

Species Status Assessment Report
for
Northwestern Pond Turtle (*Actinemys marmorata*)
and
Southwestern Pond Turtle (*Actinemys pallida*)



U.S. Fish and Wildlife Service

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EXECUTIVE SUMMARY

The U.S. Fish and Wildlife Service (Service) was petitioned to list 53 species of reptiles and amphibians, including the western pond turtle (*Actinemys marmorata*), as threatened or endangered under the Endangered Species Act of 1973, as amended (16 U.S.C. 1531–1543), in July 2012 by the Center for Biological Diversity. On April 10, 2015, we published a 90-day finding that the petition presented substantial scientific or commercial information indicating that listing may be warranted for the western pond turtle (80 FR 19262–19263). Since then, the western pond turtle was split into two separate species, the northwestern pond turtle (*Actinemys marmorata*) and southwestern pond turtle (*Actinemys pallida*). Our review assesses the status of each species separately and we will issue a 12-month finding for both the northwestern pond turtle and southwestern pond turtle.

This report summarizes the results of the Species Status Assessment (SSA) for both the northwestern and southwestern pond turtles. The SSA begins with a compilation of the best available biological information on each species and each species' habitat and demographic needs. The SSA then evaluates the threats facing the two species and provides an assessment of the two species' current and future viability based on the conservation biology principles of resiliency, redundancy, and representation. To assess the viability of the northwestern and southwestern pond turtles, two scenarios were considered, representing the range of plausible future environmental conditions based on the best available science. Viability was assessed at three timesteps (2050, 2075, and 2100) .

The historical range of western pond turtles extends along the Pacific coast from British Columbia, Canada to the northern part of Baja California, Mexico, primarily west of the Sierra Nevada and Cascade ranges. The current range of the northwestern pond turtle includes populations from the San Joaquin Valley north, all populations in California north of the middle of Monterey Bay, the Coastal and Cascade Ranges of Oregon and Washington State, and an outlying population in Nevada. The current range of the southwestern pond turtle is restricted to those populations inhabiting the central Coast Range south from the middle of Monterey Bay to the species' southern range boundary in Baja California.

Habitat needs for western pond turtles include aquatic habitat, upland habitat, and basking sites. The demographic needs are abundance, reproduction/recruitment, survival, and connectivity. We assess the threats of habitat loss and fragmentation, altered hydrology, predation, nonnative species competition, disease, road impacts, collection, contaminants, and climate change.

Assessment of past, current, and future influences on northwestern and southwestern pond turtle requirements for viability revealed the following as primary threats to the species: anthropogenic impacts (human modification/land conversion), predation, and drought. To assess the current and

future condition of the two species, we separated their ranges into analysis units that incorporate genetic, management, and ecological data. We identified 14 analysis units for northwestern pond turtle and six analysis units for southwestern pond turtle. When assessing condition of the two species, we present available information on presence and primary threats for each species. We then present results from a stochastic stage-based population model under two plausible scenarios through 2050, 2075, and 2100. Environmental conditions for the two modeled scenarios are based on the best available projections for changes in primary threats that drive western pond turtle population dynamics. Datasets used for anthropogenic influences and climate changes go out to 2100. The two scenarios are similar but represent upper and lower bounds of climate projections (Representative Concentration Pathway (RCP) 8.5 and RCP 4.5).

For northwestern pond turtles in Washington, we present data from the most recent population site estimates, and results from an existing population viability analysis (PVA) that is specific to the demographic parameters for Washington western pond turtles. For southwestern pond turtles, data specific to the Baja California analysis unit are lacking, and therefore Baja was not included in the model.

Northwestern pond turtle

In the near term, northwestern pond turtle analysis units have high probability of persistence through 2050 (i.e., 85 to 95 percent likelihood of persistence in the next approximately 25 years), indicating that the species currently maintains resiliency, defined as the ability to withstand stochastic events. Analysis units in Washington are conservation reliant, depending on headstarting, but the PVA suggests persistence through 2050. In terms of redundancy, the species is currently distributed across four states, and maintains ability to withstand most catastrophic events, particularly in the central portion of the range. In terms of representation, northwestern pond turtles exhibit ecological flexibility but have evidence of low genetic diversity in a large portion of the species' range. Taken together, this indicates that they currently maintain some ability to adapt to changing environmental conditions in the near term.

In the future, northwestern pond turtles in Washington decline rapidly towards extinction without the headstarting program. In California, Oregon, and Nevada analysis unit, chance of extinction generally follows a latitudinal trend with more southern analysis units having lower resiliency into the future, due to impacts from land conversion, bullfrog predation, and increasing drought. In 2075, probability of extinction ranges from 28 percent to 41 percent under the two scenarios, indicating that persistence is more likely than extinction in all analysis units (i.e., likelihood of extinction is less than 50 percent) for the next approximately 50 years, and thus ability to withstand stochastic events is largely maintained. By 2100, probability of extinction ranges from 46 to 60 percent under both scenarios, with 5 to 7 of the 14 analysis units more likely than not to become functionally extinct, indicating ability to withstand stochastic events becomes compromised.

Redundancy and representation of northwestern pond turtles would be reduced with extirpation of any analysis units. While Washington requires management intervention and the southern parts of the range have higher probability of extinction, persistence is more likely than extirpation in all analysis units in 2075. Therefore, the northwestern pond turtle likely maintains most of its ability to withstand most catastrophic events and ability to adapt to changing conditions over the next approximately 50 years. By 2100, the species is likely to suffer extirpations and therefore will be less likely to withstand catastrophic events and adapt to changing conditions under either future scenario.

In the future, the northwestern pond turtle has an increasing risk of extinction over time from stochasticity, catastrophic events, and inability to adapt to changing conditions. In 2075, abundances will decrease over time, but persistence of analysis units is more likely than not. By 2100, abundances decrease further and 5 to 7 of the 14 analysis units are more likely to become functionally extinct than to persist, indicating increasing species-level extinction risk in the next approximately 75 years.

Southwestern pond turtle

In the near term, the probability of extinction of southwestern pond turtles ranges from 20 to 25 percent in 2050. Low probability of extinction indicates that the analysis units currently maintain ability to withstand stochastic events. In terms of redundancy, southwestern pond turtles are currently distributed across the central coast and southern California, and Baja California, Mexico. None of the U.S. analysis units are more likely than not to be extinct by 2050, which indicates that the species maintains most ability to withstand most catastrophic events. In terms of representation, southwestern pond turtles exhibit ecological flexibility in habitat use, but exhibit high inbreeding levels across much of the range. Taken together, this indicates that they currently maintain some ability to adapt to changing environmental conditions in the near term.

In the future, southwestern pond turtle resiliency is likely to be highly reduced with extinction being more likely than persistence in all analysis units in the next approximately 50 years. Across analysis units, chance of extinction is over 50 percent by 2075 and over 70 percent by 2100. Redundancy and representation of southwestern pond turtles would be reduced with extirpation of any analysis units. Given that extinction of all analysis units is more likely than not under both scenarios and time steps, the species will be less likely to withstand catastrophic events or to adapt to near-term and long-term changes in its physical and biological environments in the next approximately 50 to 75 years.

Overall, in the future at both 2075 and 2100, the southwestern pond turtle will have substantial increasing risk of extinction from stochasticity, catastrophic events, and inability to adapt to changing conditions.

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1.0 INTRODUCTION

We, the U.S. Fish and Wildlife Service (Service), in this Species Status Assessment (SSA) report, review the biological status of the northwestern pond turtle (*Actinemys marmorata*) and southwestern pond turtle (*Actinemys pallida*), collectively referred to as the western pond turtle, in response to a petition (Center for Biological Diversity (CBD) 2012) to list the species under the Endangered Species Act of 1973, as amended (Act). Based on the best scientific information available for western pond turtles at this time, we assess the current and future status of northwestern pond turtles and southwestern pond turtles as two separate species in this SSA. Because of the relatively recent split into two separate entities (see Taxonomy and Genetics sections), the majority of available research and information refers to a single species (western pond turtle). In this document, any reference to aspects of the western pond turtle biological or ecological condition is understood to apply to either the northwestern or southwestern pond turtle depending on geographic location.

We use the SSA Framework to assess each species' biological status for the purpose of informing our decisions under the Act (Service 2016, entire). The SSA report is a compilation of the best scientific and commercial data available on each species (taxonomy, life history, and habitat) and their habitat and demographic needs. The SSA report describes the current condition of each species' habitat and demographics, and the probable explanations for past and current changes in abundance and distribution within each species' ecological settings (areas representative of geographic, genetic, or life history variation across the range of the species). Lastly, the SSA report provides information on each species' potential response to plausible future environmental conditions through future scenarios (Rowland *et al.* 2016, entire). Overall, the SSA report uses the conservation biology principles of resiliency, redundancy, and representation (collectively known as the "3Rs" (Shaffer and Stein 2000, pp. 308–311) as a lens through which we can evaluate the current and future condition of the species (Smith *et al.* 2018, entire). Ultimately, an SSA characterizes a species' ability to sustain populations in the wild over time based on the best scientific understanding of current and future abundance and distribution within the species' ecological settings.

An SSA report is a summary of the information about a species and, simultaneously, a biological risk assessment to aid decision makers who must use the best scientific and commercial data available to make policy-guided decisions. The SSA report provides decision makers with a scientifically rigorous characterization of the species' biological and conservation status, focusing on the likelihood of whether the species' will sustain populations within its ecological settings while also explicitly acknowledging uncertainties in that characterization. The SSA report does not result in a decision directly, but it provides the best scientific and commercial data available for comparison to policy standards to guide decisions under the Act.

1.1 Previous Federal Action

On July 11, 2012, we received a petition from CBD (2012), requesting that 53 species of amphibians and reptiles, including western pond turtle, be listed as an endangered or threatened species and that critical habitat be designated for those species under the Act. On June 10, 2014, CBD sent us a letter which cited a publication (Spinks et al. 2014, entire) recommending the species be split into two separate species and requested that we review the two separate species as part of their petition (CBD 2015, entire). On April 10, 2015, we published a 90-day finding in the Federal Register affirming that the petition for the western pond turtle presented substantial scientific or commercial information indicating that the petitioned action may be warranted (80 FR 19259–19263).

2.0 METHODOLOGY

This document draws scientific information from resources such as primary peer-reviewed literature, reports submitted to the Service and other public agencies, species occurrence information in Geographic Information Systems (GIS) databases, and expert experience, research, and observations. It is preceded by and draws upon analyses presented in other Service documents, including the 90-day finding (80 FR 37568–37579). Finally, we coordinated closely with our partners engaged in ongoing western pond turtle research and conservation efforts including wildlife professionals from Federal and State agencies, universities, and private entities. This assures consideration of the most current scientific and commercial data available regarding the status and conservation of the western pond turtle. Throughout this report, we refer to information on the western pond turtle (which includes southwestern and northwestern pond turtle). Where necessary, we refer to each species separately as either southwestern or northwestern pond turtle.

2.1 Analytical Framework

The SSA framework (Service 2016, entire) summarizes the information assembled and reviewed by the Service, incorporating the best scientific and commercial data available, to conduct an in-depth review of a species' biology and threats, evaluate its biological status, and assess its resources and conditions needed to sustain populations in the wild over time (viability). To do so, we use the conservation biology principles of resiliency, redundancy, and representation as a lens to evaluate the current and future condition of the two species (Smith et al. 2018, entire). In our assessment, we evaluated the condition of western pond turtle populations in the wild at 2050, 2075, and 2100. The 2100 timeframe (approximately 75 years) represents estimates of end of the century climate projections and encompasses approximately 3 western pond turtle generations (25 years).

3.0 SPECIES BACKGROUND

3.1 Taxonomy

Western pond turtles are currently identified as being in the family Emydidae; Class: Reptilia, Order: Testudines, Suborder: Cryptodira, and Superfamily: Testudinoidea. The species was first identified in 1852 as *Emys marmorata* from specimens collected from Puget Sound, Washington (Baird and Girard 1852, pp. 174–177). Past taxonomy for the western pond turtle is further detailed in Bury et al. (2012, pp. 1–3), including designation as species or subspecies; and classification in the genera *Emys*, *Actinemys*, or *Clemmys*. There has also been suggestion of three morphologically distinct groups (Holland 1994, p. 2-3) or four distinct clades based on genetic variation, with three occurring south of San Luis Obispo (Spinks and Shaffer 2005, entire). In recent publications, the genus name is interchanged between *Emys* and *Actinemys* with several authorities placing the two species in the more inclusive *Emys* or the more narrowly defined *Actinemys* (Stephens and Wiens 2003, p. 596; Fritz et al. 2011, entire; Spinks et al. 2016, entire; Turtle Taxonomy Working Group et al. 2017, pp. 75–76). Because the genus name is interchanged between *Emys* and *Actinemys*, the species names may be seen in both forms as well.

Spinks et al. (2014, entire) recommended splitting the western pond turtle into two separate species, and this split was recognized in taxonomic lists in 2017 (Crother 2017, p. 82; Turtle Taxonomy Working Group et al. 2017, p. 76). The current authoritative list of the subject, the Turtle Taxonomy Working Group checklist, refers to the two species as northwestern pond turtle (*Actinemys marmorata*) and southwestern pond turtle (*Actinemys pallida*) (Turtle Taxonomy Working Group et al. 2021, pp. 171–172). Based on the recognition by the scientific community, and in following with the Society for the Study of Amphibians and Reptiles and the Turtle Taxonomy Working Group, the Service recognizes northwestern pond turtle and southwestern pond turtle as separate species. Some common names that are associated with western pond turtles are: Pacific pond turtle, Pacific mud turtle, Pacific terrapin, and western mud turtle (Bettelheim 2005, p. 26).

3.2 Species Description

The western pond turtle is a medium-sized turtle (Figure 1). Size varies geographically, with the largest animals occurring in the northern part of the range (Holland 1994, pp. 2-3). The maximum carapace (shell) length (CL) of northwestern pond turtles is 241 millimeters (mm) (Lubcke and Wilson 2007, p. 110), and maximum CL of southwestern pond turtles is 179 mm (Germano and Riedle 2015, p. 104). Northwestern pond turtle adults typically range in size between 160 to 180 mm long and weigh between 500 to 700 grams (Bury et al. 2012, p. 4), while southwestern pond turtles range from 110 to 179 mm long and weigh between 194 to 828 grams (Germano and Riedle 2015, p. 104).

Western pond turtles are sexually dimorphic: females tend to have a smaller head, less angled snout, taller and rounder carapace, flat (rather than concave) plastron (underside of shell), and thinner tail as compared to males (Holland 1994, pp. 2-4; Rosenberg et al. 2009, p. 10). Colors and markings vary geographically and by age with most appearing olive to dark brown, or blackish, occasionally without pattern but usually with a network of spots, lines, or dashes of brown or black that often radiate from growth centers of shields (Bury et al. 2012, p. 4; Stebbins and McGinnis 2018, pp. 204–205). The plastron is yellowish, blotched with blackish or dark brown, and occasionally unmarked (Stebbins and McGinnis 2018, p. 204). Coloration of the head and neck vary by sex, geography, and age (Hays et al. 1999, p. 2). Males usually have a light-yellowish chin and underside of the throat whereas females tend to have dark prints or rosette rings that usually remain throughout their life. Hatchlings are generally a brown-olive color with visible mottling on the head and neck (Hays et al. 1999, p. 2) that darken with age. Hatchlings are 25 to 31 mm long carapace length (CL) (approximately the size of an American quarter) and weigh 3 to 7 grams at the time of emergence (Bury et al. 2012, pp. 4, 17). The shell of hatchlings is soft and pliable, and the tail is nearly as long as the shell (Ashton et al. 1997, p. 3; Stebbins and McGinnis 2018, p. 205). The shell becomes fairly hard around three to four years of age (Bury et al. 2012, p. 4). Eggs are off-white, elliptical-oval shaped, and range from 32 to 42 mm long and from 18 to 25 mm in diameter (Bury et al. 2012, p. 15).



Figure 1. Northwestern pond turtle (Photo: Adam Clause) and southwestern pond turtle (Photo: USFWS)

Seeliger (1945, entire) first proposed geographic differentiation of western pond turtles into northern and southern subspecies based on differences in coloration and the presence and shape of the inguinal scute, the plate where the carapace joins the plastron at the groin (Figure 2). Distinguishing characteristics of the northern group included the presence of relatively large triangular inguinal scutes and darker sides of the neck that contrasted with the lighter underside, whereas the southern group had reduced (34 percent of individuals) or absent (60 percent of individuals) inguinal scutes and similar coloration on the sides and underside of the neck, and an intergrade zone was identified in central California (Seeliger 1945, pp. 155–159). Recent genetic results corroborate the morphologic distinctiveness (presence/absence of inguinal scutes) as one

of the components differentiating northwestern and southwestern pond turtles (Shaffer and Scott 2022, p. 9).



Figure 2. *Inguinal scute in Actinemys marmorata (left); no inguinal scute Actinemys pallida (right) (Photos: Brad Shaffer).*

Western pond turtles sometimes co-occur in Oregon and Washington with the western painted turtle (*Chrysemys picta bellii*) (Figure 3). Western pond turtles are distinguished from the western painted turtles by lacking the yellow, orange, or red lines on the neck and legs (Oregon Department of Fish and Wildlife (ODFW) 2015, p. 3; Stebbins and McGinnis 2018, p. 205). Western pond turtles also co-occur with the nonnative red-eared slider (*Trachemys scripta elegans*) throughout portions of its range in the U.S. (Somma et al. 2022, entire) (Figure 4). Many red-eared sliders can be distinguished from western pond turtles by their broad red to orange stripe behind each eye. Western pond turtles are distinguished from melanistic forms (dark or uncolored forms) of red-eared sliders by lacking a serrated rim around the posterior edge of the carapace (Thomson et al. 2016, p. 298).



Figure 3. Western painted turtle (Photo: Sam Stukel, Service).



Figure 4. Red-eared slider (Photo: Steve Hillebrand, Service).

3.3 Genetics

Molecular analyses for western pond turtles were first conducted in the mid-1990s, with results generally following Seeliger's long-held subspecies designations based on coloration and morphological variation. Genetically unique variants, and possibly different taxa, were found in the Columbia River and isolated populations in southern California and Baja California (Gray 1995, entire; Janzen et al. 1997, entire; Ernst and Lovich 2009, pp. 174–175). Spinks and Shaffer (2005, entire) analyzed mitochondrial DNA from specimens throughout the range and found a distinct northern clade (group) with little genetic variation from the vicinity of San Francisco Bay and northward, but high variation in turtles from southern California and the Central Valley, California. The northern clade overlapped what has been previously described as the

northwestern pond turtle subspecies (Seeliger 1945, p. 156), but the intergrade zone and the southwestern “subspecies” appeared to form several distinct clades. At the time, it seemed there could be four clades within this species, but the authors did not make taxonomic recommendations based on these data (Spinks and Shaffer 2005, entire).

Subsequent analyses corroborated deep divergence between the northern and southern groups, with the divide occurring where there was a major prehistoric marine embayment in central California (Spinks et al. 2010, entire). Analysis using nuclear DNA supported two clades that matched Seeliger (1945) with a level of differentiation warranting separate species within the genus *Emys* (Spinks et al. 2014, entire). As a result of this multi-locus molecular genetic analyses, the western pond turtle is now described as two distinct species: *Emys (Actinemys) marmorata* and *E. (A.) pallida* (Spinks et al. 2014, entire). As mentioned above, in following with the Society for the Study of Amphibians and Reptiles, the Service recognizes these two as separate species.

Most recently, genetic analysis further supported the distinctiveness between northwestern and southwestern pond turtles (Shaffer and Scott 2022, entire). Using genomic data, which included increased sampling and a larger genetic dataset compared to previous genetic studies, the authors added clarity to the range of northwestern and southwestern pond turtles, especially in areas that had been identified as intergrade areas in previous publications (Seeliger 1945, p. 158; Spinks et al. 2014, p. 2234) (see Chapter 4, Range and Distribution). Northwestern and southwestern pond turtles are not reproductively isolated; where they come into contact, either at range borders or through human-mediated movements, the two species are able to interbreed. There is evidence of genetic mixing at the boundary between the two taxa in Contra Costa, Alameda, San Mateo, Santa Cruz, Santa Clara, San Benito, Monterey, San Luis Obispo, Kings, and far western Kern Counties. However, none of the sites in the study included pure (genetically 90% or greater) individuals of both species, suggesting that the two ranges come into contact but do not overlap (Shaffer and Scott 2022, pp. 4–6).

Clustering analyses identified variable numbers of genetic units within each species based on the criterion used. Analyses that best maximized the structure of the data resulted in more clusters than that which maximized marginal likelihood of the data; in this paragraph we present the genetic clusters based on structure of the data (Shaffer and Scott 2022, pp. 6–8). For northwestern pond turtles, the area north of San Francisco Bay was subdivided into three groups: a large area including the north California coast and Oregon clusters, one occupying the Sacramento Valley, and another including the Delta and due east across the Central Valley and Nevada. There was additional separation of a group in the Yosemite area, with the remainder of turtles clustering in a southern group including the San Joaquin Valley, the east and south San Francisco Bay Area, and San Francisco Peninsula. Southwestern pond turtle genetic clusters also largely followed a north/south geographic characterization. Clusters included: a Coast Range group in the central coast, from roughly Monterey Bay south to northern Santa Barbara County;

a Ventura/Santa Barbara cluster from Point Conception to the Santa Clara River; a Los Angeles group including the west-flowing Los Angeles basin drainages; a Mojave group from the east-flowing Mojave River Drainage; an Orange County/San Diego cluster encompassing southern coastal California from the Santa Ana river south through most of San Diego and Orange Counties; and, a Baja group covering populations south of the US/Mexico border. One sample in San Diego County fell into its own cluster, which was likely an artifact of the data.

4.0 RANGE AND DISTRIBUTION

The historical range of western pond turtles extends along the Pacific coast from British Columbia, Canada to the northern part of Baja California, Mexico, primarily west of the Sierra Nevada and Cascade ranges (Ernst and Lovich 2009, p. 173; Stebbins and McGinnis 2018, p. 205). Western pond turtles have been found at sites from brackish estuarine waters at sea level up to 2,048 meters (m) (6,719 feet (ft) (Ernst and Lovich 2009, p. 176) but mostly occur below 1,371 m (4,980 ft.) (Stebbins and McGinnis 2018, p. 205). Several isolated populations occur, including but not limited to those in the vicinity of Puget Sound, Columbia Gorge, the Mojave River in California, and the Carson and Truckee Rivers in Nevada (Holland 1994, p. 2-4). . Historical accounts from Vancouver Island and mainland British Columbia, Canada in the lower Fraser River watershed may represent transplanted individuals; no reports of the species are known from either region since 1966 (Gregory and Cambell 1984 in Ernst and Lovich 2009, p. 173), and western pond turtles are considered extirpated from British Columbia, Canada (Ministry of Environment 2012, p. iv). Single records from southwestern Idaho and Grant County, Oregon (Nussbaum et al. 1983 in Ernst and Lovich 2009, p. 173) are likely introduced (Ernst and Lovich 2009, p. 173), and other isolated populations within the species' native range may also represent introductions (Thomson et al. 2016, p. 300).

As discussed above, recent genetic information has led to a taxonomic split of the western pond turtle into two distinct species (Spinks et al. 2014, p. 2234), which is corroborated by recent genomic analyses (Shaffer and Scott 2022, entire). The range of the northwestern pond turtle includes populations from the San Joaquin Valley north, all populations in California north of the middle of Monterey Bay, the Coastal and Cascade Ranges of Oregon and Washington State, and an outlying population in Nevada (Shaffer and Scott 2022, pp. 5, 13). The range of the southwestern pond turtle is restricted to those populations inhabiting the central Coast Range south from the middle of Monterey Bay to the species' southern range boundary in Baja California. A new population found south of the nearest reported population represents a range extension of 95.5 kilometers (and the only oasis population within the Central Desert ecoregion in Baja California) (Valdez-Villavicencio et al. 2016, p. 265). However, we note that Spinks et al. (2014) states that populations from Baja California may represent a distinct species pending results from additional analyses (Spinks et al. 2014, p. 2238; Thomson et al. 2016, p. 300), which was not addressed in the recent genetic report (Shaffer and Scott 2022, entire). Additional sampling would help to determine the species' southern boundary.

Also described in the Genetics section, Shaffer and Scott (2022, entire) clarified areas of previous uncertainty immediately south, east, and west of the San Francisco Bay, where there were no specimens used in Spinks et al. (2014, p. 2233) when describing the two species, and the range around the San Francisco Bay presented in Thomson et al. (2016, p. 297). Based on these genomic data, Shaffer and Scott recommended that the border along the coast between the two species was in the middle of Monterey Bay (Shaffer and Scott 2022, p. 5). It also clarified the contact zone between the two species at the edge of the South Coast Ranges where they meet the floor of the Central Valley; although there are individuals with genetics from both species along the area where the species come into contact in this area, it appears that the boundaries are adjacent but do not overlap (Shaffer and Scott 2022, pp. 4–5) (Figure 5).



Figure 5. Current range of northwestern and southwestern pond turtles. As noted within this section, the species has been found between sea level to approximately 2,000 m. Habitat within this range is described in Chapter 6.

5.0 LIFE HISTORY

Western pond turtles are semi-aquatic, having both terrestrial and aquatic life history phases. Eggs are laid in upland terrestrial habitat, and hatchlings, juveniles, and adults use both terrestrial and aquatic habitat (Figure 6). The amount of time spent on land varies by location and aquatic habitat type. Terrestrial environments are required for nesting, overwintering and aestivation (warm season dormancy), basking, and movement/dispersal. Aquatic environments are required for breeding, feeding, overwintering and sheltering, basking, and movement/dispersal. Perennial (i.e., year-round) and intermittent (i.e., not year-round) bodies of water occur throughout the range. Some are flowing/lotic (e.g., streams, rivers, irrigation ditches), while others are not flowing/lentic (e.g., ponds, lakes, and reservoirs).

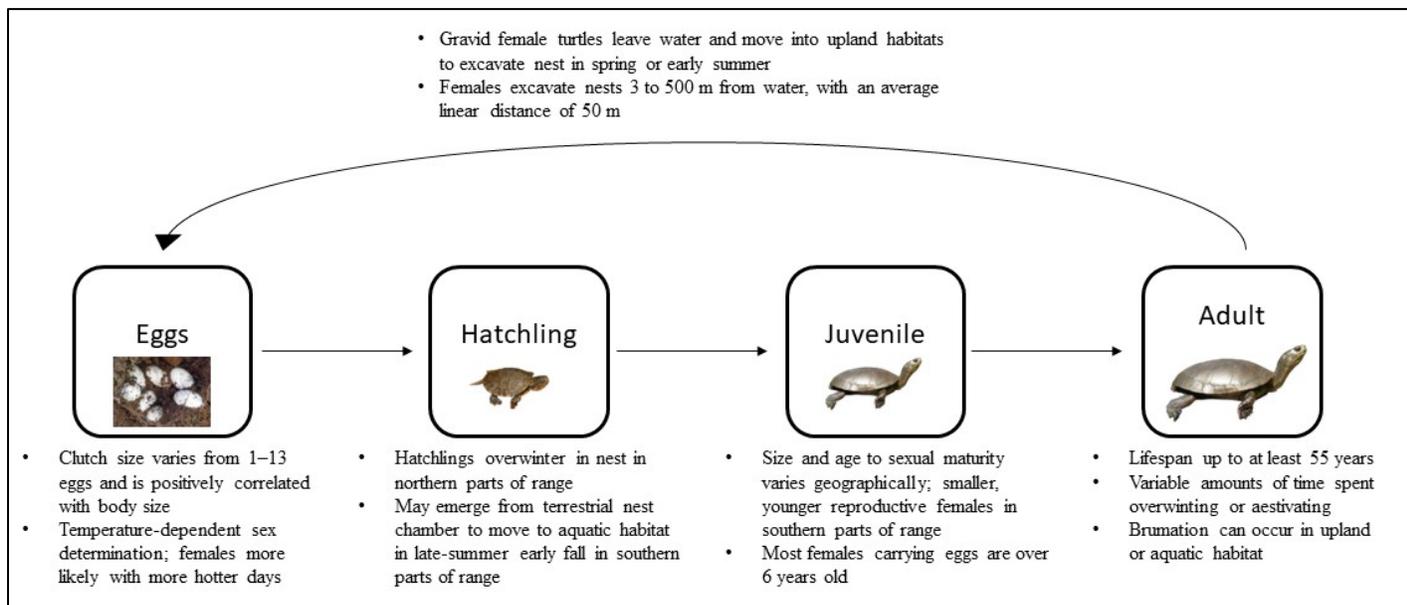


Figure 6. Life cycle diagram of the western pond turtle.

Courtship and Mating Behavior

Courtship behaviors have been observed from April through November, with mating observed in May through September (Holland 1991, p. 23). These behaviors appear to occur underwater, although this is based on very few observations (Holland 1988, pp. 87–88; Holland 1991, p. 23; Goodman Jr. 1997, pp. 33–34; Ashton 2007, entire; Bettelheim 2009, pp. 212–213). In central California, Monterey County, courtship activities were observed in mid-April within a 1.5 m deep pool with copulation documented the following day in shallow water at a depth of approximately 10 cm (Bettelheim 2009, pp. 212–213). In southern California, Holland (1988, p. 87) observed possible courtship behavior in 2 m deep water in mid-June. In northern California, mating has been observed in spring (Reese 1996, p. 224).

Nesting/Hatching

The time from ovulation of eggs to oviposition in the nest is unknown. Oviposition usually occurs from May through July, with northern populations depositing eggs later in the season than those in the south (Bury et al. 2012, p. 15). Gravid female turtles generally leave the water in the late afternoon or early evening and move into upland habitats to excavate a nest (Holland 1994, p. 2-10). Females may be out of the water for a few hours to several days with nest completion taking anywhere from 2 to more than 10 hours. Females may make several forays into the upland prior to actual oviposition, and sometimes make false scrapes where they abandon the nest prior to laying eggs, potentially as a result of hitting rocks or roots or because of disturbance, which western pond turtles are extremely sensitive to (Holland 1994, p. 2-10; Bury and Germano 2008, p. 001.5). Females will moisten the soil around the nest by urinating prior to digging the nest chamber (Holland 1994, p. 2-10; Hays et al. 1999, p. 12).

Females excavate nests 3 m to 500 m from water in compact, dry soils (Storer 1930, p. 434; Holland 1994, p. 2-10; Holte 1998, p. 54), with an average linear distance from water of 51 m (Davidson and Alvarez 2020, p. 44). Localized soil conditions, as well as the frequency and degree of disturbance in the upland habitat, probably limit nest distribution (Thomson et al. 2016, p. 300). Soils need to be loose enough to allow nest excavation, and typically have a high clay or silt component. Disturbance needs to be infrequent enough or of sufficiently low intensity that nesting females are not disturbed (Ernst and Lovich 2009, p. 178). Nests are shallow and generally occur between 9 to 12 cm below the surface (Holland 1994, p. 2-10). After the nest is excavated and eggs deposited, females pack the chamber using surrounding material such as mud, dry soil, and vegetation to form a plug that closes off the neck of the nest chamber (Holland 1994, p. 5-5).

Clutch size for western pond turtles varies from 1 to 13 eggs, and is positively correlated with body size (Holland 1994, p. 5-2; Holte 1998, p. 5). In a meta-analysis by Bury et al. (2012, p. 16), mean clutch size ranged from 4.5 to 8.5 eggs. Nesting frequency also varies across the range, based on female age, geographic location, and environmental conditions such as temperature or resource availability (Holte 1998, p. 5). The majority of females appear to deposit eggs every other year, but some may oviposit yearly (Holland 1991, p. 24). Additionally, double clutches have been documented at sites across the range, including southern California (Goodman Jr. 1997, p. 38), coastal Central California (Scott et al. 2008, p. 144; Germano and Rathbun 2008, p. 192), Oregon (Riley 2006 in Rosenberg et al. 2009, p. 25), and Washington (Hays et al. 1999, p. 12; Schmidt and Tirhi 2015, p. 8). Triple clutching may have occurred at a high elevation site in the Tehachapi Mountains, California, Los Angeles County (Germano and Riedle 2015, p. 107).

Incubation time is approximately 80 to 126 days (Holland 1994, pp. 2-10, 5-7). Hatching success is dependent on weather conditions, such as moisture and temperature, during the incubation

period. During incubation, low precipitation and warmer weather during the summer months improved hatching success, whereas cool, wet summers led to reduced hatching success (Holte 1998, pp. 96–98). Overall hatching rates average approximately 70 percent, although complete failure of nests is not uncommon (Holland 1994, p. 2-10).

Western pond turtles exhibit temperature-dependent sex determination (TSD) during incubation (Ewert et al. 1994, p. 7). In California, female hatchlings were more likely when 30 percent of the sex-determining period occurred above 29° Celsius (C) (84° Fahrenheit (F)) (Christie and Geist 2017, p. 49). In addition, lower fluctuations in temperature resulted in development of males, whereas females developed in nests with high and low temperature fluctuations. Temperatures within nests were found to fluctuate daily, varying by more than 20°C (36°F) on a daily basis (Geist et al. 2015, p. 498; Christie and Geist 2017, p. 50). Higher maximum temperatures reduce overall egg viability (Christie and Geist 2017, pp. 49, 51).

Hatchlings

While few studies have tracked hatchlings leaving the nest, available studies show variation in timing of emergence and behavior post-emergence. In southern and central California, some hatchlings may emerge from the nest chamber in late-summer to early-fall, whereas others overwinter in the nest chamber and emerge in spring (Holland 1994, p. 2-10). In the northern parts of the range, hatchlings overwinter in the nest (Holland 1994, p. 2-10; Reese and Welsh Jr 1997, p. 354). In western Oregon, hatchlings delayed emergence until spring, and typically remained within 2 m of nests for as long as 59 days after initial emergence (Rosenberg and Swift 2013, entire). During migration from their nests to aquatic habitat, hatchlings embedded themselves in soil for up to 22 days at stop-over sites. Hatchlings entered aquatic habitat on average 49 days after initial emergence and traveled an average of 89 m from their nest site. Hatchlings detected in water were always within 1 m of shore and in areas with dense submerged vegetation and woody debris (Rosenberg and Swift 2013, entire).

Growth

Hatchlings can nearly double in size by the end of the first year (Germano and Rathbun 2008, p. 189; Germano 2010, p. 95; Bury et al. 2012, p. 17). Growth rates vary based on factors including developmental conditions, environmental conditions, geography, and individual variation (Bury et al. 2012, p. 16). For example, Holland (1994, p. 2-11) notes that turtles between 100 to 110 mm are generally 4 to 5 years old, but may be as young as 3 or as old as 12. Northwestern pond turtles in Oregon were slightly larger than the same species in California, although the growth rate to achieve these sizes was slower, possibly because of cooler temperatures (Germano et al. 2022, pp. 114–115).

Age and size reached at sexual maturity is poorly understood and varies between sites and geography (Holland 1994, pp. 2-9, 5-2; Rosenberg et al. 2009, p. 22; Bury et al. 2012, p. 15). In general, males tend to exhibit external signs of sexual dimorphism around 110 to 120 mm CL (Bury et al. 2012, p. 15). In coastal central California, the average male reached 120 mm CL in 3.6 years compared to 4.1 years for females, and reached 150 mm CL in 8.3 years for males versus 11.1 years for females (Germano and Rathbun 2008, pp. 190–191). In Washington, males are thought to achieve sexual maturity when they are at least 10 to 12 years old (Hays et al. 1999, p. 12).

Wide variation occurs throughout the ranges of the two species, but in general, most females carrying eggs are over 6 years old (Bury et al. 2012, p. 15). In Oregon and northern California, females start carrying eggs when they are at least 120 mm CL and typically 8 to 10 years of age. In southern California, the smallest known reproductive female was approximately 111 mm CL and at least 6 to 7 years old, while the smallest reproductive female in Oregon was 131.3 mm CL (Holland 1994, p. 5-2). In coastal central California, a female as young as 4 years old and measuring 141 mm CL was documented carrying eggs (Germano and Rathbun 2008, p. 190). At two sewage treatment facilities in the San Joaquin Valley, California, females were documented carrying eggs at 4.4 years of age with a CL of 155 mm. In these areas, warmer water and high nutrient loads may have increased aquatic invertebrates, providing increased nutrition for faster growth rates (Germano 2010, p. 95).

Mortality and Survival

Approximations of mortality rates for adults and survivorship of earlier life stages is difficult to discern in western pond turtles due to their long life span and low detectability of young turtles. Survivorship is lowest in the smaller size classes (Holland 1994, p. 2-11). Annual mortality rates for young age classes vary greatly: mortality of juveniles less than 80 mm CL was 26.9 percent and of juveniles up to 120 mm CL was 16.2 percent at a site in the San Joaquin desert in northwestern Kern County, California described as having a robust population (Germano 2016, p. 670), while annual mortality rates for juveniles during the first three years was 85 to 90 percent according to unpublished data in a comprehensive report on the species focusing mainly on the Pacific Northwest (Holland 1994, p. 2-11). Survivorship increases as the turtles approach their reproductive years at around 120 mm CL. Beyond this point, survivorship seems to increase and appears relatively high, assumed to average 95 to 97 percent per year based upon analysis of long-term mark and recapture data (Holland 1994, p. 2-11). We note that at the same San Joaquin desert site with lower juvenile mortality than that found in the Pacific Northwest, annual adult male mortality was 18.7 and adult female mortality was 26.9 percent (Germano 2016, p. 670). The maximum lifespan of western pond turtles is unknown. However, they are long-lived species after reaching adulthood, with some living to at least 55 years of age (Bury et al. 2012, p. 17). These old individuals are rare in natural populations, but they appear to reproduce

throughout their life span based on a radiograph of a 55 year old female with eggs (Kaufman and Garwood 2022, p. 354).

Feeding

The western pond turtle is omnivorous and considered a dietary generalist (Holland 1994, p. 2-5), consuming a wide variety of food items. Prey resources are primarily found within water but can be captured or scavenged on land. Food captured or scavenged on land must be brought back to water for consumption, as they appear to be unable to swallow in the air (Holland 1994, p. 2-6). Animal matter appears to constitute a larger portion of the diet than plant material (Bury 1986, pp. 518–520; Holland 1994, pp. 2-5–2-6). Stomach contents reveal the diet consists of small aquatic invertebrates, with small vertebrates (fish, tadpoles, and frogs), carrion, and plant material (Bury 1986, p. 516; Holland 1994, pp. 2-5–2-6). In northern California, contents of 77 stomachs included aquatic insects such as dragonfly larvae, mayflies, stoneflies, caddisflies, midges, beetles, and other insects, including terrestrial prey items (e.g., grasshoppers) (Bury 1986, p. 516). Bury (1986, p. 517) found that 44 percent of the females consumed plant material compared to 10 percent of the males. Juveniles consumed mostly invertebrates (Bury 1986, p. 517), and hatchlings primarily feed on nekton and larvae of small aquatic insects (Holland 1994, p. 2-6).

Basking

Western pond turtles engage in both emergent basking and aquatic basking. Basking is an integral function of western pond turtles and is essential for thermoregulation and in turn, physiological functions such as metabolism, digestion, reproduction, and growth. Additional benefits of emergent basking by turtles include drying out the shell and skin for parasite or algal control.

The amount of time spent basking varies throughout the range depending on water and air temperature. On the Trinity River in northern California, western pond turtles spent more time emergent basking in the main fork which had cooler water temperatures than the south fork (Ashton et al. 2011, pp. 32–33). Similarly, at both an intermittent stream and perennial stream in Sequoia National Park, western pond turtles were more likely to be out of the water when air temperatures increased (Ruso et al. 2017, p. 24). At the University of California, Davis campus, also in northern California, western pond turtles were more abundant at basking sites when water temperatures were warm and the sites were unshaded (Lambert et al. 2013, p. 196). Basking structures may be especially important below dam sites with cold water discharge (Ernst and Lovich 2009, p. 175). During emergent basking, western pond turtles will retreat into the water or may seek shade once it gets too hot or sufficient internal temperatures have been achieved (Ernst and Lovich 2009, p. 176). Western pond turtles can also engage in aquatic basking, wherein they lay completely or almost completely submerged in shallow water or in a top layer

of vegetation individuals that use aquatic basking may be less conspicuous than those that use emergent basking (Holland 1991, p. 20).

Overwintering/Aestivation

Variable amounts of time may be spent overwintering and/or aestivating. Generally, overwintering is a state of little to no activity (e.g., brumation) that occurs during the cooler months of the year and can occur in either upland or aquatic environment (Holland 1994, p. 2-7; Ultsch 2006, pp. 341, 356). Aestivation is a period of inactivity, usually in response to the hottest time of year or dry conditions (Hays et al. 1999, p. 7) that occurs in terrestrial habitat.

The amount of time spent overwintering and/or aestivating varies geographically and within populations, and is likely influenced by climate and hydrological conditions. At two sites in California, western pond turtles left intermittent ponds as they dried out and overwintered in terrestrial habitat, returning to the ponds weeks or months after they refilled (Pilliod et al. 2013, entire; Zaragoza et al. 2015, entire). Similarly, western pond turtles inhabiting intermittent streams may respond to late summer drying and winter flooding by moving into upland habitat (Rathbun et al. 2002, p. 233). However, in perennial streams and rivers, turtles may remain active until fall/winter storms increase stream flows and reduce water temperatures (Belli 2016, p. 11). In northern California, beginning in September, western pond turtles spent seven months of the year away from the Trinity River to overwinter in uplands, while others moved to nearby lentic bodies of water (lake and slough) as far as 500 m from the river (Reese and Welsh Jr 1997, pp. 355–357). Movements, in this case, may have been to avoid winter flood events (Reese and Welsh Jr 1997, p. 356; Rathbun et al. 2002, pp. 228, 233). Moving to upland habitats above the flood line is generally more common for turtles occupying lotic habitats. Along the central California coast, western pond turtles that occupied pond habitat overwintered on-site, whereas most turtles from an adjacent stream left with the first heavy rains and overwintered in the upland or moved to the pond (Davis 1998, pp. 21, 25). In response to spring storms, some turtles remained within the stream under banks or within submerged shoreline or riparian vegetation, whereas others left the stream and moved a minimum of 4 m away (Belli 2016, p. 53). The range of behaviors between populations and individuals exemplifies that western pond turtle exhibit high plasticity with respect to overwintering and aestivation strategies (Holland 1994, p. 8-3; Ultsch 2006, p. 356; Zaragoza et al. 2015, p. 437). At study sites on the Trinity River and in Santa Rosa in northern California, overwintering locations across successive years was very similar, with distances between overwintering points as small as one meter (Reese 1996, p. 218). However, radio-tagged western pond turtles that were tracked for two winters on the Carrizo Plain Ecological Reserve, San Luis Obispo County, California did not have overwintering site fidelity (i.e., they did not return to the same sites) (Pilliod et al. 2013, pp. 215–2016).

Migration

We define migration for the western pond turtle to be intra-population (within local populations) movements occurring between aquatic and upland environments. Migrations are often roundtrip and reoccurring (often seasonally, but not always annually), such as when individuals are moving from aquatic to upland environment (and back) for the purpose of nesting, overwintering, and aestivation. Males generally move farther than females or juveniles (Bury 1972a, pp. 65–66). Measured home ranges of western pond turtles average 1 hectare (2.5 acres) for males, 0.3 hectare (0.7 acre) for females, and 0.4 hectare (1 acre) for juveniles (Bury 1972a). Overwintering behavior is variable, and likely more common in seasonally inundated ponds than permanent water (Pilliod et al. 2013, p. 216). Using radio-telemetry, Holland (1994, pp. 6-12–6-13) found overwintering sites at two streams/rivers that ranged from 15 to 260 m away from the aquatic environment. In northern California along the Trinity River, some turtles sought upland refuge to either overwinter or aestivate while others moved to lentic bodies of water (standing bodies of water) as far as 500 m from the river (Reese and Welsh Jr 1997, p. 356). The pattern and frequency of these migrations vary with habitat, size of the aquatic system, suitability of upland habitat, season, climate, environmental stress (e.g., drought, high stream flow), sex, and life stage (Hallock et al. 2017, p. 4). In central California, radio-tagged turtles spent over half of the year in terrestrial habitat, moving from 255 to 1,096 m over the study period but never moving farther than 343 m from seasonal ponds. Western pond turtles moved in different directions, used different microhabitats, and left ponds at different times (Pilliod et al. 2013, pp. 215–2016).

Dispersal

Dispersal of western pond turtles between populations/watersheds is generally not well understood. Genetic analyses suggest that most movements occur within drainages (Spinks and Shaffer 2005, p. 2057), but few accounts of adult and juvenile dispersal exist. Within aquatic habitat, a dispersal distance of 7 km upstream was observed (5 km overland distance) (Holland 1994, p. 7-28). Dispersal may also occur via aquatic habitats during flood events (Rosenberg et al. 2009, p. 21). Along the central California coast, Holland (1994, p. 2-9) recorded less than 10 dispersal events between drainages during a 10-year study with over 2,100 captures and recaptures across 21 drainages, suggesting that overland movements are uncommon. In that study, the longest overland distance recorded in an area considered to be under the best circumstances (mild climate and short distances between water features), was a single individual travelling 5 km. Holland (1994, p. 2-9), also states that no movements between drainages were detected from three other sites with over 1,100 hundred captures and recaptures over a 7-year period. During an extreme drought, Purcell et al. (2017, pp. 21, 24) documented a 2.6 km straight-line distance movement overland in a radio-tagged turtle, with a minimum total distance of 3.3 km moved before the individual found water.

6.0 HABITAT AND DEMOGRAPHIC NEEDS

Habitat Needs

Western pond turtles are semi-aquatic, requiring both aquatic and terrestrial habitats that are within close proximity and connected to one another (Figure 7). As habitat generalists, western pond turtles occur in a broad range of permanent and ephemeral aquatic water bodies from remote to urban landscapes, including flowing rivers and streams, lakes, ponds, reservoirs, settling ponds, marshes, vernal pools irrigation ditches, and other wetlands, including some with estuaries with tidal influence (Spinks et al. 2003, entire; Bury and Germano 2008, p. 001.3; Ernst and Lovich 2009, p. 175; Bury et al. 2012, p. 12; Stebbins and McGinnis 2018, p. 205). Despite their ability to use a wide range of aquatic features, suitable aquatic habitats are relatively rare across much of the range, exacerbated by land use changes (e.g., urbanization and agriculture) after European settlement (discussed more in Chapter 8.1: Habitat Loss and Fragmentation). Consequently, the species' distribution may be disjunct across the landscape, following the arrangement of ponds or streams, especially in areas with extensive open, dry terrain between waterways (Bury et al. 2012, p. 12). The back-and-forth movements between aquatic and terrestrial habitats (i.e., migration) are typically less than 500 m (Reese and Welsh Jr 1997, p. 357), thus the two habitat types must be adjacent. In a study in northern California, radio-tagged males used terrestrial habitat in at least ten months of the year, emphasizing the importance of upland habitat in addition to aquatic habitat (Reese and Welsh Jr 1997, p. 352).

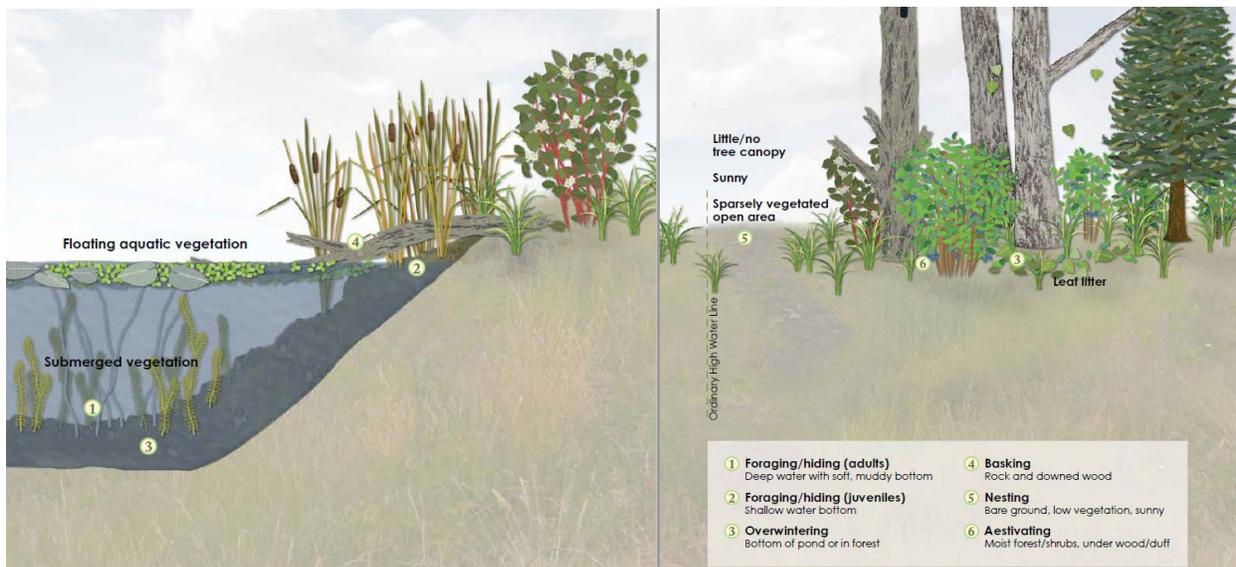


Figure 7. Aquatic and upland habitat characteristics (Figure 3 in ODFW 2015, pp. 6–7).

Aquatic Habitat

Western pond turtles use aquatic habitat for breeding, feeding, overwintering, and sheltering. Preferred aquatic conditions are those with abundant basking sites, underwater shelter sites (undercut banks, submerged vegetation, mud, rocks, and logs), and standing or slow-moving water (Holland 1991, pp. 13–14; Reese and Welsh Jr. 1998a, p. 852; Hays et al. 1999, p. 10; Bury and Germano 2008, p. 001.4; Ernst and Lovich 2009, p. 175). Western pond turtles inhabiting lentic aquatic habitat, such as ponds, lakes, and slack water habitats, often overwinter within the aquatic environment, burying themselves within the bottom substrate, such as mud. Various depths of water provide western pond turtles with habitat necessary for overwintering and hatchling growth. Primary habitat for hatchlings and young juveniles is shallow water with dense submerged vegetation and logs, which most likely provides shelter, prey, and thermoregulatory requirements or other functions for survival (Holland 1994, pp. 1-14, 2-12; Rosenberg and Swift 2013, p. 119).

