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A PETITION TO THE U.S. FISH AND WILDLIFE SERVICE
TO PLACE THE CALIFORNIA RED-LEGGED FROG (*Rana aurora draytonii*)
AND THE WESTERN POND TURTLE (*Clemmys marmorata*)
ON THE LIST OF ENDANGERED AND THREATENED WILDLIFE AND PLANTS

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This petition presents the data and interpretation to support listing the California red-legged frog (*Rana aurora draytonii*) and the western pond turtle (*Clemmys marmorata*) as threatened and endangered by the U.S. Fish and Wildlife Service. These two taxa are presented in a joint petition because they overlap significantly in their utilization of habitat. Although incompletely understood, the geographic range of the California red-legged frog is almost entirely contained within that of the western pond turtle (e.g., compare maps 48 and 59 in Stebbins [1985]). Both taxa are known to have occurred in most lowland, low-gradient aquatic systems throughout the region in which their geographic ranges overlap and the two taxa have frequently been recorded syntopically from slackwater habitats within this region (e.g., Holland 1985, Baldwin and Stanford 1987). Due to the extent of geographic and ecological overlap, important similarities exist among the array of factors that have impacted each taxon in the slackwater habitats in which they occur. These similarities will become apparent in the biological synopses of each taxon detailed below.

THE CALIFORNIA RED-LEGGED FROG (*Rana aurora draytonii*)BIOLOGICAL SYNOPSIS

This taxon was historically the common, large ranid frog over most of lowland California (Storer 1925, Slevin 1928, Stebbins 1985; Figure 1). Prior to 1900, it was abundant enough to support commercial harvests, mostly for food (Chamberlain 1898), that lasted at least 25 years (Jennings and Hayes 1985). The largest of the native ranid frogs that occur in California (Wright and Wright 1949), *Rana a. draytonii* is one of two taxa (the other being the northern red-legged frog, *R. a. aurora*) which, as their common names indicate, possess varying degrees of red pigmentation on their hindlimbs, and for which the systematic relationships remain incompletely understood (Hayes and Krempels 1986; see also Hayes and Miyamoto 1984; Green 1985a, 1986).

The name *R. a. draytonii* is currently applied to the population system of frogs with a large rather than small adult body size (males and females >100 mm SUL: Hayes and Miyamoto 1984), where males have paired vocal sacs and call in air (Hayes and Krempels 1986) rather than underwater (Licht 1969a, 1969b; Calef 1973; see also Storm 1960), and females oviposit on a vegetation brace so the egg mass is in contact with the water surface rather than typically being submerged (Hayes and Miyamoto 1984). Limited data also indicate that the population system of frogs with this suite of characteristics display some genetic differentiation from their northern counterparts (Green 1985a, 1986). Allowing for minor discrepancies that may be related to the historical, human-assisted translocation of frogs, as currently understood, *R. a. draytonii* historically extended from the vicinity of Pt. Reyes National Seashore, Marin County, California, coastally and from the vicinity of Redding, Shasta County, California, inland southward to the Río Santo Domingo system in northwestern Baja California, México (Jennings and Hayes 1985, Hayes and Krempels 1986). Over the range as currently understood, *R. a. draytonii* was distributed primarily in Pacific slope drainages, but it was also known to occur on the desert slope of four (and possibly five) drainages in southern California (Klauber 1934; Jennings and Hayes, submitted manuscript). *Rana a. draytonii* also occurs at at least two sites in Nevada, but frogs at both localities are the result of introductions (Linsdale 1938, 1940; Stebbins 1966; Green 1985b) as are the records from Santa Cruz Island, Santa Barbara County off the coast of southern California (Jennings 1988a). Work pending by two of us (MPH, MRJ) and David M. Green (McGill University, Montreal, Quebec, CANADA) is designed to characterize more precisely the geographic pattern of genetic differentiation among red-legged frogs so that the relationship between *R. a. draytonii* and *R. a. aurora* can be better understood. That work will allow more precise identification of the northern limits of the geographic distribution of *R. a. draytonii*.

Rana a. draytonii has a unique life history linked to distinctive habitat requirements that gauged on a continuum relative to other frogs, or even other vertebrates, characterizes it as a habitat specialist. *Rana a. draytonii* breeds early (late November-March: Storer 1925, Jennings and Hayes 1989) in the annual Mediterranean climatic cycle of winter rain-summer drought characteristic of its entire geographic range (Barbour and Major 1977) probably because of a low embryonic critical thermal maximum (CTM) that restricts it to depositing only a single clutch of eggs during the interval when water temperatures will be cool enough to ensure embryonic survival (the related *R. a. aurora* has the lowest embryonic CTM [21°C: Licht 1971] of any North American ranid frog; see Hayes and Jennings 1986). The season of activity for *Rana a. draytonii* appears to vary with the local climate (Storer 1925); individuals from coastal populations, which rarely experience low temperature extremes because of the moderating maritime effect, are rarely inactive (pers. observ.), whereas individuals from inland sites may hibernate for longer intervals (see Dixon in Storer 1925). Adult frogs are largely nocturnal (Storer 1925, Klauber 1932), typically very wary (pers. observ.), and are closely tied to dense riparian vegetation

(Jennings 1988b, Hayes and Jennings 1988), all features that make them difficult to observe. These are probably the reasons that the biology of the taxon has sometimes been misinterpreted (see Hayes and Jennings 1988) and that ecological studies of *Rana a. draytonii* have, until relatively recently, been limited. Males appear at breeding sites from two to four weeks before females (Storer 1925). At breeding sites, males frequently call in small, mobile groups of 2-7 individuals or rarely, individually (pers. observ.). Females individually move toward a male or male calling group, gain amplexus with a male, move to the oviposition site, and attach a egg mass that contains ca. 2,000-5,000 moderate-sized (2.0-2.8 mm in diameter), dark reddish brown eggs to an emergent vegetation brace (Storer 1925; pers. observ.). The vegetation brace on which eggs are laid is typically a vertical emergent stem; bulrushes (*Scirpus* spp.) and cattails (*Typha* spp.) are often used for oviposition, but *R. a. draytonii* may use almost any vertical emergent stem strong enough to attach eggs to (see Jennings and Hayes 1989). Embryonic development to hatching requires 6-14 days (Jennings 1988b; pers. observ.), the variation likely dependent mostly on temperature (see Licht 1971). Mortality from predation during the pre-hatching stages of development is infrequent (pers. observ.; see also Licht 1974), but pre-hatching stages are intolerant of even moderately brackish water; chronic exposure to salinity levels of >4.5 ‰ causes 100% mortality (Jennings and Hayes 1989). Larvae are thought to be algal grazers, but the foraging ecology of *R. a. draytonii* larvae is unknown. Larvae are rarely seen in the field because they are usually concealed in submergent vegetation or organic debris (pers. observ.). Larvae, which are not known to overwinter, typically metamorphose between July and September, 3.5-7 months after being laid as eggs (Storer 1925, Wright and Wright 1949, Jennings and Hayes 1989; pers. observ.). Most mortality probably occurs between hatching and metamorphosis (see Calef 1973, Licht 1974) because in those populations where comparison of an estimate of egg numbers to the number of metamorphs that appear the following summer or fall has been possible, the number of metamorphs is consistently $<1\%$ of the estimated number of eggs laid (Hayes and Jennings, unpubl. data). Postmetamorphs can grow rapidly, and sexually maturity can be attained at two years of age by males and three years of age by females (Jennings and Hayes 1985), but both sexes probably do not reproduce until three and four years of age, respectively (pers. observ.). Females attain a significantly larger body size than males (138 mm vs. 116 mm SUL; Hayes and Miyamoto 1984). No data on longevity are available from the study of marked individuals, but California red-legged frogs are suspected of being long-lived if they reach reproductive maturity. Estimates of minimum longevity of adults extrapolated from growth rates in a central California population was 8 years for males and 10 years for females (Hayes and Jennings, unpubl. data).

Unlike the related northern red-legged frog, adult California red-legged frogs do not appear to move out of riparian zones into adjacent upland forests. However, *R. a. draytonii* is known to make pronounced seasonal movements within aquatic and riparian habitats that appear to be related to the reproductive requirements of adults and seasonal changes that likely influence habitat quality (Jennings and Hayes 1989). In particular, adult *R. a. draytonii* move from breeding sites to the foraging habitat linked to dense riparian vegetation occupied in summer and early fall, and return to the vicinity of breeding sites in late fall or early winter. During low temperature ($<4^{\circ}\text{C}$) intervals and periods of high water flow, California red-legged frogs are rarely observed (S. Sweet, pers. comm; pers. observ.). Low temperature intervals and periods of high water flow usually coincide, so it is not clear whether one or both of these factors are a cue for frogs. Where California red-legged frogs go during this interval is poorly understood, but SCUBA gear-assisted surveys have revealed a few individuals concealed in pockets or animal burrows beneath undercut banks that are stabilized by shrubby willow (*Salix* spp.) growth (pers. observ.).

