Biological Effects to be considered in Status Reviews

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Taxonomy

The animal species in this status review is in the Phylum Mossusca. Molluscs are invertebrates (animals without backbones) and are further divided into nine or ten Classes. This species is in the class Gastropoda or gastropods (snails and slugs).

This species was formerly assigned to the Order Neotaenioglossa, but a 2005 classification omitted this level.
http://www.itis.gov/servlet/SingleRpt/SingleRpt?search_topic=TSN&search_value=70674

The Cahaba pebblesnail is then assigned to the Family Lithoglyphidae by ITIS and (Thompson 1984; and formerly to Hydrobiidae by the Service), a family of small freshwater snails with gills and an operculum (a cover to close off the opening in a snail's shell and protect the soft body parts). Scientists write the full name of a species as the genus and species in italics or underline text with the genus sometimes abbreviated. Thus, the common name Cahaba pebblesnail is equivalent to the scientific name *Clappia cahabensis*.

Biologists group all living organisms into various taxa or classifications: The largest (most inclusive) grouping is at the level of Kingdoms, and Kingdoms contain Phyla which in turn contain Classes, then Orders, then Families, which in turn contain Genera, which are composed of Species. Within the Animalia, there are numerous groups at the next, lower level, the Phylum and so on, down to the levels of species and sub-species.

At the time the ESA was drafted only two Kingdoms were recognized, Animals (Animalia) and Plants (Plantae); hence the conspicuous absence of statutory language referring to the three other Kingdoms (Fungi, Protista, and the Monera or Bacteria).
A commonly used mnemonic for the taxonomic hierarchy is: "King Phillip Comes Of Fairly Good Stock." As an example, humans are in the Class Mammalia, Order Primates.

The Service bases its priority for listing actions on distinctiveness in a taxon (e.g. under the Service's Listing Priority Guidance, the only species in a genus warrants priority in listing over a species that is merely one of many other species in a genus). Besides protections for species, the ESA also recognizes and requires protection of, sub-species and population segments (if the latter are distinct from other populations). The Service has an official policy on recognizing Distinct Population Segments, but oddly has no policy on recognizing sub-species; however, it was recently ordered to develop such a policy by the Ninth Circuit Court of Appeals.

Scope of Administrative Record

This Status Review incorporates by reference all citations in the Bibliography (Webb 2002), as well as the citations in the references themselves. When a Petition to List a Species under the Endangered Species Act is submitted to the US Fish and Wildlife Service (Service), the Service recognizes that it must examine "information submit[ted] with or referenced in the petition" (Deibert 1999b). The Bibliography consists of the literature cited and selected references, which are intended to help the Service in its evaluation of the plight of this species. In order to prevent extinction, management decisions may need to be made even though some level of uncertainty exists (Soule 1991c). Likewise, the Endangered Species Act (ESA) requires that the Service make a decision solely based on the "best scientific and commercial data available" and not delay in making a decision. 16 U.S.C. § 1533(b)(1)(A).

Benefits of Preserving Endangered and Threatened Species

In crafting the Endangered Species Act, Congress was cognizant of the many different types of benefits conferred by preserving species. These include religious benefits (particularly prominent in Christianity), ethical and moral concerns, recreation, and what scientists term "ecosystem services," such as clean water and clean air that are provided by the various species functioning in an ecosystem. But direct economic benefits also flow from the preservation of species, and these economic benefits can be greater in small, humble organisms than in recreation form watching large charismatic mega-fauna such as wolves.

Possibly the greatest single example of economic benefits from a humble organism is an enzyme found only in a single microbe located only in one hot springs in Yellowstone National Park. One thermophile that lives in hot springs is the source of the heat-stable DNA polymerase enzyme used in the polymerase chain reaction (PCR). PCR is the foundation for molecular genetics and the multi-trillion dollar biotechnology industry. If that single microbe had not been protected from extinction in a National Park, trillions of dollars, not to mention numerous life-saving medicines, would be lost to humanity.

Most (70%) pharmaceutical products are based on native species, despite only 0.1% of plant species having been examined for their medicinal value. Scientists analyze molecular structures of chemicals made by a species and then slightly modify the chemical structure for patenting or improved medicinal results. And, many (25 to 40%) pharmaceutical products come directly from wild plants and animals. Kellert (1996).

Population Mechanisms & Vulnerability

This species possesses certain physiological, anatomical, behavioral and population characteristics that render it more vulnerable than most species to various environmental effects in terms of extinction risk. A variety of studies have shown that increased vulnerability to extinction results from the following factors: small population size, low population density, isolation of sub-populations (demes), ineffectiveness of dispersal, large body size, large home range size, spatial concentration of individuals, non-random mating system, and low fecundity, among others (Ehrenfeld 1970; Terborgh 1975, 1976; Pimm, et al. 1988; Primack 1993). These characteristics and the others discussed throughout this review constitute threats to the species within the purview of the ESA.

Concern over this species is exacerbated because it is known that extinction can be very rapid, taking place in less than a century (Holdaway and Jacomb 2000). Additionally, the "principal demographic properties contributing to" status as a "threatened or endangered species" are "a low maximum breeding population size and a high coefficient of variation in that size" (Ehrlich and Wilson 1991, citing Pimm, et al. 1988).

Habitat loss represents a "particular threat to taxa that are ecologically specialized" (Owens and Bennett 2000) such as this species. The Service should take particular note of this issue because the research was presented in a quasi-governmental forum, published by the United States National Academy of Sciences. Davies, et al. (2004) demonstrated empirically that species which are both rare and specialized at are greater risk of extinction than would be expected because of interaction effects between traits.

Range Size & Sparse Distribution

An important concern is that extinction probabilities are more closely related to population density than to the area occupied (Bolger, et al. 1991). This is of particular importance to species, even those with large absolute populations, which are sparsely distributed across the species range. Moreover, Pimm, et al. (1988) showed that extinction risk was correlated with variation in population size – another significant concern for a species such as this, which often fluctuates in population size. These problems are exacerbated by the low intrinsic rate of increase, which also results in higher extinction risk (Pimm, et al. 1988, p. 760).

Even when the range of a species is large, range reduction is of great concern. Soule (1983) found that such range reductions are the time at which conservation efforts should be concentrated, as species may be inevitably doomed in the periods of decline following range reductions.

This species possesses many of the biological traits that are known to increase extinction risk based on both ecological and paleontological studies: low fecundity, complex behavior, low mean abundance, and high variation in abundance (McKinney 1997, Table 1, p. 499).

Population Recovery

This species has a relatively low reproductive rate and the ability to recover from population reductions is low. Such traits as low fecundity, low adult survivorship, and very low productivity increase extinction risk. The recovery time of this species when threats are removed is slow, as is that of its habitat.

Time to extinction is not the metric of interest in an endangerment analysis; instead, the time until the species is unrecoverable is the most important concern. If the species declines to a point at which it cannot be recovered, it must inevitably become extinct at some time. For species which have difficulty in increasing their numbers from small population sizes, the differences between these two time periods is significant. Once the species, subspecies, or distinct population segment is no longer recoverable, extinction is inevitable. The Service should estimate the time remaining until populations are so small as to be unrecoverable.
The Service has recognized the important effects on extinction risk of demographic stochasticity, environmental stochasticity, natural catastrophes (such as fire and drought), and genetic stochasticity (Shaffer 1981). At the time of publication, Dr. Shaffer was a FWS biologist in the

Office of Migratory Bird Management (Shaffer 1981, footnote, p. 131). The effects of these and other factors on population viability have been addressed in modeling exercises with varying degrees of empirical testing and validation. Boyce (1992) and later Beissinger and Westphal (1998) have summarized various population viability model (PVA) categories and discussed their data requirements. Viability analyses that do not estimate lower confidence intervals, or that fail to consider the effects of rare, but severe episodic events will be "unduly optimistic" in their assessment of population viability (Ludwig 1999). For example, when two known severe population declines were excluded from a data set, the probability of extinction calculated was "quite small;" however, the declines greatly increased extinction probabilities (Ludwig 1999, p. 304). Despite the importance of environmental stochasticity, other factors can add greatly to extinction risk. As Ellner and Holmes (2008) put it: "Risk analyses based solely on environmental stochasticity do not account for unanticipated changes in environmental conditions (habitat loss, climate change, impeded movement) or in ecological interactions such as introduced predators or pathogens." There is always variance around any measurement of population phenomena, and additional variance around any model estimate – consequently, the Service should err on the side of conservation in considering both data and model outputs. Importantly, the Dept. of Interior has noted that population declines have exceeded estimates of the rate of decline in past recovery efforts (Pattee 1995).

Small Population Size

This species is now typically found in small populations, and small population size is itself a threat to such species, even if no trend towards even lower numbers is seen. It has long been known that demographic and genetic stochasticity are important determinants of extinction risk (Shaffer 1981). Both these risk factors increase in small populations. The Service recognizes that stochastic perturbations can "extinguish populations even in an environment that, on average, is favorable for their growth and persistence" (Shaffer 1981, p. 131) and must take this crucial factor into account when evaluating a potential listing. The Service cannot assume that an increase in population size removes extinction risk, unless the populations are sufficiently large that stochastic effects are negligible. As Poethke, et al. (1996) note, virtually all models of population dynamics predict geometrically decreasing persistence time of a population as population size decreases. This is true of traditional birth and death demographic models, as well as Markovian chain approximations and Monte Carlo simulation approaches (Poethke, et al. 1996, p. 83). Simple analytical models are inadequate for assessing population extinction probabilities, and generally underestimate the risk of extinction (Shaffer and Samson 1985). Deterministic models, in general, also underestimate extinction risk (Beddington and May 1977, Caswell 1989). As one example of the rapidity of extinction, the Heath Hen (*Tympanuchus cupido cupido*) was given an island refuge and predators were controlled. Yet, the species became extinct in only 14 years, although over 800 individuals were present in the population (Shaffer 1981, p. 131).

Population size is a major determinant of extinction risk (Reed, et al. 2003). Population size and the trend in population size are the best of 16 predictors of extinction (O'Grady et al. 2004).

Comprehensive models which include age-structure, catastrophes, demographic stochasticity, environmental stochasticity, and inbreeding depression show that population sizes greater than 12,000 individuals are necessary to avoid extinction for 80 generations. These types of population viability analysis (PVA) predictions have proven to be very accurate, and are a valid and accurate tool for categorizing and managing endangered species (Brook, et al. 2004). A meta-analysis of studies published since the early 1970s, and covering 141 sources and 212 species found that the minimum viable population size (MVP) for most species will exceed several thousand individuals. Trailla, et al. 2007.

Small populations are at high risk of extinction for several reasons, including loss of genetic variation through inbreeding or genetic drift, demographic fluctuations (such as variation in births, deaths, or age classes), and environmental fluctuations (such as variations in predation rate, disease or parasitism rates, climate, episodic weather events, competition, food supply and a host of other abiotic or biotic factors). Inbreeding depression, and perhaps other factors, show up preferentially in fitness characters such as fecundity, juvenile mortality, and age at first breeding, because those characters typically display dominance or over-dominance (Frankel and Soule 1981, p. 65). Inbreeding depression can affect all individuals in a population (Hedrick and Kalinowski 2000). Experimental studies have shown that inbreeding substantially increases extinction risk and the effects are particularly powerful for life history traits. Wright, et al. (2008).

Small populations have reduced ability to adapt to environmental change for two reasons. First, population genetic models show that genetic variation and potential response to selection are positively correlated with population size. The empirical support for this prediction is mixed: DNA markers usually reveal low heterozygosity in small populations, whereas quantitative traits show reduced heritability only in the smallest and most inbred populations. Quantitative variation can even increase in bottlenecked populations although this effect seems unlikely to increase the adaptive potential of populations. Second, individuals in small populations have lower fitness owing to environmental stress and genetic problems such as inbreeding, which can substantially increase the extinction probability of populations in changing environments. This second reason has not been included in assessments of the critical population size to assure the ability to evolve, and thus it is likely that many small threatened populations have a decreased potential for adaptation. Willi, et al. 2006.

However, species can often be at high risk for extinction long before population sizes decrease (Abrams 2002). Population sizes remain static or even increase as conditions initially deteriorate, but as conditions continue to worsen, population sizes then decline rapidly toward extinction (Abrams 2002). This occurs because changes in the populations of other components of the food web compensate for the decline in one or more fitness components of the focal population. However, the compensatory processes are generally nonlinear and often approach their limits abruptly rather than gradually. Because of "populations lag behind the equilibrium population size specified by current environmental conditions" (Abrams 2002), the Service cannot use a lack of decline in population size as a reason to avoid listing a species under the ESA.

Until the early 1990s, classic population genetics theory had not often been expanded to consider the effects of weak selection, environmental fluctuations, strong population subdivision, different aspects of inbreeding, and cumulative weakly deleterious mutational loads (Loeschcke and Jain 1994a, p. 87-88). As analysis of population viability has expanded to incorporate such additional factors, population persistence estimations have decreased. For example, when environmental fluctuations are added, persistence times increase only linearly with population size (Shaffer 1987) or only logarithmically (Wissel, et al. 1995), resulting in increased risk of extinction at even larger population sizes (Poethke, et al. 1996, p. 83-84). Thus, if the Service finds that contemporary viability models do not include all factors affecting demographic variables, or more likely, that not all factors affecting demographic variables are known with precision, it must err on the side of caution to fulfill the legislative mandate in the ESA.

Short-term studies consistently underestimate the true variances for demographic parameters in populations. Thus, the lack of long-term studies for endangered species leads to widespread underestimation of extinction risk. Conservation programs for wild populations, that are not designed to ensure large population sizes (greater than several thousand adults) will not ensure long-term persistence (Reed, et al. 2003), and are merely plans for long-term extinction.

Shaffer (1981, p. 133) reviewed the history and limitations of arbitrary rules of thumb, such as the "50/500 rule." Briefly, the 50/500 rule considered only genetic effects and did not account for the need to preserve long term evolutionary potential. When it first arose over two decades ago (Franklin 1980, Soule 1980), the lower limit of $N_e = 50$ was thought to prevent unacceptable short-term rates of inbreeding, and the N_e of 500 was thought to maintain genetic variability. Neither figure is, or was, intended to incorporate demographic or environmental stochastic factors. Such "magic numbers" as the basic 50/500 rule are suitable for informing lay audiences about the general dangers of small population sizes, even though subject to misunderstanding and misuse (Soule 1987). NMFS recognizes this and has noted criticisms of the 50/500 rule, including the arbitrary choice of 1% inbreeding, the assumption that selection is zero in wild populations, and others (Thompson 1991, p. 6, 25). As Culotta (1995) notes, more recent studies show that substantially higher population sizes are required to prevent extinction.

The best available science requires a minimum N_e of 5,000 for population viability (Lande 1995). This viability requirement for N_e translates to a much larger total N – to a census population of 10,000 to 20,000 individuals or more (Culotta 1995, Noss 2000). This is particularly true for populations that have been gradually reduced in size over time (Lande 1995, p. 786). Minimum viable population size estimates smaller than 10,000 are based on old science and have now been superseded (Cullota 1995). A population below the minimum size for viability must, *ipso facto*, be legally endangered. All populations smaller than about 10,000 individuals should be considered endangered, and in danger of extinction. The old "50/500 rule" has been superseded and is unreliable for species protection.

A critical review of minimum population size requirements shows that conservation policy does not reflect scientific findings, with concerns about "feasibility over-riding biological risk assessment" and repeated examples showing that "scientific recommendations… are compromised by policy makers" (Traill, et al. 2010). The "evidence is clear" that thousands (not hundreds) of individuals are required for viability, yet "the required number of individuals in a population often greatly exceeds the targets proposed by conservation management" (Traill, et al. 2010).

Another factor which interacts with genetic phenomena is environmental fluctuation – such changes in the environment are likely to have fitness consequences, resulting in a high degree of variance in selection coefficients, which in turn "drastically decreases the mean time to extinction" (Lande 1995, p. 787). Such extinction risks are comparable to those from environmental stochasticity even for merely reasonable levels of variance of selection coefficients (Lande 1995, p. 788). Mutation load of even slightly deleterious mutations "considerably enhances extinction risk for small sexual populations if it acts together with demographic stochasticity" (Gabriel and Burger 1994, p. 69; Gabriel, et al. 1991). Classical models of demographic stochasticity typically underpredict extinction risk (Gabriel and Burger 1992). Moreover, smaller populations face much greater synergistic interactions among risk factors, greatly amplifying overall extinction risk (Gabriel and Burger 1994, p. 70). Population sizes of 10,000 or more may be needed to maintain single-locus traits, such as disease resistance. Moreover, synergistic interactions among risk factors must be considered in determining population viability, and such interactions likely require minimum population sizes greater than 10,000 individuals. "Smaller and more isolated disjunct populations are likely more susceptible to local declines or extinctions" (Marcot, et al. 1997; Quigley and Arbelbide 1997a, p. 77). Elphick, et al. (2001) found that a population size of at least several thousand individuals is required for recovery in birds.

A number of other factors show that this species is likely to be even more at risk than other species with similar population sizes. Marginal populations are likely to be less variable than central populations, thus increasing the likelihood of extinction in such peripheral isolates. Peripheral populations may also occupy less favorable habitat, further increasing their risk of extinction (Wilson 1975, p. 113).

Populations of this species fluctuate with environmental conditions, and the lower the population level, the greater the risk of extinction (Hays, et al. 1998). Lande (1988a, 1988b) and later Caro and Laurenson (1994) noted that while genetic variation issues are serious and perhaps irreversible problems, environmental and demographic fluctuations are likely to cause the greatest risk of extinction in the near term. Vrijenhoek (1994) showed that genetic correlations can arise in small isolated populations, and that remnant variation is likely to be correlated with fitness, thus affecting population viability. Vrijenhoek's analysis is important both for understanding extinction vortices (*sensu* Gilpin and Soule 1986) and for appreciating the importance of peripheral populations. To ignore such correlations is to fall into the trap of "beanbag genetics" criticized decades ago by Lewontin (1974).

Population trends themselves can constitute risk factors for extinction (Mertz 1971a, 1971b). Populations that have been in declining trends for several generations experience selective environments favoring delayed reproduction, reduced reproductive effort and increased longevity. These are precisely the demographic characteristics that make it more difficult to recover from sharp population declines, and thereby increase extinction risk (Wilson 1975, p. 100; Mertz 1971a, 1971b). The Service recognizes that extinction risk requires a time interval for any meaningful definition, and a minimum viable population size must be sufficiently large to "endure the calamities of various perturbations and do so within its particular biogeographic context" (Shaffer 1981, p. 132). The National Marine Fisheries Service (NMFS) has also noted that the definition of "danger" does not imply a sense of immediacy – the term peril connotes immediacy of hazard (Thompson 1991, p. 2). Thus, the Fish & Wildlife Service must define risk of extinction over some fairly lengthy quantitative time period to avoid an arbitrary or capricious listing action. A definition which requires immediacy, as has been used by FWS in the past, appears to conflict with the Congressional mandate in the statute. Moreover, the Fish & Wildlife Service should defer and coordinate its listing policies with those of its sister agency responsible for listings, NMFS. The above threat factors, and others, are discussed further under each topical heading below.