Basking Sites

Emergent basking usually takes place on logs, rocks, emergent vegetation, shorelines, and essentially any other substrate located within and adjacent to aquatic habitat (Holland 1994, p. 2-8; Hays et al. 1999, p. 10). The location of the basking site above or adjacent to aquatic features allows for quick retreat into the water if there is perceived danger (Storer 1930, p. 431). At a site in northern California, stream microhabitats containing emergent basking sites (those sites above or adjacent to water) had more turtles present than those without available emergent basking sites (Reese and Welsh Jr. 1998a, pp. 850–851). At the University of California, Davis, Arboretum, the maximum number of western pond turtles observed at a site was 14, although smaller numbers were more common (average 0.34 ± 0.03 SE) (Lambert et al. 2013, p. 195). Basking may also occur in shallow water (aquatic basking), in a top layer of vegetation, or in submerged vegetation, such as algal mats. Aquatic basking may be used when emergent basking sites are limited or not present and provide a warmer environment than that of surrounding water (Jennings and Hayes 1994, p. 100; Reese and Welsh Jr. 1998a, p. 851).

Upland Nesting Habitat

Upland habitat varies greatly across the two species' geographic ranges. However, some common attributes include that the nesting habitat is in close proximity to aquatic habitat and is typically characterized as having sparse vegetation with short grasses and forbs and little or no canopy cover to allow for exposure to direct sunlight (Holland 1994, p. 2-10; Rathbun et al. 2002, p. 232; Rosenberg et al. 2009, pp. 16–17; Riensche et al. 2019, p. 97). Females excavate nests in compact, dry soils that are 3 to 400 m from water (Holland 1994, p. 2-10; Holte 1998, p. 54). In the central coast of California, all successful and attempted nest sites were on compact

and hard soils with little vegetative cover (coastal sage scrub, exotic annual grasslands, and weed patches) (Rathbun et al. 2002, p. 232).

Additional features of nesting habitat/sites that may be important include aspect, slope, and vegetation. Nests generally occur on south or west-facing slopes, although nesting does occur on northwest, and southeast facing slopes (Holland 1994, p. 2-10; Lucas 2007 in Rosenberg et al. 2009, p. 17). Slope varies from 0 to 60° with the majority of nests on slopes of 25° or less (Holland 1994, p. 2-10). Nest site vegetation density and height can vary. At a study site in Oregon, nesting sites had low, dense vegetation with heights ranging from 0 to 20 cm (mean 4.8 cm) (Holte 1998, p. 28), while at a southwest Washington study site, nesting site vegetation heights were 24 to 45 cm (Lucas 2007 in Rosenberg et al. 2009, p. 17). At this site, where forest vegetation provided canopy cover, turtles selected more open canopies (average of 14 percent) for nesting, especially on the south southeastern slopes, and soil temperatures at nest sites were found to be warmer compared to random sites (Lucas 2007 in Rosenberg et al. 2009, p. 17).

Little is known about upland habitat requirements of hatchlings after emerging from the nest. In western Oregon, upland habitat use and movement by hatchlings varied, and hatchlings were generally found buried into soil or detritus where they were hidden from view (Rosenberg and Swift 2013, p. 116). After departing these areas, individual hatchlings made stops for varying durations in a variety of habitats. Habitat features included small patches of forest floor (embedded approximately 8 cm under detritus), small patches of forest (buried approximately 5 to 8 cm in the detritus or directly under moss in dense shrub cover), and in sparsely vegetated areas (typically embedded in soil and completely covered by moss) (Rosenberg and Swift 2013, p. 116).

Upland Overwintering/Aestivation Habitat

The upland habitat occupied by western pond turtles for overwintering and aestivation varies greatly across the range. Overwintering/aestivation habitat generally occurs in upland locations above ordinary high-water lines or beyond the riparian zone, although understanding of specific microsite conditions is limited (Reese and Welsh Jr 1997, p. 355; Rathbun et al. 2002, p. 229; ODFW 2015, pp. 6–7). In the Trinity River system in northern California the greatest distance western pond turtles traveled from their aquatic habitat to terrestrial overwintering sites was approximately 500 m (Reese and Welsh Jr 1997, p. 356). Using radio-telemetry, Holland (1994, pp. 8-12–8-13) found overwintering sites at two streams/rivers that ranged from 15–260 m away from the aquatic environment. While vegetation communities differ from site to site, open areas were avoided for overwintering, and leaf litter was present at most sites (Reese and Welsh Jr 1997, p. 355; Davis 1998, p. 19; Rathbun et al. 2002, p. 230). In central California, radio-tagged western pond turtles were generally located where they could be exposed to direct sunlight during a portion of the day (Rathbun et al. 2002, p. 230). In multiple telemetry studies in

California, radio-tagged turtles using upland habitat were found buried beneath 5 to 10 cm of leaf litter (Reese and Welsh Jr 1997, p. 354; Rathbun et al. 2002, p. 230).

Demographic Needs

The demographic needs being considered in this SSA are abundance, reproduction/recruitment, survival, and connectivity. Western pond turtles need enough individuals within a population to be able to withstand stochastic events. Reproduction/recruitment and survival are related to overall population growth, which needs to be greater than or equal to one to ensure that populations are self-replacing. Connectivity between populations is important for genetic diversity and health of populations, which can be measured by looking at factors including inbreeding and heterozygosity.

Abundance

Abundance, the number of individuals within a population, is a tenet of population resiliency (Wolf et al. 2015, p. 205) because it is an important predictor of extinction risk (Matthies et al. 2004, p. 483; Figure 1 in Pearson et al. 2014, p. 219). In general, the larger the population, the more resilient it is to stochastic demographic and environmental influences (Shaffer 1987, p. 71). The minimum number of breeding adults and rate of juvenile recruitment necessary to maintain a stable population are currently unknown for western pond turtles. Abundance data are not widely available for either western pond turtle species.

Reproduction/Recruitment

Reproduction and recruitment are required to maintain and increase population abundance. As such, they are required for population resiliency. The rate of reproduction and recruitment must compensate for the rates of juvenile and adult mortality. For successful reproduction, western pond turtles need access to mates and nesting sites. Recruitment is contingent on the survival of eggs to hatchlings, and survival of hatchlings to adult breeding condition.

Survival

Survival, like reproduction and recruitment, is positively related to population abundance and resiliency. Low survival rates are extremely limiting for population growth and abundance, and may prevent populations from being able to withstand stochastic events. Survival at all life stages (hatchling, juvenile, and adult) is required for population resiliency. Although the exact relationship varies geographically, age and size are positively correlated. Size is therefore often used as a proxy for age, though this is best done within, rather than between, populations. Survivorship is lowest for young/smaller hatchlings and juveniles, increases with size, and is highest for adult males and females.

Connectivity

For the purpose of this SSA, we define population connectivity as mobility within a population (migration) and between different populations (dispersal). Populations need a network of quality aquatic breeding and feeding habitat, and upland nesting and overwintering/aestivation sites that are connected by habitat suitable for dispersal. A population that is connected to other populations (via dispersal habitat) is more resilient because it can receive new individuals that may colonize extirpated sites and/or enhance the genetic diversity of the population and reduce effects of small population size.

Summary

In summary, the habitat needs considered most important to western pond turtles to complete their life cycle include: aquatic habitat, upland habitat, and basking sites. Table 1 below summarizes the individual habitat needs by life stage, and resource function. The demographic needs considered most important for western pond turtles are abundance, reproduction/recruitment, survival, and connectivity.

Table 1. Individual habitat needs for life history stages of the western pond turtle.

Individual Need	Life Stage	Resource Function
Aquatic Habitat	Hatchlings, Juveniles, Adults	Breeding, Feeding, Overwintering, Sheltering, and Dispersal
Upland Habitat	Eggs, Hatchlings, Juveniles, Adults	Nesting, Overwintering and Aestivation, and Dispersal
Basking Sites	Hatchlings, Juveniles, Adults	Thermoregulation

7.0 SPECIES NEEDS

Species-level conservation parameters are attributes that support viability, which is the ability of a species to maintain populations in the wild over time. Using the SSA framework, we describe the species' viability by characterizing the status of the species in terms of its resiliency, redundancy, and representation (Service 2016, entire). Although resiliency is often assessed at the population level in the SSA framework, for the northwestern and southwestern pond turtle we assessed resiliency at the analysis unit level. Analysis units are composed of multiple populations and have the demographic needs as described above. Representation and redundancy are assessed across the entire range of each species. There are currently no known differences in

species needs between the northwestern and southwestern pond turtle species, therefore the species needs identified in this SSA apply to both.

Analysis Units

To assess resiliency of the northwestern and southwestern pond turtle, we separated the species' ranges into analysis units. Species experts indicate that habitat and demographic processes likely operate at the USGS Hydrologic Unit Code subwatershed (HUC12) scale (U.S. Geological Survey et al. 2013, entire), which might be more comparable to populations, but we lacked information to analyze the species at that level. Instead, analysis units are delineated based on genetic, management, and ecological data, and stem from collaboration with stakeholders across the range of both species through the Western Pond Turtle Range-wide Conservation Coalition. For analysis units that were based on genetics, delineation along USGS HUC10 watersheds helped to bound areas. Analysis units encompass multiple populations with varying levels of connectivity within and between them.

We identified 14 analysis units for northwestern pond turtle (Figure 8), with the process for determining analysis units varying by each State. In Washington, all populations are supplemented with captive-bred or wild-bred headstarted turtles, with captive stock including one group of Columbia River Gorge origin and one likely of Puget Sound lowlands origin (Hays et al. 1999, p. 25). Therefore, we created an analysis unit around each of these regions. Results from genomic analyses found that all Oregon samples were most closely related to each other and to samples from northern California (Shaffer and Scott 2022, pp. 6–7, 19). However, rather than grouping all northern California and Oregon western pond turtles together, in Oregon, we based analysis units on management regions defined by the Oregon Department of Fish and Wildlife (ODFW 2019, in litteris). In California and Nevada, we designated analysis units primarily using genetic relatedness clusters that maximized the structure of the data in recent genomic analyses (Shaffer and Scott 2022, entire). This resulted in northwestern pond turtles in Nevada being grouped with those in the San Francisco Bay Delta and east across the Central Valley of California. In one exception, we merged two apparent genetic clusters in the Yosemite region of California based on similarity of threats and management in this area (Shaffer and Scott 2022, p. 7).

We divided the southwestern pond turtle range into 6 analysis units (Figure 9) based on genetic relatedness clusters that maximized the structure of the data in recent genomic analyses (Shaffer and Scott 2022, entire). In one exception, there was an individual from San Diego County that split into its own genetic cluster in the genomic analysis, but we combined it with other San Diego samples into one analysis unit. Other individuals from the same site did not cluster with the lone individual in its cluster, suggesting that the separate cluster may have been artifactual (Shaffer and Scott 2022, p. 7).

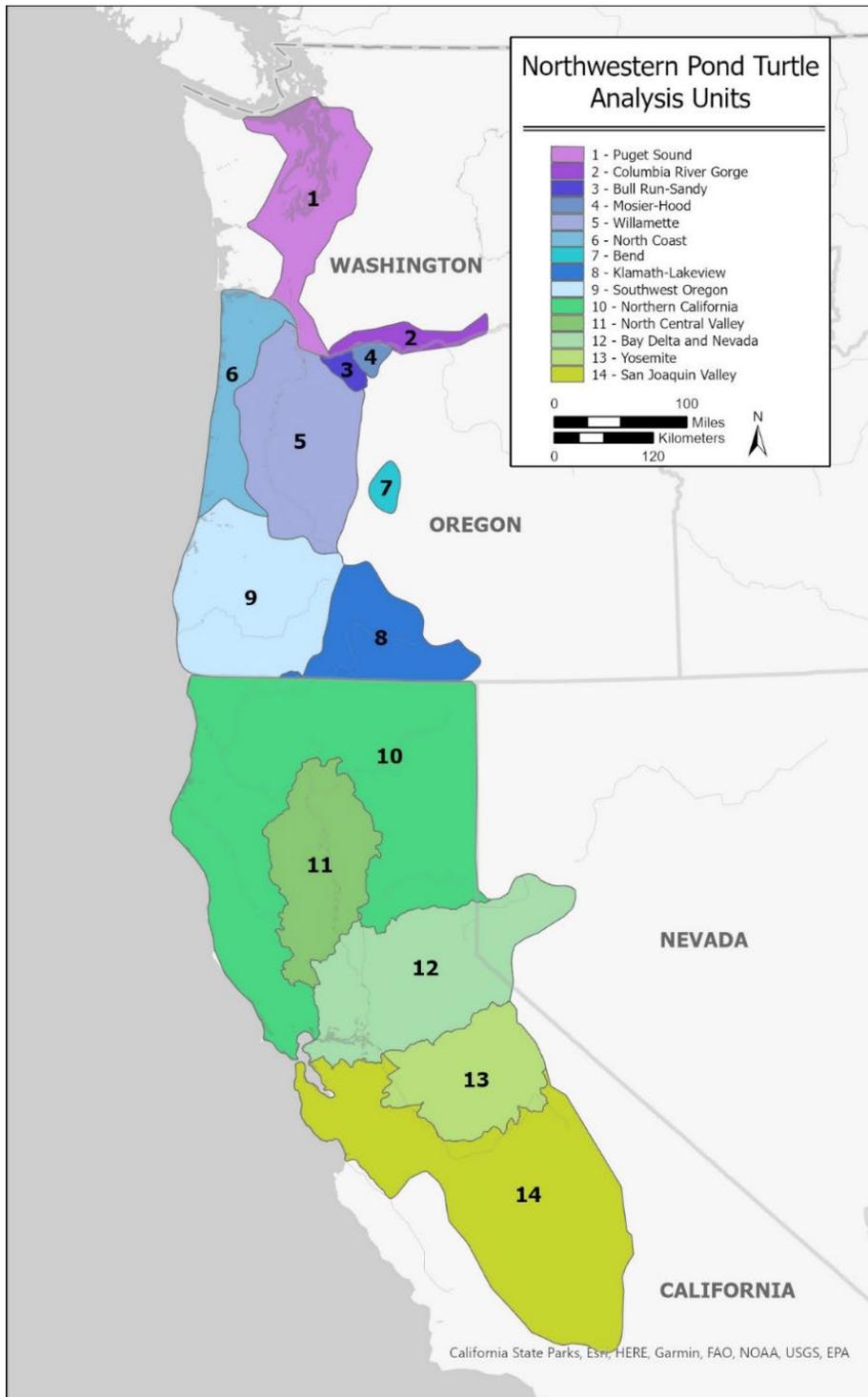


Figure 8. Analysis units for the northwestern pond turtle. Fourteen analysis units are distributed across Washington, Oregon, and California. Analysis units in Washington are based on two regions of origin for captive-bred or headstarted turtles, analysis units in Oregon are based on management regions defined by ODFW, and analysis units in California and Nevada are based on genetic relatedness clusters.

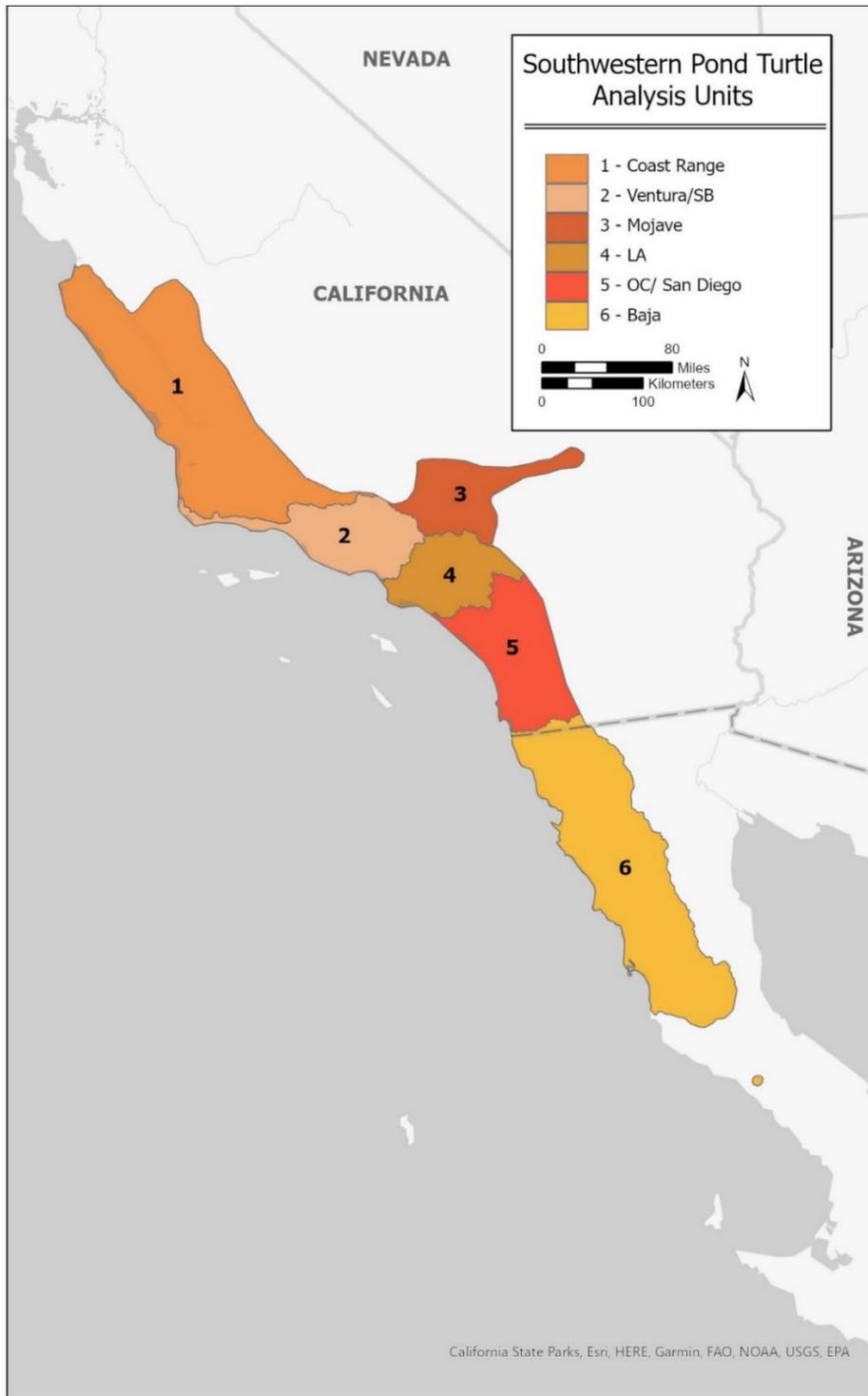


Figure 9. Analysis units for the southwestern pond turtle. Six analysis units are distributed across California and Mexico based on genetic data.

Table 2. Northwestern pond turtle analysis units. Analysis units in Washington are based on two regions of origin for captive-bred or headstarted turtles, analysis units in Oregon are based on management regions defined by ODFW, and analysis units in California and Nevada are based on genetic analyses in Shaffer and Scott (2022, entire).

Analysis Unit	State	Number
Puget Sound	WA	AU-1
Columbia Gorge	WA	AU-2
Bull Run-Sandy	OR	AU-3
Mosier-Hood	OR	AU-4
Willamette	OR	AU-5
North Coast	OR	AU-6
Bend	OR	AU-7
Klamath-Lakeview	OR	AU-8
Southwest Oregon	OR	AU-9
Northern California	CA	AU-10
North Central Valley	CA	AU-11
Bay Delta and Nevada	CA/NV	AU-12
Yosemite	CA	AU-13
San Joaquin Valley	CA	AU-14

Table 3. Southwestern pond turtle analysis units are based on genetic analyses in Shaffer and Scott (2022, entire).

Analysis Unit	State	Number
Coast Range	CA	AU-1
Ventura/Santa Barbara	CA	AU-2
Mojave	CA	AU-3
Los Angeles	CA	AU-4
Orange County/San Diego	CA	AU-5
Baja California, Mexico	Mexico	AU-6

Resiliency

Resiliency describes the ability of the species to withstand stochastic disturbance events, which is associated with population size, growth rate, and habitat quality. Analysis unit resiliency relies on sufficient suitable habitat in a condition to support multiple populations with enough individuals to withstand stochastic events. Stochastic events that may be experienced by western pond turtles include, but are not limited to, floods, droughts, high severity wildfires, disease outbreaks, and predation. A variety of factors may regulate western pond turtle numbers. These factors may be density-dependent (e.g., habitat quality, habitat abundance) or density-independent (e.g., climate).

Redundancy

Redundancy describes the ability of a species to withstand catastrophic events. Redundancy gauges the probability that the species has a margin of safety to survive and rebound after a catastrophe. We can best gauge redundancy by assessing the number and distribution of resilient analysis units relative to the scale of anticipated species-relevant catastrophic events, which entails assessing the cumulative risk of catastrophes occurring over time. Catastrophic events that could affect the western pond turtle include long-term drought, large floods, large severe wildfires, or disease epidemics. Implications of these threats are discussed in more detail in Chapter 8.

Representation

Representation is the ability of a species to adapt to both near-term and long-term changes in its physical (e.g., climate conditions, habitat conditions, habitat structure, etc.) and biological (e.g., disease, competitors, prey, predators, etc.) environments. Physical and biological changes that

are occurring, or are expected to occur, in western pond turtle habitat include hydrological management, habitat alteration, high-severity wildfire, climate change, disease, parasites, nonnative species, and dynamics of predation and competition. Implications of these changes are discussed in more detail in Chapter 8. To evaluate representation for western pond turtles, we examined the breadth of ecological and genetic diversity found within each species.

8.0 INFLUENCES ON VIABILITY

In this section, we evaluate the significant past, current, and future threats, and current conservation efforts, that are affecting western pond turtles. In our assessment, we also consider how these factors may cumulatively affect the western pond turtle both negatively and positively.

In our 90-day finding, we identified several threats that may negatively affect the western pond turtle. These threats included loss, alteration, degradation, and fragmentation of habitat due to agricultural development, flood control, water diversion projects, groundwater depletion, and urbanization. Since the 90-day finding, we have identified additional threats including predation and competition, disease, collection, and the effects of climate change (including increased temperatures, more frequent and severe droughts, extreme flood events, and high severity wildfire). Many of the threats are interrelated, have secondary impacts, or act concurrently on populations.

The Washington Department of Fish and Wildlife's periodic status review for the northwestern pond turtle identified factors affecting continued existence of pond turtles in Washington to be: natural factors (e.g. slow rate of growth, delayed sexual maturity, limited ability to disperse, complex habitat requirements and the high mortality of eggs and hatchlings, being at the northern extreme of the range), inadequacy of existing regulatory mechanisms, diseases, predation and competition with other species (especially predation by nonnative bullfrogs), habitat loss and degradation, small population size (low genetic variation), and climate change (Hallock et al. 2017, pp. 8–11). Other potential conservation concerns included trampling by livestock or large wildlife, shooting, vehicle mortality, entanglement in discarded fishing line, frequent human disturbance, rotenone use, exposure to contaminants, and catastrophic events (Hays et al. 1999, pp. ix, 31–33; Bury and Germano 2008, pp. 001.6–001.7; Pramuk et al. 2013, p. 6).

The Oregon Department of Fish and Wildlife's conservation assessment for western pond turtle identified factors limiting populations to be: loss of habitat, elevated nest and hatchling predation (including nonnative and native predators), road mortality, collection, nonnative species (red-eared slider and common snapping turtle (*Chelydra serpentina*)), and disturbance from recreation. Other threats identified included isolation and population fragmentation, research/survey disturbance, illegal shooting, stream restoration, contaminants, agricultural and

vegetation management activities, disease, and climate change (Rosenberg et al. 2009, pp. 40–47).

The California Department of Fish and Wildlife-commissioned assessment of California amphibian and reptile species of special concern identified factors limiting western pond turtles in California to be: land use changes and fragmentation of existing habitat (conversion to urban development, dams, agriculture, climate change), and competition and predation by introduced species (Thomson et al. 2016, pp. 301–302).

The Nevada Department of Wildlife identified threats to western pond turtles in the State Wildlife Action Plan to be: housing and urban areas; annual and perennial non-timber crops; invasive nonnative/alien species (including bullfrogs, crayfish, and carp); climate change and severe weather (including persistent droughts as well as unpredictable storms and flooding events); livestock farming and ranching; dams and water management/use; and roads and railways (Wildlife Action Plan Team 2022, p. 57).

A threat analysis based on peer-reviewed literature in addition to published and unpublished reports ranked threats for both northwestern and southwestern pond turtles (Manzo et al. 2021, pp. 487–488, 492–493). Based on their literature review, Manzo et al. (2021, pp. 487–488) ranked 13 threats for each of the species by scoring threat observations within papers, including the number of observations (sum score) and the average severity of the threat (mean score) as described by the cited authors. Threat severity level ranged from 0 to 2, with 0 being no observed or known effect on population health/size, and 2 being population extirpation. Results from this threats analysis are shown in Figure 10 and Figure 11. For both northwestern and southwestern pond turtles, predation (by bullfrogs and largemouth bass), drought, and land alteration were in the top four threats. Notable differences between the rankings for the two species were the higher rankings of pathogens for northwestern pond turtle and floods for southwestern pond turtles.

Threat	Rank	Sum score	Mean score	No. of observations
Predation (bullfrogs and Largemouth Bass)	1	9	0.82	11
Pathogens	2	9	0.60	15
Land alteration	3	8	1.14	7
Drought	4	7	1.40	5
Harvesting	5	4	1.00	4
Natural predators	6	4	0.80	5
Dams	7	2	1.00	2
Roadways	8	2	0.67	3
Rising temperatures	9	1	1.00	1
Competition with nonnative species	10	1	0.50	2
Contaminants	11	1	0.06	16
Flood	12	0	0.00	1
Wildfire	N/A	N/A	N/A	0
Total				72

Figure 10. Threats for northwestern pond turtle from a peer-reviewed threat assessment (Figure 5b from Manzo et al. 2021, p. 492). Manzo et al. (2021) ranked the 13 threat categories using the “sum score,” and resolved ties using the “mean score.”

Threat	Rank	Sum score	Mean score	No. of observations
Drought	1	17	1.06	16
Predation (bullfrogs and Largemouth Bass)	2	6	1.00	6
Flood	3	5	0.56	9
Land alteration	4	4	1.33	3
Wildfire	5	3	1.50	2
Natural predators	6	3	1.00	3
Roadways	7	3	0.75	4
Harvesting	8	1	1.00	1
Dams	9	0	0.00	1
Pathogens	9	0	0.00	3
Competition with nonnative species	N/A	N/A	N/A	0
Rising temperatures	N/A	N/A	N/A	0
Contaminants	N/A	N/A	N/A	0
Total				48

Figure 11. Threats for southwestern pond turtle from a peer-reviewed threat assessment (Figure 5b from Manzo et al. 2021, p. 493). Manzo et al. (2021) ranked the 13 threat categories using the “sum score,” and resolved ties using the “mean score.”

Based on the best scientific and commercial information available and our assessment of conditions across the range of the two species, we assessed the following threats that we consider are influencing the viability of the western pond turtle:

- Habitat loss and fragmentation (including recreation),
- Altered hydrology (including dams),
- Predation (native and nonnative species),
- Competition (nonnative species),
- Road mortality,
- Collection,
- Contaminants, and
- Effects of climate change (including increasing temperatures, drought, extreme flood events, and high severity wildfire).

Note that some of nomenclature for threats in the literature differs from that used in this SSA report but refer to the same general category of threats. For example, Manzo et al. (2021, p. 487) refers to harvesting, which we refer to as Collection when discussing in detail in this report.

We developed a conceptual diagram to provide a graphical representation of the threats to western pond turtles and how they may impact both habitat and demographic factors (Figure 12). Some threat categories encompass multiple components; for example, recreational activities are considered within the threat of Habitat Loss and Fragmentation. These threats, and the other factors discussed in this chapter, do not affect all western pond turtle populations to the same degree, nor are they evenly distributed throughout the ranges of the two species. Furthermore, the severity of effects may depend on the natural or anthropogenic ecological conditions in each population. We consider all of the threats and associated factors as part of our cumulative assessment of threats facing the northwestern and southwestern pond turtle.

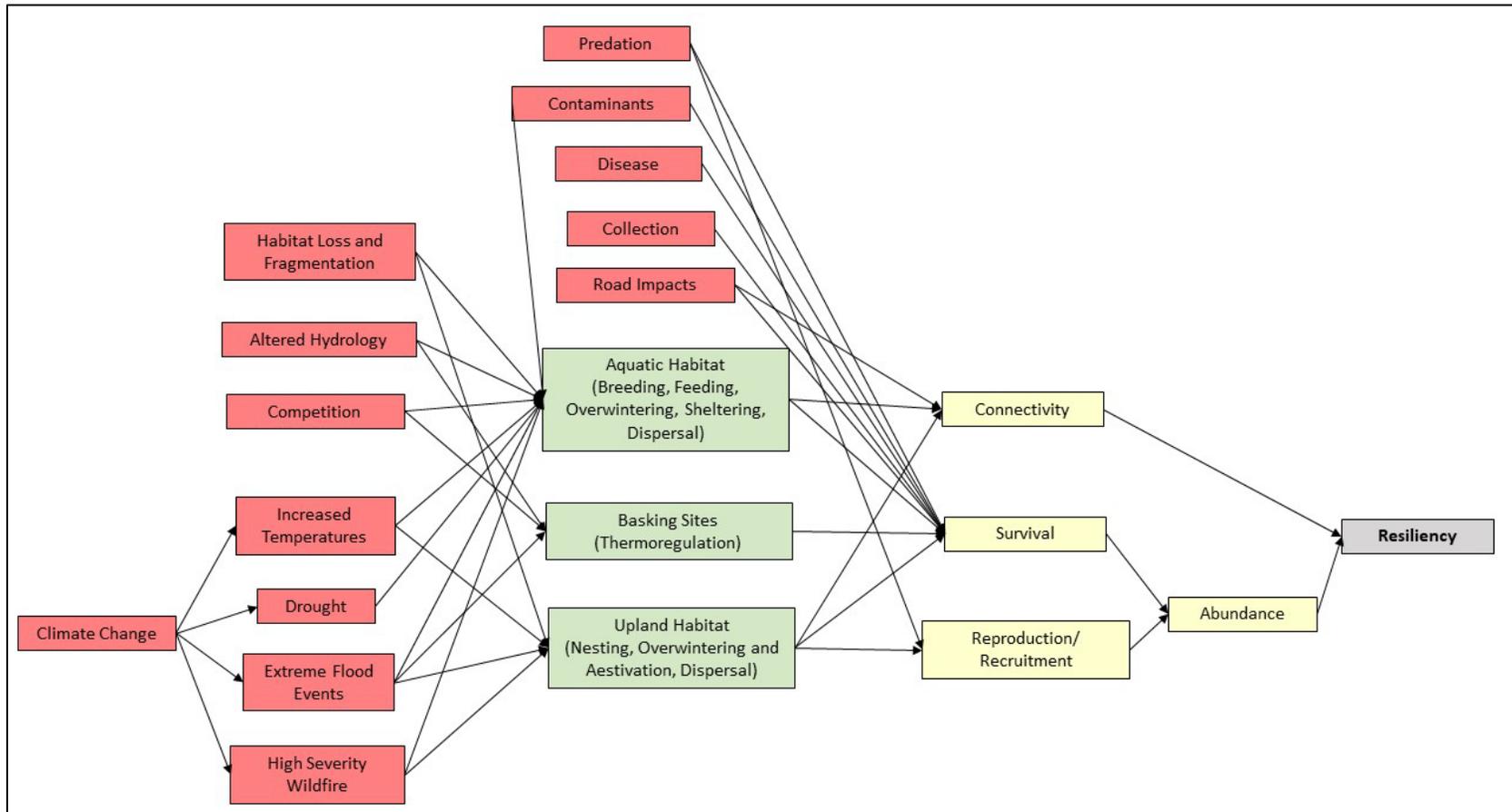


Figure 12. Conceptual diagram of the factors influencing resiliency of western pond turtles. Red boxes are threats, green boxes are habitat needs, and yellow boxes are demographic needs that contribute to resiliency. The conceptual diagram focuses on primary impacts but may leave out some individual-level effects (e.g., wildfires reduce survival of individuals through direct mortality). Synergistic impacts are also discussed in the text that may not be included in this diagram (e.g., drought increases predation when turtles move out of aquatic habitat).

8.1 Habitat Loss and Fragmentation

Extensive land conversion due to urbanization and agriculture has resulted in substantial losses to both upland and aquatic habitats across the range of the two species (Holland 1994, p. 1-23; Hays et al. 1999, pp. ix, 31; Spinks et al. 2003, p. 258; Bury and Germano 2008, p. 001.6; Rosenberg et al. 2009, p. 40; Thomson et al. 2016, pp. 300–301). Across the range, aquatic habitat for the western pond turtle is considered relatively rare and is often limited (e.g., only 1 to 5 percent of land surface area is water) (Bury et al. 2012, p. 12). As a result of habitat loss, in some areas artificial or less desirable habitat such as created stock ponds, agricultural ditches, reservoirs, and ponds associated with wastewater treatment plants are being used by western pond turtles (Germano and Bury 2001, p. 24; Rosenberg et al. 2009, p. 40; Germano 2010, entire; Polo-Cavia et al. 2010, pp. 257–258; Bury et al. 2012, p. 6; Tu and Trulio 2022, pp. 280–281).

Within this section we include discussion of historical habitat loss and the effects of habitat loss. We also include discussion of recreational activities within this section because of its relationship to land use change. Several interrelated factors associated with urbanization compound effects on western pond turtles and associated upland and aquatic habitats, such as roads, altered hydrology, invasive species, and the effects of climate change (drought, floods, and wildfire), which are discussed separately below.

Historical Habitat Loss

The loss of habitat from land conversion (urbanization and agriculture) throughout the ranges of the two species of western pond turtles has affected both aquatic and upland habitats. Loss and alteration of aquatic habitat has been significant throughout the range of western pond turtles due to human development and agriculture (Rosenberg et al. 2009, p. 6). In particular, urbanization has resulted in increased channelization and siltation, a reduction in aquatic vegetation, and fewer or less favorable basking sites (Spinks et al. 2003, p. 258). In Washington, wetland draining, filling, and development eliminated much of the western pond turtle's habitat during the past century (Lower Columbia River Fish Recovery Board 2010, p. B-204). Fifty-nine percent of the state's human population overlaps with the historical range in the lower Puget Sound, and the state continues to become more urbanized (Hallock et al. 2017, p. 10). Pressure from human development has also occurred along the lower Columbia River Gorge (Hallock et al. 2017, p. 10). The most significant example of habitat loss comes from California's Central Valley (Sacramento and San Joaquin Valleys). Kelly et al. (2005, pp. 63, 70), describe the scale of land conversion over the past 100 to 150 years as staggering, with extensive losses to wetlands, riparian and oak woodland forest, grasslands, and shrublands. Over 30 percent of wetlands and deepwater habitats in the Central Valley were lost between 1939 and the mid-

1980's, which represents an average annual net loss of over 5,400 acres for the 46-year period (Frayser et al. 1989, p. 4).

Effects of Habitat Loss

Habitat loss and associated fragmentation has reduced the number of, and connectivity between, western pond turtle populations. The Central Valley in California once represented the stronghold for western pond turtles, with an estimated population size from 10 to 14 million individuals (Holland 1991, pp. 106–107). Upland habitat loss and draining of the extensive wetland for agriculture resulted in the decline and extirpation of many populations and left the remaining western pond turtles in this region in disjunct, scattered populations (Holland 1991, p. 13; Thomson et al. 2016, pp. 300–301). In the southern part of the range, extensive urbanization and land conversion caused precipitous declines, resulting in a large fraction of the remaining habitat in southern California existing only as patches with little suitable upland habitat available for nesting (Thomson et al. 2016, p. 301). Overall, the range of the western pond turtle is fragmented to varying degrees by human activities, with some sites extirpated, and in many cases, only small, isolated groups or individuals remaining across much of the two species' ranges (Holland 1991, p. 13). Currently, populations rarely have densities similar to their historical counterparts, and age structures of extant populations tend to be skewed towards adults (Holland 1991, p. 53; Reese 1996, p. 73; Manzo et al. 2021, p. 493).

Habitat loss and fragmentation have also reduced availability of suitable upland nesting habitat adjacent to aquatic habitat. Although the rate of habitat losses described above, especially that which occurred in the San Joaquin Valley, has diminished to some degree, the lingering effects of past and current habitat loss, alteration, and fragmentation continue to impact the northwestern and southwestern pond turtle. Because western pond turtles typically nest in upland areas near aquatic habitat, suitable nest sites have become increasingly scarce and vulnerable to nearby urbanization and/or agricultural activities (Reese 1996, p. 105).

For example, in Waddell Creek in Santa Cruz County in Central California, changes in agricultural and grazing practices in upland nesting areas appear to have substantially reduced recruitment (Smith 2021, p. 1). During studies from 1995 and 1998/1999, turtles less than 100 mm long were common (25 to 28 percent of the captures) (Smith 2018, p. 1). However, in recent years, identified nest sites were found to be unsuitable for nesting. For example, one of the nest sites was a tomato field that was plowed in fall when turtle nestlings were still in the nest (Smith 2018, p. 1). Only one out of the 24 captures in 2018 was smaller than 100 mm, and only one was a female. The single small turtle found in 2018 suggests low recruitment in comparison to levels from 1995–1999 and a male-biased sex ratio compared to 2007 (Smith 2018, p. 2). In addition to nest destruction and mortality of nesting females, any activity associated with upland habitat, such as overwintering/aestivating in agricultural lands may make western pond turtles more vulnerable to mortality or injury (Gervais et al. 2009 in Rosenberg et al. 2009, p. 46).

Habitat fragmentation because of land conversion has resulted in isolation of populations across the range of the two species. Isolated populations, regardless of size, may no longer have the potential or capability of recolonization through connectivity after catastrophic events, potentially resulting in long-term extirpation of that population. Even in circumstances where there are large populations and recruitment is occurring, such as those reported by Germano and Bury (2001, p. 22) within the Central Valley, if these populations are fragmented, catastrophic events have the potential to reduce or extirpate even stable populations. This in turn may result in inbreeding and loss of genetic variability (Reese 1996, pp. 238–240), therefore reducing adaptability to environmental change. Recent genetic analyses of western pond turtles revealed that some genetic populations have higher levels of inbreeding and lower heterozygosity than others, suggested that inbreeding depression may be occurring (Shaffer and Scott 2022, p. 10). In one example, genetic similarity (e.g., sampled individuals that were more closely related to each other than are full siblings) of western pond turtles in the Columbia River Gorge region, which contained the largest northern populations in Oregon and Washington, may have resulted from fragmentation and isolation that eliminated successful dispersal between sites (Gray 1995, pp. 1250–1251).

Recreation

Recreational activities such as hiking, biking, fishing, boating, and off-highway vehicles, and the associated disturbance within or adjacent to aquatic and nest habitats, can affect western pond turtles in a variety of ways, depending on the region and type of recreation. Some forms of recreation may cause mortality of individuals through trampling, while others degrade habitat, disturb pond turtle behavior, and/or contribute to other threats. For example, recreational activities may interact with the threat of collection because humans may encounter the species while engaging in other activities.

Western pond turtles are extremely wary and will rapidly flee from basking sites into the water when disturbed by the sight or sound of people at distances of greater than 100 m (328 ft) (Bury and Germano 2008, p. 001.5). Western pond turtles at the University of California, Davis, Arboretum were more abundant in basking sites that were farther from human paths, presumably to avoid human disturbance (Lambert et al. 2013, p. 196). In another example, human activity associated with trail use and an adjacent levee road near Moffett Federal Airfield in the San Francisco Bay Area decreased emergent basking by western pond turtles, although in this case there was a higher rate of disturbance associated with vehicular use on the adjacent levee than for trail use by runners, walkers, and bicyclists (Nyhof and Trulio 2015, p. 183). Whether the disturbance is by vehicles or humans, reducing the amount of time performing this behavior has potential effects on metabolism, proper digestion, feeding, reproduction, growth, and predator avoidance.

Recreational use in reservoirs such as boating, fishing, and swimming, especially in those areas with campgrounds, often overlap with shallow and protected areas used by turtles (Hardin 1993, pp. 27–29), which may affect basking and nesting behavior. Recreational fishing can also lead to ingestion of fishing line or lures capturing or injuring of western pond turtles, with evidence of fish hooks found in this species (Lovich et al. 2017, p. 6) and other freshwater turtles (Steen et al. 2014, entire). Stocking of game fish, such as largemouth bass, results in increased presence of nonnative aquatic predators and incidental captures of western pond turtles by fishermen (Hays et al. 1999, p. 21). Off-highway vehicle recreation in or near waterways can result in sedimentation that degrades the aquatic habitat and upland habitat, including nesting areas as well as resulting in potential mortality from being run over. The California Wilderness Coalition (Shore 2001, p. 37) identified several issues related to recreational impacts on western pond turtles in the Knoxville Recreation Area northwest of Sacramento including illegal shooting, trampling of meadows and riparian areas, and accelerated erosion resulting in sedimentation and destruction of western pond turtle upland and aquatic habitat. All of these activities are expected to increase as the human population and urban centers continue to increase in size.

8.2 Altered Hydrology

Aquatic resources used by the western pond turtle have experienced high levels of loss, alteration, and degradation throughout the range of the two species (Reese and Welsh Jr. 1998b, p. 505; Germano 2010, p. 89). A substantial portion of the losses of aquatic habitat are due to anthropogenic water use (e.g., dams and diversions for the purposes of providing water for human use). Moreover, within the historical range of the western pond turtle, an extensive system of hydrologic infrastructure, including dams, reservoirs, diversions, and aqueducts, supports extensive agricultural and municipal water uses, and provides domestic water to many densely populated areas (Lund et al. 2007, p. 43; Hanak et al. 2011, pp. 19–69). These alterations include stream channelization, altered flow regimes, groundwater pumping, water diversions, damming, and water regulation for flood risk management (flood control), which affect hydrology, thermal conditions, and structure of western pond turtle aquatic and upland habitat. More recently, rapid expansion of marijuana agriculture in the western United States is associated with extensive water use. Marijuana farms are slightly closer to streams and rivers than available private parcels (Parker-Shames et al. 2022, pp. 9–11), which has potential implications to freshwater species such as the western pond turtle. Water diversions for marijuana cultivation have decreased stream flow in some areas in Northern California with negative impacts to sensitive fish and amphibians species (Bauer et al. 2015, entire), although we are not aware of specific studies on impacts to western pond turtles. Altogether, hydrologic alterations have contributed to loss of habitat for the species, which is incorporated in the above section, and can have long-lasting impacts in areas where habitat does remain.

Lentic aquatic habitat used by western pond turtles is supported and supplemented by groundwater (Rhode et al. 2019, p. 220). Because groundwater and stream surface-water systems

are connected, groundwater pumping and surface water diversions threaten western pond turtle habitat by depleting water and reducing the amount and duration of surface flows in streams. For example, groundwater pumping has depleted perennial aquatic habitat in the Mojave River, resulting in southwestern pond turtles using artificial ponds and traveling long distances to nest; no juveniles were detected in 1998/1999 despite radiographs of shelled eggs and documented nesting migrations (Lovich and Meyer 2002, pp. 541–543).

Decreases in freshwater inputs into tidal areas is of concern for western pond turtles that use these habitats. In Suisun Marsh, a tidal marsh in California, western pond turtles were more abundant and had higher survival and growth rates at a site with passive management (no water regulation) in comparison to a site with active management (water regulated for seasonal hunting). Basking activity (a proxy for habitat suitability in this study) was greatest when salinity was low and water stage was intermediate (in addition to other common correlates such as temperature), and reduced basking at the site with active management suggested habitat avoidance when environmental conditions were less suitable (Agha et al. 2020, pp. 648–651). In addition to having management implications, this study demonstrates how other alterations to hydrology (e.g., groundwater diversion, drought, climate change) that result in increased salinity can negatively impact western pond turtle habitat, with resultant impacts to populations.

Altered hydrology can be exacerbated or may be compounded by other threats to the species, such as drought and nonnative predators. During drought years, aquatic habitat with water storage reservoirs or dams upstream are more likely to go dry (Meyer et al. 2003, p. 2). For example, in southern California there is often reduced water availability in streams below dams where water is held back and diverted (Madden-Smith et al. 2004, p. 14; Madden-Smith et al. 2005, p. 5). Long-term water extractions/diversions/pumping on streams function similarly to the stream drying that occurs during extended droughts in the way they affect western pond turtles (see Drought section below). Hydrologic infrastructure and management have also been associated with the success of introduced fishes and amphibians (see *Nonnative Predators* section), many of which compete with and prey on native wildlife including western pond turtles (Moyle 1973, p. 21; Holland 1991, pp. 54–57; Holland 1994, pp. 2-11–2-12; Hays et al. 1999, pp. 13–14; Spinks et al. 2003, pp. 264–265; Cadi and Joly 2004, pp. 2515–2517).

Altered hydrology can also impact western pond turtle eggs if it results in habitat getting too much water at the wrong time of the year. Western pond turtle eggs have permeable shells that have been observed to rupture after absorbing excess moisture, killing the pond turtle embryo (Feldman 1982, p. 10). For example, this could be a problem in urban areas that are irrigated (Spinks et al. 2003, p. 263)

Dams

In general, dams and water diversions on rivers can act as barriers to migration, create stretches of unsuitable habitat (Reese and Welsh Jr. 1998a, p. 851) and/or degrade or eliminate habitat (Holland 1994, p. 1-29). Water behind dams experience artificially fluctuating water levels, which can affect aquatic and riparian vegetation and western pond turtle prey resources (e.g., invertebrates) (Madden-Smith et al. 2005, p. 5). Below dam-regulated stream systems (including reaches where stream low-flows are not largely diverted), flood risk management dam operation activities reduce the extent and frequency of flows in streams and floodplains that previously removed sedimentation and provided ephemeral floodplain open areas, resulting in dense vegetation encroachment that degrades suitable turtle nesting habitat (Ligon et al. 1995, entire; Madden-Smith et al. 2005, p. 5; Rosenberg et al. 2009, p. 40; Williams and Wolman, entire).

For northwestern pond turtles, dams appear to negatively influence recruitment. On the Trinity River in northern California, a dammed tributary had more sedimentation, decreased water temperatures, and increased canopy cover in comparison to an undammed tributary (Reese and Welsh Jr. 1998a, pp. 842, 847–848). Colder water temperatures likely contributed to the slower growth rate of western pond turtles in the dammed tributary (Ashton et al. 2015, p. 624–628). Additionally, populations in the dammed tributary had fewer juveniles than those in the undammed tributary (Reese 1996, pp. 43–44; Reese and Welsh Jr. 1998b, p. 513; Ashton et al. 2015, p. 626). However, at another river in northern California, the operation of a dam resulted in an intermittent stream becoming a perennial stream, providing for increased food availability, which allowed western pond turtles to grow larger. Similar to the study at the Trinity River though, there were fewer juveniles below the dam, which suggests an effect to recruitment (Bondi and Marks 2013, pp. 146–149). In other species of turtles, negative impacts from dams include reduction in nesting area (Norris et al. 2018, pp. 7–9), reduced diversity and extent of turtle diet (Tucker et al. 2012, pp. 15–19), and lower densities of turtles in oxbow lakes (in comparison to flowing river segments) both upstream and downstream of a dam (Selman 2020, pp. 190–191).

8.3 Predation

Western pond turtles are impacted by both nonnative and native predators. Nonnative predators include American bullfrogs (*Lithobates catesbeianus*; hereafter bullfrogs) and invasive fish, such as large and smallmouth bass (*Micropterus* sp.; hereafter bass). Native predators of western pond turtles include raccoons, skunks, foxes, coyotes, mink, herons, river otters, burrowing small mammals, and giant water bugs.

Nonnative Predators

Nonnative predators in western pond turtle habitat influence the species by increasing predation pressure on hatchlings and young juveniles. Increased predation beyond the natural levels under which western pond turtles evolved results in reduced survival and reproduction, affecting population recruitment and abundance, which in turn, lessens overall resiliency. Additional effects beyond those in natural settings are amplified when considered with other factors contributing to reduced recruitment and survival, such as urbanized areas with increased nest predators, altered hydrology, and drought. Only 10 to 15 percent of western pond turtles less than three years of age or approximately 94 mm CL survive annually (Holland 1994, p. 2-11). Therefore, additional predation pressure may result in fewer hatchlings surviving to reproductive age and recruitment failure.

Bullfrogs and bass have been linked to predation and potential decreases in recruitment of western pond turtles, although there is more literature available about bullfrogs (Moyle 1973, p. 21; Holland 1991, p. 43; Hays et al. 1999, p. 14). Some papers actually mention a lack of bass predation on other species of hatchling turtles outside of the range of the western pond turtle (Semlitsch and Gibbons 1989, pp. 1030–1031; Britson 1998, p. 386), and from here on we focus on bullfrogs in this section. Bullfrog farming for commercial use began in the United States prior to 1900, and Dodd and Jennings (2021, p. 77) report on bullfrog brochures or similar announcements related to farms in each of the states occupied by western pond turtles, leading to rapid spread of the nonnative species. The earliest known transport of bullfrogs to California was in 1896, when they were beginning to be raised on California ranches for human consumption (Heard 1904, p. 24; Jennings and Hayes 1985, p. 98). Bullfrogs have since become widespread throughout much of the western pond turtles' range (Holland 1991, p. 40). The spread of bullfrogs is facilitated by altered hydrology, land-use change, and increasing water temperatures (Moyle 1973, p. 21; Fuller et al. 2011, pp. 210–211). Although the effects of bullfrogs on western pond turtles are difficult to distinguish from co-occurring factors influencing viability, research indicates that bullfrogs and potentially other invasive predators may play an instrumental role in western pond turtle population declines through predation on hatchlings (Holland 1994, p. 2-12).