Postmetamorphic California red-legged frogs have a highly variable animal food diet (Hayes and Tennant 1986), with much variation seemingly related to the ability of different-sized frogs to handle different-sized prey. Generally, postmetamorphic *R. a.*

draytonii seem to take any prey available they can subdue that are not distasteful, with larger frogs able to take both larger prey and a greater range of prey sizes. Amphibians and small mammals may contribute significantly to the diet of adult and subadult *R. a. draytonii* (Hayes and Tennant 1986; see also Arnold and Halliday 1986, Baldwin and Stanford 1987). Adult frogs appear to use vibrations transmitted along willow branch runways that lie near water level to detect approaching small mammal prey (see Hayes and Tennant 1986; pers. observ.).

Few data are available regarding the predators of red-legged frogs. Judging from the number of dorsal spear-like wounds observed on frogs (pers. observ.), adult California red-legged frogs may be particularly wary in part because they are susceptible to being preyed upon by wading birds. Based on the behavior and patterns of habitat utilization of different wading birds, bitterns (*Botaurus lentiginosus*) and black-crowned night herons (*Nycticorax nycticorax*) are the wading birds most likely to be significant predators on adult California red-legged frogs (Jennings and Hayes 1989). Adult *R. a. draytonii* also seem to use vibrations transmitted along the willow branches or vegetation upon which they are resting to detect the approach of certain predators (e.g., raccoons [*Procyon lotor*], black bears [*Ursus americanus*], humans [*Homo sapiens*]; pers. observ.). Juvenile *R. a. draytonii* (<65 mm SUL) which are much less wary than adults, are frequently active diurnally, and often bask in the warm, surface-water layer associated with floating or submerged vegetation during the daytime (see Hayes and Tennant 1986). Thus, juveniles may be more susceptible to predation by diurnal predators; great blue herons (*Ardea herodias*) and several species of garter snakes (*Thamnophis atratus*, *T. elegans*, *T. hammondi*, and the Federally Endangered *T. sirtalis tetrataenia*) are known to take juvenile California red-legged frogs (Fitch 1940; Fox 1951, 1952; Barry 1978; Wharton et al. 1986; pers. observ.). As with other species of ranid frogs, recent post-metamorphs are probably particularly vulnerable to predation by garter snakes (see Arnold and Wassersug 1978). Little data exist on the responses of the premetamorphic stages of *R. a. draytonii* to various predators, but their infrequent co-occurrence with introduced predatory fishes (Hayes and Jennings 1986, 1988) and their cryptic behavior suggests that are relatively palatable and have a low probability of surviving where efficient aquatic predators, such as fishes, occur.

The aforementioned features of California red-legged frog life history result in a rather specific combination of habitat requirements. Adults require a dense, shrubby or emergent riparian vegetation associated with deep (>0.7 m), still or slow-moving water (Jennings 1988b, Hayes and Jennings 1988). A key element of the dense vegetation layer is that it be contact or within a few centimeters of direct contact with deep water (pers. observ.), presumably because this kind of habitat structure provides better escape and refuge for adult frogs from avian or terrestrial predators. Moreover, the dense riparian vegetation also invariably conceals refuges beneath an undercut bank or a partly submerged, but floating mat (in the case of emergents such as cattails), into which frogs can escape. The minimum water depth requirement probably also enables escape. Although the structural requirements of the habitat of adults are likely related to predation, they probably also reflect thermal requirements because adults stress when exposed to water temperatures at or above 29°C and can die if exposure is chronic (pers. observ.). Juvenile *R. a. draytonii* require at least some areas of open, submergent vegetation for daytime basking. Oviposition sites require some emergent vegetation, which typically has a vertical orientation, to which egg masses may be attached (see Hayes and Miyamoto 1984). Aquatic sites can be permanent ponds or permanent or ephemeral streams, but if an ephemeral stream, the aquatic site must retain surface water as pools somewhere in the stream system year-round (Hayes and Jennings 1988). Aquatic sites should also have a salinity <4.5 ‰ and be free of introduced aquatic predators to ensure survival of embryonic stages (Jennings and Hayes 1989).

STATUS

Collective data from our own field surveys over the past 15 years indicate that California red-legged frogs have sustained a 75% reduction in their geographic range (Figure 1) and large breeding populations of >350 adults are currently known from only three areas: Point Reyes National Seashore, Pescadero Marsh Natural Preserve, and the canals west of the San Francisco International Airport (pers. observ.). Although disappearance of *R. a. draytonii* from a significant portion of its geographic range began over 50 years ago (Jennings and Hayes 1985), well over 50% of the indicated reduction in range appears to have taken place over the last 25 years--especially in the Sierra Nevada foothills and southern California. Evidence for the disappearance of the California red-legged frog is most consistent with four types of human interference (see Soulé 1991): 1) loss of habitat; 2) fragmentation of habitat to produce deleterious area and demographic effects; 3) overexploitation; and 4) the spread of exotic (introduced) species. A fifth major class of human interference, pollution, may have contributed; and a sixth, climate change, has a significant possibility of detrimentally affecting this taxon in the near future. To better understand how this taxon probably reached its current condition, the impacts on *R. a. draytonii* are best couched in a historical context. Overexploitation is the type of human interference that has historical precedence, so evidence for the impact of the aforementioned factors on *R. a. draytonii* is presented beginning with this one.

Exploitation of *R. a. draytonii* for food in California was begun during the period following the gold rush of 1849 (Jennings and Hayes 1984, 1985). The date when significant harvests began is unknown, but by the mid-1870s, it was recognized that in the vicinity of San Francisco, the numbers of this taxon were diminishing (Lockington 1879). Documentation of frog harvest size, begun in 1888, showed a progressive increase up to 1895 (Jennings and Hayes 1984, 1985). Overharvest, driven by a significant commercial demand, was indicated by a sharp reduction in frog harvest size in the late 1890s and by the attempt to compensate for depleted wild stocks of the California red-legged frog by introducing bullfrogs (*Rana catesbeiana*) (Jennings and Hayes 1984, 1985). Nevertheless, a continued demand for frogs and a price that made frogs the most expensive food fishery commodity around the turn of the century resulted in continued exploitation of already depleted populations of California red-legged frogs (Jennings and Hayes 1985). This situation resulted in California red-legged frogs being rare enough throughout much of their geographic range just after the turn of the century that even prominent herpetologists of the era thought that these frogs had either not occurred in significant numbers or were not present over much of their lowland historic range (e.g., see Storer 1925, 1933; Stebbins 1966). Commercial exploitation did not eliminate the California red-legged frog (in fact, they continued to be sporadically harvested by individuals for research in local high schools and universities prior to 1950 [Jennings 1988b]), but it created the condition that ultimately led to their elimination from a significant portion of their geographic range, namely, it reduced lowland populations, particularly those on the floor of the Central Valley, to low levels (Jennings 1988b).

