movement. For inland populations, the researchers predicted the 95th percentile of the population distribution to be within 187 m of breeding pools, which encompassed the movements of all inland individuals in the study (Halstead et al., 2021). The researchers emphasized the need to understand site-specific characteristics to estimate habitat needs, stating that larger, more conservative conservation buffers are "prudent when faced with the accompanying variation and uncertainty in western spadefoot behavior" (Halstead et al., 2021). This suggests that conservation management and planning for western spadefoots should include at least 486 m of terrestrial habitat connected to vernal pools for populations where movement dynamics are unknown.

As mentioned previously, researchers identified two genetically distinct clusters of western spadefoot in California. The northern genetic cluster and the southern genetic cluster are separated by the Transverse ranges (Neal et al., 2018). The two clusters were found to be ecologically differentiated and occupy different climatic niches, which suggests that the habitat needs of northern populations may differ from those of southern populations (Neal et al., 2018).

Micro-scale Habitat: Burrow Locations

The species may have some flexibility regarding habitat use and burrow location depending on the available habitat and environmental conditions (**Figure 5**). For example, Baumberger et al. (2019) found that western spadefoots in coastal sites in Orange County preferred to burrow in friable, sandy/loam soils with more sand and silt and less clay (Baumberger et al., 2019). The same study found that they preferred grasslands over shrubs and were more likely to burrow in or near existing pocket gopher and ground squirrel burrows on flat slopes with south-eastern aspects, though a small number of burrows were found in the only tree-dominated habitat at one study site (Baumberger et al., 2019). The presence of duff, or dead plant material, was common at burrows, and the authors suggest that the duff and tree cover might help to conserve soil moisture (Baumberger et al., 2019). This study was conducted during a drought year (2011-2012).

A later study conducted during a relatively wet year (2018-2019) compared movement patterns of spadefoots in the same coastal areas to those at more inland sites in Orange County (Halstead et al., 2021). The researchers observed different habitat use by the coastal populations during the wet year compared to the drought year, and they found that coastal populations had different habitat preferences compared to inland populations (Halstead et al., 2021). During the wet year, spadefoots at coastal sites avoided graminoids (i.e., grasses and grass-like plants), forbs, and shrubs, and some appeared to select burrow sites under trees or tall shrubs (Halstead et al., 2021). Meanwhile, inland western spadefoots did not show strong habitat preference, though they had a slight tendency to burrow in areas with bare ground, forbs, and shrubs. These differences suggest that habitat use and preferences may vary based on geography, season, and environmental conditions.

Figure 5. *Burrow locations for western spadefoot at Crystal Cove State Park, a coastal park dominated by coastal sage scrub (top) and Limestone Canyon, an inland park dominated by black mustard and non-native grasses (bottom). Western spadefoot burrow habitats vary based on local conditions. From Halstead et al. (2021).*

<u>Macro-scale Habitat: Landscape Connectivity and Vernal Pool Complexes</u>

Western spadefoots rely on well-connected vernal pools and vernal pool complexes with sufficient upland burrowing habitat for their long-term survival. Therefore, it is critical to consider their habitat requirements at a landscape level.

Multiple studies indicate that northern western spadefoots are more likely to occur in areas where there is 60% or more grassland cover within 2000 m of ephemeral pools (Halstead et al., 2022; Rose et al., 2020). In addition, Rose et al. (2020) found that they were more likely to occur in areas with sandy soils and a high proportion of grassland within 2000 m of vernal pools. They were also more likely to occur on slopes between four to 12 degrees in the foothills on the edge of the Central Valley at mid-elevations, rather than the valley floor. The lower suitability on the valley floor may reflect predation by introduced fish and bullfrogs at lower elevations (Rose et al., 2020).

In Southern California, western spadefoot habitat use was positively related to grassland or shrub/scrub cover and up to about 60% sand in the soil within 1000 m of vernal pools (Rose et al., 2022). They were negatively associated with slope, elevation, and distance from pools. This aligns with Neal et al. (2020), who found that depth to bedrock and slope were important habitat characteristics that impact suitability and facilitate population connectivity, with greater depth to bedrock and lower slope being the most ideal for spadefoots. This further emphasizes the importance of intact and well-connected vernal pool complexes, particularly in areas that are mostly flat or have gentle slopes with sandy soils, grasslands, and/or shrub/scrub.

V. Range and Distribution

The western spadefoot is nearly endemic to California, with the very southern part of its historical range extending from Southern California into northern Mexico (US Fish and Wildlife Service, 2005). Western spadefoots were historically distributed throughout lowland areas from southern Shasta County to northwestern Baja California, Mexico, occurring throughout the Central Valley, Sierra Nevada foothills, and coastal California south of the San Francisco Bay Area (**Figure 6**). (Thomson et al., 2016; US Fish and Wildlife Service, 2005). As of March, 2025 the California Natural Diversity Data Base lists 1,443 occurrences from 31 counties.[1] As noted by (US Fish and Wildlife Service, 2005), these records include data from as far back as 1911, and do not represent a systematic survey. Many sites from which older records are known have not been re-surveyed in recent years, and the status of many of the sites recorded before 2000—which includes all but 93 of the CNDDB occurrences—are unknown (US Fish and Wildlife Service, 2005).

As described above, the northern and southern populations of western spadefoots are genetically and ecologically distinct (García-París et al., 2003; Neal et al., 2018). The Transverse Ranges in Southern California present a barrier of unsuitable habitat and split the northern and southern clades (Neal et al., 2018).

[1] California Dept of Fish and Wildlife, CNDDB RareFind tool, query “western spadefoot.” Accessed 11 March 2025.

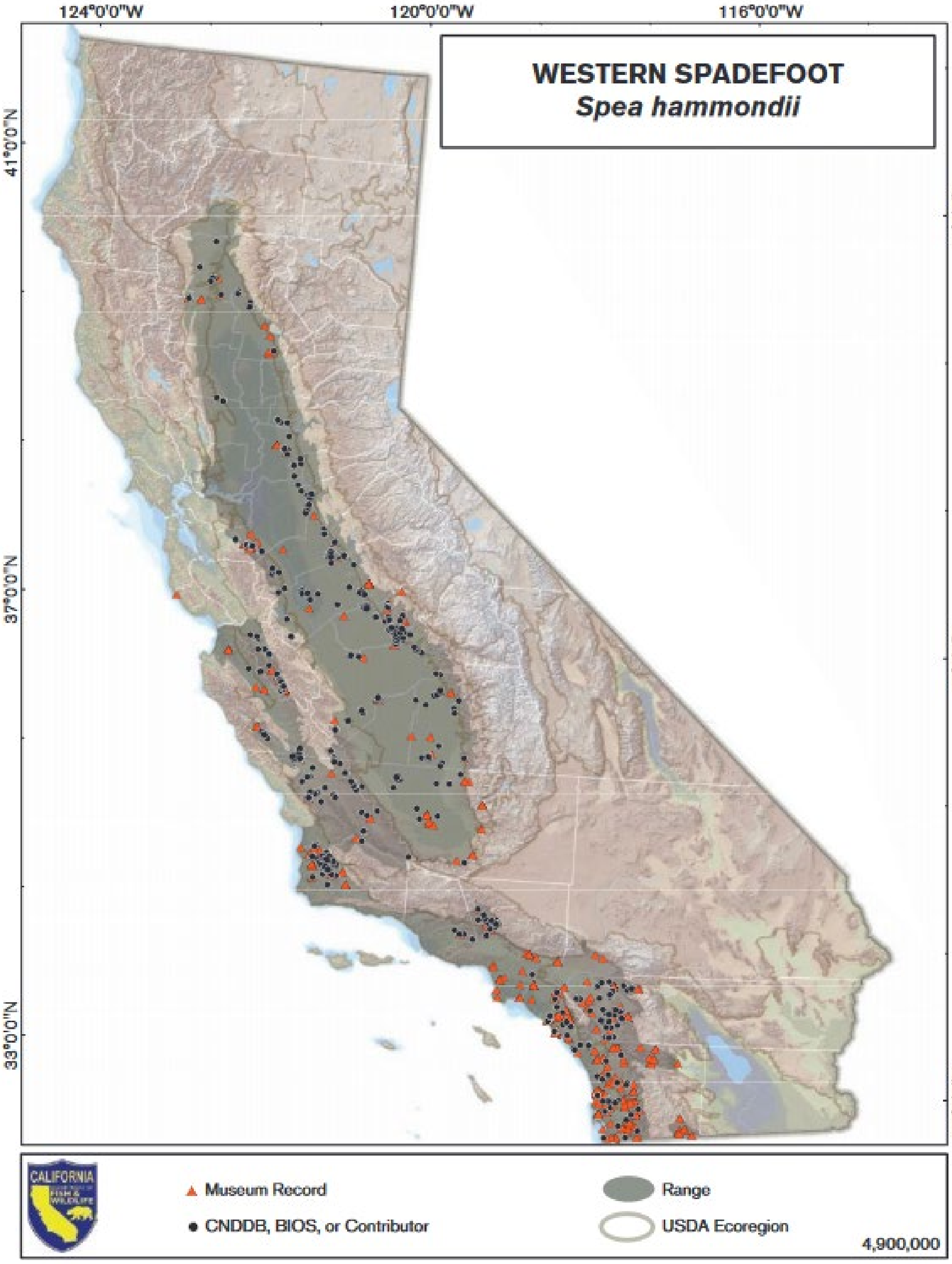

Figure 6. *Western spadefoot range, including contemporary observations and museum records from numerous sources. Source: Thomson et al., (2016).*

The species is now extinct through much of lowland Southern California as well as many historical locations in the Central Valley (R. N. Fisher & Shaffer, 1996; Jennings & Hayes, 1994; Stebbins & McGinnis, 2012; US Fish and Wildlife Service, 2005). In the 1990s researchers estimated that western spadefoot populations in Southern California had lost up to 80% of their native habitat, including vernal pools, while in northern and central California at least 30% of western spadefoot habitat had been lost (Jennings & Hayes, 1994). Recent studies indicate that the majority of remaining suitable habitat for southern populations is located in the southern half of its historical range (Rose et al., 2022), and only a small portion of the historical range for northern populations remains suitable (Rose et al., 2020).

The populations that have persisted in the San Joaquin Valley are on average higher in elevation than the historical range, suggesting that lowland populations experienced the greatest declines (Fisher & Shaffer, 1996; U.S. Fish and Wildlife Service, 2005). Predation by introduced fish and bullfrogs at lower elevations may be to blame (R. N. Fisher & Shaffer, 1996; Rose et al., 2020). A similar pattern was observed in Orange County, with most currently existing populations restricted to the less suitable uplands surrounding the Los Angeles Basin, rather than the lowland areas that historically harbored western spadefoots (Neal et al., 2020).

Vernal pools utilized by the species occur in grasslands, coastal sage scrub, oak woodlands, and chaparral. These areas have been decimated by urban and agricultural development, and as a consequence, the western spadefoot is now extirpated across much of its range in Southern California (Davidson et al., 2002; Thomson et al., 2016), and is declining in central and northern regions as well (R. N. Fisher & Shaffer, 1996; Jennings & Hayes, 1994; Rose et al., 2020).

As noted in the *U.S. Fish and Wildlife Service Recovery Plan for Vernal Pool Ecosystems of California and Southern Oregon* (2005), western spadefoots commonly co-occur in their breeding pools with numerous other federally and state listed threatened and endangered species, including California tiger salamander, California red-legged frog, vernal pool tadpole shrimp, vernal pool fairy shrimp, and California fairy shrimp.

VI. Abundance and Population Trends

Western spadefoots have experienced significant population declines over the past several decades. Concern over western spadefoot populations in the Central Valley and Southern California was documented as early as the 1970s (Jennings & Hayes, 1994), and a study conducted by Fisher & Shaffer (1996) reported a “severe pattern of decline” by the 1990s, with western spadefoots being completely extirpated from the Sacramento Valley and experiencing reduced densities in the eastern San Joaquin Valley.

Western spadefoot breeding pools occur in multiple habitat types that have been significantly reduced by urban and agricultural development—including grasslands, coastal sage scrub, oak woodlands, and chaparral—leading to their extirpation throughout much of Southern California (Neal et al., 2020; Thomson et al., 2016). Historically, regions including the Los Angeles Basin and San Diego County had extensive distributions of lowland vernal pools that provided suitable habitat for western spadefoot breeding, but human development has eliminated the natural landscape (Neal et al., 2020). In Southern California, more than 80% of known western spadefoot habitat has been destroyed or rendered unsuitable via urban or agricultural development (Jennings & Hayes, 1994). In central and northern California, at least 30% of western spadefoot habitat has been similarly lost (Jennings & Hayes, 1994). However, this has likely increased, as much of the region has been developed since the 1990s and only a small portion of the historical range for northern populations remains suitable (Rose et al., 2020). As urbanization and agricultural development continue, western spadefoots will continue to decline.

While explicit estimates of abundance across the species' range are uncommon, available data indicate that populations are struggling. For example, Thomson et al. (2016) describes recent surveys of Mather Airport (formerly Mather Air Force Base) in Sacramento County, which estimated that the population of breeding adults numbered only a few dozen. Breeding, or census, population size (N_c) is generally estimated by on-the-ground surveys and represents a count of mature individuals in the population. Effective population size (N_e) is not synonymous with census population size and is instead determined by population genetic analysis. However, effective population sizes are often significantly smaller than census population sizes; a common rule of thumb to convert census population size to effective population size is to apply the N_e/N_c ratio, which is generally close to 0.1. Thus, a census population of 100 individuals would likely have an effective population size around 10 individuals. Following this estimation, the effective population size of the small Mather Airport population is likely less than 10, well below the widely-accepted 50/500 rule (Clarke et al., 2024; Frankham et al., 2014; Traill et al., 2010), or the more recently updated 100/1000 rule (Frankham et al., 2014), under which an effective population size of 100 is assumed sufficient to prevent inbreeding depression and an effective population size of 1000 is sufficient to retain evolutionary potential in perpetuity.

A genetic analysis of ponds in Orange County also revealed low effective number of breeders and very low effective population sizes, among the lowest recorded for pond-breeding amphibians (Neal et al., 2020), indicating reduced genetic health and declining populations. Although limited, these surveys and studies suggest that western spadefoot population trends are declining.

VII. Factors Affecting the Ability of the Species to Survive

Amphibians are the most threatened vertebrate group with more than 40% of species threatened and approximately 200 species collapsing to or near extinction since the 1970s (Alroy, 2015; Luedtke et al., 2023; Stuart et al., 2004). According to researchers at the U.S. Geological Survey (USGS), amphibian populations in the U.S. are declining at an alarming rate of almost 4% per year (Grant et al., 2016).

It is increasingly clear that the western spadefoot is exceptionally vulnerable to extinction. They have a biphasic life history that requires connected sensitive aquatic and terrestrial habitats, much of which have already been lost or are increasingly vulnerable to development. Although information specific to western spadefoots is limited, multiple stressors including habitat loss and fragmentation, roads, urban development, agriculture development, extractive development, pollutants, off-road vehicles, disease, invasive species, and climate change threaten the western spadefoot's long-term survival. These impacts are not mutually exclusive and likely occur synergistically.

A. Habitat Loss and Fragmentation

Anthropogenic habitat loss and degradation are some of the biggest threats to amphibian populations worldwide, currently impacting 88% of threatened amphibian species (Baillie et al., 2004; Cushman, 2006; Sodhi et al., 2008; Willson & Dorcas, 2003). Human activities such as urban and agricultural development, deforestation, and draining of wetlands are primary mechanisms of habitat loss (Gallant et al., 2007; Grant et al., 2016).

Habitat loss and fragmentation due to urban and agricultural development are the primary threats to western spadefoots (Thomson et al., 2016). California has been the most populous state since 1970 and currently has over 39 million residents; in fact, one in eight residents of the United States lives in California. While growth has slowed in the last decade, California's population is expected to continue increasing, reaching 40 million by 2032 (California Department of Finance, Demographic Research Unit, 2025); other estimates are even higher (Landis & Reilly, 2004). Continuing poorly planned development, particularly in grasslands and shrublands, will further destroy spadefoot habitat and their ability to survive.

Western spadefoots most commonly occur in grasslands and shrublands. As poignantly stated by Augustine et al. (2021), "grasslands live in mortal fear of anthropogenic activities." Despite their importance in carbon cycling, water cycling, and ecosystem health, grasslands are historically undervalued, and have been systematically destroyed and disturbed for agriculture and urban development (Augustine et al., 2021; Buisson et al., 2022; Samson et al., 2004; Yap et al., 2023). By the 1990's, California had already lost 99% of its native grassland (Noss et al., 1995). Today's grasslands, which cover approximately 10-25% of the state, are largely dominated by non-native annual grasses, with few small patches of native plants remaining (Barry et al., 2020; Stromberg et al., 2007). As a result, grassland species including western spadefoots have experienced dramatic declines, especially in Southern California where large-scale sprawl development is rampant.

Shrublands are also highly threatened. For example, in California as much as 85% of coastal sage scrub has been lost to urbanization, development, and altered fire regimes, while low-elevation chaparral has been highly altered by invasive grasses which were introduced during California's early agricultural period and have thrived in the resulting disturbed landscapes (O'Leary, 1995; Riordan et al., 2018). These trends are unlikely to slow for decades, if not longer. Under a "business as usual" scenario, model projections for California indicate that developed lands could more than double by 2100, and grasslands and shrublands will continue to bear the greatest declines (Sleeter et al., 2017). Western spadefoots therefore will also continue to decline as they lose these remaining primary habitats to urbanization and agricultural development.

Vernal pools, which are essential components of western spadefoot habitat, have similarly declined due to agricultural and urban development. Over 90% of California's historic vernal pool systems have since been lost (National Fish and Wildlife Foundation, 2018). Starting in the 1800s, vernal pools were commonly drained and converted to either agricultural lands or water conveyance and storage, especially in the Central Valley. In the 1700's, approximately half of the Central Valley was characterized by vernal pool landscapes, covering an estimated 7 million acres (Holland, 2009). An estimated 75% to 87% of this habitat was lost by 1997 (Holland, 2009; National Fish and Wildlife Foundation, 2018; US Fish and Wildlife Service, 2005). More recent studies show that vernal pool habitat loss continues to occur; between 1997 and 2005, 13% of existing vernal pool habitat in the Central Valley disappeared (Holland, 2009), and another 9.3% of vernal pool habitat was lost between 2005 and 2018 (Witham, 2021). Since the Vernal Pool Recovery Plan was created in 2005 (US Fish and Wildlife Service, 2005), approximately 5,200 acres of vernal pool habitat in the Central Valley have been converted to other land uses per year (Witham, 2021).