Teasing apart the impacts of nonnative predators from other factors may best be observed by testing the effects of removing them from the system and measuring the response by western pond turtles. For example, at Sycuan Peak Ecological Reserve in San Diego County, California, removal of invasive predators including bullfrogs resulted in observations of hatchling and young juvenile western pond turtles (less than 80 mm CL) for the first time in over a decade (Brown et al. 2015, pp. 24, 110). In another example, an all-life stage bullfrog removal initiated in Washington in 2014 resulted in an observed reduction in bullfrogs and an increase in western pond turtle hatchlings in 2016 (Hallock et al. 2017, pp. 13–14). While there has been some progress in western pond turtle recovery, natural recruitment remains low in Washington in some areas due to low hatching success and predation on hatchlings (Hallock et al. 2017, p. iv). In another promising success story, in Yosemite National Park a removal effort initiated in 2005 led

to the first successful eradication of bullfrogs at the landscape level after bullfrogs were intentionally introduced in the 1950s (Kamoroff et al. 2020, entire). However, in many cases after successful eradication in a local area, control efforts may be challenged by repeated introductions or invasions if bullfrogs are present in surrounding areas.

Native Predators

Western pond turtle viability can be negatively influenced by native predators, especially near urbanized areas where increased predation is caused by meso-predator release and/or predator subsidies (Soule et al. 1988, p. 84; Marchand and Litvaitis 2004, pp. 758–759). Predators such as raccoons, skunks, foxes, and coyotes are known to predate western pond turtle nests (Holland 1994, p. 2-12). Because habitat loss and degradation may result in reduced nesting habitat, these predators may cause elevated nest predation as they concentrate their foraging efforts in these remaining areas (Geller and Parker 2022, pp. 1–2). Raccoons are commonly mentioned as one of the leading nest predators, and have been reported as one of the primary predators causing decline in several other turtle species (Christiansen and Gallaway 1984, entire; Browne and Hecnar 2007, p. 426). Including other nest predators, Holland (1994, p. 5-8) states that predation on western pond turtle nests in some areas may be a primary factor in recruitment declines. For example, observations at a site in Oregon in 1991 indicated that an estimated 90 percent of the nests were preyed upon. This pattern was confirmed with similar and more extensive observations of predation at several sites in the Willamette and Umpqua drainages in 1992/1993 (Holland 1994, p. 5-8). Throughout the range of the western pond turtle, Holland (1994, p. 5-8) notes that in 1992, of the 106 nests discovered, 97 had been predated. However, predated nests are inherently easier for humans to detect than undisturbed nests (Bettelheim et al. 2006, pp. 214–215).

While predation may be heightened in urbanized areas, especially affecting nests, the threat from natural predators on adults in the wild may pose little risk to populations of western pond turtle when acting alone. However, when acting in concert with other factors such as long-term drought, risk from predation may increase (Leidy et al. 2016, pp. 72–73; Purcell et al. 2017, pp. 21–22). In general, the risk of increased predation may occur during drought because western pond turtles spend more time on land or moving between aquatic habitats. For example, during a drought from 1987–1991, western pond turtles in drying aquatic habitat increased movement within stream systems, resulting in heavy predation by bears and coyotes, and causing one population to be effectively extirpated (S. Sweet, pers. comm in Holland 1991, p. 54). The effect of drought and resultant observations of potential increased predation are further discussed below under Climate Change. Within parts of the northwestern pond turtles' range, river otters are known to prey upon western pond turtles by severing one or more limbs (Holland 1991, p. 40; Studebaker 2008, pp. 463–464; Green Diamond 2022, in litteris). According to researchers along the Mad River in northern California, these turtles are not always killed and appear in good health after recovery from these injuries; however, the frequency of these injuries suggest that

some individuals within the population do not survive from mammalian predation events (Green Diamond 2022, in litteris). Reproductive females are frequently observed with extensive scarring on the shell indicating attempted predation (Holland 1994, p. 2-12), most likely a result of time spent on land during nesting activities. In small populations or those that may be male-biased, predation may be of greater concern as the loss of a breeding female can affect recruitment. Injuries were noted in the majority of western pond turtles captured at three sites along the Mojave River, with no difference based on sex or mean carapace length of injured or noninjured turtles (Cummings et al. 2022, pp. 723–725)

Effects of Elevated Predation Levels

Elevated rates of predation pressure, regardless of whether it is from nonnative or native predators, results in increased pressure on survival, and ultimately impacts recruitment into the population. These affects are further compounded when considered together with other factors influencing survival such as habitat loss, altered hydrology, contaminants, recreation, and effects of climate change. Increased predation is mentioned extensively throughout the literature as one of the influences affecting western pond turtle survival and recruitment (Rosenberg et al. 2009, pp. 40–41). Regardless of the origin of the predator, predation appears to be a major factor in western pond turtle persistence, particularly when acting in concert with other threats (Manzo et al. 2021, pp. 492–493).

Nonnative predators occur throughout the range of both species in aquatic environments, and both native and nonnative predators are more common in areas near urbanization (often within or near areas of artificial hydrology) than in remote areas. Within the aquatic environment, hatchlings and small juveniles are most at risk, whereas in the uplands, all life stages are vulnerable to predation. Females seeking nesting sites in the uplands are more vulnerable during this stage when more time is spent on land. However, predation of any life stage increases in uplands during migration activities (aestivation, overwintering, males seeking mates, etc.) and dispersal events as they are easier targets and susceptibility to predation increases (Leidy et al. 2016, p. 72). This is also true during times of reduced water availability, such as natural drying of aquatic resources or from drought (see Climate Change below), when western pond turtles often congregate in remaining pools or spend more time on land, including exposed shorelines. With limited resources in the environment, terrestrial predators may also be drawn to remaining water on the landscape, increasing potential for higher levels of predation at these locations.

8.4 Nonnative Species Competition

Competition with nonnative species may be a threat to the western pond turtle, particularly when resources are otherwise limited, such as basking sites and/or prey items. The red-eared slider has been identified as the main potential competitor for western pond turtles, but direct evidence of competition is limited.

Red-eared sliders are listed as one of the “world’s worst invasive species” by the International Union for Conservation of Nature (IUCN) (Lowe et al. 2000, p. 6). They are common in areas near dense human populations, with red-eared slider numbers likely reinforced by releases or escapes of pets (Thomson et al. 2010, p. 300; Lambert et al. 2013, p. 196). Because red-eared sliders are often found in habitat heavily degraded by human activities, identifying the negative impacts from red-eared sliders versus effects from other coexisting threats can be difficult, especially in complex environments (Dupuis-Desormeaux et al. 2022, pp. 2–3). However, red-eared sliders have been tied to declines in Sonora mud turtles (*Kinosternon sonoriense*) because of interference for basking sites in a before-after study in an undisturbed natural environment (Drost et al. 2021, entire). Under experimental conditions, red-eared sliders negatively impacted weight and survival of European pond turtles (*Emys orbicularis*) (Cadi and Joly 2004, pp. 2514–2515) and negatively impacted basking activity for Spanish terrapins (*Mauremys leprosa*) (Polo-Cavia et al. 2010, p. 2144–2147).

Red-eared sliders may impact western pond turtle occupancy and behavior, but specifics are difficult to generalize. Both species were found in surveys in agricultural lands across the Sacramento Valley and the Sacramento-San Joaquin River Delta, with western pond turtles more commonly found in wide canals farther from urban areas, and red-eared sliders found in wetlands near major roads and urban areas (Fulton et al. 2022, pp. 102–104). In these surveys, western pond turtles and red-eared sliders only co-occurred at 6 sites. The authors caution that the lack of a relationship between red-eared slider occurrence and western pond turtle occupancy in this study does not necessarily mean that red-eared sliders do not have negative impacts on western pond turtles. Indeed, red-eared sliders may have outcompeted western pond turtles in human-dominated environments where the former species was more common, and/or may continue to expand their range beyond introduction sites near major roads, eventually occupying the irrigation canals habitat occupied by western pond turtles (Fulton et al. 2022, p. 106). Both species have co-occurred in Clear Lake, California for more than 50 years with no clear evidence of strong competition. The opportunistic nature of the data collection from 2010–2018 makes it unclear whether western pond turtles have declined, remained stable, or even increased at Clear Lake during this time (Hayes et al. 2018, p. 311). However, the short duration of this study relative to the generation time of western pond turtles makes it hard to make definitive conclusions. At the University of California, Davis, Arboretum, removal of red-eared sliders improved body condition of western pond turtles through weight gain, suggesting either indirect or direct competition with red-eared sliders for food. Western pond turtles also shifted their basking behavior after red-eared slider removals, but surprisingly, the behavioral shift was not consistent with strong competition (Spinks et al. 2003, pp. 264–265; Lambert et al. 2019, pp. 11–12).

Additional invasive species that may compete for resources include, but are not limited to, bullfrogs, bass, and crayfish, but direct evidence of competition between these species and western pond turtles is not available. It is possible that competition for prey may occur

depending on availability and abundance. Availability of prey, particularly small invertebrate prey, appears to be important for the survival of all mobile life stages of the western pond turtle.

8.5 Disease

Disease has been and is an emerging concern for western pond turtle populations. Documented diseases in western pond turtles include respiratory disease and shell disease. In 1990, an unidentified pathogen causing an upper respiratory disease killed more than a third (at least 36 individuals) of the extant western pond turtles in Washington at that time (Hays et al. 1999, p. 14; Hallock et al. 2017, p. 9). Additional cases of respiratory disease have not been detected in Washington, but two cases were documented in Oregon (B. Bury, pers. comm. in Hallock et al. 2017, p. 9). Pathogen sampling in western pond turtles in California detected *Mycoplasma* species (a type of bacteria) in northern, central, and southern California turtles, with infected turtles having lower body weights. They did not find evidence that sympatry with nonnative red-eared sliders correlated with pathogen occurrence (Silbernagel et al. 2013, pp. 41–43). We are also aware of leeches, including in the genus *Hellodella* and the introduced *Placobdella parasitica*, that have been reported on western pond turtles (S. Barnes 2023, in litteris; D. Ashton 2023, in litteris), but the ecology between these leeches and western pond turtles is poorly understood and is not discussed further in this report. For the remainder of this section, we focus on shell disease.

Emydomyces-associated shell disease has been diagnosed in over 22 species of aquatic and semi-aquatic turtle including free-living turtles (Woodburn et al. 2021, pp. 580–582). Shell disease associated with the fungus *Emydomyces testavorans* (*Emte*) has been documented in both wild and headstarted western pond turtle populations in Washington, and in free-living red-eared sliders within the same pond as western pond turtles in California (Hallock et al. 2017, p. 9; Haman et al. 2019, entire; Lambert et al. 2021, entire; Woodburn et al. 2021, entire). The fungus is closely related to the causative agent of Snake Fungal Disease (genus *Ophidiomyces*) (Haman et al. 2019, p. 500; Woodburn, Miller, et al. 2021, p. 9). Although associated with shell disease, *Emte* has not been verified as the sole causative agent of shell disease in western pond turtles (Haman et al. 2019, p. 500).

The shell disease manifests itself primarily through shell degradation, and results in shell pitting, lesions (sometimes penetrating into the internal cavity of the turtle), and hollow areas beneath keratin, as well as other shell deformities or weaknesses (Hallock et al. 2017, p. 9; Haman et al. 2019, p. 497; Lambert et al. 2021, p. 458; Woodburn et al. 2021, pp. 580–582). However, the gross appearance of the disease is often subtle with few traits that would suggest the presence of severe lesions below the surface of the scutes. Superficially this disease may sometimes appear similar to Septicemic Cutaneous Ulcerative Disease (a bacterial infection), a disease observed in captive turtles (Haman et al. 2019, pp. 497–498). Woodburn et al. (2021, p. 582) also identified characteristic shell lesions associated with *Emte* infection (i.e., epithelial inclusion cysts).

Shell disease has most likely been present in Washington since at least 2003 based on photographic evidence going back to that year (Hallock et al. 2017, p. 9; Haman et al. 2019, p. 497). In Washington, shell disease is present at all six sites with western pond turtles (Holman and Anderson 2014, p. 3; Schmidt and Tirhi 2014, p. 13; Schmidt and Tirhi 2015, p. 11; Haman et al. 2019, entire). CT scans taken during health assessments for shell disease from 2015–2019 revealed that most turtles CT scanned (199/232; 86%) had at least one lytic lesion. For turtles with multiple scans, shell disease increased in lesion extent over time. All turtle lesions tested by qPCR (48/48; 100%) were positive for *Emte* DNA. The disease was primarily observed in headstarted turtles (those hatched and/or raised in captivity for 10–12 months then released into the wild). Evaluation of wild turtles in Washington was limited as the six remaining sites are now composed almost entirely of headstarted turtles, with less than about 30 wild adults captured in recent years. The disease was identified in a small number of adults that were never headstarted but were kept in a captive breeding program. One turtle, that was never documented to spend time in captivity, had shell disease. Of the turtles included in the study, individuals with shell disease were more likely to have been in captivity than those without shell disease (OR=6.5, p-value = 0.0003; K. Haman, unpublished data).

Some western pond turtles do not show clinical signs of shell disease, suggesting a potentially higher prevalence of the disease than can be observed through external examination (K. Haman, pers. comm. in Hallock et al. 2017, p. 9). Based on photographs and scans, the disease may progress slowly with some western pond turtles having the disease for more than a decade. Severity varies and the effects of the shell disease on life span, reproduction, and recruitment in the wild are unknown. The extensive shell damage in some animals suggests some with disease will result in mortality (Hallock et al. 2017, p. 9). In 2022, two western pond turtles with shell disease died after being brought in for treatment, and an additional two sick western pond turtles were euthanized after being brought in for examination. The latter two individuals had severe shell disease and had lost weight since their most recent capture several months earlier (Bergh and Wickhem 2022, p. 6). In 2020, *Emte* was first documented in free-living turtles in California (Lambert et al. 2021, p. 960). In this study, red-eared sliders tested positive for the disease (n=3), though only 1 had evidence of the disease. In contrast, western pond turtles did not test positive, despite one western pond turtle having shell bleaching, which is thought to be a symptom of early shell disease. CT scans may be necessary to determine if abnormal shells observed in the western pond turtles were in fact the result of this disease but went undetected through the sampling method used. Recent unpublished data indicates that *Emte* is present in multiple western pond turtle populations around the Bay Area, including areas that contain western pond turtles but not red-eared sliders (H. Anderson 2023, in litteris) More study is needed to determine the status of diseases on wild populations of western pond turtles across their range.

Disease is a current and future threat for both northwestern and southwestern pond turtles, but is of greater concern for the former because of the prevalence of shell disease in headstarted western pond turtles and the reliance of Washington recovery sites on continued headstarting.

8.6 Road Impacts

Although roads are tightly linked to urbanization and development, roadways also exist as a standalone threat since their presence is not always associated with urban or developed areas. Thus, we address roadways independently but also consider that the effects are synergistic with urbanization. Roads can affect western pond turtle viability because of vehicles killing or injuring individuals or disturbing basking behavior, and by reducing connectivity between populations, which reduces migration between upland and aquatic habitat (Rosenberg et al. 2009, p. 41; Nyhof 2013, p. 43; Thomson et al. 2016, p. 301; Nicholson et al. 2020, entire; Manzo et al. 2021, p. 494, S1 text supplement). Railroad tracks can also serve as barriers to migration, as is observed in other turtle/tortoise species (Rautsaw et al. 2018, pp. 138–139). There have been no documented western pond turtle population extirpations attributed directly to roadways. Additional threats that have associated effects with roads include: increased fragmentation (roads further break up the landscape), recreation (roads increase access to habitat), collection (roads increase access for humans), contamination (runoff of contaminants), and predation (roads increase access for predators), and an interaction with drought (drought causes turtles to spend more time in upland habitat, increasing potential to interact with roads). Despite the high likelihood that these threats have compounding impacts, there is limited direct evidence in the literature about their combined effects on western pond turtle.

Many direct mortality events have been documented on roads, but these effects have mostly been documented at the individual rather than the population level. Although roads are known to create dispersal barriers, there are no formal assessments of the impact of roads on connectivity of western pond turtles at the population level. Thus, it is difficult to assess the impact of roads on population-level parameters. However, in a road risk assessment ranking susceptibility of California herpetofauna to road mortality and habitat fragmentation, Brehme et al. (2018, p. 921) classified northwestern pond turtles and southwestern pond turtles as very high risk (both in the top 10 out of 160 species evaluated).

There is some support for the hypothesis that more roads have led to demographic changes in populations, including increasingly male-biased sex ratios. Using museum specimens as unbiased samples (in comparison to field collections and trapping seasons), Nicholson et al. (2020, pp. 11–13) found that populations of western pond turtles within 219 m of roadways often have heavily male-skewed sex ratios, and suggest that this correlation was consistent with road mortalities of females (Nicholson et al. 2020, pp. 12–13). Nicholson et al. (2020, pp. 13, 16) also found that populations of western pond turtles became increasingly male-biased as the proportion of land covered by roads within 400 m of a water body increased. Studies of other species have suggested that road kills result in male-biased turtle populations because nesting female turtles are more likely to be hit and killed by passing vehicles as they travel upland in search of suitable nesting sites (Steen and Gibbs 2004, pp. 1145–1146).

In Oregon, there are numerous reports of individual western pond turtles found dead on or alongside of roads, often in areas where roads bisect nesting habitat (examples in Rosenberg et al. 2009, p. 41). Over a four-month period in the Willamette Valley in Oregon, 25 individuals were found crossing a road and brought to the Oregon Department of Fish and Wildlife or the Northwestern Ecological Research Institute; of these, at least 3 were killed. Assuming 25 to 50 turtles are potentially lost each year, this would represent an annual potential loss of 3 to 5 percent of the Willamette Valley population at that time (Holland 1994, p. 2-13).

In California, western pond turtles left basking sites significantly more often in response to motorized vehicles compared to recreational users along the San Francisco Bay Trail (Nyhof and Trulio 2015, p. 183). As motorized vehicles passed, turtles left their basking site 45 percent of the time (Nyhof and Trulio 2015, p. 183). Overall, disturbed western pond turtles had shorter basking periods compared to undisturbed. Less basking affects thermoregulation, which can lead to required physiological processes not being met. This study also found a male-skewed ratio, suggesting road mortality among nesting females (Nyhof 2013, p. 45).

A more general study across nine watersheds in San Diego County, California investigated the synergistic impact of collection and roadways with drought. Relatively few juveniles and female turtles were captured or documented during the study, which may have resulted from human activities (i.e., collection facilitated by a dense network of roads) (Madden-Smith et al. 2005, p. 85). Since females travel upland for nesting sites, they may be more vulnerable to collection by humans and/or getting struck by vehicles. As noted earlier, this could lead to high male bias and less recruitment (Madden-Smith et al. 2005, pp. 43, 45).

8.7 Collection

Collection of western pond turtles directly removes individuals from a population and can lead to reduced reproduction and recruitment. This is especially the case in populations that are fragmented or where numbers of individuals are already low. Extensive collection is widely reported along with habitat alteration and habitat loss (discussed above) as primary factors initially responsible for declines of both species (Holland 1994, p. 2-13; Hays et al. 1999, p. 16; Bettelheim 2005, entire; Rosenberg et al. 2009, p. 42; Thomson et al. 2016, p. 301). The true extent of these declines associated with collection remains largely unknown. However, Bettelheim (2005, entire) and Bettelheim and Wong (2022, entire) provide a thorough review of collection for commercial harvest occurring between the mid to late 1800s and early 1900s. At the height of collection in 1895, approximately 63,000 individuals from the San Francisco Bay area and Central Valley of California were marketed (Bettelheim and Wong 2022, p. 9). This was followed by approximately 53,935 individuals marketed for several years until the turn of the century from San Joaquin, Solano, Sonoma, Stanislaus, and Contra Costa Counties for the commercial terrapin fishery in California (Bettelheim 2005, p. 32; Bettelheim and Wong 2022, p. 9). Numbers in the thousands from several counties throughout California accounted for turtles

collected for the San Francisco market with large numbers coming from the Sacramento and San Joaquin regions (Bettelheim 2005, pp. 32–33). In 1883, one trapper on Tulare Lake collected a minimum of 3,600 individuals. In 1904, not accounting for other collections during that time, an estimated 12,740 individuals were collected from San Joaquin and Sacramento Counties (Bettelheim 2005, pp. 32–33). Bettelheim and Wong (2022, p. 10) suggest that historic collection between 1863 and 1931 resulted in the collection of approximately 524,100 individuals and could be over a million individuals (Bettelheim and Wong 2022, p. 10) collected for the San Francisco market, and likely other markets in California, Oregon, and Washington (Holland 1991, p. 44). Collecting for commercial harvest likely had an impact on turtle populations by removing a greater number of reproductively viable adults and, consequently, acted as an intense population suppressant (Bettelheim and Wong 2022, p. 11).

While the impact of collection on western pond turtles has declined, it is still currently occurring, typically for the pet trade, food, or for use as a personal pet. For example, ODFW has records of western pond turtle hatchlings that were collected as pets, individuals collected while crossing the road, etc. (Barnes in litteris 2023). As mentioned above, the proximity to urbanized areas or roads probably contributes to collection. For example, in the 1980s and 1990s, two herpetologists witnessed individuals filling burlap sacks with western pond turtles from Piru and Sespe Creeks, and from the Santa Ynez River in Santa Barbara and Ventura Counties (S. Sweet pers. comm. in Bettelheim 2005, p. 42). Potential collection from a remnant slough surrounded by agricultural lands in western Fresno County (adjacent to Highway 180) may have caused reduced numbers and decline of a population of western pond turtles and thereby increased its risk to extirpation due to its isolation and lack of surrounding habitat (Germano 2021, p. 240).

8.8 Contaminants

Although western pond turtles are exposed to a variety of toxins throughout their range, sensitivity of individuals to pesticides, heavy metals, pollutants, and other contaminants is largely unknown. However, contaminants in general have been identified as a significant threat in freshwater ecosystems both through indirect or through direct toxicity to organisms (Reid et al. 2019, p. 9). Potential affects to long-lived species such as the western pond turtle are discussed in Rowe (2008, entire). For example, because western pond turtles take multiple years to reach reproductive maturity (see Chapter 5.0 Life History), potential effects from contaminants include mortality before reproduction, or chronic accumulation of contaminants that could be transferred to offspring (Rowe 2008, p. 626).

Sources of contaminants affecting western pond turtles include run-off or drift from agricultural activities, run-off from mining sites, diesel spills, run-off from urbanized areas, and roadways. Pesticides and mercury are the most studied contaminants, but little to nothing is known about the biological implications. For example, variable amounts of organochlorine pesticides, polychlorinated biphenyls (PCBs), and mercury were detected in western pond turtle eggs at a

site in Oregon, but differences in concentrations of these contaminants were not related to egg hatchability in the study (Henny et al. 2003, pp. 49–51). Contaminants can be toxic to aquatic prey items of western pond turtle such as amphibians and small invertebrates (Davidson 2004, p. 1892; Relyea 2005, p. 1118; Brühl et al. 2013, p. 1). Thus, a potential reduction of prey due to contaminants may have negative impacts at the individual and population level of western pond turtle. Per- and polyfluoroalkyl substances (PFAS) are common contaminants in the environment that bioaccumulate in other turtle species, with negative metabolic impacts for individuals (Beale et al. 2022, entire). The specific impacts from PFAS to western pond turtle populations are not known.

Pesticides are of particular concern as their use in California has historically been and continues to be widespread, and they can expand beyond the area to which they are applied via spray drift, sorption, leaching, volatilization, and surface runoff (Majewski and Capel 1995, entire; Tudi et al. 2021, pp. 6–8). Differences in exposure to pesticides depend on the proximity of the population to agricultural pollution. For example, pesticides (semi-volatile organic compounds; SOCs) were detected in the plasma of populations of western pond turtles at higher concentrations in two sites closest to agricultural sources (Meyer et al. 2016, p. 330). Some pesticides, such as organophosphates and carbamates, are known to inhibit cholinesterase enzyme (ChE) in wildlife, thus ChE activity can be used as an indicator of pesticide exposure (Meyer et al. 2013, p. 692). Western pond turtles from areas within the Sierra Nevada had significantly depressed ChE activity by 31 percent compared with other areas farther north in the range (Meyer et al. 2013, pp. 695–696). Despite direct evidence of the presence of ChE depression occurring in the northwestern pond turtle, the effects of it are still unknown in the species. However, it could impact neurotransmission and neuromuscular function (Meyer et al. 2013, p. 696).

In addition to pesticide exposure in agricultural areas, additional noteworthy sources of contaminants are old mines and diesel spills. Mercury has been found in western pond turtles and is still found in ecosystems surrounding historic gold mining sites throughout California (Meyer et al. 2014, p. 2994) and historic mercury sulfide (cinnabar) mining sites in Santa Clara County in California (Service 2013, pp. 43–44; AECOM 2021, p. 1). Elevated concentrations of mercury, lead, and arsenic have been found in fish and waterfowl species in the Carson River area in Nevada, but western pond turtles in this area have not been tested (NDOW 2022, in litteris). In several populations of western pond turtles, blood plasma analyses revealed consistent relationships between mercury concentrations in red blood cells and evidence of disruption of thyroid hormones, which are known to be critical to growth, development, and reproduction (Meyer et al. 2014, p. 2994). It is unknown at what level of exposure to mercury and/or pesticides would have biologically detrimental effects at the individual level or population level in western pond turtles. Also, even when contaminants occur in blood at concentrations below many diagnostic thresholds, it is possible that multiple contaminants at low concentrations could interact synergistically (Meyer et al. 2016, p. 333). In a study documenting a variety of

contaminants (organochlorines, PCBs, and metals) in Eugene, Oregon, no relationship was found between egg hatchability and contaminant levels. However, these contaminants are known to disrupt proper sexual development, immune function, or survival of hatchlings. Although diesel spills in freshwater are uncommon, there is evidence that when they do occur they can result in mortality. In California, a diesel spill from a truck into freshwater resulted in mortality of at least one small individual western pond turtle, and negatively impacted the health and behavior of other individuals that were observed (Bury 1972, p. 294). In Oregon, at Yonkalla Creek, in January of 1993, a diesel spill resulted in the death of least 50 (and probably in excess of 100) northwestern pond turtles (Holland 1994, p. 2-13). Of an additional 30 animals collected, 3 died due to delayed reactions and complications.

8.9 Climate Change

Climate change is defined by the Intergovernmental Panel on Climate Change (IPCC) as the change in the mean or variability of one or more measures of climate that persist for an extended period, whether the change is due to natural variability or human activity (IPCC 2015, p. 120). Overall trends in climate across the range of the western pond turtle include increasing temperatures, greater proportion of precipitation falling as rain instead of snow, earlier snowmelt, and increased frequency and severity of extreme events such as droughts, heat waves, wildfires, and floods (Bedsworth et al. 2018, pp. 19–33; Oregon Climate Change Research Institute 2019, pp. 5–7). The increased frequency and severity of extreme events increases extirpation risk of western pond turtles from catastrophic events. Impacts in climate trends and change are expected to vary throughout the range of the species.

Increasing Temperatures

Western pond turtle nest sites and embryo development are sensitive to temperature because the species exhibit TSD (Ewert et al. 1994, p. 7) (see *Nesting/Hatching*). In a field study, female hatchlings were more likely when 30 percent of the sex-determining period occurred above 29° C (84° F) (Christie and Geist 2017, p. 49). Therefore, an increase in temperature may result in skewed sex ratios with higher numbers of females being produced. In the same study, lower fluctuation in temperature resulted in development in males, whereas females developed in nests with high and low fluctuations (Christie and Geist 2017, p. 49). The maximum incubation temperature expected to result in a 90 percent rate of inviability was 45°C (113° F) (Christie and Geist 2017, pp. 49, 51). Therefore, increases in maximum temperatures could potentially result in consequences to western pond turtle demography and reduced number of hatchlings being produced. If habitat is available, females may be able to select for areas with reduced temperatures, such as areas providing less direct sun exposure, although it is unclear to what degree females select nest sites based on microhabitat differences related to temperature. There is some evidence in other turtle species that plasticity in female nesting behavior could compensate

to some degree to changes in climatic changes to the environment; adaptive potential of this as well as other traits is discussed in Refsnider and Janzen (2016, pp. 64–66).

There is also the potential that rising temperatures could increase the number of warm days for developing embryos, potentially enhancing reproductive success in the wild for individual northwestern pond turtles at the northern extent of the range (Washington Department of Fish and Wildlife 2015, p. C–56). In other species of reptiles, studies focusing on the active season predict a largely positive response to warming because increases in temperature can prolong the active season, but results from a meta-analysis of winter warming on reptile traits were less clear, with some positive but some negative effects (Moss and MacLeod 2022, p. 264–266). Benefits to reproduction may be outweighed by the cumulative negative impacts (e.g., isolation of populations, skewed sex ratios, loss of aquatic habitats, etc.) to individuals and their habitats.

Drought

Desiccation of waterways from drought has led to declines and extirpations of western pond turtle populations by negatively affecting the quality and/or quantity of its aquatic habitat, impacting survival, recruitment, and connectivity, and exacerbating the effects of other threats. Western pond turtle mortality during drought is well documented, and appears to occur as a result of drought-induced starvation (Lovich et al. 2017, p. 7) and/or drought-induced predation (Purcell et al. 2017, p. 21). Goodman Jr (1997, p. 23) documented a size-class distribution in western pond turtles suggesting lower reproductive output and a gap in recruitment during drought in the late 1980s. Although the exact drivers are not clear, several factors could lead to lower reproductive output and lower recruitment during drought, including fewer mating opportunities (since they mate in aquatic habitat), deferred reproduction in response to reduced resources (Pires 2001, pp. 42–43), and high drought-related mortality. Negative effects appear to prevail despite the potential for drought to reduce aquatic predator abundance (U.S. Forest Service, Los Padres National Forest 2022, in litteris).

The frequency, severity, and/or duration of drought are expected to increase in response to climate change. California has experienced extreme drought conditions in recent decades, including in 2007–2009 and 2012–2014 (Williams et al. 2015, pp. 6823–6824). Anthropogenic warming likely contributed to the 2012–2014 drought anomaly (Williams et al. 2015, pp. 6819, 6826) and will likely continue. According to the U.S. Drought Monitor, since 2000 the longest duration of drought in California lasted 376 weeks, beginning on December 27, 2011, and ending on March 5, 2019. The most intense period of drought occurred the week of July 29, 2014, where “exceptional drought” affected over 58 percent of California (NOAA 2022, p. 4). The period from 2000–2021 was the driest period in southwestern North America since at least the year 1800 (Williams et al. 2022, p. 232). Over the past 20 years (2000–2020), the incidence, extent, and severity of drought in the Northwest has increased and it is predicted to continue (Dalton and Fleishman 2021, pp. 37–42). In Oregon, for example, climate models project factors that will

increase the likelihood of drought, including warmer, drier summers and decreases in mountain snowpack (Dalton and Fleishman 2021, entire). Several severe droughts occurred in Oregon from 2010–2020 with different causes, from low winter precipitation to snowpack. The most severe of these droughts occurred from 2013–2015. The latest extreme drought was in 2020, when most of the state was in a historically significant drought (Dalton and Fleishman 2021, pp. 38–39). Prolonged (multi-year) droughts and/or drought periods that occur in close sequence may be especially problematic for western pond turtles, since patterns may fall outside of the range of selection pressures the species experienced in the past.

Drought is clearly implicated in population declines and extirpation at sites in the southern portion of the northwestern pond turtle's range (i.e., California) (Holland 1991, pp. 65–71; Germano and Bury 2001, p. 31; Purcell et al. 2017, entire). During the 1987–1992 drought, populations in the Sierra Nevada and San Joaquin Valley experienced severe declines estimated to range from 75–90 percent (Holland 1991, pp. 69–70; Germano and Bury 2001, p. 21). A telemetry study also in the San Joaquin Valley, spanning from 2009–2015 (thus including the severe drought of 2012–2015), clearly documented drought-related mortality (Purcell et al. 2017, entire). During the first few years of the study, no mortality was observed. In the fourth year of the study, effects of drought resulted in mortality of two adult western pond turtles followed by additional mortalities documented through the remainder of the study. Mortality was attributed to drought-induced predation, although the authors could not rule out scavenging. However, some individuals were healthy only a week beforehand, therefore, in those cases, starvation could be ruled out as the cause of death. Potential predators included coyotes and raccoons, which were documented at the site. The ability to track individuals with telemetry provided additional insight regarding the relationship between drought and predation by documenting where predation occurred relative to the pond itself (Purcell et al. 2017, p. 20). All but one of the observed mortalities of radio-tagged turtles occurred during transit between terrestrial sites. Presumably, western pond turtles were attempting to disperse over terrestrial habitat in search of aquatic habitat, increasing vulnerability to predation while in the uplands. Once the water was dry, some turtles remained in upland habitat for long periods, including one surviving individual that remained out of water for 617 consecutive days, which is the longest documented continuous time out of water for this species (Purcell et al. 2017, pp. 21, 24). Only two individuals successfully found nearby aquatic resources (a trough and a pond), but otherwise mortality was high, indicating that populations with limited access to alternative bodies of water are at risk of extirpation from drought.

Ongoing surveys in central California showed that the current drought conditions in 2020–2021 are having negative impacts on western pond turtles (EBRPD 2022a, b, in litteris). A pond in eastern Contra Costa County that was constructed over 70 years ago went dry for the first time during this period. At least 90 turtles were negatively impacted by the drought: 79 died and 15 were collected and transferred to a zoo. When the pond was low in April, two dead turtles were collected. By October 2021, this pond and other ponds nearby were all dry. At this time, they

found over 40 carcasses that were clearly emaciated, with the cause of death determined to be mostly due to extreme emaciation. Just a month prior, over 151 turtles were observed in the pond, an example of extremely high density resulting from drought. Water quality tests found microcystins of greater than 50 ppb, which may have contributed to the poor health. Many turtles responded to drying conditions by moving into the nearby upland habitat but were preyed upon or scavenged. Although this population had sustained several droughts, western pond turtles were substantially reduced after many successive drought events (EBRPD 2022b, in litteris). A study by Leidy et al. (2016, p. 72) that occurred during the 2012–2015 drought along Coyote Creek in Santa Clara County, California documented 39 shells of western pond turtles that were found scattered along a 3.7 km stretch of the creek. It remains unclear whether the die-off resulted from drought-induced desiccation or drought-induced predation.

Few studies have examined drought effects to western pond turtles in Oregon and Washington, and information is limited regarding their sensitivity to increasing drought under a changing climate. While the species has evolved with and can tolerate periodic drought conditions, alterations to hydrology could limit movements of northwestern pond turtle between habitats, further isolating local populations (Holland 1994, p. 2-14; Leidy et al. 2016, pp. 73–74). Additionally, increased duration of summer droughts could result in ponds drying for prolonged periods of time, compounding declines in local populations (Hallock et al. 2017, pp. 10–11).

In populations of southwestern pond turtles, evidence of declines and extirpation due to drought are common. Here, the negative impacts that have been documented most commonly include mortality due to drought-induced starvation and/or drought-induced predation, which are sometimes difficult to tease apart. Extended drought occurring during 1986–1987 through at least 1991 caused major population declines and extirpations in many areas, but most significantly in southern and central California (Holland 1991, p. 65). During this time, turtles in small to moderate sized watercourses were fairly abundant until 1988–1989, but as water continued to dry, resulting in major increases in distance to the next water source, turtles concentrated in the few remaining pools exhausted available prey, and were exposed to increased predation. The number of carcasses recovered during the middle of the drought in 1989–1990 was approximately 400 percent greater than prior to the drought during 1987–1988. High mortality was attributed to drought-induced starvation because the majority of carcasses did not have visible body fat reserves and showed obvious signs of decreased muscle mass, while signs of predation were lacking (Holland 1991, p. 65). In a study during extreme drought years (2012–2015), a western pond turtle population in Topanga, California (Santa Monica Mountains) experienced high mortality likely due to drought-induced predation. Prior to the drought, from 2002–2009 biologists documented only 12 adult and 4 hatchling mortalities, whereas during the 2012–2015 drought, when the only water available occurred in small refugial pools, biologists documented mortality of over 50 individuals (Resource Conservation District of Santa Monica Mountains 2022, in litteris). Although the exact cause of mortality could not be determined in all

cases, over that same time period, they rescued 15 individuals that were missing limbs or otherwise injured, suggesting that drought-induced predation was likely responsible.

During normal drought conditions, when water levels are low, western pond turtles can aestivate in upland habitat or move to another water body if one is within migration and/or dispersal distance. Aestivating southwestern pond turtles remained in upland habitat for approximately 7 months (mean 201 days, range 154 to 231 days) during the 2011–2012 drought (Belli 2016, p. 57), suggesting that even in a severe drought, individuals could remain alive to repopulate the water body once conditions become suitable again (see Purcell et al. 2017). However, extended drought conditions and/or increased frequency of droughts, could have substantive effects on populations, and other synergistic effects could also make repopulation by aestivating individuals unlikely. In addition, because females often forego nesting when conditions are unfavorable, extended drought can result in reduced reproduction and recruitment opportunities.

A study during the 2012–2015 drought in an area also impacted by a wildfire in 2013 reveals potential synergistic effects between drought and fire (Lovich et al. 2017, entire). Across two sampling points in 2014 and a follow-up visit in 2015, the authors documented extirpation of a population of approximately 170 turtles at Elizabeth Lake in northern Los Angeles County. The carcasses exhibited signs of emaciation, suggesting mortality from starvation (Lovich et al. 2017, p. 7). Starvation was attributed to not only the depletion of prey but also high levels of salinity, which appears to curtail feeding in western pond turtles (Agha et al. 2019, p. 6). Fire likely reduced water quality and negatively affected the food web through inputs of ash, sediment, and nutrients (Burton et al. 2016, pp. 12–22). Lovich et al. (2017a, p. 9) suggested that the effects of drought or fire alone would have been less devastating, but the combination likely led to the extirpation of this population.

In addition to wildfires, a number of other threats may interact with the effects of drought on western pond turtle populations. Although western pond turtles, especially in the more southern portions of the range, have evolved in areas with regular periods of drought historically, manmade stressors (e.g., habitat loss, altered hydrology, increased predation, pollution) may compound the effects of drought on populations. Negative impacts associated with drought are numerous and can also work synergistically to cause mortality, population declines, and/or extirpation. Other threats that can have synergistic effects with drought include: predation (turtles are vulnerable at high density when ponds are drying and or when moving to upland habitat); increased fragmentation/decreased connectivity (increased pond drying means fewer neighboring locations to disperse); increased competition (fewer resources not only for western pond turtle but for potential competitors, such as red-eared sliders); and potentially dams (during drought years, less water is released) (Madden-Smith et al. 2005, p. 5). Within these synergistic relationships, drought may be the catalyst for a cascade of negative impacts.

Extreme Flood Events

Flooding is a natural event that occurs throughout the range of the western pond turtle. Effects of flooding to western pond turtles include flushing of individuals from aquatic and terrestrial habitat and inundation of nesting sites (Rathbun et al. 1992, p. 323; Nerhus 2016, p. 45). Strong winter flows from heavy precipitation are typical in western pond turtle habitats and floods can maintain and improve nesting habitat quality (Risley et al. 2010, p. 64). However, extreme flood events have the potential to cause severe habitat destruction and can act in concert with other stressors leading to potential extirpation, as may have occurred at two sites in the Mojave Desert, San Bernardino County, California (Lovich pers. comm. in Nerhus 2016, p. 44; Puffer et al. 2020, p. 5). Western pond turtles are known to leave the water during times of highwater events and mostly aestivate or overwinter in the uplands above the highwater marks (Reese and Welsh Jr 1997, p. 356). In Oregon, most hatchlings overwinter in the nest; however, fall emergence was observed in response to a heavy precipitation event (Rosenberg and Swift 2013, p. 117) exposing the hatchlings to both environmental and predation risk that may have resulted in reduced survival of those individuals because protection usually provided by the nest was no longer available. Extreme flood events can also cause nest failure, as a result of prolonged inundation or too much moisture during the incubation period, and/or cause drowning of hatchlings (Bury et al. 2012, p. 17).

A potential benefit of flood events may be aided dispersal. Hatchlings that overwinter in nests along the Mojave River may be dispersed by floods (Lovich and Meyer 2002, p. 542). Rosenberg et al. (2009, pp. 20–21) report anecdotal accounts of young and adult turtles being flushed to the mouth of rivers after the floods of 1995 in Ventura County, California. While some pond turtles were most likely injured or killed, long distance dispersal from these infrequent but large flood events likely occurred (Rosenberg et al. 2009, pp. 20–21) and may have provided opportunities for genetic exchange.

High Severity Wildfire

Wildfire occurrence and/or severity are positively correlated with urbanization, roads, recreation, climate change, and drying and drought. Very little is known about the direct effects of wildfire on western pond turtles. Although fires are a natural part of the environment within the range of the western pond turtle, increased fire activity on the landscape is likely exacerbated by years of fire suppression, and by climate change (Abatzoglou and Williams 2016, entire). Fire suppression activities in the south Puget Sound area contributed to an increased distribution of coniferous trees with less than 10 percent of the historical grassland habitat remaining (Lang 1961, pp. 84–86; Crawford and Hall 1998, pp. 13–14). The increased shade cover resulted in the reduction of available nesting habitat and may have contributed to the decline of western pond turtles in Washington (Hays et al. 1999, p. 11). Therefore, natural lower intensity fire most likely benefits western pond turtles by maintaining habitat quality suitable for nesting by decreasing canopy cover and increasing habitat heterogeneity (Hays et al. 1999, p. 11). Severe wildfire on the other hand, has potential to affect western pond turtles through direct mortality, injury, and/or

loss and degradation of aquatic and upland habitat. Western pond turtles that do survive fire may be challenged to find suitable aquatic and/or upland habitat, which could contribute to reduced survival, reproduction/recruitment, and abundance. Observationally, trapping at Malibu Creek State Park within the Santa Monica Mountains in an area that burned in the 2018 Woolsey Fire only resulted in the capture of one western pond turtle in 2022 (Dagit and Morgan 2022, in litteris), although the impacts of the fire itself on pond turtles at this site is unclear.

Other than direct mortality or injury, the loss and/or degradation of upland and aquatic habitat following fires may lead to western pond turtles being more vulnerable to predation and starvation. In the years following the Zaca and Rey fires in the Los Padres National Forest, California, precipitation events caused sedimentation that reduced the depth of streams and a pool resulting in bear and raccoon predation on western pond turtles (U.S. Forest Service, Los Padres National Forest 2022, in litteris).

Activities associated with wildfire suppression (e.g., construction of emergency roads and fire breaks, fire retardant application) are likely to have negative effects on western pond turtles and their habitats because emergency firefighting situations cannot typically accommodate careful avoidance and minimization measures. Rare observations of such an event occurred when western pond turtles were scooped from a pond for fire-fighting purposes in two separate locations, one in California and one in Oregon (L. Wold, USFS, pers. comm. in Holland 1994, p. 2-13). The use of fire retardants and suppressants during wildland fire fighting can affect water quality and can be toxic to western pond turtle food sources (Service 2018, pp. 42–44). Exposure of water bodies to fire retardant chemicals can also disrupt trophic systems by impacting algae and invertebrates (McDonald et al. 1996, pp. 62, 69, 71; Finger et al. 1997, pp. 136–137), potentially resulting in the unavailability of prey and starvation.

Both a warming climate trend and increased woody fuel connectivity are influencing upward trends in fire size and severity and contributing to habitat loss (Moritz et al. 2018, pp. 2, 5). In dry mixed-conifer forests of the Inland Northwest and Pacific Southwest, there has been an increase in high severity fires and an increase in the potential for fires of higher severity (Moritz et al. 2018, p. 3). Observed and projected trends in warmer and drier fire seasons in the western U.S. are likely to continue the trend toward higher-severity wildfires and larger burn areas (Parks and Abatzoglou 2020, pp. 1, 5–6). There is broad agreement among fire scientists that dry forests are becoming less resilient to fire under current and projected climate (Moritz et al. 2018, p. 3).

8.10 Beneficial Influences (Conservation Efforts and Regulatory Mechanisms)

Within this section, we provide examples of projects or efforts that are benefitting western pond turtles, including initiatives that are species-specific and/or more broadly habitat based. We do not attempt to provide an exhaustive list of restoration, monitoring, management, outreach, and research efforts.

State, federal, and international management statuses of western pond turtles are listed in Table 4. These designations can provide additional protection and conservation for the species, and conservation actions associated or stimulated by the designations can act as roadmaps towards recovery. For example, in Washington, recovery actions are guided by the Washington Department of Fish and Wildlife and the state recovery plan (Hays et al. 1999, entire). In Oregon, where the species is designated State Sensitive-Critical and a Species of Greatest Conservation Need, the Oregon Conservation Strategy (ODFW 2016, entire) has identified multi-species Conservation Opportunity Areas to guide agencies and private landowners to areas where long-term success of conservation efforts is most likely. Written guidance on best management practices for western pond turtles is available to facilitate conservation actions in these areas (ODFW 2015, entire). Similarly, the Department of Defense, through its Partners in Amphibian and Reptile Conservation (PARC) network, and the Service developed recommended conservation implementation strategies and best management practices for military base installations, which may also be suitable for other areas (DoD and Service 2020, entire). At a broader scale, partnership between State and Federal agencies and private organizations involves a Memorandum of Understanding (MOU) signed by participating parties, which serves to provide for cooperation and coordination in the conservation and management of the western pond turtle throughout the range of both species. Specifically, the MOU points to the implementation of the range-wide management strategy (RCC 2020, entire), State and local conservation strategies, and/or other conservation plans for these species.

Table 4. State, Federal, and International Management Status (updated from Management Status table from Western Pond Turtle Range-wide Conservation Coalition (RCC) 2020, pp. 8–9).

Entity	NatureServe State Rank	Status
Washington	S1: critically imperiled	State Endangered; Species of Greatest Conservation Need, State Wildlife Action Plan
Oregon	S2: imperiled	Protected Wildlife, State Sensitive–Critical, Species of Greatest Conservation Need (Strategy Species), State Wildlife Action Plan
California	S3: rare, uncommon or threatened	Species of Special Concern (SSC), <i>A. marmorata</i> Priority 3, <i>A. pallida</i> Priority 1, Species of Greatest Conservation Need, State Wildlife Action Plan
Nevada	S2: imperiled	Species of Conservation Priority, State Wildlife Action Plan
Mexico	--	None

Entity	NatureServe State Rank	Status
Canada	--	Extirpated; Species at Risk Act (Priority 2); 2015 Recovery Strategy
IUCN Red List	--	VU – vulnerable
U.S. Forest Service	--	Regions 5 (California) and Region 6 (Oregon and Washington), Sensitive Species.
U.S. Bureau of Land Management	--	California, Oregon/Washington, and Nevada, Sensitive Species

Several regulatory mechanisms provide some protection for the species or reduce or eliminate impacts to habitat from threats, and we provide some examples here. The California Environmental Quality Act requires minimizing significant effects to special-status species such as the western pond turtle, and CDFW often requires mitigation measures for western pond turtles in Lake and Streambed Alteration Agreements (Fish and Game Code section 1602). The Sikes Act requires Integrated Natural Resources Management Plans (INRMPs) for most military installations, leading to Service- and state-approved plans that include protections for western pond turtles on a handful of military installations throughout the range. Twenty habitat conservation plans (HCPs), which describe how the anticipated effects of actions (permitted under section 10(a)(1)(B) of the Endangered Species Act) will be minimized or mitigated, are currently being implemented that include western pond turtles as a covered species: 10 each for northwestern and southwestern pond turtles, respectively. Several of these are also joint Natural Community Conservation Plans (NCCPs), which is the state counterpart to the HCP. Implications of being included as a covered species vary depending on the plan, but in general assures that habitat will be set aside and managed for the species as compensation for covered activities, such as planned urban development, within the area the HCP covers, and that measures will be implemented to avoid or minimize take of the covered species. For example, the East Contra Costa County HCP/NCCP has preserved as mitigation or under existing easements approximately 13,590 acres of suitable habitat for the western pond turtle (Table 8a in East Contra Costa County Habitat Conservancy 2022, pp. 65–66). Specifics for each HCP are included within the agreements. More information about HCPs that include the western pond turtle as a covered species can be found at: <https://ecos.fws.gov/ecp/species/1833> or on websites for the various plans.

Headstarting, Captive Breeding, and Reintroductions

Headstarting is the process of collecting eggs or young from the wild and rearing them in captivity through the most vulnerable stages of their life cycle, and then releasing those

individuals back into wild populations. The goal of headstarting is to increase recruitment, particularly by rearing individuals in a protected environment until they are large enough to avoid predation. Reintroductions can involve release of headstarted individuals, or translocation between sites. Captive breeding is another tool that has been implemented for western pond turtles, with release of juveniles produced through the programs. While these tools are useful for augmenting populations in the wild and preserving genetic diversity, they do not address the causes of decline themselves.