Loss of Genetic Variation

Small populations face increased susceptibility to chance environmental and demographic effects, and also typically lose genetic variation due to drift, inbreeding or other population phenomena (Lande 1988a, 1988b; Caro and Laurenson 1994). Loss of genetic variation reduces mean population fitness and decreases the ability of the population to respond to environmental stressors (Westemeier, et al. 1998). Factors causing reduced genetic variation such as genetic drift, founder effect, and inbreeding increase as population size decreases and ultimately reduce both fitness and survivorship (Allendorf and Leary 1986; Roelke, et al. 1993; Lacy 1987). Rare alleles are lost more rapidly in small populations, as is the proportion of heterozygous genes per individual and in the entire population (Avise 1994, Gyllensten 1985). This effect is known from populations of wild vertebrates (Soule and Mills 1998). That low genetic variation causes species extinction is not merely theoretical: Johson and Dunn (2006) found that the Heath Hen (*Tympanuchus cupido cupido*) had low levels of mitochondrial genetic variation prior to its extinction.

The evidence is "compelling" that inbreeding depression and loss of genetic diversity are important factors increasing extinction risk (Frankham 2005). Disregarding the influence of inbreeding depression on extinction risk will lead to serious overestimates of the survival prospects of threatened species (O'Grady, et al. 2006). As Frankham (2005) noted, if these genetic factors are ignored, extinction risk will be underestimated and recovery strategies will be inappropriate. It is known that ecological disturbance causes genetic changes that severely affect the ability of a population to persist, including inbreeding effects and reduced heterozygosity. Vignieri 2010.

Many populations of this species may become trapped in an "extinction vortex" (*sensu* Gilpin and Soule 1986) where loss of genetic variation interacts synergistically with demographic and environmental stochastic effects to drive populations to extinction. Populations of this species, like those of the greater prairie chicken, exist in a series of isolated relicts with few habitat connections and only sporadic gene flow to restore genetic variation. Prairie chicken population size, productivity, genetic heterogeneity, and fitness decreased even though both habitat quality and quantity increased, and predation and nest parasites were controlled (Westemeier, et al. 1998, p. 1697). Thus, both current and planned methodology to enhance populations may prove ineffective.

Demographic Stochasticity

In small populations, *r* fluctuates due to demographic effects, even in a constant environment. Small populations thus suffer erratic swings in size due to demographic stochasticity, and at the same time, such small numbers provide no buffer against declines in numbers resulting in extinction. Extinction risk depends strongly on factors contributing to all types of stochasticity, including variation in vital rates, not merely environmental stochasticity. (Melbourne and Hastings 2008).

Because this species exists in small demes (sub-populations) they are particularly susceptible to such effects. Moreover, this species is subject to strong demographic effects of environmental variation on vital rates. This has strong effects on extinction risk. Populations "numbering many thousands may be at risk" in species "subjected to intense environmentally driven variation in life-history parameters" (Quinn and Karr 1993, p. 452; Leigh 1981). This is just such a species. Extreme variability in population size suggests "considerable risks of extinction in even locally abundant populations" (Quinn and Karr 1993, p. 452). This environmentally driven variation in vital rates "can lead to a substantial short term risk of extinction" even when a population is growing in size (Quinn and Karr 1993, p. 460).

The recovery time of populations subject to demographic stochasticity strongly affects extinction risk. Extinction risk increases sharply as carrying capacity (K) decreases (MacArthur and Wilson 1967), and is strongly affected by r_m (Lande 1993). These factors are low in this species, further increasing the likelihood of extinction. When the long-run growth rate is negative, extinction risk scales logarithmically with initial population size (Lande 1993, p. 912, 923).

Effective population size is also affected by the variance in production of progeny among individuals (Lande and Barrowclough 1987). Thus, all census estimates of population size must be reduced by the operation of demographic effects to adequately assess extinction risk. Demographic stochasticity can be an even greater threat to small populations than is genetic stochasticity (Westemeier, et al. 1998).

Environmental Stochasticity

May (1973b) may have been the first to distinguish between the effects of demographic and environmental stochasticity. Contemporary models of metapopulation dynamics often assume that persistence depends on the balance of extinction and colonization in a static environment (Hanski 1996). Environments are not static, however, particularly the habitat of this species. Besides extreme climatic variation, these habitats are being reduced in size, degraded in quality and fragmented at rapid rates. Large and unpredictable fluctuations in climatic and other environmental factors are known to occur frequently across the entire range of the species. These are thus anticipated risk factors that must be considered by the Service – no one can claim that these fluctuations are unanticipated natural events. It is known that environmental degradation can cause a tipping point in population dynamics, beyond which decline to extinction is almost certain (Drake and Griffen 2010).

Moreover, biotic ecosystem components exhibit more variability than do abiotic components, and animals exhibit more variability than do plants (Kratz, et al. 1995). Saether, et al. (2005) found that short-term risk of extinction for numerous bird populations is most highly correlated with environmental stochasticity. Finally, environmental stochasticity can be more important than genetic or demographic stochasticity, especially in populations of moderate or greater size (Lande 1988a). This fact has been recognized by both NMFS (Thompson 1991, p. 4) and FWS (Shaffer 1987).

Environmental stochasticity can be fully examined only if the factors of predictability and amplitude of effect, as well as the periodicity and contingency of environmental variation are considered (Colwell 1974, Stearns 1981). Duration of effect, recovery time, spatial distribution, and frequency are other important metrics for evaluating disturbance effects (examples in: Sousa 1984; White and Pickett 1985; Wiens 1989b). One of the major errors in the conservation planning for spotted owls was that environmental stochasticity was not considered (Noon and McKelvey 1996, p. 148); consequently, the population models were overly optimistic and owls remain at risk.

Importantly, variance in environmental factors alone can cause extinction, even in the absence of pronounced amplitude of those factors (Saltz, et al. 2006). In the Saltz, et al. study, good years did not enhance reproductive success beyond certain limits, but bad years reduced reproductive success, causing local extinctions. While Saltz, et al. studied a particular species (Equidae), the result obtained is general and must be considered by the Service for this species.

Climatic and weather effects have long been recognized as primary factors affecting terrestrial populations (Allee, et al. 1949, chap. 22; Andrewartha and Birch 1954, part III; Watt 1968, chap. 11.3; Gessaman and Worthen 1982). Climate and its short-term analog, weather, are particularly variable and unpredictable in these habitat types. Because it is located in a continental interior, the range of this species is often completely affected by climatic effects such as drought – no spatial refugia exist for the species.

Genetic variation, including metrics such as population mean heterozygosity and number of alleles per locus, were significantly lower after a population bottleneck (Bellinger, et al. 2003).

Moreover, environmental variation affects the ability of humans to manage ecosystem effects. Short-term variations in local or regional precipitation – upon which management planning often is based – are greater than the predicted change in the mean value of precipitation for North America (Shuttleworth, 1996).

Drought is a major climatic variable affecting population productivity and viability. Drought events appear to be associated with extra-tropical oceanic circulation patterns (Miller, et al. 1993) and are exacerbated by global warming. The hydrological cycle of the western United States has changed significantly over the last half of the 20th century, mainly because of human influences, and a crisis in water supply for the western United States will occur in the near future (Barnett, et al. 2008). The same is true for other areas of North America. Both the frequency and intensity of drought in the United States is expected to increase due to climate change (Field et al. 2007, Cook et al. 2008).

Effective Population Sizes

Effective population size (N_e) is a term used to incorporate various environmental, demographic, and genetic effects on population dynamics. N_e has been calculated in various ways, and can incorporate the effects of drift, inbreeding, and loss of alleles at segregating loci (Burgman, et al. 1993, p. 238). The population dynamics of small populations cannot be accurately estimated without consideration of effective population size. This is not a new result; instead, it dates from at least the time of Sewell Wright in the 1940s.

Various estimates of the ratio of N_e to N indicate that effective population sizes will be only 25% to 33% the size of the census population (Soule 1980, Wilcox 1986). Salwasser and Marcot (1986) suggested 50%, but they used unrealistically low minimum viable population sizes of 100 and 1,000. For vertebrates, minimum viable populations should be approximately 5,500 or more (Thomas 1990). NMFS independently derived similar a size of 5,000 (Thompson 1991; p. 8, Appendix B). The Service has used unrealistically low recovery goals for other species, for example the Southern Sea Otter, for which it used a "genetically effective population size of 500" (FWS Southern Sea Otter Recovery Team 2000, p. 25), which led to census population sizes of more than 2,650 for recovery, of 1,850 to 2,650 for the threatened category, and of 1,850 or less for the endangered category (id., p. ix). The Service did not adequately include environmental stochasticity and other effects in its draft determination, however. Thus, N_e will be lower than those calculated using the formula given by Schroeder (1998b). However, if used cautiously and conservatively, Schroeder's formulation is a useful metric until population viability models are established for the species. The Service must make its listing determination on the basis of the best available science regarding minimum viable population sizes, however. That science indicates that relatively large census populations are required.

Other factors will also increase the census population size required. N_e also will vary with the spatial dispersion of individuals across a landscape (Hartl and Clark, p. 90), an effect that might be particularly large for this species. Because there are a large number of different factors that affect population dynamics, there are a plethora of ways to calculate N_e – at least as many as the number of mechanisms affecting genetic drift (Boyce 1992, p. 486). Effects of overlapping generations, dispersal and spatial dispersion have been considered by Reed, et al. (1986); Chepko-Sade, et al. (1987); and Harris and Allendorf (1989). Generally, the effects of overlapping generations can cause severe underestimation of the size of a census population required to maintain N_e at a given size and the method of Reed, et al. (1986) will usually underestimate census population sizes (Koenig 1988, p. 232). As with other species having overlapping generations, juveniles and adults are not equally likely to breed. N_e has also been calculated for extinction-repopulation events in a metapopulation (Gilpin and Hanski 1991). Incorporation of additional factors in more complete models will further reduce effective population size and hence persistence time.

Allee Effects and Social Disruption

The term Allee effect refers to the negative effects on population processes of low population size or density (Allee 1938, 1951; Drickamer and Vessey 1992), sometimes also termed "undercrowding." Thus, as population size decreases, social mechanisms collapse at some threshold value. Allee effects include the difficulties of finding a mate (Dennis 1989), and the ability to evaluate numerous potential mates for suitability, difficulty in detecting, evading or fending off predators or competitors (Boyce 1992, p. 493). Allee effects appear ubiquitous and occur across a wide range of taxa: for example, Allee noted that groups of fish survived toxic conditions better than single fish, as did groups of flatworms. Moreover, all the benefits of group size for predator protection and other ecological interactions (summarized in Wilson 1975) are lost when group sizes decrease. Soule (1983) notes that social disruption is likely to cause extinction long before genetic effects such as drift or inbreeding and long before demographic stochasticity. Even species with "high reproductive potential are vulnerable" and that a minimum viable population size may be two orders of magnitude larger when social disruption is involved (Soule 1983, p. 118).

Because of Allee effects, decreases in population size are unlikely to be linear; instead, as population size decreases to some threshold value, rapid declines and extirpation are likely. Such declines may be so rapid as to be undetectable before they actually occur, or if detected, extirpation may not be preventable by that point. Certain aspects of the environment combine with Allee effects to greatly increase the likelihood of extinction for this species. First, Dennis (1989) has shown that environmental stochasticity amplifies Allee effects. As explained elsewhere in this review, these populations fluctuate greatly and habitat is subject to great amplitudes of environmental stochasticity as well as unpredictability of these fluctuations. Second, "harvesting also amplifies those [Allee] effects" (Dennis 1989, p. 481). Thus, hunting can greatly increase extinction risk, and this risk will not be accounted for in conventional models of population harvest. Petitioners request that the Service consider this and all non-linear effects of population declines in evaluating the status of this species. This is particularly important because non-linear, or threshold effects may cause populations that "appear safe for many years" to "decline suddenly" (Meffe and Carroll 1997, p. 218). Pulliam (1992) already modeled a situation similar to that for this species –an example in which the population abruptly declines as the proportion of agricultural areas in the landscape increases. Such declines are difficult to predict "because the problem may well go undetected until a critical threshold is reached" and causes "catastrophic population collapse" which has been documented repeatedly (Meffe and Carroll 1997, p. 219). One of the major errors in the conservation planning for spotted owls was that Allee effects were not considered (Noon and McKelvey 1996, p. 148); consequently, the population models were overly optimistic and owls remain at risk. Habitat loss and fragmentation are likely to disrupt social neighborhoods resulting in lower levels of social interaction and reduced productivity (Cale 2003).

Higher density may also inhibit population persistence. For example, reproductive conflict is more likely as ranges shrink. In social species, reproductive conflict is known to inhibit population recovery and increase the likelihood extinction (Lopez-Sepulcre, et al. 2008). In the critically endangered birds studied by Lopez-Sepulcre, et al. (2008) reproductive conflict was expressed as territoriality; however, any type of social behavior could increase extinction risk, and management must consider the impacts of such effects regardless of population size.

Cultural Inheritance

Modern behavioral ecology has documented numerous instances of cultural inheritance in non-human species (sometimes termed "memes"). Young learn from older animals, and thus important survival behaviors can be lost from the population if cultural transmission is interrupted. This effect operates most strongly in small populations, just as alleles are lost most frequently in small populations. Social disruption, and the removal of older more experienced individuals from a population, as in trophy hunting, can also be expected to remove learned behaviors from the population (Wilson 1975, p. 152, 168-172). Connelly, et al. (1988) suggested that seasonal movements may be traditional, hence culturally inherited. For example, roads often eliminate traditional movements from these populations because older, more experienced individuals are eliminated from the population by road deaths. Thus, deaths of relatively old, post-reproductive individuals may have important population effects.

Reintroductions

Reintroductions of species are difficult and expensive. Once community structure is altered, introduction of a species – even one previously present in the community – can be difficult (Diamond 1975b). Reintroductions from captive breeding programs may have lower fitness than natural populations, and such programs may allow deleterious genes or gene combinations to increase as captives are provided with ad lib food, medical care, and a predator free environment (Caughley 1994). Captives are also likely to lose essential behaviors if there is any component that is transmitted culturally. Moreover, it has never been demonstrated that an extinction can be prevented by any reintroduction (Fyfe 1978, Berger 1978). Reintroductions also run the risk of introducing disease if a small natural population already exists. Because reintroductions are so difficult, it is imperative that existing populations of this species be conserved wherever found. It may not be possible to reintroduce this species where it has been extirpated; thus, peripheral isolates are of great importance to recovery of the species.

Importance of Peripheral Populations

Contemporary understanding of evolution and population biology emphasizes the importance of populations at the extremities of a species range. It is in these peripheral populations that the evolutionary potential of a species is greatest (Gadgil and Bossert 1970, Levin 1970, Gadgil 1971). Peripheral populations often differ genetically from more centrally located populations, thus adding genetic diversity to the species and providing genetic backgrounds where natural selection can more easily increase the gene frequency of novel alleles or combinations. "Such populations are often of evolutionary significance" (Scott, et al. 1993, p. 35), and "preserv[e] unique genetic material" which is "restricted to peripheral populations of native species" (Scott, et al. 1993, p. 36; Quinn and Karr 1993). Peripheral populations may be disproportionately important for conserving genetic diversity (Squires, et al. 1998). Genetic differentiation of peripheral populations from more central populations is well documented (e.g., Gibson, et al. 2009, Lammi et al. 1999, Rossum et al. 2003, Hampe and Petit 2005, Hamilton & Eckert 2007), and conserving such genetic variation is important to preventing extinction of the larger species because it is "significant to future adaptation potential" (Gibson, et al. 2009). Thus, protection of such peripheral populations is critical whether they are distinct population segments or not. The effects of genetic isolation, exposure to different selection pressures, and small population sizes may make the periphery of a species' range particularly evolutionarily dynamic (Lesica & Allendorf 1995, Garcia-Ramos and Kirkpatrick 1997, Hewitt & Nichols 2005). Hence, the preservation of peripheral populations is critical to persistence of the entire species.

Peripheral populations are also often located at the ecological limits of the species, thus exposing these novel genetic combinations to environmental circumstances that may later become prevalent in central populations, such as global warming effects. Such testing of the periphery can act to stabilize the entire species in the face of environmental change. However, peripheral populations are much more likely to go extinct than are core populations. Populations along the periphery of the range occupy less favorable habitats and tend to be more fragmented and, as a result, are less likely to receive immigrants from other populations. A population's probability of extinction is directly correlated with its variability and inversely correlated with density and immigration rate. This has led to the prediction that, when a species becomes endangered, its geographical range should contract inwards, with the core populations persisting until the final stages of decline. Convinced by these logical but untested deductions, conservation biologists and wildlife managers have been instructed to avoid the range periphery when planning conservation strategies or allocating resources for endangered species. However, studies of range contraction in 245 species from a broad range of taxonomic groups and geographical regions do not support the above assumptions; instead, most species examined (98%) persist in the periphery of their historical geographical ranges (Channell and Lomolini 2000). For these reasons, conservation strategies, including critical habitat designations must take more account of the outer fringes of an endangered species' geographical range instead of just concentrating on core regions where the species is still most plentiful (Channell and Lomolini 2000). Populations on the edge of their range are sometimes assumed to be less dense, less fit, less genetically variable, and, therefore, more prone to extinction (Brown, et al. 1996). However, such peripheral populations are sometimes more likely to show increased persistence than more centrally located populations. Lomolino & Channell 1995, 1998, Channell & Lomolino 2000. (These authors studied range contractions of mammals in North America.)

Gibson, et al. (2009) suggested that persistence on an edge may be more likely when extirpation is due to extrinsic factors that move through the range like a contagion, such as climate change, rather than to intrinsic factors, such as inbreeding or bottlenecks.

Rapid evolution is likely when a peripheral population is isolated from gene flow, allowing a local deme to evolve to a local ecological optimum (Garcia-Ramos and Kirkpatrick 1997). Such evolution can be the first step to speciation, independent of genetic drift ("genetic revolutions," *sensu* Mayr 1963, 1982) or founder event mechanisms (Carson and Templeton 1984). There is "growing evidence that microevolutionary changes may often be rapid and, in many cases, occur on time frames comparable to human disturbance and anthropogenic change." Ashley, et al. (2003). Quantitative traits have very high mutation rates (1 per 1,000) as opposed to the mutation rates typical of other loci, and are thought to be more closely related to fitness than are other traits (Lande and Barrowclough 1987; Nei 1987).

The importance of peripheral populations is likely to increase with climatic change (Hunter 1991; Quinn and Karr 1992; Scott, et al. 1993). Because of climate change and other factors, the ranges of numerous species can no longer be assumed to stay more or less the same over time, and instead these ranges will often shift over the next few decades (Gibson, et al. 2009). As the species range shifts, if it does not purely contract, it may advance along a leading edge and retreat along a trailing edge (or rear edge, *sensu* Hampe and Petit 2005). If the leading edge is threatened by other anthropogenic pressures, such as land-use change, a species' range may be squeezed on both sides and incapable of persisting (Kerr & Deguise 2004), leading to extinction. Peripheral isolates are also of great importance to recovery of species where reintroduction is difficult, such as this species.

Population Isolation

Of particular import for the assessment of population size, and hence viability, is the degree to which panmixis occurs across the landscape. If individuals interbreed only between small demes, then actual population sizes will be quite small. Because many landscape features such as roads, powerlines, reservoirs, and agricultural fields now subdivide habitat, it is highly likely that the range now consists of a large number of isolated populations, rather than a large panmictic population or even a well-connected metapopulation. The implications for extinction risk are enormous.

Matings, and production of viable offspring would likely be much lower than these movement rates. Such movements, even if accompanied by matings, would not constitute effective levels of gene flow unless impregnated females were able to raise viable offspring who could prevail during all life-history stages and raise viable young themselves.