In more recent years, urbanization has been a primary driver in the loss of vernal pools. Over half of California's vernal pools lost since 1994 were destroyed as a result of residential, commercial, or industrial projects (National Fish and Wildlife Foundation, 2018; US Fish and Wildlife Service, 2005). On the Central California coast, at least 90% of vernal pools have also been lost, and many of those that remain have been degraded (US Fish and Wildlife Service, 2005). In Southern California, vernal pools have declined drastically due to urbanization, with estimated losses of more than 95% (Bauder et al., 1998; Bauder & McMillan, 1998; Mattoni & Longcore, 1997; US Fish and Wildlife Service, 2005). The loss of vernal pool habitat throughout California has been a major factor in the decline of western spadefoots (Thomson et al., 2016). Continued destruction of vernal pools and associated grassland and shrubland habitats will drive the northern and southern spadefoot populations to extinction.

Fragmentation compounds the effects of direct habitat loss. As more suitable habitat is destroyed, populations become more and more isolated, reducing gene flow and increasing the likelihood of irreversible local extirpation and inbreeding depression. In a study of populations throughout California, Neal (2019) found extremely low numbers of effective breeders in all populations surveyed, indicating the risks of genetic isolation and inbreeding are likely widespread throughout the entire species.

Most western spadefoot habitat is not currently protected. Those areas that are protected are relatively small and therefore still highly susceptible to disturbance, destruction, and other threats (US Fish and Wildlife Service, 2005).

Urban Development

Urbanization threatens over one third of all amphibian species (Hamer & McDonnell, 2008). Urban development and expansion causes habitat loss and fragmentation, alters hydrology—including hydroperiod of ephemeral water bodies, which are extremely important for pond-breeding amphibians—modifies soils, and impacts ecological interactions (Hamer & McDonnell, 2008). Urban development can also enhance invasion by exotic species that directly harm or outcompete native amphibians (Riley et al., 2005). Residential and commercial developments also often cause devastating direct harm to native wildlife, including stress, injury, mortality and local extinctions (McKinney, 2002).

Urban development is one of the primary causes of population decline in western spadefoots (US Fish and Wildlife Service, 2005). Urbanization, especially in Southern California, has destroyed a significant amount of existing western spadefoot habitat. For example, from 1973 to 2000, developed land in California increased by approximately 38%, destroying western spadefoot habitat primarily within the Central Valley, chaparral and oak woodland habitats, and the Mojave Basin (Sleeter et al., 2011). In addition, edge effects of such development also impacts spadefoots, as researchers found that western spadefoots were less likely to be encountered as surrounding urban land use increased (Davidson et al., 2002). If the rate of urban development continues apace, we can expect the amount of developed land to increase by another 102% by the end of the century (Sleeter et al., 2017). Even if the rate of urbanization slows, developed area is still likely to increase by 40%-90% (Sleeter et al., 2017).

In Southern California, over 80% of the habitat that was historically occupied by western spadefoots has been lost to development (Jennings & Hayes, 1994). In Northern California, habitat loss has not been as extensively monitored; however, experts estimate that over 30% of suitable habitat had been destroyed by the 1990s (Jennings & Hayes, 1994). The percentage of lost habitat is likely much higher after three decades of continued development. Urbanization leads to direct destruction of habitat, degrades adjacent habitat, and fragments and potentially isolates existing populations. Due to their dependence on ephemeral aquatic breeding sites and vernal pool complexes, western spadefoots are particularly sensitive to habitat disruption. When breeding ponds are eliminated or cut off from upland habitat due to urban development, the populations that rely on them are doomed to extirpation.

Western spadefoots experience large annual fluctuations in population sizes due to annual variation in precipitation, which means they are particularly sensitive to habitat alterations that may interfere with recolonization and reestablishment after unsuccessful recruitment years (R. N. Fisher & Shaffer, 1996). Urbanization and development reduce regional and local connectivity across the landscape, isolating remaining populations. When isolated populations experience years with low reproductive success or become locally extirpated, recovery is difficult because they no longer have adjacent subpopulations to supply individuals and boost the population or facilitate re-establishment. Therefore, urban development that destroys remaining habitat and continues to fragment dwindling spadefoot populations leads to higher risks of local and regional extinction (Neal et al., 2020).

In Southern California, suitable western spadefoot habitat has become rare. In Los Angeles County, which historically harbored the species, suitable low elevation habitats have essentially all been developed into urban and suburban land uses. The majority of remaining habitat for western spadefoots in Southern California occurs in Riverside County, San Diego County, and Orange County, with small patches of suitable habitat occurring in Ventura and San Bernardino counties as well (Rose et al., 2022). Yet San Bernardino, Riverside, and San Diego are also under the greatest threat of additional habitat loss due to urbanization over the next 100 years, with Ventura County following not far behind (Landis & Reilly, 2004).

The threat from urban development in Southern California is evident by numerous recently approved sprawl development projects for mid- and high-income residential units that would destroy and fragment known western spadefoot populations and habitats (Table 1). For example, in 2019 Los Angeles County approved the Northlake development, which, according to the California Department of Fish and Wildlife ("CDFW"), would eliminate one of the region's last surviving populations of western spadefoot (B. J. Courtney, personal communication, June 15, 2017). In 2019 and 2020 San Diego County approved Otay Village 14 and Otay Village 13, respectively, which proposed large-scale planned developments on sites with thousands of acres hosting vernal pools, where some of the last-remaining, intact, high quality western spadefoot habitat in Southern California is located. And in 2025 the City of Santee in San Diego County approved the Fanita Ranch development, which would destroy and fragment large vernal pool complexes where a healthy spadefoot population is established. Otay Village 14 was blocked after successful litigation, and legal challenges are ongoing for Otay Village 13 and Fanita Ranch. Northlake was initially blocked after successful litigation, but the project proponents have indicated their intent to continue project development and revised environmental review documents have recently been released.

In Northern California, the City of Chico approved sprawl development in spadefoot habitat: the Stonegate Project and Valley's Edge Specific Plan in 2021 and 2022, respectively (Table 1). Despite legal challenges, development is still expected to occur at these sites. These projects would destroy and fragment vernal pool habitat that the USFWS identified as core areas necessary for the recovery of vernal pools and vernal pool species (US Fish and Wildlife Service, 2005).

Most western spadefoot habitat is not protected, and areas that are protected are relatively small (US Fish and Wildlife Service, 2005). Further habitat destruction, alteration, and fragmentation from urban development, particularly in Southern California, will drive the species towards extinction.

Table 1: Representative examples of development projects approved in the past few years in areas where western spadefoot occurs.

	Approved Sprawl Project (Year Approved)	Project Site (acres)	Impact to Western Spadefoot (Threatened Population)	Status
Los Angeles	Northlake (2019)	>1,300	Destroy habitat for one of the region's last-remaining populations. (Southern Population)	Project seeking reapproval after being blocked by successful litigation
San Diego	Otay Village 14 (2019)	>1,000	Destroy 12 of 16 known breeding pools; destroy, degrade, and fragment 57 potential breeding pools and associated upland habitat. (Southern Population)	Legal agreement led to permanent conservation of lands
San Diego	Otay Village 13 (2020)	>1,800	Destroy occupied spadefoot habitat adjacent to core habitat identified by the USFWS (2005). Limited analyses were conducted to determine the extent of impacts to spadefoot. (Southern Population)	Conditional settlement will allow more limited development
City of Santee	Fanita Ranch (2025)	>2,600	Destroy, degrade, and fragment vernal pool complexes with at least 42 occupied breeding pools, ~200 potential breeding pools, and ~400 acres or more of upland habitat. (Southern Population)	Lawsuit is ongoing
City of Chico	Stonegate Project (2021)	314	Destroy and degrade vernal pool and upland habitat within core habitat designated by the USFWS (2005). (Northern Population)	Court order invalidated biological opinion in 2025, but the project is expected to move forward
City of Chico	Valley's Edge Specific Plan (2022)	1,448	Destroy and degrade vernal pool and upland habitat within core habitat designated by the USFWS (2005). (Northern Population)	Project approvals rescinded after successful referendum; future development proposals for the site are likely

Roads

Human activity and development are accompanied by the construction of roads, which are detrimental to amphibian populations, including western spadefoots. Roads create physical barriers to amphibian movement that cause fragmentation and habitat isolation, reduce genetic diversity, introduce exotic species, and increase pollution in the form of road runoff (Holderegger & Di Giulio, 2010; Trombulak & Frissell, 2000). Vehicular traffic is a major source of amphibian mortality (Carr & Fahrig, 2001). Species that migrate to and from breeding sites and/or move slowly are particularly vulnerable (Carr & Fahrig, 2001; Hels & Buchwald, 2001). For example, thousands of Pacific newts (California newts [*Taricha torosa*] and rough-skinned newts [*Taricha granulosa*]) are killed on a 4-mile stretch of road every year during the rainy season, when newts migrate from upland burrows to breeding pools; scientists documented a 39.2% road mortality rate and predicted local extinction in less than 60 years for this population if connectivity is not improved (H.T. Harvey and Associates, 2021). Negative effects from roads have been detected even thousands of meters away from wetlands (Beebee, 2013; Hamer et al., 2021; Houlahan & Findlay, 2003).

Roads present a threat to western spadefoot survival and persistence. Western spadefoot mortality on roads appears to be common and widespread (US Fish and Wildlife Service, 2005). Western spadefoots are slow-moving, migrate en masse when rains begin, and exhibit strong natal philopatry and are therefore less likely to avoid roads that are located between their burrows and their breeding pools. In a study analyzing the impacts of roads on herpetofauna in California, western spadefoots were ranked as high risk, meaning roads are likely negatively impacting the species' range and conservation status (Brehme et al., 2018). As noted above, western spadefoot populations have high demographic instability, which makes them vulnerable to local extinction. Barriers like roads can reduce successful recruitment, and if local extirpation occurs for any reason, such barriers can prevent recolonization and re-establishment from neighboring populations. Roads increase fragmentation of regional western spadefoot populations and lead to further species decline.

Road construction can also result in direct mortality of western spadefoots. Western spadefoots are notoriously hard to detect, as they spend the vast majority of their lives in underground burrows. Any roads constructed or expanded in western spadefoot habitat may directly harm and kill individual western spadefoots. Roads also introduce pollutants in runoff, like carcinogenic polycyclic aromatic hydrocarbons, heavy metals, and tire dust from vehicles; herbicides and nutrients from nearby agriculture; or toxic chemicals used to clean or maintain roads, all of which may be harmful to western spadefoots (see **Pollutants** section below).

Roads also likely impact western spadefoot habitat connectivity. While studies on the impacts of roads on western spadefoots specifically have not been conducted, due to the likely high rates of road mortality, roads present significant barriers to movement, thereby fragmenting western spadefoot populations (Brehme et al., 2018; Thomson et al., 2016).

Agriculture

Agriculture is an extremely prominent component of California's land use. California has been the country's most agriculturally productive state for the past 50 years at least, producing over

13% of the entire nation's agricultural production value (United State Department of Agriculture & Farm Service Agency, 2011). Agriculture (including rangeland) accounts for approximately 42% of all land in California. Approximately 71% of agricultural land is concentrated in the Central Valley, while 21% is located in chaparral and oak woodland ecosystems (Sleeter et al., 2011). Demand for agriculture has led to significant land use changes over time, likely affecting resident western spadefoots with the draining of vernal pools and the conversion of grasslands and shrublands to farmland. The majority of current farmland lies on lands that were originally predominantly grasslands, composed of prairie and woodland plant communities (Larson-Praplan, 2014).

Much of the conversion from native landscapes to farmed and ranched land occurred in the late 19th and early 20th centuries. Loss of grassland continues to this day through conversion to agriculture and urbanization, as well as through conversion of former grazing rangeland to high-value crops like almonds and vineyards (Sleeter et al., 2011). Additionally, agricultural land use shifted over the late 21st century; while the overall amount of agricultural land use cover did not significantly change, conversion of agricultural lands to urban development often pushed agriculture to new areas, including chaparral, oak woodlands, and the foothills along the Central Valley (Sleeter et al., 2011)—all of which are prime habitats for western spadefoot.

Throughout much of lowland California—the western spadefoot's historical range—many native grasslands and shrublands have been converted to cropland, destroying western spadefoot habitat. Suitable vernal pool networks are concentrated on valley terraces along the edges of the Central Valley, and many have been lost or fragmented due to agricultural expansion and conversion (Jennings & Hayes, 1994; US Fish and Wildlife Service, 2005). Researchers found that agricultural land use negatively affected western spadefoots within a 5-km radius (Davidson et al., 2002), which suggests that ongoing land use changes are destroying and degrading remaining suitable spadefoot habitat. In addition, agriculture also introduces pesticides, herbicides, and fertilizers into spadefoot habitat, which have been found to have negative effects on various amphibian species, including reduced growth, immunosuppression, and malformations (e.g., T. B. Hayes et al., 2006).

Agriculture has also led to the introduction of invasive plants, especially grasses. Non-native annual grasslands increased by over 8,600% in the last century (Noss et al., 1995). Invasive grasses alter the community structure and hydrology of vernal pools, such that the few remaining vernal pools that exist in agricultural areas may not be as ecologically or hydrologically stable as they were historically. Invasive grasses outcompete and shade out numerous native plant species, reducing diversity (Hamilton, 2008; US Fish and Wildlife Service, 2005). These grasses also create a layer of thatch around the pool that alters habitat and hydrology, creating a feedback loop in which pool hydrology and ecology is permanently shifted to an alternative state (Faist & Beals, 2017). The potential impacts of altered vernal pool dynamics present additional challenges to western spadefoots in these environments.

Ranching is a significant component of California's agricultural industry. Cattle and other livestock have been grazed throughout California's grasslands and shrublands since the mid-eighteenth century. While cattle grazing may not be as directly destructive to western spadefoot habitat as conversion to cropland, grazing has nonetheless led to the degradation of many native grasslands and shrublands through the introduction of invasive plant species, which alter the community composition, ecological interactions, and ecosystem functioning of native habitats (Koteen et al., 2011). Although there are no studies on the impact of the altered composition of California grasslands on western spadefoots, research on other amphibians indicates that degraded grasslands and shrublands likely play a role in the species' decline. For example, a study of American toads (*Anaxyrus americanus*) in Georgia found that toads living in habitats dominated by an invasive grass experienced significantly reduced survival rates, likely due to altered predator-prey dynamics (DeVore & Maerz, 2014).

Even when agricultural practices maintain existing vernal pools, as is sometimes the case with ranching, pool hydrology and ecology are often altered. Timing, frequency, and duration of pool inundation are critical to a pool's ecological function and to survival of vernal pool species. The creation of reservoirs, stock ponds, and conveyance systems in agricultural lands can lead to dewatering of vernal pools. By contrast, increasing water flow and longer inundation can allow harmful invasive species like American bullfrogs to more readily colonize pools, which can drive native species out through competition and predation. Runoff from agricultural practices can also harm vernal pools by increasing contaminant loads and contributing to erosion and siltation, which make pools uninhabitable for numerous species, including western spadefoot larvae (US Fish and Wildlife Service, 2005). Pool ecology is also impacted by livestock. Without livestock, vernal pools may experience a reduced capacity to take up nitrogen, leading to increased nitrate levels, yet with livestock, vernal pools may be subject to ammonia and nitrites from animal waste (Huntsinger et al., 2007). Both scenarios disrupt vernal pool ecosystem production and functioning and may harm sensitive vernal pool species. including western spadefoots.

Livestock grazing also has several direct negative impacts on vernal pools, including trampling and nutrient input (Robins & Vollmar, 2002). Livestock may directly harm and kill adult and juvenile western spadefoots by trampling them. Similarly, livestock may crush or unintentionally consume eggs while using ponds. Livestock may also deplete ponds earlier than they would have dried naturally, which can prevent successful metamorphosis (US Fish and Wildlife Service, 2005). In addition, increased nutrient loads from livestock manure and urine in overgrazed areas can lead to vernal pool eutrophication, which harms native fauna (Robins & Vollmar, 2002).

Direct and indirect effects of agricultural development and practice have been major contributing factors to the decline of western spadefoots and continue to be a prominent threat to the long-term survival of the species.

However, sustainable grazing practices that carefully regulate the timing and intensity of grazing may support western spadefoots. Researchers and land managers have observed that in some cases grazing can have a positive impact on western spadefoot habitats. For example, cattle and sheep grazing decreases invasive plant cover in uplands and around vernal pools, which helps to protect vernal pool hydroperiods (Robins & Vollmar, 2002). One study of western spadefoots in grazed and non-grazed areas found that cattle grazing activity reduced the abundance of invasive grasses surrounding vernal pools and therefore reduced evapotranspiration rates, giving spadefoot tadpoles more time to develop and metamorphose (Marty, 2004). Some studies also show that trampling, while potentially harmful to western spadefoots and other sensitive species, may also increase soil compaction and decrease pool infiltration rates, maintaining long hydroperiods and supporting overall vernal pool ecosystem functioning (Robins & Vollmar, 2002), although other studies show no such effect (Michaels et al., 2022). It is important to note, however, that studies of grazing impacts on vernal pools are generally conducted on land that has been ranched for 100+ years, not undisturbed areas. Many remaining vernal pools are located on private lands with long histories of grazing (Michaels et al., 2022). Considering that much of the Central Valley and other grasslands and shrublands throughout California have been subjected to grazing for centuries, it is important to acknowledge that sustainable grazing practices, if done properly, can be beneficial in the current land-use system. However, researchers caution that the impacts of grazing are complex and not well understood, and inappropriate grazing can harm vernal pools; such efforts should be done carefully, with consideration of local site-specific biotic communities and climatic conditions, and with rigorous monitoring and management (Robins & Vollmar, 2002; US Fish and Wildlife Service, 2005).

Extractive Development

Extractive development like mining and drilling for oil and gas has negative impacts on western spadefoots. Although the full extent of impacts of such development has not been determined, mining for gravel and clay has destroyed and degraded vernal pool habitat in many areas, disrupting soil formations, hydrology, and seed banks and introducing harmful invasive species (US Fish and Wildlife Service, 2005).

In addition to directly killing individuals and destroying spadefoot habitat, activities that disturb the soil and produce low frequency noise and vibration, such as mining, grading for development, and seismic exploration for natural gas may negatively impact western spadefoots (US Fish and Wildlife Service, 2005). Dimmitt & Ruibal, (1980) found that western spadefoots were extremely sensitive to these types of stimuli and would emerge early from their burrows in response to these ground-moving activities, disrupting their natural dormancy patterns. Such emergence at inappropriate times could lead to non-lethal detrimental effects that may result in reduced fitness or mortality (US Fish and Wildlife Service, 2005).