Headstarting was initiated in Washington in 1990, through a joint effort involving the Woodland Park Zoo, Center for Wildlife Conservation, and Washington Department of Wildlife (Hays et al. 1999, pp. 25–26). At this time, only 2 populations remained in the Columbia Gorge, with only approximately 150 western pond turtles left in the wild, and they were effectively extirpated from the Puget Sound (Hays et al. 1999, entire; Pramuk et al. 2013, p. 3; Hallock et al. 2017, p. iv). The Washington program also involved a captive breeding component, with breeding stock (9 adult turtles from Washington and 3 adult turtles from Oregon) forming three groups including one of Columbia Gorge origin, one of Puget Sound origin, and one from Oregon (Hays et al. 1999, p. 25). Rearing under protected conditions allows the hatchlings to grow quickly, and after 6–12 months, they are released back into the wild at the size of 3 year-old wild turtles (Hallock et al. 2017, p. 11). From 1991–2015, 2,200 captive-bred and wild-bred western pond turtles raised at the Woodland Park Zoo and Oregon Zoo have been released, increasing the number of sites in Washington from two at state listing in 1993 to six sites today (Hallock et al. 2017, p. iv): two remnant populations in the Columbia River Gorge and four reintroduced populations (two in Puget Sound and two in the Columbia River Gorge). Annual survival estimates of western pond turtles released at Pierce National Wildlife Refuge were high based on radio-telemetry of a subset of released individuals, ranging from 86 to 97 percent (Haegen et al. 2009, pp. 1403–1404). A population viability assessment for Washington suggested that the sites in that state are reliant on continuation of population augmentation via the headstarting program until bullfrog predation is reduced and adult mortality remains below 12.5% (Pramuk et al. 2013, entire). The Washington State Recovery Plan indicates that headstarting and captive breeding should continue until populations are sustainable without such intervention (Hays et al. 1999, p. 39).

Headstarting both northwestern and southwestern pond turtles has been implemented by additional zoos and other partners after the program in Washington. For example, at the University of California, Davis, Arboretum, headstarting was successfully used to supplement the western pond turtle population after mark-recapture surveys documented a population decline of 40 percent in less than 10 years (Spinks et al. 2003, pp. 260–261). In another example, releases from a headstarting program at the San Diego Zoo, in combination with nonnative aquatic predator removals, effectively doubled the western pond turtle population at the Sycuan Peak Ecological Reserve in San Diego County (Brown et al. 2015, pp. 4–16). Other reintroduction efforts in San Diego County have occurred that involved translocating western

pond turtles from private ponds into restored habitat, often in conjunction with nonnative species removals (Molden et al. 2022, p. 2).

8.11 Summary

In this section, we discussed significant past, current, and future threats, and current conservation efforts, that are affecting western pond turtles. We began by listing the threats that we identified in the 90-day finding, as well of those included by Washington, Oregon, Nevada, and California in state-specific documents. We also presented summary tables from a recent review assessing threats to both northwestern and southwestern pond turtles. After reviewing these lists, we included sections within this chapter of the SSA report discussing the following threats: habitat loss and fragmentation; altered hydrology; predation; nonnative species competition; disease; road impacts; collection; contaminants; and climate change. Each section discussed historical, current, and future impacts, highlighting any notable differences between northwestern and southwestern pond turtles.

9.0 CURRENT CONDITION

Current condition may be described in terms of past and ongoing changes in a species' habitat, demographics, and distribution (Smith et al. 2018, p. 306). To assess the current condition of the northwestern pond turtle and southwestern pond turtle, we used the best scientific and commercial data available to describe past and ongoing changes in the species' occupancy and impacts from primary threats, and assess population trends within analysis units. Specifically, we present information about western pond turtle survey results (where available), impacts of primary threats to the species, and results from a model that incorporates these same primary threats and projects population growth and probability of extinction through 2050, which is 27 years from now, approximately one western pond turtle generation. Although this chapter includes information related to each of these themes, our analysis of resiliency focuses on probability of extinction at 2050 because this result can be consistently applied across all analysis units for both species. Analysis unit level resiliency is then used to discuss current representation and redundancy at the species level.

9.1 Presence

Although there have been numerous surveys throughout the range of both species, and an increase in survey effort in recent years, in the majority of cases survey information provides presence information only. The long-lived nature of western pond turtles means that repeated presence of adults across years could represent detections of the same individual(s) over time and does not indicate if there is recruitment. In most cases, western pond turtle presence data yields little information that can inform population status other than observations of various life stages or a high count for any given time period. Repeated surveys at the same sites are needed to

predict occupancy across the range, and these data are not available for most western pond turtle occurrence locations.

The information we have related to survey data varies by species. For both species, we mapped presence using a dataset including western pond turtle observations across the species' ranges, including observations from peer-reviewed literature, reports, and unpublished data from State, Federal, and regional/local partners, academic institutions, and independent researchers (Supplemental Table 1 in Appendix A). Observations are denoted at the HUC12 level, such that any presence observation in a HUC12 would denote that entire HUC12 as occupied. We also present results from Manzo et al. (2021, p. 493) on population size estimates throughout the species' ranges. In this study, the definition of a population was determined by field researchers who collected the data and was therefore not standardized across sites. We stress that this dataset is not inclusive of all western pond turtle populations, but instead reflects those data gathered that met the criteria for inclusion in the Manzo et al. (2021, entire) study. For example, the study excluded data from visual surveys of basking turtles because it could include recounts of the same individuals and assumed that field efforts maximized capture effort of as many western pond turtles as possible at each site. Manzo et al. (2021, p. 488) also excluded estimates of population size from mark-recapture efforts because these data were not available for most sites. To compile population estimates, Manzo et al. (2021, p. 488) calculated the average number of unique individuals captured annually, which they refer to as the census population size.

For the northwestern pond turtle, we also include population data from all sites in Washington and results from mark-recapture and occupancy surveys in Oregon.

For the southwestern pond turtle, we also show results from surveys conducted by USGS, including information on adult and juvenile detections.

Northwestern Pond Turtle Observations

Northwestern pond turtle presence data in Oregon, California, and Nevada are shown in Figure 13, including observations from 1879 through 2022. This dataset also includes historical observations from Washington; more recent data since 1991 initiated reintroductions from headstarting are not included. The figure shades in HUC12 subwatersheds that have had positive detections. Detections do not denote the number of observations at the point, or the number of western pond turtles observed at any given time; they only convey presence. Similarly, although we denote recent observations in the figure, the absence of a recent observation does not necessarily mean that a HUC12 is not currently occupied. The number and proportion of subwatersheds within each analysis unit with observations is shown in Table 5 (HUC12s that straddle multiple analysis units are considered within the analysis unit that has the larger portion). Positive detections in a HUC12 subwatershed do not necessarily mean that

subwatershed is currently occupied, nor does the absence of observations mean that the species is not present.

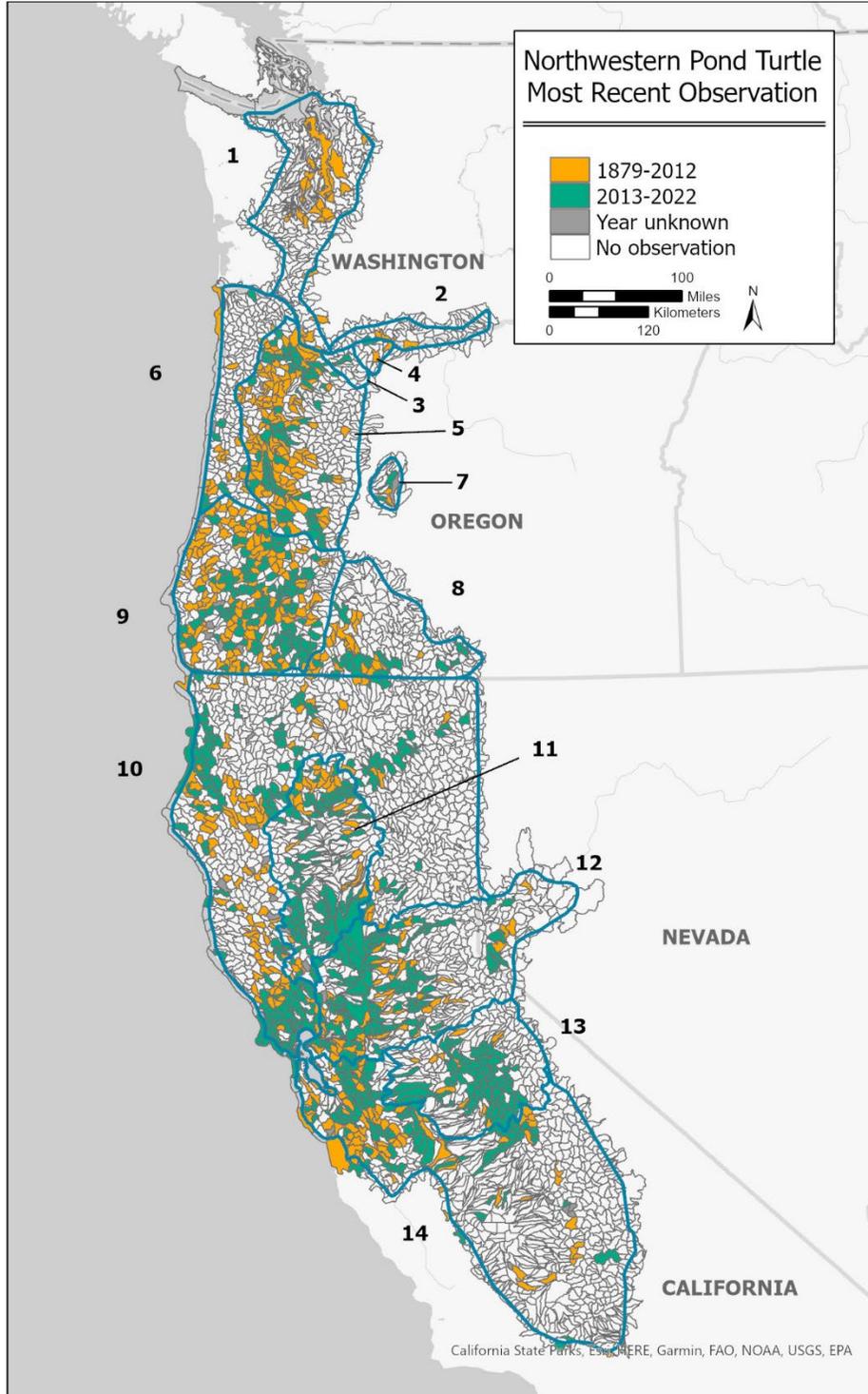


Figure 13. Northwestern pond turtle occurrences from 1879 through 2022. USGS HUC12 subwatersheds are drawn on the map and shaded in orange in areas that have had positive surveys from 1879 through 2012, and green in areas with recent positive surveys in the last 10 years (2013 through 2022).

Table 5. Proportion of subwatersheds with occurrence observations between 1979 through 2022 for each northwestern pond turtle analysis unit.

Analysis Unit	State/Country	Number	Total HUC12s	Occupied HUC12s	Proportion HUC12s Occupied
Puget Sound	WA	AU-1	275	22	8%
Columbia River Gorge	WA	AU-2	85	3	4%
Bull Run - Sandy	OR	AU-3	21	2	10%
Mosier - Hood	WA	AU-4	19	3	16%
Willamette	OR	AU-5	403	180	45%
North Coast	OR	AU-6	198	9	5%
Bend	OR	AU-7	28	4	14%
Klamath - Lakeview	OR	AU-8	197	34	17%
Southwest Oregon	OR	AU-9	433	250	58%
Northern California	CA	AU-10	1065	229	22%
North Central Valley	CA	AU-11	276	113	41%
Bay Delta Nevada	CA/NV	AU-12	430	132	31%
Yosemite	CA	AU-13	241	78	32%
San Joaquin Valley	CA	AU-14	727	156	21%

Manzo et al. (2021, pp. 488, 493, 495) collated rough estimates of northwestern pond turtle population sizes from available peer-reviewed literature, reports, and unpublished data sets (Figure 14). In the population estimates gathered by Manzo et al. (2021, p. 495) most populations contained between 1 and 50 individuals; the mean number of individuals within this category was 20.7. Sites with the highest population estimates occurred along the Trinity River in Trinity County, California, and in parts of California’s Central Valley (Fresno and Kern counties). While there were several populations estimated over 100 individuals in California and one site with over 100 individuals in Nevada, there was only one population estimated to be over 50 individuals in Oregon. Two sites with a mean annual capture of less than 1 individual per year were located in Kern County (Manzo et al. 2021, p. 493). Note that this figure and information relates only to the population estimates that fit the criteria for inclusion as detailed by Manzo et al. (2021, p. 493), and we present additional information below related to populations in Washington and Oregon below.

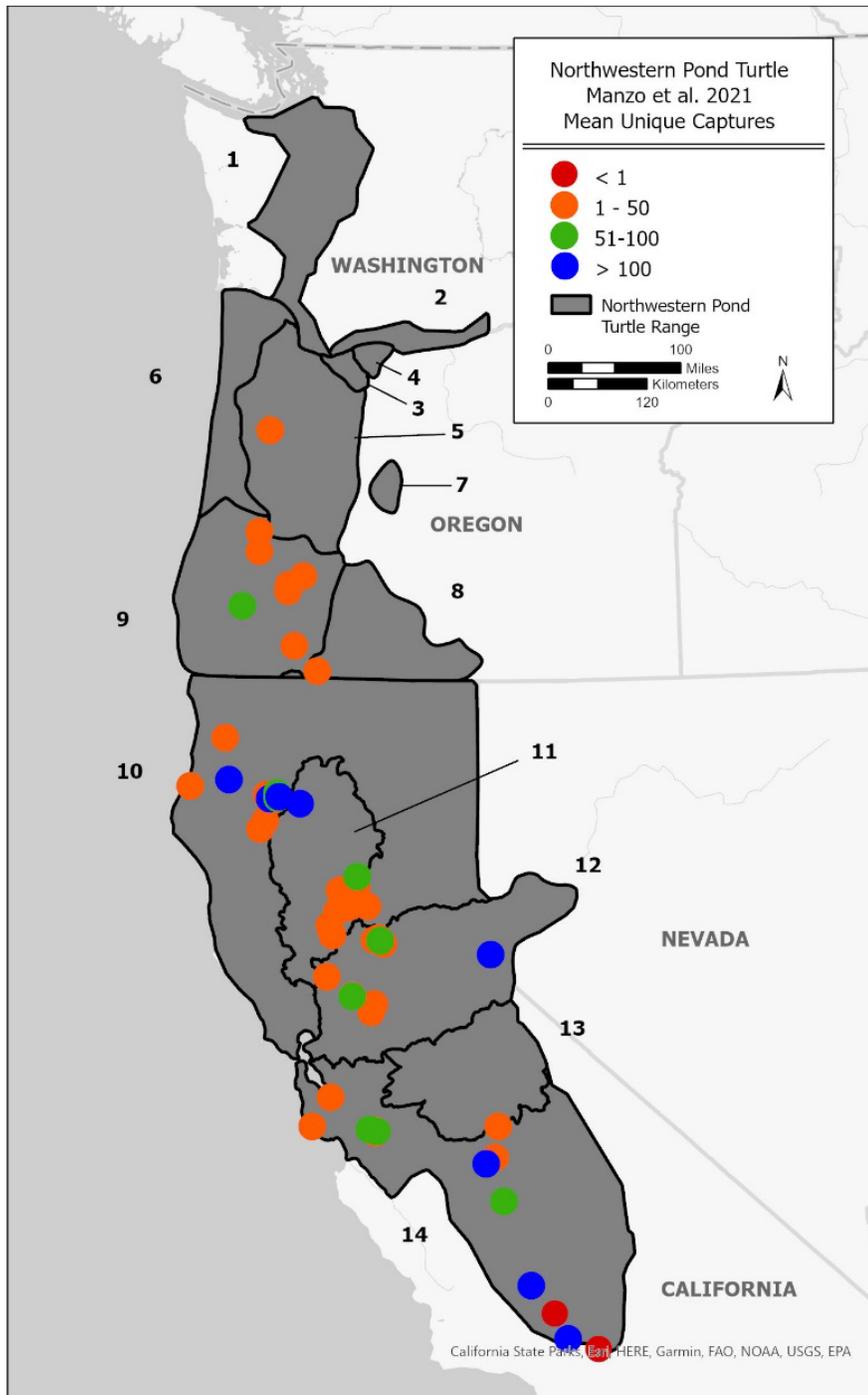


Figure 14. Population size estimates for northwestern pond turtles from Manzo et al. (2021, p. 495) based on mean annual captures per site. Data is based on surveys across 50 sites from 1993 to 2019. Populations are defined as a group of individuals sampled at a site, as reported by the researchers who collected those data. Population sizes are binned into categories: <1, 1–50, 51–100, and >100 individuals.

In Washington, current population estimates are derived from mark/recapture efforts, population models, and the minimum numbers of northwestern pond turtles observed during surveys at all six northwestern pond turtle sites (Hallock et al. 2017, p. 6; Bergh and Wickhem 2022, pp. 4–6, WDFW 2022, in litteris) (Table 6). The total minimum estimated population of northwestern pond turtles in Puget Sound (AU-1) and Columbia Gorge (AU-2) is approximately 481 and 281, respectively (although this total involves summing population sizes across years).

Table 6. Northwestern pond turtle estimated population sizes from most recent surveys at 6 sites in Washington.

Analysis Unit	Site	Year	Estimate population size	Lower confidence interval	Upper confidence interval	Method
1	Pierce County Site	2018	183	na	na	Basking and hoop-trap captures
1	Mason County Site	2013	98	na	na	Snorkel captures; census
2	Pierce NWR	2017	60	56.0	71.0	Mark/recapture models
2	Sondino	2018	219	212.2	233.7	Mark/recapture models
2	Collins-Bergen	2022	162	159.8	169.6	Mark/recapture models
2	Beacon Rock (Homestead Lake)	2022	40	37.9	47.2	Mark/recapture models

There are numerous sites in Oregon that are included in our presence map (Figure 13) but were not represented in the Manzo et al. (2021, Supplemental Table 2) dataset. Of these, several are sites with between 50 and 100 or over 100 marked northwestern pond turtles, including sites in Mosier-Hood (AU-4), Willamette (AU-5), Klamath-Lakeview (AU-8), and Southwest Oregon (AU-9) analysis units (Barnes 2023, in litteris). Additionally, standardized occupancy surveys were conducted in 2018, 2019, and 2020 at 138 historical sites and 176 new sites in Oregon (Samara Group, LLC 2021, entire). These data were then used in models to predict the proportion of area occupied in both lotic and lentic habitat in a historical site analysis (defined as

within 1000 m of a historical survey point) and a relaxed site analysis (including points within and outside the historical survey radius, and controlling for distance to a historical point). Probability of presence was 100 percent at locations where western pond turtles were observed in a survey, and presence in other locations was predicted based on a number of covariates including, but not limited to, presence of habitat features (e.g., basking sites, nesting habitat), quality and variation of habitat, and land cover classes (Samara Group, LLC 2021, pp. 24–26). Estimated percentage of occupied locations in lentic habitat sites (e.g., ponds) was 61.6 percent [95 percent CI [55.3, 73.7]] at historical sites and 48.5 percent (95 percent CI [42.3, 66.4]) across all sites. Similarly for lotic habitat sites (e.g., streams), the estimated percentage of occupied locations was found to be 59.8 percent (95 percent CI [59.7, 61.3]) at historical sites and 44.0 percent (95 percent CI [43.0, 46.1]) across all sites. The researchers point out that the estimates are positively biased for two reasons: (1) the occupancy models only use locations that did not have extreme habitat loss (2) the models cannot distinguish sustainable populations from populations where adults are present but there is no active recruitment. Therefore, occupancy estimates should only be applied to locations without habitat loss, and some of the occupying populations may be on a path to local extinction.

Southwestern Pond Turtle Observations

Southwestern pond turtle presence data from California and Baja California, Mexico are shown in Figure 14. This figure shades in HUC12 subwatersheds that have had positive detections. These detections do not denote the number of observations at the point, or the number of western pond turtles observed at any given time; they only convey presence. Similarly, although we denote recent observations in the figure, the absence of a recent observation does not necessarily mean that a HUC12 is not currently occupied. The number and proportion of subwatersheds within each analysis unit with observations is shown in Table 7. Positive detections in a HUC12 subwatershed do not necessarily mean that subwatershed is currently occupied, nor does the absence of observations mean that the species is not present.

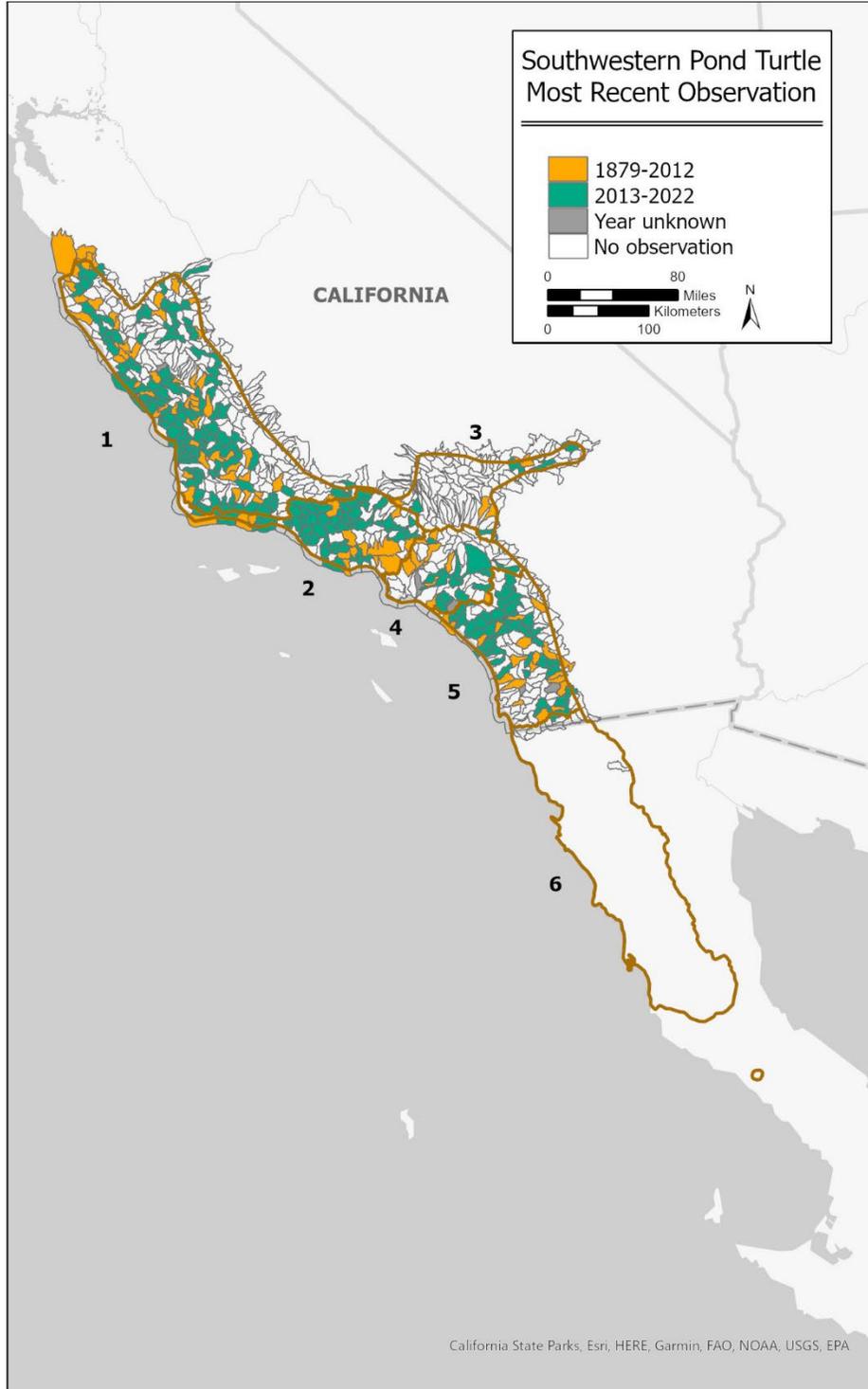


Figure 15. Southwestern pond turtle occurrences from 1879 through 2022. USGS HUC12 subwatersheds are drawn on the map and shaded in orange in areas that have had positive surveys from 1879 through 2012, and green in areas with recent positive surveys (2013 through 2022).

Table 7. Proportion of subwatersheds with occurrence observations for each southwestern pond turtle analysis unit.

Analysis Unit	State/Country	Number	Total HUC12s	Occupied HUC12s	Proportion HUC12s Occupied
Coast Range	CA	AU-1	318	151	47%
Ventura/Santa Barbara	CA	AU-2	104	66	63%
Mojave	CA	AU-3	102	9	9%
Los Angeles	CA	AU-4	56	20	36%
Orange County/San Diego	CA	AU-5	138	61	44%
Baja	Mexico	AU-6	NA	NA	NA

Figure 16, which depicts population estimates from Manzo et al. (2021, p. 495), shows sites with southwestern pond turtle survey data. Sites with the highest population estimates occurred sparsely along the central and southern California coast. However, most sites were reported to have 1–50 western pond turtles, and the mean within this category was 10 individuals (Manzo et al. 2021, p. 495). Sites with the lowest mean annual captures (<1 individual per year) occurred in San Diego County and in San Bernardino County along the Mojave River (Manzo et al. 2021, p. 493).

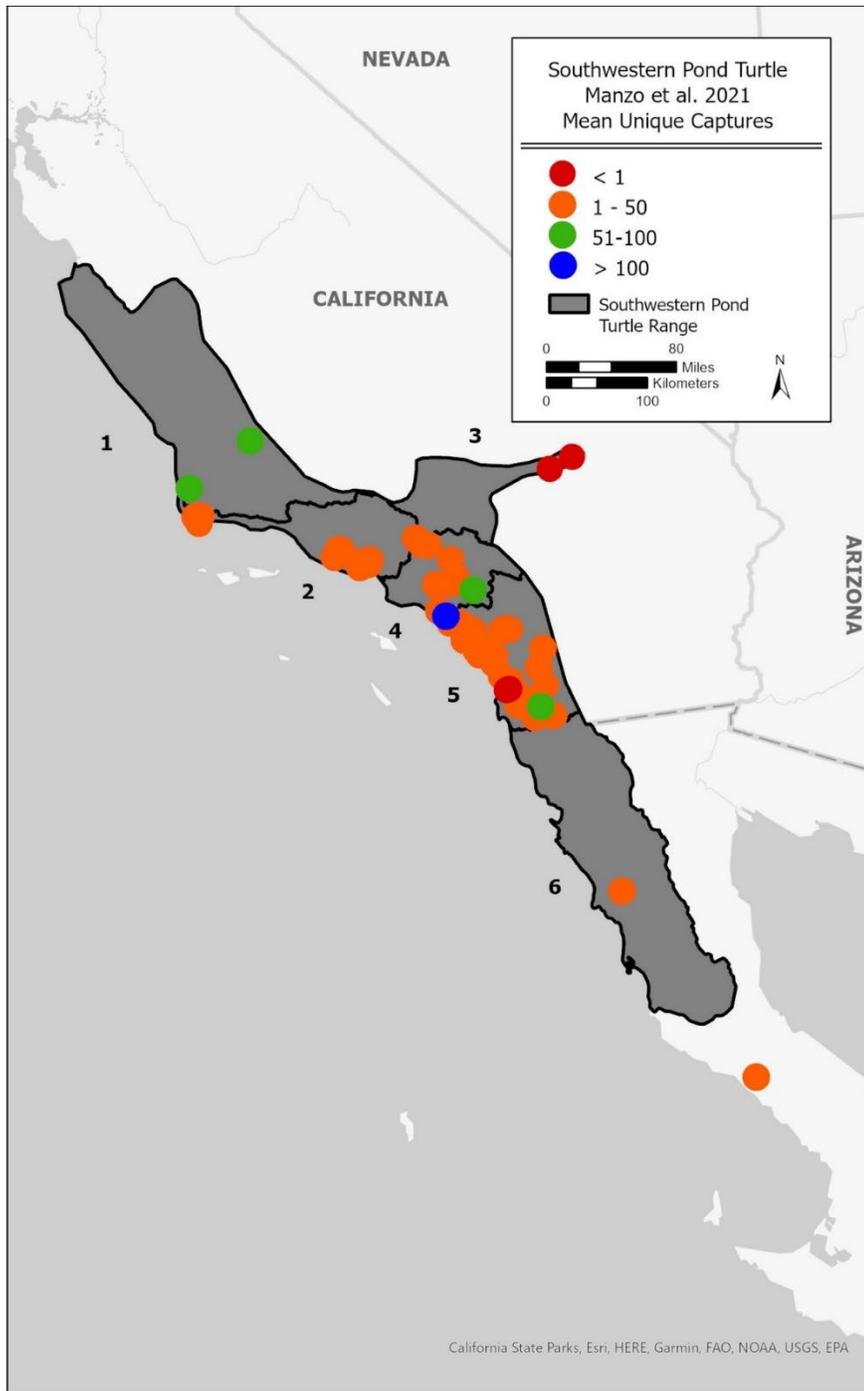


Figure 16. Population size estimates for southwestern pond turtles from Manzo et al. (2021, p. 495) based on mean annual captures per site. Data is based on surveys across 81 sites from 1993 to 2019. Populations are defined as group of individuals sampled at a site, as reported by the researchers who collected those data. Population sizes are binned into categories: <1, 1–50, 51–100, and >100 individuals.

Increased sampling in the past several years has resulted in a more robust dataset across a subset of the southwestern pond turtle range (USGS 2022, in litteris) (Figure 17). Using data from various survey types, including but not limited to visual encounter surveys and trapping surveys, USGS has performed over 3,200 western pond turtle surveys since 2000 across the range of the species. Over 450 of these surveys were in 2021 and 2022. These data provide important insight into where surveys have been conducted, and importantly, where adults and juveniles have been detected. In particular, the data reveal that in many areas, only adults are being detected in surveys, which raises concerns about reproduction and recruitment in those areas. These data also indicate gaps in the range where surveys have occurred without positive detections, although as we note above, that lack of detections does not necessarily mean that western pond turtles are absent from an area. Specifically, Los Angeles (AU-4) and San Diego/Orange County (AU-5) have had many surveys with adult-only detections or no detections. The Mojave (AU-3) only includes surveys with no detections from this dataset (but see Lovich et al. 2021, pp. 326–327).

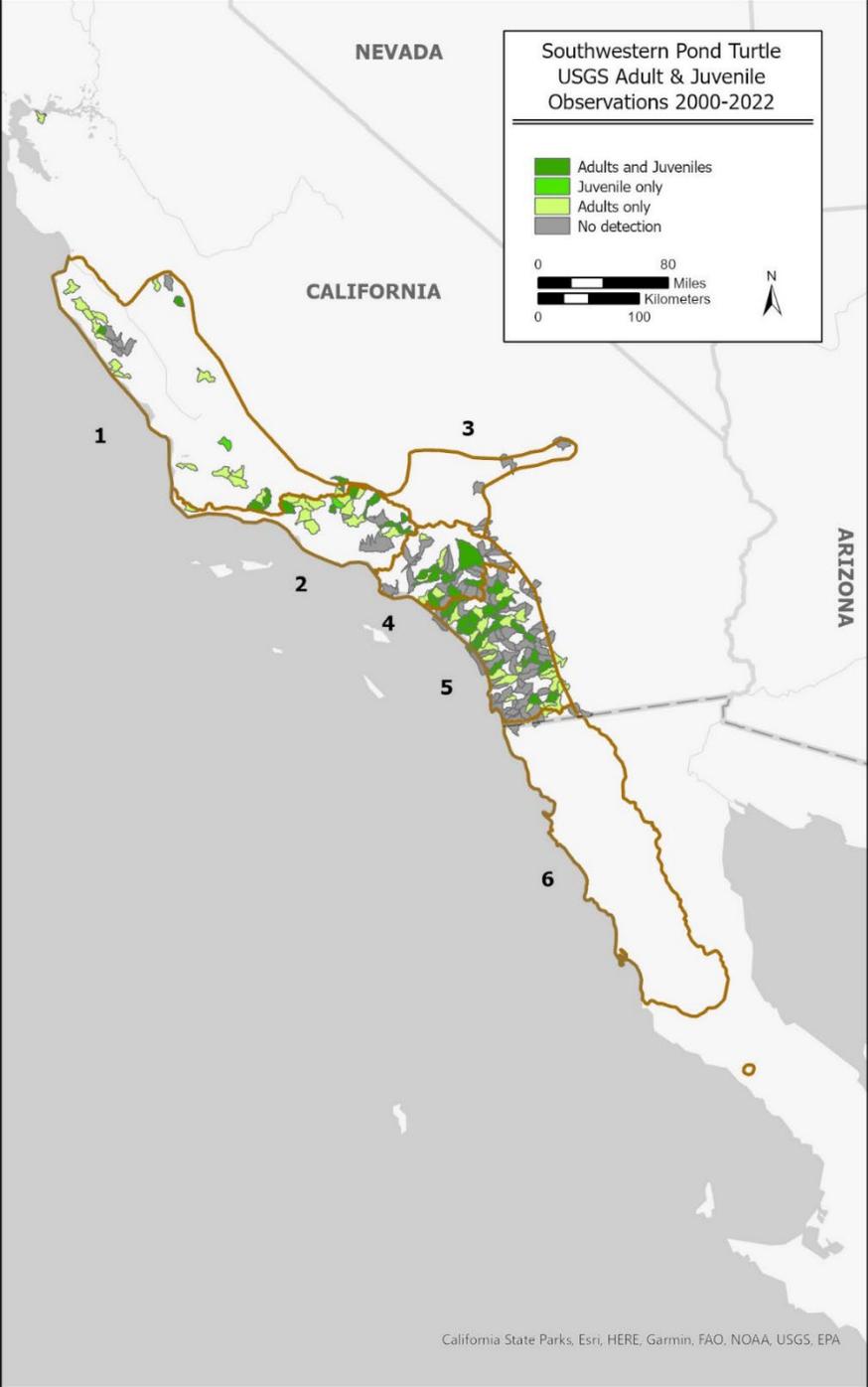


Figure 17. Survey results from USGS conducted across the range of the southwestern pond turtle from 2000 through 2022. Within surveyed HUC12 subwatersheds, results denote whether both adults and juveniles, adults only, or juveniles only have been observed, or if there have been surveys without detections.

9.2 Primary Threats

Based on our assessment of Influences to Viability in Chapter 8, we identified three key factors as most influential in driving the western pond turtle’s current and future condition: anthropogenic impacts, predation by bullfrogs, and drought. These factors have had substantial population-level effects that are anticipated to continue, and to be the primary drivers of species viability. They encompass or relate to the other factors influencing viability discussed in Chapter 8 (Figure 18). Assigning these three threats as primary drivers of condition for the species is supported by the literature. Predation by non-native predators and drought are both top threats in a peer-reviewed analysis of threats acting on the two species, and anthropogenic impacts overlaps with land alteration, another top threat in that paper. Anthropogenic impacts also touches upon many of the other threats within Manzo et al. (2021, pp. 492–493).

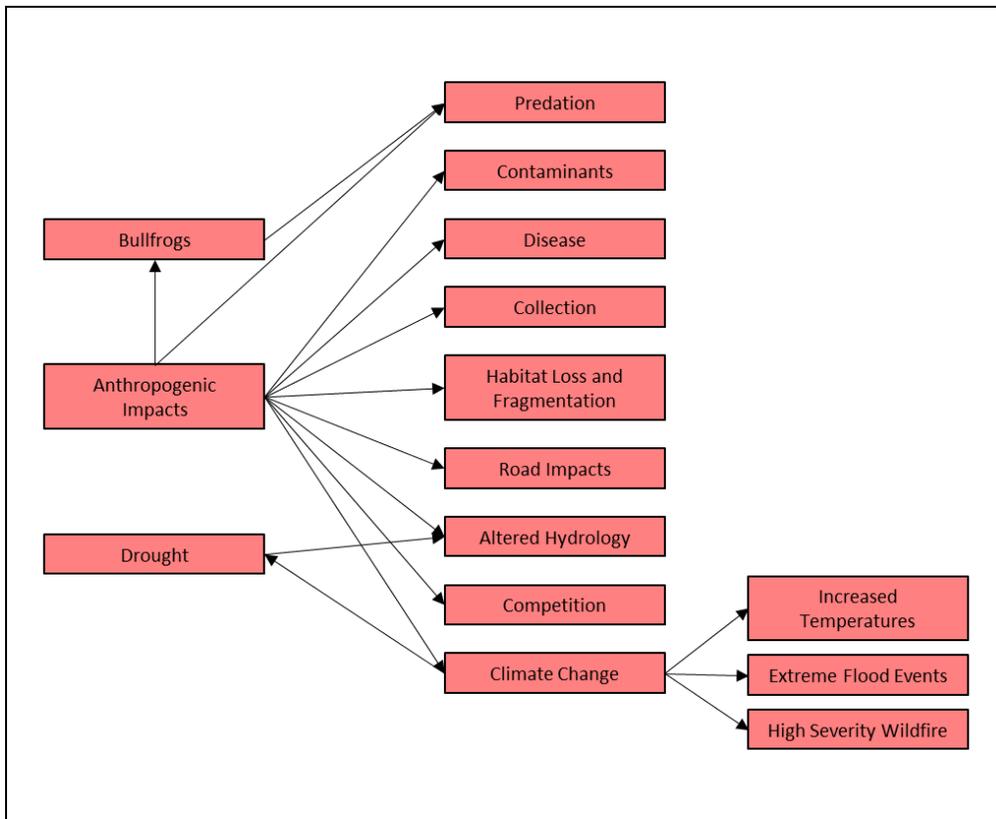


Figure 18. Relationship between primary driving influences included in scenarios and other threats to western pond turtles, as identified in Chapter 8 “Influences to Viability”

Although these primary threats are already discussed in Chapter 8 in relation to their historical, current, and future impacts on western pond turtles, here we provide additional information about these threats relative to the analysis units for each of the species.

Anthropogenic Impacts

We look at past, current, and future impacts of anthropogenic impacts on western pond turtles and their habitat using two spatial datasets: human modification and land conversion. Both are associated with increases in habitat loss and fragmentation, road mortality, altered hydrology, contaminants, disease, collection, competition, and climate change (Figure 18). They impact all western pond turtle life stages.

To summarize impacts from human modification across the range of the western pond turtle, we present data from a human modification index developed by Theobald (2021, entire). Human modification in the index is based on the following stressors and data sources: urban and built-up, crop and pasture lands, livestock grazing, oil and gas production, mining and quarrying, power generation (renewable and nonrenewable), roads, railways, power lines and towers, logging and wood harvesting, human intrusion, reservoirs, and air pollution (Theobald et al. 2020, p. 1954), many of which are known to impact western pond turtles and their habitats. The scale for datasets used to develop the human modification score index in Theobald et al. (2020, p. 1965) varies, for example from 0.0009–0.9 km² for the Global Human Settlement Layer used to assess the urban and built up class, and 10 km² for the Gridded Livestock of the World source used to assess the grazing stressor. The resultant dataset quantifies human modification on a continuous scale from 0 to 1, where higher numbers reflect more extensive modification.

Mean human modification values from Theobald (2020, entire) are shown for the northwestern pond turtle range in Figure 19 and within HUC12 subwatersheds occupied by northwestern pond turtles in Figure 20. For southwestern pond turtles, mean human modification values are shown across the range and within occupied HUC12 subwatersheds in Figure 21 and Figure 22, respectively. Human modification values do not extend fully into Baja California, Mexico, and we cropped the map accordingly to focus on the available data. Increased levels of human modification on the landscape surrounding a water body and riparian habitat can result in: fewer suitable nesting sites (Legler 1954, p. 142), increased meso-predator populations that consume nests and juvenile turtles (Wang et al. 2015, p. 27), increased adult mortality while crossing roads to find nest sites (Gibbs and Shriver 2002, entire), and less available wintering habitat (Davis 1998, pp.16–19).

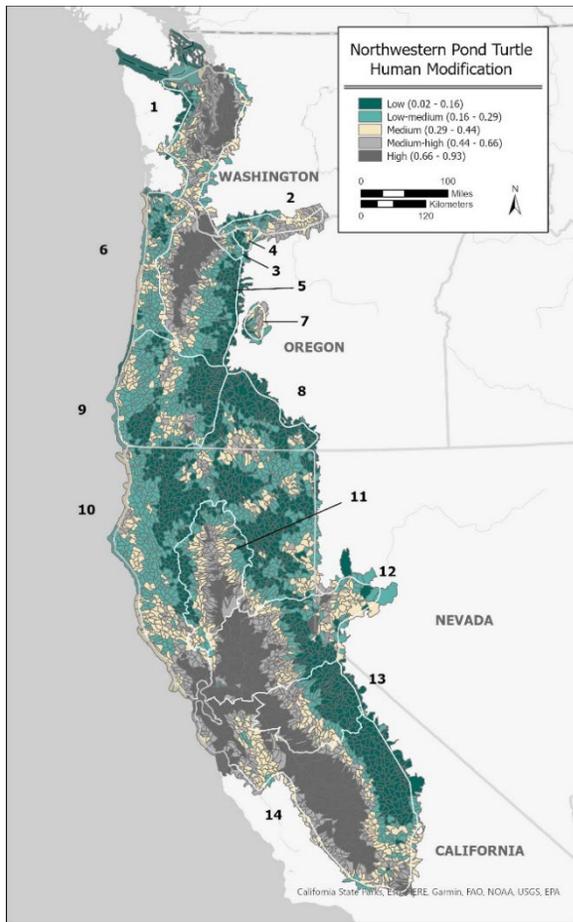


Figure 19. Mean human modification values in HUC12s in the northwestern pond turtle range. Human modification values from Theobald et al. (2020, entire) are averaged across HUC12s.

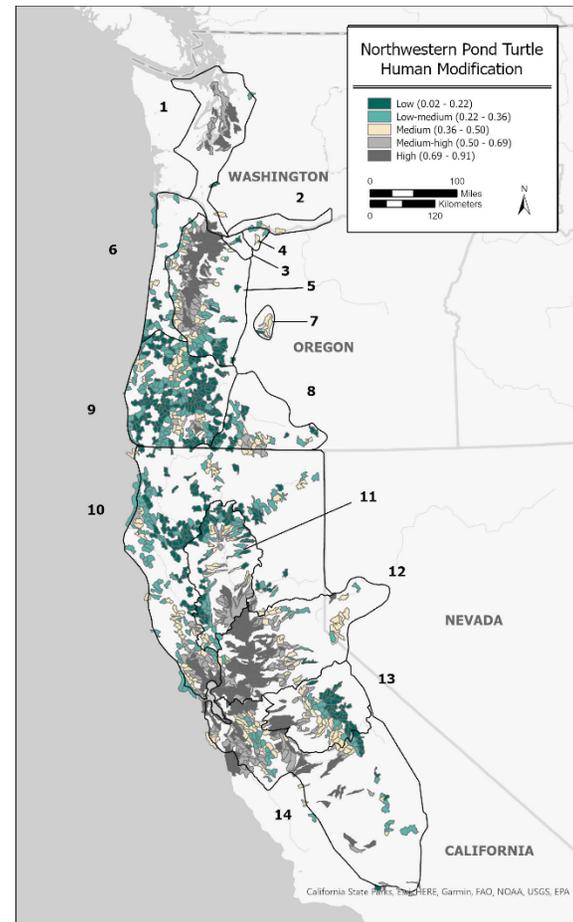


Figure 20. Mean human modification values in HUC12s occupied by northwestern pond turtles.

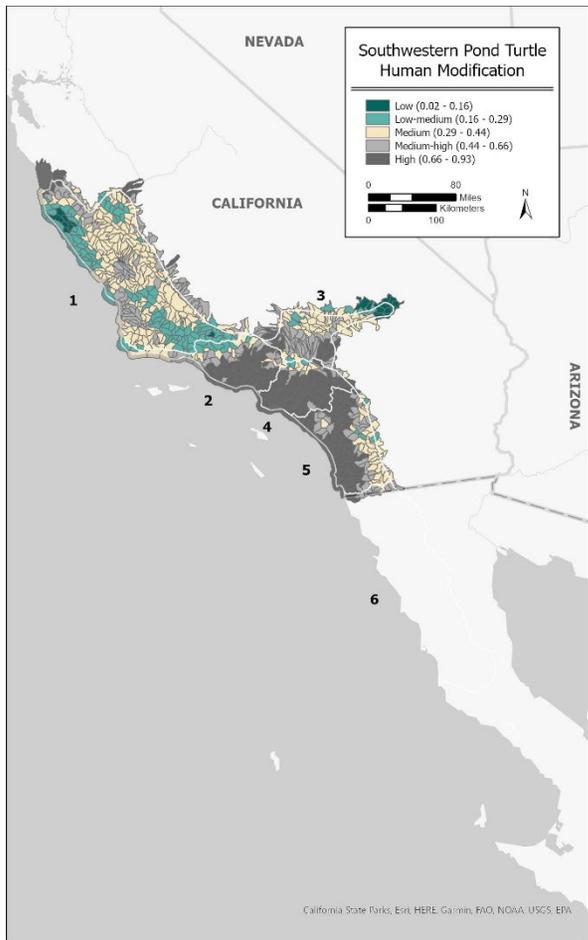


Figure 21. Mean human modification values in HUC12s in the southwestern pond turtle range. Human modification values from Theobald et al. (2020, entire) are averaged across HUC12s.

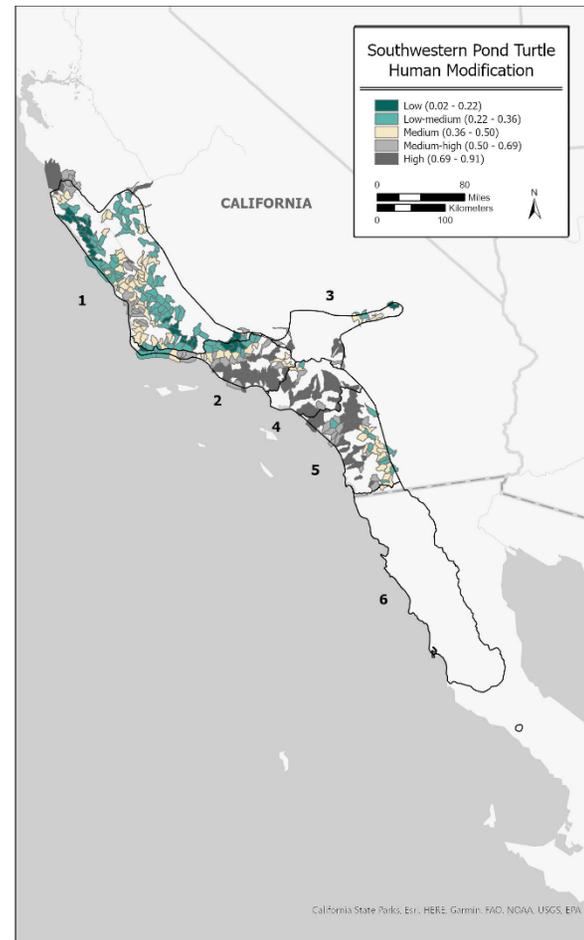


Figure 22. Mean human modification values in HUC12s occupied by southwestern pond turtles.

While the human modification snapshot focuses on historical modification across the range, our focus on land conversion is forward-looking. The Integrated Climate and Land-Use Scenarios (ICLUS) project produced spatially explicit projections of human population and land-use through 2100 (EPA 2016, entire). These products incorporate Shared Socioeconomic Pathways (SSPs), which are scenarios of projected socioeconomic global changes, and RCPs, which are greenhouse gas concentration trajectories adopted by the International Conference on Climate Change. The projected socioeconomic pathways represent a “middle-of-the-road” projection as opposed to a rapidly growing and flourishing economy. For example, the U.S. population under SSP2/RCP 4.5 is predicted to be 455 million by 2100, while under SSP5/RCP8.5 it is predicted to be over 730 million. The ICLUS project includes 19 land use categories, of which wetland is the land use type most closely associated with western pond turtle habitat. These data incorporate both losses and increases in wetlands over time and are incorporated into the modeling described in Chapter 9.3.

Drought

Drought affects all life stages of the western pond turtle, with the largest documented impacts to adult survival. There are additional but lesser reductions in reproduction/recruitment. Recent droughts have led to declines and extirpations of western pond turtle populations by negatively affecting the quality and/or quantity of its aquatic habitat, impacting survival, recruitment, and connectivity, and exacerbating the effects of other threats. There are more documented impacts in southwestern pond turtle than northwestern pond turtle, but drought has led to extirpations within both species.

The Palmer Drought Severity Index (PDSI) from the National Oceanic and Atmospheric Administration (NOAA) Monthly U.S. Climate Divisional Database (Vose et al. 2014, entire) uses temperature and precipitation data to estimate relative dryness. The index spans -10 (dry) to +10 (wet), with most maps showing a range of values from -4 to +4. Index values between -2 and -4 are classified as moderate drought, and <-4 are classified as extreme drought. As an example, PDSI values in the continental US in July 2022 are shown in Figure 23. Within the western pond turtle range, California and Nevada were in drought at this time step, while Oregon and Washington were relatively moist. Note that this is just a snapshot of drought conditions in one month of one year.

Palmer Drought Severity Index
July, 2022

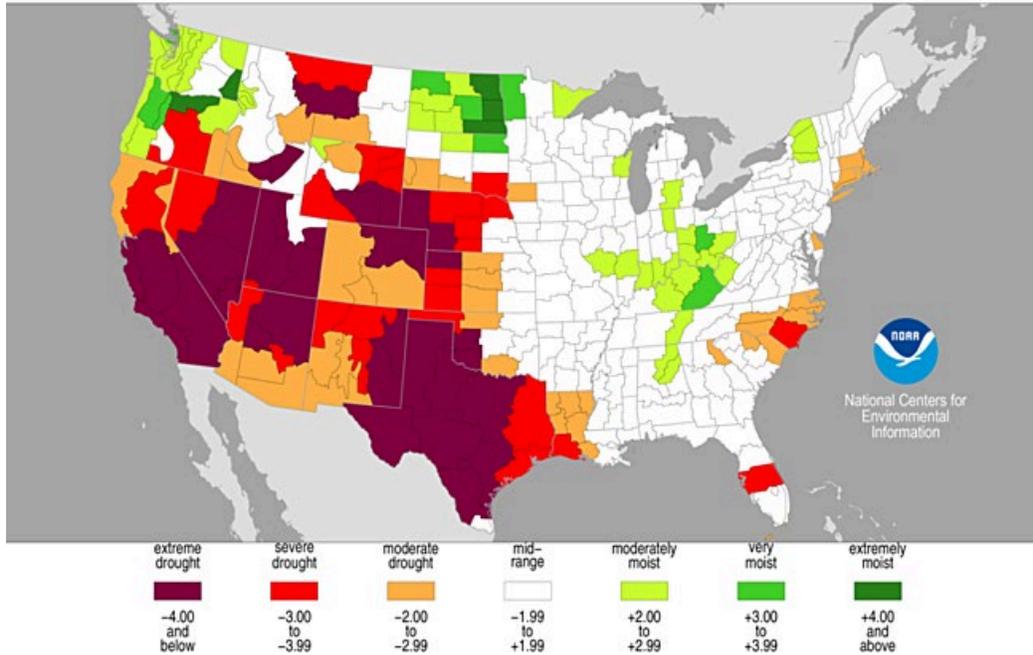


Figure 23. Palmer Drought Severity Index values in the continental US in July 2022. Data from NCEI/NOAA.