The Service acknowledges that population isolation increases the risk of extinction from stochastic genetic and environmental events including drought, flooding, and toxic spills (US FWS 2009; 74 Fed. Reg. 31114). Habitat modification and cumulative habitat degradation from non-point source pollution are also major threats for species which exist in isolated populations. The Service also admits that limited dispersal ability causes isolated populations to "gradually and quietly perish" as habitat conditions deteriorate (US FWS 2000, p. 14).

Habitat and Geographic Range

Most vegetation classifications are based on potential natural vegetation, not on the actual vegetation present currently. For example, a commonly used classification such as Kuchler's map thus does not delineate original from natural vegetation or include disclimax communities formed by disturbance effects (Loveland and Hutcheson 1995). Kuchler's map and other similar ones are thus "little more than informed guess as to what might be" present as vegetative communities on the ground (Scott, et al. 1987; Botkin, et al. 1984). Potential vegetation is "far removed from what actually occurs on an area" currently and "provides little guidance" as to habitat quality and biodiversity protection (Scott, et al. 1989, p. 85). Perhaps the first comparison between Kuchler's potential natural vegetation and extant plant communities was that of Klopatek, et al. (1979), which used data valid to 1967.

Remote sensing has also been used to estimate extant vegetation types and land uses. The most recent coarse level vegetation maps are from a 1990 USGS dataset, interpreted from the Advanced Very High Resolution Radiometer; however, radiometry data are nonetheless quite coarse (Loveland and Hutcheson 1995). That study did not include urban areas, but urbanization data are available from the Defense Mapping Agency (Loveland and Hutcheson 1995). Other classification schemes have been developed, for example the USFS's ecoregions (Bailey 1976), and the Nature Conservancy's National Vegetation Classification System (M. Anderson, et al. 1998; Grossman, et al. 1998), which uses a hierarchical classification system with over 4,100 plant associations, recognized at the lowest level of hierarchy.

The amount of its historic range that a species occupies is one of the two most important factors in determining whether a species can be recovered (Abbitt and Scott 2001). Small geographical range size is the single best predictor of threat of extinction in terrestrial species (Harris and Pimm 2008). For forest-dependent bird species, the threshold at which the species become threatened with extinction, was determined to be 11,000 km^2 (4,347 mile^2). Harris and Pimm (2008).

The area of habitat available for a species to persist in must not be confused with the much larger area shown on range maps. Range maps always overestimate the actual area occupied by a species (Harris and Pimm, 2008). Ranges must be reduced in size or trimmed by including only those parts of the ranges that meet the species' requirements of elevation and habitat types suitable for breeding, or the rearing of juveniles. Harris and Pimm (2008). Species range maps based on extents of occurrence (EOO maps) greatly overestimate the ranges of species that are at risk of extinction (Jetz, at al. 2008).

This species is found in an isolated, small habitat area, making it highly vulnerable to extinction. Even large populations found in small isolated areas can be extirpated by a single catastrophic event (e.g. FWS 2001, stating that a "restricted location makes [a species] vulnerable to extinction from a single catastrophic event").

Birds should be considered threatened if the actual area available for breeding is below a threshold of 11,000 km^2 (4,347 mi^2) Harris and Pimm (2008). However, species may be threatened even if they have larger ranges than this because of other (non-spatial) criteria (Harris and Pimm 2008).

For viability analysis, even a large area of habitat may be less important than fragmentation effects, and amount of area containing sufficient resources for good habitat may be greatly overestimated by simply summing the areas of habitat. Proximity analysis, termed adjacency analysis by Mladenoff (1997), is also important – areas without cover may used for feeding so long as they are near enough to landscape features which can function as cover for escape from predators. Such analyses are easily done using GIS (ASPRS/ACSM/RT 1992).

Habitat Degradation

Degraded habitats typically do not support as many individuals as higher quality habitats, but may support an equal number of breeding males (Gibbs and Faaborg 1990). Thus, surveys of breeding males may overestimate population viability (Gibbs and Faaborg 1990). Mechanistically, this effect can arise when degraded or marginal habitat is adequate for males but not for breeding females. Habitat loss and degradation are the "most frequent causes of species endangerment" (Schwartz 1999, p. 86; Wilcove, et al. 1993; Foin, et al. 1998).

Virtually all habitat types in the United States are being rapidly degraded and fragmented. The prognosis is poor for US forest ecosystems in general. Southeastern forests are undergoing profound changes, and the dry forest ecosystems of the American West, especially those once dominated by open ponderosa pine forests, are in widespread collapse. "We are now witnessing sudden leaps in aberrant ecosystem behavior long predicted by ecologists and conservation professionals" (Covington 2000). As forest ecosystems decline, pollution and hydrologically mediated strains are put on rivers, streams, lakes, and cave systems, even those many miles away from the affected forests.

Habitat degradation causes severe endangerment to species. The greatest causes of habitat degradation and loss by 2100 will likely be land-use alterations and climate change (Sala, et al. 2000). Successful recovery of species under the ESA has primarily involved species threatened with predation and pollutants. "It is much more difficult to recover species where habitat degradation and loss are the primary causes of endangerment" (Schwartz 1999, p. 86). Because future prospects for species recovery are significantly lower than for past recovery efforts, the Service should act promptly and comprehensively to list this species and to restore habitats. A second important reason to act based on habitat effects, such as habitat loss and degradation, is that, statistically, most species go extinct because of habitat effects.

Habitat Fragmentation and Landscape Effects

Habitat fragmentation is one mechanism that has been proposed to explain declines in a number of species, and has perhaps been most extensively studied in forest dwelling birds, particularly neotropical migrants (Wilcove, et al. 1986; Finch 1991; Faaborg, et al. 1993; Morrison, et al. 1992a; Sherry and Holmes 1993). Habitat fragmentation is particularly dangerous form of habitat effect, because the researcher may see healthy individuals when conducting population sampling, only to find that the number of individuals in each isolated patch are inadequate to prevent the loss of genetic diversity and consequent extinction risk. Moreover, this species may be unusually sensitive to fragmentation effects because of its reliance on a specialized habitat type. Both coarse-grained and fine-grained fragmentation could affect this species.

Habitat fragmentation occurs when a large tract of habitat is dissected into smaller patches isolated by other habitats or vegetation types different from the original (Wilcove, et al. 1986; Morrison, et al. 1992a; Faaborg, et al. 1993). These patches (also referred to as fragments, islands, or isolates) are redistributed into variable sizes, shapes, and locations from the original area (Diamond 1975a; Wilcove, et al. 1986; Morrison, et al. 1992a; Faaborg, et al. 1993). This dissection of intact habitat can be caused by various effects, including pipeline or powerline construction and operation, roading, or logging. For example, Gascon et al. (2000) showed that logging and the building of roads through forests results in their fragmentation. Around the perimeter of each forest fragment is an edge where ecological changes take place. Eventually, the edge effect leads to recession of the forest edges and diminution of the fragment until it disappears (Gascon et al. 2000). Fragmentation is often accompanied by habitat loss *sensu strictu* (Franklin, et al. 2002) and such effects may need to be distinguished if extinction mechanisms are to be properly studied for recovery planning.

Fragmentation causes increased extinction risk and population declines as compared to habitat loss alone (Andren 1994). Fragmentation affects nearly all species. Bender, et al. (1998) found that habitat fragmentation caused population declines in both edge and interior species, and that only generalist species that use both the edge and the interior of a habitat patch did not show effects from fragmentation itself. Fragmentation effects reduce species abundance below that expected from the total amount of habit present, and abundance is more closely related to a smaller metric – the area of patch interior present, termed "core areas" (Temple 1986a). Forman and Godron (1981) offer a readable introduction to patch dynamics, and a variety of textbooks offer simple introductions to fragmentation concepts (Primack 1993, Meffe and Carroll 1997). Wilcove (1987) identified four ways that fragmentation can cause extinction: (1) a species can be excluded from protected patches by the loss of internal heterogeneity due to invasion of edge plant species; (2) it creates isolated populations that are susceptible to catastrophes and genetic drift; (3) it interferes with ecological relationships; and (4) fragmentation creates edge environments which typically increase predation. Fragmentation and insularization (the creation of disconnected, "island" like habitat patches) are known to cause higher extinction rates than those from reductions in area size alone (Wilcove, et al. 1986). Spatial scale is important in fragmentation effects such as dispersal (Doak, et al. 1992); however, the spatial arrangement of habitat patches does not mitigate habitat loss in fragmentation processes (Fahrig 1997). Fragmentation and insularization are well advanced across the range of this species.

Another issue regarding geographic scaling effects is that studies conducted at smaller, local scales are often insufficiently powerful to detect population declines. Chalfoun, et al. (2002). Studies conducted at patch or edge scales are also less likely to detect nest predation of bird species or to detect the effects of habitat fragmentation. Studies at the landscape level should be relied upon when assessing these effects (Stephens, et al. 2004).

Meffe and Carroll (1997, p. 75) recently summarized the theoretical and empirical studies on habitat fragmentation effects: fragmentation "sets the stage for rapid local extinctions" because "extinction probabilities increase greatly in small populations" and extinction often occurs quickly – "in a matter of years or decades." Even in cases where a species can evolve rapidly in response to fragmentation, such population mechanisms may render the species more vulnerable to extinction. Patten, et al. (2005).

Many of the effects of fragmentation are explicable by the MacArthur-Wilson theory of island biogeography, which predicts a balance between immigration and extinction rates represented by the number of species on an island (MacArthur and Wilson 1967; Diamond 1975a; Whitcomb, et al. 1981; Morrison, et al. 1992a). This equilibrium number of species is dependent upon island size, distance from other colonizing populations, dispersal abilities, and population densities. Most importantly, equilibrium species number decreases with island size. Mechanistically, the negative impact of invasive species is higher on island species, and is particularly high in species with small distribution ranges destruction (Claveroa, et al. 2009). Mainland bird species and island species with large ranges tended to be affected by habitat destruction (Claveroa, et al. 2009).

Habitat fragments are similar to islands because there is an obstacle to dispersal, whether it is an agricultural area, a road, or a utility corridor that isolates them from other similar habitats (Diamond 1975a; Wilcove, et al. 1986). Fragments are also particularly susceptible to incursions by predators, invasive alien species, and competitors. Fragments are subject to higher invasion rates by parasites, parasitoids, and disease vectors. Populations in isolated fragments have lower growth rates than those in connected areas, and are thus more prone to extinction (Fahrig and Merriam 1985).

Fragmentation can affect species diversity, population persistence, and community structure, because it isolates individuals, breeding units, and sub-populations of patch-interior species into smaller sub-populations or demes. Smaller populations experience negative genetic effects, such as higher genetic drift and inbreeding depression (Lacy 1987, Wiens 1995), as well as being more susceptible to environmental and demographic fluctuations. Models based on island biogeography are not adequate to analyze fragmentation (Laurance 2008). Moreover, corrected for range size, continental species are more—not less—likely to be threatened than island species. Manne, et al. (1999).

Smaller patch sizes may be unable to effectively contain the home ranges of individuals in a species (Wilcove, et al. 1986), and also increase the risk of extinction by altering microclimates, decreasing cover availability, increasing predation, competition, or parasitism, and increasing the chances of human encroachment. In addition, the quality and quantity of resources decrease while the susceptibility of fragments to disturbance, such as wind blown weed seeds and fires increase (Morrison, et al. 1992a). All of these pressures on habitat-interior species increase as the size of the habitat fragment decreases.

Fragmentation not only causes a decrease in effective area size, but also affects habitat heterogeneity (Wilcove, et al. 1986). In forested areas, forest-interior bird species are dependent upon large expanses of their preferred habitat (Wilcove, et al. 1986; Morrison, et al. 1992a). When an area is fragmented, individual fragments may not have all the habitat types that were initially found in the original block. Therefore, species that require specific habitats are vulnerable to local extinction (Wilcove, et al. 1986). If a fragment lacks a required habitat for a given species, then establishment of breeding populations in that fragment cannot occur (Wilcove, et al. 1986). Local abundances of individual species are influenced by the structural and floristic characteristics of the vegetation and these vegetation characteristics vary with area size (Lynch and Whigham 1984; Wilcove, et al. 1986). Many species require more than one habitat type for survival and reproduction.

Another important effect of fragmentation is the creation of edge (Wilcove 1985; Wilcove, et al. 1986; Morrison, et al. 1992a). Gates and Gysel (1978) observed higher densities of nests along forest edges which may have resulted in increased predator densities or predator search efforts in edge habitats. Habitat edges caused by fragmentation effects and elongate patch shapes can function as ecological traps, luring interior dependent species to attempt breeding in areas where nest predators congregate (Weldon and Haddad 2005). Such effects may operate on this species. Plant and animal species associated with patch interior conditions are sensitive to early serial stages and edge habitats. Habitat fragments are susceptible to drying, wind penetration, and invasions by early successional plant species along edges and large openings (Morrison, et al. 1992a). Wind penetration into fragments would be an especially severe problem during winter. Edges increase predation on avian nests because a wide variety of avian, mammalian, and reptilian predators are abundant in such areas (Wilcove 1985; Wilcove, et al.

1986; Morrison, et al. 1992a). In flat habitats such as prairies, vertical elements that fragment such habitat can result in a tripling of nest predation rates (Burger, et al. 1994). Negative effects of fragmentation may also be indirect – fragmentation is known to affect community development, vegetation dynamics, and succession (Robinson, et al. 1993), all of which can reduce habitat quality for animal consumers.

This species lives in visually obvious locations and reproduction rates are low. In these respects it is similar to forest dwelling birds that have been found particularly susceptible to reduction in productivity by fragmentation (Whitcomb, et al. 1981). Such fragmentation may allow for high rates of nest predation (Askins, et al. 1990). Wilcove (1985) showed that open-cup ground nests were more susceptible to predation than low-canopy cavity nests. The effects of edge on nest predators in forested areas can extend over 600 m into a fragment (Wilcove 1985), meaning that a fragment as large as 100 hectares would have only edge and no interior, reducing its value to essentially zero. The exact relation of juvenile predation with respect to distance from an edge, and of the type of edge formed, is not known for this species, but prudence in conserving the species dictates that wide buffers be provided around any habitat.

When conceptualizing patch dynamics and meta-populations on a landscape, the approach typically taken is to partition habitats into core areas (often term core reserves) embedded in a matrix of less suitable habitat types. Sometimes, corridors for movement (dispersal) of a given life-history stage are included. Patch area, shape and isolation (most often by distance) are the typical measures of interest. Ricketts (2001) showed that the surrounding matrix can significantly influence the "effective isolation" of habitat patches, altering the isolation effect from the predictions of simple distance or classic models. A later and more comprehensive study of 1,015 bird, mammal, reptile, amphibian, and invertebrate population networks on 6 continents shows that while patch area and isolation are indeed important factors affecting the occupancy of many species, properties of the intervening matrix must not be ignored (Prugha, et al. 2008). Improving matrix quality may be critical to conservation and recovery of many listed species (Prugha, et al. 2008).

Fragmentation effects make analysis of population viability difficult and introduce time lags. The population size of a declining species is not a reliable guide to its risk of extinction because it takes some time following habitat loss before the extinction will occur (Hanski and Ovaskainen 2002). The Service must include such time lags and time delays (termed extinction debt by Hanski and Ovaskainen) in its analyses of listings, critical habitat designations, consultations, conferencing, and recovery plans in order to fulfill its duties under the ESA.

As Hanski and Ovaskainen (2002) put it: "landscapes that have recently experienced substantial habitat loss and fragmentation are expected to show a transient excess of rare species, which represents a previously overlooked signature of extinction debt… and ignoring time delays in population and metapopulation dynamics will lead to an underestimate of the number of effectively endangered species. Such landscapes that have recently experienced habitat loss are precisely the ones in which most species that are listed or petitioned to be listed under the ESA are found.

Both the range of this species and the ecosystem itself are severely fragmented.

Metapopulation Effects

Fragmentation splits a single large, cohesive population into a system of small sub-populations (demes) that are linked by gene flow into a metapopulation. (Alternatively, the sub-populations are not linked and, if small, become extinct.) In either event, it is critical that demes be recognized as such and not aggregated into a single large population – such errors will cause the observer to underestimate extinction rates (Wilson 1975, p. 108). A metapopulation is the subdivision of a population into networks of smaller, partly isolated, subunits (Harding and McNamara 2002). Such metapopulations can have extremely complex dynamics, and affects genetic differentiation, the spread of disease, and population stability, and many other important population processes (Harding and McNamara 2002). It is often assumed that a metapopulation will exhibit lower extinction risk because emigration from a source population can "rescue" a smaller population from extinction. However, in some cases, high migration rates can be deleterious to population persistence, a phenomenon that has been largely ignored in metapopulation conservation theory, and management regimes that ignore a significant antirescue effect can increase extinction risk.

Metapopulation concepts date at least to the early population genetics syntheses of Sewell Wright (1940), and the term was apparently first introduced by Levins (1969, 1970). Extinction risk is generally higher for metapopulations than for intact populations of equal size – often significantly higher (see below). One possible advantage of a metapopulation is the spatial "spreading of risk" from environmental fluctuations (den Boer 1968). This will be an advantage only if the spatial extent of the metapopulation is greater than that of the intact population. There is no evidence that this was ever the case for this species – instead, any metapopulation structure for this species arises from its extirpation from various areas and the creation of human caused barriers to dispersal as well as destruction of corridors of habitat. Also, spatial spreading of risk will be ineffective if environmental fluctuations are spatially correlated as is generally true for the mid-continental climate throughout the range of this species.

Various types of metapopulation concepts have been elaborated: Boorman and Levitt (1973) postulated a large source population with geographically static sub-population sinks which experience rapid and recurrent cycles of colonization, population turnover, and extinction. Levins (1970) postulated a system of interacting sub-populations where most of a fixed number of habitat patches were empty at any given time due to dispersal difficulty. In both models, the balance of immigration and extinction rates determines deme dynamism – the occupancy of patches and the size of colonies in each patch. Wilson (1975, p. 112) provides a simple comparison of these two models, and a tripartite spectrum of situations ranging from a mainland with satellites to equally distributed and sized isolates is illustrated in Poethke, et al. (1996, p. 86). Gill (1978) suggested a model in which patches were ephemeral to the point of altering reproductive success – a given patch would change from a source of emigration to a sink (no emigration) with a minority of patches serving as sources. Harrison (1991) noted that a decline of a species across a large region is often accompanied habitat fragmentation, a process strikingly similar to that confronting this species today.

Two aspects of patch geometry—area and isolation—currently dominate the field of metapopulation dynamics. Under this area-and-isolation paradigm, models commonly assume that the probability of local extinction decreases as patch area increases and that the probability of colonization increases as patch connectivity increases. Environmental variables other than patch area and isolation are assumed to have relatively little effect on metapopulation dynamics. Studies of a metapopulation of the butterfly *Speyeria nokomis apacheana* highlights the need for a broader view of metapopulation dynamics. In this system, neither occupancy nor turnover patterns were best modeled as functions of patch area or isolation. Instead, other measures of habitat quality explained the most variance in occupancy and turnover. Factors other than patch geometry, such as low habitat quality, may drive extinction processes in metapopulations, especially when environmental variability is present (Fleishman, et al. 2002).

Although patch occupancy surveys are often used as input to metapopulation models, field studies show that they are not valid substitutes for detailed experimental tests of population processes, particularly when conservation concerns are involved. Particularly disturbing is that static or declining patterns of patch occupancy can easily mistakenly attributed to dynamically stable metapopulation processes thus under-estimating extinction likelihood (Clinchy, et al. 2002).