Off-road Vehicles

Motorized recreation, including off-road vehicles ("ORVs"), is a significant threat to wildlife. ORVs include dirt bikes, all-terrain vehicles, 4x4s, and snowmobiles, all of which are made to be ridden in backcountry wilderness areas, creating significant and long-lasting impacts (Shore, 2001). ORVs alter and degrade habitats, cause disturbances, and lead to direct death of many wildlife species (Kassar, 2009). ORVs have numerous direct and indirect impacts on amphibians in particular. Amphibians are susceptible to direct mortality from ORVs, especially during dispersal and migration. However, habitat degradation likely causes a greater impact. ORV use can alter hydrological patterns, potentially affecting the ecology of an entire area, as well as increasing erosion and sedimentation, thereby degrading aquatic habitat for larval and adult amphibians (Kassar, 2009). For western spadefoots, which have already experienced significant habitat destruction, preservation of intact and functional habitat is essential. Highly used trails can also functionally become roads, creating barriers to amphibian movement and fragmenting populations, increasing the likelihood of local extirpations (see *Roads* section above).

Public lands in California experience more ORV use than any other state. In Southern California's wildland-urban interface, ORV use is common in the mountains and wildlands that surround cities (Shore, 2001). As described above, western spadefoots have been largely extirpated from urban and suburban areas and remain in fragmented surrounding lands that may also be used for ORV recreation. Systematic studies of ORV impacts to western spadefoots have not been conducted, but ORV use in spadefoot habitat likely poses a threat to remnant populations throughout California. In a herpetological survey of the Carnegie State Vehicular Recreation Area near Tracy, CA, researchers observed that metamorphic western spadefoots often sheltered in desiccation cracks in and near dried breeding pools, generally remaining 5-7cm below the surface of the ground (Alvarez & Kerss, 2023). This microhabitat use makes metamorphs like western spadefoot extremely susceptible to ORVs. The researchers note that entire cohorts may be killed or injured by vehicles driving over recently dried pools, while habitat may be rendered completely unusable for this critical life history phase (Alvarez & Kerss, 2023). Thus, ORV use in spadefoot habitat on both public and private lands may pose a threat to remnant populations.

B. Pollutants

Amphibians are highly susceptible to pollutants and chemical contaminants from agricultural practices, vehicles, road runoff, industrial facilities, and more, which can have direct and indirect effects on amphibian populations (Blaustein et al., 2003). The complex life cycles of many amphibians, including western spadefoots, leave numerous opportunities and routes for exposure to such chemicals. For example, many amphibians are exposed to prolonged periods of low concentrations of pesticide mixtures, which can have dramatic adverse effects on amphibian development, growth and survivorship (T. B. Hayes et al., 2006), although effects vary across species. For example, Davidson (2004) found a strong association between amphibian declines and total upwind pesticide use for four frog species, while the co-occurring *Bufo* species was unaffected. In addition, atrazine, the most commonly-used herbicide in the U.S., has endocrine-disrupting effects on African clawed frogs (*Xenopus laevis*), causing hermaphroditism and demasculinized larynges of exposed males (T. B. Hayes et al., 2002).

Western spadefoots are exposed to a variety of toxins throughout their range, but their sensitivity to pesticides, heavy metals, air pollutants, and other contaminants is not well-studied (US Fish and Wildlife Service, 2005). Every year, millions of pounds of chemicals—including fertilizers, insecticides, herbicides, and fungicides—are applied on crops, forests, roads, and urban landscapes, some of which are extremely toxic to amphibians (US Fish and Wildlife Service, 2005). Additionally, industrial facilities and motor vehicles regularly release contaminants like carcinogenic polycyclic aromatic hydrocarbons, heavy metals, tire dust and other toxic chemicals into the environment. Vehicles powered by internal combustion engines emit nitrogen oxides (NO_x), while catalytic converters designed to reduce NO_x emissions emit ammonia gas (NH_3). Thus, traffic on roads results in excess nitrogen spread throughout the region, which promotes the growth of invasive, non-native grasses and disrupts ecosystem health and function (Weiss & Longcore, 2020). Nitrogen deposition has been linked to declines in many sensitive vertebrate, invertebrate, and plant species and could have negative impacts to western spadefoots (Hernández et al., 2016). While direct studies of the impacts of contaminants on western spadefoots have not been conducted, such contaminants may reduce their ability to survive by reducing fitness, directly killing them, and/or damaging and degrading their breeding pools and upland habitat (US Fish and Wildlife Service, 2005).

C. Disease

Emerging infectious diseases have been implicated as a factor in amphibian declines worldwide (Blaustein et al., 2018; Daszak et al., 1999, 2003; Storfer et al., 2007). Chytridiomycosis is an emerging infectious disease primarily caused by the fungal pathogen *Batrachochytrium dendrobatidis* (*Bd*). This pathogen has significantly affected global amphibian biodiversity, infecting over 500 species (Olson et al., 2013) and causing declines and extinctions in hundreds of species since the 1970s (Alroy, 2015; M. C. Fisher & Garner, 2020; Skerratt et al., 2007). Global trade likely played a role in the current *Bd* pandemic by spreading non-native, infected animals worldwide and exposing naïve populations to *Bd* (M. C. Fisher & Garner, 2007; Liu et al., 2013; Schloegel et al., 2012).

Bd attacks the keratin and skin of amphibians (Berger et al., 1998), often causing a thickening of the outer layer and disruption of necessary physiological processes like fluid and electrolyte balance (Voyles et al., 2012), ultimately leading to morbidity and mortality (Marantelli et al., 2004; Rosenblum et al., 2010). In tadpoles infected with chytrid fungus, jaw sheaths and tooth rows are abnormally formed or lack pigment, and this type of deformity likely inhibits tadpole foraging ability (Fellers et al., 2001). Adult anurans infected with chytrid exhibit symptoms such as lethargy and reluctance to flee, skin abnormalities, loss of righting reflex, extended back legs, and eventually death by cardiac arrest (Berger et al., 1998; Fellers et al., 2001).

Bd has been documented in California since at least the 1930s, (Vredenburg et al., 2019). *Bd* has multiple strains (some more virulent than others) and not all species react similarly to infection; some species are infected and do not exhibit any symptoms but can transmit disease, others are infected and show signs of disease but then recover, and others are infected, show signs of disease, and die. In California, *Bd* is responsible for the drastic population declines of numerous native species, including the federally endangered mountain yellow-legged frog (*Rana muscosa*) and the Sierra Nevada yellow-legged frog (*Rana sierrae*). (Padgett-Flohr & Hopkins II, 2009; Reeder et al., 2012; Sette et al., 2015; Yap et al., 2018). There is no known available data regarding *Bd* infection in western spadefoots; however, much of the species' range is in or near areas where *Bd* has been detected and within areas identified as having moderate or high suitability for *Bd* (Yap et al., 2018). In addition, western spadefoots may come into contact with co-occurring species that are known to be *Bd* reservoirs and potential "supershedders" like the native Pacific chorus frogs (*Pseudacris regilla*) (Reeder et al., 2012) and non-native American bullfrogs (*Rana catesbeiana*) (Yap et al., 2018). Furthermore, *Bd* was detected on the closely related Mexican spadefoot (*Spea multiplicata*) (Christman & Jennings, 2018), which suggests that western spadefoots may be susceptible to infection. *Bd* could be a significant threat to small, fragmented remnant western spadefoot populations. In addition, although it has not yet been detected in the U.S., a second chytrid fungal pathogen called *Batrachochytrium salamandrivorans* (*Bsal*) poses a potential additional threat to western spadefoots should it be introduced (Martel et al., 2014; Stegen et al., 2017; Yap et al., 2015, 2017).

Ranaviruses (genus *Ranavirus*, Family Iridoviridae) are also important amphibian pathogens, infecting at least 105 species across 25 countries (Duffus et al., 2015). They exhibit high virulence, lack of host specificity, and wide global distribution (Gray & Chinchar, 2015; S. A. Smith et al., 2016). Ranavirus infections usually cause mortality in larvae and metamorphs, and adults of some species may also be impacted. Mortality is often sudden, with significant die-offs (up to 90% of a local population) occurring within several days (J. L. Brunner et al., 2015). In the United States, ranaviruses have been associated with mass mortality in the federally listed Sonoran tiger salamander (*Ambystoma tigrinum stebbinsi*) and several other salamanders and frog species (Davis & Kerby, 2016; Gray et al., 2009; Green et al., 2002). Ranaviruses can also cause sublethal effects, including reduced rates of growth and development (Echaubard et al., 2010). Similar to the case of chytridiomycosis, the prevalence and impact of ranaviruses on western spadefoots have not yet been studied. However, ranaviruses have been linked with larval mass mortality events in related species, including eastern spadefoots (*Scaphiopus holbrookii*) in Illinois (Kirschman et al., 2017) and Delaware (S. A. Smith et al., 2016) as well as plains spadefoots (*Spea bombifrons*) in Nebraska (Davis & Kerby, 2016). Therefore, ranaviruses pose a potential threat to dwindling western spadefoot populations as well.

D. Invasive species

Invasive species can negatively impact amphibian populations through competition, predation, hybridization, or as carriers of infectious disease (Collins & Storfer, 2003). They are spread both intentionally and accidentally for sport, biocontrol, as a food source, or simply because they are unwanted pets (Kats & Ferrer, 2003). Introduced predators such as mosquitofish, bullfrogs and crayfish have played a major role in amphibian population declines broadly (Kats & Ferrer, 2003), and are also a direct threat to western spadefoots (Thomson et al., 2016). Eggs and tadpoles are particularly susceptible to predation, although bullfrogs have also been shown to prey on adults. Fisher & Shaffer, (1996) observed that invasive fish and bullfrogs did not generally co-occur with western spadefoots in the Central Valley. Instead, these invasive predators occupied lower-elevation sites and western spadefoots occupied higher-elevation sites. The authors hypothesize that this pattern suggests invasive predators may be a contributing factor in the decline of low-elevation western spadefoots.

Non-native mosquitofish (*Gambusia affinis* and *G. holbrooki*) historically were commonly stocked into ephemeral pools and permanent water bodies in California and throughout the world as a biocontrol agent to reduce mosquito populations and manage vector-borne diseases (G. H. Pyke, 2008). Introduced mosquitofish can prey on tadpoles and may negatively impact native amphibian populations with which they co-occur (Kats & Ferrer, 2003). While there are no published studies determining whether mosquitofish prey on western spadefoot tadpoles, it remains a possibility (US Fish and Wildlife Service, 2005). Additionally, mosquitofish may act as a reservoir for pathogens that can infect western spadefoots, including some ranaviruses (Brenes et al., 2014).

Non-native crayfish (order Decapoda) have similarly been introduced to waters throughout California in the late 1800s and early 1900s. Crayfish are aquatic predators, and have been shown to prey on eggs and larval amphibians (Axelsson et al., 1997). They may inhibit successful recruitment of larval western spadefoots in pools where they co-occur (Jennings & Hayes, 1994; US Fish and Wildlife Service, 2005).

American bullfrogs may also have a negative effect on western spadefoot populations. Invasive American bullfrogs are present throughout California,[2] co-occurring with western spadefoots throughout their range. American bullfrogs often compete with native amphibians for prey and will also consume native amphibians directly. They commonly eat tadpoles of other anurans, and may consume western spadefoot tadpoles as well (US Fish and Wildlife Service, 2005). In Arizona, where juvenile bullfrogs and developing spadefoots co-occur, spadefoot larvae and metamorphs make up a large proportion of bullfrog diets (Morey & Guinn, 1992). They have even been documented to consume adult western spadefoots (M. P. Hayes & Warner, 1985). American bullfrogs also act as reservoirs for amphibian pathogens, including Bd and ranaviruses, and can spread infections to other co-occurring amphibian species (A. J. Adams et al., 2017; J. Brunner et al., 2019; Greenspan et al., 2012). Bullfrogs rely on permanent bodies of water to survive, so while they may not commonly co-occur with western spadefoots in vernal pools or other temporary water bodies, they do pose a threat to spadefoots that breed in or near perennial waters (Stebbins & McGinnis, 2012).

E. Climate Change

Climate change is worsening ecosystem stress and species extinction risk (Trisos et al., 2020). Increasing variability and extremes in temperature, wind, and precipitation are all products of a warming climate, leaving species struggling to adapt. As a result, species' genes are changing, physiological and physical features such as body size are changing, ranges are shifting as species try to maintain a suitable climate space, and numerous species are expressing new breeding and migration behaviors (Scheffers et al., 2016). Climate-related local extinctions have already occurred in hundreds of plant and animal species (Wiens, 2016). If climate change goes unabated, scientists predict that more than one-third of all plant and animal species could become extinct in the next 50 years (Román-Palacios & Wiens, 2020).

Climate change is one of the greatest threats to amphibians worldwide (Luedtke et al., 2023). Changes in temperature and precipitation may impact reproduction, development, feeding, dispersal, ecological interactions, and immune function (Blaustein et al., 2010; Corn, 2005). Extreme climate events are increasing in frequency and magnitude (IPCC, 2023) and may push species past their temperature or desiccation thresholds and lead to direct mortality or sublethal effects like reduced growth or impaired immune function (Blaustein et al., 2010; Corn, 2005).

Human-induced global warming has led to higher temperatures and more frequent and extreme weather events (IPCC, 2023). Temperatures are predicted to continue to increase over the coming decades throughout the western spadefoot's range, leading to warmer winters and summers and earlier spring warming (Cayan et al., 2008; Thomson et al., 2016), increasing drought risk (Diffenbaugh et al., 2015) and shifting precipitation regimes. Some studies predict that rainfall may decrease by up to 30% (Snyder and Sloan 2005, PRBO 2011).

Such climatic shifts threaten spadefoot habitat and their survival. Western spadefoots, which rely on ephemeral ponds in semi-arid ecosystems, will likely experience habitat loss or degradation due to climate impacts, which in turn could lead to population declines and extirpations (Blaustein et al., 2010). Although western spadefoots are adapted to withstand occasional

[2] https://wildlife.ca.gov/Conservation/Invasives/Species/Bullfrog

drought and associated unsuccessful recruitment years, prolonged drought and reduced precipitation would heighten the likelihood of local extinction (Baumberger et al., 2019) and further decline of the species. Their breeding is triggered by rainfall and requires standing water (e.g. vernal pools) that lasts long enough for tadpoles to complete metamorphosis and for juveniles to disperse out of the breeding pools, but not so long that non-native bullfrogs or fish can establish and prey upon the larvae. Montrone et al., (2019) found that hydroperiods in vernal pools in Northern California are expected to decrease with climate change, which suggests that some northern populations are increasingly vulnerable to impacts of climate change.

By contrast, C. R. Pyke, (2005) determined that inundation period may increase in the Central Valley. While at first glance this might seem beneficial for vernal pool species like western spadefoots, such a shift in hydroperiod could be ecologically disruptive. Like many native amphibian species, western spadefoots are adapted to successfully reproduce in seasonally drying wetlands. Longer inundation periods allow for more extensive colonization by aquatic predators, including invasive American bullfrogs (C. R. Pyke, 2005), that prey upon and outcompete native species like western spadefoot (C. R. Pyke, 2005). Such hydrological changes could therefore result in decreased larval survival and eventual declines or extirpations in northern populations.

Changes in the timing and amount of precipitation and increases in extreme weather events may also lead to a mismatch between western spadefoot emergence and the occurrence of suitable breeding habitat, decreasing reproductive success. For example, Ervin et al., (2005) observed abnormally early western spadefoot breeding after a prolonged dry period, which was largely unsuccessful due to breeding pools drying up before larvae had metamorphosed. The adult spadefoots had emerged to breed before adequate breeding pools were available and instead used pools that quickly dried up. This illustrates the potential mismatch between environmental cues and breeding behavior that western spadefoots may experience with climate change (Thomson et al., 2016).

Increased drought risk due to climate change will also continue to impact western spadefoots. Although many native amphibian species, including western spadefoot, are adapted to ephemeral wetlands and are historically resilient to drought, severe and extended drought could cause steep declines very quickly. In drought years, when ephemeral vernal pools are not formed and permanent water bodies dry up, western spadefoots are unable to breed. With prolonged drought, individuals may not have an opportunity to reproduce in their lifespan and populations may not be able to breed for multiple years, potentially heightening the likelihood of local extinction (Baumberger et al., 2019). Such additional demographic stress on the generally small and fragmented western spadefoot populations makes them vulnerable to extirpation from other unpredictable environmental conditions, like disease outbreaks or landslides after wildfire, ultimately leading to further decline of the species. Some species that rely on ephemeral pools for breeding are already being impacted by increasing drought. For example, researchers documented a 20% reduction to mean body condition of California newts (*Taricha torosa*) in Southern California coinciding with warmer and drier conditions (Bucciarelli et al., 2020).

Although the full extent of impacts of climate change is uncertain, it is clear that shifts in precipitation regimes and potential changes to vernal pool hydrology and ecology due to climate change threaten remaining western spadefoot populations.

F. Synergistic Effects

Western spadefoots face numerous diverse threats. Many of these threats interact synergistically, presenting additional and sometimes heightened challenges to western spadefoot survival. Any combination of habitat loss, reduced connectivity, pollutants, disease, and climate change heightens the risk of extinction for remnant spadefoot populations.

Habitat loss interacts with climate change to impact western spadefoots. As described above, climate change is likely to have myriad impacts on breeding and upland habitat, with shifts in hydroperiod and increased temperatures expected to become more extreme (Montrone et al., 2019; C. R. Pyke, 2004; Thomson et al., 2016). Western spadefoot populations are often able to persist in the face of environmental variability due to metapopulation connectivity; when one population experiences declines, migrants from surrounding populations can bolster the population and prevent local extirpation. As climate change impacts become more severe and extreme weather events continue to occur, such metapopulation dynamics are likely to be increasingly important for species persistence. However, development will continue to destroy such connectivity. As vernal pools and upland habitats continue to be destroyed by urban and agricultural development, fragmented populations of western spadefoot may not be able to survive in the few and/or isolated vernal pool complexes that remain. Due to reduced connectivity between suitable habitats, populations that are unable to successfully breed due to climate impacts on breeding pools may decline and disappear and will be unable to be re-established by neighboring populations, leading to further permanent species decline.

Disease likely works synergistically with other threats to amphibians as well (Collins & Storfer, 2003; Kiesecker et al., 2001). For example, sublethal environmental stressors such as those present in developed areas (e.g. increased pollution and runoff, invasive predators) may suppress immune systems (Carey, 1993) and allow disease agents to kill weakened animals (Alford & Richards, 1999). Western spadefoots are likely experiencing numerous physiological stressors, from shifts in temperature with climate change, increased contamination from agricultural and urban runoff, and increased competition and predation from invasive species. Together, such stressors may impair immune function, leading western spadefoots to be more susceptible to emerging infectious diseases like ranavirus and chytridiomycosis. As described above, studies of disease in western spadefoot are scant, and more research is needed to determine whether pathogens and related compounding factors may be impacting western spadefoots.