The aforementioned reduction in population levels became especially significant in the context of two types of human interference, habitat loss and introduction of exotic species that ultimately led to a third type of interference, habitat fragmentation. Habitat loss and exotic species introductions, each which began during the interval that California red-legged frogs were extensively exploited for food (Jennings and Hayes 1985, Hayes and Jennings 1986), became particularly significant from the turn of the century and over the next 40 years. These two factors fragmented habitat for *R. a. draytonii* over a large portion of its geographic range by eliminating suitable habitat from the floor of the Central Valley, which isolated the remaining populations in drainages around the periphery of the valley. Exotics of importance were aquatic vertebrates and invertebrates with a high probability of being efficient predators on one or more of the life stages of California red-legged frogs, namely, bullfrogs (Moyle 1973), red swamp crayfish (*Procambarus clarkii*), signal

crayfish (*Pacifastacus leniusculus*), and various species of fishes, but especially basses (*Micropterus* spp.), catfishes (*Ictalurus* spp.), sunfishes (*Lepomis* spp.) and mosquitofish (*Gambusia affinis*) (Hayes and Jennings 1986, 1988). The earliest known introductions of the exotic predatory fishes in California took place well before 1900 and many successful introductions occurred repeatedly until well after the turn of the century (see citations in Hayes and Jennings 1986). Bullfrogs were first introduced to California in 1896 (Jennings and Hayes 1985), but many intentional and later accidental introductions associated with the frog-farming boom after the turn of the century resulted in bullfrogs expanding their range in California at a rate much faster than expected had they simply dispersed from a single focal point of introduction (Hayes and Jennings, unpubl. data). Red swamp and signal crayfish were also introduced into California around the turn of the century, in part as a forage for bullfrogs (Riegel 1959), and largely spread by fishermen using them as bait (Reigel 1959, Baker 1980). Although evidence exists for members of this suite of exotic aquatic predators having detrimentally affected California red-legged frogs, it remains unclear which predator or predators were most important (Hayes and Jennings 1988). Moreover, simultaneous changes in habitat tended to be unfavorable to *R. a. draytonii*, and these same changes also tended to be favorable to the suite of introduced exotic aquatic predators discussed previously. Such changes included reduction or elimination of dense riparian vegetation and reduction in the water depth of pools associated with such vegetation, both of which typically result in increased water temperatures (see Hayes and Jennings 1986, 1988; Jennings 1988b). Such habitat changes were extensive on the floor of the Central Valley between 1900 and 1940 because of canal construction to divert water for agriculture and the reclamation of marshlands that were viewed as mosquito-infested habitat of little value (e.g., see Preston 1981).

Regardless of how the negative effects of habitat change and the introduction of exotic aquatic predators on *R. a. draytonii* may be confounded, the result was the elimination of this taxon from the floor of the Central Valley probably sometime before 1960. The last verifiable record of a California red-legged frog on the valley floor was a subadult found under a house near Lodi in 1957; the circumstances of this record make it likely that it was a waif dispersal following winter high water from a nearby foothill location rather than an individual representing the remnant of a population on the valley floor. To our knowledge, the last records of a reproducing California red-legged frog population on the valley floor are based on several sightings that, although considered reliable, are unverifiable, and these occurred in the vicinity of Gray Lodge National Wildlife Refuge around 1947 (J. Cowan, pers. comm.). The additional report of an adult *R. a. draytonii* taken during the summer of 1981 on the valley floor in the old Kern River channel below the Kern National Wildlife Refuge (N. Euliss, pers. comm.), may also be a valid record but we have been unable to verify it.

Elimination of *R. a. draytonii* from the valley floor was particularly significant because it eliminated lowland populations as a source from which the now isolated foothill populations could depend for immigrants. In 1960, we estimate that isolated populations of *R. a. draytonii* occurred in at least 30 foothill drainage systems bordering the Central Valley. Several factors, frequently acting in concert, seem to be responsible for the elimination of virtually all of these populations over the next 30 years. First, the construction of large reservoirs either directly eliminated (e.g., Lake Oroville, Whiskeytown Reservoir) or further fragmented and isolated remaining populations into smaller areas of habitat (e.g., Don Pedro Reservoir, Lake Berryessa, San Luis Reservoir) (see Hayes and Jennings 1988 for a general discussion). Fragmentation was inevitable because reservoirs were sited at or just below the juncture of several tributaries (often just before the streams started greatly increasing in gradient), and filling of the reservoirs isolated frog populations in upstream tributaries because the intervening habitat of the new reservoir was structurally unsuitable and/or had exotic aquatic predators. This is significant because recolonization of locations within a drainage seem to occur because of immigrants from other nearby populations within that drainage rather than immigration from an

adjacent drainage. Second, continued expansion of the exotic aquatic predator assemblage into foothill streams resulted in a change in habitat utilization by *R. a. draytonii* from its historical use of both ephemeral and perennial streams to the nearly exclusive use of ephemeral streams or isolated springs (Hayes and Jennings 1988). Since ephemeral streams suitable for *R. a. draytonii* are but a subset of the streams in which they might occur if no habitat alteration or introduction of exotic aquatic predators had occurred, this phenomenon further isolated remaining populations and enhanced the chance of extinction from natural random events such as fires, droughts, and floods. Third, additional habitat alteration because of the presence of domestic livestock has occurred at every known historical locality within the Central Valley hydrographic basin where California red-legged frogs have been observed since 1960. The ubiquitous nature of livestock grazing in the western United States has resulted in its identification as the major cause of excessive habitat disturbance in most riparian areas in that region (Mosconi and Hutto 1982, Kauffman and Krueger 1984, Taylor 1986). While it is difficult to determine precisely the relative importance of cattle grazing when compared to other important factors that impact riparian systems, our observations indicate that the impact of even moderate levels of cattle grazing on California red-legged frog habitat is both negative and severe (Jennings 1988b). In particular, grazing cattle remove or trample riparian vegetation (Kauffman and Krueger 1984, Marlow and Pogacnik 1985), an alteration that frequently severely negatively impacts adult California red-legged frogs because dense riparian vegetation is a crucial component of their habitat (see previous discussion of habitat requirements). Cattle also markedly affect the physical characteristics of stream margins because they tend to concentrate there (Colman 1953) and streambanks themselves are more susceptible to trampling than adjacent upland habitat because of greater soil moisture (Marlow and Pogacnik 1985). Trampling often increases soil compaction, contributes to streambank erosion, decreases water quality, fills in pools, and makes stream channels wider, but shallower (Behnke and Raleigh 1978, Bohn and Buckhouse 1985, Kauffman and Krueger 1984, Kauffman et al. 1983). Well-grazed stream margins also lack extensive undercut banks (Platts 1981). For California red-legged frogs, the loss of undercut banks and reduced water levels is particularly critical because refuge plunge pool habitat would be reduced or eliminated and water temperatures would tend to be higher than in habitat not altered in this way, a feature not only detrimental to *R. a. draytonii*, but which also tends to favor a number of introduced exotic aquatic predators (see previous discussion of habitat requirements). Finally, most recently, California experienced four sequential years of severe drought (1986-1990) that severely affected remaining California red-legged frog populations in the Central Valley hydrographic basin. In several thousand hours of field checks of historical California red-legged frog localities on the Sierran slope of the Central Valley, we have observed only one California red-legged frog since before 1985. Many localities in ephemeral streams that before the drought years ponded in late summer and early fall, were entirely dry during our field checks. The ones with water had water levels so low that predators had easy access and would have efficiently removed any frogs that may have been present. Moreover, at many sites, the damaging effects of domestic livestock discussed above were exacerbated by the fact that drought limited or prevented recovery of the riparian vegetation. If the California red-legged frog still exists on the Sierran slope of the Central Valley, it is extraordinarily rare. Our collective data indicate that at this writing, we know of no more than six small drainages in the hydrographic basin of the Central Valley that have some probability of still harboring California red-legged frogs, all of which occur on the Coast Range slope of the San Joaquin valley. The only one of these of which we are certain still harbors California red-legged frogs, the Corral Hollow Ecological Reserve (a State reserve), is threatened by siltation because adjacent and upstream habitat has been nearly denuded by abusive grazing practices and the presence of a State off-road vehicle park (created after the area was designated a reserve), the effects of both of which have been augmented by drought. Of the other five drainages, one (Frank Raines Regional County Park) is also subjected to heavy livestock grazing, off-road vehicle

use, and human disturbance. The rest of the sites are slated as locations for future reservoirs (e.g., Los Vaqueros and Los Banos Grandes).

In several ways, the pattern of disappearance of the California red-legged frog in southern California is similar to that seen in the Central Valley. Differences are largely in the degree to which different factors have played a role and how they played that role rather than an entirely different set of factors being important. In southern California, *R. a. draytonii* was the most common native frog in San Diego, Orange, and parts of Los Angeles, Riverside, and San Bernardino counties (Storer 1925; Slevin 1928; Klauber 1934; Sloan 1964; Jennings and Hayes, submitted manuscript). Significant historic populations of this frog were located in the freshwater marsh portions of coastal lagoons and estuaries, and to a lesser degree, along floodplains at more inland locations. Most of these populations disappeared at a relatively early date because of elimination of wetland habitat due to development of harbors, urbanization, agriculture, stream channelization, large reservoirs, and the introduction of much of the same suite of exotic aquatic predators discussed previously. Virtually all the best inland *R. a. draytonii* habitats in this region are now either under cemented flood control channels (e.g., Arroyo Seco, Los Angeles River), reservoirs (e.g., Lake Silverwood, Lake Piru, Pyramid Lake, Lower Otay Lake), freeways, or cities. Crayfish and bullfrogs were present at several locations in southern California prior to 1930 (Storer 1922, Reigel 1959) and bullfrogs became widespread with the frog-farming boom of the 1930s (Storer 1933). Introduced predatory fishes were also well established by the 1930s (Miller 1961). California red-legged frogs seem to have progressively disappeared in most drainages in an upstream direction, a change corresponding to the slow, progressive expansion of the exotic aquatic fauna directionally from the downstream portions of drainages where most of the latter were introduced coupled with habitat alterations that favored the latter. The rate of extinctions greatly increased after 1960, so that by 1975, *R. a. draytonii* had virtually disappeared from southern California south of Ventura County. Today, *R. a. draytonii* is known from only three small drainage systems in this region: two tributaries of the Santa Clara River and a 2 km reach of stream in the headwaters of the Santa Margarita River system (Hayes and Jennings, unpubl. data).