Metapopulation biology is concerned with the dynamic consequences of migration among local populations and the conditions of regional persistence of species with unstable local populations. Well established effects of habitat patch area and isolation on migration, colonization and population extinction have now become integrated with classic metapopulation dynamics (Hanski 1998). This has led to models that can be used to predict the movement patterns of individuals, the dynamics of species, and the distributional patterns in multispecies communities in real fragmented landscapes.

Simple metapopulation models with random disturbances are often used in species conservation or management, and such models rely on the condition for persistence that the per-patch colonization rate be greater than the per-patch extinction rate. In more general models which incorporate the role of succession or patch age–dependent extinction, persistence can only occur if the per-patch colonization rate to be greater than the inverse of mean patch age, where age is the time since the patch became available for colonization (Hastings 2003). Currently persisting metapopulations may be persisting despite small colonization rates, and management approaches that do not consider the role of mean patch age may unnecessarily doom populations to extinction (Hastings 2003).

As with most species in severe decline, this species probably occupies all available patches of adequate habitat, thus the empty patch model of Levins (1970) seems inappropriate here – and is more likely applicable to colonizing species (Harrison 1991, 1994; Thomas 1994). As Thomas (1994) puts it: local extinctions "rarely generate empty patches of suitable habitat." Moreover, no suitable, but empty patches of habitat are known. Instead, some of the features of Gill's model seem to fit this species. Because of severe habitat degradation and various threats, there are no source populations left. As with various other species declining in abundance because of habitat loss and fragmentation, it is not likely that sink patches will become source patches – thus extinction processes will predominate over immigration processes. Further, it is known that for some species, conservation policies are "erroneously protecting sink habitats" and are "failing to prioritize key source habitats" (Heinrichs et al. 2010).

Model outcomes differ significantly between mainland-satellite metapopulation models (Fig. 3C in Poethke, et al. 1996, p. 86) and those with nearly equally distributed and sized isolates (Fig. 3A in Poethke, et al. 1996, p. 86). It is both intuitively obvious and easily shown mathematically that the latter situation carries higher risk of metapopulation extinction – local extinction rates must be smaller than colonization probabilities, else the entire metapopulation will become extinct (Hanski and Gilpin 1991; Poethke, et al. 1996, p. 87). This occurs because there is no large mainland source population which is so well buffered that it always serves as a source of immigration into patches. Instead, colonization probability is a function of the number of occupied patches. The number of linked patches is an important determinant of metapopulation extinction. For a finite number of linked isolates, metapopulation persistence depends on the number of interconnected local populations and the ratio of colonization probability to extinction rate must be 50% greater than with a very large number (near infinite) of isolates (Poethke, et al. 1996, p. 87).

In areas where the spatial cohesion of the habitat is below the critical level of metapopulation persistence, the expansion of ranges will be blocked. An increased frequency of large-scale disturbances caused by extreme weather events will cause increasing gaps and an overall contraction of the distribution range, particularly in areas with relatively low levels of spatial cohesion (Opdam and Wascher 2004).

Even when subpopulations are protected and appear viable, extinction risk may remain high. Metapopulation persistence is not possible below a threshold minimum metapopulation size – the number of subpopulations required to support metapopulation survival (Hanski, et al. 1996a). Extinction thresholds also result from the minimum amount of suitable habitat present in a region (Lande 1987). Moreover, time lags can exist for the effects of patch disturbances – in one annual species, the time scale of the time lags was two years. Fleishman, et al. 2002. Metapopulation concepts are of particular importance for this species because extinction can occur even when a great deal of suitable habitat exists if barriers prevent movement so that extinction rates of local patches exceed colonization rates (Levins 1969, 1970; Lande 1987).

Area Size

One of the most important insights of contemporary landscape ecology is that small areas of habitat are of limited value in conservation. Moreover, that value decreases as area size decreases and reaches zero before the size of the area reaches zero, thus creating a minimum threshold for the size of an area that will support a viable population. There is little value to a small conservation reserve or a series of small reserves that are not well-connected for the species at issue. Such "living museums" are necessarily subject to species-area and edge effects because of their isolation and small areal extent (Diamond 1975a; Wilcove, et al. 1986; Wilcove 1987; Wilcox and Murphy 1985; Harris 1984). It is well established that vulnerability to extinction is correlated with range size (Schwartz, et al. 2006), and species with small or shrinking range distributions are particularly vulnerable to extinction (Davies, et al. 2009).

Determination of minimum area sizes for various life-history stages and behaviors is complicated by the non-linearity of species responses to different sized areas. These incidence functions are "usually not linear" and instead show "sharp breaks" as area size varies (Wiens 1994, p. S99).

In determining the minimum area sizes needed for this species, it is important to also consider trophic linkages and other community and ecosystem processes. For example, areas might appear large enough for this species based on present habitat conditions, but be too small when disturbance processes are considered. Area size can strongly affect patch occupancy by birds (Ferraz, et al. 2007) and this effect likely applies to other species. Moreover, predators and predation are known to be an important effects determining minimum area size (Wilcove, et al. 1986).

Minimum Dynamic Area

Pickett and Thompson (1978) noted that reserve design should focus on the disturbance dynamics inside the reserve because extinction processes will predominate over immigration processes. This is intuitively obvious – by the time a reserve is established, the surrounding habitat will be degraded, and recolonization rates will be low or nonexistent. This is particularly true for this species because vast expanses of habitat have been degraded, and because successful reintroductions are unknown.

Pickett and Thompson (1978) proposed the term "minimum dynamic area" to denote the smallest area in which metapopulations could be viable given natural disturbance processes generating patches of suitable habitat, and the frequency and longevity of those patches. Meffe and Carroll (1997, p. 215) referred to the size a landscape must become before population dynamics within that landscape are essentially independent from events external to that landscape, and referred to the term "spatial autonomy" used by Pulliam, et al. (1995). Other authors have noted that "disturbance regimes … must be protected to preserve associated genetic, population, and assemblage dynamics" (Karr and Freemark 1985, p. 167; internal citations omitted). US Forest Service scientists have noted that such protection is a "key premise of ecosystem management" (Swanson, et al. 1994, p. 80). Pickett and Thompson's phrase connotes more readily the concept involved, at least as a noun, and that term is used in this review. The minimum dynamic area size must be substantially larger than the largest disturbance patch size (Pickett and Thompson 1978, p. 34). For this species, this means that areas to be preserved as suitable habitat must be substantially larger than the typical disturbed area in its habitat. Moreover, natural catastrophes will occur even if climatic regimes are stable (Noss 1992), so that sufficient geographic area must be included in reserves for these natural fluctuations in weather, as well as more broad scale climatic change. Both factors are substantial in the interior continental range of this species.

The minimum dynamic area must also include internal recolonization sources (or "hot spots" *sensu* Diamond 1975a), different ages of patch types, and separate minimum dynamic areas of each included habitat patch type. The latter two criteria have been discussed above in the various Habitat sections of this review. The former criterion is essentially a joint requirement for large and growing demes, with concomitant successful dispersal, and hence gene flow, from those demes – this requires that the macro-habitat (*sensu* D. R. Webb 1981) be permeable to dispersing individuals. Expansive reserves are needed because species are extirpated from even relatively large and protected reserves (Newmark 1987, Belovsky 1987). For some species, habitat protection plans have been based on 95% likelihood of persistence for 100 years (Noon and Murphy 1997, p. 437). For persistence over evolutionary time scales, even larger reserves will be needed, on the order of a few million mile^2 (Belovsky 1987, Soule and Terborgh 1999).

Habitat Connectivity and Permeability

The ability of individuals to cross intervening areas between habitat patches and to venture across various types of boundaries has been termed permeability. The likelihood of an individual crossing a boundary between two vegetation types has been termed "boundary permeability" (Wiens 1989b, II, p. 220), and this likelihood is a function of the sharpness of the boundary discontinuity or contrast between patches (Wiens, et al. 1985a). Such boundary discontinuities have also been termed hard edges or soft edges, depending upon the degree of discontinuity (e.g. Rolstadt 1991). These boundary effects determine the degree of gene flow among demes in metapopulations, and hence influence extinction probabilities. When discussion focuses on habitats or landscapes rather than species, the term connectivity is often used. Spatial areas are considered connected when at least certain life-history stages of various species can move from one to another. Connectivity thus summarizes movement probability among habitat patches in a landscape (Merriam 1991; Taylor, et al. 1993).

An important aspect of habitat connectivity is that different life-history stages of a species often require different habitat types, and must therefore transit to a different habitat at a different life-history stage. Human-induced disconnection between habitats used by different life-history stages of a species can significantly affect species viability (Becker, et al. 2007).

Species with highly fluctuating populations and high frequencies of local extinctions living in changeable environments are especially susceptible to curtailment of dispersal and restriction of habitat. Green (2003). While Green (2003) studied pond- and torrent-breeding amphibians, the study results are not taxonomically limited.

A special case of habitat connectivity is the movement corridor, which is a discrete area that often serves as a valuable linkage among core habitat areas (Beier and Noss 1998). However, corridors must be sufficiently wide, contain adequate dispersal habitat, and not have such hard or abrupt edges as to form barriers to dispersal. The effect of corridors on gene flow and population linkage is probably species specific (e.g. Mech and Hallett 2001). Soule (1991b) identified three types of movement needs: corridors for periodic migrations, corridors for access to seasonally moving resources and for movement among different patches, and for immigration into smaller, less viable populations. All are applicable to at least some populations. Corridors must not be too narrow, and broad swaths of intact habitat will best serve movement needs.

For this species, many types of objects, facilities, landforms, and land uses intervene between different areas of habitat. Moreover, these potential barriers to movement are of variable widths. At one extreme, dams completely eliminate movements. Burned areas significantly alter migration (Fischer, et al. 1997, p. 89). Very narrow facilities such as powerlines and high speed highways (such as Interstate freeways) may also serve as dispersal barriers. Reservoirs and lakes may constitute barriers. In conducting further research into barrier effects, it may prove useful to ordinate studies along the factors such as the width of the hypothesized barrier, the shape ratio or length of the barrier in relation to its width ("peninsula effect," *sensu* Emlen 1978), the visual appearance, and the degree of risk in crossing the barrier. The degree of risk may need to be evaluated as apparent risk. Even if individuals are able to cross certain barriers, they may be reluctant to do so because of predation risk.

Another aspect of the use of different habitats and consequent connectivity is that different habitats, sometimes in widely dispersed geographic areas are required for different life history stages. Species must then migrate from type of habitat to another, and the intervening areas must facilitate such migration. Prugh et al. (2008) compiled and analyzed raw data from 785 animal species in over 12,000 discrete habitat fragments on six continents. The quality of the matrix surrounding a fragment had a greater influence on persistence than fragment size and isolation. Moreover, loss and degradation of wintering habitat is known to have significant effects on reproduction, even in species which winter and breed thousands of miles away (Norris, et al. 2004).

Habitat Vegetation Analysis

Traditional Clementsian succession theory (Clements 1916, also known as monoclimax theory) was applied to arid ecosystems early on (Sampson 1919) and even forms the basis for government manuals and early range management textbooks (Fleischner 1994). At its simplest, community development theory encompasses essentially monotonic successional changes from priseres to cliseres (cliseres are often dubbed "old growth" or "ancient forests" for various forest ecosystems). Typically, only a single pathway to climax seral stages was recognized, and the process envisioned was one of "progress" towards that "vegetative goal." Clementsian succession concepts no doubt were influenced by then current notions of evolutionary and social "progress" and goal directedness. Ellison (1960a, 1950b) summarized this development. Computer simulation models predict that invaded and degraded areas in the Snake River region cannot recover in less than a century and then only if optimal moisture, fire control, and seed dispersal conditions are present (Knick 1998, USDI 1996). A large body of theory on community development, together with numerous experimental tests and extended field data, exist for forest ecosystems, particularly North American forests. This body of work has led to numerous predictions on the effects of forest management practices on birds (such as spotted owls), various forest carnivores (wolverine, fisher, lynx), and on fungi, amphibians and many other species. Donahue (1999) summarizes successional theory development.

What is key to such state and transition models of community development is that if correct, then grazing or alterations to the fire regimes in areas of present or former habitat may produce situations from which the community cannot recover naturally (West 2000). Recent government studies suggest that this is indeed the case (USGS 1999).

Threat Analysis

A wide variety of threats affect this species. Most threats are anthropogenic (human caused), although some are natural or only indirectly caused by human activities. One range-wide effect is climate change (de Hann, et al. 1996). Although vehicular and power plant CO_2 emissions receive the bulk of the attention in the popular news media, methane has 21 times the radiative absorption capacity of CO_2 (Kauffman and Pyke 2001, p. 47). Anthropogenic impacts are qualitatively different from natural disturbances because "human impacts tend to be chronic," cumulative and persistent (Aplet and Keeton 1999). It is perhaps not surprising that anthropogenic threats to this species are so serious. Anthropogenic threats endanger many species – perhaps most of the species extant. This is to be expected because approximately 40% of all net primary productivity (NPP) for the entire terrestrial surface of Earth is sequestered for human use (Vitousek, et al. 1986). NPP is roughly the food supply for all animals and decomposers (Ehrlich and Wilson 1991). Most threats occur throughout the range and all range-wide threats are caused either directly or indirectly by humans.

This species, like 85% of the species analyzed by Wilcove, et al. (1998), is greatly threatened by habitat degradation and destruction. For many taxa, habitat destruction is "the most significant cause of endangerment" (e.g. for birds as a group; King 1977, p. 10, evaluating IUCN Red Book listings; accord Collar, et al. (1994)) and is likely a strong effect on mammals and other taxa as well). Every category of habitat destruction and degradation identified by Wilcove, et al. (1998), is also a threat to this species. The categories identified by Wilcove, et al. (1998) include: agriculture, livestock grazing, mining and oil and gas exploration and development, logging, infrastructure development, road construction and maintenance, military activities, outdoor recreation, off road vehicle use and developments, water developments, dams, pollutants (including pesticides, herbicides, and pollutants from mining and oil and gas developments), land conversion, and disruption of natural fire ecology.

It is particularly important to analyze threats in combination, and not merely separately. Threats in combination can have synergistic effects (the cumulative effect can be greater than the additive sum of the parts). Such synergy of effects is so common in ecology that it is discussed in major texts (e.g. Meffe and Carroll 1997, p. 152-154), and such extintions are common – one striking example is that of the Heath Hen (Simberloff 1986b). Porter, et al. (1984) demonstrated a method of statistical analysis to simplify the analysis of such complex interactions.

Importantly, many threats are correlated. For example, military training exercises are both directly harmful, and also increase the likelihood of fire. Threats can inhibit population processes even without direct death or injury to individuals. For example, individuals may practice behavioral avoidance of intrusive threats such as noise sources, antennas, transmission towers or other raptor perches. This avoidance can disrupt dispersal patterns, foreclose the use of traditional breeding sites, or otherwise reduce population viability even without noticeable increases in mortality rates near the intrusion. The amount of habitat affected by such factors is huge, and is continually increasing.

As detailed in previous sections of this review, this species possesses certain demographic, physiological and ecological characteristics that renders it susceptible to extinction and extremely difficult to reestablish after extirpation. Likewise, the species habitat is easily damaged, and recovers only slowly if at all from damage. These sections are included as threats to the species by reference in this petition and the Service is requested to give its comprehensive consideration to the suite of characteristics rendering this species susceptible to population declines and extinction.

A species must be listed if it "is endangered or threatened" because of any "natural or manmade factors affecting its continued existence " 50 C.F.R.§ 424.11(c)(5); 16 U.S.C. § 1533(a)(1)(E). This section of the ESA is meant to incorporate any factors not explicitly listed in the four sections preceding that section in the statute. The Secretary must conduct a "review of the species' status." 50 C.F.R. § 424.11(c). The determination to list the species must be made "solely on the basis of the best scientific and commercial data." 16 U.S.C. § 1533(b)(1)(A); 50 C.F.R. § 424.11(b). The Service "cannot ignore available biological information." Connor v. Buford, 848 F.2d 1441, 1454 (9th Cir. 1988). The Secretary may not consider actual or "possible economic or other impacts" in the listing decision. 50 C.F.R. § 424.11(b). To make the Service's task easier, and to insure prompt action on the petition, Petitioners have prepared a very thorough and comprehensive status review for the Service. Petitioners incorporate all parts of this review, particularly the section designated "Population Mechanisms & Vulnerability" and its sub-sections, into the threat analysis as "other natural or manmade factors affecting" the continued existence of this species.

Effects on Habitat and Range

A species must be listed if it "is endangered or threatened" because of "present or threatened destruction, modification, or curtailment of its habitat or range." 50 C.F.R.§ 424.11(c)(1); 16 U.S.C. § 1533(a)(1)(A). The Secretary must conduct a "review of the species' status." 50 C.F.R. § 424.11(c). The determination to list the species must be made "solely on the basis of the best scientific and commercial data." 16 U.S.C. § 1533(b)(1)(A); 50 C.F.R. § 424.11(b). The Secretary may not consider actual or "possible economic or other impacts" in the listing decision. 50 C.F.R. § 424.11(b).

The range of this species has been significantly curtailed in historic times. Much of the extant range will not be available to this species because it is (1) degraded, and (2) even if not degraded, is too small, or (3) is subject to proximity effects (too close to powerlines, roads, or trees) for individuals or populations to use. Habitat may also be subject to proximity effects if it is too far away from another habitat type for individuals to use, or to use without significant predation or other dangers. These range contractions are ongoing, and are virtually certain in the future. Virtually all habitat has been degraded and much has been destroyed. The threats which have produced habitat degradation are ongoing, and additional habitat modification and degradation is certain. Many, but not all threats, are anthropogenic. Human impacts tend to be chronic, arising from cumulative and persistent actions over broad areas (Aplet and Keeton 1999; Johnson, et al. 1994). Species behavior is affected by heterogeneous and changing landscapes, especially human-induced environmental changes (Knowlton and Graham 2010).

Habitat and Range

The Cahaba River begins in the Valley and Ridge region, and runs south of Birmingham, Alabama to its junction with the Alabama River west of Selma, Alabama. It is one of the most biologically diverse rivers in the United States, but is threatened by at least 14 non-native species, pollution, transportation, erosion, and hydropower dams. Lydeard and Mayden (1995). These threats have already caused severe declines in what was originally a highly diverse fish fauna and snail fauna, as well as the extirpation of nearly a quarter of the species of mussels. The US EPA has found that the habitat of this species is threatened by sediment, excess nutrient pollution and algal growth. http://www.epa.gov/region4/sesd/reports/2002-0809/addendum.pdf

The US Dept. of the Interior has noted that the Cahaba River drainage "is heavily affected by nonpoint-source runoff, siltation, acid mine drainage, pollution from wastewater treatment plants, and water withdrawn for domestic water use." (Bogan, et al. 1995). Service employees also called for "immediate action to prevent further declines and extinctions." Id. However, no action was taken. Water quality in the Cahaba Basin has been degraded, and continues to be degraded, by a variety of factors (Herrig and Shute 2002, Buckner et al. 2002).

The Service admits to serious water pollution problems in the habitat of this species: "In the Cahaba River Basin there are 10 municipal wastewater treatment plants, 35 surface mining areas, one coalbed methane and 67 other permitted discharges…." US FWS, 1992. Cahaba Shiner Recovery Plan, p. 3. Southeast Region, US Fish and Wildlife Service. Atlanta, GA.