In addition, development and human activity may increase disease risk as well. Chytrid prevalence has been shown to be higher near urban areas in California, suggesting a synergistic effect between the urban environment and disease risk (M. J. Adams et al., 2010). Ranavirus may also be spread by human activity. An evolutionary analysis of ranaviruses in salamanders indicated recent spread through range expansion and long-distance colonization, patterns consistent with human-facilitated viral movement (Jancovich et al., 2004). As discussed above (see **Range and Distribution** section above), western spadefoots occur in areas with high human impacts, especially in the southern portion of their range. As such, they may be vulnerable to these pathogens that are often spread via human activity.

VIII. Degree and Immediacy of Threat

As described in the previous sections, the threats to western spadefoots are both immediate and ongoing. Populations in Southern California meet the definition of endangered; they are in serious danger of becoming extinct throughout all or a significant portion of their range (Fish & Game Code § 2062). Northern populations meet the definition of "threatened;" they are likely to become an endangered species in the foreseeable future in the absence of special protection and management efforts (Fish & Game Code § 2067). As development and climate change continue to alter, destroy, and degrade suitable habitats and further fragment already-precarious populations, western spadefoots face surmounting threats to their survival.

Urbanization and agricultural development have destroyed nearly all suitable western spadefoot habitat in Southern California, as well as significant amounts of habitat in Central and Northern California. Over 90% of California's historic vernal pool systems have been lost (National Fish and Wildlife Foundation, 2018). From 1973 to 2000, developed land in California increased by approximately 38%, including multiple suitable habitat areas within the western spadefoot's range (e.g. the Central Valley, vernal pools with upland grassland, chaparral, and oak woodland habitats (Sleeter et al., 2011)). If the rate of urban development continues apace, the amount of developed land is likely to increase by another 102% by the end of the century, while important spadefoot habitats like grasslands and shrublands may decline up to 10% and 4% respectively (Sleeter et al., 2017).

In Southern California, the majority of remaining western spadefoot habitat occurs in counties with the highest likelihood of continued urbanization over the next century (Riverside, San Diego, Ventura, San Bernadino) (Landis & Reilly, 2004; Rose et al., 2022). This is exemplified by multiple large development approvals in the little remaining spadefoot habitat in LA and San Diego counties (see Table 1). Without additional means to provide critical habitat protection, western spadefoots may disappear from Southern California.

In Central and Northern California, agricultural development, including shifting from rangeland to irrigated crops, as well as conversion of agricultural lands to urban and industrial development will continue into the future (Sleeter et al., 2011), destroying, fragmenting, and degrading remaining western spadefoot habitat. If land-use planning continues business-as-usual, such habitat loss and degradation combined with the many other threats described above, including climate change, pollution, emerging infectious disease, off-road vehicles, and invasive species, will drive the northern western spadefoot populations towards extinction.

As noted below (see **Protected Lands** section), most current western spadefoot habitat is not protected. Without immediate action to protect remaining habitats, restore historical habitats, and enhance population connectivity, both the southern and northern populations of western spadefoots may disappear within our lifetimes.

IX. The Inadequacy of Existing Regulatory Mechanisms and Impact of Existing Management Efforts

A. Federal Regulatory Mechanisms

<u>National Environmental Policy Act</u>

The National Environmental Policy Act ("NEPA") NEPA is "our basic national charter for protection of the environment" (*Conservation Cong. v. Finley*, 774 F.3d 611, 615 (9th Cir. 2014).) NEPA is designed to ensure that environmental information is available to the public before decisions are made or actions taken and to help public officials make decisions based on an understanding of the environmental consequences. Historically, pursuant to NEPA federal agencies prepared an environmental impact statement ("EIS") if it was known that a major federal action would significantly affect the environment, or an environmental assessment ("EA") if the extent of effects was unknown. (*See* 42 U.S.C § 4332; *see also* former 40 C.F.R. §§ d1502.3 & 1508.9 [repealed].) NEPA further requires federal agencies to analyze reasonable alternatives to the proposed project. (42 U.S.C. § 4332(C)(iii); *see also* former 40 C.F.R. § 1502.14(a)-(c) [repealed].) NEPA requires the federal agency performing environmental review to consider the degree of adverse effect on a species or its critical habitat designated pursuant to the Federal Endangered Species Act. (*Conservation Cong. v. United States Forest Serv.* (E.D.Cal. 2017) 235 F.Supp.3d 1189, 1207.) Courts have interpreted NEPA as primarily a "procedural" statute. While NEPA requires federal agencies to consider detailed information regarding a project's environmental effects, "NEPA itself does not mandate particular results." (*Winter v. NRDC, Inc.* (2008) 555 U.S. 7, 23.)

Recent legal developments have drastically changed the landscape of how federal agencies can be expected to interpret, apply, and implement NEPA going forward. Historically, key components of federal agencies' environmental review were guided by regulations first established in 1978 by the federal Council on Environmental Quality ("CEQ"). Among other things, the regulations directed the preparation of EAs, required analysis of a no-action alternative, and required analysis of mitigation measures. In 2024, the D.C. Circuit Court of Appeals held that the CEQ lacked authority to issue binding NEPA regulations. (*Marin Audubon Soc'y v. FAA* (D.C.Cir. 2024) 121 F.4th 902.) In February 2025, the CEQ repealed the NEPA regulations that had been in place for more than forty years. (*See* 90 FR 10610, 10616, Feb. 25, 2025.) Federal permitting agencies will now be responsible for developing their own NEPA processes consistent with the law and will be doing so amid ongoing staff shortages and workforce restructuring.

However, even under the former CEQ regulations, agencies did not interpret NEPA as requiring analysis of impacts to species populations not listed as threatened or endangered, such as the western spadefoot. Furthermore, NEPA applies only to "major federal actions." Many of the land use decisions in California that negatively affect western spadefoots are the purview of local governments and do not constitute a "major federal action" requiring environmental review under NEPA. This will be especially true given the recently reduced permitting authority of the U.S. Army Corps of Engineers under the Clean Water Act, described below.

Because of its limited application, NEPA is insufficient to protect the western spadefoot.

Clean Water Act

Under Section 404 of the Clean Water Act, 33 U.S.C. § 1251 et seq. ("CWA"), the discharge of pollutants, including dredged or fill material, into "waters of the United States" is prohibited absent a permit from the U.S. Army Corps of Engineers. Theoretically the CWA should provide protections for stream and wetland habitats that are used by western spadefoots. However, the implementation of the CWA regulatory scheme and the Section 404 program in particular have fallen far short of Congress's intent to protect wetlands and water quality. A National Research Council report entitled "Compensating for Wetland Losses Under the Clean Water Act" concluded that the goal of no net loss of wetlands has not been achieved through the Army Corps regulatory program, and that applicants often do not follow through on promised mitigation packages (National Research Council, 2001). These failures of the Army Corps regulatory scheme are due in part because the Corps' implementation of the individual permitting process has allowed too much development while requiring too little avoidance and mitigation. Also, in permitting projects, the Army Corps often takes a very limited view of a project, looking only at impacts in the project footprint.

Even when vernal pools were included within the jurisdictional waters of the U.S., the CWA did not prevent them from being degraded or destroyed. For example, in a study of Placer County vernal pools subject to CWA Section 404 permits, researchers found that most permit files were incomplete and missing key documents that included important information regarding mitigation; in many cases, it was unclear whether mitigation measures were even implemented (AECOM et al., 2009). A larger analysis of Section 404 permits issued from 2000-2006 for projects that impacted vernal pool habitats in the Central Valley found that on the whole mitigation is generally replacing impacted acreage, but information on mitigation methods is lacking for many permits, and data on ongoing management (including required annual monitoring reports) were almost entirely lacking (AECOM & Vollmar Consulting, 2009). However, in Placer County, CWA permitting proved insufficient to prevent significant impacts, as the county experienced a net loss in vernal pool habitat over the period studied (AECOM & Vollmar Consulting, 2009).

Moreover, due to recent legal developments, the vast majority of breeding habitat used by western spadefoots is no longer subject to the Clean Water Act, as vernal pools are no longer considered waters of the United States. The Clean Water Act's application is limited to waters of the United States ("WOTUS"). In 2006 the United States Supreme Court established that WOTUS include navigable waters, interstate waters, territorial seas, impoundments, tributaries, wetlands adjacent to jurisdictional waters, and wetlands "with a significant nexus" to a traditionally navigable water (*Rapanos v. U.S.* (2006) 547 U.S. 715, 742). Vernal pools with a "significant nexus" to a relatively permanent water were therefore included in the definition of WOTUS. Not all vernal pools meet this standard, so whether a vernal pool was covered by the CWA depended on local conditions. In 2023, the United States Supreme Court narrowed the definition of WOTUS, defining them as relatively permanent bodies of water connected to traditional navigable waters, interstate waters, territorial seas, and wetlands with a continuous surface connection to these jurisdictional waters. (*Sackett v. EPA* (2023) 598 U.S. 651.) As a result, vernal pools, isolated wetlands and temporary wetlands and streams are now excluded from the definition of WOTUS. Thus, the vast majority of western spadefoot habitat is not covered under the Clean Water Act and is not protected by the Act's provisions.

Endangered Species Act

The Endangered Species Act ("ESA"), enacted in 1973, establishes protections for fish, wildlife, and plants that are threatened or endangered. When a species is listed, the ESA requires the listing agency to designate critical habitat and draft a recovery plan. The law also includes various requirements to prohibit further harm to listed species.

Under ESA section 7(a)(2), federal agencies must consult with the U.S. Fish and Wildlife Service prior to authorizing, funding, or carrying out activities that may affect listed species. (16 U.S.C. § 1536.) The USFWS then determines if the action will jeopardize a species; a jeopardy determination is made for an action that is reasonably expected, directly or indirectly, to appreciably reduce the likelihood of both the survival and recovery of a listed species in the wild by reducing its reproduction, numbers, or distribution; a non-jeopardy opinion may include reasonable and prudent measures that minimize the amount or extent of incidental take of listed species associated with a federal action (50 CFR 402.02). The FWS must also determine whether the action will result in the destruction or adverse modification of designated critical habitat.

ESA section 9 separately prohibits the taking of any federally listed endangered or threatened species. (16 U.S.C. § 1538.) Section 3(18) of the Act defines "take" to mean "to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, or collect, or to attempt to engage in any such conduct." (16 U.S.C. § 1532(19).) The ESA provides for civil and criminal penalties for the unlawful taking of listed species. Incidental take refers to taking of listed species that results from, but is not the purpose of, carrying out an otherwise lawful activity by a federal agency or applicant. (50 CFR 402.02.) For non-federal projects, the USFWS may issue incidental take permits pursuant to section 10(a)(1)(B) of the ESA that are supposed to provide protection of sensitive species through the approval of HCPs that "detail measures to minimize and mitigate the potential impacts of projects to the maximum extent practicable." (*See* USFWS 2009b pp. 19–20.) As described below, several HCPs that cover the western spadefoot have been approved and, unfortunately, do not provide adequate protection for the species.

Listing under the federal ESA would certainly help to protect western spadefoots and assist in preventing further declines. Such listing is unlikely to happen any time soon, however. The species was petitioned for listing under the federal Endangered Species Act in 2012 (U.S. Fish and Wildlife Services, 2015), and in December 2023 the USFWS proposed to list the western spadefoot as threatened. The final rule is unlikely to occur in the next several years. The USFWS national domestic listing workplan addresses ESA listing decisions in the coming years. As of May 23, 2024, the workplan for fiscal years 2024-2028 did not include western spadefoot.[3] The current federal administration has since focused on limiting the application and reach of the ESA. For example, in April 2025, the administration proposed to rescind the longstanding definition of "harm" to species under the ESA. (90 FR 16102.)

[3] https://www.fws.gov/sites/default/files/documents/2024-05/national-domestic-listing-workplan-2024.pdf

Western spadefoots do benefit indirectly from the federal ESA listing of other co-occurring vernal pool species. For example, 20 vernal pool plant and animal species are listed as threatened or endangered under the federal ESA, and are therefore subject to habitat conservation and management efforts that will also benefit western spadefoot populations (US Fish and Wildlife Service, 2005). However, these protections are not always fully implemented, enforced, or adequate (US Fish and Wildlife Service, 2005). Despite the listing of numerous vernal pool species under the Federal Endangered Species Act, projects with the potential to destroy, degrade, or fragment vernal pool habitat—including conversion for human uses—have nonetheless been approved, in spite of the resident species' endangered status (See Table 1 above). Thus, while the protections applied to other vernal pool species under the federal ESA may benefit western spadefoots where they co-exist, these protections are not sufficient for effective overall conservation of the species.

National Wildlife Refuge System Improvement Act of 1997

The National Wildlife Refuge System Improvement Act established the protection of biodiversity as the primary purpose of the National Wildlife Refuge system, prioritizing wildlife conservation over other uses (P.L. 105-57, 111 Stat. 1252 (1997)).

CNDDB records show western spadefoot occurrences in 7 National Wildlife Refuges ("NWR") (number of CNDDB observations in parentheses):

- Bitter Creek NWR (1)
- Pixley NWR (4)
- San Diego Bay NWR (1)
- San Diego NWR (42)
- San Joaquin River NWR (1)
- San Luis NWR (6)

These occurrences make up only 56 of the 1443 occurrences included in the CNDDB, and most of them are clustered in the San Diego and San Luis NWRs. Additionally, like the State Protected Areas discussed below, these protected areas are scattered throughout the large range of the western spadefoot. While all protected habitat is valuable, National Wildlife Refuges represent only a small fraction of the western spadefoot's range, and do not provide adequate protections for the species as a whole. California's NWR's are not sufficient to ensure western spadefoot survival and recovery.

Sikes Act

The Sikes Act requires the U.S. Department of Defense to develop and prepare Integrated Natural Resource Management Plans ("INRMPs") for military installations; these plans provide for the conservation and rehabilitation of natural resources on military lands consistent with military uses. (16 U.S.C. § 670.) Western spadefoots occur at Camp Roberts in Lan Luis Obispo County (INRMP 2022), Fort Hunter Liggett in Monterey County (INRMP 2012), and while both plans include habitat protection, neither include specific management goals or strategies for western spadefoots. The March Air Force Base INRMP (2021) notes that western spadefoots have not been observed, but suitable habitat does occur on the base. While INRMPs provide benefits to covered species when fully implemented, they are not regulatory mechanisms because their implementation is subject to funding availability. They are insufficient to ensure western spadefoot survival and recovery.

Federal Land Policy and Management Act

The Federal Land Policy and Management Act ("FLPMA") regulates the management of public lands administered by the BLM; specifically the management, protection, development, and enhancement of public lands with the intention to, among other things, "…preserve and protect certain public lands in their natural condition; that will provide food and habitat for fish and wildlife and domestic animals…" as well as recreation and human occupancy and use. (43 U.S.C. § 1701.) This Act could protect western spadefoots on BLM lands, but only a small portion of the species' range falls on BLM lands.

A query of CNDDB shows 20 occurrences in the Carrizo Plain National Monument, one in the California Coastal National Monument, and two in the BLM Otay Mountain Wilderness Area. The Carrizo Plain National Monument Resource Management Plan (2010) does include management of western spadefoots, including vernal pool protection and population monitoring. The California Coastal National Monument and Otay Mountain Wilderness Area do not include species-specific management efforts but do limit development and intensive uses. The remaining western spadefoot occurrences on BLM lie outside of established National Conservation Areas of National Monuments. The FLPMA is therefore unlikely to provide significant protection for the western spadefoot.

B. State Regulatory Mechanisms

California Species of Special Concern

In California, the western spadefoot is a Priority I Species of Special Concern ("SSC") (Thomson et al., 2016). SSC status applies to animals not listed under the federal ESA which are declining at a rate that could result in listing or historically occurred in low numbers and known threats to their persistence currently exist. The SSC designation is intended to result in special consideration by CDFW, land managers, and others, to focus research and management attention on the species. SSC are supposed to receive this special consideration during preparation of CEQA documents. But as discussed in the section on CEQA below, the CEQA process has proven inadequate to protect western spadefoots or reverse their declines in California. The practical benefit of the SSC designation for the western spadefoot has been minimal. Such status may call attention to the species and prompt more information to be collected about the loss of its habitat in Environmental Impact Reports and other documents, but it has not halted habitat loss for spadefoot, or other factors causing the decline of the species. SSC species also do not benefit from the prohibitions against "take" that a species listed under the California or federal Endangered Species Act would receive.

California Environmental Quality Act

The California Environmental Quality Act ("CEQA") is California's landmark environmental law and establishes a state policy to prevent the "elimination of fish or wildlife species due to man's activities, ensure that fish and wildlife populations do not drop below self-perpetuating levels, and preserve for future generations representations of all plant and animal communities...." (Cal. Pub. Res. Code § 21001(c)). Under CEQA, state and local agencies must analyze and disclose the potentially significant environmental impacts of discretionary activities that they approve or carry out. CEQA provides that agencies should not approve projects as proposed if there are feasible alternatives or mitigation measures which would substantially lessen the significant environmental effects of such projects. (Cal. Pub. Res. Code § 21002.)

CEQA requires a "mandatory finding of significance" if a project may "substantially reduce the number or restrict the range of an endangered, rare or threatened species." (Cal. Code Regs., tit. 14, § 15065(a)(1)). CDFW has interpreted this provision to apply to species of special concern, which are species that are "experiencing, or formerly experienced, serious (noncyclical) population declines or range retractions (not reversed) that, if continued or resumed, could qualify it for State threatened or endangered status" (California Department of Fish and Wildlife, 2025). CDFW further provides that species of special concern "should be considered during the environmental review process." (*Id.*; Cal. Code Regs., tit. 14, § 15380). Thus, a potentially substantial impact on a species of special concern *could* be construed as "per se" significant under CEQA. (*Vineyard Area Citizens for Responsible Growth, Inc. v. City of Rancho Cordova* (2007) 40 Cal.4th 412, 449).

However, CEQA does not provide any specific legal protection for species of special concern aside from the requirement that projects triggering CEQA review must analyze the impacts of the proposed action on such species. Importantly, lead agencies have discretion to develop their own thresholds of significance. (*See East Sacramento Partnerships for a Livable City v. City of Sacramento* (2016) 5 Cal.App.5th 281, 300; Cal. Code Regs., tit. 14, § 15064(d)). Local agencies—who are often under pressure from developers to approve projects—can sometimes make significance determinations that are inconsistent with independent scientific analysis, including CDFW's analysis.