Bullfrogs

Bullfrogs have been introduced into western pond turtle habitat and influence viability of the species by increasing predation pressure on hatchlings and small juveniles. Although predation by other nonnative species, and increased predation pressure in urbanized areas by meso-predator release and/or predator subsidies, are also threats to western pond turtles, we consider bullfrog predation to be having the largest impact on western pond turtle demography. In doing so we acknowledge that focusing only on bullfrog predation may underestimate effects of predation.

We mapped bullfrog presence across of the range of western pond turtles using observational data from the Global Biodiversity Information Facility (GBIF.org 2022, entire). The Global Biodiversity Information Facility is an international network and data infrastructure that provides open access to species occurrence records, providing information across the western pond turtle range. Most bullfrog records in GBIF are from iNaturalist and natural history museums, and the dataset includes observations from the early 1900s through May 2022, plus one outlier from 1818. Over 90 percent of the observations were from 2000 through May 2022, with half between 2020 and May 2022. Because this dataset relies on detections reported to the database, it is not a

true measure of presence, and is probably more likely to underestimate bullfrog presence than overestimate.

Bullfrog observations within the range of the species are shown in Figure 24 and Figure 25. The proportion of subwatersheds occupied by bullfrogs and northwestern pond turtles varies from 0 to 0.6 (Table 8). The proportion of subwatersheds occupied by bullfrogs and southwestern pond turtles varies from 0.3 to 0.7 (Table 9).

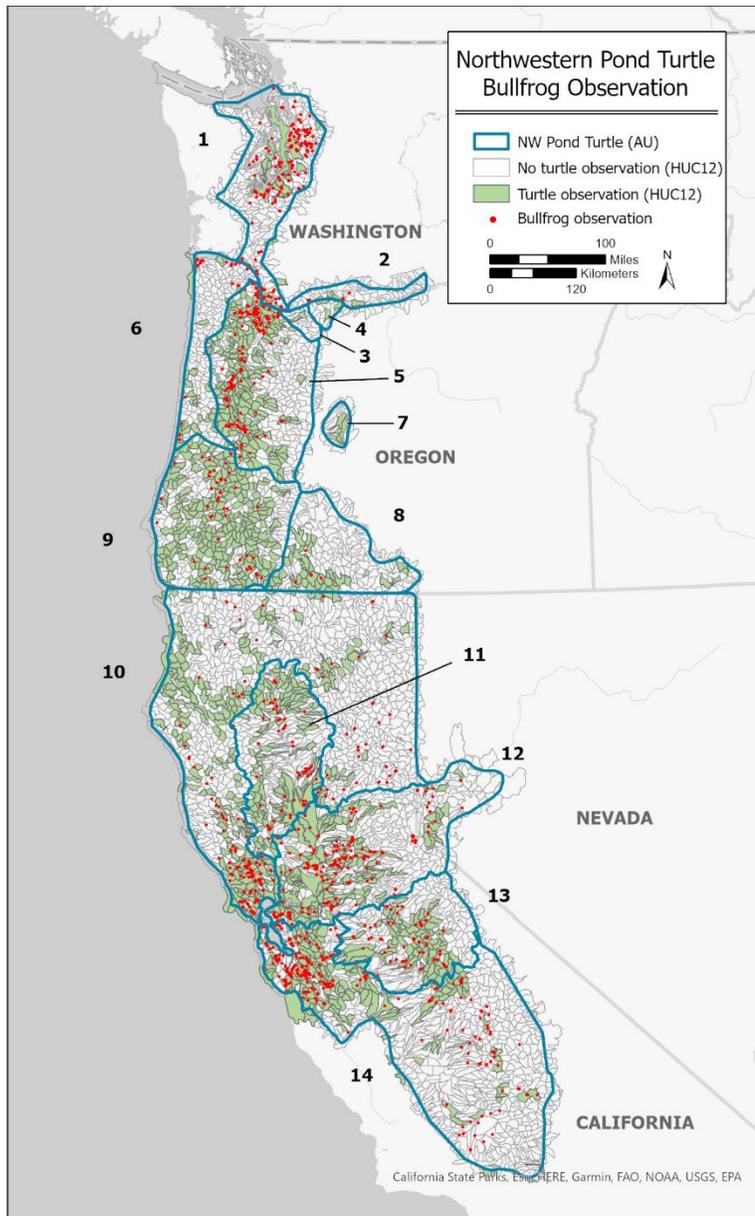


Figure 24. Bullfrog observations within the range of the northwestern pond turtle. Bullfrog observations are from GBIF (GBIF.org 2022). Subwatersheds with northwestern pond turtle presence observations are also shown for reference.

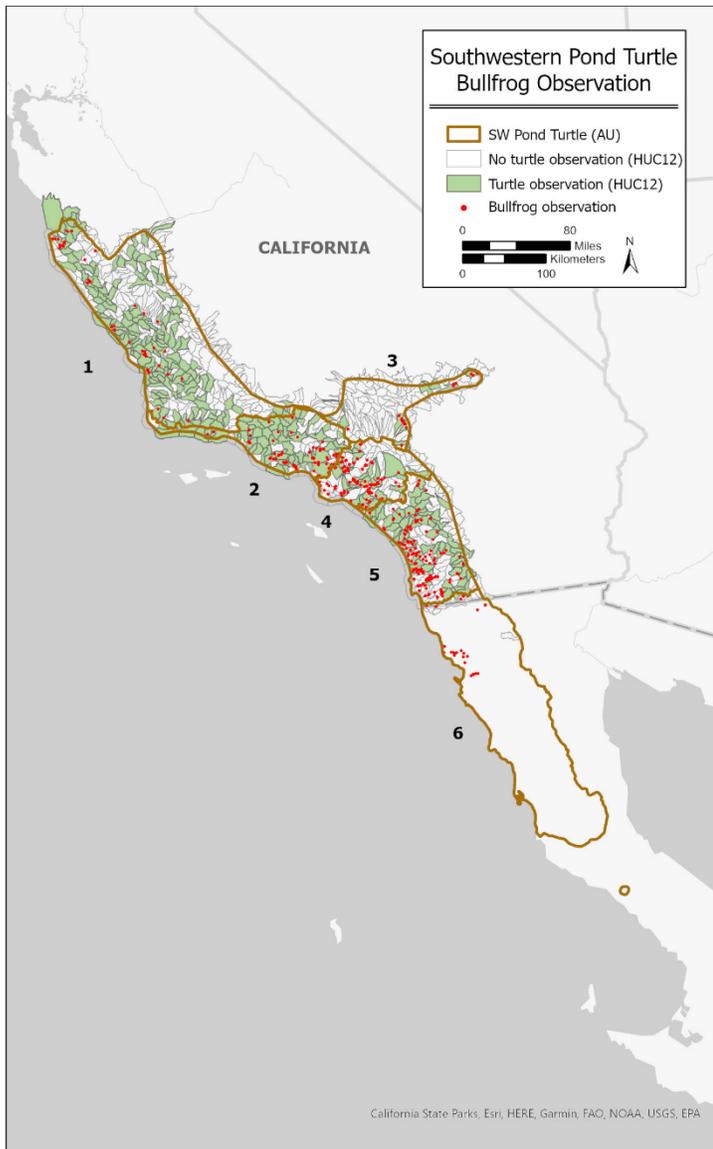


Figure 25. Bullfrog observations within the range of the southwestern pond turtle. Bullfrog observations are from GBIF (GBIF.org 2022). Subwatersheds with southwestern pond turtle presence observations are also shown for reference.

Table 8. Proportion of HUC12s occupied by northwestern pond turtles that have bullfrog observations.

Analysis Unit	State	Number	Proportion of occupied HUC12s with bullfrogs
Bull Run-Sandy	OR	AU-3	0
Mosier-Hood	OR	AU-4	0.3
Willamette	OR	AU-5	0.4
North Coast	OR	AU-6	0.1
Bend	OR	AU-7	0
Klamath-Lakeview	OR	AU-8	0.1
Southwest Oregon	OR	AU-9	0.2
Northern California	CA	AU-10	0.3
North Central Valley	CA	AU-11	0.2
Bay Delta and Nevada	CA/NV	AU-12	0.6
Yosemite	CA	AU-13	0.4
San Joaquin Valley	CA	AU-14	0.5

Table 9. Proportion of HUC12s occupied by southwestern pond turtles that have bullfrog observations.

Analysis Unit	State	Number	Proportion of occupied HUC12s with bullfrogs
Coast Range	CA	AU-1	0.3
Ventura/Santa Barbara	CA	AU-2	0.6
Mojave	CA	AU-3	0.7
Los Angeles	CA	AU-4	0.9
Orange County/San Diego	CA	AU-5	0.7
Baja California	Mexico	AU-6	N/A

9.3 Modeling Population Growth and Probability of Extinction

To quantitatively assess current condition, we present results from a single sex (female) stochastic stage-based (hatchling, juvenile, adult) matrix population model developed by Gregory and McGowan (2023, entire), which is also included in this SSA report as Appendix A. We consider model results through 2050, in approximately 25 years, to be the current condition of western pond turtle analysis units, which is equivalent to approximately one generation.

Model Methods

The model incorporates information on western pond turtle presence, specifically occurrence observations (as shown in Figure 13 and Figure 14), as well as data from Manzo et al. (2021, supplemental Table 2) and primary threats (anthropogenic impacts, drought, and bullfrogs) as described above. Before we describe the model more fully, we first list several things that the model does not do. Headstarting was not explicitly included in the model because of uncertainty in the status/scope moving forward. The model also does not incorporate impacts from disease. Disease in western pond turtles is not well-understood, but is especially of concern in Washington, where all populations are supplemented with headstarted turtles that have been in captivity (in which shell disease is more prevalent than in wild turtles; see Chapter 8.5 Disease).

For northwestern pond turtles, Gregory and McGowan (2023, entire) do not model abundance in Washington State because the repeated reintroductions of headstarted individuals would confound demographic parameters used in their model. In Washington, most northwestern pond

turtles have been headstarted and released, with robust capture and population data yielding an estimated population for the State, across all six sites. We consider the current condition of the Washington analysis units (Puget Sound, AU-1; Columbia Gorge, AU-2) to be conservation reliant, as all six recovery sites require management efforts to maintain their existence (Hallock et al. 2017, p. 14).

For southwestern pond turtles, the model does not include Baja California (AU-6), for which we consider the current condition to be unknown.

Starting Abundance

Because available information in the literature and available survey data lack information to quantify current abundance within most of the analysis units, the model began by estimating initial abundance. To do so, Gregory and McGowan (2023, pp. 9–14) performed a stepwise process to incorporate presence data, available habitat information combined with anthropogenic impacts, and the effects of the other two primary threats: drought and bullfrog predation. Using presence observations (from the same dataset as shown in Figure 13 and Figure 15), Gregory and McGowan (2023, p. 9) calculated probability of occupancy in HUC12 subwatersheds using a distribution of adult survival rates that was applied based on the number of years since the most recent western pond turtle observation. Next, to generate an abundance within occupied subwatersheds, they used a negative binomial linear generalized linear regression model with habitat area (riparian area, delineated by the National Riparian Areas Base Map from the U.S. Forest Service; Abood et al. 2022, entire) and the human modification index (Theobald et al. 2020, entire) as potential explanatory variables. For the response variable in the model, they transformed population size estimates from Manzo et al. (2021, Supplemental Table 2) using detection probabilities from Fulton et al. (2022, p. 102). In the regression, western pond turtle abundance was higher with increasing habitat area and lower human modification (Gregory and McGowan 2023, p. 11). They summed these abundances across occupied HUC12s within each analysis for what the model calls “habitat-based abundance”. They then applied information related to recent droughts and bullfrog presence to “habitat-based abundance,” and again summed the HUC12 totals to derive starting abundance in the model. For this calculation, they divided the habitat-based abundance into age classes used in the model (hatchlings, juveniles, and adults), and applied threat survival metrics based on the age class and threat. Additional information about threat parameters is detailed below and explained more fully in Gregory and McGowan (2023, entire), but described briefly here. To incorporate recent droughts, Gregory and McGowan (2023, pp. 13–14) examined PDSI data beginning in 2022, going back one year at a time to determine the duration of the current drought, but stopping once a non-drought year was encountered. For example, all analysis units in the southwestern pond turtle range were classified as having extreme drought in 2022 and 2021, but not 2020, so the examination ended there, with survival rates based on two years extreme drought applied to the habitat-based abundance (i.e., reduced by 16.7 percent for the first year and 44 percent for the second year

based on Table 4 in Gregory and McGowan 2023). To incorporate information on bullfrog presence, Gregory and McGowan (2023, p. 14) substituted hatchling and juvenile abundance for their respective survival values based on the proportion of area of HUC12s occupied by bullfrogs in comparison to the entire analysis unit area.

Impacts from Threats

This section describes the threats that are incorporated into the population model in Gregory and McGowan (2023, entire). Incorporation in model scenarios is described in the next section

Land Conversion: To assess the impact of land conversion, Gregory and McGowan (2023, pp. 18, 22) used ICLUS data on projected wetland land use change through 2100. Gregory and McGowan (2023, pp. 18, 22) isolated the wetland land cover type, assuming that rates of change in riparian area are an index to wetland area. The ICLUS projections report area of land cover types for each state in 20-year periods. For each 20-year period, Gregory and McGowan (2023, p. 22) calculated an annual rate of habitat loss or increase under two scenarios. The projections of annual habitat loss were assigned to each analysis based on the state they are in: Washington, Oregon, or California. The Bay Delta and Nevada analysis unit (AU-12) was assigned the California rate because it is primarily in that state.

The model used a ceiling type density function to mimic habitat loss, such that the maximum amount of suitable habitat was reduced each year based on annualized projected wetland loss. In this way, Gregory and McGowan (2023, pp. 18, 22) reduced the future carrying capacity based on land use change projections, assuming that less habitat would be suitable for western pond turtles in conjunction with land use change; this change did not result in demographic changes to western pond turtles in the model (i.e., they did not make changes to survival, etc.), instead focusing only on available habitat.

Drought: To assess impacts of drought, Gregory and McGowan (2023, pp. 15–17) used PDSI data, focusing on data from June to August of each year because this period corresponds with when western pond turtles are active and breeding. They calculated the weighted average of PDSI values across NOAA climate divisions to determine an annual PDSI value for each analysis unit. They classified annual PDSI values of less than -2 but greater than -4 as moderate drought, and annual PDSI values of less than -4 as extreme drought, based on categorization in the US Drought Monitor; note that this categorization scheme also defines PDSI <-5 as exceptional drought, which was not differentiated from extreme droughts in the model (National Drought Mitigation Center 2022). This categorization is consistent with that used in studies on the effects of drought on western pond turtle survival (Lovich et al. 2017, p. 3; Purcell et al. 2017, p. 20). The model included reductions in survival and proportion of breeding females based on drought severity, duration, and combinations thereof (Gregory and McGowan 2023, Table 4).

To project drought conditions through 2100, Gregory and McGowan (2023, pp. 19–21) used annual rates of increase of moderate and extreme droughts. These rates of increase were determined by comparing annual drought frequencies of historical and recent time periods, which they defined as 1986–2005 and 2006–present, respectively. Using 1986–2005 as the baseline for comparison is consistent with IPCC climate scenarios that use this same baseline period and 2006 as a starting point for RCP scenarios (Collins et al. 2013, p. 1034).

Bullfrog predation: Because direct measures of demographic effects from bullfrogs are difficult to quantify, Gregory and McGowan (2023, p. 17) developed a way to incorporate bullfrog effects into hatchling and juvenile survival rates using size class data for western pond turtles reported in areas with and without bullfrogs present. They assumed that bullfrogs can depredate any hatchlings, and juveniles less than 3 years of age, because these small turtles would be vulnerable to bullfrogs. Survival of western pond turtles in analysis units with bullfrogs was the juvenile survival rate from the literature in areas with bullfrogs (ranging from 0 to 0.87) multiplied by the proportion of the population that are small turtles (hatchlings and juveniles less than 3 years old) in areas with bullfrogs, divided by the proportion of the population that are small turtles \leq 3 years old) in areas with no bullfrogs.

To model effects of bullfrogs into the future, they determined the rate of bullfrog spread throughout the range of the species. To do so, Gregory and McGowan (2023, pp. 21–22) compared observations of bullfrogs in 2006 to observations in 2022, calculating the annualized difference between proportion of HUC12s with bullfrogs in each analysis unit between these two time periods. HUC12s not occupied by western pond turtles were considered in this analysis under the assumption that bullfrogs could spread through adjacent neighboring HUC12s into those occupied by western pond turtles.

Model Scenarios

Under two plausible scenarios, Gregory and McGowan (2023, pp. 18–39) projected probability of extinction and changes in abundance through the year 2100 because this is how far anthropogenic influences and climate changes are projected in the datasets used. We consider results from the model at 2050 (approximately 1 western pond turtle generation) to represent current condition of western pond turtles, and also present model results at 2075 (approximately 2 generations) and 2100 (approximately 3 generations) in Chapter 10 Future Condition. Specifically, the model uses quasi-extinction as the threshold under which the western pond turtle numbers within an analysis unit would be so small that it would no longer be viable; this threshold was 5 percent of the habitat-based abundance of each analysis unit (described within the *Starting abundance* section). This approach is similar to that used in other population viability analyses for reptiles: McGowan et al. (2017, p. 122) set quasi-extinction thresholds at 2 percent and 4 percent of the maximum estimated population size, and the Service (2021, p. 70) considered a decline to less than 5 percent of the initial starting size as quasi-extinction.

Extinction is used synonymously with quasi-extinction in the remainder of this document, under the assumption that if analysis units fall to these levels then they are functionally extirpated and unlikely to sustain populations in the wild. We use both probability of extinction and probability of extirpation interchangeably when discussing results from the model.

Gregory and McGowan (2023, pp. 18–22) developed two plausible future scenarios for their stochastic simulation model that bound the plausible range of future influences on western pond turtles. They used the three primary threats as discussed above (anthropogenic impacts, drought, and bullfrogs) to model potential impacts to the two species into the future and provide modeled outputs of probability of extinction. Scenario 1, based on Representative Concentration Pathway (RCP) 8.5 in which greenhouse gas emissions continue to increase into the next century, incorporates the more impactful plausible limit for each of the three threats. Scenario 2, based on RCP 4.5 in which greenhouse gas emissions stabilize by mid-century and then decline to levels seen in the 1990s by the end of the century, includes the less impactful plausible limit of projections for each influence.

To model impacts from land conversion, Gregory and McGowan (2023, p. 22) used data from ICLUS. In Scenario 1, they used the RCP 8.5/SSP5 projection from the ICLUS model and in Scenario 2, they used the RCP 4.5/SSP2 scenario. Rates of land conversion under these scenarios are shown in Gregory and McGowan (2023, Table 8).

The model projected drought conditions into the future by calculating annual rates of increase of moderate and extreme drought for RCPs 4.5 and 8.5 at the analysis unit scale. Under Scenario 1, the model projected these same rates of increase into the future, because RCP 8.5 aligns most closely with our current trajectory of greenhouse gas emissions (Schwalm et al. 2020, entire). Under the assumption that annual rates of increase would be reduced compared to current under RCP 4.5, for Scenario 2, annual rates of increase were multiplied by 0.64 based on the difference in mean PDSI values between the RCPs by 2100 reported in Cook et al. (2015, pp. 2–3). The effects of droughts based on drought severity and duration were modeled, reducing survival of all age classes and the proportion of adults that bred in any given year as shown in Gregory and McGowan (2023, Table 4).

The model incorporated impacts from bullfrog predation by looking at rates of change in bullfrog occupancy within analysis units between 2006 and 2022, assuming that these rates would be maintained into the future (Gregory and McGowan 2023, Table 7). If there was no spread of bullfrogs over that time period, then bullfrog presence was not increased in the future projections. They modeled the impacts of bullfrogs on western pond turtles by decreasing survival of hatchlings and vulnerable juveniles based on the proportion of bullfrogs in an analysis unit through 2100. Vulnerable juveniles were determined based on a proportion predicted to be small enough to be predated by bullfrogs, as described in Gregory and McGowan (2023, p. 17). Impacts from bullfrogs were the same in both scenarios.

Table 10 summarizes the ways that land conversion, drought, and bullfrog predation are incorporated into Scenario 1 and Scenario 2. As stated earlier, the model outputs include probability of extirpation through 2100, with extinction defined as 5 percent of the habitat-based abundance calculated in the modeling exercise.

Table 10. Summary of conditions of primary threats to western pond turtle under two plausible scenarios.

Threat	Scenario 1 (RCP 8.5)	Scenario 2 (RCP 4.5)
Land conversion	Ceiling type density function mimicking wetland loss by State under RCP 8.5/SSP 5	Ceiling type density function mimicking wetland loss by State under RCP 4.5/SSP 2
Drought	Drought frequency under RCP 8.5; annualized rates of moderate and extreme drought within analysis units, with impacts to survival of all age classes and to the proportion of breeding females, after 4+ years of moderate drought, 1, 2, or 3 years of severe drought, or a combination	Drought frequency under RCP 4.5; annualized rates of moderate and extreme drought within analysis units (0.87 of RCP 8.5 rates of increase), with impacts to survival of all age classes and to the proportion of breeding females, after 4+ years of moderate drought, 1, 2, or 3 years of severe drought, or a combination
Bullfrog presence	Bullfrog spread based on continuation of rate of change over last 17 years across analysis units, with impacts to survival of hatchlings and small juveniles	Bullfrog spread based on continuation of rate of change over last 17 years across analysis units, with impacts to survival of hatchlings and small juveniles

Model Results

Northwestern Pond Turtle

For northwestern pond turtle analysis units, population growth through 2050 was below 1, indicating a decreasing population size. Probability of extinction in analysis units 3–14 in 2050 was found to be low, ranging from 0.064 to 0.152 in Scenario 1 and 0.055 to 0.146 in Scenario 2, indicating a 5 to 15 percent likelihood of extinction in the next approximately 25 years.

Scenario 1, P(extinction) in 2050

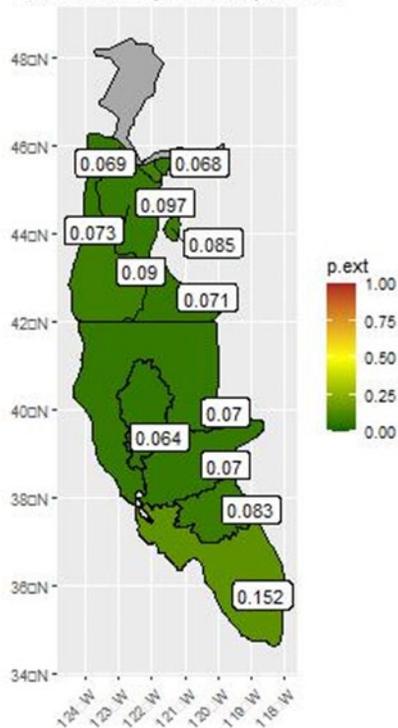


Figure 26. Probability of extinction in 2050 for Scenario 1 in each northwestern pond turtle analysis unit, except for analysis units in Washington.

Scenario 2, P(extinction) in 2050

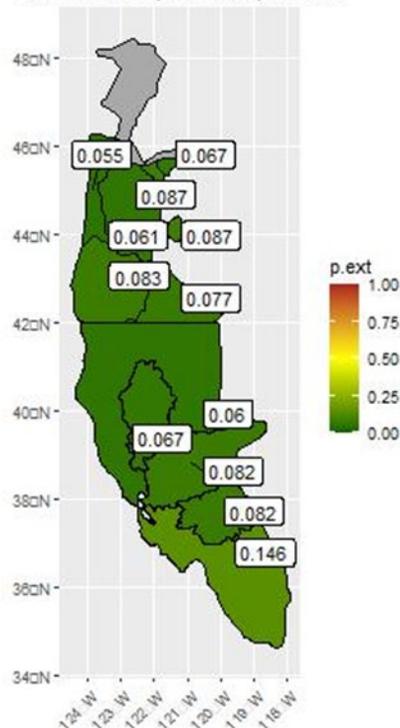


Figure 27. Probability of extinction in 2050 for Scenario 2 in each northwestern pond turtle analysis unit, except for analysis units in Washington.

Southwestern Pond Turtle

For southwestern pond turtle U.S analysis units, population growth through 2050 was below 1, indicating a decreasing population size. Probability of extinction for U.S analysis units in 2050 was found to be low, ranging from 0.211 to 0.244 in Scenario 1 and 0.197 to 0.228 in Scenario 2 (Figure 28, Figure 29), indicating a 20 to 24 percent likelihood of extinction in the next approximately 25 years.

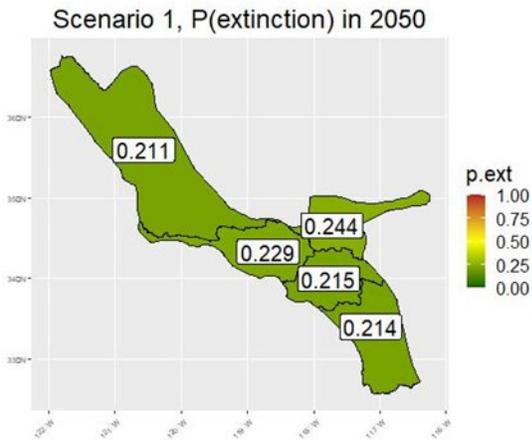


Figure 28. Probability of extinction in 2050 for Scenario 1 in each southwestern pond turtle analysis unit, except for the analysis unit in Baja California (AU-6).

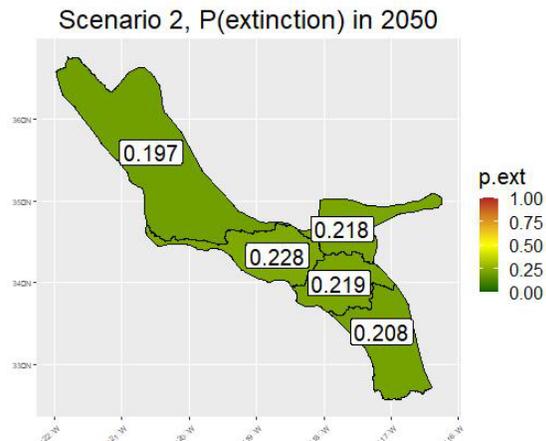


Figure 29. Probability of extinction in 2050 for Scenario 2 in each southwestern pond turtle analysis unit, except for the analysis unit in Baja California (AU-6).

9.4 Current Analysis Unit Resiliency

Probability of extinction in approximately one generation (approximately 25 years, through 2050) is our measure of resiliency for northwestern pond turtles for all analysis units except those in Washington. For Washington analysis units, we also consider a separate population viability analysis (PVA) conducted by Pramuk et al. (2013, entire), with the methodology for that PVA discussed in more detail in Chapter 10.1.

Northwestern Pond Turtle

Under both scenarios, probability of extinction at 2050 ranged from approximately 5 to 15 percent across analysis units. Analysis units in Washington (AU-1, and AU-2), are conservation reliant, as all six recovery sites require management efforts to maintain their persistence, including headstarting. Although a Washington PVA indicates rapid decline in population size without management intervention, in 2050 the PVA indicates the species will persist in Washington. Overall, the probability of persistence values of 85 to 95 percent in modeled analysis units, as well as persistence in the Washington PVA, suggest that the northwestern pond turtle analysis units are likely to withstand stochastic events in the next approximately 25 years.

Southwestern Pond Turtle

Under both scenarios, probability of extinction at 2050 ranged from approximately 20 to 24 percent across the U.S. analysis units. The probability of persistence values of 75 to 80 percent

suggest that U.S. analysis units are likely to withstand stochastic events in the next approximately 25 years.

Current resiliency of the Baja California analysis unit (AU-6) is unknown.

9.5 Current Species Redundancy

We assess redundancy by evaluating the distribution of resilient analysis units for northwestern and southwestern pond turtles across their range.

Northwestern Pond Turtle

Northwestern pond turtles are currently distributed across 4 states, although most of the species' range is within Oregon and California. This wide distribution fundamentally protects the species from catastrophic events. Although long-term severe drought is a potential threat that may impact large extents of the species' range, the latitudinal spread, particularly across the three Pacific states, reduces the risk of this threat, particularly in the near term. The analysis units in the more southern portion of the range have higher probability of extinction, but none of the analysis units are likely to become extirpated by 2050 (85 to 95 percent probability of persistence across modeled analysis units, declining abundance but persistence in Washington PVA). Currently, while Washington requires management intervention and the southern parts of the range have higher probability of extinction, the species likely maintains its ability to withstand most catastrophic events.

Southwestern Pond Turtle

Southwestern pond turtle populations are currently distributed across the central coast and southern California, and Baja California, Mexico. Impacts from long-term drought are currently impacting most of the species' range at similar trajectories, resulting in similar probabilities of extinction in the U.S. analysis units, and any localized extirpations would reduce redundancy for the species. However, given that probability of extinction of the analysis units by 2050 is low (75 to 80 percent probability of persistence), this indicates that the species likely maintains its ability to withstand most catastrophic events.

9.6 Current Species Representation

To assess the current representation for the western pond turtle, we considered the current ecological and genetic diversity throughout the range of the species.

Northwestern Pond Turtle

Both northwestern and southwestern pond turtles exhibit ecological flexibility in habitat use. As discussed in Chapter 6, western pond turtles use a diverse set of permanent and ephemeral

aquatic water bodies, including flowing rivers and streams, lakes, ponds, reservoirs, settling ponds, marshes, vernal pools, brackish waters, irrigation ditches, and other wetlands (Spinks et al. 2003, entire; Bury and Germano 2008, p. 001.4; Bury et al. 2012, p. 12; Stebbins and McGinnis 2018, p. 205). Vegetative characteristics of upland habitat used by the species also vary across study sites. The various types of aquatic and upland habitat used by the species demonstrates environmental diversity, which contributes to the species' representation. Further, the species uses habitat at a variety of elevations, from sea level to about 2000 m in elevation. Although Germano and Riedle (2015, p. 107) report that population traits are similar across elevational differences, the use of habitats across this gradient adds to representation, as these areas may face differences in threats (e.g., drought, wildfires).

Severe reductions in the range and low population sizes in Washington, combined with reduced genetic diversity, have lowered representation for the northwestern pond turtle in Puget Sound (AU-1) and the Columbia River Gorge (AU-2). Specifically, genetic analysis in the mid-1990s revealed a lack of genetic variation in the Bergen and Sondino populations (Gray 1995, entire). Although headstarting has increased the numbers of northwestern pond turtles in the wild and reintroduced the species to some areas, the Pierce County and Mason County populations originate from a founder population of just 12 turtles, indicating that these areas also likely have low genetic diversity (Hallock et al. 2017, p. 10).

Northwestern pond turtles in Oregon and northern California are part of the same genetic relatedness cluster. These northwestern pond turtles that are farther north in the range (i.e., north of the Bay Area into Oregon; analysis units 3 through 10) had the lowest genetic variation of the northwestern pond turtle genetic clusters, and had nearly double the inbreeding levels compared to the San Joaquin Valley (AU-14) genetic cluster (Shaffer and Scott 2022, pp. 6–7).

Taken together, northwestern pond turtles exhibit ecological flexibility, but have evidence of low genetic variation in a large portion of the species' range. However, overall we do not expect severe population declines or extirpations in the near-term (85 to 95 percent probability of persistence across analysis units in 2050). Therefore, the species is likely to maintain its ability to adapt to changing environmental conditions in the next 27 years.

Southwestern Pond Turtle

Southwestern pond turtles exhibit ecological flexibility in habitat use as discussed above for northwestern pond turtles.

Within the southwestern pond turtle, most genetic clusters have relatively high levels of inbreeding, and there was very elevated inbreeding level in both the Mojave and Baja genetic units (AU-3 and AU-6).

In terms of representation, southwestern pond turtles exhibit ecological flexibility in habitat use, but high inbreeding levels across much of the range. However, overall we don't expect severe populations declines or extirpations in the near-term (75 to 80 percent probability of persistence in 2050). Therefore, the species is likely to maintain its ability to adapt to changing environmental conditions in the next 27 years.

9.7 Current Condition Uncertainty

In our analysis, we used the three threats that we determined to be most influential in driving western pond turtle viability. Although these threats interact with many of the other known threats to the species, the analysis likely underestimates the overall impact of threats to the species.

Assuming uniformity across HUC12 subwatersheds could lead to under- or overestimates of western turtle populations and impacts from threats, although taken together the results likely overestimate condition. For example, we assumed that an observation of 1 bullfrog within a HUC12 equated to the whole HUC12 being occupied, which may have overestimated the threat. At the same time, we assumed that observation of 1 western pond turtle in a HUC12 indicated uniform occupancy across all of the available habitat, which may have overestimated condition. Additionally, much of the available data suggests that populations are skewed towards males and adults, which suggests that populations may be in worse condition than that presented here.

10.0 FUTURE CONDITION

Below we discuss the potential future condition of the northwestern and southwestern pond turtles based on future threats facing the two species and their responses to those threats, in terms of their resiliency, redundancy, and representation. We assessed future condition of western pond turtles for the same analysis units for which we presented the current condition. For analysis units where we were unable to present information on modeled abundance in current condition, we consider the future condition to be unknown. For southwestern pond turtles, this includes the Baja California population (AU-6). Methods and results of a model (Gregory and McGowan 2023, entire) and our future condition assessment for the two species are described below. For detailed information on modeling methods please see Appendix A of this document (Gregory and McGowan 2023 entire).

10.1 Future Condition Summary of Methods

For Washington analysis units, we present results from a PVA that incorporates demographic data specific to these sites (Pramuk et al. 2013, entire). The PVA for Washington northwestern pond turtles is an individual-based model in the program Vortex, which models population dynamics as discrete sequential events according to parameters as defined by the modelers. At the time of the analysis, the estimated population of northwestern pond turtles in Washington

was approximately 1,200 to 1,500 individuals (Pramuk et al. 2013, p. 19), compared to a total estimate of 800 to 1,000 individuals in 2015 (Hallock et al. 2017, p. 5) and approximately 800 individuals when summing the most recent population estimates (Hallock et al. 2017, p. 6; Bergh and Wickhem 2022, pp. 4–6, WDFW 2022, in litteris). However, the initial population size used in the PVA was 194, reflective of the Pierce County site population (Pramuk et al. 2013, p. 25). The PVA explored what levels of survival were needed at various age classes for populations to be maintained without continued releases of headstarted individuals (Pramuk et al. 2013, entire). To determine how sensitive the PVA was to input parameters, Pramuk et al. (2013, p. 26) also calculated elasticity values – the measure of the proportional change in a population’s growth rate given changes to a specific parameter while all others are held constant. Bullfrog predation was incorporated into the PVA in that mean survival rate for hatchlings that were reared in captivity was 45 percent, while mean survival of hatchlings in the wild was only 5 percent (Pramuk et al. 2013, p. 24). Bullfrog mitigation that lowered mortality of wild hatchlings to 85 percent was also included as a scenario (Pramuk et al. 2013, p. 28). Drought was not explicitly incorporated into the Washington PVA.

10.2 Future Condition Results

Northwestern Pond Turtle

Specific to northwestern pond turtles in Washington State, Pramuk et al. (2013, pp. 28–29), found that populations declined rapidly towards extinction in the absence of headstarting. When looking at adult females only, there was an initial bump in abundances that reflected the transition of sub-adults to adults, where the number of adult females increased even as the overall population declined, after which female abundance declined towards extinction. The model was highly sensitive to survival rates of adults (>10 years) and subadults (>2 years), but less sensitive to changes in hatchling survival, juvenile survival, age of first reproduction, and maximum age of reproduction (Pramuk et al. 2013, pp. 26–27). When exploring the survival rates needed to sustain populations without headstarting, the key adult mortality rate that could lead to a stable population hinged around 12.5 percent, with hatchling mortality rates in conjunction with that adult mortality level ranging from 80 to 85 percent depending on the initial population abundance (Pramuk et al. 2013, pp. 30–32).

Across the analysis units in California, Nevada, and Oregon, population growth rate was below 1 for the duration of the model simulation, and probability of extinction rises over time (Gregory and McGowan 2023, pp. 23–32). Probability of extinction in 2075 ranges from 0.284 to 0.428 under Scenario 1 and 0.287 to 0.4115 under Scenario 2, indicating a 28 to 43 percent likelihood of extinction in the next approximately 50 years. Probability of extinction in 2100 ranges from 0.468 to 0.589 under Scenario 1 and 0.456 to 0.594 under Scenario 2 (Figure 30, Figure 31), indicating a 46 to 59 percent likelihood of extinction in the next approximately 75 years. Probability of extinction showed a geographic trend, with more northern analysis units having

lower probability of extinction. This is consistent with expectations based on the model inputs: drought had significant impacts on adult survival, and future projected drought frequencies are higher in more southern analysis units.

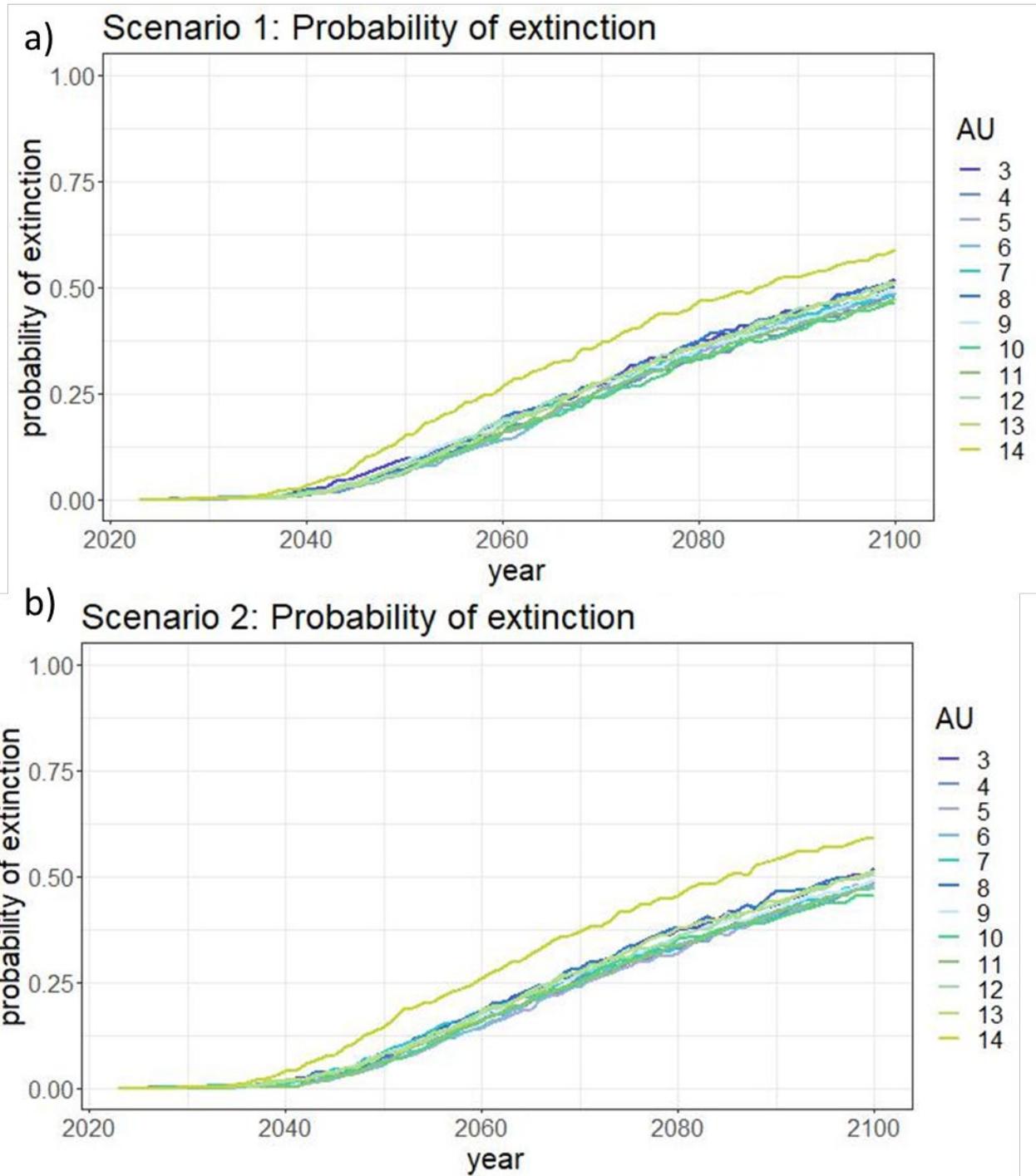


Figure 30. Probability of extinction of northwestern pond turtles in CA, NV and OR analysis units 3 through 14 through 2100 under a) Scenario 1) and b) Scenario 2. See Appendix A for individual plots showing both scenarios for each analysis unit. Colors are the same as that shown in the analysis unit map (Figure 8).

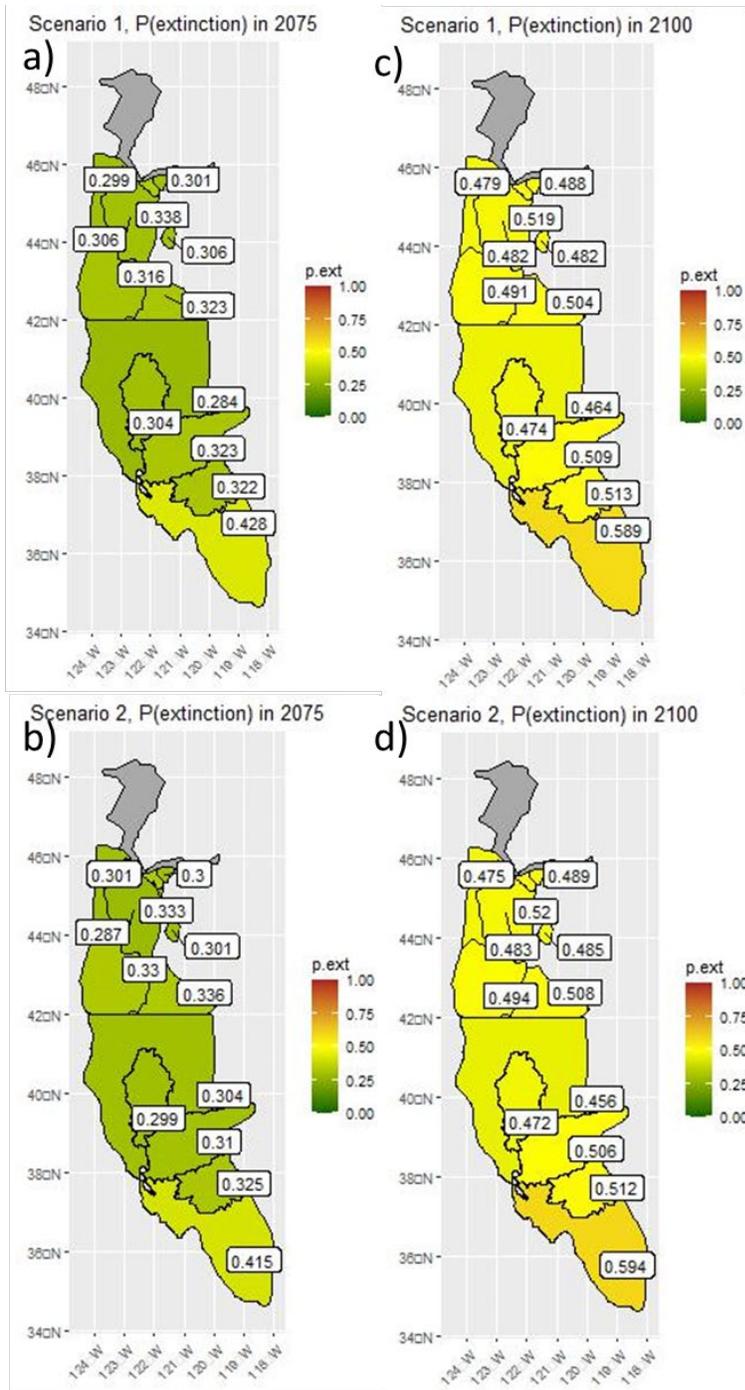


Figure 31. Probability of extinction of northwestern pond turtles in CA, NV, and OR analysis units 3 through 14 in 2075 under a) Scenario 1) and b) Scenario 2, and in 2100 under c) Scenario 1 and d) Scenario 2.

Southwestern Pond Turtle

Across all southwestern pond turtle U.S. analysis units, population growth rate was below 1 for the duration of the model simulation, and probability of extinction rises over time. Probability of extinction in 2075 ranges from 0.544 to 0.570 under Scenario 1 and 0.509 to 0.554 under Scenario 2, indicating a 51 to 57 percent likelihood of extinction in the next approximately 50 years. Probability of extinction in 2100 ranges from 0.734 to 0.776 under Scenario 1 and 0.703 to 0.732 under Scenario 2 (Figure 32, Figure 33), indicating a 70 to 78 percent likelihood of extinction in the next approximately 75 years. Model results were most sensitive to increases in drought, especially in the LA (AU-4) and Orange County/San Diego (AU-5) analysis units, where probability of extinction in 2100 increased by up to 18.5 percent when annual drought frequency was increased by only 25 percent.

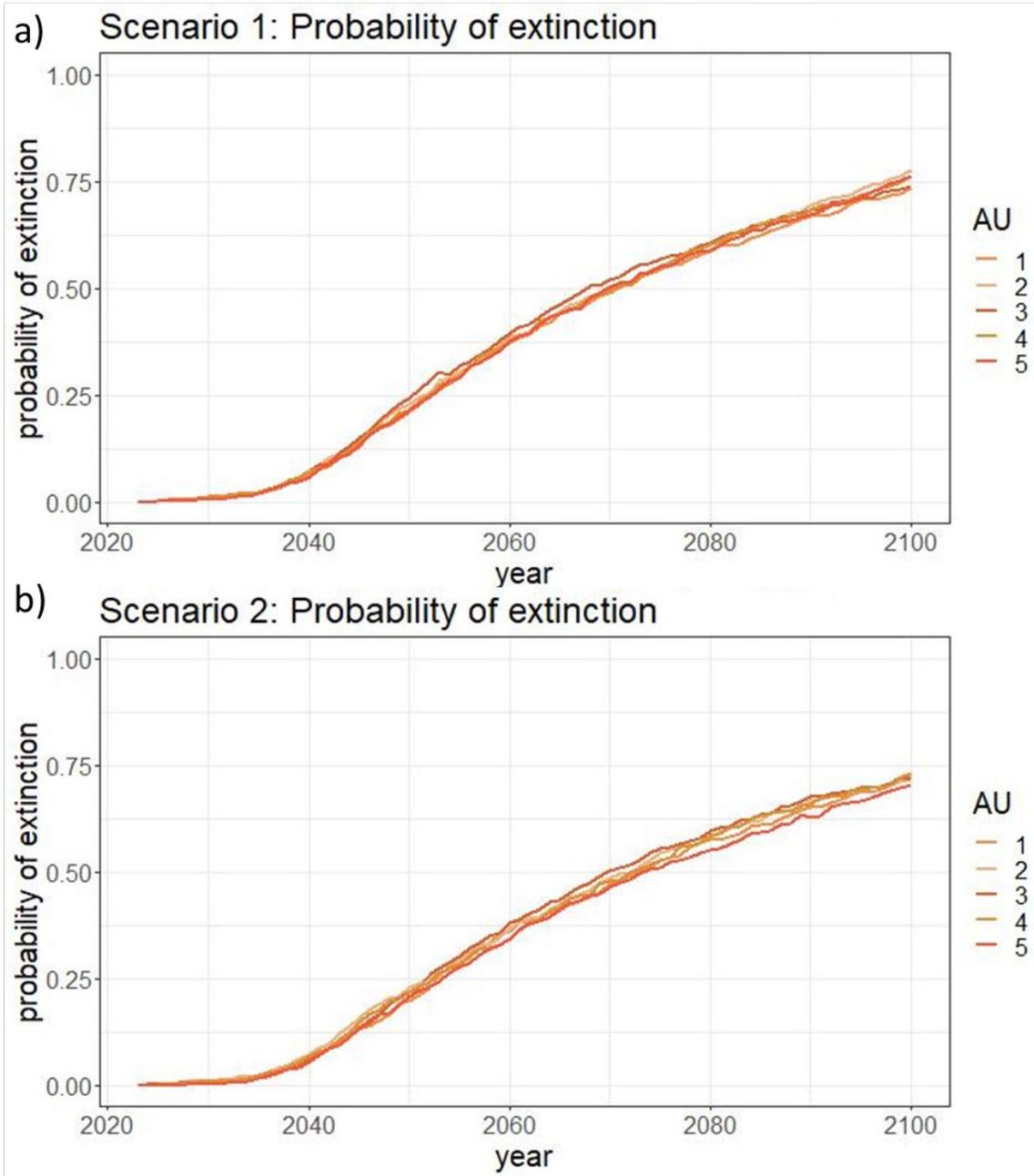


Figure 32. Probability of extinction of southwestern pond turtles in the U.S. by 2100 under a) Scenario 1) and b) Scenario 2. See Appendix A for individual plots showing both scenarios for each analysis unit. Colors are the same as that shown in the analysis unit map (Figure 9).