California red-legged frogs were also once widespread and abundant in the inner Coast Ranges between the Salinas River system and the Central Valley. The combination of habitat alteration largely due to years of livestock grazing, agriculture, water use patterns, and the recent 4-year drought are probably the reason that recent records of California red-legged frogs are lacking for much of this region. Fewer than a dozen post-1985 records of *R. a. draytonii* exist for this area, and all are from the region of Pacheco Pass, Merced and Santa Clara counties, north (Hayes and Jennings, unpubl. data).

The small coastal drainages between Pt. Reyes National Seashore, Marin County and Carpinteria in Santa Barbara County is the only remaining region in California where *R. a. draytonii* can still be found in significant numbers. The coastal drainages with the best remaining populations are usually those in which the riparian habitat has been least altered and that lack exotic aquatic predators. Based on our most recent surveys and discussions with Drs. Samuel S. Sweet (University of California at Santa Barbara) and Paul W. Collins (Santa Barbara Museum of Natural History), California red-legged frogs still occur in over 50% of coastal drainages from which they were historically recorded. Nevertheless, the exotic aquatic fauna continues to expand in this region and several coastal areas are continuing to undergo extensive development. Bullfrogs co-occur with California red-legged frogs in four of the systems that we consider some of the most intact and are established with exotic fishes in at least 12 others that no longer have *R. a. draytonii*. Although bullfrogs may not be able to effectively reproduce in some of these coastal systems, they may continuously leak into the coastal lagoon areas from sites upstream in the same drainages where they can reproduce (Jennings and Hayes 1989). Indications also exist that such non-reproductive bullfrogs may interfere with California red-legged frog reproduction even if bullfrogs do not prey on *R. a. draytonii* directly (Jennings and Hayes

1989). An equally serious concern is the potential effects of drought on reproduction. Decreased flows during drought years coupled with agricultural demands and releases from upstream community wastewater treatment facilities, can result in increased salinities of water at oviposition sites sufficient to kill nearly the entire complement of one year's reproduction (Jennings and Hayes 1989). Since increased salinities in coastal lagoons were a widespread phenomenon during the decreased flows of the 1986-1990 drought (C. Swift, K. Worcester, pers. comm.) and significant populations of California red-legged frogs are localized in the coastal lagoon portion of these coastal drainages, drought conditions have the potential to eliminate a significant proportion of the reproductive effort of California red-legged frogs over the region where the only significant populations of this taxon remain. Prolonged drought conditions that last several years have the potential for actually extirpating populations in the region as adult *R. a. draytonii* are unable to discern aquatic habitats where salinity levels are lethal to eggs and developing embryos (Jennings and Hayes 1989). The recent trends in increased temperature and decreased rainfall, linked by some to global climate changes (Kerr 1990, Schlesinger et al. 1990), suggest that an increased probability exists that drought conditions will be more frequent in the immediate future. This trend, for several reasons already noted, will result in conditions progressively less favorable for the California red-legged frog, and progressively more favorable for the introduced exotic aquatic fauna.

The California red-legged frog (*R. a. draytonii*) is estimated to have disappeared from over 99% of the inland and southern California localities within its historic range and at least 75% of all localities within its entire historic range despite being protected from exploitation since 1971 by the California Department of Fish and Game (Bury and Stewart 1973). Many of the factors believed responsible for the extirpation of *R. a. draytonii* from localities within its historic range still affect the large majority of populations that remain, and future conditions are anticipated to become even less favorable for this taxon. As a result, pursuant to the definition provided in the Federal Endangered Species Act of 1973 as amended, existing data strongly support listing the California red-legged frog (*R. a. draytonii*) as:

Threatened: California - coastal slope drainages from Marin County south Santa Barbara County

Endangered: California - throughout the rest of its range within the state; the entire Central Valley hydrographic basin [including the Sacramento-San Joaquin, Kings, Kaweah, and Kern River systems] and Ventura County south to the Mexican border).

THE WESTERN POND TURTLE (*Clemmys marmorata*)

BIOLOGICAL SYNOPSIS

The western pond turtle (*Clemmys marmorata*; hereafter WPT) is the only freshwater turtle native to most of the west coast of temperate North America (Stebbins 1985). Historically, the WPT was very abundant over large parts of its range, and it was commercially harvested extensively from at least the 1870s (Lockington 1879, Brown 1940) to the 1920s (Storer 1930). Currently, two subspecies are recognized (Seeliger 1945): the northwestern pond turtle (*Clemmys m. marmorata*), which occurs from the vicinity of the American River (California) northward to the lower Columbia River (Oregon-Washington), and the southwestern pond turtle (*Clemmys m. pallida*), found in coastal drainages from the vicinity of Monterey (California) south to northwestern Baja California Norte in the vicinity of the Sierra San Pedro Martir (Mexico). Turtles that occur in Central Valley from south of the American River to the vicinity of Tejon Pass were described as representing an area of

intergradation of the two subspecies (Seeliger 1945). Outlying populations of *C. m. marmorata* are recorded from the vicinity of Puget Sound (Washington), Grant County (Oregon), and the Carson and Truckee Rivers (Nevada). The Grant County turtles were probably introduced in the 1970s (Nussbaum et al. 1983). Outlying populations of *C. m. pallida* occur in the Mojave River (California). A generalized map of the the range of the WPT is provided in Stebbins (1985).

Low fecundity, low hatchling and juvenile survivorship, high adult survivorship, and a potentially long life-span characterizes the unique life history of *Clemmys marmorata*. Size and age at first reproduction vary. Individuals from the southern part of its geographic range (south of the Transverse Ranges in California) may reproduce at body sizes (carapace length (CL)) as small as 110-115 mm CL and seven years of age, but at this latitude, most females first reproduce at 120+ mm CL and 8-9 years of age (D. Holland, unpubl. data). A parallel situation occurs on the central California coast, where the smallest recorded gravid female was 122 mm CL. In northern California, the smallest female recorded with eggs was 130 mm CL and probably 10-12 years old. In central Oregon, the smallest gravid female recorded was 138 mm CL and probably 12-14 years old. Examination of the reproductive status of 1735 females from 1987-1991 indicates that most females in a population probably oviposit biennially, although a few females may deposit eggs every year. Clutch size ranges from 1-13 (mean = 5.7) and is positively correlated with CL ($r = 0.754$). Prior to nesting, females leave the watercourse in late afternoon or early evening and typically move some distance overland to the nest location. All known nest sites are beyond the often densely vegetated riparian zone adjacent to the watercourses that turtles inhabit, and the distance between the watercourse and nest location ranges from a minimum of 16 m to a maximum of 402 m (Storer 1930; Hayes and Holland, unpubl. data). Females are sensitive to disturbance during their overland movements to nest sites, and may excavate one or more nests in which no eggs are laid (Holland 1991, Rathbun et al., in press). Incubation time in captivity is 73-80 days (Feldman 1982) and the few records from natural nests (Holland 1991) were incubated from 95-106 days. Hatchling western pond turtles range from 23 mm to 31 mm CL, and weigh 1.5-5 g. Most hatchlings probably overwinter in the nest (Holland 1985a, 1991) and emerge from the nest and move to the watercourse in early spring, usually March or April. Hatchlings and first-year juveniles have low survivorship, averaging about 8-12% (Holland, unpubl. data). Growth is rapid, and hatchlings double their size in the first year of growth. By their fourth year, juvenile WPTs typically reach 80-90 mm. After this point, rates of growth vary both within and among populations. Secondary sexual characteristics usually become apparent at 110-120 mm CL in both males and females. Once this size is reached, survivorship seems high and the adults average a 3-5% turnover rate/year (Holland, unpubl. data).