Though state and federal wildlife agencies can weigh in, protection of non-listed species through CEQA is at the discretion of the lead agency involved. CEQA provides that when overriding social and economic considerations can be demonstrated, project proposals may go forward, even in cases where the continued existence of the species may be threatened, or where adverse impacts are not mitigated to the point of insignificance. Even when a lead agency acknowledges that an effect is "significant," CEQA allows a lead agency to adopt a "statement of overriding considerations" and approve a project if the agency finds that other factors outweigh the environmental costs of the project or that further mitigation is infeasible (Cal. Pub. Res. Code § 21081; Cal. Code Regs., tit. 14, § 15093(b)). This means that lead agencies may approve projects that they determine will have a significant effect on a wildlife population like the western spadefoot. Additionally, environmental review under CEQA is project-specific and rarely considers population-wide impacts to species like the spadefoot. Although CEQA requires that agencies consider a project's "cumulative" impacts, these analyses are usually limited in geographic scope and do not account for regional or statewide impacts—including habitat loss.

Additionally, the legislature has continued to amend CEQA to make it easier for local agencies and developers to avoid conducting environmental review for a host of projects. For example, in

June 2025, the state legislature passed and Governor Newsom signed sweeping rollbacks to CEQA under Senate Bill (SB) 131 (formerly SB 607, authored by Sen. Scott Wiener) as a budget trailer bill. The Bill exempts certain projects from environmental review and makes legal challenges more difficult and in some cases impossible, even when a project would have significant environmental impacts, making sensitive habitats for species like western spadefoot even more vulnerable to destruction and fragmentation.

Although CEQA's environmental review provisions have provided some benefit to western spadefoots, they have not prevented local and regional species decline. While helpful, CEQA is inadequate to protect the western spadefoot.

State Ecological Reserves and State Wildlife Areas

The California Fish and Game Code provides for the establishment of ecological reserves that "protect threatened or endangered native plants, wildlife, or aquatic organisms or specialized habitat types, both terrestrial and nonmarine aquatic" (Cal. Fish & Game Code § 1580). "Take" of wildlife is prohibited in State Ecological Reserves ("SERs") (Cal. Code Regs. Tit. 14, § 630), which are generally governed by Land Management Plans. The California Code of Regulations also provides for the establishment of state wildlife areas ("SWAs"), which allow for a broader use: "wildlife areas are maintained for the primary purposes of developing a statewide program of ecological conservation, restoration, preservation, development and management of wildlife and wildlife habitat and hunting" (Cal. Code Regs. Tit. 14, § 551 (a)).

CNDDB records show western spadefoot occurrences in 22 SERs and 6 SWAs (number of CNDDB observations in parentheses):

- Alkali Sink Ecological Reserve (1)
- Allensworth Ecological Reserve (9)
- Box Springs Reserve (1)
- Burton Mesa Ecological Reserve (1)
- Buttonwillow Ecological Reserve (2)
- Carrizo Plains Ecological Reserve (13)
- Corral Hollow Ecological Reserve (1)
- Del Mar Mesa -- Lopez Ridge Ecological Reserve (1)
- Elliott Chaparral Reserve (1)
- Kerman Ecological Reserve (3)
- Lokern Ecological Reserve (1)
- Motte Rimrock Reserve (1)
- North Carrizo Ecological Reserve (2)
- North Grasslands Wildlife Area (1)
- Rancho Jamul Ecological Reserve (15)
- Santa Margarita Ecological Reserve (1)
- Santa Rosa Plateau Ecological Reserve (6)
- Semitropic Ecological Reserve (5)
- Stone Corral Ecological Reserve (6)
- Stone Ridge Ecological Reserve (1)
- Thomes Creek Ecological Reserve (1)
- Tule Elk State Reserve (1)

- French Valley Wildlife Area (1)
- Hollenbeck Canyon Wildlife Area (7)
- Los Banos Wildlife Area (1)
- Mendota Wildlife Area (2)
- Oroville Wildlife Area (1)
- San Jacinto Wildlife Area (12)

Such records may appear to provide significant benefits for the western spadefoot, but it is important to note that these occurrences make up only 99 of the 1443 occurrences included in the CNDDB. Additionally, like the National Wildlife Refuges discussed above, these protected areas are scattered throughout the large range of the western spadefoot. While all protected habitat is valuable, SERs and SWAs represent only a small, fragmented fraction of the western spadefoot's range, and do not provide adequate protections for the species as a whole. SERs and SWAs are insufficient to ensure western spadefoot survival and recovery.

Porter-Cologne Water Quality Control Act

The Porter-Cologne Water Quality Control Act ("Porter-Cologne") of 1969 governs water quality regulation in California. Porter-Cologne requires permits for waste discharges and dredge and fill into Waters of the State, which includes WOTUS and additional waters including surface, ground, saline waters, and wetlands (Cal. Water Code § 13000 et seq.). Porter-Cologne's purpose is to protect the quality of the state's water resources— any such "activities and factors" affecting water quality "shall be regulated to attain the highest water quality which is reasonable." (Cal. Water Code § 13000).

Under Section 401 of the CWA, California Regional Water Quality Control Boards (the "Water Boards") exercise authority to regulate discharges into WOTUS in California. For waters that do not meet the definition of WOTUS, but are nonetheless included within Waters of the State, Porter-Cologne requires Waste Discharge Requirement permits for any discharges, including dredge and fill material.

Vernal pools and other temporary wetlands as well as temporary streams are surface waters and are included in the definition as Waters of the State under the Porter-Cologne Act. (*See* Cal. Water Code § 13050(e).) Discharge into or dredge and fill of any vernal pool requires permits from the California Water Boards, which in turn require compensatory mitigation for fill and excavation impacts.

Although the *Sackett* decision narrowed only the scope of federal jurisdiction and did not weaken California's more stringent wetlands protections, the ruling has placed an incredible burden on state regulators. Without federal permitting and federal enforcement, the Water Boards are now tasked with regulating a vastly expanded number of water bodies with fewer administrative and legal resources and less enforcement capacity (State Water Resources Control Board, 2023a, 2023b). Legislation introduced in 2025 (SB 601 Allen) could have alleviated some of this burden by strengthening Porter-Cologne, but the bill was not passed. Porter-Cologne is therefore unlikely to adequately prevent destruction or modification of western spadefoot aquatic habitat.

California Coastal Act

The California Coastal Act of 1976 charges the California Coastal Commission with regulating the water quality of coastal wetlands by “minimizing adverse effects of wastewater discharges” within the coastal zone (Cal. Pub. Res. Code § 30231). The coastal zone is defined as “the land and water area of the State of California from the Oregon border to the border of the Republic of Mexico, specified on maps [adopted by the State legislature]…, extending seaward to the state’s outer limit of jurisdiction, including all offshore islands, and extending inland generally 1,000 yards from the mean high tide line of the sea.” (Cal. Pub. Res. Code § 30103). There are a few exceptions in which the coastal zone extends further inland, but it generally covers only a narrow strip of the California coast. While some western spadefoot populations may occur within this zone, the vast majority occur further inland and therefore do not benefit from the Coastal Act’s protections.

C. Local and Regional Regulatory Mechanisms

Regional and local conservation plans in some areas of the spadefoot’s range provide limited management. In Southern California, the majority of suitable spadefoot habitat areas fall within the footprint of regional conservation plans implemented during the past 25 years (Rose et al., 2020). Yet even within regions covered by a conservation plan, 61% of the potentially suitable spadefoot habitat is not currently protected (or lies within Department of Defense (“DoD”) land), highlighting the unfortunate reality that the existence of a conservation plan on paper does not guarantee protection of habitat for all species subject to said plan (Rose et al., 2020).

Natural Community Conservation Plans and Habitat Conservation Plans

The Natural Community Conservation Planning Act is a voluntary conservation planning mechanism for proposed development projects within a planning area to avoid or minimize impacts to wildlife. (Cal. Fish & Game Code § 2801(f).) The NCCP Act is designed to promote coordination among agencies and landowners to conserve unfragmented habitat areas and multihabitat management. (Cal. Fish & Game Code § 2801(d).). Natural Community Conservation Plans (“NCCPs”) are state-approved local or regional plans that authorize “take” of state-protected species and allow for development and activities in the covered areas. Compared with Habitat Conservation Plans (see below), NCCPs promote conservation actions that benefit landscape-level natural communities and are less focused on individual species conservation goals.

The Habitat Conservation Plan Program is defined in Section 10 of the federal ESA. Habitat Conservation Plans (“HCPs”) are local or regional planning documents that guide development on private lands on which federally-threatened and -endangered species occur to ensure compliance with the federal ESA. HCPs authorize “take” of protected species for covered activities, and in exchange generally require conservation measures intended to avoid a net adverse impact on covered species. HCPs can be applied at different scales from individual projects to whole regions.

Overall, coverage of western spadefoots in HCPs/NCCPs is piecemeal and limited and does not provide adequate mechanisms for species-level protection and recovery. Although HCPs and

NCCPs that list the western spadefoot as a covered/target species offer some regulatory protection or conservation consideration for the species, conservation measures and mitigation requirements vary from plan to plan. For example, some HCPs provide detailed information on avoidance and minimization of take during development activities, while others provide vague requirements of “no net loss” of habitat. Some HCPs/NCCPs cover areas within the western spadefoot’s range, but do not consider the species at all, although wetland protections included in some of these plans may incidentally benefit spadefoot (Table 2). In the East Contra Costa County HCP and Yolo County HCP/NCCPs, western spadefoots were considered either in the HCP itself or in previous drafts but were ultimately excluded from coverage. Four additional HCP/NCCPs within the species’ range are currently in development, and it is unclear whether they will include protections for western spadefoots (San Benito County NCCP/HCP, San Diego East County MSCP, San Diego North County MSCP, and Yuba-Sutter NCCP/HCP).

Several HCPs/NCCPs that do not cover the western spadefoot do provide protections for vernal pool habitats (Table 2), which may also benefit western spadefoots. However, western spadefoots also require suitable upland habitat to live in during the non-breeding season. Merely protecting breeding pools without protecting sufficient connected upland habitat will drive the species to extinction. For example, the City of San Diego Vernal Pool Habitat Conservation Plan focuses exclusively on protecting pools for federally endangered fairy shrimp species and vernal pool plants that do not as desperately require connected upland habitat. As such, the management activities outlined in the plan do not focus on habitat needs of western spadefoots, including the importance of suitable upland habitat surrounding vernal pools and the connectivity between vernal pools. Vernal pool protections are certainly necessary, and do benefit western spadefoots, but they are not sufficient to protect or restore western spadefoot populations.

While HCPs can provide some benefits for covered species, they are not always effectively implemented. An early nationwide study of some of the first HCPs developed was conducted by the National Center for Ecological Analysis & Synthesis and the American Institute of Biological Sciences (Kareiva et al., 1999). Kareiva et al. (1999) found that most HCPs contributed to habitat losses for the targeted species, failed to meet recovery goals, and suffered from poor planning and plan evaluation. Among the failures of HCPs discussed by Kareiva et al. (1999) were:

- nearly 30% of HCPs allowed “take” of 100% of the focal species’ populations or habitat in the permit area
- about 50% of HCPs allowed 50% or more of the species’ populations or habitat in the plan area to be “taken”
- 43% of the time, HCPs failed to provide sufficient mitigation measures
- 23% of the time, species and their habitats would be “taken” before mitigation measures had been implemented and found effective
- most HCPs failed to reduce allowed “take” levels or use other more conservative approaches in the face of inadequate information or uncertainties
- 33% of HCPs failed to secure up-front funding to ensure that mitigation actually occurs
- 81% of HCPs studied would have irreversible impacts

Not surprisingly, Kareiva et al. (1999) found that HCPs which fail to adequately conserve species also tend to lack rigorous impact assessments and planning. The Kareiva et al. (1999) study found that:

- 75% of the time, impacts to species were not adequately studied by HCPs
- 42% to 49% of the time, HCPs failed to quantify how much of a species' habitat and population, respectively, would be "taken"
- most HCPs used low quality data to evaluate their mitigation measures
- 25% of the time, sufficient information did not exist to determine how HCPs would affect the species' viability

Subsequently, Rahn et al. (2006) reviewed the species selected for coverage in 22 multispecies HCPs from USFWS Region 1 that were approved by the end of 2004. HCPs frequently cover multiple species, some federally listed and others not. Rahn et al. (2006) focused their evaluation exclusively on such multiple species HCPs (MSHCPs). Federal and state wildlife agencies promote the multispecies approach because it both increases certainty for the permittee in case of future listings and increases the "biological value" of the plans by providing "more opportunities for strategically placing appropriate conservation in an ecosystem context" and early consideration of the needs of unlisted species (US Fish and Wildlife Service & US National Marine Fisheries Service, 2016). Rahn et al. (2006) sought to evaluate the claim that MSHCPs provide special conservation value. While a comprehensive planning approach at the community, habitat, or ecosystem level may seem reasonable and efficient, it carries the risk that the needs of particular species may be overlooked. However, many MSHCPs that are created to provide comprehensive coverage for multiple species actually end up focusing on just one species, generally the most prominent threatened or endangered species in the plan area (Smallwood et al., 1998). In fact, two studies found that species covered under multiple-species plans were generally less likely to show improving trends in status than species covered under single-species plans (Boersma et al., 2001; Taylor et al., 2005). California lacks any single-species HCPs or NCCPs focused on western spadefoots, making them vulnerable to these weaknesses in implementation even when they are covered under a multi-species plan.

To gauge the extent to which MSHCPs incorporate science-based conservation planning, Rahn et al. (2006) evaluated whether or not covered species were confirmed in the planning area, and whether or not the plan contained specific conservation measures for the covered species. Rahn et al. (2006) found that the conservation benefits of multispecies plans to individual covered species may be overestimated, that conservation measures were often not clearly defined, and that the presence of the species in the planning area was not even confirmed for 41% of covered species. While Rahn et al. (2006) do not question the conservation value of multispecies plans, their study suggested that changes were needed to achieve full conservation potential. They identified three shortcomings of MSHCPs that can substantially limit their conservation potential. First, many plans were overbroad, covering species for which they provided no localized scientific information. The lack of information makes it difficult to predict the effectiveness of a plan when an incidental take permit is issued, or to evaluate it during the permit term. Second, most unconfirmed species also did not have specific conservation actions. Finally, they found high levels of variability across plans in the species they covered, the levels of justification for that coverage, and the extent to which they offered species-specific conservation actions.

As described in Table 2, many HCPs/NCCPs that include western spadefoots do not include specific locality data within the plan area and do not provide species-specific conservation measures. Given the results of the meta-analyses described above, it is therefore unlikely that all HCPs/NCCPs that include western spadefoots are effectively protecting the species within the plan areas.

Regional Conservation Investment Strategies

California Department of Fish and Wildlife also implements the Regional Conservation Investment Strategies ("RCIS") Program, which encourages landscape-level conservation assessment and planning. Importantly, RCISs are voluntary and are not enforced, and are therefore a potentially useful but insufficient mechanism for protecting species or habitats. Several counties and regions within the western spadefoot's range have established RCISs, yet only four cover western spadefoots (Kaweah Groundwater Subbasin (2022), Yolo County (2020), San Bernadino County (2024)). Recommended approaches in the Yolo County and San Bernardino County RCISs include habitat protection and management; only the Yolo County plan provides a species-specific habitat preservation goal (407 acres of habitat). The Kaweah Groundwater Subbasin RCIS provides more thorough recommended actions, including protecting existing population and enhancing habitat through invasive species removal, managed grazing, ceasing use of rodenticides, disease monitoring/management, creation of wildlife crossings, and minimization of disturbance around breeding habitat.

Three others, East Bay (2021), Monterey (2021), and the draft San Joaquin Valley plan (draft as of 2025) exclude western spadefoots, despite the fact that they have been documented in the RCIS plan areas (CNDDB 2025). The Monterey and San Joaquin Valley RCISs designate the western spadefoot as a non-focal species, a species that is "associated with focal species and focal other conservation elements and will benefit from the same conservation and habitat enhancement actions," but which is not itself the primary focus of conservation or management goals or actions. Such determinations are made based on criteria including occurrence, availability of data, status, and rarity in the RCIS area. For a species like the western spadefoot, such criteria do not capture the dire reality of the species' conservation status. Their above-ground occurrence is cryptic, highly seasonal, and weather-dependent, they do not receive significant monitoring efforts, and they are experiencing compounding negative stressors across its range, so occurrence data are unlikely to provide a reliable understanding of the presence or absence, let alone population dynamics, in any given area.

D. Protected Lands

As described above, western spadefoot populations are present on several National Wildlife Refuges, National Monuments, State Ecological Reserves, and State Wildlife Areas. They are also present in various other protected lands that prohibit development or other intensive uses, including agricultural and conservation easements, one national park, and state marine conservation areas (**Figure 7**). While western spadefoot populations on these lands are at low risk of habitat loss or fragmentation due to development, many of these lands do not actively manage for western spadefoots. Without targeted conservation efforts, populations on protected lands can still experience harm from roads, recreation, invasive species, and other threats.

Overall, about 21% of CNDDB records occur on protected lands (304 of the 1443 occurrences). While this number may at first seem high considering protected lands make up a very small portion of the species' range (**Figure 7**) (US Fish and Wildlife Service, 2005), it is important to consider sampling bias. Protected lands are more likely to have been surveyed, and both detection and reporting are likely higher on public protected lands and conservation easements than on non-protected lands or private lands. Protected lands are not sufficient to prevent further species decline.