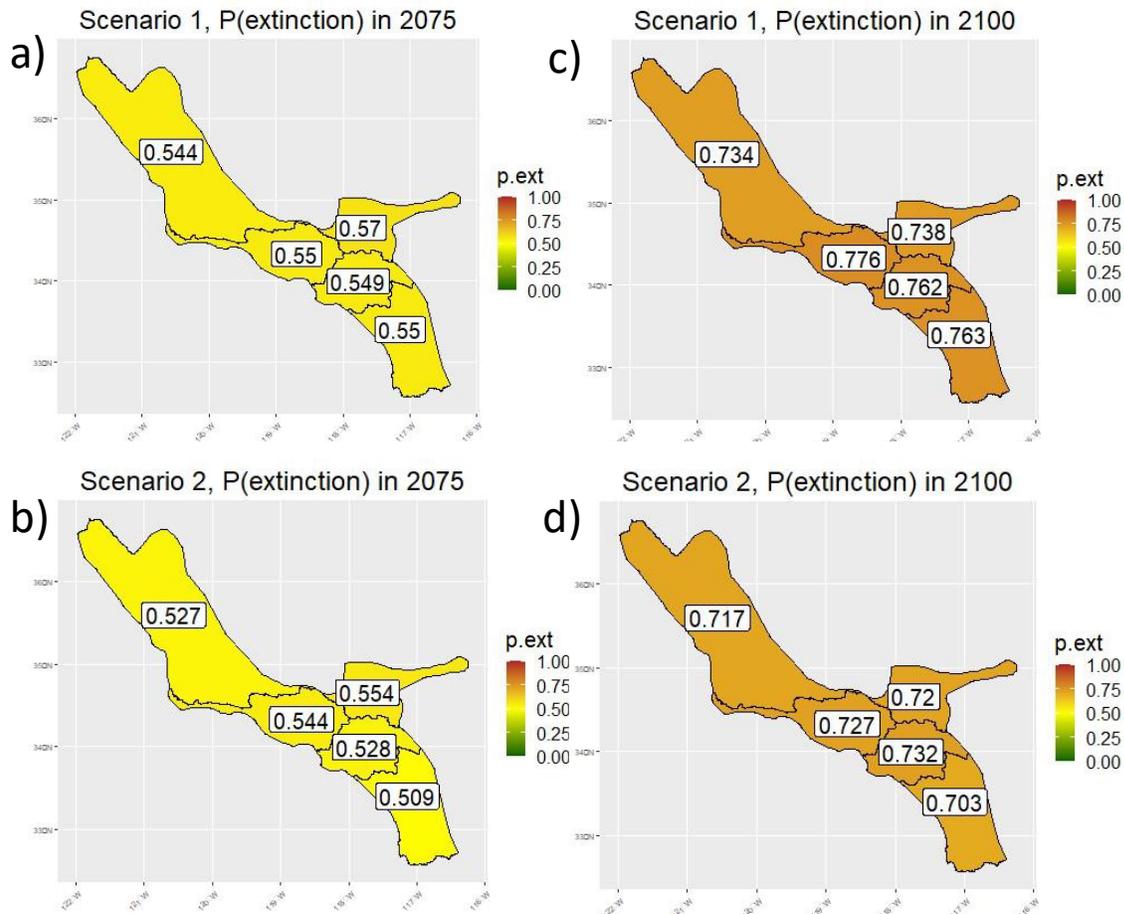


Figure 33. Probability of extinction of northwestern pond turtles in the U.S. in 2075 under a) Scenario 1) and b) Scenario 2., and in 2100 under c) Scenario 1 and d) Scenario 2.

10.3 Future Condition Analysis Unit Resiliency

Probability of extinction in approximately two and three generations (approximately 50 years through 2075 and 75 years through 2100, respectively) is our measure of resiliency for western pond turtles for all analysis units except those in Washington for northwestern pond turtles.

Northwestern Pond Turtle

In Washington, populations declined rapidly towards extinction in the absence of headstarting in the Pramuk et al. (2013, pp. 28–29) PVA, indicating that these analysis units are not able to withstand stochastic events without human intervention into the future.

For the remainder of the analysis units, due to impacts from future land conversion, bullfrog predation, and increasing drought, model results showed increasing probability of extinction for northwestern pond turtle analysis units over time. Probability of extinction in 2075 ranges from 28 percent to 41 percent under the two modeled scenarios, indicating that persistence is more likely than extinction in all analysis units (i.e., likelihood of extinction is less than 50 percent) for the next approximately 50 years. Probability of extinction in 2100 ranges from 45 percent to 60 percent under the two modeled scenarios, indicating that extinction becomes more likely than not in some analysis units in approximately 75 years. Even without overall analysis unit extirpation, increasing probability of population-level extinction is associated with decreases in resiliency as abundance declines over time. Resiliency of these analysis units generally follows a latitudinal trend, with more southern analysis units having higher likelihood of extinction and therefore lower ability to withstand stochastic events in the future.

Southwestern Pond Turtle

Due to impacts from future land conversion, bullfrog predation, and especially increasing drought, southwestern pond turtle analysis unit resiliency is likely to be highly reduced in the future, with model results showing rapidly increasing probability of extinction. Chance of extinction rises in all analysis units under both scenarios between 2075 and 2100. In 2075, probability of extinction across analysis units is over 50 percent, indicating that extinction is more likely than persistence in approximately 50 years. By 2100, likelihood of extinction rises to over 70 percent across the range in approximately 75 years. Overall, model results indicate steeply increasing likelihood of analysis unit extinction and therefore substantially reduced resiliency across all analysis units as compared to current conditions. Substantially reduced resiliency means that the species would be significantly less able to withstand stochastic events in the future.

10.4 Future Species Redundancy

We assess redundancy by evaluating the future distribution of resilient analysis units for northwestern and southwestern pond turtles across their range.

Northwestern Pond Turtle

In 2075, none of analysis units 3 through 14 are more likely than not to be functionally extinct under either scenario, although probabilities of extinction are over 30 percent for most analysis units. While Washington requires management intervention and the southern parts of the range have higher probability of extinction, the species likely maintains its ability to withstand most catastrophic events over the next approximately 50 years.

By 2100, 5 out of these 12 analysis units are more likely than not (greater than 50 percent) to be functionally extinct under both scenarios, and likelihood of extinction for the other all analysis

units is above 45 percent. Both analysis units in Washington decline rapidly towards extinction in the absence of headstarting. Therefore, 5 to 7 of the 14 analysis units are more likely than not to become functionally extinct by 2100, and the other half of the analysis units are declining. The species will likely maintain spatial spread across much of the current distribution, which will continue to offer some protection from catastrophic events for the species. However, increasing probability of extinction across analysis units and contraction of the range means that the species would be less likely to withstand catastrophic events under either future scenario in approximately 75 years.

Southwestern Pond Turtle

Based on projections of probability of extinction, loss of all 5 analysis units in the U.S. is greater than 50 percent under both scenarios by 2075. Therefore, all U.S. analysis units are more likely than not to become functionally extinct in approximately 50 years. There is a possibility that the species could maintain some of its current distribution in those waterbodies most resistant to anthropogenic impacts, bullfrog predation, and/or drought, which would continue to offer some low-level of redundancy for the species. Increasing probability of extinction across analysis units and contraction of the range mean that the species would be less likely to withstand catastrophic events under either future scenario in approximately 50 years.

By 2100, all California analysis units are substantially likely (greater than 70 percent) to be functionally extinct under both scenarios. Given the increasing probability of extinction predicted across analysis units and contraction of the range, the species would be much less likely to withstand catastrophic events under either future scenario in approximately 75 years.

10.5 Future Species Representation

To assess the future representation for the western pond turtle, we considered projected ecological and genetic diversity throughout the range of the species.

Northwestern Pond Turtle

Representation of northwestern pond turtles would be reduced with extirpation of any analysis units. As stated above, based on probability of extinction, all of the analysis units are likely to be extant in 2075. However, even without overall analysis unit extirpation, loss of individuals over time leads to an overall decline in species genetic diversity due to increased probability of inbreeding, genetic drift, and the potential fixation of deleterious alleles, which decreases genetic diversity and adaptive potential (Palstra and Ruzzante 2008, entire). While abundance is projected to decrease leading to likely reductions in genetic diversity, no complete analysis unit extirpations are projected. Therefore, the species likely maintains most of its ability to adapt over the next approximately 50 years.

By 2100, 5 to 7 of the 14 analysis units are more likely than not to become functionally extinct. Analysis units that are not extinct will continue to offer some representation, depending on which of the diverse set of permanent and ephemeral aquatic water bodies used continue to support the species. Besides analysis units in Washington, the southern-most northwestern pond turtle analysis unit (San Joaquin Valley, AU-14) has the highest probability of extinction. Given that these turtles are at the lowest latitude and experience some of the highest temperatures across the range, loss of these individuals may result in loss of adaptive capacity for increasing temperatures with climate change. In addition, the more southern analysis units have greater genetic diversity than in the more northern parts of the range (Shaffer and Scott 2022, p. 8) and therefore the potential loss of the San Joaquin Valley (AU-14) in the next approximately 75 years would be a significant loss of overall species genetic diversity. Therefore, under both future scenarios representation in northwestern pond turtles is likely to be reduced in approximately 75 years.

Southwestern Pond Turtle

Representation of southwestern pond turtles would be reduced with extirpation of any analysis units. As stated above, based on probability of extinction, all analysis units in the U.S. are more likely than not to become functionally extinct by 2075 and are very likely to be functionally extinct by 2100. Therefore, the species may lose most of its current distribution. Inbreeding depression and loss of genetic diversity would be exacerbated as abundance declines across analysis units with increasing probability of population-level extirpations. Even without overall analysis unit extirpation, loss of individuals over time leads to an overall decline in species genetic diversity due to increased probability of inbreeding, genetic drift, and the potential fixation of deleterious alleles, which decreases genetic diversity and adaptive potential (Palstra and Ruzzante 2008, entire). Therefore, under both future scenarios representation in southwestern pond turtles is likely to be severely reduced in the next approximately 50 and 75 years.

10.6 Comparison to Other Future Projections

In this section, we compare the results from our future scenarios from Gregory and McGowan (2023, entire) with a previous PVA that encompassed western pond turtles overall (Manzo et al. 2021, entire).

Manzo et al. (2021, entire) modeled population dynamics of both northwestern and southwestern pond turtles together in a single PVA model, using the population at Goose Lake in Kern County, California (from Germano 2016, entire) as a baseline population that was healthy under

normal non-drought conditions, with initial population size based on site. As with the model presented in this SSA report, Manzo et al. (2021, pp. 489–492) drew many of the life history parameters from Germano (2016, entire) and Holland (1994, entire). However, Manzo et al. (2021, entire) considered parameters from Germano and Holland independently, whereas Gregory and McGowan (2023, entire) used parameters from each, with distributions that allowed for uncertainty. In Manzo et al. (2021, pp. 493–494), western pond turtle probability of extinction was 0.00 using life history parameters from Germano, while probability of extinction was 1.00 using parameters from Holland. Sensitivity analyses in the PVA showed that the difference in first-year mortality, which was a distinction between the parameters used from Germano and Holland, was important in determining population viability over time (Manzo et al. 2021, p. 494). In contrast, the adult mortality rates did not change model outcomes. However, sensitivity analyses of the data within the PVA showed that the probability of extinction increased when adult mortality was over 35 percent per year, and was close to 1.00 at adult mortality of about 50 percent (Manzo et al. 2021, p. 494). Population viability also varied based on drought frequency, with the PVA incorporating long-term drought as a catastrophic event. Starting with the Germano baseline parameters that predicted viable populations, the PVA found that probability of extinction was 15 percent using the baseline drought frequency (3 catastrophic droughts over a 100-year period). Probability of extinction increased when drought frequency was increased in the model, reaching 1.00 when drought frequency was 14 percent (14 catastrophic droughts over a 100-year period). In comparison, Gregory and McGowan (2023, pp. 21–22) used baseline drought frequencies that varied by analysis units and found that their model for southwestern pond turtles was most sensitive to increases in drought.

10.7 Future Condition Uncertainty

The two future scenarios used in this report are very similar, thus potentially underrepresenting the range of potential extirpation probabilities for the species. In some cases the probability of extinction for analysis units is largely overlapping under RCP 4.5 and RCP 8.5.

By using anthropogenic impacts as an umbrella to capture many of the threats influencing western pond turtles, we are likely underestimating future impacts of threats to the species. Similarly, we pulled out drought as the most impactful component of climate change moving forward, but other climatic changes (e.g., wildfires and extreme flooding events) are also expected to negatively impact the species. Therefore, we may be providing an overly optimistic view of future probability of persistence for each species.

Gregory and McGowan (2023, entire) modeled impacts of drought across the range of the two species assuming that drought impacts would be equivalent across habitat types. However, water depth, waterbody type, and land use can lead to notable differences in how areas respond to drydowns. In some cases, synergistic impacts between drought and other threats (e.g., altered hydrology, wildfires) may lead to more severe consequences at local levels than that predicted by

their models. On the other hand, some areas with deep perennial waterbodies may be less impacted by drought. For example, several wastewater treatment plants in California's Central Valley have been reported to have persistent western pond turtle populations (Germano 2010, entire), and it is less likely that these types of habitats would have substantial mortality in the face of drought in comparison to an intermittent stream. By assuming that drought acts in the same way across an analysis unit, we disregard perennial/deep waterbodies that would persist and could continue to support the species. Assessing water permanency at the scale relevant to western pond turtles may require groundtruthing or site visits rather than coarse spatial analyses. For example, in a habitat suitability assessment for southwestern pond turtles conducted by USGS, numerical scores for habitat conditions were assigned for categories including water depth (most points for >3 m), and aquatic refugia; water permanency was identified as one of the habitat characteristics that was used to categorize sites (Baumberger et al. 2022, p. 37).

Future bullfrog expansion may vary throughout the range of the species differently than we predicted in this SSA. For example, the model predicted only modest differences in bullfrog expansion throughout California, whereas environmental niche modeling predicted that areas with the highest predicted risk of future bullfrog expansion in California include the southern Central Valley, the coast of northern California, and mid elevations in the mountains of northern California (Nelson and Piovia-Scott 2022, pp. 1767, 1777). However, this same environmental niche modeling suggested that bullfrogs may be approaching environmental equilibrium in California (Nelson and Piovia-Scott 2022, p. 1789).

Gregory and McGowan (2023, entire) used land conversion data to simulate impacts from anthropogenic influences in the future, specifically looking at changes in wetland habitat as a proxy for western pond turtle aquatic habitat, whereas they modeled initial abundance using available habitat based on riparian areas. Differences in the way that riparian areas versus wetland areas were characterized across the landscape would lead to different amounts of available habitat considered in current and future conditions, but the uncertainty that this could bring into the model was reduced by focusing on rate of change rather than acreage itself. We also acknowledge that concentrating on changes to wetland habitat ignores future losses in upland habitat for western pond turtles; losses in upland habitat could reduce nesting and dispersal, and limit connectivity between populations. The model also used land conversion data as a ceiling that reduced carrying capacity, but because of rapid population declines (after a time lag consistent with that found in Pramuk et al. (2013, pp. 28–29), wherein the number of adult females continues to grow as hatchlings and juveniles transition despite overall population declines) in all analysis units in both scenarios, reductions in carrying capacity had little impact on model results. For this reason, impacts from land conversion were likely underrepresented.

Because we focused on probability of extinction as the metric for analysis unit resiliency, decreases in connectivity were not assessed quantitatively in relation to resiliency. Increasing probability of extinction went hand in hand with decreasing abundances, which would also be

linked to decreases in connectivity, which is important for both resiliency and representation. Impacts to demographic conditions were likely underrepresented because of this.

11.0 Overall Assessment of Resiliency, Redundancy, and Representation

Northwestern Pond Turtle

Washington State analysis units are currently conservation reliant, depending on headstarting and management. In 1990, populations were almost extirpated in Puget Sound and down to 2 populations in the Columbia Gorge totaling only 150 individuals, prompting the initiation of the headstarting program, which improved resiliency in the currently 6 extant populations; although as noted, these analysis units still rely on management. A WDFW PVA suggests that despite projected declines, these do not happen until later in the century, and in 2050 northwestern pond turtles in Washington are expected to persist in reduced numbers.

In California, Nevada, and Oregon northwestern pond turtle analysis units, population growth rate and abundance is declining, but probability of extinction in 2050 is low. Under both scenarios, probability of extinction at 2050 ranged from approximately 5 to 15 percent across analysis units. These estimates, as well as persistence in the Washington PVA, suggest that the northwestern pond turtle is likely to withstand stochastic events in the near term.

In terms of current redundancy, northwestern pond turtles are currently distributed across 4 states, with the majority of turtles in California and Oregon. This spatial spread protects the species from many catastrophic events. While Washington requires management intervention and the southern parts of the range have higher probability of extinction than those in northern California and Oregon, the species maintains its ability to withstand most catastrophic events, particularly in the center of the range.

For current representation, the species exhibits ecological flexibility in habitat use, particularly different types of waterbodies. Severe reductions in Washington, combined with reduced genetic diversity, restrict representation in the northern-most part of the range. Northwestern pond turtles in Oregon and northern California had the lowest genetic variation of the northwestern pond turtle genetic clusters in a genetic study despite covering a larger geographic area than the San Joaquin Valley (AU-14) analysis unit, which had the greatest genetic diversity. However, overall we do not expect severe population declines or extirpations in the near-term (85 to 95 percent probability of persistence across California, Nevada, and Oregon analysis units, and persistence in Washington PVA at 2050). Therefore, the species likely maintains its ability to adapt to changing environmental conditions in the near-term.

In the future, due to impacts from land conversion, bullfrog predation, and increasing drought, chance of extinction rises in all northwestern pond turtle analysis units. Washington analysis units decline rapidly towards extinction in the absence of headstarting. In Oregon, California,

and Nevada analysis units, probability of extinction in 2075 ranges from 28 percent to 41 percent under the two scenarios, indicating that persistence is more likely than extinction in all analysis units (i.e., likelihood of extinction is less than 50 percent) for the next approximately 50 years, and thus ability to withstand stochastic events in this part of the range is largely maintained. By 2100, probability of extinction ranges from 46 to 60 percent under both scenarios. Resiliency of Oregon, California, and Nevada analysis units generally follows a latitudinal trend, with more southern analysis units having higher likelihood of extinction and therefore lower ability to withstand stochastic events in the next approximately 75 years.

Redundancy of northwestern pond turtles would be reduced with extirpation of any analysis units. Washington analysis units rely on headstarting to be maintained in the wild. Based on extinction projections, none of the California, Nevada, or Oregon analysis units (3 through 14) are more likely than not to be lost by 2075. Therefore, while Washington requires management intervention and the southern parts of the range have higher probability of extinction, the species likely maintains most of its ability to withstand most catastrophic events over the next approximately 50 years. By 2100, loss of five of the analysis units is greater than 50 percent under both future scenarios. This suggests that those five units are more likely than not to become functionally extinct by 2100 in the next approximately 75 years. The species will thus likely maintain some of the spread of its current distribution, which will continue to offer some redundancy for the species, but is likely to suffer losses. Overall, given increasing probability of extinction across analysis units with potential extirpations in several analysis units, the species would be less likely to withstand catastrophic events under either future scenario in 2100.

Representation of northwestern pond turtles would be also reduced with extirpation of any analysis units. In 2075, while abundance is projected to decrease across all analysis units leading to likely reductions in genetic diversity, no analysis unit extirpations are projected. Therefore, the species likely maintains most of its ability to adapt over the next approximately 50 years. By 2100, 5 to 7 of the 14 analysis units are more likely than not to become functionally extinct. Besides analysis units in Washington, the southern-most northwestern pond turtle analysis unit (San Joaquin Valley, AU-14) has the highest probability of extinction. Given that these turtles are at the lowest latitude and experience some of the highest temperatures across the range, loss of these individuals may result in loss of adaptive capacity for increasing temperatures with climate change. In addition, the more southern analysis units have greater genetic diversity than in the more northern parts of the range, therefore the loss of the San Joaquin Valley unit (AU-14) would be a significant loss of overall species genetic diversity. Loss of individuals over time leads to an overall decline in species genetic diversity. Therefore, by 2100 representation in northwestern pond turtles is likely to be reduced, making the species less able to adapt to changing environmental conditions.

Overall, in the future, the northwestern pond turtle has an increasing risk of extinction over time from stochasticity, catastrophic events, and inability to adapt to changing conditions. In 2075,

abundances will decrease over time, but persistence of analysis units is more likely than not. By 2100, abundances decrease further and 5 to 7 of the 14 analysis units are more likely to become functionally extinct than to persist, indicating increasing species-level extinction risk in the next approximately 75 years.

Southwestern Pond Turtle

Currently, population growth rate in southwestern pond turtles is declining, and probability of extinction by 2050 ranges from 20 to 24 percent across the U.S. analysis units. The probability of persistence values of 75 to 80 percent suggest that U.S. analysis units are resilient to withstand stochastic events in the near term.

In terms of current redundancy, southwestern pond turtles are currently distributed across the central coast, southern California and Baja. All U.S. analysis units are more likely than not to persist in 2050, indicating that the species maintains some ability to withstand most catastrophic events.

As for current representation, particularly high inbreeding levels in the Mojave (AU-3) and Baja (AU-6) suggest that the species may have limited ability to adapt to changing conditions. However, the species exhibits ecological flexibility in habitat use and we do not expect extirpations in 2050 (i.e., 75 to 80 percent probability of persistence across analysis units). Therefore, the species likely maintains its ability to adapt to changing environmental conditions in the near-term.

In the future, due to impacts from future land conversion, bullfrog predation, and especially increasing drought, southwestern pond turtle analysis unit resiliency is likely to be highly reduced. Chance of extinction rises in all analysis units in the U.S. In 2075, probability of extinction across analysis units is over 50 percent, indicating that extirpations range-wide are more likely than not. In 2100, probabilities of extinction rise to over 70 percent across U.S. analysis units. There are not sufficient data for Baja, Mexico to estimate probability of extinction. Overall, model results demonstrate that all analysis units are more likely than not to go extinct, indicating substantially reduced resiliency and that the species would be significantly less able to withstand stochastic events in the future.

Redundancy of southwestern pond turtles would be reduced with extirpation of any analysis units. Based on projections of likelihood of extinction, all U.S. analysis units are more likely than not to become functionally extinct by 2075, with an increasing chance of becoming functionally extinct by 2100 (i.e., probability of extinction greater than 70 percent across U.S. analysis units). Given reduced abundances and potential extirpations, the species would be much less likely to withstand catastrophic events under either future scenario.

Representation of southwestern pond turtles would also be reduced with extirpation of any analysis units. As stated above, based on probability of extinction, all analysis units in the U.S. are more likely than not to become functionally extinct by 2075 and are very likely to be functionally extinct by 2100. Therefore, the species may lose most of its individuals and current distribution. Under both future scenarios, representation in southwestern pond turtles is likely to be substantially reduced, making the species less able to adapt to changing environmental conditions.

Overall, in the future at both 2075 and 2100, the southwestern pond turtle will have substantial increasing risk of extinction from stochasticity, catastrophic events, and inability to adapt to changing conditions.

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APPENDIX A

Modeling appendix for the Northwestern and Southwestern pond turtle (*Actinemys marmorata*, *Actinemys pallida*)

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Summary

To predict future status of the northwestern pond turtle (*Actinemys marmorata*) and southwestern pond turtle (*Actinemys pallida*) species, we developed a stochastic stage-based matrix population model to simulate future population conditions. We constructed a demographic population viability analysis for each species based on a post-breeding, single sex, stage-based life history diagram elicited from taxa experts and derived from relevant literature. Demographic parameters were based on estimates from published literature and data provided to the U.S. Fish and Wildlife Service (USFWS). Using the most recent observations of turtles, available habitat, local abundances, and current threat conditions, we calculated spatially explicit initial abundances to initialize our stochastic projection. In order to incorporate multiple types of uncertainty (ecological, parametric, temporal), we built three embedded simulation loops within the simulation model. Representing ecological uncertainty, species status was projected into the future using multiple plausible future scenarios based on two representative concentration pathways (RCP 4.5, 8.5) and two shared socioeconomic pathways (SSP 2, 5) to reflect plausible alternative future trajectories of relevant environmental conditions. Parametric uncertainty was included for survival estimates of all life stages due the inconsistency of estimates across the species' range. Temporal variability or environmental stochasticity was included in the form of randomized variation from the mean demographic parameter values in each year of the approximately 80-year simulation.

The model output included probability of extinction and estimated abundance through 2100 for each unique Analysis Unit (AU) and for the full geographic range of the species except populations in the state of Washington. The AUs in Washington are conservation dependent and sustained by a head-starting and reintroduction program. Thus, the population dynamics do not match our model for the rest of the range and therefore the Washington AUs were included in this projection modeling effort. There is already pre-existing, detailed PVA for these specific populations (Pramuk et al. 2012, p.41-60), and the Status assessment report can use those results for inference about future status. We discuss the results of Pramuk et al. (2012, p.41-61) alongside our own. Probability of extinction was overall higher for the southwestern pond turtle as compared to the northwestern species and population growth rates were strongly negative for both species (approximately -3% annually for all AUs for all scenarios). This appendix is organized into three primary sections: 1) a description of the life history, the core population dynamics model, and demographic parameters, 2) a description of methods for establishing initial abundances of the populations for the future viability modeling, and 3) a description of the methods for modeling effects of various threats on future demographic rates and the results of future conditions scenarios.

Model Description

Structure & Demographic Rates

A stage-based life history diagram was elicited from taxa experts and a review of northwestern and southwestern pond turtle literature (hereafter, WPT, turtle, and pond turtle are used interchangeably when referring to both species). Life history was described and modeled in life stages, versus age classes, because of the lack of information on age-specific demographic rates and effects of threats to the species. Consistent with published literature (Bury et al. 2010 entire; Bury et al. 2012 Chapter 2), we used a stage-based model to represent the diversity of threats and demographic rates for WPT over their lifetimes. Experts described three stage classes: hatchling (H), juvenile (J), and adult (A) in a single sex (female-only), post-breeding census (Figure 1). Hatchlings are defined as small individuals ages 0-1 years. Juveniles are non-reproducing individuals with a carapace length <120 mm, aged approximately 1-7 years old. Adult turtles are characterized by a greater size (>120 mm) and sexual maturity (Bury et al. 2010 p.445).

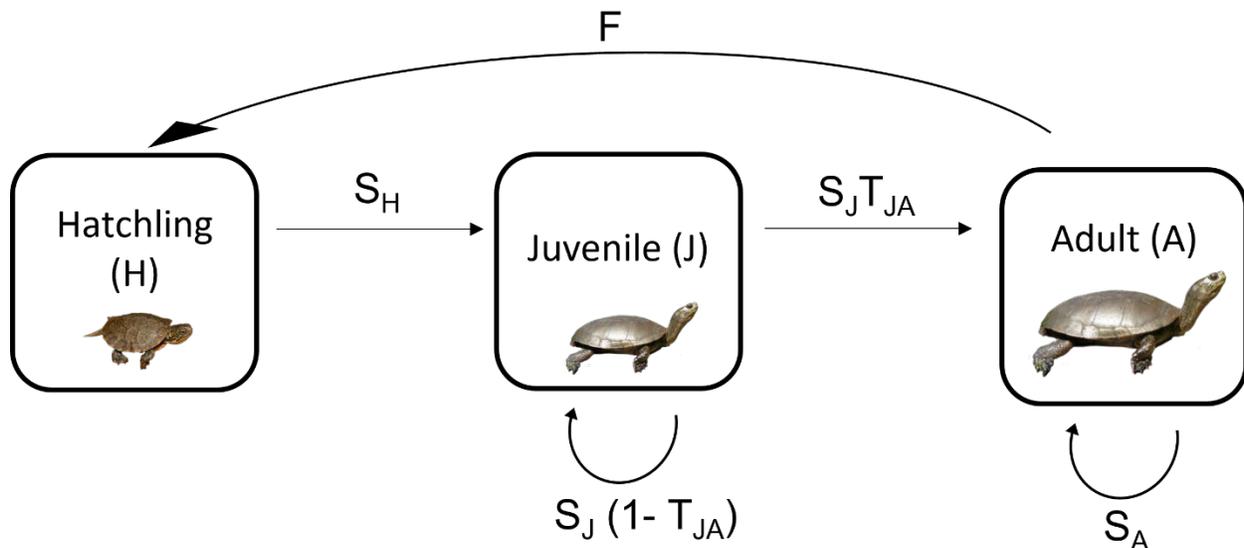


Figure 1. Stage-based life history diagram for both WPT species, representing three life history stages: hatchling (H), juvenile (J), and adult (A). Arrows represent moving from one stage to another, or remaining in a stage, by the next time step ($t+1$). Definitions and estimates of each parameter are described in Tables 1 and 2.

We developed a population projection matrix (Eqn.1) based on the life history diagram (Caswell 2006 entire). For each time step in the model, parameter values were drawn from randomized statistical distributions to incorporate parametric uncertainty and mimic environmental variability over time, as described below. At each time step a population vector was multiplied by the randomized demographic matrix to predict the abundance of each life stage in the next time step (Eqn. 2). Additionally, we used the R package ‘popbio’ (Stubben and Milligan 2007, entire) to perform an elasticity analysis to determine the magnitude of effect of each life history parameter on population growth rate (λ). With this process at its core, a triple loop stochastic simulation model was used to predict the species abundance in the year 2100 (inner loop, Eqn.2), with 1,000 replicates for each (middle loop), for each unique Analysis Unit (hereafter, AU; middle loop). There are 14 AUs for *A. marmorata* and 6 for *A. pallida*. The triple loop structure allows for incorporation of various future plausible scenarios, spatially

explicit population parameters, and uncertainty in demographic rates, initial abundances, current conditions, and effects of threats to the species (McGowan et al. 2011 entire; McGowan et al. 2017 entire).

$$\mathbf{Eqn\ 1.}\ A = \begin{bmatrix} 0 & 0 & F \\ S_H & S_J(1-T_J) & 0 \\ 0 & S_J T_J & S_A \end{bmatrix}$$

$$\mathbf{Eqn\ 2.}\ \begin{bmatrix} N_{t+1}^H \\ N_{t+1}^J \\ N_{t+1}^A \end{bmatrix} = A * \begin{bmatrix} N_t^H \\ N_t^J \\ N_t^A \end{bmatrix}$$

Estimates of demographic parameters were sourced from the best available information on WPT, in the form of published literature, and data provided to the USFWS by researchers and state and federal agencies (Table 1, Table 2). Appropriate statistical distributions were selected for each demographic parameter based on possible values of each parameter and the patterns of values reported in the literature (see below). For some parameters, there was no species-specific information, so we assumed the distributions to be the same for both species. For each time step and replicate in the model, a value was drawn randomly drawn from these distributions to account for uncertainty and mimic annual variability.

We modeled survival for all age classes using the same distribution for both species (Table 1). Holland, (1994 p.2-11) estimates hatchling survival to be 0.1 – 0.15, and in the absence of likelihood of values within that range, we modeled hatchling survival using a uniform distribution (Figure 2). We also used a uniform distribution to represent juvenile survival (Figure 3), with upper and lower limits drawn from Germano (2016 p.667-668). Uniform distributions assume that all values between the upper and lower bound are equally likely for be drawn and are appropriate in the case because we do not have errors stand errors or other distribution descriptors to specify a probability distribution. Estimates of adult survival are dominated by values close to 1.0 (Holland 1994 p.2-11; Manzo et al. 2021 p.41). However, Germano (2016 p.670) estimated survival to be as low as 0.731. We used a four-parameter beta distribution to represent the skewedness of adult survival data such that lower values around the Germano (2016 p.670) estimate were possible, but less likely (Figure 4). Overall, estimates of WPT survival in the literature are inconsistent and the model output was highly sensitive to these inputs, adult survival specifically. Thus, we included parametric uncertainty for these parameters (McGowan et al. 2011 entire). The values drawn from the ranges in Table 1 were used as mean values, with a coefficient of variation of 0.1 for hatchling and juvenile survival, and 0.01 for adult survival, to estimate alpha and beta parameters. A lower coefficient of variation was used for adult survival because mean values were close to 1.0, and any higher coefficient of variation can result in an estimate over 1.0, violating the requirements of a Beta distribution and prohibiting proper function of the model. The generated alpha and beta parameters from the parametric uncertainty portion of the simulation were used in a Beta distribution within the innermost loop of the simulation to generate values of survival for each age class in each year. Juveniles mature to the adult life stage after approximately 7 years, so we defined the probability of transitioning from juvenile to adult as 1/6 such that by after 6 years in the juvenile stage, juveniles will on-average have transitioned to the adult stage. This is similar to a method used in (Sweka et al. 2007 p.278-280; McGowan et al. 2011 p.128; McGowan et al. 2017 p.121) and others in demographic population viability models where the inter-stage transitions rates were not estimated from empirical data. We acknowledge that Kendall et al. (2019 p.36) suggested an

alternative method for calculating transition probabilities, but our populations, given threats from non-native predators, likely do not meet the stable age and asymptotic growth assumption of that approach so we used the simpler method with fewer assumptions and tested model sensitivity to the parameter.

Table 1. Description of stage-based survival and transition probability distributions and relevant sources for both WPT species.

Parameter	Species	Mean value or range of values	Figure #	Source(s)
Hatchling survival (S_H)	<i>A. marmorata</i> , <i>A. pallida</i>	0.1 - 0.15	2	(Holland 1994 p.211)
Juvenile survival (S_J)	<i>A. marmorata</i> , <i>A. pallida</i>	0.731 - 0.838	3	(Germano 2016 pp.667-668)
Adult survival (S_A)	<i>A. marmorata</i> , <i>A. pallida</i>	0.731 - 0.99	4	(Germano 2016 p.670; Manzo et al. 2021 p.491; Holland 1994 p.2-11)
Juvenile to adult transition probability (T_{JA})	<i>A. marmorata</i> , <i>A. pallida</i>	1/6	N/A	Standard method, see McGowan et al 2017 entire

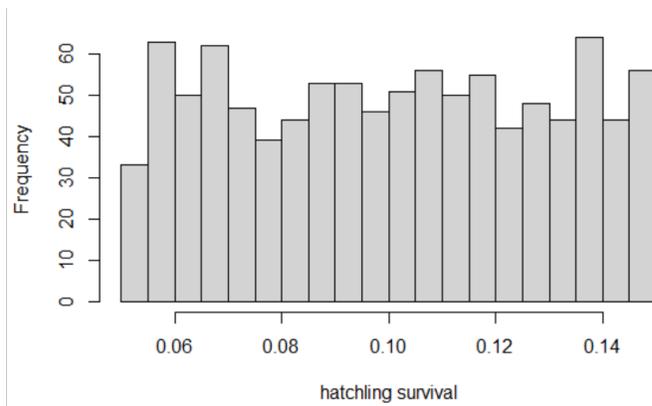


Figure 2. Histogram of 1,000 simulated samples of mean hatchling survival rate for both WPT species drawn from a Uniform distribution used to incorporate parametric uncertainty into the simulation model.

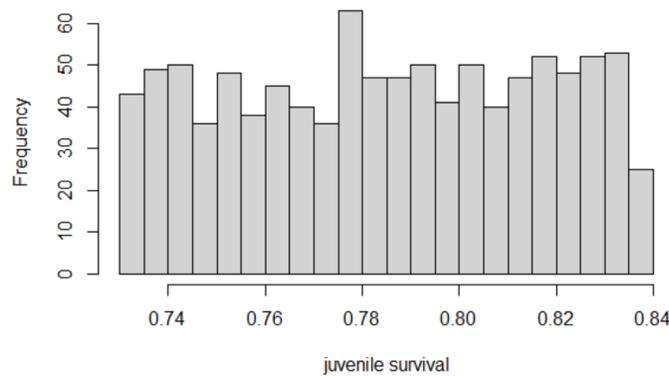


Figure 3. Histogram of 1,000 simulated samples of mean juvenile survival for both WPT species drawn from a Uniform distribution used to incorporate parametric uncertainty into the simulation model.

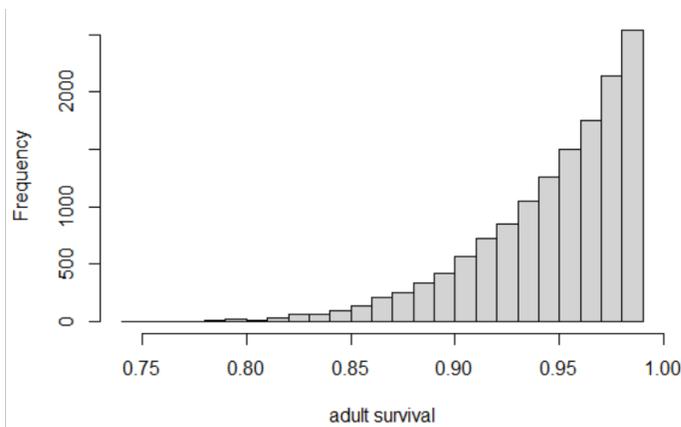


Figure 4. Histogram of 1,000 simulated samples of mean adult survival for both WPT species drawn from a four-parameter Beta distribution used to incorporate parametric uncertainty into the simulation model.

In WPT life history, fertility (F) represents the number of female offspring that one adult female contributes to the hatchling population in a single year. We defined fertility as the product of multiple components of recruitment and reproduction (Eqn.3). Fecundity (Fec), the number of eggs laid per female, was higher for the northwestern species as compared to the southwestern (Lovich and Meyer 2002 p.540; Germano 2016 p.668). The probability of double clutching, however, was higher for the southwestern species (Table 2). Hatching success was the probability that eggs hatched successfully and was parameterized equally for both species. Nest survival, the probability that a nest was not predated or destroyed in any way during the egg incubation process, was highly uncertain in the literature, with estimates ranging from 0% survival (entire nest or all eggs destroyed) to 90% survival (Table 2). Nest survival was modeled as a Uniform distribution to represent the wide uncertainty in reported rates. Because this is a female-only model, total fertility was then multiplied by a hatchling sex ratio (HSR) value drawn from a Uniform distribution between 0.329 and 0.690 (Table 2) to reflect uncertainty because this is a female-based model. Lastly, because this is a post-breeding census (Kendall et al. 2019 p.34), we included adult survival in the fertility term by multiplying everything by an adult survival value drawn from the distribution described in Figure 4.

$$\text{Eqn. 3 } F = (Fec * NS * HS * PB + p(\text{double clutch}) * Fec * NS * HS * PB) * HSR * S_A$$

Table 2. Description of distributions of parameters used to calculate fertility for both WPT species.

Parameter	Species	Mean value or range of values	Figure #	Source(s)
Fecundity (Fec)	<i>A. marmorata</i>	6.24	5	(Germano and Rathbun 2008 pp.190,192; Germano 2016 p.668; Germano et al. 2022 p.113)
Fecundity (Fec)	<i>A. pallida</i>	4	6	(Lovich and Meyer 2002 p.540)
Hatching success (HS)	<i>A. marmorata</i> , <i>A. pallida</i>	0.72–1.0	7	(Holland 1994 pp.2-10; Rosenberg 2013 pp.115-118)
Nest survival (NS)	<i>A. marmorata</i> , <i>A. pallida</i>	0.09–0.78	8	(Holte 1988 p.48; Holland 1994 pp.2-12; Rosenberg 2013 pp.115-118)

Proportion breeding (PB)	<i>A. marmorata</i>	0–0.9	9	(Ashton et al. 2015 p.626; Germano 2016 pp.668-669)
Proportion breeding (PB)	<i>A. pallida</i>	0.125–0.53	10	(Lovich and Meyer 2002 p.540; Belli 2015 p.45)
Probability of double clutching	<i>A. marmorata</i>	0.05–0.1	11	(Germano and Rathbun 2008 p.192; Germano 2016 p.668)
Probability of double clutching	<i>A. pallida</i>	0.04–0.5	12	(Goodman 1997 entire; Lovich and Meyer 2002 pp.540-541; Scott et al. 2008 pp.144-145; Belli 2015 pp.45,70)
Hatchling sex ratio (HSR)	<i>A. marmorata</i> , <i>A. pallida</i>	0.329–0.69	13	(Gordon 2009 p.33; Dallara 2011 p.25; Christie and Geist 2017 p.50; Nicholson et al. 2020 p.9)

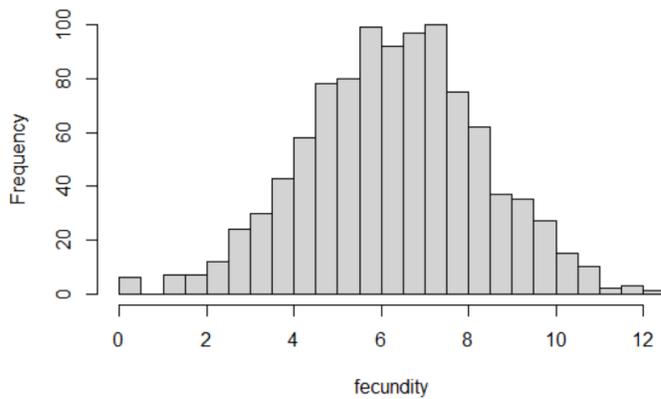


Figure 5. Histogram of 1,000 simulated samples of fecundity (Fec) for the northwestern species, *A. marmorata*.

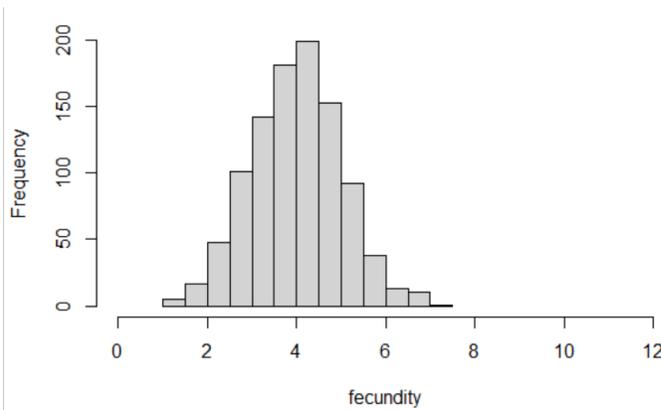


Figure 6. Histogram of 1,000 simulated samples of fecundity (Fec) for the southwestern species, *A. pallida*.

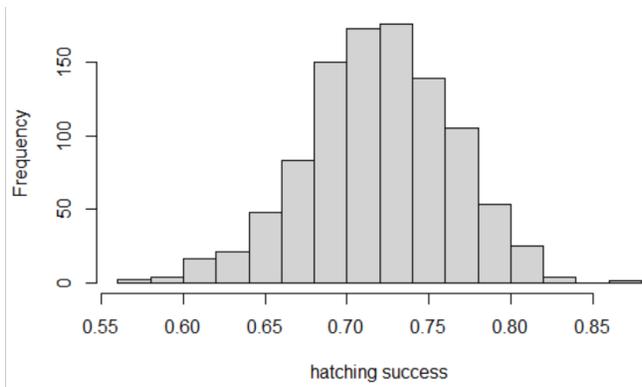


Figure 7. Histogram of 1,000 simulated samples of hatching success (HS) for both WPT species.

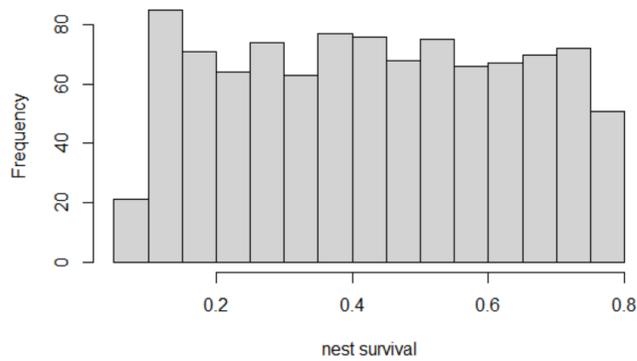


Figure 8. Histogram of 1,000 simulated samples of nest survival (NS) for both WPT species.

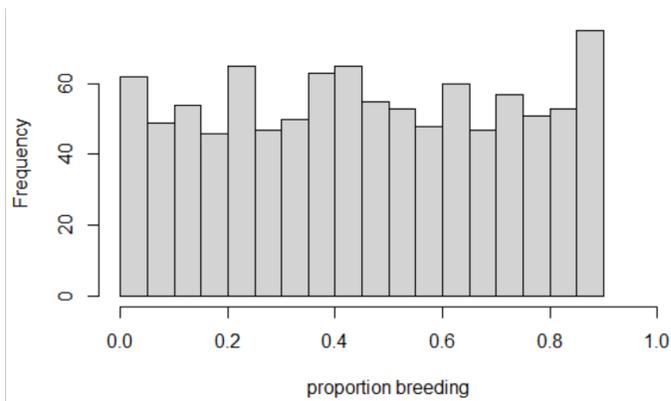


Figure 9. Histogram of 1,000 simulated samples of proportion of individuals breeding (PB) for the northwestern species, *A. marmorata*.

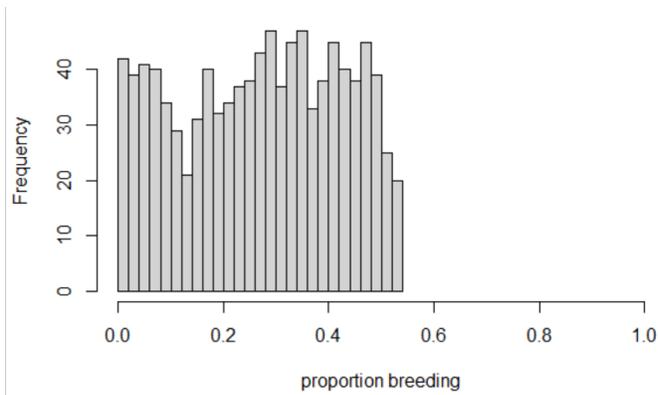


Figure 10. Histogram of 1,000 simulated samples of proportion of individuals breeding (PB) for the southwestern species, *A. pallida*.

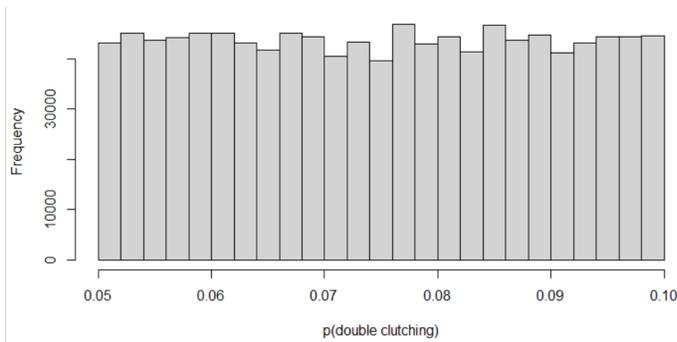


Figure 11. Histogram of 1,000 simulated samples of probability of double clutching for the northwestern species, *A. marmorata*.

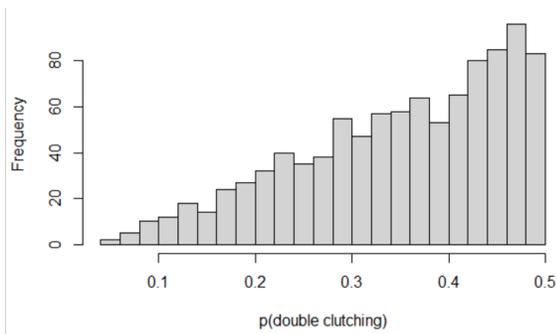


Figure 12. Histogram of 1,000 simulated samples of probability of double clutching for the southwestern species, *A. pallida*. Note that the x-axis scale is different than that in Figure 11.

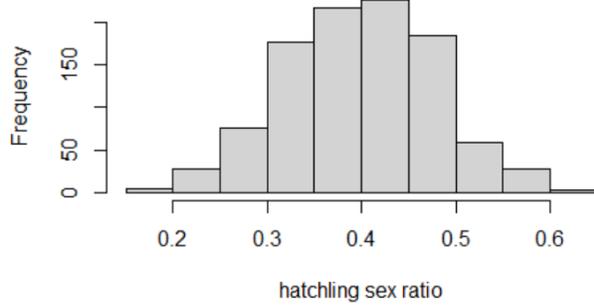


Figure 13. Histogram of 1,000 simulated samples of hatching sex ratio (HSR) for both WPT species.

Determining Initial Abundance

Habitat-based abundance

To estimate initial population size for each AU, we performed a stepwise process to incorporate best available presence data, available habitat information combined with human landscape modification metrics, and then the effects of recent drought and invasive species (Figure 14). We used the HUC (hydrologic unit code) 12 spatial unit (the hydrological units at the 12-digit level “HUC12”, U.S. Geological Survey and U.S. Department of Agriculture, Natural Resources Conservation Service 2013) as the base spatial scale for estimating initial population size because experts believe habitat and demographic processes happen at this small spatial scale. Once abundance was estimated for each HUC12 unit, we summed all the units within an AU to calculate initial population size for each AU.