Western pond turtles are active year-round in the southern part of the range and along the central coast of California (Holland 1985a). In the Central Valley and areas north, activity typically commences in March and peaks in June-July, decreases gradually in August, increases briefly in September, and usually terminates by November. At least some of the population apparently leaves the watercourse in October-November and moves into adjacent upland habitats. Turtles have been found overwintering several hundred meters from the watercourse.

Western pond turtles spend considerable amounts of time engaged in thermoregulatory behavior (Bury 1972, Bury and Wolfheim 1973, Holland 1985a). Emergent basking typically begins shortly after emergence from overwintering sites. Animals utilize logs, rocks, emergent vegetation, mud or sand banks, or human-generated debris as basking sites. Aggressive interactions on these sites are common (Bury 1972, Bury and Wolfheim 1973, Holland 1985a), and as many as 70+ turtles have been observed on a single site. Turtles also engage in "aquatic" basking, utilizing thermal microenvironments within the aquatic habitat to engage in thermoregulatory behavior (Holland 1985a).

Clemmys marmorata can be classified as a dietary generalist with most of the diet being comprised of small aquatic invertebrates (Holland 1985a, 1985b; Bury 1986). However,

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some small vertebrates (fish and anuran larvae) may be taken (Bury 1986) and carrion is frequently utilized (Holland 1985a, 1989; Bury 1986). Plant material typically makes up a small percentage of the diet. WPTs typically forage throughout the water column, utilizing both vision and scent to locate prey items. Most foraging appears to occur during daylight hours, however turtles may be active throughout the night during the summer months. Turtles will also take trolled, floating or bottom-set baits used by fishermen (Holland, pers. obs.).

Many species are known to prey on western pond turtles, including black bear (*Euarctos americanus*), coyote (*Canis latrans*), grey fox (*Urocyon cinereoargenteus*), domestic and feral dogs (*Canis familiaris*), river otter (*Lutra canadensis*), mink (*Mustela vison*), raccoon (*Procyon lotor*), bald eagle (*Haliaeetus leucocephalus*), bullfrog (*Rana catesbeiana*) and largemouth bass (*Micropterus salmoides*) (Holland 1985a; Holland, unpubl. data). Suspected predators include striped skunk, opossum, beaver, nutria, bobcat, feral pig, osprey, great blue heron, red-shouldered hawk, black-crowned night heron, raven, common crow, giant garter snake, two-striped garter snake, California red-legged frog, striped bass, white bass, smallmouth bass, the larger species of catfish, rainbow trout (steelhead), and giant water bugs. Few species can prey on adult turtles in the water (e.g., mink, river otters), but turtles are particularly vulnerable on land. Raccoons and possibly skunks are major nest predators; raccoons also eat adults. Adult female turtles typically show evidence of attempted predation (shell scarring, missing limbs) at a level 6-7 times that of males. Mutilations are probably incurred during overland nesting movements. Loss of a hind limb may prevent a turtle from successfully excavating a nest.

The WPT is a habitat generalist, occurring in a wide range of both permanent and intermittent aquatic environments. This species historically occurred from sea level to 5000 feet, although turtles are scarce anywhere above 4500 feet. Two records above 6000 feet exist which may represent introductions. Turtles occur in brackish-water habitats along the California coast, and can tolerate prolonged immersion in sea water (Holland, unpubl. data). Historically, WPTs occurred in most watercourses throughout its range. Optimal habitat was the series of warm, shallow lakes and the extensive slough systems that formerly covered much of the floor of the Central Valley (Elliot 1883, Brown 1940, Harding 1960, Preston 1981). In the few remaining microhabitats that approximate these conditions, the WPT typically achieves densities in excess of 1,000 animals per hectare of water surface, and densities as high as 3,700 animals/ha are known. Given these densities, this species was historically probably the dominant element of the vertebrate biomass in many aquatic ecosystems on the west coast of North America. Currently, the primary habitat for this species are the small-to-medium sized streams in foothill areas. Turtles are patchily distributed along some of the larger rivers (e.g., San Joaquin, Sacramento, Klamath and Umpqua). Turtles are also found in some small farm ponds and other modified watercourses such as canals and reservoirs. Although WPTs are habitat generalists, hatchlings and juveniles have relatively specialized habitat requirements, and the microhabitats used by these age groups are locally frequently very limited, and particularly susceptible to disturbance.

STATUS

From 1976 to the present, Dan C. Holland has conducted research on the WPT. Between 1976 and 1981, he made over 150 surveys of the distribution and abundance of the WPT in central California. Over 100 of these surveys were made at 4 sites, but over 40 additional different sites were also surveyed. Since 1981, Holland has conducted over 700 surveys at over 250 sites throughout the range of the WPT. The areas surveyed included many localities where WPTs were historically present based on museum specimens or literature records. Overall, turtles were present at 65-70% of the localities surveyed. In addition, over 100 additional sites were examined that based on their habitat characteristics, turtles were likely to have historically occupied, but for which no historical records of their

presence were available. Habitat alteration at these sites precluded the presence of WPTs. Although the WPT still occurs in most of its former geographic range, it has been completely or ecologically extirpated from many areas. The total population size of the WPT has declined significantly from the levels present prior to the advent of European man on the west coast of North America, and the few remaining areas that hold moderate-to-large viable populations face several threats. Based on Soulé (1991), the major types of threats that historically and currently affect WPTs include (1) loss of habitat; (2) habitat fragmentation with concomitant effects on population viability; (3) overexploitation; and (4) the spread of exotic species. Two other major categories of threats, pollution and long-term climatic change, may also pose significant problems. The overview that follows describes the impacts of each of these factors on the WPT.

Several native American cultures (e.g., the Modoc: Howe 1968) have used WPTs as food. Based on midden data, the southern Yokuts appear to have used WPTs extensively (Porcasi and Dillon 1991). Apparent lack of mention of this use among Yokuts in the Tulare Lake Basin (Latta 1949) is probably because his data were collected after the turn of the century, postdating the period when extensive turtle populations were present on the floor of the Central Valley. Commercial exploitation probably began shortly after the California gold rush in 1849, but the first published record is that of Lockington (1879). Demand resulting from the restaurant trade had already depleted populations of California red-legged frogs (*Rana aurora draytonii*) and WPTs in the vicinity of San Francisco by this date, and market hunters for both species were moving as far afield as the area of Visalia (central California) to supply the market. During this period, extensive commercial operations were conducted on Tulare Lake; the schooner "Water Witch" was used to support a fishing and turtling operation there from the late 1870s to early 1880s. Harvest figures are scarce, but it is known (Elliot 1883) that one operation shipped at least 2,160 turtles in one year to the markets in San Francisco. Other information (M. Jennings, pers. comm.) indicates that the total catch was significantly higher than this. Smith (1895) estimated that 18,000 turtles were offered for sale annually in San Francisco markets. Based on incidental commercial collecting records sporadically maintained by the California Department of Fish and Game, commercial trade continued until at least the late 1920s with a minimum of several hundred to several thousand turtles being reported as taken each year. Commercial harvest of WPTs for food undoubtedly played a significant role in initial population reductions in this species. The impact was probably most significant on populations where habitat conditions facilitated mass harvest, such as in the lakes on the floor of the San Joaquin valley (Kern, Buena Vista and Tulare) and in slough and backwater habitats throughout the San Joaquin and Sacramento valleys. However, records from the California Department of Fish and Game also indicate that substantial collections of turtles occurred in many areas of California that lacked the types of habitats previously mentioned in the early part of the 20th century. Illegal commercial harvest of this species for food continues today, particularly among recent immigrants whose traditional culinary repertoire included turtles, and may have a significant impact on remaining populations, particularly in southern and central California (H. DeLisle, S. Sweet, pers. comm.).