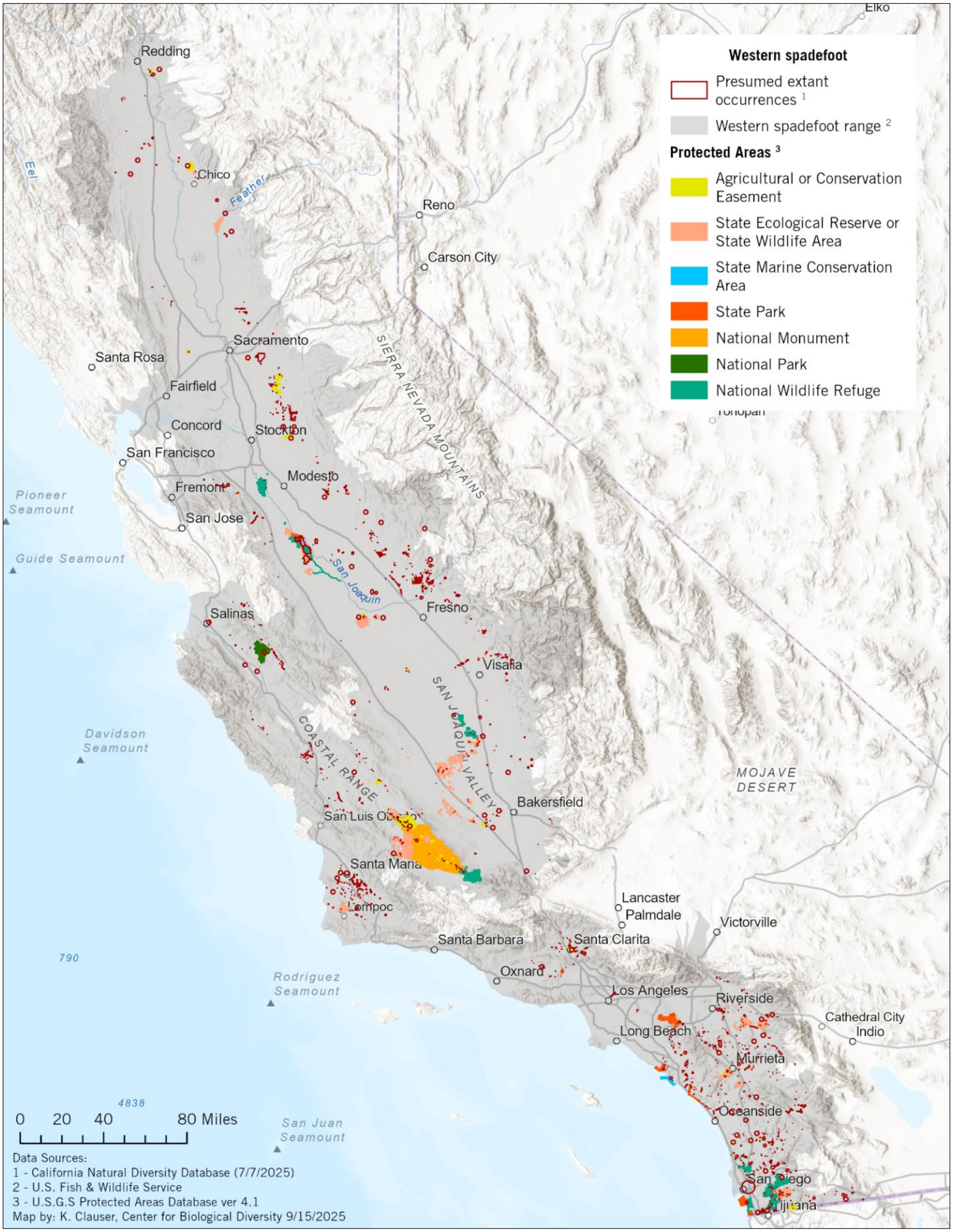

Figure 7. *Map of CNDDB western spadefoot occurrences and protected lands. Only 21% of western spadefoot occurrences in the CNDDB database are on protected lands. Most western spadefoot populations occur outside of protected lands.*

X. Suggestions for Future Management and Recovery Actions

Western spadefoots are facing a multitude of threats throughout their range in California that are driving both the northern and southern populations towards extinction. However, numerous management actions can help to alleviate these threats and aid the species' recovery. Recommendations for the management and recovery of western spadefoots are as follows:

- Formally protect both the northern and southern populations of western spadefoots as threatened and endangered, respectively, under CESA and prepare a recovery plan pursuant to Cal. Fish & Game Code § 2079.1, including management efforts aimed at reducing habitat loss, degradation, and fragmentation.
- Conduct systematic surveys of known historical localities that have not been assessed in the past 20 years, particularly in the Central Valley, to produce up-to-date occurrence data and guide protection efforts.
- Permanently protect known occupied breeding and upland habitat. Once habitat is disturbed or destroyed, it is not easily restored. Minimizing or eliminating disturbance around existing occupied upland and breeding habitats is the best and most cost-effective way to protect the species (Thomson et al., 2016).
- Protect and enhance suitable habitat that promotes connectivity within and between metapopulations. Western spadefoots move between upland and wetland breeding habitats and can travel hundreds of meters. Maintaining connectivity and accommodating short- and long-range dispersers is vital for continued population persistence and/or recolonization following a local extinction (Cushman, 2006; Semlitsch & Bodie, 2003).
- Implement buffer zones around protected habitat to help protect individuals and populations from edge effects from surrounding land use, including human activity, development-related disturbance, invasive species, noise, and pollutants. Buffers provide connectivity among suitable habitats and resiliency in the face of climate change, which will likely cause shifts in species ranges and distributions, including those of western spadefoots (Cushman et al., 2013; Heller & Zavaleta, 2009; Warren et al., 2011).
- Avoid or minimize the construction of new roads in western spadefoot habitat and enhance connectivity at existing barriers. Elevated road segments, underpasses, and culverts with directional fencing can improve spadefoot connectivity.
- Continue currently experimental efforts to restore breeding and upland habitat, including the creation of artificial ponds and the reestablishment of native grasslands, to help maintain healthy populations, increase climate resilience, and improve connectivity (Neal et al., 2020; Thomson et al., 2016). It is important to note that the success of artificial ponds is not guaranteed (Baumberger et al., 2020; Groff et al., 2012); such efforts should be carefully planned, monitored, and adaptively managed, and they should not replace conservation of existing habitat.
- Invest in monitoring and research programs that investigate the species' natural history and behavior, current connectivity and distribution, genetics, and disease threats (i.e., *Bd*) as well as other potential adaptive management strategies, like conservation grazing, prescribed fire, and conservation genetics.
- Research the potential for translocation and re-introduction in previously occupied areas with remaining suitable habitat or restored/created habitat to help spadefoots establish

local metapopulations and counter the effects of inbreeding due to genetic isolation (Frankham et al. 2019).

- In highly modified rangelands, implement sustainable grazing practices that promote natural vernal pool hydrology while minimizing direct impacts to breeding spadefoots. Disincentivize the conversion of rangeland to row crops or irrigated crops that require discing, planting, and harvesting (US Fish and Wildlife Service, 2005).

XI. Conclusion

In this petition, the Center for Biological Diversity has carefully assessed the best scientific information available regarding the western spadefoot. We have reviewed the best scientific and commercial information available regarding the historic, present, and future threats faced by the two western spadefoot populations (northern and southern) and have determined that the southern population is in danger of extinction throughout its range, and the northern population is likely to become endangered if it does not receive protection. The species' past and ongoing declines are largely attributable to habitat loss and fragmentation, particularly destruction and fragmentation of vernal pool complexes, although numerous other disturbances play roles as well. In the United States, the western spadefoot is endemic to California and despite its vulnerability to threats associated with California's growth and development, it does not currently receive either California or federal species-specific protection. As such, we urge the California Fish and Game Commission to protect the two western spadefoot populations (northern and southern) as threatened and endangered, respectively, under the California Endangered Species Act.

Table 2. Coverage of western spadefoot in NCCPs and HCPs.

Plan name	Year	Coverage of Spadefoot	Spadefoot Conservation Measures	Spadefoot occurrence in plan area	Acres of spadefoot (or other) habitat covered
Central/Coastal Orange County NHCCP/HCP	1996	Targeted[4]	Western spadefoot habitat preservation, restoration, enhancement, and management. (via Reserve System) Mitigation for loss of vernal pool habitat (minimization, compensatory restoration/creation).	Found in numerous localities within Orange County. 9500+ acres of potential habitat and 10 known breeding sites conserved, 12,000+ acres potential habitat and 3 known breeding sites taken.	9500+ acres of potential habitat and 10 known breeding sites conserved, 12,000+ acres potential habitat and 3 known breeding sites taken.
Kern water bank HCP-NCCP	1997	Targeted	Take avoidance and minimization. Habitat management/restoration on previous ag lands, not specific to spadefoot - managed for of 30 animals and 15 plants. Leaves possibility for introduction of spadefoot into managed wetlands but does not require it.	No specific information. *Website shows spadefoot do occur there (https://www.kwb.org/sensitive-wildlife-species). CNDDB shows numerous occurrences in and around the plan area.*	960 acres habitat preserve, 9,389 additional acres managed vegetation (p. S-2). No specific details on habitat types.
Natomas Basin, Sacramento HCP	2003	Targeted	Pre-construction surveys, take avoidance and minimization. Vernal pool mitigation via avoidance and preservation, vernal pool resources relocation, or payment into a conservation bank (p. V-5,6).	Low probability of occurrence in the plan area (p. II-36). *CNDDB shows numerous occurrences nearby in western Placer County.*	Very small wetland area; not specifically defined but Fig 10 shows little coverage.
South Sacramento HCP	2018	Targeted	Mitigation for direct impacts (work timing, exclusion fencing, monitoring, avoiding entrapment, erosion control, encounter protocol) (p.5-96) and habitat preservation and management (p.7-224).	41 occurrences in plan area, 20 in Urban Development Area (p3-78).	Covered activities will impact 23,000 acres of modeled aquatic and upland habitat (p.6-236). The plan will preserve and link at least 1,647 acres of modeled aquatic habitat and 22,061 acres modeled upland habitat (p. 7-224).

[4] "Targeted" indicates western spadefoot was included as a focal species of the plan. "Incidental" indicates that the plan does not include western spadefoot as a focal species but may benefit the species via habitat protections.

San Diego County Water Authority NCCP	2010	Targeted	Avoidance and minimization of impacts (surveys, exclusion fencing, invasive species management), including mitigation for impacted habitat. Vernal Pool Protection Policy states there should be no permanent impacts to vernal pools, requires mitigation for temporary impacts.	13 occurrences in plan area, 3 within probable impact zone.	Estimated maximum 373 acres total, all habitats (ES-1).
San Diego Gas & Electric NCCP	1995	Targeted	Avoidance and minimization of take, Avoidance, minimization and mitigation for loss of habitats (p.101).	No specific information.	Maximum 400 acres of impacts total, all habitats (p. vi).
Western Riverside County MSHCP	2003	Targeted	Maintain or improve vernal pool habitat. Management of alteration of hydrology, non-native plant species, farming, mining, grazing, off-road vehicle use and predation. (Table 5-2)	No specific information. *CNDDB shows many occurrences in plan area.*	Plan area covers 7,910 acres (2.1.3), 6,750 conserved (Table 3-1).
San Joaquin County MSCP	2000	Targeted	Retain known breeding sites, compensate for habitat loss (3:1 for natural lands, 2:1 preservation plus 1:1 creation for vernal pools) (p. 4-18, 5-51, 6-6). Vernal pool protections including preserve formation with natural, restored, and artificial pools (p. 5-33, 6-3).	Specific locality data not provided. Lots of occupied habitat.	12,488 acres occupied, 61,864 potential habitat. Expected conversion of occupied/potential habitat is 5,103 acres (7% of total avail habitat).
Final MSHCP Plan, SANDAG for the cities of Carlsbad, Encinitas, Escondido, Oceanside, San Marcos, Solana Beach, and Vista	2003	Targeted	Western spadefoot habitat preservation and management, avoidance, and mitigation for habitat loss (no net loss of wetlands) (4-216). Conserve 3/4 known populations.	4 known population locations.	22 acres of vernal pool.

Lake Mathews HCP	1995	Targeted	Conserve habitat, minimize and mitigate project habitat impacts via mitigation bank credits, habitat restoration, enhancement, acquisition of replacement habitat.	2 known occurrences in existing reserve (surrounding lake), also present in mitigation bank lands (surrounding reserve) (p. 121).	2722 acres estimated habitat in Multiple Species Reserve, 236 acres in operations areas and plan area projects (p. 44).
Western Placer County NCCP	2020	Incidental	Vernal pools mitigation via preservation of existing vernal pools or pool restoration. Focused on vernal pool invertebrate habitat.	Western spadefoot has been found in vernal pool complexes in the non-participating city of Roseville and may occur in the vernal pool complexes in the Plan Area, though there are no known occurrences to date (p. 3-64). *CNDDB shows numerous occurrences northwest of Roseville.*	Protect 17,000 acres of existing vernal pool complex, including 790 wetted acres of vernal pool constituent habitat. Restore/create 3,000 acres of vernal pool complex in the Reserve System (p. 5-17).
City of San Diego Vernal Pool Habitat Conservation Plan	2019	Incidental	Avoidance and minimization of impacts to vernal pools; mitigation via vernal pool restoration/enhancement/preservation. Focused on vernal pool invertebrates and plants.	No mention of spadefoot. *CNDDB shows numerous occurrences in plan area.*	Conserve 38.3 acres of vernal pools at 73 sites (at least 2473 pools total). Manage 62 vernal pool sites in perpetuity. Restore 19 vernal pool sites (within 12 complexes). Note that 9 vernal pool sites in the plan area will not be conserved or managed under this plan (p. 5-2).
East Contra Costa County	2007	Incidental	Vernal pools mitigation via preservation of existing vernal pools or pool restoration/creation. Focused on federally-listed vernal pool invertebrate habitat.	No specific information. *CNDDB shows no occurrences in plan area.*	121 acres of seasonal wetlands were mapped, vernal pool expected to be a small portion (p. 3-18).
San Diego MSCP South County Subarea Plan	1998	Incidental	Avoidance, minimization and mitigation for loss of wetlands (p.3-22). No specific mention of vernal pools or western spadefoot.	No mention of spadefoot. *CNDDB shows numerous occurrences in and around the plan area.*	Conserving 738/928 acres disturbed wetlands (p. 3-19).

Yolo County HCP	2018	Incidental	Create reserve system for habitat preservation, restoration. Preservation of grassland and wetland habitat for California Tiger Salamander (approx. 2100 acres) could benefit spadefoot.	No specific information. *CNDDB shows several occurrences in plan area.*	No specific information.
Orange County Transportation Authority NCCP HCP	2016	Incidental	None	No mention of spadefoot. Marginally suitable habitat. Limited potential to occur (App. p. 36).	Maximum 140 acres of impacts total, all habitats (ES-3).

XII. References

Adams, A. J., Kupferberg, S. J., Wilber, M. Q., Pessier, A. P., Grefsrud, M., Bobzien, S., Vredenburg, V. T., & Briggs, C. J. (2017). Extreme drought, host density, sex, and bullfrogs influence fungal pathogen infection in a declining lotic amphibian. *Ecosphere*, *8*(3), e01740.

Adams, M. J., Chelgren, N. D., Reinitz, D., Cole, R. A., Rachowicz, L. J., Galvan, S., McCreary, B., Pearl, C. A., Bailey, L. L., Bettaso, J., Bull, E. L., & Leu, M. (2010). Using occupancy models to understand the distribution of an amphibian pathogen, Batrachochytrium dendrobatidis. *Ecological Applications*, *20*(1), 289–302.

AECOM, Platenkamp, G., Vollmar Consulting, & Holland, R. F. (2009). *Loss of Central Valley Vernal Pools: Land Conversion, Mitigation Requirements, and Preserve Effectiveness*. Placer Land Trust.

AECOM & Vollmar Consulting. (2009). *U.S. Army Corps of Engineers Permitting and Mitigation of Central Valley Vernal Pool Impacts, 2000–2006*. Placer Land Trust.

Alford, R. A., & Richards, S. J. (1999). Global Amphibian Declines: A Problem in Applied Ecology. *Annual Review of Ecology and Systematics*, *30*(1), 133–165.

Alroy, J. (2015). Current extinction rates of reptiles and amphibians. *Proceedings of the National Academy of Sciences of the United States of America*, *112*(42), 13003–13008.

Alvarez, J. A., & Kerss, T. S. (2023). Microhabitat use by post-metamorphic western spadefoot (Spea hammondii) in Central California. *Sonoran Herpetologist*, *36*(1), 13–14.

Alvarez, J. A., & Woodall, J. (2024). Above ground phenology of terrestrial forms of western spadefoot with management recommendations. *California Fish and Wildlife Journal*, *110*(4).

Augustine, D., Davidson, A., Dickinson, K., & Van Pelt, B. (2021). Thinking Like a Grassland: Challenges and Opportunities for Biodiversity Conservation in the Great Plains of North America. *Rangeland Ecology & Management*, *78*, 281–295.

Axelsson, E., Nyström, P., Sidenmark, J., & Brönmark, C. (1997). Crayfish predation on amphibian eggs and larvae. *Amphibia-Reptilia*, *18*(3), 217–228.

Baillie, J. E. M., Hilton-Taylor, C., & Stuart, S. N. (Eds.). (2004). *2004 IUCN Red List of Threatened Species: A Global Species Assessment*. IUCN - The World Conservation Union.

Barry, S., Larson, S., & George, M. (2020). *California Native Grasslands: A Historical Perspective*.

Bauder, E. T., Kreager, D. A., & McMillan, S. C. (1998). *Vernal Pools of Southern California Recovery Plan*. U.S. Fish and Wildlife Service.

Bauder, E. T., & McMillan, S. (1998). *Current Distribution and Historical Extent of Vernal Pools in Southern California and Northern Baja California, Mexico*.

Baumberger, K. L., Backlin, A. R., Gallegos, E. A., Hitchcock, C. J., & Fisher, R. N. (2020). Mitigation Ponds Offer Drought Resiliency for Western Spadefoot (Spea hammondii) Populations. *Bulletin, Southern California Academy of Sciences*, *119*(1), 1–12.

Baumberger, K. L., Eitzel, M. V., Kirby, M. E., & Horn, M. H. (2019). Movement and habitat selection of the western spadefoot (Spea hammondii) in southern California. *PLoS ONE*, *14*(10), 1–16.

Beebee, T. J. C. (2013). Effects of Road Mortality and Mitigation Measures on Amphibian Populations. *Conservation Biology*, *27*(4), 657–668.

Berger, L., Speare, R., Daszak, P., Green, D. E., Cunningham, A. A., Goggin, C. L., Slocombe, R., Ragan, M. A., Hyatt, A. D., McDonald, K. R., Hines, H. B., Lips, K. R., Marantelli, G., & Parkes, H. (1998). Chytridiomycosis causes amphibian mortality associated with population declines in the rain forests of Australia and Central America. *Proceedings of the National Academy of Sciences*, *95*(15), 9031–9036.

Blackburn, D. C., & Wake, D. B. (2011). Class Amphibia Gray, 1825. In Z. -q. Zhang (Ed.), *Animal biodiversity: An outline of higher-level classification and survey of taxonomic richness: Vol. Zootaxa 31* (pp. 39–55).

Blaustein, A. R., Romansic, J. M., Kiesecker, J. M., & Hatch, A. C. (2003). Ultraviolet radiation, toxic chemicals and amphibian population declines. *Diversity and Distributions*, *9*(2), 123–140.

Blaustein, A. R., Urbina, J., Snyder, P., Reynolds, E., Dang, T., Hoverman, J., Han, B., Olson, D., Searle, C., & Hambalek, N. (2018). Effects of Emerging Infectious Diseases on Amphibians: A Review of Experimental Studies. *Diversity*, *10*(3), 81.

Blaustein, A. R., Walls, S. C., Bancroft, B. A., Lawler, J. J., Searle, C. L., & Gervasi, S. S. (2010). Direct and Indirect Effects of Climate Change on Amphibian Populations. *Diversity*, *2*(2), 281–313.

Boersma, P. D., Kareiva, P., Fagan, W. F., Alan Clark, J., & Hoekstra, J. M. (2001). How Good Are Endangered Species Recovery Plans? *BioScience*, *51*(8), 643.