First, we calculated a probability of current occupancy for each HUC12 using historic observational data of the WPT. The historic observational data set was composed of a variety of data sets provided to us and the USFWS by species experts and State and Federal agencies, among others (Supplemental Table 1). We determined the number of years since the most recent observation in each HUC12, which we used in a Bernoulli trial-based model where the probability of success was calculated by raising adult survival to the power of the number of years since turtles had been observed in the area (Eqn.4). This function would be equivalent to the probability that at least one adult that was observed at some time in the past was still present in 2022 (the starting point of our future simulation model). The adult survival rate was drawn from the previously described distribution (see Structure and Demographic Rates, Figure 4, Table 2). While there are likely issues of detection in the data on which this current occupancy method is based, the data set represents the best available information for the species, and we included many replicates and uncertainty in the adult survival (S_A) estimates to account for this.

$$\text{Eqn. 4 } \textit{occupancy} = \text{Bernoulli}(S_A^{\# \textit{ years since observed}})$$

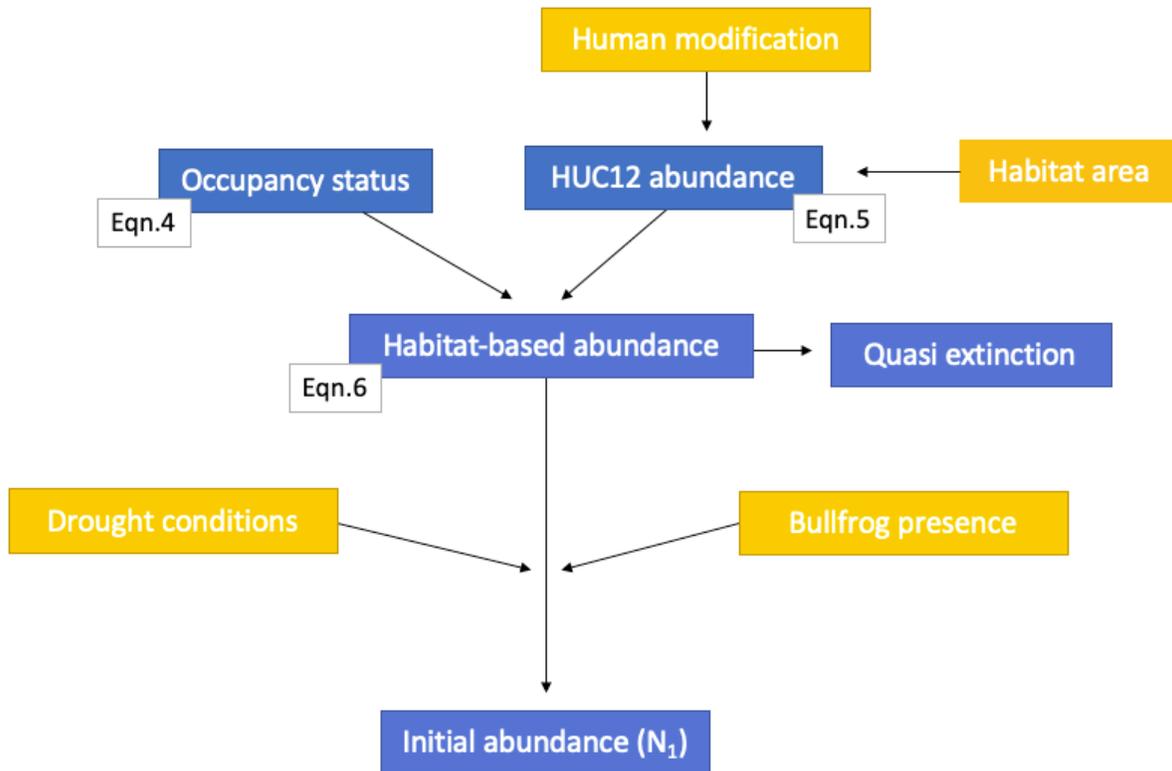


Figure 14. Conceptual diagram of process for calculating initial abundance for each Analysis Unit (AU) for both WPT species.

To determine the abundance of turtles in a given HUC12 on the landscape, we used a negative binomial linear generalized linear regression model with habitat area and human modification as potential explanatory variables. Robust population estimates are largely not available for the WPT, so we transformed unique capture data (Manzo et al. 2021, Supplemental Material) using detection probability estimates from Fulton et al. (2022 p.102). We calculated approximate abundances by dividing capture estimates by a detection probability randomly drawn from a Uniform distribution with a lower bound of 0.09 and an upper bound of 0.17 (Fulton et al. 2022 p.102). Using the location information from Manzo et al. (2021 Supplemental Material), we determined the HUC12 specific habitat area and human modification values for each corresponding abundance estimate. The human modification value (H) combines numerous remotely sensed data sets into a single metric (Theobald et al. 2020 entire), including a number of factors that affect turtle life history, namely roads which pose a female-biased road mortality risk (Nicholson et al. 2020 entire), among other factors in the metric such as urban build up, agricultural development, and human intrusions (Theobald et al. 2020 p.1955). We hypothesized that human modification (H) likely has a negative effect on HUC12 abundance, based on the probability that increasing levels of human modification in the landscape surrounding a water body and riparian habitat leads to fewer suitable nesting sites (Legler 1954, p.142), increased meso-predator populations to consume nests and juvenile turtles (Wang et al. 2015 p.27), increased probability of adult mortality while crossing roads to find nest sites (Gibbs and Shriver 2002, entire), and less available wintering habitat for the turtles (Davis 1998, pp.16-19). We classified habitat area as riparian area, delineated by the National Riparian Areas Base Map from

the U.S. Forest Service (Abood and Wieczorek 2022, entire), which we predicted would have a positive relationship with HUC12 abundance.

The results of the negative binomial regression analysis revealed that both factors are explanatory variables for HUC12 abundance, with abundance increasing as habitat area increases and as human modification decreases, as expected. (Figure 15, Eqn.5). We then used the negative binomial relationship (Eqn. 5, Table 3) to determine the HUC12 abundances for all sites with historic WPT observational data. The habitat area and human modification information was extracted for each of observation and input into Eqn. 5 with relevant uncertainty around the intercept and beta coefficient terms as defined by the regression results (Table 3).

$$\text{Eqn. 5 } HUC12 \text{ abundance} = \text{Intercept} + \beta_1(\text{habitat area}) + \beta_2(H)$$

Table 3. Coefficient mean and standard error values of negative binomial regression for HUC12 abundance (Eqn.5)

Coefficient	Mean	Standard error
Intercept	5.421354	0.489778
β_1	7.0625537×10^{-6}	2.815185×10^{-6}
β_2	-1.385846	0.5424998

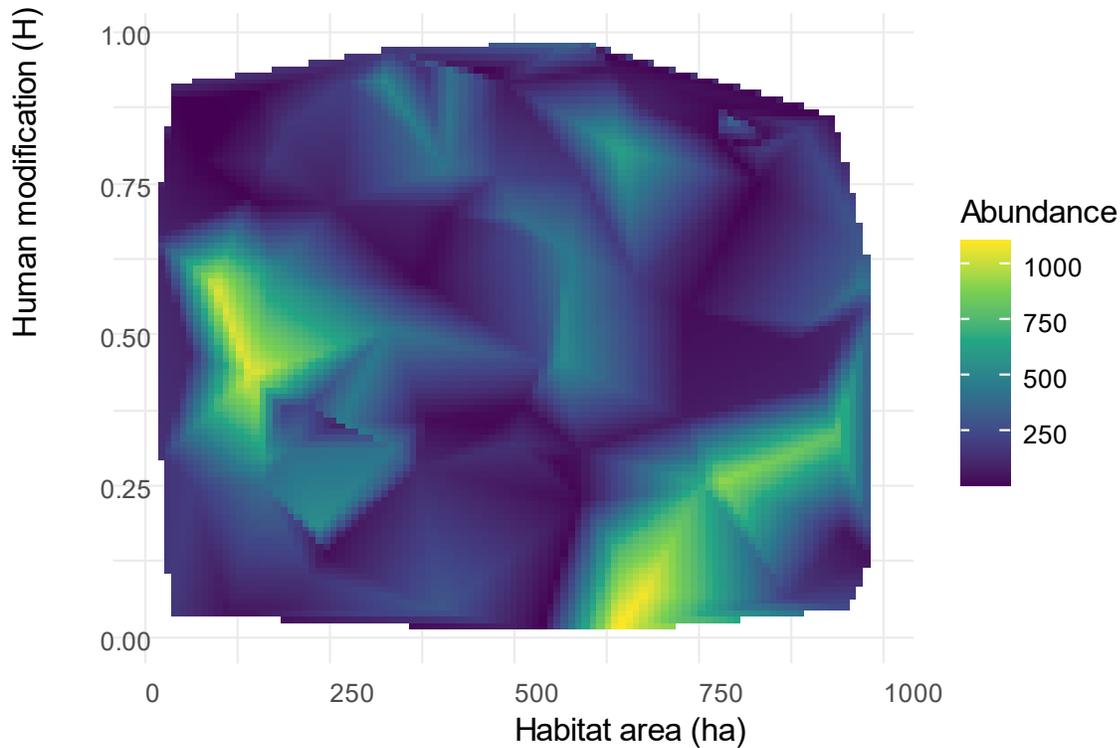


Figure 15. Surface plot of predicted HUC12 abundance values based on habitat area and human modification (H) using the mean value of the regression coefficients (Table 3).

The HUC12 abundance values were then multiplied by the current occupancy status and adult sex ratio (ASR) to determine the habitat-based abundance of each HUC12 (Eqn.6). For each calculation, ASR was sampled from a Normal distribution with a mean of 0.4 (Nicholson et al. 2020 p.9-10) and a 10% coefficient of variation, in the absence of standard deviation information (Figure 16).

Eqn. 6 *habitat based* $N_1 = \text{occupancy} * \text{density} * \text{ASR}$

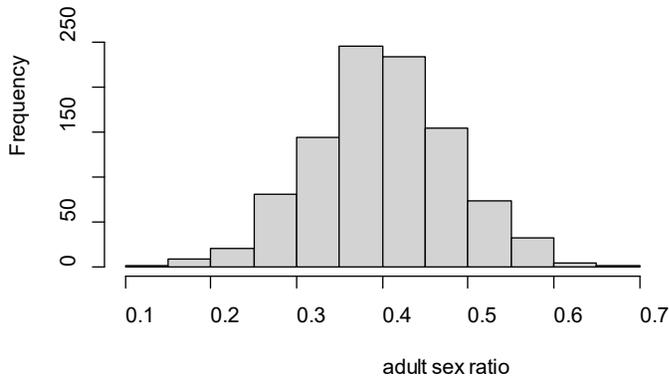


Figure 16. Histogram of 1,000 simulated samples of adult sex ratio (ASR) for both WPT species.

The estimates of initial population size in Eqn.6 are considered ‘habitat-based’ because they represent what the population would theoretically be on the landscape in the absence of current demographic threats (e.g., recent droughts, bullfrogs). Based on this reasoning, we determine quasi-extinction thresholds for each AUs using the habitat-based estimates of abundance to understand the current status of populations as it relates to extinction levels. Quasi-extinction thresholds are the population size below which a population is considered functionally extinct. Some previous species status assessments (SSAs) have used a percentage of the initial population size to set the quasi-extinction threshold and asserted that the threshold represented the probability of catastrophic population collapse (e.g., 5% of the initial population, McGowan et al. 2017 p.122; USFWS 2021 p.ix). However, that approach does not account for the possibility that some populations may already be below the quasi-extinction threshold. To deal with these uncertainties, we disassociated the estimates of the quasi-extinction threshold from the initial abundance by defining the threshold relative to the habitat-based abundance in the absence of environmental stressors (see Eqn. 6, above). We defined quasi-extinction as 5% of the total habitat-based abundance for each AU for use in the simulation scenarios described below. AU-specific estimates of quasi-extinction are also valuable to account for spatially explicit initial conditions and future threats.

Initial abundance

Because our current condition threats have varied impacts on WPT life stages, we divided the habitat-based abundance into the three life stages. In the absence of stage distribution information, stable stage distribution is often used for initial conditions in population viability analysis; however, in the case of WPT, populations are largely dominated by adults, with most estimates of proportions of adults between 0.75-1.0, with some values as low as 0.55 (Holland 1994 p.4-24; Germano and Bury 2001 p.25; Lovich and Meyer 2002 p.5; Spinks et al. 2003 pp.260-261; Germano and Rathbun 2008 p.190; Bury et al. 2010 pp.447-448; Sloan 2012 p.15; Belli 2015 p.41). To capture this skewed distribution, we used a four-parameter Beta distribution with a minimum of 0.55 (Figure 16). For each replicate in the model, a value of proportion of adults is drawn from this distribution, which is then subtracted from 1 to calculate the combined remaining proportion of hatchlings and juveniles. There was no information available on the

relative proportions of hatchlings to juveniles, so we assumed a stable stage distribution for these two stages. We calculated the mean and variance of proportion of hatchlings to proportion of juveniles when all three stages are in a stable stage distribution for 100 replicates of each AU. To account for parametric uncertainty, we drew values of hatchling to juvenile proportions from a Beta distribution with parameters calculated from the ratio mean and variation values (Figure 17). The drawn values were then multiplied by the remaining non-adult proportions and habitat-based abundance to calculate the habitat-based hatchling and juvenile abundances.

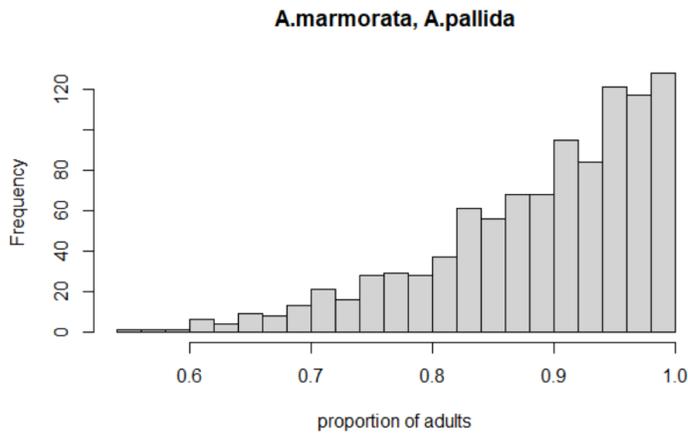


Figure 17. Histogram of 1,000 samples of proportion of adults for the initial stage-based abundance for both WPT species.

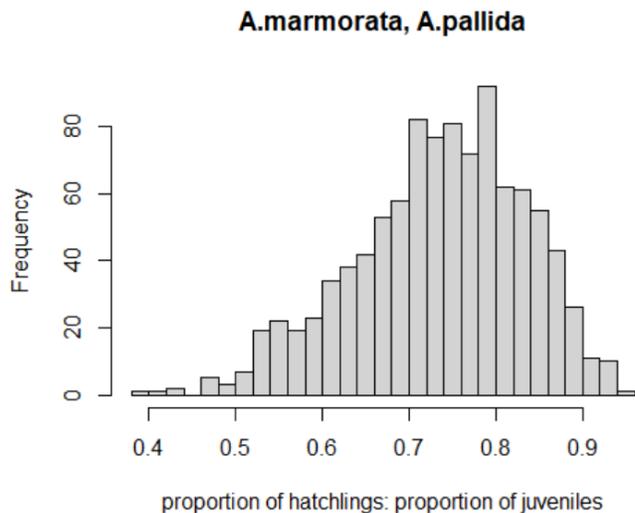


Figure 18. Histogram of 1,000 samples of the proportion of hatchlings as compared to juveniles in a stable stage distribution for the initial stage-based abundance for both WPT species.

To account for recent drought effects on initial abundance, we examined drought data from the previous 5 years for each AU to determine the occurrence of the four drought conditions that negatively affect WPT life history (see ‘Influences on Future Viability’). To estimate recent and historical drought conditions, we used Palmer Drought Severity Index

(PDSI) data from the National Oceanic and Atmospheric Administration (NOAA) Monthly U.S. Climate Divisional Database (Vose et al. 2014). Specifically, we used values from the months of June – August to determine average drought conditions for that year. These months are generally the most arid, and turtles are active and breeding during this time (Bury et al. 2012 entire, St. John 2015 entire) and thus most vulnerable. NOAA records PDSI values for each of their climate divisions, which are areas NOAA has determined to be climatically homogeneous. California, Oregon, and Washington have 7, 8, and 10 unique climate divisions, respectively (Vose et al. 2014 entire). To apply these climate division data to the AU scale, we used a weighted average of PDSI values. The proportion of the area of each climate division within each AU served as the weighting factor on the PDSI value, resulting in an annual value of PDSI for each AU. Because of the lack of information about a continuous relationship between WPT survival and a quantified drought metric, we categorized drought conditions based on the U.S. Drought Monitor categories to enable incorporation of temporal and spatial stochasticity in the drought data itself. We implemented the negative effects of drought based on recent conditions, as a reduction in initial population abundance below the habitat-based estimate. This approach applies the estimated effect of droughts on survival (see ‘Influences on Future Viability’) to the initial abundance, assuming that droughts in recent years will have caused mortality in the population and thus lowered abundance. Effects of drought on probability of breeding (PB) were not translated into abundance effects here because the stochastic uncertainties of drought interacting with breeding probability to affect current abundance are complicated by the lag time between breeding and recruitment to the adult population, so drought effects were not included in the calculations of initial abundance.

Similarly, we implemented the effect of bullfrog presence on initial abundance using the methods described in the Threats section below, but with hatchling and juvenile abundance substituted for their respective survival values. Bullfrog presence was measured in terms of proportion of area in an AU occupied by bullfrogs. Observation data were downloaded from Global Biodiversity Information Facility (GBIF; GBIF.org 2022). HUC12 was the primary unit of analysis, such that if a bullfrog was observed in a HUC12, it was assumed present in the entire HUC12 unit. The area of the occupied HUC12s within an AU was divided by the total AU area to attain the relevant proportion, which was then assumed to represent the proportion of WPT within an AU that are impacted by bullfrogs.

Influences on future viability

Based on literature and discussions with taxa experts, we included three primary potentially negative environmental influences on future population demographics in our model: drought, invasive bullfrogs, and land conversion. While there are many threats to the species enumerated in the SSA, our three selected threats were determined to be highly likely to influence the future viability of the species (Nicholson et al. 2020 entire; Manzo et al. 2021 entire). These same factors influence the estimates of current condition and initial abundance, except that the human modification index was used as a proxy for habitat loss and quality. Additionally, inclusion of these factors in the model was limited by availability of information on future trajectory of the factors and their effects on WPT demography.

Due to the correlated nature of climate change effects, we chose to only model drought as our main climate change factor because it had a direct and previously estimated effect on adult survival, the most influential parameter on model output (Lovich et al. 2017 entire; Purcell et al. 2017 entire). We also modeled the potential effect of invasive bullfrogs on WPT recruitment as

bullfrogs are a growing concern and non-native predation pressure on pond turtles (Nicholson et al. 2020 entire; Manzo et al. 2021 entire). Lastly, we included a habitat loss effect to represent the land conversion threat. Land conversion may include direct mortality effects on turtles from roadways as well as indirect effects such as loss of nesting habitat leading to density dependent effects on nest survival (Steen et al. 2006 entire; Nicholson et al. 2020 entire). However, none of these habitat loss effects on demography are directly estimated in the literature, thus we implemented a simple ceiling-type carrying capacity function to mimic the effect of habitat loss in each region (Morris and Doak 2002 p.311-322; McGowan et al. 2017 p.123).

Despite the recent increased attention to disease in WPT, specifically the respiratory disease that affected Washington populations in the 1990s (Hallock et al. 2017 p.9) and shell disease found in Washington and recently detected in California (Lambert et al. 2021 entire), we did not include disease effects in our model. Disease experts indicated that not enough was known about the disease dynamics to quantify demographic effects on WPT and that disease is probably a secondary or tertiary factor that affects individuals but only rises to the point of a population level effect for already very small populations. Applying a quasi-extinction threshold instead of modeling full extinction is a widely accepted approach to account for these types of small population effects without speculating on the functional form of the demographic effects. Though there were no data available or published literature to quantify the effects of disease on demographics, it is worth noting that disease may represent an additional contributor to extinction risk above and beyond the results of our model. Because all threats to the species cannot be included in the model, estimates of abundance may be overestimated and probability of extinction might be underestimated. In the following paragraphs we describe the methods for modeling the effects of the three primary factors (drought, bullfrogs, and habitat loss) on the likely future status of the species.

Drought

Drought intensity and duration have both been documented to negatively affect WPT survival rates (Holland 1992 pp.65-70; Holland 1994 p.2-14; Lovich et al. 2017 entire; Purcell et al. 2017 entire). Previously we translated the drought effects on survival into an effect on initial abundance, but in this portion of the modeling analysis we are retaining the direct effect on survival. We estimated historical drought intensities in each AU using the PDSI and the intensity classification categories by the U.S. Drought Monitor (Drought Classification | U.S. Drought Monitor 2022). We selected the PDSI as our primary metric of drought because it is an index of factors including temperature, precipitation, and evapotranspiration (Heddinghaus and Sabol 1991 entire; Heim 2002 entire; Dai 2011 entire), which are known to be influential of WPT life history parameters. Purcell et al. (2017 entire) used PDSI to quantify drought; however, for studies reporting negative effects of drought on survival not using the PDSI (Holland 1992 pp.65-70; Lovich et al. 2017 entire), we cross-referenced study-specific time and location information with detailed PDSI data from the NOAA Monthly U.S. Climate Divisional Database. Once we had a consistent metric for drought and its effects, we classified drought intensity in terms of two categories of moderate ($PDSI < -2$ and > -4) and extreme ($PDSI < -4$) intensities.

We incorporated the effects of a variety of combinations of drought intensities and durations into our projection model (Table 4). Studies of drought effects on survival only provide information on adult survival despite the fact that drought likely impacts all life stages, so in the absence of stage-specific information, we assume that the effect of drought on survival of adults

is equal to that for the hatchling and juvenile life stages. In instances of moderate drought lasting at least four years, we reduced survival of all stage classes by 59.8% (Manzo et al. 2021 p.491) and proportion of breeders (PB, Eqn. 3) to anywhere between 0 - 0.2 for the fourth year of the drought, and each year after if moderate drought conditions continued (Holland 1992 p.5-2; Manzo et al. 2021 p.491). The value of proportion of breeders was drawn from a Uniform distribution with a minimum value of 0 and maximum value of 0.2, such that all numbers between those limits are equally probable. Droughts of extreme intensity (or worse, $PDSI \geq 4.0$) have been observed to alter adult survival within the first year, so we include a 16.7% reduction in survival of all stage classes (Purcell et al. 2017 p.21-22). In the second and third consecutive years of extreme drought, survival of all stage classes is reduced by 44.4% (Purcell et al. 2017 p.21-22) and 95.3% (Lovich et al. 2017 p.5-6), respectively. Lovich et al. (2017 entire) did not report survival of turtles in a non-drought year, so we used the non-drought survival value reported by Purcell et al 2017 p.21-22 for the purposes of our calculation. There is no available information on demographic effects of extreme droughts lasting longer than three years, so we assume that survival effects in such instances to be equal to that of the three year extreme drought effect (95.3%, Lovich et al. 2017 p.5-6). In the absence of information, we include the effect on proportion of breeders previously described for all extreme drought effects under the assumption that extreme droughts likely also effect the proportion of breeding individuals. Lastly, we included a drought effect with a combination of intensities for droughts that occur for at least three years and are followed by a year of extreme severity (e.g., three years on moderate followed by a year of extreme drought. These types of moderate-extreme drought combinations were not reported in literature or data provided to this SSA team, but the effect of sequential moderate and extreme drought likely affect populations more severely than a severe drought or moderate drought alone. To account for this, we summed the proportional effects of moderate and extreme droughts described previously. For all effects of drought on survival, values of proportional reduction in survival were drawn from Beta distributions which we converted from Normal distributions with a mean of the proportional survival reduction (Table 4).

Table 4. Descriptions of demographic effects of various combinations of drought intensities and durations on WPT life history.

Intensity	Duration (years)	Affected parameter(s)	Proportional reduction or range of values	Source
Moderate	4+	Hatchling survival (S_H) Juvenile survival (S_J) Adult survival (S_A)	59.8%	(Manzo et al. 2021 pp.491)
		Proportion breeding (PB)	0–0.2	(Holland 1992 p.5-2; Manzo et al. 2021 p.491)
Extreme	1	Hatchling survival (S_H) Juvenile survival (S_J) Adult survival (S_A)	16.7%	(Purcell et al. 2017, pp.21-22)
		Proportion breeding (PB)	0–0.2	(Holland 1992 p.5-2; Manzo et al. 2021 p.491)
	2	Hatchling survival (S_H) Juvenile survival (S_J) Adult survival (S_A)	44.4%	(Purcell et al. 2017, pp.21-22)
		Proportion breeding (PB)	0–0.2	(Holland 1992 p.5-2; Manzo et al. 2021 p.491)
	3	Hatchling survival (S_H)	95.3%	(Lovich et al. 2017 pp.5-6)

		Juvenile survival (S_j) Adult survival (S_A)		
		Proportion breeding (PB)	0–0.2	(Holland 1992 p.5-2; Manzo et al. 2021 p.491)
Moderate AND extreme drought in last year	4+	Hatchling survival (S_H) Juvenile survival (S_j) Adult survival (S_A)	59.8% + 16.7%	(Lovich et al. 2017 pp.5-6; Purcell et al. 2017 pp.21-22; Manzo et al. 2021 p.491)
		Proportion breeding (PB)	0 – 0.2	(Holland 1992 p.5-2; Manzo et al. 2021 p.491)

Invasive bullfrog predation

While the negative effects of bullfrogs on WPT populations are well-documented within the literature, most evidence is largely anecdotal (Sloan 2012 entire; Hallock et al. 2017 entire) or based on distribution of size classes (Holland 1992 p.54-64; Hays et al. 1999 p.15; Nicholson et al. 2020 entire), neither of which are direct measures of demographic effects. We devised a way to incorporate bullfrog effects into hatchling and juvenile WPT survival rate using size class data for WPT reported in areas with and without bullfrogs present (Holland 1992 p.54-64, Hays et al. 1999 p.15). Hallock et al. (2017 p.5) stated that WPT less than or equal to three years old are vulnerable to bullfrog predation, but Holland (1992 p.54-64) and Hays et al. (1999 p.15) reported these populations by size, not age, so we define 90 mm as the maximum standard carapace length of three year old female turtles (Fig.2 in Germano et al. 2022). We divided the vulnerable proportion of the population (WPT ≤ 3 years old, Hallock et al. 2017 p.5) for samples with bullfrogs by that for comparable samples without bullfrogs to obtain the effect of bullfrog presence on the proportion of vulnerable individuals within the populations (Eqn.8). To translate to our model framework, we assumed that these effects on vulnerable population proportions are approximately equal to effects on survival of vulnerable populations.

$$\text{Eqn. 8} \quad S_{BF} = S \times \left(\frac{P_{BF}}{P_{NBF}} \right)$$

S_{BF} is the survival of hatchlings or juveniles in areas with bullfrogs (BF), S represents either hatchling or juvenile survival (S_H, S_j), P_{BF} is the proportion of the population that are small turtles (≤ 3 years old) in areas with bullfrogs and P_{NBF} is the proportion of the population that are small turtles (≤ 3 years old) in areas with no bullfrogs. The effect described in Eqn. 8 was applied to both hatchling and juvenile life stages.

To account for uncertainty in effect size of bullfrogs, demonstrated by the variation in calculated values from the published studies of S_{BF} (0, 0.2415, 0.3717, 0.6278, 0.8688), we used a Uniform distribution with minimum of 0 and maximum of 0.8688 from which to draw values of annual values of hatchling and juvenile survival when bullfrogs are present. Because not all juveniles are affected by bullfrogs, we calculated juvenile survival when bullfrogs are present as a weighted average of non-bullfrog and bullfrog juvenile survival rates. Here, the weighting was defined as the proportion of the juveniles vulnerable, and not vulnerable to bullfrogs. We modeled the proportion of juveniles as a Uniform distribution ranging from 1/3 to 1/2 to incorporate uncertainty into the assumption that WPT ≤ 3 years old are affected (Hallock et al. 2017 p.5)

Habitat loss

To model habitat loss, we used a ceiling-type density dependence function (Morris and Doak 2002 pp.311-322) to limit maximum population size and then used annual reductions in the population ceiling to mimic habitat loss in an AU. This is a similar approach used by McGowan et al. (2017 p.123) to model the potential effects of habitat loss in Sonora Tortoise. Here, the initial ceiling, or carrying capacity (K), was calculated using the same methods for initial abundance (see ‘Current Conditions’), but with maximum HUC12 abundance value possible for its corresponding values of human modification and habitat area. That is, we used the same calculation for estimating habitat-based abundance but used the upper bound of the confidence interval for the intercept and slope terms of the HUC12 abundance relationship (Table 3). We then simulated carrying capacity into the future as a function of annual habitat loss rate (Eqn. 9).

$$\text{Eqn. 9} \quad K_t = K_{t-1} \times (1 - hlr)$$

Where K is the carrying capacity, t is the year in the simulation, and hlr is the habitat loss rate. For example, if the carrying capacity of a region is 10,000 turtles in year-1 and we expect a habitat loss rate of 1% per year, the carrying capacity in year-2 would be: $10,000 \times (1-0.01) = 9,900$. We varied habitat loss rate across scenarios (see ‘Future Condition Scenarios’) to evaluate alternative future plausible habitat loss rates.

The carrying capacity function was designed to limit population level reproduction if the population exceeds the annual value of K . There are likely a variety of factors by which density and habitat loss affects WPT and their populations, e.g., leaving water bodies to find nest sites leads to increased risk of road mortality, limited nesting habitat leads to increased nest densities and higher risk of predation by meso-predators, etc. However, there are no empirical quantifiable studies that show a specific demographic effect of habitat loss. Thus, in the absence of a quantified functional relationship, we applied a simple ceiling type density dependence function that sets population fertility (F) to 0 if the ceiling is exceeded. This approach allowed us to implement and evaluate the possible effects of habitat loss on population viability, without speculating on the ecological relationships and functional form of density dependence.

Future Condition Scenarios

We developed two plausible future scenarios for our stochastic simulation model (Scenarios 1, 2). Scenarios 1 and 2 represent the upper and lower plausible limits, respectively, of our projections. We project our three modeled threats (drought, invasive bullfrogs, habitat loss) into the future to best estimate the future conditions of WPT. Methods for developing the scenarios were identical between species. We also created three additional scenarios for each species to test the sensitivity of results to each threat. In the sensitivity analysis scenarios, for each threat, the annual rate of change of each threat was increased by 25% while that of the remaining two threats was held constant.

We based our scenarios of drought on the Representative Concentration Pathways (RCPs) described by the Intergovernmental Panel on Climate Change (IPCC, IPCC 2014 p.8). We selected RCP8.5 and RCP4.5 to represent the ‘medium-low’ and ‘high’ emission pathways, for Scenarios 1 and 2, respectively. We projected drought conditions into the future using AU-specific annual rates of increase of moderate and extreme drought for RCPs 8.5 and 4.5. These rates of increase were determined by comparing annual drought frequencies of historical and recent time periods, which we defined as 1986-2005 and 2006-present, respectively (Figure 19).

The definitions of these time periods are consistent with the IPCC’s 5th Climate Assessment’s baseline and recent time periods for comparisons of climate change effects (Collins et al. 2013 p.1034). The drought frequencies for each time period were calculated using the methods described above (Figure 19, see ‘Current Conditions’). The annual rates were calculated by dividing the drought frequencies by the number of years in each period. We then subtracted the recent annual frequencies from the historical annual frequencies to arrive at an annual rate of increase (Table 5, Table 6), which we defined as the RCP 8.5 scenario under the assumption that we have largely been following that pathway since 2006. The annual rates of increase for RCP 4.5 were defined as anywhere from 54.5-74.5%% of that of RCP 8.5, based on the difference in average PDSI values between the RCPs by 2100 reported in (Table 5, Table 6; Cook et al. 2015 p.2-3). The difference in PDSI values in 2100 in Cook et al., (2015 p.2-3) is 64%, but we incorporated additional uncertainty into this value by drawing it from a uniform distribution with lower and upper limits +/- 10% (e.g. Uniform(0.545, 0.745)). When measured annual rates of increase in an AU were negative, we rounded them to 0 under the assumption that drought frequency may be unchanged into the future, assuming drought frequencies will not decrease in the WPT range (Weiss et al. 2009 entire; Cook et al. 2015 entire). There were only three instances of negative values out of 26 total frequencies for *A. marmorata*, and none for *A. pallida*.

In the stochastic simulation, we modeled droughts in a stepwise process to approximate the different intensities. A Bernoulli trial was performed to determine if there was a drought of at least moderate intensity in a single year in the simulation (Table 5, Table 6). If it was at least a moderate drought year, another Bernoulli trial was performed to determine if that drought was of extreme intensity. The extreme drought frequency values reported in Tables 5 and 6 are thus the probability that a drought will be of extreme intensity in years that have droughts of moderate intensity.

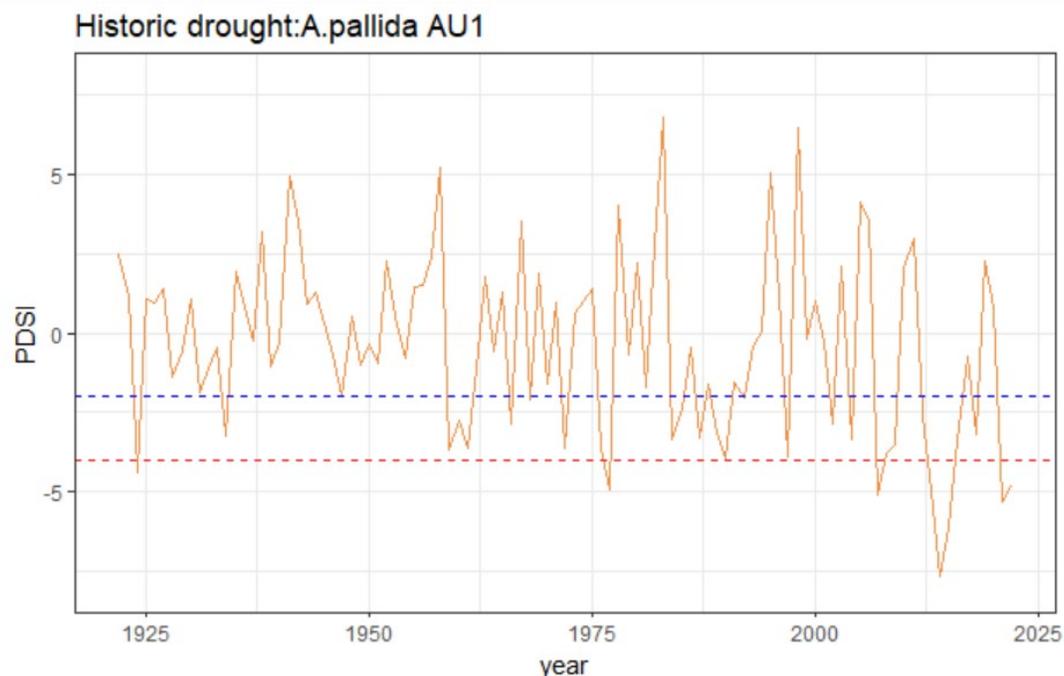


Figure 19. Example of historic drought conditions for *A. pallida* in Analysis Unit (AU1). This process was performed for all AUs of both species. The blue line represents the threshold for a moderate drought

(Palmer Drought Severity Index, PDSI < -2) and the red line represents the threshold for an extreme drought (PDSI < -4).

Table 5. Initial frequencies and annual rate of change (ROC) of moderate drought and extreme drought for both scenarios for *A. marmorata*. Extreme drought frequency is defined as the probability of a drought of at least moderate intensity being of extreme intensity. *indicates ROC values that were rounded to 1. U(0.545, 0.745) indicates a value drawn from a uniform distribution with a minimum of 0.545 and maximum of 0.745.

AU	Moderate drought			Extreme drought		
	Initial frequency	ROC Scenario 1	ROC Scenario 2	Initial frequency	ROC Scenario 1	ROC Scenario 2
3	0.176	1.001	1.001 x U(0.545, 0.745)	0	1.000	1.000 x U(0.545, 0.745)
4	0.235	1.0*	1.0* x U(0.545, 0.745)	0.059	1.001	1.001 x U(0.545, 0.745)
5	0.235	1.004	1.004 x U(0.545, 0.745)	0	1.000	1.000 x U(0.545, 0.745)
6	0.235	1.007	1.007 x U(0.545, 0.745)	0	1.000	1.000 x U(0.545, 0.745)
7	0.529	1.012	1.012 x U(0.545, 0.745)	0.059	1.0*	1.0 x U(0.545, 0.745)
8	0.353	1.004	1.004 x U(0.545, 0.745)	0.059	1.001	1.001 x U(0.545, 0.745)
9	0.235	1.002	1.002 x U(0.545, 0.745)	0	1.000	1.000 x U(0.545, 0.745)
10	0.412	1.010	1.010 x U(0.545, 0.745)	0.0589	1.004	1.004 x U(0.545, 0.745)
11	0.412	1.010	1.010 x U(0.545, 0.745)	0.0589	1.004	1.004 x U(0.545, 0.745)
12	0.471	1.014	1.014 x U(0.545, 0.745)	0.235	1.015	1.015 x U(0.545, 0.745)
13	0.588	1.013	1.013 x U(0.545, 0.745)	0.412	1.026	1.026 x U(0.545, 0.745)
14	0.647	1.022	1.022 x U(0.545, 0.745)	0.353	1.022	1.022 x U(0.545, 0.745)

Table 6. Initial frequencies and annual rate of change (ROC) of moderate drought and extreme drought for both representative concentration pathway (RCP) scenarios for *A. pallida*. Extreme drought frequency is defined as the probability of a drought of at least moderate intensity being of at least extreme intensity.

AU	Moderate drought			Extreme drought		
	Initial frequency	ROC Scenario 1	ROC Scenario 2	Initial frequency	ROC Scenario 1	ROC Scenario 2
1	0.647	1.022	1.022 x U(0.545, 0.745)	0.353	1.022	1.022 x U(0.545, 0.745)
2	0.647	1.028	1.028 x U(0.545, 0.745)	0.529	1.025	1.025 x U(0.545, 0.745)
3	0.647	1.015	1.015 x U(0.545, 0.745)	0.412	1.015	1.015 x U(0.545, 0.745)
4	0.647	1.028	1.028 x U(0.545, 0.745)	0.529	1.025	1.025 x U(0.545, 0.745)
5	0.647	1.028	1.028 x U(0.545, 0.745)	0.529	1.025	1.025 x U(0.545, 0.745)

The projection of invasive bullfrogs was calculated similarly to that of drought, comparing the presence of bullfrogs in HUC12 units in 2006 with 2022. As in the ‘Current Conditions’ section, we used data from GBIF (GBIF.org 2022) as our observations of bullfrogs each observation year. The annual rate of increase in bullfrog spread was calculated for each AU as the annualized difference between proportion of HUC12s with bullfrogs in an AU in 2022 versus 2006. There was not enough information on bullfrog spread into the future to create multiple scenarios, so bullfrog rate of spread is the same for Scenarios 1 and 2. We incorporated uncertainty into the spread of bullfrogs by drawing the rate from a uniform distribution with limits +/- 10% of the calculated value, for each replicate of the simulation. If there was no expansion of bullfrogs within an AU between 2006 and 2022, the annual rate of increase into the future is 0. It is possible that bullfrogs already occupy all the suitable habitat in these AUs and thus will not spread further, such that their annual rate of change would be 0 (Table 7). The annual rate of change for AU 2 for *A. marmorata* was also 0, because there were no observations of bullfrogs in the AU between 2006 and 2022 (Table 7). In both cases where annual rate of increase was 0, if there were unoccupied areas with suitable habitat in these AUs but there had not been an observed spread in recent years, our simulations likely underestimate the effect of bullfrogs on WPT viability.

Table 7. Annual rate of spread of bullfrogs by Analysis Unit (AU) for both WPT species. *indicates an AU where there were no bullfrogs were observed between 2006 to 2022.

Species	AU	Annual rate of spread of bullfrogs	
<i>A. marmorata</i>	3	0*	
	4	0	
	5	0.0004	
	6	0	
	7	0	
	8	0.001	
	9	0.001	
	10	0.001	
	11	0.003	
	12	0.002	
	13	0.003	
	14	0.003	
	<i>A. pallida</i>	1	0.003
		2	0.012

	3	0.005
	4	0.033
	5	0.025

For habitat loss scenarios, we used projections from the Environmental Protection Agency’s Integrated Climate and Land Use Scenarios (ICLUS) project (U.S. EPA 2017 entire). The ICLUS project incorporates RCP information with Shared Socioeconomic Pathway (SSP) projections to predict habitat loss. For Scenario 1, we used the RCP 8.5/SSP5 projection from the ICLUS model. SSP5 represents the ‘fossil fueled development’ scenario. For Scenario 2, we used the RCP 4.5/SSP2 scenario, where SSP2 is the ‘middle of the road scenario’. To be consistent with the climate models used in the California Climate Change Assessment (Pierce et al. 2018 entire), we used the HadGEM2-ES projections within the ICLUS project. The ICLUS projections report area of land cover types for each state in 20-year periods. We isolated the wetland land cover type, assuming that rates of change in riparian area are an index to wetland area. For each 20-year period, we calculated an annual rate of habitat loss or increase (Table 8). Wetland total area oscillated, although overall it decreased throughout the rest of the century (hereafter, we refer to this rate as “annual habitat loss” for brevity). The projections of annual habitat loss were assigned to each AU based on the state they were in: Oregon, or California (Table 8).

Table 8. Annual rate of habitat loss for Scenarios 1 and 2 (RCP8.5/SSP5, RCP4.5/SSP2, respectively) by state (Oregon, OR; California, CA).

State	2022-2040		2041-2060		2061-2080		2081-2100	
	Scenario 1	Scenario 2						
OR	0.0002947	0.0000932	0.0004486	0.0000754	0.0005276	0.0000412	0.0004726	0.0000132
CA	0.0005998	0.0002084	0.0009054	0.0002874	0.0008533	0.0002308	0.0007530	0.0000669

Results

Initial abundance

Northwestern pond turtle, A.marmorata

Initial abundance was lowest for the smallest AU’s (3,4,7). (Table 9). On average, the abundance of the *A. marmorata* AU’s ranged from 396 to 22,954 individuals for a range-wide sum of 97,171, excluding the Washington AUs.

Table 9. Mean initial abundance, standard deviation, standard error of 1,000 replicates for each Analysis Unit (AU) for *A. marmorata*.

AU	Mean	Standard deviation	Standard error
3	396	330	10
4	273	305	10
5	11555	2872	91
6	3113	1162	37
7	265	284	9
8	3696	1263	40

9	22954	5384	170
10	22496	5275	167
11	12287	3140	99
12	9118	2360	75
13	7106	1857	59
14	3912	1134	36

Southwestern pond turtle, A.pallida

Initial abundance was highest for AU 1 and lowest for AU 4 for the southwestern species. Mean AU abundance ranged from 436 to 8,622 individuals, with a range-wide sum of 15,407 (Table 10).

Table 10. Mean initial abundance and standard error of 1,000 replicates for each Analysis Unit (AU) for *A. pallida*.

AU	Mean	Standard Deviation	Standard Error
1	8622	2181	69
2	3465	1050	33
3	436	266	8
4	770	343	11
5	2114	720	23

Future condition

Northwestern pond turtle, A. marmorata

Probability of extinction was near zero until 2040 for both Scenarios. After 2040, probability of extinction increased throughout the simulation for both Scenarios (Figure 20, Figure 21, Figure 23, Figure 24). The range-wide mean probability of extinction in 2100 was 50.% for both Scenarios (Figure 21). Population growth rate (λ) was always less than 1 (Figure 26) meaning populations were in decline, and abundance precipitously declined in all AUs over the projection timeline (Figure 25). For the Washington populations, Pramuk et al. (2012 p.50-51) predict significantly higher extinction risk for these State-endangered populations. In a ‘no conservation action’ scenario, the population declines rapidly towards extinction, with less than 50 individuals left in the state by 2100. Only some scenarios of conservation action resulted in an increase in population size (Pramuk et al. 2012 p.50-51). Sensitivity testing (Table 11) revealed that the probability of extinction was moderately sensitive to an increase in the annual rate of change of bullfrog spread. Probability of extinction in 2100 increased by a maximum of 0.126 in AU 9, with sensitivity higher for the southern AUs (Table 13). These sensitivity results are consistent with the findings of Nicholson et al., (2020 entire). Despite only demonstrating moderate sensitivity to one of the threats modeled here, probability of extinction is fairly high by the end of the projection, suggesting that there is an underlying extinction risk for the species. Outcomes of the stochastic simulation and population growth rate were most sensitive to values of adult survival, so the underlying extinction risk can reasonably be attributed to the estimates of adult survival. The importance of adult survival for population viability is typical for long-lived species and therefore factors that negatively affect adult survival become important conservation issues. Manzo et al (2021 p.496) report two drastically

different projections of population abundance, based on different estimates of adult survival, confirming our findings of population sensitivity to adult survival values.

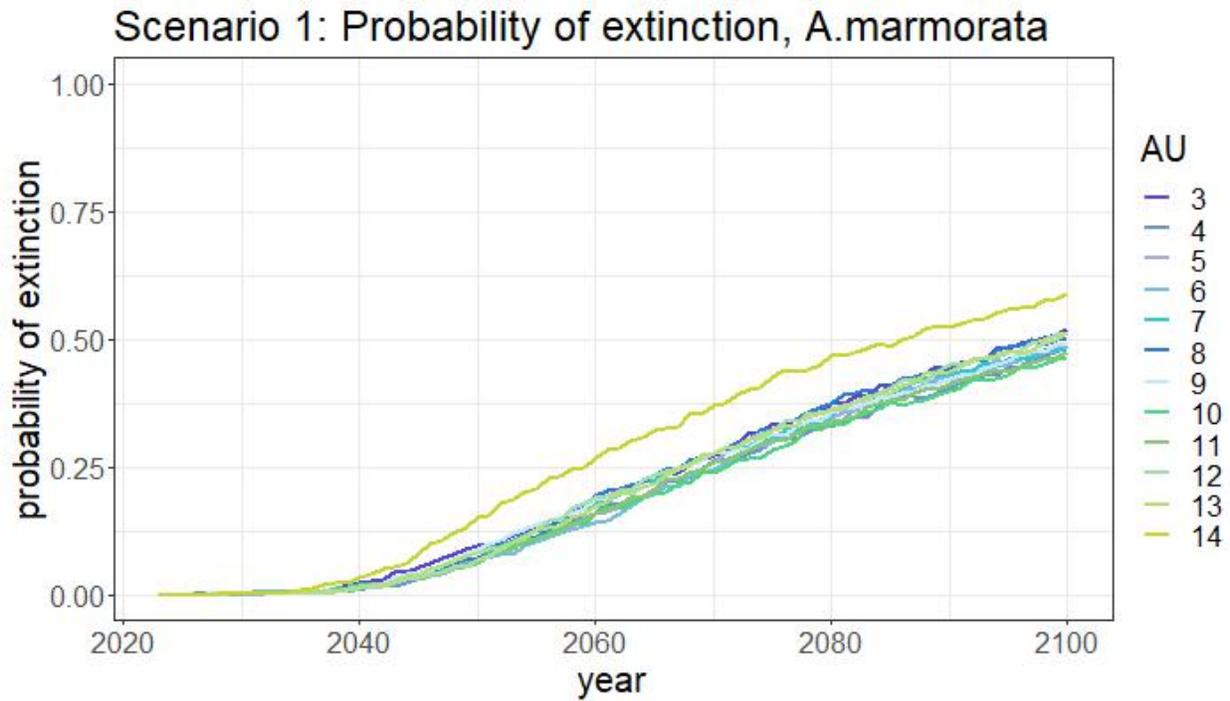


Figure 20. Probability of extinction overtime in the Scenario 1 simulation for *A. marmorata*.

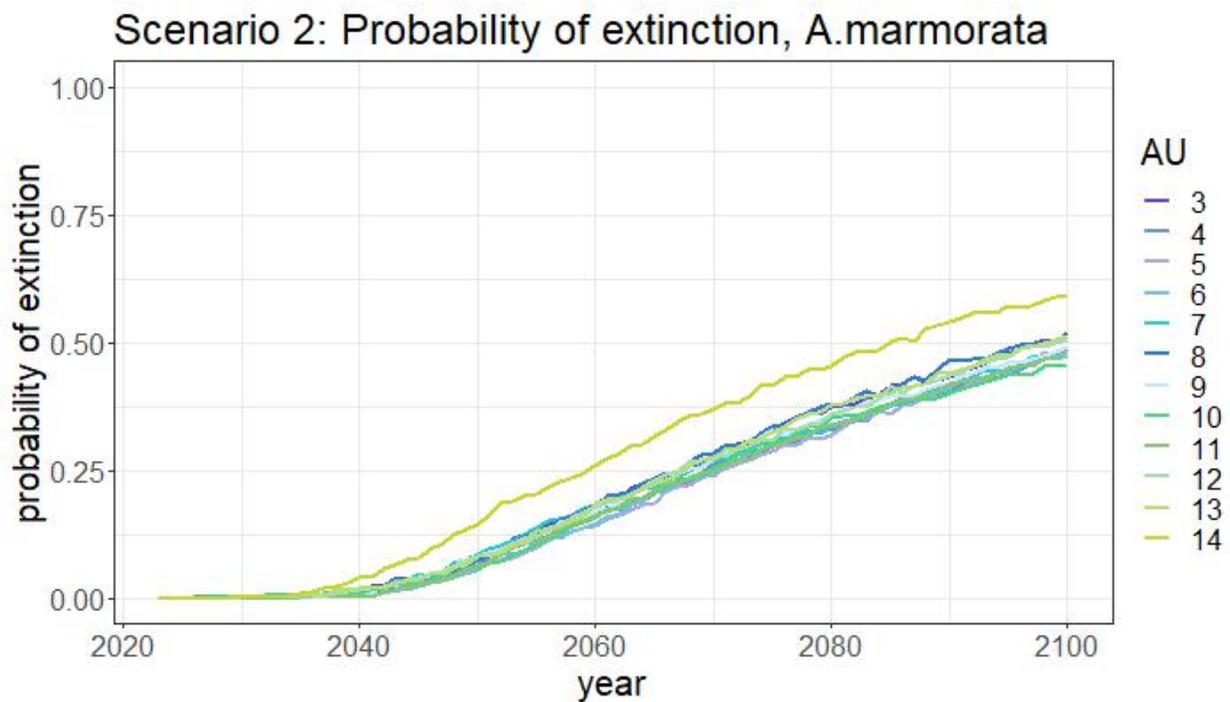


Figure 21. Probability of extinction overtime in the Scenario 2 simulation for *A. marmorata*.