Concurrent with the initial population reductions caused by commercial harvesting, WPTs also sustained two other important impacts, habitat alteration and the introduction of exotic species. Widespread alteration of WPT habitat was already well underway during the 1880s as extensive "reclamation" of "swamp and overflow lands" became commonplace (Elliot 1883, Brown 1940, Harding 1960, Preston 1981). Extensive water diversion efforts for agricultural and other purposes led to the alteration, reduction, or elimination of WPT habitat throughout much of its range. Among the most significant losses were in the Central Valley of California and the Klamath and Willamette drainages in Oregon. Additional habitat losses were incurred through the construction of dams and the creation of reservoirs along many watercourses. Creation of these impoundments had several negative effects, including: 1) alteration of historical patterns of water flow within a drainage and negative impacts on downstream habitats; 2) direct elimination of WPT habitat in the area of the dam

and reservoir; 3) creation of habitat suitable for the maintenance and spread of exotic species which have deleterious effects on WPTs; 4) fragmentation of existing WPT populations and interference with normal movement patterns; and 5) creation of barriers to normal dispersal and concomitant decreases in the probability of re-establishment of extirpated populations. These factors have all had moderate-to-severe negative impacts on populations of WPTs. Surveys of all reservoirs in the southern Sierra Nevada from south of the Merced River from 1981-1991 and selected northern Sierra reservoirs from 1987-1991 indicated that WPTs are occasionally observed in these systems, but no viable populations are known to occur in these reservoirs. Parallel situations have been identified along river systems in southern California; central coastal California; northwestern California; the Klamath, Rogue and Willamette drainages in Oregon; and along the Columbia drainage in both Oregon and Washington.

Other forms of habitat alteration have affected and continue to affect WPT populations. Widespread channelization of watercourses for flood control and water diversion have eliminated considerable WPT habitat along small-to-moderate-sized watercourses, especially in the Central Valley of California and southern California (see Harding 1960). Urbanization encouraged by water diversion efforts has also played a major role in the elimination of WPT habitats, particularly in southern California. Grazing may also play an important role in modifying WPT habitats. California has a history of extensive grazing (Burcham 1957), which is known to adversely affect the structure of aquatic and riparian habitats (Kauffman and Krueger 1984, Marlow and Pogacnik 1985). In particular, trampling or consumption of emergent vegetation along watercourses may adversely affect survivorship of WPTs as this decreases or eliminates microhabitats hatchlings and juveniles use. Cattle also trample streambanks and modify or eliminate undercut banks (Platts 1981), areas typically utilized as refugia by turtles, thus increasing their susceptibility to predation.

Another form of habitat alteration with significant consequences for WPT populations has been the five years of drought suffered by the west coast from 1986-1991. Surveys on several turtle populations in southern and central California during this time period have revealed an interesting pattern. Turtles were moderately abundant during 1987-1988 and capture rates (quantified as individuals captured per hour) were equivalent to many northern California watercourses presumably less affected by the drought. Increased drying of the watercourses concentrated turtles in smaller and smaller areas of the watercourse, such that by 1989-1990, capture rates were very high and turtles might be found in only a few small pools within several miles of river or stream channel. All turtles in these situations were stressed with no apparent body fat reserves. Increased mortality was anticipated and subsequently observed during 1990-1991; some populations were completely extirpated, whereas others displayed 66% (minimum) to over 90% reductions in size. The drought also exacerbated population fragmentation and problems associated with other forms of disturbance.

A less obvious form of habitat alteration, but one which is crucial to the viability of WPT populations is alteration or destruction of nesting habitat. WPTs are known to make extensive overland movements to nests locations; distances of up to 0.25 miles are known (Storer 1930) and other information (Holland 1991) indicates that distances of over 150 m from the watercourse are commonplace. Alteration of upland areas or other habitats adjacent to watercourses in which WPTs occur is frequent, and in many of these sites, WPT populations are extremely adult-biased, indicating limited or lack of recruitment. In many of these situations, lack of recruitment is not easily assigned to a single cause because introduced predators are frequently also present. However, sufficient data indicate that patterns of alteration of nesting habitat is a major cause for concern.

Introduced predators have also had a major impact on WPT populations, the most significant of which are bullfrogs and bass. Predatory fishes, such as bass, and bullfrogs are postulated to impact native frogs (Jennings and Hayes 1985, Hayes and Jennings 1986), and parallel effects on WPTs are known or postulated. Bullfrogs are known predators on WPTs (Moyle 1973; H. Basey, T. Papenfuss, pers. comm.; Holland, unpubl. data,), as

are bass (Holland, pers. obs.). Bullfrogs are particularly significant because of their catholic diet, their widespread occurrence and large population sizes within the range of the WPT, and their habit of foraging in microhabitats favored by hatchling and juvenile WPTs (Bury and Whelan 1985). A negative correlation exists between the occurrence of bullfrogs and the presence of hatchling and juvenile WPTs (Holland, unpubl. data). A parallel situation frequently exists between bass and hatchling WPTs, although coexistence may be possible where habitat complexity provides a refuge for hatchlings. More subtle in their action, but also of concern are the effects of other introduced fishes, specifically carp (*Cyprinus carpio*) and goldfish (*Carassius auratus*), which may impact WPT populations through elimination or modification of emergent vegetation; and sunfishes (*Lepomis* spp. and *Pomoxis* spp.), well-known for stunting their own populations (e.g., Swingle and Smith 1940), which may deplete available food for young turtles (see Holland 1985b), depressing their rates of growth and increasing the time interval they are vulnerable to a suite of predators. Larger species of catfishes (*Ictalurus* spp.) may also prey on turtles or compete for the available prey base.

Incidental human predation may also affect WPT populations. Accidental capture, either by hook and line methods or by nets, has a high probability of leading to the death of the animals involved (Mader 1988; Hayes, Holland, Jennings, pers. observ.). Surveys in California (1981-1991), Oregon (1987-1991), and Washington (1990) indicate that this may be a significant source of mortality. Mortality from vehicles may also be a significant problem in many areas because roads frequently separate aquatic foraging and upland nesting habitats. Loss of nesting females, which represent a small, but significant portion of any population, may be a major factor in the decline of the species in some areas. Off-road vehicles pose a significant threat to WPT populations in many areas due to direct loss through crushing or illegal collection and/or alteration of the aquatic or nesting habitat. In most areas impacted by off-road vehicles, WPT populations are small and exist in marginal habitats, thus, are less likely to be able to tolerate this type of disturbance. Furthermore, the isolation of such sites essentially eliminates the possibility of re-establishment of these populations through immigration from adjacent habitats. In many areas, boat traffic may pose a direct or indirect threat through disturbance of normal activity, alteration of hatchling/juvenile microhabitat, vandalism (shooting) or illegal collecting, and/or increased fishing access.

Disease may also play a role in the decline of WPT populations. Evidence from a study in Washington in 1990 (Holland 1991) indicates that WPTs are susceptible to an Upper Respiratory Disease syndrome similar to that noted in the desert tortoise (*Gopherus agassizi*). The precise causal agent is unknown, but may be a virus or mycoplasma. Dead and moribund turtles were first observed in May and continued to be noted throughout the study. Thirty-eight turtles were collected during the epidemic and provided with extensive veterinary care through the efforts of the Washington Department of Wildlife, the Woodland Park Zoo and the Center for Wildlife Conservation. Despite the expenditure of over \$60,000 and extraordinary efforts by staff and consulting veterinarians, over 50% of the turtles in this group died. Known mortality in this population in the summer of 1990 was over 40%. To date, this remains the only documented example of this syndrome in WPTs. However, at least one animal from the Willamette drainage during the same period displayed lung lesions similar to those noted in turtles from the Washington population. The sole positive aspect of this epidemic was that it occurred in an isolated population under intensive study. If this disease became established in an area where turtles are continuously distributed (e.g., the Trinity River system), it is unlikely that any effective measures could prevent massive mortality.

The results of the additive and sometimes synergistic effects of the various factors noted above has been a substantial decrease in the total population size of the WPT. Although turtles are distributed throughout most of the historic geographic range of the species, numerous localized extirpations have occurred. The WPT has been completely extirpated from the type locality of the species, Puget Sound. Localized extirpations have also

occurred over most of its range within the Columbia River drainage; in many areas of the San Joaquin Valley; and over the metropolitan areas of San Francisco, Los Angeles, and San Diego. Populations in the remaining portions of the range are fragmented to varying degrees. In Washington, WPTs are known from only two localities along the Columbia River, and the total number of individuals at both these sites combined probably does not exceed 110 animals. One meta-population, probably not numbering over 200, occurs over about 4 square miles on the Oregon side of the Columbia River drainage east of the Cascades. Over 200 sites with potentially suitable habitat were surveyed in the Willamette River drainage during 1991, and WPT populations with indications of significant recruitment were found at only three sites. Eighty-four percent of all turtles observed in the Willamette surveys were adults. Western pond turtles were observed at 62% of 41 sites surveyed in the Umpqua River drainage, but only two of the surveyed sites appeared to have viable populations. The two largest WPT populations encountered in the lower Umpqua River system consisted entirely of adults. Over the same time period, WPTs were found at 23% of ca. 30 sites surveyed in the Rogue River drainage. Only one moderately large WPT population was noted during the Rogue River surveys, and that population consisted of mostly adults. In the Klamath River system, WPTs were observed at 33% of 41 sites with potentially suitable habitat. Only one population that could be considered viable was found in the extreme upper end of this drainage. Adults made up 89% of all WPTs observed in the Klamath River surveys. As noted above, although turtles were found at a significant number of sites, the size (age) structure of these populations indicates that they are comprised primarily of adults, indicating little or no recruitment is taking place.