Proliferation of development in the area is causing increased recreational developments and activities. The development of retirement communities and recreational areas is increasing in the southeast and is threatening freshwater biodiversity. Housing developments, strip malls, and resorts are being constructed in rural areas, and small towns are now burgeoning in previously undeveloped areas including the Cahaba River headwaters outside Birmingham (Buckner et al. 2002). Numerous dams are being constructed on headwater streams, often in areas that were recently remote and inaccessible, with resultant impacts on aquatic species (Buckner et. al 2002). The development of housing and recreational facilities on lakeshores and in riparian areas results in the degradation of water quality and aquatic habitat (Tennessen 1997).

The Cahaba River is impacted by disturbance, sediment, and other pollutants. http://www.epa.gov/region4/sesd/reports/2002-0809.html The river receives domestic and industrial wastewaters, and there are at least 103 industrial discharge permits in the Cahaba Basin, which release a variety of toxic metals, chemicals and other substances (US Fish and Wildlife Service 2007). There are six municipal wastewater treatment plants in the upper basin with a combined discharge of 19 million gallons a day. Water quality tests show high levels of phosphorus, nitrogen, heavy metals, low dissolved oxygen, organic enrichment, siltation, and chemical spills in the upper basin. Water quality is degraded by historic and current coal mine drainage. The US FWS (2007) states: "Characterization of Cahaba River water quality by the U.S. Environmental Protection Agency (Howard et al. 2002) documented the following problems in the basin:

- Excessive sedimentation and nutrient enrichment are affecting watershed biology;
- A decline in pollution-sensitive fish species with a concomitant increase in pollution-tolerant fish species;
- A prominence of the filamentous green algae Cladophora, which is often associated with nutrient enrichment and nuisance conditions;
- Total phosphorus and total nitrogen ranged from 12 to 960 ppb and 230 to 21,094 ppb, respectively (12 ppb TP and 230 ppb TN considered adequate);
- Excessive sediments have degraded and altered benthic community and species diversity in portions of the river;
- Dramatic increase in "disturbed land" in the basin since 1990; and
- High incidence of NPDES permit violations for nutrient or nutrient related parameters over the last several years."

Grazing

Grazing of domestic livestock reduces the forest understory, alters tree recruitment, alters fire regimes, and entices humans to cut down adult trees. BLM acknowledges that grazing is the major activity affecting wildlife habitat on its lands (USDI 1994, Draft EIS, p. 25-27). Grazing is the predominant land use on 70% of the land in the 11 states west of the 100th meridian (Cooperrider 1991). Livestock grazing is the most ubiquitous land use in western North America, and a study at one of the largest grazing exclosures in the American West, showed that long-term protection with no or reduced grazing significantly enhanced species richness (Floyd, et al. 2003).

Grazing effects can be categorized into two major groupings. First, grazing directly affects vegetation structure, soil characteristics, and other habitat characteristics. Second, when lands – even public lands – are used for grazing, they are usually altered by one or more "treatments" to produce more grass and forb cover for the livestock and to reduce trees. These treatments involve a vast agro-industrial infrastructure of fences, roads, dams, water developments, powerlines, vegetation manipulations and more. Although much of the grazing effects in the West takes place in grassland, desert and shrubland ecosystems, significant effects are known in ponderosas and oak forest types.

Indeed, livestock grazing is the most widespread influence on native ecosystems of western North America (Fleischner 1994, Wagner 1978, Crumpacker 1984). In 16 western states, approximately 165 million acres of BLM land (94% of the total BLM land) and 103 million acres of USFS land are grazed by 7 million head of livestock, primarily cattle (Fleischner 1994, GAO 1988b) and on 70% of all lands in the west (GAO 1988b). Approximately 70% of the of the West is grazed, and this figure includes the entire area of lakes, rivers, forests, parking lots and urbanized areas (Crumpacker 1984; Fleischner 1999, p. 64). Even 35% of all wilderness areas have active livestock grazing allotments (Reed, et al. 1989). The 35% figure represents an average over the entire US, so the figure for the West is probably substantially higher. Effects on wildlife are severe: Smith (1977) found that grazing is "the single most important factor limiting wildlife production in the West," and Cooperrider (1991) noted that grazing is "one of the primary threats to biological diversity."

Grazing changes habitat structural characteristics and species composition in both upland and riparian sites, spreads exotic invasive species, and causes erosion, degradation, and shrub encroachment into riparian areas (Rasmussen and Griner 1938; Patterson 1952c; Autenrieth, et al. 1982; Klebenow 1982, 1985; Call and Maser 1985, Belsky, et al. 1999).

The amount of grazing on public lands is under-estimated by the standard measure of grazing intensity, the animal unit month (AUM), essentially the forage consumed by a cow and its calf in one month. Many, if not most, of the existing grazing allocations by public agencies were determined in the mid-1950s and were based on cows that ranged in size from about 850 to 1000 pounds. In the last 40 years, however, genetic techniques have been used to breed significantly larger cattle, which now often have body masses of 1,350 pounds or more (GBCP 1997, p. 43). This factor alone is a 35% to 59% increase. Moreover, the calves are also larger, and mature more rapidly (GBCP 1997, p. 43). These two factors increase the plant consumption for each cow and calf, so that the consumption today may approach twice that per AUM when grazing allocations were determined. Even if the grazing allocations today did not exceed those of the mid-1950s, those previous allocations were too high.

However, cattle grazing has actually increased in many states (Rich 2001). While not commonly appreciated, most beef cattle production is located in the southeastern states. Domestic livestock grazing reduces water infiltration rates, reduces cover of herbaceous plants and litter, disturbs and compacts soils (creating microsites for invasion of exotics), and increases soil erosion, which reduces the productivity of vegetation. A large and robust literature exists on these alterations of ecosystem processes, much of it based on exclosure studies and other experimental manipulations. Unfortunately, many of these studies involved areas too small to understand ecosystems processes or lasted only a few years (Saab, et al. 1995). Exclosures should be several hundred hectares in size (Rotenberry 1998, p. 270), and studies need to continue for decades. Belsky and Blumenthal (1997) recently reviewed this literature.

Grazing retards vegetative recovery from fires, grazing itself, revegetation after strip mining and from other disturbances. Grazing can also completely prevent any vegetative recovery after a disturbance, permanently altering the ecosystem to one of lower productivity and lower vegetative cover, reduced biomass and biodiversity, soil deterioration, and other aspects of desertification (Sheridan 1981, Noss and Cooperrider 1994). Overgrazing is "the most potent desertification force, in terms of total acreage affected, within the United States" (Sheridan 1981). An estimated 1.1 million mile^2 (36.8% of North America's arid lands) have already undergone "severe desertification," and desertification of an additional 10,500 mile^2 has been "very severe" (Sheridan 1981). Moreover, twice that area is threatened with desertification (Sheridan 1981). Indeed, there are few lands that have not undergone at least some desertification (Noss and Cooperrider 1994).

Recovery time of habitat once grazing is halted can require decades or centuries. However, removal or reduction of cattle grazing is known to result in increases of herbaceous vegetation (Ellison 1960a, 1960b; Sneva, et al. 1984; Miller, et al. 1993, p. 126). Grazing also leads to feedback loops in ecosystems, and these feedbacks can damage native species. For example, grazing increases the frequency and severity of grasshopper outbreaks. Managers then use more pesticides to control the grasshoppers, harming other species. Grasshoppers then proliferate even more, amplifying the feedback cycle. Interaction effects abound in such systems – for example, the overgrazing starves other species, putting them into poor nutritional status, where they are likely more susceptible to pesticide effects. The Service will need to thoroughly evaluate the magnitude of all such interactions and feedback effects.

Apologists for livestock interests often note that heavy grazing in the past may be affecting ecosystem health more than current grazing. Even if correct, this assertion is irrelevant to a listing determination – if the effects on habitat create threats or endangerment then listing is required, even if the ultimate causes of those threats are in the past. Second, because ecosytems have been degraded and destabilized, even lower levels of grazing in the present can produce severe effects.

Livestock grazing in stream and riparian ecosystems can negatively affect water quality and quantity, channel morphology, hydrology, soils, instream and streambank vegetation, and aquatic and riparian wildlife (Belsky et al. 1999).

This section discusses the effects of grazing *per se*. However, it is important to realize that and livestock operations have many more negative effects besides those due to grazing. For example, in order to graze an area with domestic cattle, the area will usually be fenced, water developments will be installed, and unintended effects will also result, such as the invasion of exotic plants and insects, soil erosion, lowering of water tables and the dewatering of wetlands. Thus grazing cannot be thought of as merely a few cattle or sheep as depicted in a gentle pastoral painting. Instead, these effects result from the conversion of native ecosystems to agro-industrial uses, and are discussed in other sections below.

Trampling of Vegetation

Livestock trample undergrowth that may be needed for both food and shelter. Trampling of vegetation by grazing livestock is detrimental to most upland wildlife (Fleischner 1994, Belsky and Blumenthal 1997). Cattle generally trample riparian areas first, moving into the uplands after they have depleted the valley bottoms and areas near water sources (Stoddard, et al. 1975). However, removal of cattle from near streams to protect fish would force cattle to make greater use of uplands.

Alteration of Ecosystem Processes

Grazing also affects plant community composition by fostering interspecific competition between plants tolerant of grazing and those less tolerant. Grazing is known to cause large increases in small herbivore populations, such as grasshoppers and jackrabbits (Miller, et al. 1993, p. 130). Grazing creates microsites for germination of invasive weed seeds. Despite much talk about using livestock to "trample native seeds into the soil" for germination, there is little data to support such a notion (Miller, et al. 1993, p. 130). It is obvious that such processes would be superfluous for native plants, since they did not evolve with large mammals.

Grazing alters nutrient cycles, water cycles, fire return intervals, and energy flow (Miller, et al. 1993, p. 130). Grazing also reduces competition of grasses and forbs with tree seedlings, promoting tree establishment (Belsky and Blumenthal 1997).

Invasion of Exotics

Livestock grazing is known to destabilize plant communities by increasing their susceptibility to invasion by exotic alien species (Fleischner 1994). Livestock help spread exotics by (1) dispersing their seeds in fur, in mud on their hooves, and in dung, (2) creation of microsites for establishment of exotics (Gould 1951, Mack 1981), and (3) by reducing the competition from native species by eating them (Fleischner 1994). Alien grass invasions in North America are closely associated with grazing (D'Antonio and Vitousek 1992).

Livestock are also known to severely degrade cryptogamic crusts (Fleischner 1994). Cryptogamic crusts (which consist of bacteria, blue-green algae, fungi, mosses and lichens) are important in providing favorable sites for the germination of vascular plants (St. Clair, et al. 1984) and have important effects on soil hydrology (Fleischner 1994), on stabilization against wind and water erosion, on retention of soil moisture, on reduction of wind erosion (Belnap 2000, p. 57), and on promoting equable soil temperature regimes (Belnap 1993, 1994; St. Clair and Johansen 1993; Kaltenecker 1997). Crusts are also known to enhance the survival and biomass of native perennials in cool deserts (Belnap 2000). Once damaged, cryptogamic crusts can require 50 to 100 years to recover (St. Clair and Johansen 1993).

Numerous studies show that livestock grazing contributes to both the invasion and dominance of noxious weeds (Lacey 1987, Bedunah 1992, Hobbs and Huenneke 1992, Dwire, et al. 1999). Cattle can transport invasive plant seeds and other pest propagules into nearly all areas, except those with the steepest slopes and areas farthest from water (Daubenmire 1970, Belsky and Gelbard 2000). A single cow can transport over 900,000 viable seeds per season (Dore and Raymond 1942). Other studies confirm the ability of cattle to transport viable seeds in dung and on their coats (review by Belsky and Gelbard 2000). Besides promoting the invasion of noxious species, cattle create an environment that is susceptible to invasion (review by Belsky and Gelbard 2000).

Conversion of Habitat

Conversion of habitat to other uses, such as agriculture or housing, completely eliminates that habitat. Consequently, species are extirpated from that area (Swenson, et al. 1987). Even minor reductions in area from habitat conversion can cause large reductions in populations. Worse, conversion is a permanent habitat change and precludes restoration or recovery of populations (unless one contemplates completely plowing up fields, burning down suburban houses, and replanting forests). As such, these are the only threats that are likely to be greater than livestock grazing. Conversion of habitat has limited the amount of habitat available throughout the range. Settlement of, and agriculture on, habitat began in the mid-1800's and was enhanced by passage of the series of acts known as the Homestead Acts, beginning in 1862 (Todd and Elmore 1997). Much of the land originally homesteaded was ploughed and trees removed. Some of this land reverted to the public domain after agriculture and homesteading failed. Irrigation projects, including both taxpayer subsidized large dams on major rivers, and small dams on streams, made irrigation water unusually cheap.

Habitat can also be converted by the invasion of exotic alien species. Initially, these species merely degrade habitat, but after a period of time some invasions can destroy habitat, rendering it totally unsuitable.

Agriculture is the most widespread form of habitat destruction for all species, followed by commercial development (Wilcove, et al. 1998). For this species, agriculture (including grazing) is clearly a widespread threat to the species existence, with development being the worst threat in some relatively local areas, such as near urban centers and in scenic areas near roads. Habitat conversion has been overwhelming in some areas, such as various scenic river and feeder creek valleys.

Vast areas of natural communities have been converted to anthropocentric land uses, especially for agriculture. In 1989, USDA Soil Conservation Service analyses showed that cropland accounted for 421 million acres, and pastureland for 132 million acres (Langner and Flather 1994, p. 14). Although much of that conversion took place in the East and Midwest, native grasslands are known to have suffered much higher conversion rates than have forests (Langner and Flather 1994, p. 14). Much agriculture does not produce food for human consumption, but instead merely feeds cattle. Approximately 37% of the Columbia Basin has been converted from native vegetation to agricultural use (Quigley and Arbelbide 1997b, p. 459). "Agricultural and urban development, livestock herbivory, the introduction of exotic plants, and changes in disturbance regimes have resulted in unprecedented changes." (Quigley and Arbelbide 1997b, p. 765).

Habitat Conversion to Agriculture

It is not surprising that agriculture, including grazing, is so damaging: agriculture is the key variable explaining the endangerment patterns of birds, mammals and plants (Dobson, et al. 1997). Wilcove, et al. (1998) identified habitat loss as the single greatest threat to biodiversity, as have virtually all other authors (Noss, et al. 1995, p. 2; Noss and Peters 1995, p. 45; Ehrlich and Ehrlich 1981; Ehrlich and Wilson 1991; Diamond 1984a, 1984b). Besides the direct removal of habitat, agricultural fields serve as barriers to dispersal and isolate populations (Mader, et al. 1990; Mader 1984). By attracting individuals into areas with little predator cover, fields may also create high risk "kill zones." Agricultural conversion is also accompanied by the vast infrastructure of modern corporate agriculture – fences, electrical wires and poles (via chemically treated posts and poles), herbicides, and water developments – all are threats to this species.

Development and Habitat Conversion to Suburbs and Ranchettes

Suburbanzation (including dispersed ranchettes and vacation homes) as well as the buildings and home sites of working ranches also directly remove habitat. Although buildings and development may affect much more habitat than they directly displace because of their proximity to other habitat and the introduction of domestic animals, disturbance, and disruption of population connections. Suburbanzation also tends to occur near water, where higher quality soils are located, so its population impacts are also often larger than the amount of habitat affected. Riparian areas are particularly vulnerable to fragmentation associated with residential development (Smith and Wachob 2006).

The West is the fastest growing area of the country, and has a population growth rate that is <u>greater</u> than that of Mexico (Knight 1997). Moreover, suburbanization rates are disproportionately greater than population growth. In Oregon and Washington the area of developed land grew faster than population in the decade leading up to 1992 (Durning 1996, p. 26). The amounts of habitat that are degraded by suburbanization can be startling. Only one western state, Oregon, has land use controls and urban growth boundaries. However, even in Oregon, these land use controls are frequently violated in the part of the state containing required habitat. Worse, Oregon's land use controls have not demonstrably reduced conversion of either natural areas or of agricultural lands (Kline and Alig 1999).

Suburbanization also unleashes large numbers of domestic pets, which can prey upon or otherwise disturb both adults and their young. Domestic dogs (*Canis familiaris*) and cats (*Felis catus*) commonly forage along edge habitats near human dwellings (Oehler and Litvaitis 1996). Free-ranging domestic cats kill 1.4 to 3.7 billion birds and 6.9 to 20.7 billion mammals annually in the United States. Un-owned cats, as opposed to owned pets, cause the majority of this mortality. Such free-ranging cats cause substantially greater wildlife mortality than previously thought and are likely the single greatest source of anthropogenic mortality for US birds and mammals. Developments not only remove habitat, they also constitute direct threats to this species.

Suburbs do not exist in a vacuum, but are tied to urban centers by networks of roads, powerlines, and pipelines. This infrastructure also affects this species, by directly removing their habitat, by degrading adjacent habitat, and by fragmenting habitat.

Mining as Habitat Conversion

Mining, particularly strip mining, directly eliminates habitat. Mining operations also release a diversity of pollutants, many of them toxic, and create additional roads and additional traffic on existing roads. Surveys and explorations for mineral, oil, gas, and coal deposits also entail habitat degradation. If the area is adequately reclaimed, individuals may be able to reestablish populations after some decades, but only if migration corridors from population source areas are available.

Spatial Analysis of Habitat Conversion

Remote sensing data can be used to gain fairly accurate estimates of the amount of landscape currently in agriculture, suburbanized, or mined. Indeed, wholesale habitat conversion is the threat that is most susceptible to spatial analysis, because of the ease of acquiring geo-data via remote sensing. The amount of landscape that was formerly habitat can be estimated from potential natural vegetation mapping, and the fraction of potential habitat that might not have been available in historical times because of disturbance induced displacements from its potential seral stage (e.g. by fire) can be estimated by paleoecological techniques, such as pollen counts. Many of these data probably already exist, and a good portion of those data may be in spatial form, suitable for use in a Geographic Information System (GIS). GIS is now commonly used for resource conservation and reserve design (ASPRS/ACSM/RT 1992). A habitat quality model suitable for remote sensing (thematic mapper) and GIS applications has already been developed (Homer, et al. 1993; Edelmann, et al. 1998). Their model does not appear to include any landscape features such as proximity analysis, barriers, shape ratios or other characteristics, but is an important first step. For example, the Interior Columbia Basin Ecosystem Management Plan (ICBEMP) generated a wealth of spatial data that may be applicable to habitat studies, even though that data probably overestimates the availability of habitat to particular wildlife species. Thus, the degree of habitat conversion can be estimated fairly accurately without a great deal of effort. Nonetheless, a higher priority may exist for population viability analysis and habitat fragmentation studies of those populations still existing, than for studies of the importance of previous effects on population decline. Results from remote sensing can be valuable for aquatic species as well, because forest canopy or effects from other areas often permeate into aquatic environments.

Landscape data on habitat types and land uses will be important for recovery planning. Such datasets are available from a number of sources. Most USFS data are collected for conifer forestry, such as Timber Survey maps based on aerial photointerpretation, and more recent GIS products. Pipeline and utility companies often have maps of their facilities in GIS form. BLM has variable coverage of habitat and vegetation maps, and has not yet gathered comprehensive GIS information. Similarly, USGS-BRD (Biological Resources Div., US Geological Survey) has land use and land cover maps, and has a GIS project underway (S. Knick, personal communication). The EPA compiles ecoregion maps and the Natural Resources Conservation Service (NRCS), formerly the Soil Conservation Service (SCS) has soil type maps, although agricultural use mapping is not comprehensive. NRCS has also compiled a National Resources Inventory, which contains information on land uses. State natural heritage programs also have various data, often in GIS form. These and other sources are described by Scott, et al. (1993, p. 15-16) and in more recent sources. Many states have completed or are near completion of GAP analysis, and various GIS layers will be useful for habitat analysis and protection. GAP analysis is not a panacea however. Two major limitations on the use of GAP analysis are that location data for species typically lack specificity and that data layers on species distributions are typically mere estimates of habitat affinities (Schwartz 1999, p. 98).