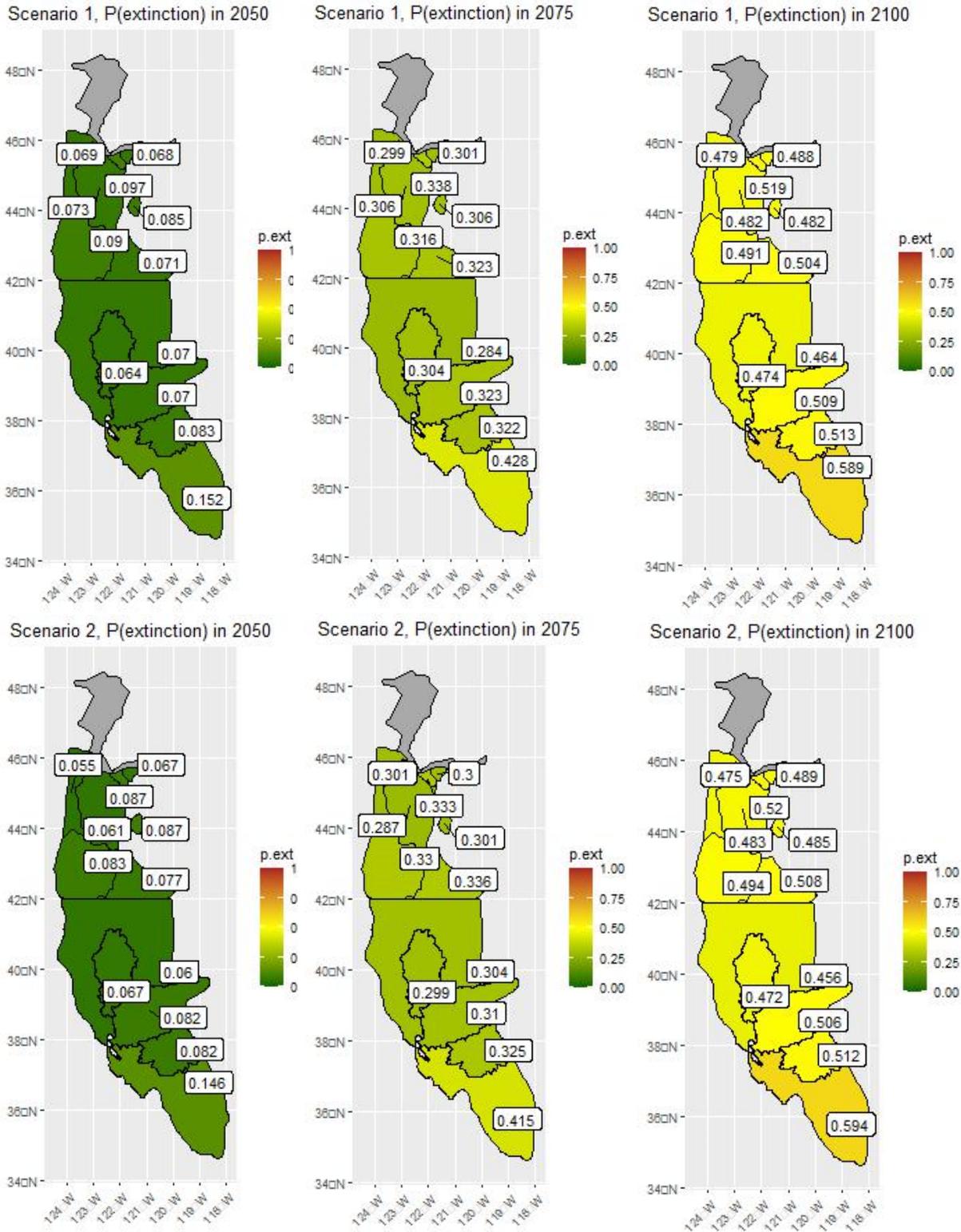


Figure 22. Maps of probability of extinction in 2050, 2075, and 2100 (columns) for Scenarios 1 and 2 (rows) in each Analysis Unit (AU) in the *A. marmorata* range. Labels are the AU-specific value of extinction probability in 2100. Washington AUs were not modeled and are thus greyed out.

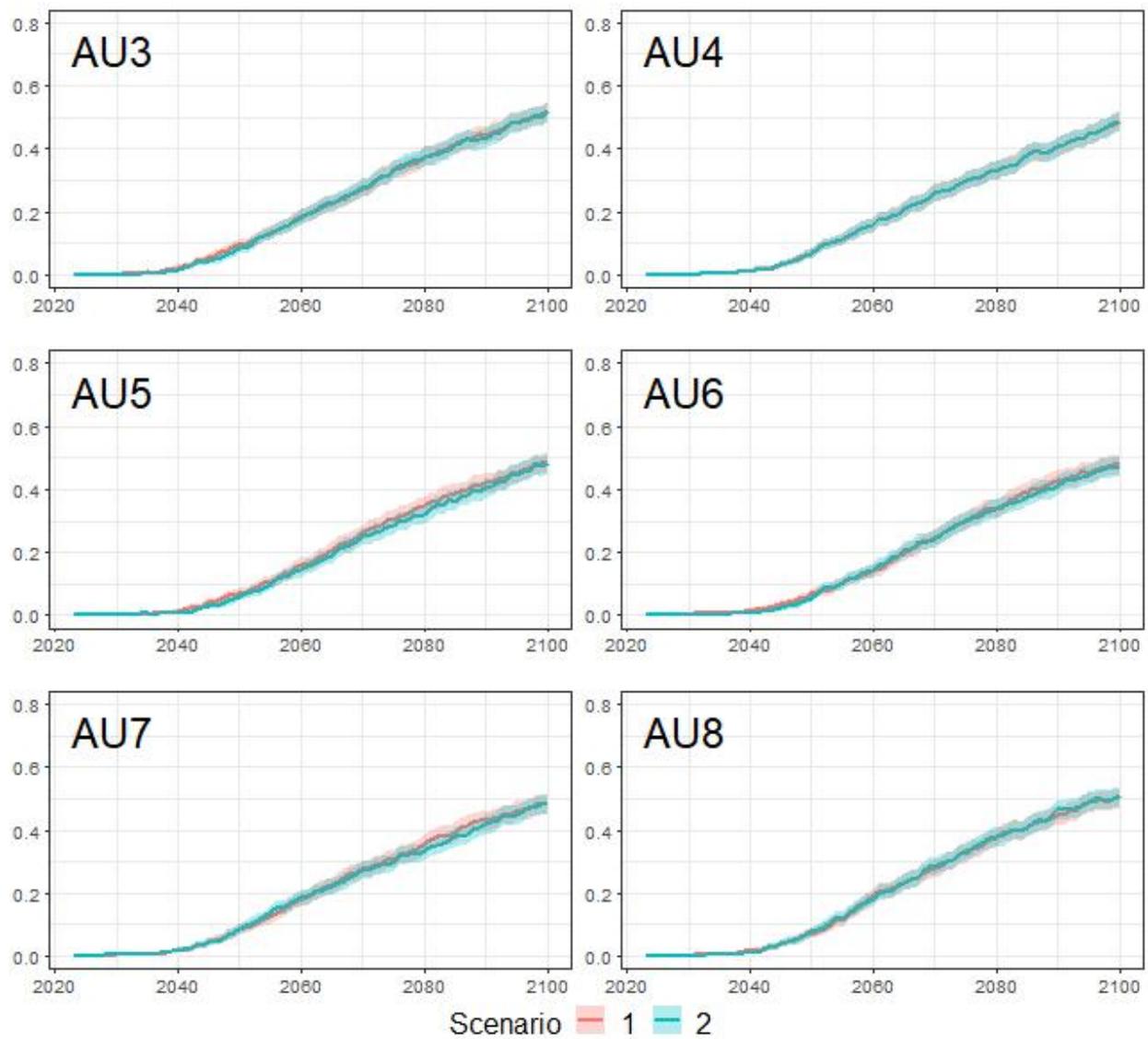


Figure 23. Probability of extinction overtime for Scenarios 1 (red) and 2 (blue) for Analysis Units (AUs) 3-8 in the *A. marmorata* range.

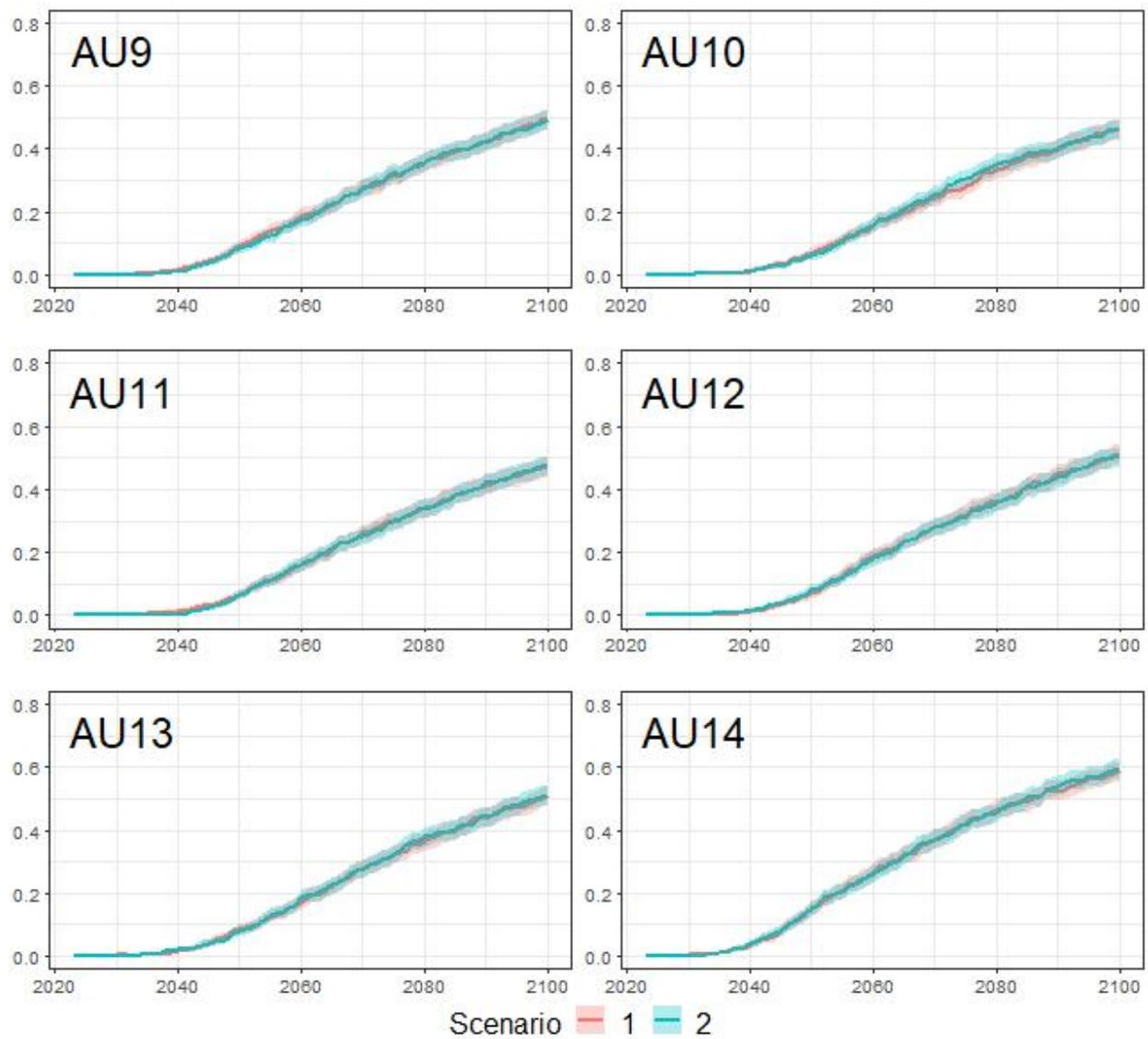


Figure 24. Probability of extinction overtime for Scenarios 1 (red) and 2 (blue) for Analysis Units (AUs) 9-14 in the *A. marmorata* range.

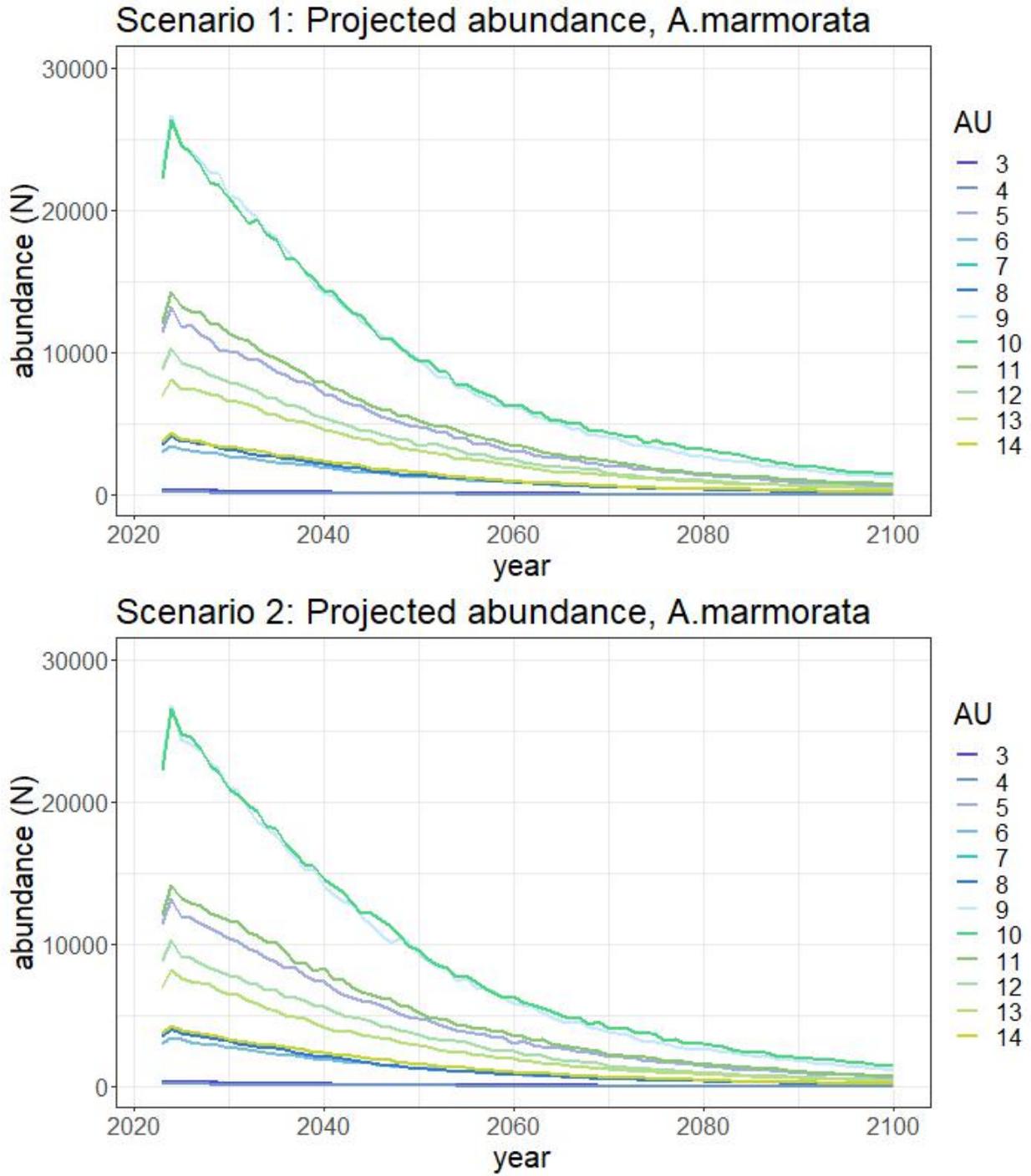


Figure 25. Projected abundance overtime of *A. marmorata* for Scenario 1 (top) and Scenario 2 (bottom) for each AU.

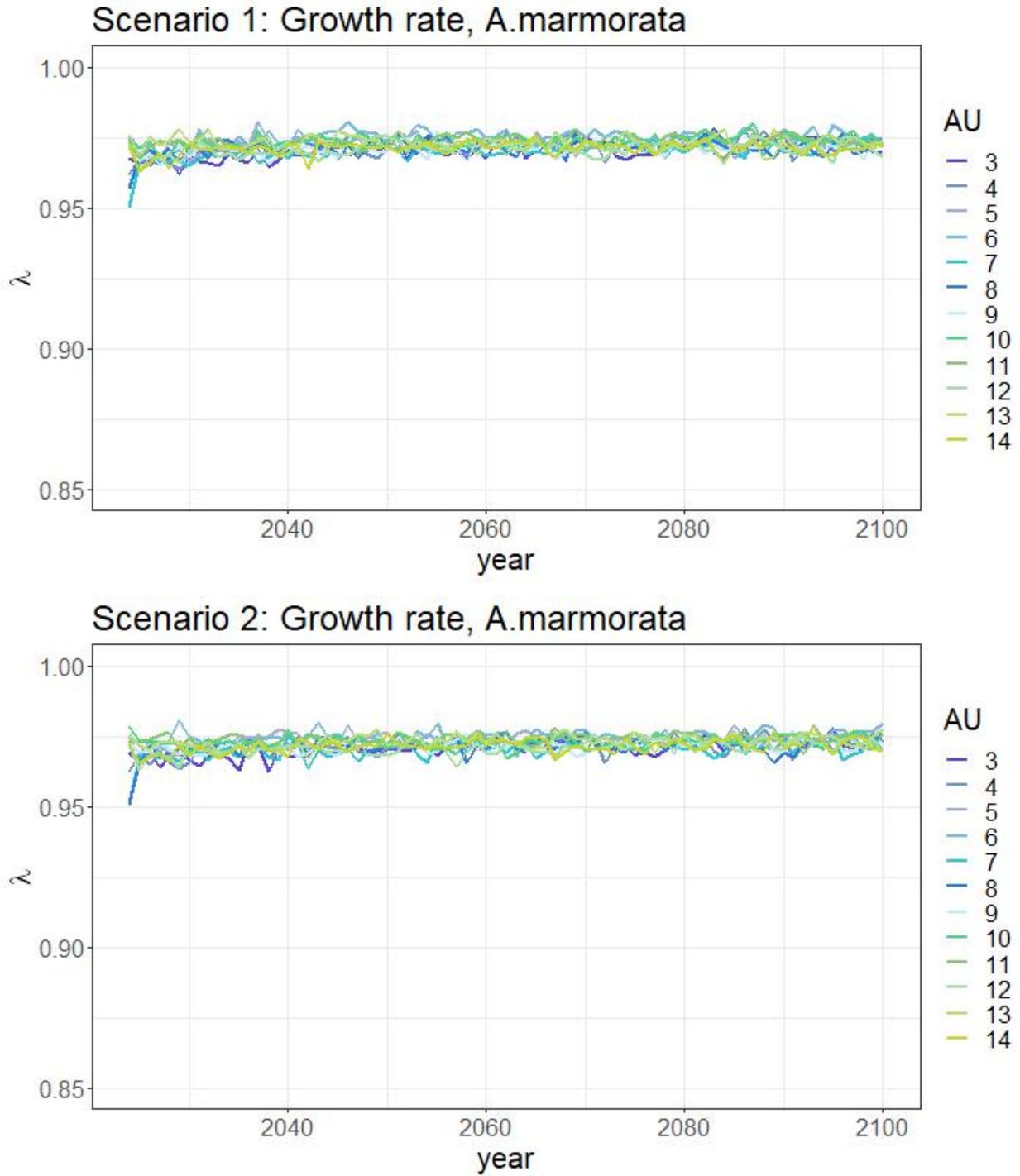


Figure 26. Population growth rate (lambda, λ) overtime for Scenarios 1 (top) and 2 (bottom) for each AU in the *A. marmorata* range.

Table 11. Mean and standard error of population growth rate (lambda, λ) in 2050, 2075, and 2100 for Scenario 1 for each AU in the *A. marmorata* range.

Year	AU	Lambda	Standard error
2050	3	0.9726	0.0024

2050	4	0.9713	0.0023
2050	5	0.9731	0.0025
2050	6	0.9704	0.0026
2050	7	0.971	0.0025
2050	8	0.9705	0.0026
2050	9	0.9696	0.0026
2050	10	0.9746	0.0023
2050	11	0.9762	0.0024
2050	12	0.9756	0.0023
2050	13	0.971	0.0025
2050	14	0.9725	0.0026
2075	3	0.9686	0.0025
2075	4	0.9736	0.0024
2075	5	0.9753	0.0024
2075	6	0.9733	0.0026
2075	7	0.9745	0.0024
2075	8	0.9716	0.0027
2075	9	0.9741	0.0024
2075	10	0.9756	0.0024
2075	11	0.9761	0.0025
2075	12	0.9748	0.0025
2075	13	0.9741	0.0026
2075	14	0.969	0.0027
2100	3	0.9697	0.0025
2100	4	0.9743	0.0024
2100	5	0.9733	0.0024
2100	6	0.9713	0.0026
2100	7	0.9734	0.0025
2100	8	0.9727	0.0025
2100	9	0.974	0.0023
2100	10	0.9731	0.0026
2100	11	0.9736	0.0025
2100	12	0.9681	0.0028
2100	13	0.9737	0.0025
2100	14	0.972	0.0025

Table 12. Mean and standard error of population growth rate (λ) in 2050, 2075, and 2100 for Scenario 2 for each AU in the *A. marmorata* range.

Year	AU	Lambda	Standard error
2050	3	0.9702	0.0026
2050	4	0.9715	0.0023

2050	5	0.9764	0.0024
2050	6	0.9713	0.0027
2050	7	0.9708	0.0025
2050	8	0.9715	0.0026
2050	9	0.9722	0.0028
2050	10	0.9722	0.0025
2050	11	0.9709	0.0025
2050	12	0.968	0.0026
2050	13	0.9718	0.0027
2050	14	0.9758	0.0023
2075	3	0.9681	0.0025
2075	4	0.9742	0.0024
2075	5	0.9705	0.0026
2075	6	0.9736	0.0025
2075	7	0.9699	0.0028
2075	8	0.9737	0.0024
2075	9	0.9699	0.0025
2075	10	0.9701	0.0026
2075	11	0.9763	0.0025
2075	12	0.9722	0.0025
2075	13	0.9764	0.0024
2075	14	0.9713	0.0026
2100	3	0.9726	0.0026
2100	4	0.975	0.0024
2100	5	0.9726	0.0026
2100	6	0.9798	0.0024
2100	7	0.9704	0.0028
2100	8	0.9774	0.0022
2100	9	0.9733	0.0024
2100	10	0.9701	0.0026
2100	11	0.976	0.0024
2100	12	0.9697	0.0027
2100	13	0.9773	0.0023
2100	14	0.9698	0.0027

Table 13. Change in probability of extinction as compared to a baseline for each threat sensitivity scenario for *A. marmorata* for each AU.

AU	Drought	Bullfrog	Habitat loss
3	-0.007	0.087	0
4	-0.005	0.085	0.002
5	0.013	0.115	0
6	-0.012	0.104	0.001

7	-0.011	0.101	-0.001
8	0.005	0.103	0
9	0.006	0.126	0
10	0.001	0.096	0
11	0.017	0.109	-0.001
12	-0.006	0.101	0
13	-0.002	0.108	0
14	0.024	0.091	-0.001

Southwestern pond turtle, A. pallida

Similar to *A. marmorata*, there was little to no extinction risk for *A. pallida* for the first 20 years of the projection, after which there is a rapid increase. By 2100, mean range-wide probability of extinction is 75.5% and 71.0% for Scenarios 1 and 2, respectively (Figure 27, Figure 28, Figure 29, Figure 30). The two scenarios rendered very similar probabilities of extinction across all AUs (Figure 30), likely due to the similarity between the underlying drought frequency parameters. Estimated abundance was drastically reduced in all AUs in both scenarios (Figure). Population growth rate (λ) was less than 1 for the duration of the simulation, indicating that populations are decreasing through 2100 (Figure 32). Towards the end of the simulation, λ began to decrease in Scenario 1, whereas it remained more stable in Scenario 2 (Figure 32, Table 14, Table 15).

Unlike *A. marmorata*, results indicated that extinction risk for *A. pallida* was sensitive to increases in annual drought frequency (Table 16). Probability of extinction in 2100 increased by up to 18.5% when annual drought frequency was increased by only 25% (Table 16). Because the population is most sensitive to values of adult survival throughout the stochastic simulation, changes to adult survival rates have the greatest impact on the model. It follows that drought effects, modeled as negative impacts on all survival parameters, including adult survival, had the greatest effect on *A. pallida* probability of extinction. This same effect is likely seen for *A. pallida* and not *A. marmorata* because of the former's higher measured annual drought frequencies (Table 5, Table 6). The population viability analysis by Manzo et al. (2021 p.496) also found that results were sensitive to drought risk which was consistent with our results.

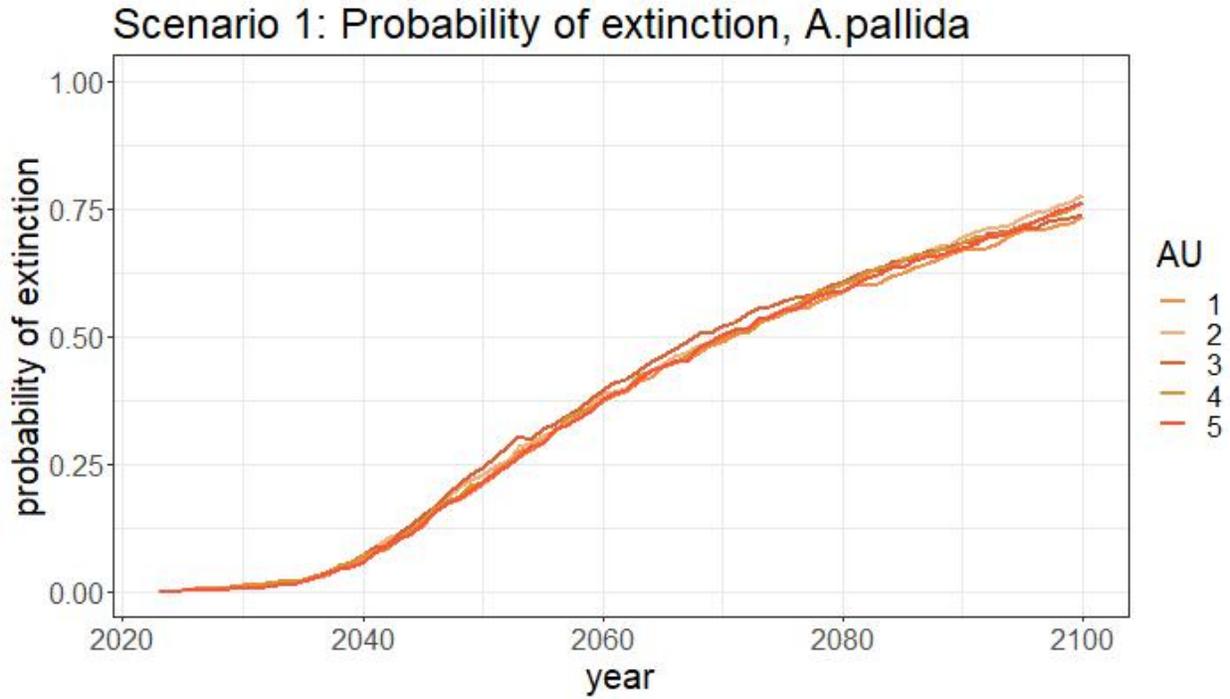


Figure 27. Probability of extinction overtime for each Analysis Unit (AU) in the *A. pallida* range for Scenario 1.

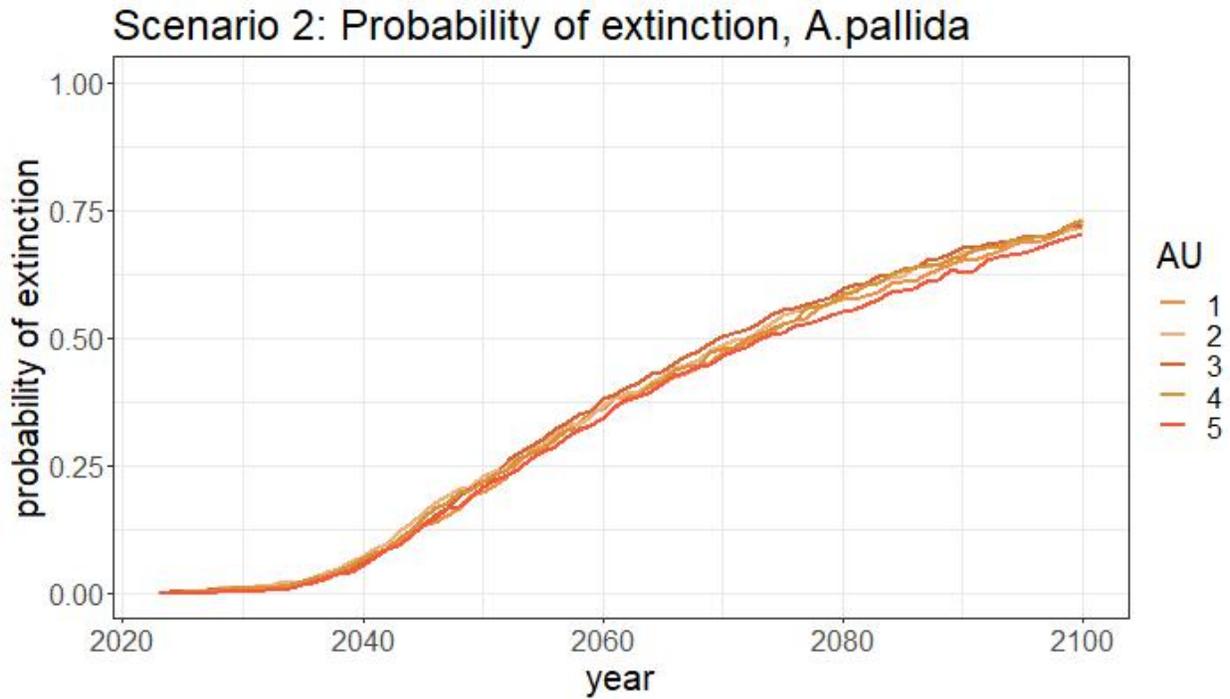


Figure 28. Probability of extinction overtime for each Analysis Unit (AU) in the *A. pallida* range for Scenario 2.

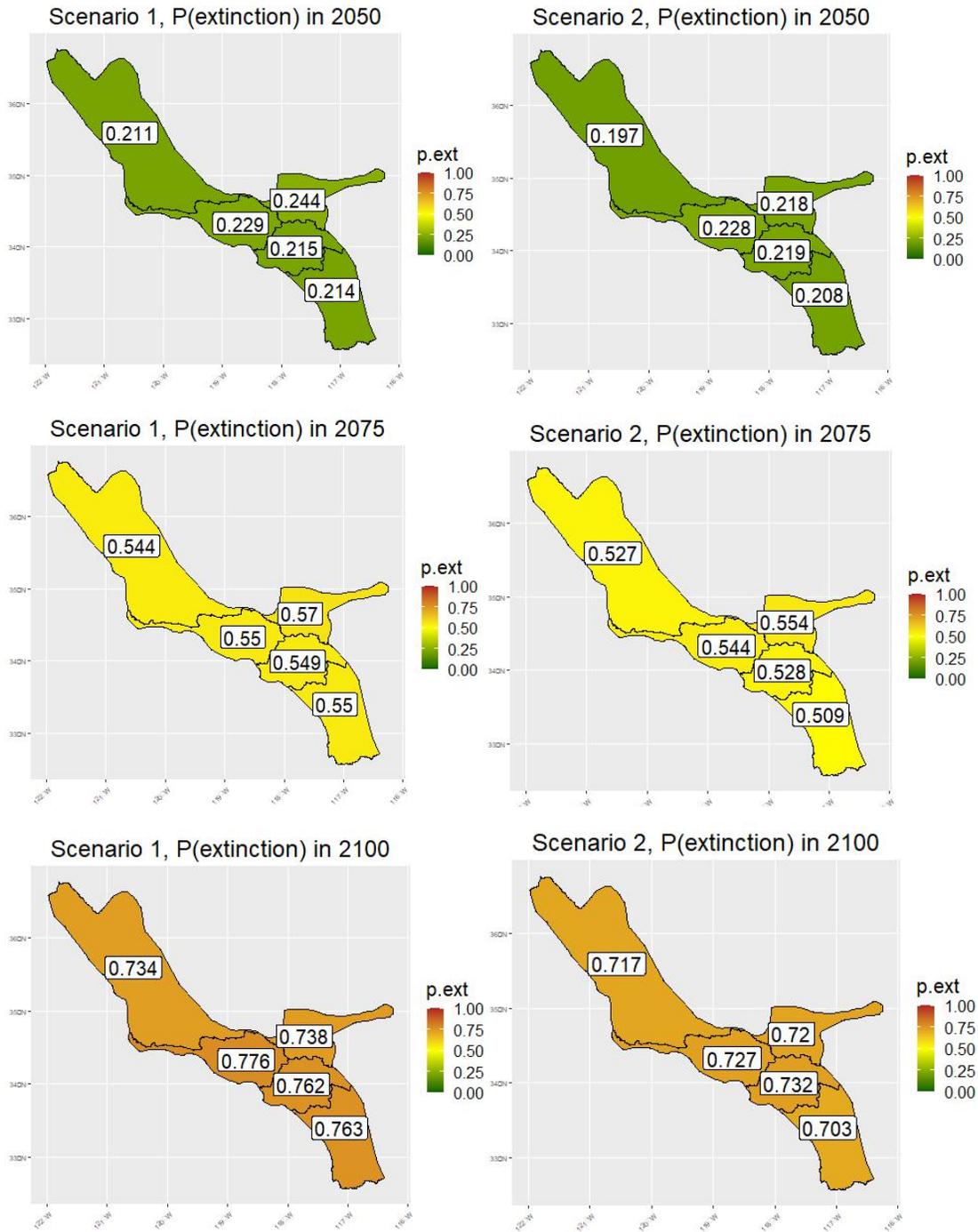


Figure 29. Maps of probability of extinction in 2050, 2075, and 2100 (rows) for Scenarios 1 and 2 (columns) in each Analysis Unit (AU) in the *A. pallida* range. Labels are the AU-specific values of extinction probability.

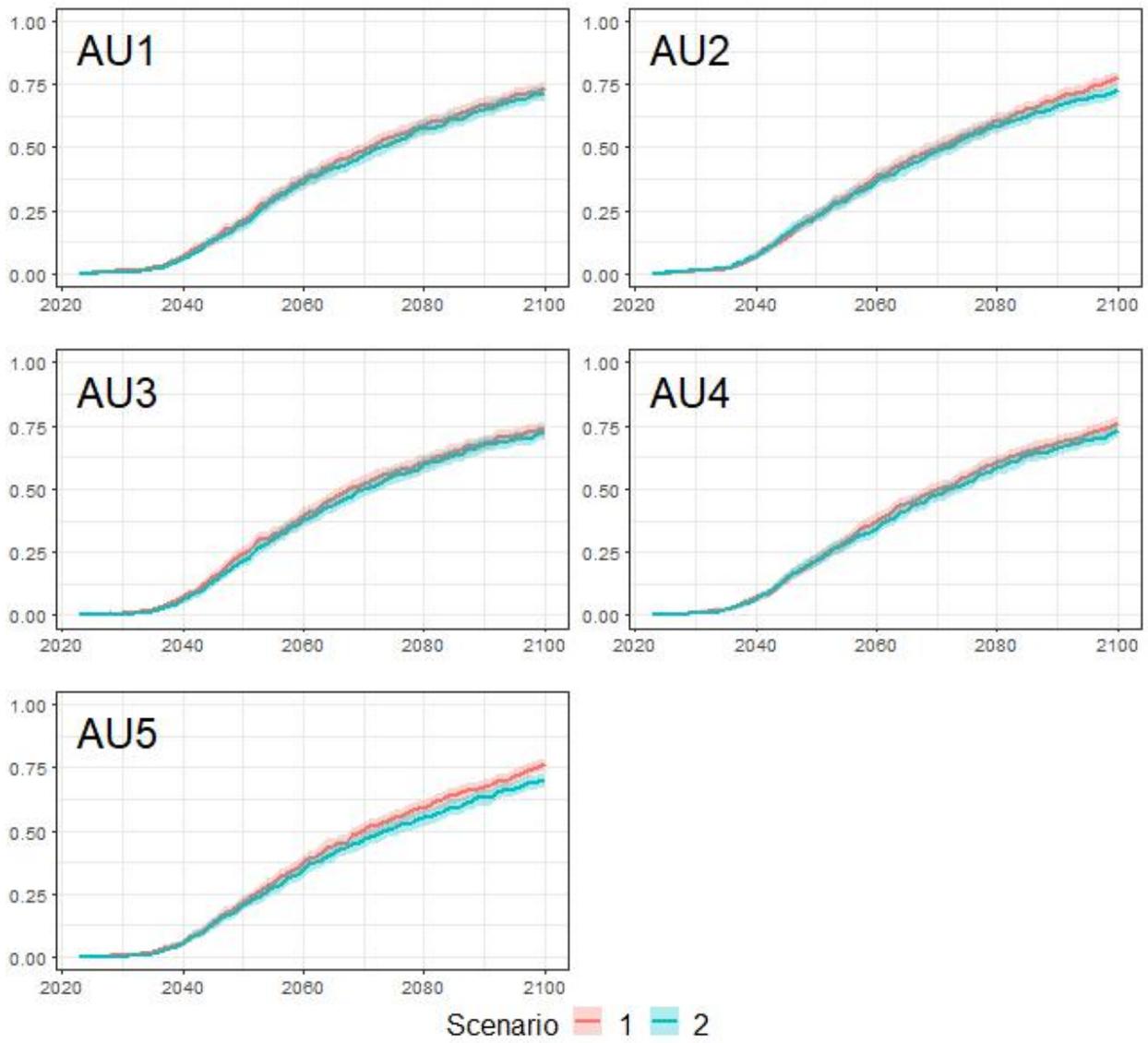
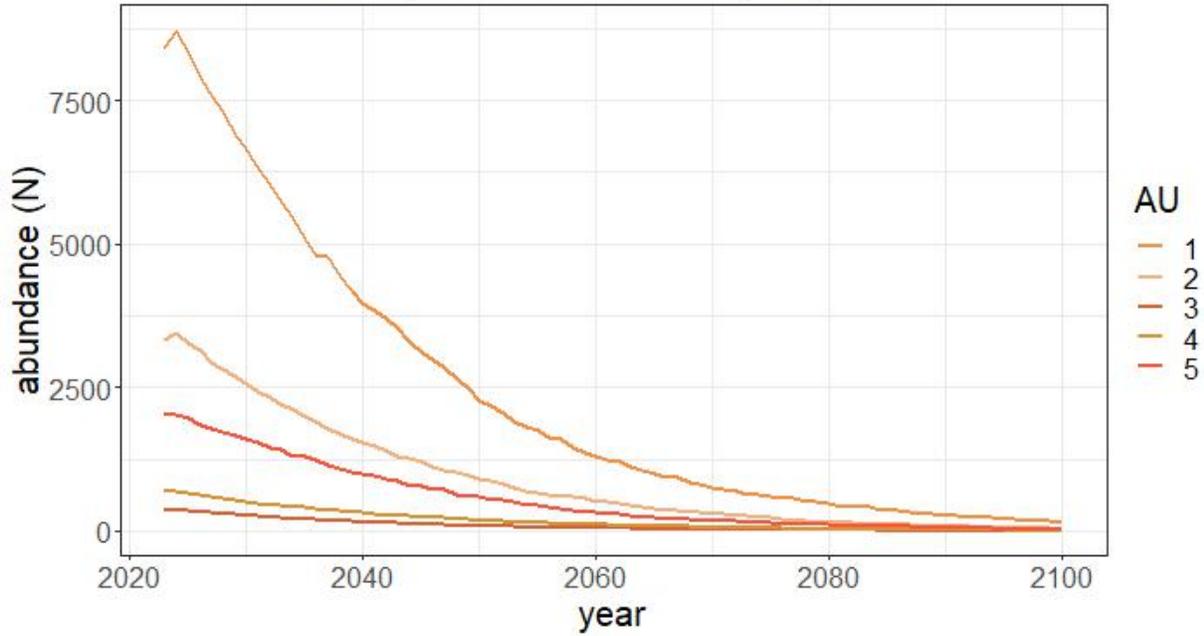


Figure 30. Probability of extinction overtime for Scenarios 1 (red) and 2 (blue) for each Analysis Unit (AU) in the *A. pallida* range.

Scenario 1: Projected abundance, *A.pallida*



Scenario 2: Projected abundance, *A.pallida*

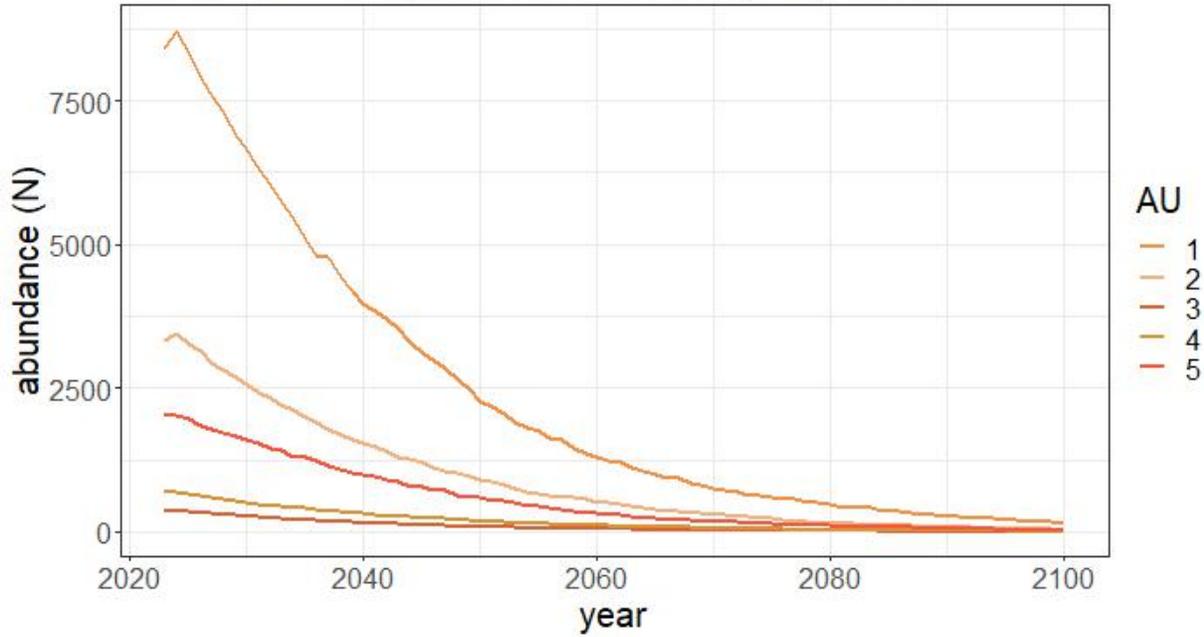


Figure 31. Projected abundance overtime of for each AU in the *A.pallida* range Scenario 1 (top) and 2 (bottom) for each AU

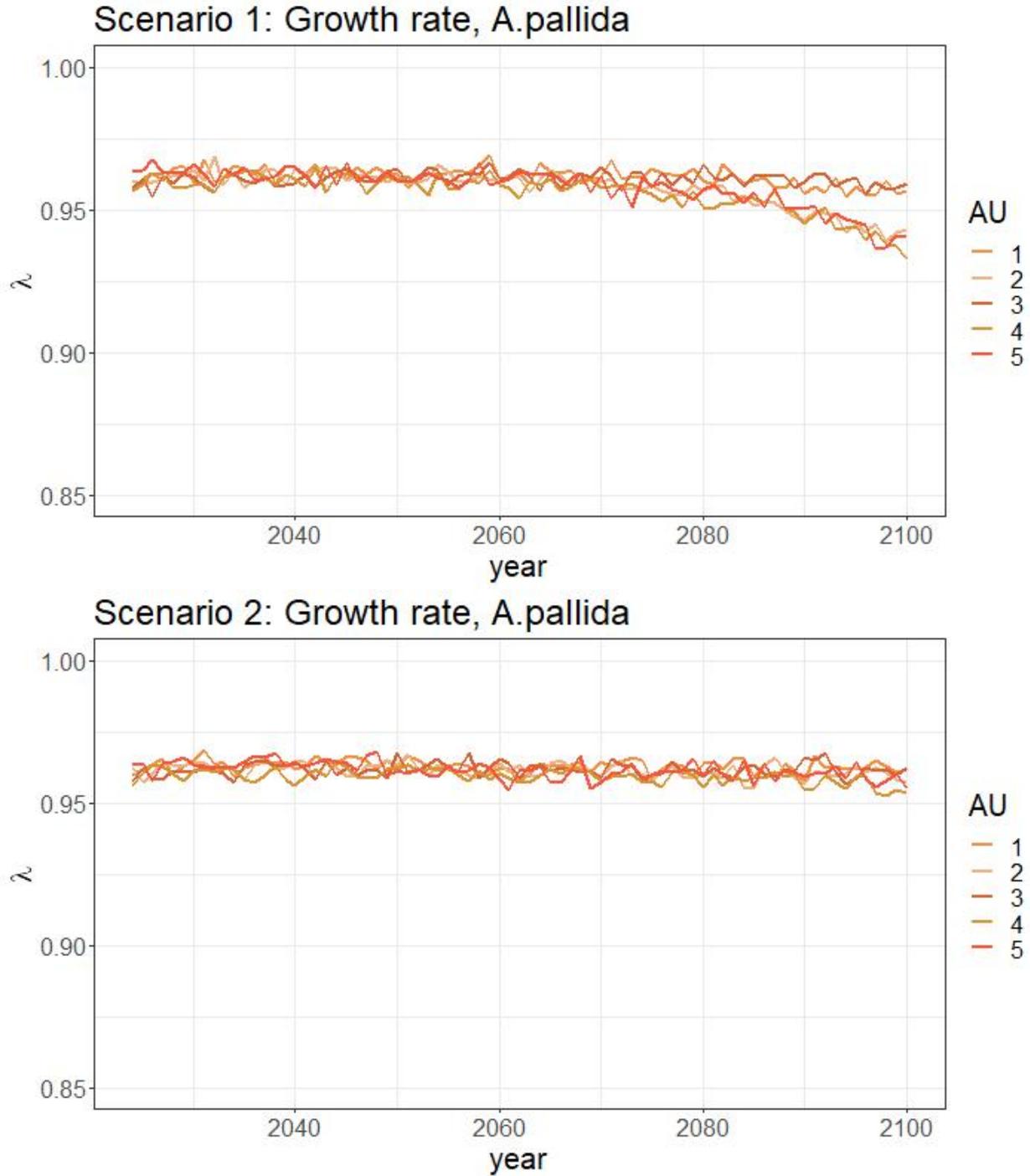


Figure 32. Population growth rate (lambda, λ) overtime for Scenarios 1 (top) and 2 (bottom) for each AU in the *A. pallida* range.

Table 14. Mean and standard error of population growth rate (lambda, λ) in 2050, 2075, and 2100 for Scenario 1 for each AU in the *A. pallida* range.

Year	AU	Lambda	Standard error
2050	1	0.9631	0.0025

2050	2	0.9613	0.0025
2050	3	0.9643	0.0024
2050	4	0.9604	0.0025
2050	5	0.9605	0.0026
2075	1	0.9647	0.0024
2075	2	0.959	0.0028
2075	3	0.9576	0.0026
2075	4	0.9558	0.0027
2075	5	0.958	0.0027
2100	1	0.9565	0.0029
2100	2	0.9433	0.0034
2100	3	0.9592	0.0026
2100	4	0.933	0.0037
2100	5	0.9404	0.0037

Table 15. Mean and standard error of population growth rate (λ) in 2050, 2075, and 2100 for Scenario 2 for each AU in the *A. pallida* range.

Year	AU	Lambda	Standard error
2050	1	0.9626	0.0024
2050	2	0.9634	0.0025
2050	3	0.9675	0.0022
2050	4	0.9621	0.0026
2050	5	0.9614	0.0026
2075	1	0.9582	0.0027
2075	2	0.958	0.0027
2075	3	0.9599	0.0025
2075	4	0.9577	0.0026
2075	5	0.9598	0.0026
2100	1	0.9617	0.0026
2100	2	0.957	0.0027
2100	3	0.9622	0.0025
2100	4	0.9533	0.0027
2100	5	0.9552	0.0027

Table 16. Change in probability of extinction as compared to a baseline for each threat sensitivity scenario for each AU in the *A. pallida* range.

AU	Drought	Bullfrog	Habitat loss
1	0.052	0.044	0
2	0.16	0.021	0
3	0.016	0.033	0

4	0.183	0.017	0
5	0.185	0.014	0

Supplemental Table 1. Sources of data and information provided to us and the U.S. Fish and Wildlife Service, listed in alphabetical order.

Source
Academic researchers
Bureau of Land Management (BLM)
California Department of Fish and Wildlife (CDFW)
California Natural Diversity Database (CNDDDB)
City of Eugene, Oregon
Clean Water Services
Department of Defense (DoD)
East Bay Regional Park District (EBRPD)
Green Diamond Resource Company
Independent researchers
Marin County Parks, California
National Park Service (NPS)
Nevada Department of Wildlife (NDOW)
North American Herpetological Education and Research Project (NAHERP)
Oregon Department of Fish and Wildlife (ODFW)
Portland Parks and Recreation
Riverside County, California
Tualatin Hills Park and Recreation District
U.S. Geological Survey (USGS)
Washington Department of Fish and Wildlife (WDFW)

*****Disclaimer*****

Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the U.S. Government. Empirical data used in this assessment were provided to us by the Oregon Department of Fish and Wildlife and the U.S. Fish and Wildlife Service. Access to those data can be sought through each agency.

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