Brehme, C. S., Hathaway, S. A., & Fisher, R. N. (2018). An objective road risk assessment method for multiple species: Ranking 166 reptiles and amphibians in California. *Landscape Ecology*, *33*, 911–935.

Brenes, R., Gray, M. J., Waltzek, T. B., Wilkes, R. P., & Miller, D. L. (2014). Transmission of Ranavirus between Ectothermic Vertebrate Hosts. *PLoS ONE*, *9*(3), e92476.

Brown, H. A. (1967). Embryonic Temperature Adaptations and Genetic Compatibility in Two Allopatric Populations of the Spadefoot Toad, Scaphiopus hammondi. *Evolution*, *21*(4), 742–761.

Brown, H. A. (1976). The status of California and Arizona populations of the western spadefoot toads (genus Scaphiopus). *Los Angeles County Museum of Natural History Contributions in Science*, *286*, 1–15.

Brunner, J. L., Storfer, A., Gray, M. J., & Hoverman, J. T. (2015). Ranavirus Ecology and Evolution: From Epidemiology to Extinction. In *Ranaviruses: Lethal Pathogens of Ectothermic Vertebrates*. Springer International Publishing.

Brunner, J., Olson, A., Rice, J., Meiners, S., Le Sage, M., Cundiff, J., Goldberg, C., & Pessier, A. (2019). Ranavirus infection dynamics and shedding in American bullfrogs: Consequences for spread and detection in trade. *Diseases of Aquatic Organisms*, *135*(2), 135–150.

Bucciarelli, G. M., Clark, M. A., Delaney, K. S., Riley, S. P. D., Shaffer, H. B., Fisher, R. N., Honeycutt, R. L., & Kats, L. B. (2020). Amphibian responses in the aftermath of extreme climate events. *Scientific Reports*, *10*, 3409–3409.

Buisson, E., Archibald, S., Fidelis, A., & Suding, K. N. (2022). Ancient grasslands guide ambitious goals in grassland restoration. *Science*, *377*(6606), 594–598.

California Department of Finance, Demographic Research Unit. (2025). *State And County Population Projections 2020-2070: P-1A Total Population for California [computer file]*. https://dof.ca.gov/forecasting/demographics/projections/

California Department of Fish and Wildlife. (2025). *Species of Special Concern*. https://wildlife.ca.gov/Conservation/SSC

Carey, C. (1993). Hypothesis Concerning the Causes of the Disappearance of Boreal Toads from the Mountains of Colorado. *Conservation Biology*, *7*(2), 355–362.

Carr, L. W., & Fahrig, L. (2001). Effect of Road Traffic on Two Amphibian Species of Differing Vagility. *Conservation Biology*, *15*(4), 1071–1078.

Cass, T. L. (2007). Natural history notes: Spea hammondii. Reproductive pattern. *Herpetological Review*, *38*(2), 196–197.

Cayan, D. R., Maurer, E. P., Dettinger, M. D., Tyree, M., & Hayhoe, K. (2008). Climate change scenarios for the California region. *Climatic Change*, *87*(S1), 21–42.

Christman, B. L., & Jennings, R. D. (2018). *Distribution of the amphibian fungus Batrachochytrium dendrobatidis in New Mexico*. New Mexico Department of Game and Fish.

Clarke, S. H., Lawrence, E. R., Matte, J., Gallagher, B. K., Salisbury, S. J., Michaelides, S. N., Koumrouyan, R., Ruzzante, D. E., Grant, J. W. A., & Fraser, D. J. (2024). Global assessment of effective population sizes: Consistent taxonomic differences in meeting the 50/500 rule. *Molecular Ecology*, *33*(11), e17353.

Collins, J. P., & Storfer, A. (2003). Global amphibian declines: Sorting the hypotheses. *Diversity and Distributions*, *9*(2), 89–98.

Corn, P. S. (2005). Climate change and amphibians. *Animal Biodiversity and Conservation*, *28*(1), 59–67.

Courtney, B. J. (2017, June 15). *Northlake Specific Plan Project (PROJECT) DRAFT SUPPLEMENTAL ENVIRONMENTAL IMPACT REPORT (DSEIR), SCH #2015031080* [Letter from CDFW to Los Angeles County Department of Regional Planning].

Crother, B. I., Bonett, R. M., Boundy, J., Burbrink, F. T., de Queiroz, K., Frost, D. R., Highton, R., Iverson, J. B., Jockusch, E. L., Kraus, F., Krysko, K. L., Leaché, A. D., Moriarity Lemmon, E., McDiarmid, R. W., Mendelson III, J. R., Meylan, P. A., Reeder, T. W., Ruane, S., & Seidel, M. E. (2017). Scientific and Standard English Names of Amphibians and Reptiles of North America North of Mexico, with Comments Regarding Confidence in Our Understanding. In J. J. Moriarty (Ed.), *Herpetological Circular No. 43* (8th Edition, p. 102). Society for the Study of Amphibians and Reptiles. http://www.ssarherps.org

Cushman, S. A. (2006). Effects of habitat loss and fragmentation on amphibians: A review and prospectus. *Biological Conservation*, *128*, 231–240.

Cushman, S. A., McRae, B., Adriaensen, F., Beier, P., Shirley, M., & Zeller, K. (2013). Biological corridors and connectivity. In D. W. Macdonald & K. J. Willis (Eds.), *Key Topics in Conservation Biology 2* (1st ed., pp. 384–404). Wiley.

Daszak, P., Berger, L., Cunningham, A. A., Hyatt, A. D., Green, D. E., & Speare, R. (1999). Emerging Infectious Diseases and Amphibian Population Declines. *Emerging Infectious Diseases*, *5*(6), 735–748.

Daszak, P., Cunningham, A. A., & Hyatt, A. D. (2003). Infectious disease and amphibian population declines. *Diversity and Distributions*, *9*(2), 141–150.

Davidson, C. (2004). Declining downwind: Amphibian population declines in California and historical pesticide use. *Ecological Applications*, *14*(6), 1892–1902.

Davidson, C., Shaffer, H. B., & Jennings, M. R. (2002). Spatial Tests of the Pesticide Drift, Habitat Destruction, UV-B, and Climate-Change Hypotheses for California Amphibian Declines. *Conservation Biology*, *16*(6), 1588–1601.

Davis, D. R., & Kerby, J. L. (2016). First Detection of Ranavirus in Amphibians from Nebraska, USA. *Herpetological Review*, *47*(1), 46–50.

Denver, R. J., Mirhadi, N., & Phillips, M. (1998). Adaptive plasticity in amphibian metamorphosis: Response of Scaphiopus hammondii tadpoles to habitat desiccation. *Ecology*, *79*(6), 1859–1872.

DeVore, J. L., & Maerz, J. C. (2014). Grass invasion increases top-down pressure on an amphibian via structurally mediated effects on an intraguild predator. *Ecology*, *95*(7), 1724–1730.

Diffenbaugh, N. S., Swain, D. L., & Touma, D. (2015). Anthropogenic warming has increased drought risk in California. *Proceedings of the National Academy of Sciences of the United States of America*, *112*(13), 3931–3936.

Dimmitt, M. A., & Ruibal, R. (1980). Environmental correlates of emergence in spadefoot toads (Scaphiopus). *Society for the Study of Amphibians and Reptiles*, *14*(1), 21–29.

Duffus, A. L. J., Waltzek, T. B., Stöhr, A. C., Allender, M. C., Gotesman, M., Whittington, R. J., Hick, P., Hines, M. K., & Marschang, R. E. (2015). Distribution and Host Range of Ranaviruses. In *Ranaviruses: Lethal Pathogens of Ectothermic Vertebrates* (pp. 9–57). Springer International Publishing.

Echaubard, P., Little, K., Pauli, B., & Lesbarrères, D. (2010). Context-Dependent Effects of Ranaviral Infection on Northern Leopard Frog Life History Traits. *PLoS ONE*, *5*(10), e13723.

Ervin, E. L., Smith, C. D., & Christopher, S. V. (2005). Natural history notes: Spea hammondii. Reproduction. *Herpetological Review*, *36*(3), 309–310.

Faist, A., & Beals, S. (2017). Invasive plant feedbacks promote alternative states in California vernal pools. *Restoration Ecology*, *26*.

Feaver, P. E. (1971). *Breeding pool selection and larval mortality of three California amphibians: Ambystoma tigrinum californiense (Gray), Hyla regilla (Baird and Girard), and Scaphiopus hammondi hammondi (Girard)* (p. 63).

Fellers, G. M., Green, D. E., & Longcore, J. E. (2001). Oral Chytridiomycosis in the Mountain Yellow-Legged Frog (Rana muscosa). *Copeia*, *2001*(4), 945–953.

Fisher, M. C., & Garner, T. W. J. (2007). The relationship between the emergence of Batrachochytrium dendrobatidis, the international trade in amphibians and introduced amphibian species. *Fungal Biology Reviews*, *21*(1), 2–9.

Fisher, M. C., & Garner, T. W. J. (2020). Chytrid fungi and global amphibian declines. *Nature Reviews Microbiology*, *18*(6), 332–343.

Fisher, R. N., & Shaffer, H. B. (1996). The Decline of Amphibians in California's Great Central Valley. *Conservation Biology*, *10*(5), 1387–1397.

Frankham, R. R. (1995). Effective population size/adult population size ratios in wildlife: A review. *Genetics Research*, *66*, 95–107.

Frankham, R. R., Bradshaw, C. J. A., & Brook, B. W. (2014). Genetics in conservation management: Revised recommendations for the 50/500 rules, Red List criteria and population viability analyses. *Biological Conservation*, *170*, 56–63.

Gallant, A. L., Klaver, R. W., Casper, G. S., & Lannoo, M. J. (2007). Global Rates of Habitat Loss and Implications for Amphibian Conservation. *Copeia*, *2007*(4), 967–979.

García-París, M., Buchholz, D. R., & Parra-Olea, G. (2003). Phylogenetic relationships of Pelobatoidea re-examined using mtDNA. *Molecular Phylogenetics and Evolution*, *28*, 12–23.

Gherghel, I., & Martin, R. A. (2020). Postglacial recolonization of North America by spadefoot toads: Integrating niche and corridor modeling to study species' range dynamics over geologic time. *Ecography*, *43*(10), 1499–1509.

Goldberg, S. R. (2023). Notes on reproduction of western spadefoot toads, Spea hammondii (Anura: Scaphiopodidae). *Sonoran Herpetologist*, *36*(1), 10–12.

Grant, E. H. C., Miller, D. A. W., Schmidt, B. R., Adams, M. J., Amburgey, S. M., Chambert, T., Cruickshank, S. S., Fisher, R. N., Green, D. M., Hossack, B. R., Johnson, P. T. J., Joseph, M. B., Rittenhouse, T. A. G., Ryan, M. E., Waddle, J. H., Walls, S. C., Bailey, L. L., Fellers, G. M., Gorman, T. A., … Muths, E. (2016). Quantitative evidence for the effects of multiple drivers on continental-scale amphibian declines. *Scientific Reports*, *6*(1), 25625–25625.

Gray, M. J., & Chinchar, V. G. (2015). Introduction: History and Future of Ranaviruses. In *Ranaviruses: Lethal Pathogens of Ectothermic Vertebrates*. Springer International Publishing.

Gray, M. J., Miller, D. L., & Hoverman, J. T. (2009). *First Report of Ranavirus Infecting Lungless Salamanders*.

Green, D. E., Converse, K. A., & Schrader, A. K. (2002). Epizootiology of Sixty-Four Amphibian Morbidity and Mortality Events in the USA, 1996-2001. *Annals of the New York Academy of Sciences*, *969*(1), 323–339.

Greenspan, S. E., Calhoun, A. J. K., Longcore, J. E., & Levy, M. G. (2012). Transmission of Batrachochytrium dendrobatidis to wood frogs (Lithobates sylvaticus) via a bullfrog (L. catesbeianus) vector. *Journal of Wildlife Diseases*, *48*(3), 575–582.

Groff, L. A., Duffy, W. G., Kahara, S. N., & Chapin, S. J. (2012). Temporally irregular breeding of western spadefoot toads (Spea hammondii) in managed wetlands. *Northwestern Naturalist*, *93*(1), 79–83.

Hall, J. A. (1998). Scaphiopus intermontanus. In *Catalogue of American Amphibians and Reptiles* (Vol. 650, p. 650.1-650.17). Society for the Study of Amphibians and Reptiles.

Halstead, B. J., Baumberger, K. L., Backlin, A. R., Kleeman, P. M., Wong, M. N., Gallegos, E. A., Rose, J. P., & Fisher, R. N. (2021). Conservation implications of spatiotemporal variation in the terrestrial ecology of western spadefoots. *Journal of Wildlife Management*, *85*(7), 1377–1393.

Halstead, B. J., Rose, J. P., Clark, D., Kleeman, P. M., & Fisher, R. N. (2022). Multi-scale patterns in the occurrence of an ephemeral pool-breeding amphibian. *Ecosphere*, *13*(3), 1–14.

Hamer, A. J., Barta, B., Bohus, A., Gál, B., & Schmera, D. (2021). Roads reduce amphibian abundance in ponds across a fragmented landscape. *Global Ecology and Conservation*, *28*, e01663.

Hamer, A. J., & McDonnell, M. J. (2008). Amphibian ecology and conservation in the urbanising world: A review. *Biological Conservation*, *141*(10), 2432–2449.

Hamilton, W. O. (2008). *Removal of invasive grass thatch in vernal pool complexes leads to an increase in native plant diversity*. California State University, Sacramento.

Hayes, M. P., & Warner, J. (1985). Life history notes: Rana catesbeiana, food. *Herpetological Review*, *16*(4), 109.

Hayes, T. B., Case, P., Chui, S., Chung, D., Haeffele, C., Haston, K., Lee, M., Mai, V. P., Marjuoa, Y., Parker, J., & Tsui, M. (2006). Pesticide mixtures, endocrine disruption, and amphibian declines: Are we underestimating the impact? *Environmental Health Perspectives*, *114*(Suppl 1), 40–50.

Hayes, T. B., Collins, A., Lee, M., Mendoza, M., Noriega, N., Stuart, A. A., & Vonk, A. (2002). Hermaphroditic, demasculinized frogs after exposure to the herbicide atrazine at low ecologically relevant doses. *Proceedings of the National Academy of Sciences of the United States of America*, *99*(8), 5476–5480.

Heller, N. E., & Zavaleta, E. S. (2009). Biodiversity management in the face of climate change: A review of 22 years of recommendations. *Biological Conservation*, *142*(1), 14–32.

Hels, T., & Buchwald, E. (2001). The effect of road kills on amphibian populations. *Biological Conservation*, *99*(3), 331–340.

Hernández, D. L., Vallano, D. M., Zavaleta, E. S., Tzankova, Z., Pasari, J. R., Weiss, S., Selmants, P. C., & Morozumi, C. (2016). Nitrogen Pollution Is Linked to US Listed Species Declines. *BioScience*, *66*(3), 213–222.

Holderegger, R., & Di Giulio, M. (2010). The genetic effects of roads: A review of empirical evidence. *Basic and Applied Ecology*, *11*(6), 522–531.

Holland, R. F. (2009). *California's Great Valley Vernal Pool Habitat Status and Loss: Rephotorevised 2005*. Placer Land Trust.

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

H.T. Harvey and Associates. (2021). *Alma Bridge Road-related Newt Mortality Study* (pp. 60–60).

Huntsinger, L., Bartolome, J. W., & D'Antonio, C. M. (2007). Grazing Management on California's Mediterranean Grasslands. In M. R. Stromberg (Ed.), *California Grasslands: Ecology and Management*. University of California Press.

IPCC. (2023). *Climate Change 2023 Synthesis Report: Summary for Policymakers*. Intergovernmental Panel on Climate Change (IPCC).

Jancovich, J. K., Davidson, E. W., Parameswaran, N., Mao, J., Chinchar, V. G., Collins, J. P., Jacobs, B. L., & Storfer, A. (2004). Evidence for emergence of an amphibian iridoviral disease because of human-enhanced spread: SALAMANDER RANAVIRUS PHYLOGEOGRAPHY. *Molecular Ecology*, *14*(1), 213–224.

Jennings, M. R., & Hayes, M. P. (1994). *Amphibian and Reptile Species of Special Concern in California*.

Kareiva, P., Andelman, S., Doak, D., Elderd, B., Groom, M., Hoekstra, J., Hood, L., James, F., Lamoreux, J., LeBuhn, G., McCulloch, C., Regetz, J., Savage, L., Ruckelshaus, M., Skelly, D., Wilbur, H., Zamudio, K., & NCEAS HCP working group. (1999). *Using Science in Habitat Conservation Plans*. National Center for Ecological Analysis and Synthesis.

Kassar, C. (2009). *Environmental impacts of ORVs on the Rubicon Trail*. Center for Biological Diversity.

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

Kiesecker, J. M., Blaustein, A. R., & Belden, L. K. (2001). Complex causes of amphibian population declines. *Nature*, *410*(6829), 681–684.

Kirschman, L. J., Palis, J. G., Fritz, K. A., Althoff, K., & Warne, R. W. (2017). *Two Ranavirus-Associated Mass-Mortality Events among Larval Amphibians in Illinois, USA*.

Koteen, L. E., Baldocchi, D. D., & Harte, J. (2011). Invasion of non-native grasses causes a drop in soil carbon storage in California grasslands. *Environmental Research Letters*, *6*.

Laabs, D. M., Orloff, S. G., & Allaback, M. L. (2001). Pond and Stream-Breeding Amphibians. In J. E. Vollmar (Ed.), *Wildlife and Rare Plant Ecology of Eastern Merced County's Vernal Pool Grasslands* (pp. 191–230). Vollmar Consulting.

Landis, J. D., & Reilly, M. (2004). *How We Will Grow: Baseline Projections of the Growth of California's Urban Footprint through the Year 2100*.

Larson-Praplan, S. (2014). History of Rangeland Management in California. *Rangelands*, *36*(5), 11–17.

Lemm, J. (2006). *Field guide to amphibians and reptiles of the San Diego region*. University of California Press.

Liu, X., Rohr, J. R., & Li, Y. (2013). Climate, vegetation, introduced hosts and trade shape a global wildlife pandemic. *Proceedings of the Royal Society B: Biological Sciences*, *280*(1753), 20122506.

Luedtke, J. A., Chanson, J., Neam, K., Hobin, L., Maciel, A. O., Catenazzi, A., Borzée, A., Hamidy, A., Aowphol, A., Jean, A., Sosa-Bartuano, Á., Fong G., A., de Silva, A., Fouquet, A., Angulo, A., Kidov, A. A., Muñoz Saravia, A., Diesmos, A. C., Tominaga, A., … Stuart, S. N. (2023). Ongoing declines for the world's amphibians in the face of emerging threats. *Nature*, 1–7.