Surveys conducted in Nevada in 1987-88 indicate that WPTs are known mostly from the Carson River drainage, although a few animals may persist in the Truckee River. Adults comprised over 90% of the animals observed, and the number of WPTs in the state probably does exceed a few hundred animals.

Surveys of drainage systems in California reveal several patterns, many similar to those observed in Oregon and Nevada. Turtle populations in the Klamath River drainage and its tributaries vary from a small population size which is heavy adult-biased to the largest remaining WPT populations known. The latter occurs in a very small portion of the drainage (one stream) and is subject to a variety of threats. Other populations in this area have been impacted by dam construction, introduced predators and other factors. In the Central Valley, most WPT populations have undergone dramatic declines in size and have become increasingly fragmented. Conservative extrapolation from knowledge of the historical distribution of suitable habitat, and historical WPT densities based upon historical anecdotes, current densities in suitable habitat, and the size and status and of remaining populations indicates that turtle populations have been reduced by 99.9% of their historic extent in the southern portion of the San Joaquin Valley, and by probably 95-99% in the remainder of this area. At present, only one, perhaps viable population is known from the area south of Tulare County. Currently, the Pajaro-Salinas system contains relatively few WPT populations. The few that exist are severely reduced, a number having been recently extirpated by the effects of drought. Remaining viable populations in this drainage are moderate in size, highly disjunct, and exist in very restricted habitats. The central coast region from Monterey south to the Santa Clara River contains several moderate-sized populations. Some of these populations seem stable, but they face a growing number of threats and their stability cannot be guaranteed long-term. Other populations in this region are beginning to manifest signs of recruitment declines or failures. Many of the populations in this area were also severely affected by the drought with population declines ranging from 66% to 90% or more during 1990-1991. The Mojave River system currently has only one known WPT population restricted to a few hectares of habitat. Surveys in 1989 indicated that the total population in this system probably does not exceed 100 individuals, and may actually be considerably smaller. The disjunct nature of this population effectively precludes *any possibility of natural re-establishment should the population be extirpated*. Brattstrom and Messer (1988) and Holland examined the status of the WPT in southern California.

Brattstrom and Messer noted that south of Ventura County, only 12.8% of 218 sites surveyed held turtles, and only 5 sites (2.2%) held viable populations. Surveys by Holland have verified the general conclusions of this report, south of the Santa Clara River (inclusive), no more than 9 viable WPT populations are known to exist. Most of these were severely reduced during the drought with losses of up to 66% being recorded.

In summary, the aforementioned data and that provided in the enclosed report indicate that the WPT is in a general state of decline through most of its range. At present, populations appear to be stable in only about 20-25% (maximum) of the total area of the range. Local extirpations have occurred in many areas, and significant reductions in population size and distribution have taken place in others. In many areas, WPT populations are experiencing recruitment failures due to a multiplicity of causes that are manifest as adult-biased populations. Reasons for the overall decline of the species and recruitment failures are numerous, complex, and not easy to disentangle in terms of relative importance. The most important factors are thought to be alteration of aquatic and associated upland habitats, historical commercial exploitation, introduced predators, population fragmentation, and drought. In addition, pollution, and disease and disturbance due to a variety of other human activities may play a role. The occurrence of WPTs in many areas testifies to their resilience, but the mere presence of turtles in an area is no guarantee of lack of problems. Because WPTs have a potentially long lifespan, significant numbers of WPTs may occur in an area for decades after the population has been **effectively** extirpated through recruitment failures. Existing WPT populations face numerous threats throughout their range, many of which are becoming increasingly important, and the future of this species cannot be assured given the current level(s) of protection afforded it under state law(s). Thus, pursuant to provisions of the Endangered Species Act of 1973, we hereby formally petition the United States Fish and Wildlife Service to list the WPT (*Clemmys marmorata*) as Endangered or Threatened as specified, depending on the degree of threat, in different parts of its range.

Endangered: Washington - entire state
 Nevada - entire state
 Oregon - Columbia, Willamette and Klamath drainages and associated tributaries
 California - Klamath River and tributaries downstream to confluence with Scott River, all Clear Lake internal drainages, all Central Valley drainages from the Mokelumne River (inclusive) south, Mojave River and all desert drainages, all southern California drainages from the Santa Clara River (inclusive) south.

Threatened: Remainder of range, including but not limited to:
 Oregon - Umpqua, Rogue and coastal drainages and tributaries
 California - Klamath River and tributaries downstream from Scott River confluence, all other north coast drainages, all Central Valley drainages from Mokelumne River north, all non-Central Valley drainages from San Francisco Bay south to the Santa Clara River

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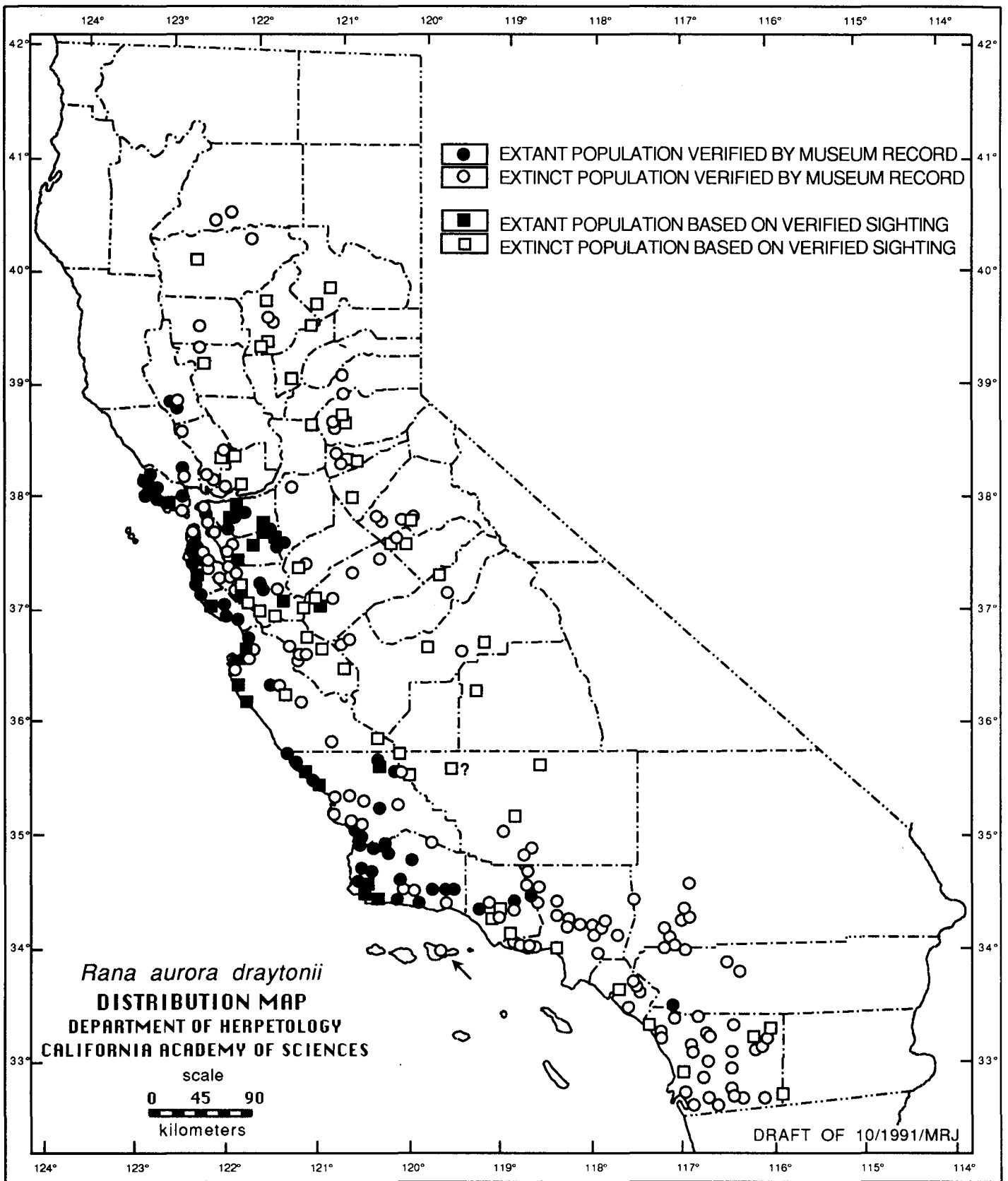


Figure 1. Historic and current distribution of *Rana aurora draytonii* in California based on 1205 museum records and 250 records from other sources.