Pursuant to Executive Order 12906, the Federal Geographic Data Committee (FGDC) of USGS has set up a Clearinghouse for geospatial data at: http://www.fgdc.gov/clearinghouse/clearinghouse.html.

Predation

Direct removal of individuals from a population, e.g. by predation or hunting, can obviously cause endangerment of a species. However, predators do not need to actually kill or even physically harm their prey to cause population declines; instead, the prey's perception of predation risk reduced the number of offspring produced per year by 40% in a bird species (Zanette, et al. 2011). This strong result that the perception of predation risk is itself powerful enough to affect wildlife population dynamics, and should thus be given great consideration in vertebrate conservation and management (Zanette, et al. 2011).

Predation, and its effects on population persistence, occurs in a particular environmental context. For example, habitat degradation can expose prey to predators, as can habitat fragmentation as the need for individuals to transit outside of core habitat may expose them to detection by predators. And, it is known that climatic disturbance compounded by predation can cause extinction of prey populations (Schoener, et al. 2001).

Disease and Parasitism Effects

Parasites and diseases differ from predation in the sense that the "predator" is resident inside the body of the target organism, and often inhibits reproductive output without killing the organism. Novel diseases can have serious impacts on naive wildlife populations, which in turn can have substantial impacts on ecosystem integrity (Frick, et al. 2010). Changes in environmental conditions, such as global warming, allow novel diseases and disease vectors to move into new areas. Rapid population declines or species extinctions from infectious diseases will increase with warming (Harvell 2000). Infectious disease is one of the top five causes of global species extinctions; however, Smith, et al. (2006) note that there are temporal biases and some uncertainty in the data. Disease and infectious organisms can have profound effects on animal populations and these effects should not be neglected in the Service's analysis of endangerment merely because of some uncertainty as to effects. Moran 2002. Novel diseases or diseases that spread into new areas are of particular concern. Emerging infectious diseases have caused some bird species to decline by up to 45% and few species have recovered (LaDeau, et al. 2007). Climate change can affect infectious diseases in a nonlinear fashion, and infectious diseases are expected to invade more northern areas, such as the United States (Lafferty 2009).

Diseases can also cause species extinction even without directly infecting the species at risk, if they damage that species habitat. For example, Sudden Oak Death (SOD), a disease induced by the fungus-like pathogen *Phytophthora ramorum*, threatens to seriously reduce or eliminate several oak species endemic to the west coast of North America (Monahan and Koenig

2006). Even if the oak trees are not killed, SOD reduces their output of acorns, resulting in population declines in species that depend on those acorns as a food source.

Invasive Species

Overall, invasive species are a major cause of species extinction, second only to habitat loss (Flather, et al. 1994; Wilcove, et al. 1998). An exotic "invader" is any organism that is able to colonize and persist in an area where it has never existed before (Mooney and Drake 1986). Similar definitions have been offered by Pres. Executive Order 13112, and by the Service (invasives.FWS.gov). The US Office of Technology Assessment estimated that at least 4,500 alien species have invaded the United States and established sustainable populations here (Office of Technology Assessment 1993; Campa and Hanaburgh 1999). Invasions by exotic species can reduce biodiversity, spread disease and alter ecosystem processes and constitute serious threats to native ecosystems (Liebhold, et al. 1995; Vitousek, et al. 1996). Invasive species can alter ecosystem processes and components by disrupting normal rates of system level resource supply, altering trophic structures, and altering disturbance regimes (D'Antonio and Vitousek 1992, p. 64). They also can destroy wildlife habitat and damage agricultural crops, rangelands, forests and wildlands (Vitousek, et al. 1996). Bird families with the highest extinction risk are primarily associated with threats posed by invasive species, once species richness and phylogeny are taken into account (Claveroa, et al. 2009).

Infestations of noxious weeds alone are increasing by at least 14% annually (R. Johnson, et al. 1999, p. 13). Over 860 exotic plant species have invaded the Pacific Northwest (Rice 1994 in Quigley and Arbelbide 1997c). Invasive plant species have a large effect in many areas. Not all invasive alien species appear on state or federal noxious weeds lists – in some cases, control is so difficult or economically burdensome that regulatory powers have simply given up on control (Pyke 2000, p. 45). This abdication of responsibility – though the frustration is perhaps understandable – constitutes an inadequate regulatory mechanism.

Attempts to control invasive species can expose individual animals, and plants they need for food and cover, to herbicides, pesticides, and endocrine disruptors. Alien species are often associated with agricultural lands and invade natural areas from agricultural lands (Janzen 1986; Alberts, et al. 1993; Smallwood 1994). Livestock are a known vector for the spread of alien exotics.

The term Invasive Species can also be applied to native species that escape natural biological controls. For example, overgrazing, drought, and destruction of competitors and predators by pesticide spraying contribute to grasshopper outbreaks (Lockwood, et al. 1988). Domestic livestock are the most damaging alien species in western North America, with the possible exception of hominids *per se*, but both are discussed in their own sections of this review.

Invasions by exotic organisms are facilitated by disturbance (Elton 1958 (reprinted 2000), Mooney and Drake 1986). Disturbance to the soil surface, such as is caused by cattle, is known to facilitate the establishment and spread of invasives. Pickett and White (1985a) defined disturbance as any discrete temporal event that disrupts ecosystem, community or population structure and changes resources availability or the physical environment. Grime (1979) views disturbance as any process that removes or damages biomass. The disturbance caused by roads, ORVs, and livestock grazing allows exotic invaders access to undisturbed ecosystems. Global climate change is expected to accelerate the spread of invasive species (Field et al. 2007).

Agriculture, including livestock grazing, is the principal source of nonnative plant introductions (Miller, et al. 1994; Pieper 1994). As the USGS notes, one of the most important factors "making an ecological community susceptible to invasion by nonindigenous species is the level of human-induced disturbance" (Williams and Meffe 1998, p. 120; Hobbs 1989). Livestock grazing and other threats have substantially increased the invasibility of habitats. For example, creation of microsites by grazing and other soil disturbing activities is a major factor in the invasion of alien plants (Mack 1981).

Noise, Acoustic Interference, and Disturbance

Noise can interfere with mating displays (Morton 1975), reduce the ability to detect predators, and cause individuals to move away from the noise source (Witkin 1977, Wiley and Richards 1978, Richards and Wiley 1980). In some species, male mating success is strongly related to a very small number of different acoustical components of display (Gibson 1996a, 1996b; Gibson and Bradbury 1985), thus noise can have profound effects on population viability. Contact enhancement between a mother and newborns is often mediated by acoustical signals (Girard 1937, Patterson 1952c). Therefore, noise effects will be particularly important during the brooding period. Animals that vocalize at frequencies similar to those of traffic noise (<3 kHz) are 10 times less likely to be found in noisy than in quiet plots because the noise masks their acoustic signals (Goodwin and Shriver 2011). The same effects are likely to occur for insects, amphibians and other species using vocal communication or acoustic detection of predators in the same frequency ranges.

Noise and other disturbances are known to cause physiological stress in wildlife (Geist 1978) but the effects may not appear for days, weeks or months after the disturbance event (Gutzwiller 1991). Such stresses exacerbate the effects of disease and competition (Gutzwiller 1991), and stress, particularly if prolonged, can cause decreases in lifespan or in reproductive output (Geist 1978). Disease susceptibility is apparently mediated by immunosuppression, acting either via glucocorticosteroid levels and adrenal activity or via cortisol mediated pathways (Van Mourik, et al. 1985). Wildlife exposed to disturbance are known to experience reduced weight gain, higher mortality, and reduced productivity (Titus and van Druff 1981, Gutzwiller 1991), as well as nesting failures (Boeker and Ray 1971), reduced nesting success (Wiley 1975, White and Thurow 1985), displacement, and alterations in wintering distribution, summer home ranges, activity areas, and behavior (Stalmaster and Newman 1978; Andersen, et al. 1990; Fyfe 1969; Enderson and Craig 1974; Portnoy 1974; Swenson 1975; Zarn 1974; Dunstan 1968). Recreational activity and the presence of humans is also known to reduce nest success (Anderson undated). More subtle effects are also known from noise exposure: proximity to noise sources such as airports and freeways is known to reduce IQ in humans (Doidge 2007) and white noise exposure is known to cause premature closure of the critical period for learning (Zhang et al. 2002). Such effects can easily affect survivorship and fecundity.

ORVs present particular problems with respect to disturbance. Supervision and enforcement is virtually impossible if the area is large (Malaher 1967). The same is true of other ORVs – operators can simply drive around any closure notices or barriers because there is no forest cover to constrain the routes taken. Often abuses involve several snowmobiles and even aircraft, all in communication by two-way radio (Malaher 1967). Military operations can also severely disturb wildlife. Gunfire, even from 0.22 caliber weapons affect wildlife (White and Thurow 1985). Roads also cause noise effects from passing traffic that can disrupt activities, inhibiting mating. Braun (1998a) estimated that noise effects would be disruptive as far away as

1 km from a road.

Frequency effects are likely important for disturbance effects. Acoustical interference can obscure behavioral signals at a wide range of frequencies. Data comparing the frequency spectra of noise sources to the detection ability of this species at each frequency are lacking. The transmission of different sound frequencies through forest habitats types has been determined (Wiley and Richards 1978, Richards and Wiley 1980).

Disturbance increases movements and avoidance behavior in wildlife (Bleich, et al. 1990, 1994; Côté 1996). Wildlife exposed to ORVs alter feeding and activity behaviors, stay in cover more often, and flee more easily from disturbances (Yarmoloy, et al. 1988). In various wildlife species, disturbance can also cause social disruption, including disruption of broods and consequent increases in brood mortality (Bartelt 1987, Côté 1996). Wildlife do not habituate to disturbances such as humans on foot, or low level helicopter or fixed-wing aircraft flights (Bunch and Workman 1993), and such disturbances are known to disrupt important breeding, feeding, and social behaviors in a variety of wildlife (Dwyer and Tanner 1992). Other prey species, such as geese, respond adversely to people, boats, hunting activities, aircraft (Owens 1977, Kramer, et al. 1979), loud noises (Owens 1977), and automobile traffic (Madsen 1985) with aircraft causing particular disturbance (Miller 1994). Geese can be driven away from preferred areas when aircraft approach, and aircraft disturbance can cause significant weight loss (Miller 1994). Wildlife react to helicopter noise at levels as low as 60 db, A weighted, and show strong reactions at 77 dbA (Luz and Smith 1976). Helicopter disturbance is known to have serious effects on brant (Derksen, et al. 1992).

Breeding male passerines, particularly older experienced males, avoid roads: densities of breeding males within 200 m of a highway can be 40% less than densities of comparable habitat further away (Reijnen and Foppen 1994). For all sex and age classes, 7 of 12 studied grassland bird species showed reduced densities near roads, and density reductions occurred as far away as 1.7 km from highways (Reijnen, et al. 1996). Density reductions were correlated with noise effects, not visual effects (Reijnen, et al. 1995).

Even low noise levels can obscure important signals that individuals attempt to detect – for example, the wind whistling through a raptor's wings as it stoops to dive is one such signal that individuals need to detect quickly and accurately. Besides any effects caused while the noise source is operating, noise exposure can reduce the capability to detect acoustic signals (partial deafness) for an extensive period after the exposure or permanently (Dr. Howard Wilshire, quoted in ABC News 1999). There is an extensive human literature on such effects, which are likely to affect both mating activities and predator detection.

Finally, little is known about visual disturbances as distinct from acoustic ones. However, it is likely that a visual disturbance will be accompanied by noise and the disturbance effects from the combined stimuli will be enhanced.

An important concern is that there appear to be synergistic interactions among disturbance sources. Visual stimuli often potentiate the effects of auditory disturbances (Taylor, et al. 1962). Thus, effects from two different disturbances are greater than if they were merely additive.

Roads

Roads have numerous negative effects on birds and other vertebrates. Roads and other infrastructural development are major threats to biodiversity and can affect populations over 2 miles away (Benitez-Lopeza, et al. 2010). One of the few studies to examine the degree to which roadless areas contribute to biodiversity conservation found that roadless areas contributed to regional connectivity in important ways and that even small roadless areas were an important component of the foundation for development of comprehensive regional conservation strategies (Strittholt and Dellasala 2001).

The well-known direct effects of roads include habitat loss and fragmentation, vehicle-caused mortality, pollution, and poisoning (Kociolek, et al. 2011). Worse, indirect effects may cause even greater harm to populations, and these effects include noise, artificial light, barriers to movement, and edge effects associated with roads (Kociolek, et al. 2011). Indirect and direct effects may also act synergistically to cause larger decreases in population density and species richness than the mere summation of each effect (Kociolek, et al. 2011). The review by Kociolek, et al. (2011) suggests that road mortality and traffic noise may have the most substantial effects on birds. Various stresses on populations act synergistically, and increasing habitat loss and fragmentation and predicted species distribution shifts due to climate change are likely to compound the overall effects of roads (Kociolek, et al. 2011).

Despite assumptions that the major effect of roads is direct impact mortality, traffic volume is not a good guide to death rates on roads. Both mammal and bird road-kill indices were consistently higher on a low volume parkway than on a high-speed, high volume highway (Clevenger, et al. 2003). Moreover, birds were even more vulnerable to collisions than mammals on the high-speed highway (Clevenger, et al. 2003).

Fragmentation and Isolating Effects of Roads

Besides the obvious effects of collision induced mortality, roads affect habitat directly because the road surface itself and the band of altered vegetation on both sides of the road and its drainage structures alters vegetation and microclimate in the habitat. It is known that large highways can seriously impede genetic exchange in large vertebrates (Strasburg 2006), reducing population viability.

Roads can restrict movements of individuals. Roads are particularly pernicious in their fragmenting effects on populations because they constitute linear isolating elements in the landscape – there may be no way to transit a landscape without crossing a road. Many of the effects of roads are also present for railroad lines, pipelines, and power line corridors. Roads are typically built along drainages (Miller and Eddleman 2000, p. 23) and thus differentially affect riparian areas.

Roads also serve to greatly increase human impacts such as hunting, poaching, and recreational use. Over 90% of all wildfires are human caused, and 75% of that fraction start within 265 feet of a road (Wilson 1979).

Noise Effects of Roads

Roads also induce noise effects from passing traffic that can disrupt activities, inhibiting mating. Most bird species will avoid an area that is subject to highway noise, even if no highway itself is present. (McClure, et al. 2013). This study removes confounding effects of collisions, visual disturbance, chemical pollutants and habitat alteration, showing that noise from roads in and of itself harms most bird populations.

Destabilization, Landslide, Compaction and Dusting Effects of Roads

Roads destabilize slopes and greatly increase landslide risk. East of the Cascade crest, 80% or more of all landslides are road related (Megahan 1980), while west of the Cascade crest, approximately 35% of all landslides are road related (Weaver and Hagans 1996). Such slides can greatly diminish water quality and affect aquatic habitats, besides the loss of terrestrial habitat.

Even gravel or dirt roads produce soil compaction which can persist for decades after use of a road ends, inhibiting plant growth and altering heat storage and water vapor transport on and near the road (Trombulak and Frissell 2000). Traffic on roads, particularly unpaved ones, causes dust mobilization which inhibits plant growth and nutrition – in some cases the entire structure of the plant community is altered (Trombulak and Frissell 2000).

Roads Increase Predation Effects

Predators, such as raptors, may use road signs and utility poles along roads as perches (Bevanger 1994). Some predators and invasive species are known to frequent roadsides and to travel along roads. Generalist predators, such as coyote (*Canis latrans*), frequently hunt along roads in forests (May and Norton 1996). Human development, including campgrounds and roads, increases nest predation by crows and ravens (Marzluff 2006). Because roads cause increased kill rates of animals and increase the visual prominence of carcasses, this inflates the numbers of raptors, corvids, and other scavengers along the roadside (Knight and Kawashima 1993). Such secondary impacts then increase predation and disturbance (Hall and Haney 1997, p. 26). Three separate studies show that ravens are more common near highways than in open areas (Boarman and Berry 1995). Corvids are also more common where linear rights of way, such as roads, parallel each other (Knight, et al. 1995b).

Effects of Roads on Invasive Species

Roads greatly increase the invasion of alien species and are a major contributing factor to the ongoing spread of exotic plants (Gelbard and Belnap 2003). Roads alter ecosystem structure by facilitating the transport and growth of invasive species. Roads foster the entry of exotic plants by providing access for dispersal through human activities. Livestock often travel along roads, and vehicles are major transport mechanisms that spread exotic plants. When a vehicle drives through a weed infested area, seeds or other plant parts often become lodged in the tire treads and undercarriage. Propagules can travel for miles before becoming dislodged in uninfested areas (Cale and Hobbs 1991; BLM 1993c; Sheley, et al. 1997). As one example, the arrival of tansy ragwort (S*enecio jacobaea)* in Montana is directly attributed to seeds moving in on logging equipment from Oregon (Kollmeyer 1997). Tansy Ragwort is a noxious Eurasian weed that is toxic to livestock, and can outcompete native vegetation. Tansy thrives in grasslands and disturbed sites, including the compacted soil of roadbeds (Kollmeyer 1997).

Soil disturbance plays a major role in the spread of exotic plants (Elton 1958 (reprinted 2000), Mooney and Drake 1986, Hobbs and Huenneke 1992, Pickett and White 1985b). Soil disturbance caused by road building changes the microclimate of the area, allowing opportunistic exotic plants to colonize. Two studies on experimental soil disturbance (Kotanen 1997; Zink, et al. 1996) showed that when soil was excavated and biomass removed, exotic plant species colonized quickly and completely, outcompeting native vegetation. Johnstone (1986) notes that plant invasion is caused by removing a barrier that previously excluded a plant from a site. An exotic seed or propagule can lie dormant as a seed or maintain itself as a suppressed seedling until a disturbance destroys or weakens its native competitors. The exotic then enters a growth and reproductive phase, spreading throughout the area. Roads clearly remove barriers (vegetation/biomass) that exclude some exotic plants from a site. Roads also provide access for intentional or unintentional human introduction of exotics, besides the stresses they exert on native species (Trombulak and Frissell 2000). Humans sometimes introduce exotics purposefully for erosion control along or near roadsides (Trombulak and Frissell 2000).

Roads open up paths for weed invasions and improved roads spread more invasive weeds than primitive roads. Each step of road improvement converts an increasing area of natural habitat to habitat that is infested with weeds and other exotic invasive plants. Belnap and Gelbard (2003). Roadless areas act as refuges for native species against invasions. Gelbard and Harrison (2003).

Federal land management agencies, such as the US Forest Service, recognize the deleterious impacts of roads on weed invasion. Ferguson et al. (2003) note that noxious weeds typically invade highly disturbed sites, such as areas along rivers and streams, trails, trailheads, roadsides, building sites, wildlife bedding grounds, overgrazed areas, and campgrounds (citing Baker 1986; Sheley and Petroff 1999). Noxious and invasive plant cover is greatest closer to roads (Gelbard and Harrison 2003), while native plant cover and plant species diversity is greatest only after one moves over 0.6 miles from the nearest road (Ferguson et al. 2003). Weeds and exotic invasives can be transported long distances on the undercarriages of off-road vehicles, and maintenance equipment used for work that disturbs the ground is likely to transport weed seed more readily than recreational vehicles (Ferguson et al. 2003).