Marantelli, G., Berger, L., Speare, R., & Keegan, L. (2004). Distribution of the amphibian chytrid Batrachochytrium dendrobatidis and keratin during tadpole development. *Pacific Conservation Biology*, *10*(3), 173.

Martel, A., Blooi, M., Adriaensen, C., Van Rooij, P., Beukema, W., Fisher, M. C., Farrer, R. A., Schmidt, B. R., Tobler, U., Goka, K., Lips, K. R., Muletz, C., Zamudio, K. R., Bosch, J., Lotters, S., Wombwell, E., Garner, T. W. J., Cunningham, A. A., Spitzen-van Der Sluijs, A., … Pasmans, F. (2014). Recent introduction of a chytrid fungus endangers Western Palearctic salamanders. *Science*, *346*(6209), 630–631.

Marty, J. T. (2004). Effects of Cattle Grazing on Diversity in Ephemeral Wetlands. *Conservation Biology*, *19*(5), 1626–1632.

Mattoni, R., & Longcore, T. (1997). The Los Angeles Coastal Prairie, a Vanished Community. *Crossosoma*, *23*, 71–102.

McKinney, M. L. (2002). Urbanization, Biodiversity, and Conservation. *BioScience*, *52*(10), 883.

Michaels, J. S., Tate, K. W., & Eviner, V. T. (2022). Vernal pool wetlands respond to livestock grazing, exclusion and reintroduction. *Journal of Applied Ecology*, *59*(1), 67–78.

Montrone, A., Saito, L., Weisberg, P. J., Gosejohan, M., Merriam, K., & Mejia, J. F. (2019). Climate change impacts on vernal pool hydrology and vegetation in northern California. *Journal of Hydrology*, *574*, 1003–1013.

Morey, S. R. (1998). Pool duration influences age and body mass at metamorphosis in the western spadefoot toad: Implications for vernal pool conservation. In C. W. Witham, E. T. Bauder, D. Bek, W. R. Feren Jr., & R. Ornduff (Eds.), *Ecology, Conservation, and Management of Vernal Pool Ecosystems—Proceedings from a 1996 Conference* (pp. 86–91). California Native Plant Society.

Morey, S. R. (2005). Spea hammondii (Baird, 1859, “1857”): Western Spadefoot. In M. Lannoo (Ed.), *Amphibian Declines: The Conservation Status of United States Species* (pp. 514–517). University of California Press.

Morey, S. R., & Guinn, D. A. (1992). Activity Patterns, Food Habits, and Changing Abundance in a Community of Vernal Pool Amphibians. *Endangered and Sensitive Species of the San Joaquin Valley, California: Their Biology, Management, and Conservation*. San Joaquin Valley Endangered Species Conference, California State University, Bakersfield.

Morey, S. R., & Reznick, D. (2001). Effects of larval density on postmetamorphic spadefoot toads (Spea Hammondii). *Ecology*, *82*(2), 510–522.

Morey, S. R., & Reznick, D. N. (2004). The relationship between habitat permanence and larval development in California spadefoot toads: Field and laboratory comparisons of developmental plasticity. *Oikos*, *104*, 172–190.

National Fish and Wildlife Foundation. (2018). *Sacramento District California In-Lieu Fee Program Compensation Planning Framework*.

National Research Council. (2001). *Compensating for Wetland Losses Under the Clean Water Act*. National Academies Press.

Neal, K. M. (2019). *An Integrative Population and Landscape Genomic Approach to Conservation of a Threatened California Amphibian at Multiple Spatial Scales*. University of California Los Angeles.

Neal, K. M., Fisher, R. N., Mitrovich, M. J., & Shaffer, H. B. (2020). Conservation Genomics of the Threatened Western Spadefoot, *Spea hammondii*, in Urbanized Southern California. *Journal of Heredity*, *111*(7), 613–627.

Neal, K. M., Johnson, B. B., & Shaffer, H. B. (2018). Genetic structure and environmental niche modeling confirm two evolutionary and conservation units within the western spadefoot (Spea hammondii). *Conservation Genetics*.

Noss, R., LaRoe, E. T., & Scott, J. M. (1995). *Endangered Ecosystems of the United States: A Preliminary Assessment of Loss and Degradation*. US Department of the Interior, National Biological Service.

O’Leary, J. F. (1995). Coastal sage scrub: Threats and current status. *Fremontia*, *23*(4), 27–31.

Olson, D. H., Aanensen, D. M., Ronnenberg, K. L., Powell, C. I., Walker, S. F., Bielby, J., Garner, T. W. J., Weaver, G., The Bd Mapping Group, & Fisher, M. C. (2013). Mapping the Global Emergence of Batrachochytrium dendrobatidis, the Amphibian Chytrid Fungus. *PLoS ONE*, *8*(2), e56802.

Padgett-Flohr, G. E., & Hopkins II, R. L. (2009). Batrachochytrium dendrobatidis, a novel pathogen approaching endemism in central California. *Diseases of Aquatic Organisms*, *83*.

Pfennig, D. (1990). The adaptive significance of an environmentally-cued developmental switch in an anuran tadpole. *Oecologia*, *85*, 101–107.

Pyke, C. R. (2004). Habitat Loss Confounds Climate Change Impacts. *Frontiers in Ecology and the Environment*, *2*(4), 178–182.

Pyke, C. R. (2005). Assessing Climate Change Impacts on Vernal Pool Ecosystems and Endemic Branchiopods. *Ecosystems*, *8*(1), 95–105.

Pyke, G. H. (2008). Plague Minnow or Mosquito Fish? A Review of the Biology and Impacts of Introduced Gambusia Species. *Annual Review of Ecology, Evolution, and Systematics*, *39*(1), 171–191.

Rahn, M. E., Doremus, H., & Diffendorfer, J. (2006). Species Coverage in Multispecies Habitat Conservation Plans: Where's the Science? *BioScience*, *56*(7), 613.

Reeder, N. M. M., Pessier, A. P., & Vredenburg, V. T. (2012). A Reservoir Species for the Emerging Amphibian Pathogen Batrachochytrium dendrobatidis Thrives in a Landscape Decimated by Disease. *PLoS ONE*, *7*(3), e33567.

Richardson, J. S., & Oaten, D. (2013). *Critical breeding, foraging, and overwintering habitats of Great Basin spadefoot toads (Spea intermontana) and western toads (Anaxyrus boreas) within grassland ecosystems: 2013 final report.* Prepared for Can. Wildl. Fed., Kanata, ON.

Richmond, J. Q., Backlin, A. R., Tatarian, P. J., Solvesky, B. G., & Fisher, R. N. (2014). Population declines lead to replicate patterns of internal range structure at the tips of the distribution of the California red-legged frog (Rana draytonii). *Biological Conservation*, *172*, 128–137.

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

Riordan, E. C., Montalvo, A. M., & Beyers, J. L. (2018). *Using species distribution models with climate change scenarios to aid ecological restoration decisionmaking for southern California shrublands* (No. PSW-RP-270; p. PSW-RP-270). U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station.

Robins, J. D., & Vollmar, J. E. (2002). Chapter 11: Livestock Grazing and Vernal Pools. In *Wildlife and Rare Plant Ecology of Eastern Merced County's Vernal Pool Grasslands*. Vollmar Consulting.

Román-Palacios, C., & Wiens, J. J. (2020). Recent responses to climate change reveal the drivers of species extinction and survival. *Proceedings of the National Academy of Sciences*, *117*(8), 4211–4217.

Rose, J. P., Halstead, B. J., & Fisher, R. N. (2020). Integrating multiple data sources and multi-scale land-cover data to model the distribution of a declining amphibian. *Biological Conservation2*, *241*, 108374–108374.

Rose, J. P., Halstead, B. J., Packard, R. H., & Fisher, R. N. (2022). Projecting the remaining habitat for the western spadefoot (Spea hammondii) in heavily urbanized southern California. *Global Ecology and Conservation*, *33*, e01944.

Rosenblum, E. B., Voyles, J., Poorten, T. J., & Stajich, J. E. (2010). The Deadly Chytrid Fungus: A Story of an Emerging Pathogen. *PLoS Pathogens*, *6*(1), e1000550.

Ruibal, R., Tevis, L., & Roig, V. (1969). The terrestrial ecology of the spadefoot toad Scaphiopus hammondii. *Copeia*, *3*, 571–584.

Samson, F. B., Knopf, F. L., & Ostlie, W. R. (2004). Great Plains ecosystems: Past, present, and future. *Wildlife Society Bulletin*, *32*(1), 6–15.

Scheffers, B. R., De Meester, L., Bridge, T. C. L., Hoffmann, A. A., Pandolfi, J. M., Corlett, R. T., Butchart, S. H. M., Pearce-Kelly, P., Kovacs, K. M., Dudgeon, D., Pacifici, M., Rondinini, C., Foden, W. B., Martin, T. G., Mora, C., Bickford, D., & Watson, J. E. M. (2016). The broad footprint of climate change from genes to biomes to people. *Science*, *354*(6313), aaf7671-1-aaf7671-11.

Schloegel, L. M., Toledo, L. F., Longcore, J. E., Greenspan, S. E., Vieira, C. A., Lee, M., Zhao, S., Wangen, C., Ferreira, C. M., Hipolito, M., Davies, A. J., Cuomo, C. A., Daszak, P., & James, T. Y. (2012). Novel, panzootic and hybrid genotypes of amphibian chytridiomycosis associated with the bullfrog trade. *Molecular Ecology*, *21*(21), 5162–5177.

Searcy, C. A., Gabbai-Saldate, E., & Bradley Shaffer, H. (2013). Microhabitat use and migration distance of an endangered grassland amphibian. *Biological Conservation*, *158*, 80–87.

Semlitsch, R. D., & Bodie, J. R. (2003). Biological criteria for buffer zones around wetlands and riparian habitats for amphibians and reptiles. *Conservation Biology*, *17*(5), 1219–1228.

Sette, C. M., Vredenburg, V. T., & Zink, A. G. (2015). Reconstructing historical and contemporary disease dynamics: A case study using the California slender salamander. *Biological Conservation*, *192*, 20–29.

Shore, T. (2001). *Off-Road to Ruin: How Motorized Recreation is Unraveling California's Landscapes*. California Wilderness Coalition.

Skerratt, L. F., Berger, L., Speare, R., Cashins, S., McDonald, K. R., Phillott, A. D., Hines, H. B., & Kenyon, N. (2007). Spread of Chytridiomycosis Has Caused the Rapid Global Decline and Extinction of Frogs. *EcoHealth*, *4*(2), 125.

Sleeter, B. M., Wilson, T. S., Sharygin, E., & Sherba, J. T. (2017). Future Scenarios of Land Change Based on Empirical Data and Demographic Trends. *Earth's Future*, *5*(11), 1068–1083.

Sleeter, B. M., Wilson, T. S., Soulard, C. E., & Liu, J. (2011). Estimation of late twentieth century land-cover change in California. *Environmental Monitoring and Assessment*, *173*(1–4), 251–266.

Smallwood, K. S., Wilcox, B., Leidy, R., & Yarris, K. (1998). Indicators Assessment for Habitat Conservation Plan of Yolo County, California, USA. *Environmental Management*, *22*(6), 947–958.

Smith, M. A., & Green, D. M. (2005). Dispersal and the metapopulation paradigm in amphibian ecology and conservation: Are all amphibian populations metapopulations? *Ecography*, *28*(1), 110–128.

Smith, S. A., Monsen-Collar, K. J., Green, D. E., Niederriter, H. S., Hall, M. L., Terrell, K. A., Gipe, K. D., Urban, C. A., Patterson, C. A., Seigel, R. A., Zarate, B., Kleopfer, J. D., Campbell-Grant, E. H., & Driscoll, C. P. (2016). Detecting the Extent of Mortality

Events from Ranavirus in Amphibians of the Northeastern U.S. *Report to the Northeast Association of Fish and Wildlife Agencies (NEAFWA) for Regional Conservation Needs (RCN) Grant #2012-01*.

Sodhi, N. S., Bickford, D., Diesmos, A. C., Lee, T. M., Koh, L. P., Brook, B. W., Sekercioglu, C. H., & Bradshaw, C. J. A. (2008). Measuring the Meltdown: Drivers of Global Amphibian Extinction and Decline. *PLoS ONE*, *3*(2), e1636.

State Water Resources Control Board. (2023a, May 5). *State Water Board: U.S. Supreme Court decision decreases federal wetlands protection*.

State Water Resources Control Board. (2023b, October 23). *Frequently Asked Questions regarding the U.S. Supreme Court's ruling in Sackett v. EPA*.

Stebbins, R. C. (2003). *Western Reptiles and Amphibians* (3rd edition). Houghton Mifflin Company.

Stebbins, R. C., & McGinnis, S. M. (2012). Western spadefoot (Spea hammondii). In *Field Guide to Amphibians and Reptiles of California* (Revised edition, pp. 156–158). University of California Press.

Stegen, G., Pasmans, F., Schmidt, B. R., Rouffaer, L. O., Van Praet, S., Schaub, M., Canessa, S., Laudelout, A., Kinet, T., Adriaensen, C., Haesebrouck, F., Bert, W., Bossuyt, F., & Martel, A. (2017). Drivers of salamander extirpation mediated by Batrachochytrium salamandrivorans. *Nature*, *544*(7650), 353–356.

Storer, T. I. (1925). *A Synopsis of the Amphibia of California*.

Storfer, A., Alfaro, M. E., Ridenhour, B. J., Jancovich, J. K., Mech, S. G., Parris, M. J., & Collins, J. P. (2007). Phylogenetic concordance analysis shows an emerging pathogen is novel and endemic. *Ecology Letters*, *10*(11), 1075–1083.

Stromberg, M. R., Corbin, J. D., & D'Antonio, C. (Eds.). (2007). *California grasslands: Ecology and management*. University of California Press.

Stuart, S. N., Chanson, J. S., Cox, N. A., Young, B. E., Rodrigues, A. S. L., Fischman, D. L., & Waller, R. W. (2004). Status and trends of amphibian declines and extinctions worldwide. *Science*, *306*(5702).

Tanner, W. W. (1989). Amphibians of Western Chihuahua. *Great Basin Naturalist*, *49*(1), 38–70.

Taylor, M. F. J., Suckling, K. F., & Rachlinski, J. J. (2005). The Effectiveness of the Endangered Species Act: A Quantitative Analysis. *BioScience*, *55*(4), 360.

Thomson, R. C., Wright, A. N., & Shaffer, H. B. (2016). *California amphibian and reptile species of special concern*. University of California Press.

Timm, B. C., McGarigal, K., & Cook, R. P. (2014). Upland Movement Patterns and Habitat Selection of Adult Eastern Spadefoots (Scaphiopus holbrookii) at Cape Cod National Seashore. *Journal of Herpetology*, *48*(1), 84–97.

Traill, L. W., Brook, B. W., Frankham, R. R., & Bradshaw, C. J. A. (2010). Pragmatic population viability targets in a rapidly changing world. *Biological Conservation*, *143*, 28–34.

Trisos, C. H., Merow, C., & Pigot, A. L. (2020). The projected timing of abrupt ecological disruption from climate change. *Nature*, *580*, 496–501.

Trombulak, S. C., & Frissell, C. A. (2000). Review of ecological effects of roads on terrestrial and aquatic communities. *Conservation Biology*, *14*(1), 18–30.

United State Department of Agriculture, & Farm Service Agency. (2011). *California State Fact Sheet*.

US Fish and Wildlife Service. (2005). *Recovery Plan for Vernal Pool Ecosystems of California and Southern Oregon*.

US Fish and Wildlife Service & US National Marine Fisheries Service. (2016). *Habitat Conservation Planning and Incidental Take Permit Processing Handbook*.

U.S. Fish and Wildlife Services. (2015). Endangered and threatened wildlife and plants; 90-day findings on 31 petitions. *Federal Register*, *80*(126), 37578–37578.

Voyles, J., Vredenburg, V. T., Tunstall, T. S., Parker, J. M., Briggs, C. J., & Rosenblum, E. B. (2012). Pathophysiology in Mountain Yellow-Legged Frogs (Rana muscosa) during a Chytridiomycosis Outbreak. *PLoS ONE*, *7*(4), e35374.

Vredenburg, V. T., McNally, S. V. G., Sulaeman, H., Butler, H. M., Yap, T. A., Koo, M. S., Schmeller, D. S., Dodge, C., Cheng, T., Lau, G., & Briggs, C. J. (2019). Pathogen invasion history elucidates contemporary host pathogen dynamics. *PLoS ONE*, *14*(9), 1–14.

Wang, I. J., Johnson, J. R., Johnson, B. B., & Shaffer, B. B. (2011). Effective population size is strongly correlated with breeding pond size in the endangered California tiger salamander, Ambystoma californiense. *Conservation Genetics*, *12*, 911–920.

Warren, R., Price, J., Fischlin, A., de la Nava Santos, S., & Midgley, G. (2011). Increasing impacts of climate change upon ecosystems with increasing global mean temperature rise. *Climatic Change*, *106*(2), 141–177.

Weiss, S. B., & Longcore, T. (2020). *Effects of nitrogen deposition on sensitive species and habitats resulting from the Southern California Association of Governments Regional Transportation Plan*.

Wiens, J. J. (2016). Climate-related local extinctions are already widespread among plant and animal species. *PLoS Biology*, *14*(12), 1–18.

Wiens, J. J., & Titus, T. A. (1991). A phylogenetic analysis of Spea (Anura: Pelobatidae). *Herpetologica*, *47*(1), 21–28.

Willson, J. D., & Dorcas, M. E. (2003). Effects of Habitat Disturbance on Stream Salamanders: Implications for Buffer Zones and Watershed Management. *Conservation Biology*, *17*(3), 763–771.

Witham, C. W. (2021). *Changes in the Distribution of Great Valley Vernal Pool Habitats from 2005 to 2018*. San Francisco Estuary Institute.

Yap, T. A., Koo, M. S., Ambrose, R. F., & Vredenburg, V. T. (2018). Introduced bullfrog facilitates pathogen invasion in the western United States. *PLOS ONE*, *13*(4), e0188384.

Yap, T. A., Koo, M. S., Ambrose, R. F., Wake, D. B., & Vredenburg, V. T. (2015). Averting a North American biodiversity crisis. *Science*, *349*(6247), 481–482.

Yap, T. A., Nguyen, N. T., Serr, M., Shepack, A., & Vredenburg, V. T. (2017). Batrachochytrium salamandrivorans and the Risk of a Second Amphibian Pandemic. *EcoHealth*, *14*(4), 851–864.

Yap, T. A., Prabhala, A., & Anderson, I. (2023). *Hidden in Plain Sight: California's Native Habitats are Valuable Carbon Sinks*. Center for Biological Diversity.