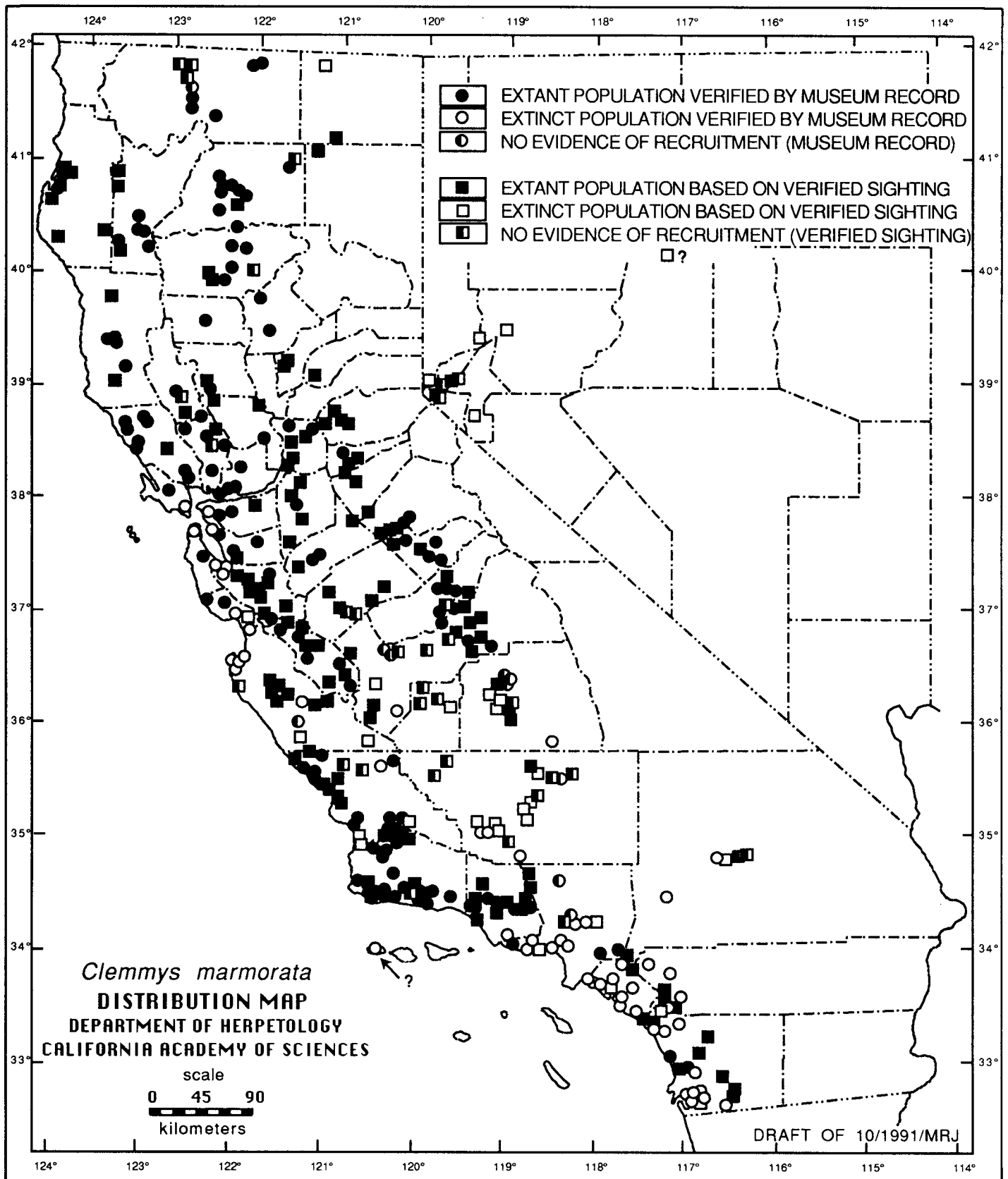


Figure 2. Historic and current distribution of *Clemmys marmorata* in California and Nevada based on 484 museum records and 417 records from other sources.

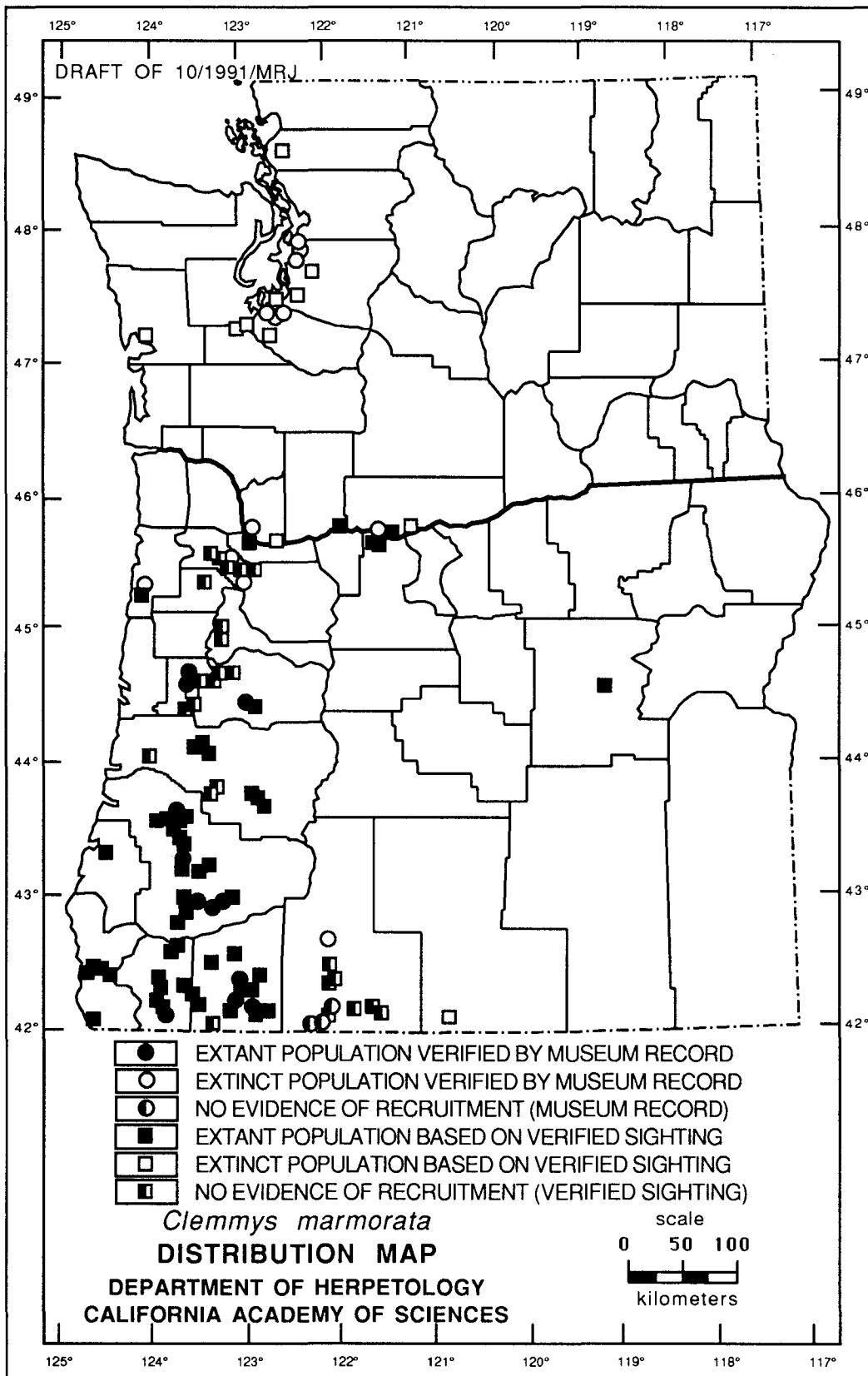


Figure 3. Historic and current distribution of *Clemmys marmorata* in Oregon and Washington based on 57 museum records and 175 records from other sources.