Road maintenance greatly increases weed invasion because it involves both transport and soil disturbance. Road grading typically moves road surface material up and down the road system, disturbing soil and vegetation on the roadway and shoulders, and transporting soil and gravel that may carry weed seeds or vegetative propagules. The freshly graded road provides a disturbed soil seedbed suitable for weed germination. Grading a series of roads without washing the blade may transport weeds from one road to the next. Cleaning roadside ditches also moves soil from place to place, creating an ideal seedbed by disturbing soil and removing competitive, desirable vegetation. Equipment used to maintain road drainage structures can spread weeds by transporting soil and weed seeds from one culvert to another. Seeds from equipment can be deposited in stream crossings and washed downstream, creating infestations along the riparian corridor. Treatment of weeds in riparian zones is difficult and chemical treatment can affect animal species, hence expensive techniques, such as hand removal, may be required. Brush cutting transports weed seeds on the mower head. Ferguson et al. 2003.

Roads facilitate invasions by exotic pests and pathogens. As with exotic plants, disturbance can cause "outbreaks" of exotic pests and pathogens. For example, an exotic species may be present at low levels and not drastically impact the ecosystem. With human disturbance, outbreaks can occur where one or two species rise to higher levels of abundance than in undisturbed areas (Dobson and May 1986). Roads also alter microclimates, causing outbreaks that can have serious ecological implications.

Effects of Roads are Extremely Widespread

Roads directly affect over 20% or the land area of the United States (Forman 2000). The effects of roads can never be fully mitigated unless the road is completely removed and revegetated (Trombulak and Frissell 2000). Roads are persistent landscape features that constitute threats to species for long periods of time.

Most of the continental US is impacted by roads with 83% of all land areas within 1 km of a road (Riitters and Wickham 2003). The mainland US contains approximately 6.3 million kilometers of public roads, nearly a quarter of which were classified as non-local roads in rural areas (2001 data from US Department of Transportation (DOT)). "Effects of roads - what we call influence zones – extend tens to hundreds of meters from the roads themselves, altering habitats and water drainage patterns, disrupting wildlife movement, introducing exotic plant species, and increasing noise levels." Riitters and Wickham (2003).

The land development that follows roads into rural areas inevitably leads to more roads, and these roads tend to cross natural boundaries rather than going around, creating new patterns of movement within ecosystems. "Though there is extensive research on how roads impact ecosystems at the site level, there is not much information about roads in relation to ecosystems at the regional level." Riitters and Wickham (2003).

Riitters and Wickham (2003) found that 20% of all land area was located within 127 meters (417 feet) of the nearest road, and 50% was within 382 meters (1253 feet). Only 18% of U.S. land area was more than 1000 meters (0.6 mile) from a road, and only 3% was more than 5,000 meters (3.1 miles) away from a road. Overall, forest land was slightly more remote than other landcover types.

As one US Forest Service scientist put it: "Imagine that the U.S. has been divided into 8.6 billion parcels the size of a baseball diamond infield, and that you could stand on home plate in each of these parcels. According to our model, in one out of every 22 cases there would be a road no further away from you than second base. In one in five cases, the road is no further away than the center field fence. While the actual size of a road influence zone depends on local circumstances, the sheer pervasiveness of roads means that few places in the U.S. are immune to their influences." This quotation is posted by the USDA – Forest Service at http://www.srs.fs.usda.gov/news/109 and is thus entitled to agency deference.

Off Road Vehicles

The Geological Society of America convened a special committee of experts to assess the effects of off-road vehicles (ORVs). That expert committee found that off-road vehicle (ORV) use caused "severe physical and biological consequences" (Wilshire, et al. 1977). Dr. Wilshire, at that time a USGS employee, was attacked by administrators within USGS and Interior, such as Charles Kay (Principal Deputy Assistant Secretary for Policy, Budget and Administration of the Interior Department), and a gag order was issued prohibiting contact with conservation groups (Wilkinson 1998, p. 323-328). Many studies confirm that ORVs cause significant harm to desert areas (e.g. Eckert, et al. 1979; Webb and Wilshire 1983). This damage occurs even when ORV use is minimal (Iverson, et al. 1981). In 1995, the US Government Accounting Office (GAO) studied the impacts of ORVs, and found that land management agencies such as BLM were not complying with Executive Orders 11644 and 11989; for example, monitoring of ORVs was casual and ineffective rather than systematic, "adverse impacts were seldom being documented," and corrective actions were not implemented or even "prioritized" for action (excerpted in Wilkinson 1998, p. 310).

ORVs are one of the primary mechanical toys of industrial tourists (sometimes dubbed "tourons"). ORVs are also used to some extent for industrial operations, for example, in oil and gas exploration, and by ranchers and farmers. For this species, wheeled vehicles (small ATVs as well as larger SUVs) are of concern. ORVs cause alterations of grass and forb cover and reduce plant species diversity (Clampitt 1993). ORV operation is a well known cause of soil compaction and erosion, reduced water infiltration rates, and negative effects on vegetation (Adams, et al. 1982; Eckert, et al. 1979; Iverson, et al. 1981). In arid lands, the soils are "exceptionally vulnerable to ORV attack" (Sheridan 1979). Many of the effects of ORVs are discussed further in the section Military Operations.

Even light use of a truck can damage vegetation and soils (Vollmer, et al. 1976; Iverson, et al. 1981). Not surprisingly, ORV use causes decreased diversity, density and biomass of breeding birds, and ORVs have a negative effect on desert wildlife over large areas (Bury, et al. 1977). Even moderate ORV use is known to cause substantially reduced bird densities (Bury, et al., 1977; Luckenbach 1978). Affected areas can take "centuries or millennia" to recover, or may not recover at all (Wilshire, et al. 1977; Iverson, et al. 1981). Besides damage to vegetation, even slight ORV use increases the amount and frequency of water runoff and erosion, decreases soil porosity, infiltration capacity, effectiveness of soil stabilizers, and hydraulic resistance to overland water flow (Iverson, et al. 1981). ORVs and other motorized vehicles tend to travel in valley bottoms, which are particularly critical because these areas are highly productive. Roads and trails formed by ORVs become corridors for predators and for invasive plant species (GBCP 1997, p. 50).

ORV use also is a major cause of invasion of weed seeds and other pests into intact ecosystems (Tyser and Worley 1992, Hobbs and Humphries 1995, BLM 1996a). Seeds commonly lodge in the vehicle's chassis and can be transported "hundreds of miles" (Pyke 2000, p. 46). Landscape scale is important in understanding the invasion of weeds, and ORVs and livestock are of primary importance in introducing weeds from roadsides into areas away from roads (Belsky and Gelbard 2000). Extant native grasses, forbs, and shrubs will be destroyed by even moderate ORV use, and even tree roots can be undermined, or damaged by soil compaction even though damage is not apparent (Wilshire, et al. 1977).

ORVs present particular problems with respect to disturbance. Supervision and enforcement of snowmobile use is virtually impossible if the area is large (Malaher 1967). Often abuses involve several snowmobiles and even aircraft, all in communication by two-way radio (Malaher 1967). During the winter months, animals are especially vulnerable to this harassment because they are already burdened by increased levels of stress due to low temperatures, inclement weather, reduced food supply, and the need to gain weight for the energetically demanding breeding season. Snowmobile use can also cause disruption in movement patterns, making it more difficult to locate reliable food sources. These impacts are best understood and documented for ungulates (Cain, et al.,1997; Parker, et al. 1984; Moen, et al. 1982; Severinghaus and Tullar 1975); nevertheless, many other wildlife species suffer the same sorts of direct impacts from exposure to and harassment by ORVs.

Repeated ORV use can lead to changes in plant density and species composition and set back seral stages (Aasheim 1980, Wanek and Schumacher 1975), and the associated loss of vegetative cover generally leads to increased soil erosion (Montana Fish, Wildlife and Parks 1993). Because ORVs generate pollution levels hundreds of times those of a modern automobile, there is significant opportunity for vegetation damage from air pollution in basins with stable air masses.

ORV use is accelerating and as BLM states, has shown a "dramatic increase" in just the last few years (ABC News 1999). Over half of all ORV use takes place on BLM lands (Donahue 1999, p. 187). Other federal lands are also experiencing rapid increases in ORV use (ABC News 1999). In Wyoming, one driver recently ran his vehicle "right into the middle" of a strutting ground (High Country News, May 10, 1999, Vol. 31, No. 9, p. 15).

Military Operations

Use of tracked vehicles ("tanks") can cause even greater damage than use of ORVs. Both operations in a straight line and turning or stopping of the vehicle (causing divots) are significant causes of erosion, tree destruction and understory destruction (Watts 1998). Effects on cryptogamic crust are particularly severe (Watts 1998). Both vehicle and foot traffic are known to compact soils, increase erosion, reduce vegetative cover, facilitate the spread of alien plants, and increase fire frequency (Watts 1998).

Firing of tracer bullets and use of pyrotechnic devices are major sources of fire (YTC CA 1994, p. 4, section VI.H). Troop training also involves the excavation of soil for foxholes, latrines, and other uses, and the establishment of bivouacs, which damage vegetation.

Oil and Gas Operations, Mining, and Prospecting

Prospecting and operations for oil and gas, mining, and other such resource development typically involves the use of ground vehicles and road construction. Prospecting often involves setting off underground explosions that can interfere with the acoustic communication and otherwise disturb the animals. Moreover, any type of intensive development greatly increases poaching – Bay (1989) estimated a 3x increase in game violations in such areas.

Exploration activities cause noise, road creation, and disturbance that may have long-term effects (Braun 1987a). For example, in Sage Grouse, hens from areas where leks were disturbed by natural gas development had lower nest initiation rates, traveled twice as far to reach nest sites, and were more sensitive to nest cover than hens from leks that were not disturbed (Lyon 2000). This is particularly troubling because BLM and other land management agencies attempt to mitigate disturbance by protection areas within 2 miles (or even less) of a lek. But it is precisely these areas in which females will attempt to nest farther than 2 miles from a lek (Lyon 2000, p. 23). The use of low-flying helicopters, in an attempt to avoid ground disturbing activities, can also enhance the dispersal of weed seeds, as well as create high noise levels that interfere with activities. Aircraft over-flights are apt to be particularly disturbing to prey species, as explained elsewhere in this review.

Utility Corridors and Powerlines

Pipelines, electrical transmission lines, telephone lines, and the like cause degradation of natural vegetation, soil disturbance, and the hydrological regime (Artz 1989). Recovery times for vegetation on these areas are 30 to 100 years or even longer (Artz 1989).

Utility poles also represent perches for aerial predators and can serve as a behavioral deterrent to movements. Three separate studies show that ravens are most numerous near powerlines (Boarman and Berry 1995). However, the greatest danger near powerlines comes from raptors which use the poles as perches, providing excellent point from which to sight prey. Typically, such poles range from 13 m to over 20 m in height (Hall and Haney 1997, p. 11). Utility poles and their lines can permanently disrupt populations. Any vertical element that can serve as a raptor perch can affect prey. This is not limited to powerline or telephone poles, but also includes trees (live or dead), microwave towers, military and civilian radar installations, cell phone towers, oil rigs, and similar infrastructure. These vertical elements in the landscape serve as perches for predators, such as raptors and corvids. Corvids are known to be more common along powerlines because they serve as perches and as nest sites (Knight and Kawashima 1993).

Weather Effects

By weather, this review adopts the conventional view that weather encompasses relatively short-term changes in such variables as precipitation, temperature, wind, and solar insolation, while climate refers to longer-term changes in these factors. Weather events have direct effects on adults (Walsberg 1978, 1983a, 1983b; Walsberg and King 1978b; Gessaman and Worthen 1982; Root 1988a, 1988b), and on juveniles (Walsberg and King 1978a; Webb and King 1983a, 1983b). Episodic weather events can also alter habitat and vegetation structure and are a recognized element in plant survival (Nelson and Tiernan 1983).

Climate Change and Global Warming

Climate refers to long term changes in weather. The greenhouse effect is the term used to describe the trapping of heat in planetary atmospheres by various gases. This effect is not closely related to ozone layer depletion. The greenhouse effect is minimal on Mars, very strong on Venus, and moderate, but rapidly increasing, on Earth. Carbon dioxide (CO_2) is estimated to account for about 49% of the contribution to the greenhouse effect on Earth (Hansen, et al. 1988). Methane, nitrogen oxides (NO_x), and chloroflurocarbons (CFCs) are the main gases accounting for the remainder of the greenhouse effect (Hansen, et al. 1988). Livestock account for about 15% of methane inputs to the atmosphere – each cow emits 400 liters of methane per day because it farts or belches every 90 seconds. When summed over the number of cattle on Earth, this is a very large amount of methane.

Peters and Lovejoy (1992) described global warming as the most ominous of all potential threats to biodiversity. Grover (1990) and Kareiva, et al. (1992) discuss biotic effects of global warming. Climate change is a major and nearly all-encompassing threat to conserving species fauna in the United States and worldwide. Karl et al. 2009, Thomas et al. 2004, IPCC 2002. The U.S. Fish and Wildlife Service (US FWS) recognizes that climate change is the greatest threat it has ever faced in causing extinctions of the nation's wildlife species. US FWS 2008a. In California alone, climate change will cause most endemic plant taxa to experience 80% or greater reductions in range within a century (Loarie et al. 2008). Increases in temperature over the last century have clearly been linked to shifts in species distributions, and in the near future, climatic change will cause the local loss of at least 10% of the vertebrate fauna over much of North and South America. Lawler, et al. 2009. Regional temperature changes can be much more extreme than changes in global averages (Root and Schneider 1993).

Regardless of the sources of the greenhouse effect or how significantly the greenhouse effect itself contributes to planetary warming, the warming trends are real and will have severe effects on this species and its habitat. The "magnitude of climate-driven extinctions is potentially very large," so large as to be "alarming." Schwartz, et al. (2006). Throughout North America, climate change will be the most important factor in reducing biodiversity (Sala, et al. 2000).

Herman and Scott (1992) identified several factors rendering species vulnerable to global warming effects. Climate change is "almost certain to become" a threat to many species "in the foreseeable future due to increasing concentrations of greenhouse gases from fossil-fuel use, land-use changes, and agriculture" (Wilcove, et al. 1998). Indeed, it appears that the ranges of some species are already being affected by global warming (Parmesan 1996). The inland West is "particularly vulnerable to global warming and to extreme moisture stress" (Covington, et al. 1994). Climate change will continue and worse, will accelerate in the future (IPCC 1996, McCarty 2001).

Although the predicted magnitudes of warming are severe, "it is the predicted rate of temperature change that poses the greatest threat to biodiversity" (Morse, et al. 1995). Climate change has been and is projected to be "rapid," and the "ability of species to survive rapid climate change may partially depend on the rate at which they can migrate to newly suitable areas" (Morse, et al. 1995). However, migration rates are largely irrelevant because the required habitat will not trend northward sufficiently rapidly to avoid extinction.

Limitations imposed on species ranges by the climatic, ecological, and physiological effects of elevation are important determinants of extinction risk (Sekercioglu, et al. 2008). Examples include, but are not limited to, species living in montane environments, where lack of intervening habitat inhibits dispersal to similar environments. Elevational limitation of range size explained 97% of the variation in the probability of a species of landbird being at high risk for extinction (Sekercioglu, et al. 2008). These scientists found that even an intermediate estimate of surface warming (2.8 °C) would result in 400–550 land bird extinctions in the near term, and that approximately 2150 additional species would be at risk of extinction by 2100. Only 21% of the species predicted to become extinct in their study are currently considered threatened with extinction by governmental organizations and the like (Sekercioglu, et al. 2008), indicating that current species at risk lists grossly underestimate the actual number of species facing extinction. If surface warming exceeds the intermediate estimates, then extinction rates will be even greater than predicted by Sekercioglu, et al. (2008).

The World Meteorological Organization's Intergovernmental Panel on Climate Change (IPCC) estimates that the central portion of North America will warm 2 °C to 4 °C by 2030 (Houghton 1990). Other models predict even greater warming of 4 °C to 7 °C (Morse, et al. 1995). Soil moisture is predicted to decline by 15% to 20% (Houghton 1990). Even intermediate warming trends, which will occur sooner, will cause a broad array of negative impacts on species and their habitat, including increased length and severity of droughts. Warming would push the entire area suitable for this species and their required habitat northward. Fragmentation interacts with climate change to restrict migration because climate change creates new barriers to movement (Peters and Darling 1985).

Two factors are sometimes assumed by lay people to mitigate the effects of climate change on species extinction likelihoods: migration and adaptation. However, studies have shown that neither effect is adequate to forestall extinction. Most vertebrate species will be unable to adapt to climate change. Quintero and Wiens (2013). The authors did not study invertebrates, but the evolutionary rates required indicate a similar result in those taxa. As Quintero and Wiens (2013) put it: "Our results are striking: matching projected changes for 2100 would require rates of niche evolution that are > 10,000 times faster than rates typically observed among species, for most variables and clades."

Nor is migration any remedy for climate change: Many "species won't be able to move fast enough… also, moving may require unimpeded access to habitats that have not been heavily disturbed by humans. Or consider a species living on the top of a mountain. If it gets too warm or dry up there, they can't go anywhere." Interview with author J. J. Wiens by Univ. Arizona press office.[better citation). Climate warming has resulted in a significant upward shift in species optimum elevation averaging 29 meters per decade. The shift is larger for species restricted to mountain habitats and for grassy species, which are characterized by faster population turnover (Lenoir, et al. 2008). Lenoir, et al. (2008) have shown that climate change affects the spatial core of the distributional range of plant species, in addition to their distributional margins, and managers cannot ignore the effects on the spatial core of the range.

Foden, et al. (2013) note that species which are highly vulnerable to climate change will be those with low ability to adapt or adjust to the changes in the specific habitats required by any life-history stage of that species. Because the rapidity of climate change greatly exceeds the ability of vertebrates (and likely other species) to evolve (Quintero and Wiens 2013), the ability of species to adapt to climate change will often be exceeded, increasing the degree of endangerment.

Equilibrial general circulation models predict greater drought and decreased summer soil moisture within 50 years (Ferguson 1997) – a threat that is thus easily foreseeable. Warming will lead to increased invasion of alien plants (Morse, et al. 1995). Warming reduces the size of, or even eliminates, existing habitat areas in reserves (Peters and Darling 1985). Climate change will cause increased fragmentation by creating barriers to movement, and dispersal areas will be reduced in number (e.g. some stepping stones along a pathway will be removed, making dispersal impossible). Peters and Darling (1985).

Thus, habitat – already degraded, fragmented and reduced – will contract and fragment even more. As American Association for the Advancement of Science Fellow John P. McCarty put it: "It is no longer safe to assume that all of a species' historic range remains suitable." McCarty, 2002.

Moreover, variability of precipitation has increased (Tsonis 1996). The effects could be severe because it is the population lows that are of concern for extinction risk, and such lows are exacerbated by drought years. Similarly, the impacts of climate change take place in landscapes already severely dissected by habitat fragmentation. Willis and Bhagwat (2009). The combination of all these factors greatly increases the impacts of climate change.

Besides warming, other effects of increased CO_2 in the atmosphere include ocean acidification, acidification of precipitation, and enhancement of growth of certain plants over other plant species, depending upon photosynthetic pathway.

Global warming also affects drought incidence, severity and the length of drought periods. Many geographic areas are expected to convert into deserts over time. Saltz et al. (2006) found that even minor die-offs (on the order of 15%) following droughts increased extinction probability nearly 10-fold. Nothing in their study limits these results to the particular species studied, and the results indicate that an increase in environmental stochasticity due to climate change may, in itself, pose a significant extinction risk.

Climate change is already exacerbating drought:

> Climate change affects water demand and the ways water is used within and across regions and economic sectors. The Southwest, Great Plains, and Southeast are particularly vulnerable to changes in water supply and demand. Changes in precipitation and runoff, combined with changes in consumption and withdrawal, have reduced surface and groundwater supplies in many areas. These trends are expected to continue, increasing the likelihood of water shortages for many uses. Increasing flooding risk affects human safety and health, property, infrastructure, economies, and ecology in many basins across the United States… Increasing resilience and enhancing adaptive capacity provide opportunities to strengthen water resources management and plan for climate-change impacts.

US National Climate Summary (2014).

The Federal Government expects "increased competition for water" in southern latitudes, as well as "decreased water availability." US National Climate Summary (2014).

Climate change is known to exert strong effects on some species of consumer populations even without affecting them directly. For example, Martin (2007) found strong effects of climate change, including extirpation and severe population declines, on several consumer species because of effects on trophic levels below (plants) and above (predators) the consumer populations. Similarly, Voigt, et al. (2003) found that the sensitivity of a species to climate change is correlated with trophic level, with predators and consumers at greater risk than herbivores and plants.

Indirect impacts from climate change may be even more damaging to species viability than direct impacts (Ockendon, et al. 2014).

The effects of climate change need not be immediate: Thomas, et al. (2004) note that climate change will cause 15–37% of species to be 'committed to extinction.' Climate change can also operate indirectly to endanger a species, even if the species is not directly affected. For example, the unprecedented future rates of climate change, coupled with land use changes that impede gene flow, can be expected to disrupt the interplay of adaptation and migration, likely affecting productivity and threatening the persistence of many tree species (Davis and Shaw 2001) which provide habitat for many other species.

The Service is aware that climate change poses severe threats to species, and has been aware of these dangers for several decades. As one example, the Service acknowledges these risks to existence in the 1984 Recovery Plan for the Iowa Pleistocene snail (*Discus macclintocki*) where the Service identifies climate change as a factor in causing the endangerment of this species (on p. 6 the Service notes that "the major long-term cause of decline is cyclic climate change"). Frest and US FWS (1984). In 2009, the Service further expressed its concern over climate change, noting that: "Global warming is perhaps the largest looming threat to the Iowa Pleistocene snail." (p. 15). Similarly, the Service has identified climate change as a threat to the Dakota skipper butterfly, a candidate species (now proposed for the listing as Threatened). In its Proposed Rule for Listing of the of the Dakota skipper Butterfly as Endangered (Federal Register, Vol. 78, No. 206 on Thursday, October 24, 2013), the Service reflects the scientific certainty surrounding global climate change and its effects on the United States:

> Scientific measurements spanning several decades demonstrate that changes in climate are occurring, and that the rate of change has been faster since the 1950s. Examples include warming of the global climate system, and substantial increases in precipitation in some regions of the world and decreases in other regions.

78 Fed. Reg. 63612, col. 2.

Worse, the Service acknowledges that detrimental climate change will continue in the future: "there is strong scientific support for projections that warming will continue through the 21st century." 78 Fed. Reg. 63612, col. 3. The Service also acknowledges that "adverse effects of climate change[] includ[e] climate variability and extremes" and that "increased variability in weather patterns and greater frequency of severe weather events" are aspects of climate change. 78 Fed. Reg. 63613, col. 1.

Climate change affects numerous other threats, such as fire risk and magnitude, invasion of pest pathogens, competitor and predator species, drought, aridity and altered water-flow regimes, and many others. Climate change will also accelerate invasion and habitat conversion by exotic pests which predate, out-compete and/or parasitize this species. Studies show that 15–37% of terrestrial species will be 'committed to extinction' or completely extinct by 2050 because of climate change operating alone (Thomas et al. 2004). But climate change does not operate alone. Instead, it operates in combination with other threats, such as habitat degradation, fire, drought, conversion to agriculture and other development, and landscape threats such as oil and gas development and electric powerline and cell telephone tower sitting. These threats in combination will eliminate even formerly abundant species in a much shorter time than climate change alone. When the various threats are considered in combination, the outlook for this species is grim indeed.

Ozone Layer Depletion

Thinning of the layer of ozone in the Earth's atmosphere removes the primary barrier to the transmission of ultraviolet rays (UV). Increases in high energy UV radiation can damage plant tissues (thereby inhibiting plant growth and vigor and affecting photosynthesis), can cause thinning of avian eggshells, and can affect insect production (GBCP 1997, p. 45). Thus ozone depletion can affect species directly, as well as by reducing their food supply. Effects on juveniles will be particularly strong in areas where disturbance (e.g. livestock grazing) has removed radiative cover from nesting areas (Webb 1993b).

Pollution

Environmental toxins can cause effects not only on the affected individual but on its progeny, and studies have shown that such effects can persist for several generations, perhaps indefinitely. Such transgenerational effects can occur either by a chromosomal or epigenetic alteration in the germ line. Transient exposure of a gestating female rat during the period of gonadal sex determination to endocrine disruptors caused decreased spermatogenic capacity (cell number and viability) and an increased incidence of male infertility in the next generation (Anway, et al. 2005). These effects were transferred through the male germ line to nearly all males of all four subsequent generations examined. The ability of an environmental factor (for example, an endocrine disruptor) to reprogram the entire germ line and to promote a trans-generational disease state has significant implications for evolutionary biology and disease etiology (Anway, et al. 2005.

Water Pollution

Sediment, toxins, alterations to pH and other pollutants can affect organisms directly and at various life history stages, or can affect an organism indirectly altering its habitat. Pollution from far away can reach rivers, lakes and streams or percolate into ground water.

Air Pollution

Several Clean Air Act non-attainment areas (i.e. polluted air sheds) lie within the range of this species. Power plants constructed in the area produce a number of harmful emissions including sulphur compounds that can produce significant environmental effects. Sulphur emissions are a "regional issue because the sulphur may travel 1,000 km in a few days" (Schoettle, et al. 1999, p. 5). Air pollution is known to cause genetic mutations in animals (Somers, et al. 2004), thus lowering mean population fitness.

Sulphur dioxide (SO_2) and other pollutants can affect vascular plants. Sulphur oxide (SO_x) and nitrogen oxide (NO_x) emission sources are common throughout the range of this species (Schoettle, et al. 1999, p. 8-9). Industrial facilities have proliferated in recent years, as have vehicular sources. Consequently, "even remote areas are subject to high concentrations of [air] pollutants" (Scruggs 1991). Nearly every county within the range of this species has SO_x and NO_x sources producing hundreds of tons, and in some cases thousands of tons, of emissions per year (Schoettle, et al. 1999, p. 8-9). Another potentially important emissions source for NO_x is agriculture – "air emissions from fertilized agricultural land may be substantial," yet such emissions "are not accounted in the NO_x inventory" (Schoettle, et al. 1999, p. 41).

Ozone (O_3) emissions are increasing and are particularly damaging because ozone is a known phytotoxin and can "threaten remote ecosystems and resources far from pollutant sources" (Schoettle, et al. 1999, p. 10). Even if ozone does not kill plants outright, it can weaken native vegetation sufficiently to allow invasion by alien plant species. Indeed, "ozone has the greatest potential of any air pollutant to directly reduce growth and vigor of vegetation" in the Interior Columbia Basin (Schoettle, et al. 1999, p. 43).

Acid Precipitation

Acid precipitation – often termed acid rain – can occur as rain, snow, or particulate fallout carried by any type of precipitation. It occurs when nitrogen or sulfur oxides are released into the atmosphere. Vehicle emissions are the major source of nitrogen oxides and industrial plants are the major source of sulfur oxides.

The susceptibility of certain organisms such as lichens to acid precipitation is quite high. The susceptibility of oaks and ponderosa pine should be considered by the Service. What is important in analyzing acid precipitation is not whether the absolute amounts generated in the west are comparable to the amounts generated in the mid-western and eastern United States. Instead, it is whether the susceptibility of plants needed by this species is within the range of acid precipitation reaching them.

Effects of Chemical & Radiological Agents

Use of herbicides, pesticides, and other chemical agents is known to have damaged animal populations, even though the phenomenon has been little studied. Like humans, wildlife are subjected to a mix of numerous synthetic chemicals. Synergistic effects of this mixture may exceed the effects of individual contaminants by several orders of magnitude (Arnold, et al. 1996; Colborn 1995). Besides the effects of synergistic mixtures of contaminants, and the effects on susceptible life-history stages, the manner in which chemical agents are tested also tends to underestimate effects in nature. Chemicals are routinely administered to well fed and well watered, healthy animals. In contrast, wildlife often go hungry, experience water stress, disease and parasite loads, face mechanical injury and immuno-challenges, often in combination – all while being subjected to chronic or acute exposures to chemical agents.

Besides the effects on habitat, water sources can be contaminated by spraying of nearby fields that drain into the water source. Breakdown of chemicals in dry soils may be particularly slow, due to lack of microbial activity in low moisture environments.

Effects of Herbicides

A variety of chemical herbicides have been used in habitat areas. Besides any direct effects on individuals, such chemicals may produce indirect effects on food or other resources. Effects may be acute or chronic, and may interact with other chemicals. Herbicides are known to increase infection rates of wildlife by parasites, and mechanisms include both reduced immunocompetancy in the host, and augmentation of intermediate hosts for the parasite (Rohr, et al. 2008).

Effects of Animal Pesticides

Pesticides have been used to kill various insects occurring in habitat areas. Pesticides harm individuals by depleting their food supply, by acute poisoning, by chronic poisoning, and perhaps by disrupting neuronal and endocrinological systems affecting immune function, development and behavior. Even low doses of organochlorine pesticides are known to alter important behaviors, such as aggression levels, alertness, discriminatory behaviors and territorial activity (Jeffries 1973). Organophosphide pesticides are known to cause such sub-lethal effects as weight loss, reduced visual acuity, auditory detection, vigilance, food seeking behaviors, offspring caretaking, and greater susceptibility to environmental stressors, including weather effects (Grue, et al. 1983). Thus, low doses can cause death or reproductive decrements by acting synergistically with other, natural effects in the environment. These threats are continuing.

Effects of Endocrine Disruptors

A number of chemical compounds, some otherwise thought to be benign, have been implicated as causing subtle, but long-lasting effects, including behavioral alterations, and disruption of development. Behavioral alterations include aberrant behavior during reproductive activities (Mac 2000) – of particular import for viability. Importantly, these effects are hypothesized to occur at concentrations several orders of magnitude below those at which either acute or chronic effects are known from conventional chemical agents. A second important concern is that effects are believed to have occurred at concentrations below detectability limits, even using the most modern analytical techniques, such as HPLC (high performance liquid chromatography) or mass spectrometry. The US Geological Survey noted that such compounds can act by mimicking natural estrogens in the body, as well as by altering the action of other sex hormones, and glucocorticoids and thyroxine (Mac 2000). In humans, large reductions in sperm counts have been attributed to endocrine disruptors, as have the recent increases in cancers of the prostrate, breast, and ovaries – all tissues which are sensitive to sex hormones (Mac 2000). Effects of these compounds have been demonstrated in birds and are known to produce transgenerational effects (Colborn, et al. 1993; Facemire, et al. 1995). Effects are a particular concern during embryonic development and endocrine disruptors " can permanently modify the organization of the reproductive, immune, and nervous systems" (Guillette, et al. 1995).

Natural Factors and Environmental Variation

Environmental variation in climate, food sources, and predators is high in the areas used by this species. One prediction from global warming models is increased drought in continental interiors, such as the range of this species.

Fragmentation

Fragmentation is discussed extensively elsewhere in this review, and is also known to affect social behavior in vertebrates (D. R. Webb 1981). Webb postulated that the increase in agonistic behavior, and the decrease in amicable behavior seen in fragmented areas was caused by the difficulty of juvenile dispersal to new areas, and was unrelated to such factors as elevation, foraging time, and other non-landscape factors. Habitat fragmentation could cause similar behavioral alteration in this species, increasing the likelihood of extinction.

Habitat Recovery Time

Lags in population recovery may also add to lags in habitat recovery – even if populations are able to maintain some viability in a degraded area, the local population is unlikely to be capable of serving as a source population in a landscape sense for many decades. The Service must consider the many decades required to restore habitat in its assessment of threats to the species because areas of habitat will continue to be degraded and eliminated before other areas can recover and individuals can reestablish populations in those areas. Forests are particularly slow to reestablish after logging.

Inadequacy of Existing Regulatory Mechanisms

A species must be listed if it "is endangered or threatened" because of "the inadequacy of existing regulatory mechanisms." 50 C.F.R.§ 424.11(c)(4); 16 U.S.C. § 1533(a)(1)(D). The Secretary must conduct a "review of the species' status." 50 C.F.R. § 424.11(c). The determination to list the species must be made "solely on the basis of the best scientific and commercial data." 16 U.S.C. § 1533(b)(1)(A); 50 C.F.R.§ 424.11(b). The Secretary may not consider actual or "possible economic or other impacts" in the listing decision. 50 C.F.R. § 424.11 (b).

Only a small amount of the land in the United States is managed to support biodiversity (Dietz and Czech, 2005). On federal lands, specific statutes apply to various agencies that manage the public lands and most such lands are managed for multiple uses. Even lands administered by the National Park Service are managed for tourism (enjoyment by humans), not merely for protection. Legal protections inside a National Park, designations as Wilderness, much less as Roadless Study Areas, do not ensure actual protection of habitat for Threatened or Endangered species. Besides changes in statutory of administrative status, trans-boundary ecological effects such as CO^2 concentrations, increasing temperature or aridity, or acid deposition are all known to have caused habitat alterations in the recent past. Such changes have negatively affected rely on the impacted habitat, are likely to do so in the future. For example, CO^2 concentrations have favored the invasion of juniper into sage grouse habitat.

Besides the ESA itself, only two relevant statutes extend across agency land boundaries: the National Environmental Policy Act and the Clean Water Act. The latter is rarely applicable to habitat issues. Multiple-use land management agencies typically have organic acts relating to how they manage the public's land that they are allowed to administer. For the BLM, the Federal Lands Policy and Management Act (FLPMA) applies, and for the Forest Service, the National Forest Management Act (NFMA) applies.

The Service cannot reasonably rely on NEPA, FLPMA, NFMA and other laws to conserve the species, because these laws are not adhered to by federal agencies. Just as is the case with the Service, hundreds – if not thousands – of federal judges have ruled that the BLM, the Forest Service and other land management agencies have broken the nation's environmental laws. In many other cases, plaintiffs have not been successful in getting courts to reach the merits of a case because of a plethora of procedural, deferential, and jurisdictional hurdles that shield federal agencies from judicial review. The erection of doctrines barring or hampering citizens from court contributes to the inadequacy of existing regulatory mechanisms. The Service must consider the track record of agency protections and the ability of citizens to review agency action when determining the adequacy of regulatory mechanisms. When agencies have a long record of deliberate violations, those agencies cannot be relied upon to obey the law in the future.

Other agencies with populations or habitat have no particular mandate to protect the species. These agencies include various branches of the Dept. of Defense, the Bureau of Reclamation, and others. They are governed by no substantive statutes requiring wildlife protection. Thus, the protection of the ESA is especially critical on these lands.

National Environmental Policy Act (NEPA)

The National Environmental Policy Act of 1969 (NEPA, 42 U.S.C. § 4321 – et seq., Pub. Law No. 91-190) is merely a procedural act and requires no substantive outcome. Vermont Yankee Nuclear Power Corp. v. NRDC, 435 U.S. 519 (1978). It thus does not constitute a regulatory mechanism. Moreover, because substantial declines in this species populations have occurred since NEPA was first passed in 1969 it has clearly been inadequate to conserve the species. Congress recognized the inadequacy of existing statutes when it passed the Endangered Species Act in 1973.

National Forest Management Act (NFMA)

The National Forest Management Act (NFMA, 16 U.S.C. § 1600 – et. seq., Pub. Law No. 94-588, 90 Stat. 2949, as amended) governs actions of the US Forest Service on the public lands it administers. One NFMA provision requires that the Forest Service "provide for diversity of plant and animal communities," 16 U.S.C. § 1604(g)(3)(B). To carry out its statutory duty, the Forest Service promulgated an administrative regulation that requires maintenance of the viability of vertebrate species on its lands (36 C.F.R.§ 219.19). The Forest Service has not maintained the viability of vertebrate species, including the spotted owl, nor has the Forest Service maintained biodiversity on the lands it manages. For example, several botanists were forced to sue the agency because forest plans did not rely on well understood landscape ecology concepts (Mlot 1992). As late as the 1990s, scientists noted that the Forest Service "abysmally misunderstood, misconstrued, or missed altogether" vast amounts of information on the effects of fragmentation from the 1970s and 1980s (Mlot 1992). Although this information had been in the scientific literature for "20 to 25 years" before forest plans came out, and was "widely accepted scientifically," the plans did not incorporate that information. A Justice Dept. attorney, who represented the Forest Service in the ensuing litigation, stated that that was "how government works. They're going to be behind the curve" (Mlot 1992). Unlike the FWS, the Forest Service is not required to use the best available science, leading to mis-management of rare species. Moreover, the viability requirement has been weakened by several court decisions that have given overweening deference to the USFS even when the top population biologists in the world disagreed (Inland Empire Public Lands Council v. United States Forest Service, 88 F.3d 754, 760 (9th Cir. 1996)), and have allowed USFS to substitute mere habitat quantity for population viability (Sierra Club v. Marita, 46 F.3d 606, 619-20 (7th Cir. 1995)). More recently, the Forest Service has attempted to do away with the administrative rule altogether. Ultimately, there is simply not sufficient habitat on USFS lands to preserve this species, no matter how well USFS acted as a steward.

Other Federal Laws

The Sikes Act, 16 U.S.C. §§ 670a-670o, September 15, 1960 (74 Stat. 1052, Public Law 86-797), as amended 1968, 1974, 1978, 1982, 1986, 1988 and 1989) requires the development of conservation plans for wildlife on all public lands. Federal agencies have not developed a comprehensive plan for this species, and the species has declined under the operation of the Sikes Act.

Conservation Reserve Program

The federal Conservation Reserve Program (CRP), administered by the USDA Farm Service Agency, is often touted as protecting wildlife, but does not do a good job of protecting wildlife. Lands for the CRP are not selected on the basis of wildlife values and farmers often volunteer marginal and low productivity lands for the CRP. Additionally, lands in the CRP are often isolated fragments (see sections above), provide poor habitat values, and may even be detrimental to wildlife species by acting as population sinks.

Surprisingly, the CRP has no requirements for planting native vegetation. Farmers and ranchers often plant exotic grasses as a cover crop, thus providing minimal habitat value to wildlife. During drought periods ungrazed lands are of particular value to wildlife compared with times of higher precipitation, yet farmers and ranchers are allowed to graze such lands in the CRP system, thus removing food and cover habitat for wildlife.

Enforcement

Multiple use land management agencies, such as the BLM and Forest Service, may be structurally incapable of adequately managing species at risk. One important reason for this arises from the agency mission to provide for extractive commercial interests such as grazing and mining, as well as recreational interests such as off-road vehicle (ORV) use that have strong support from large industrial manufacturers. Such uses are often at odds with wildlife protection, and agencies often fall prey to the familiar phenomenon of administrative "capture" by the interests it is supposed to be regulating: "controls directed by a public regulator on the private sector…[but] <u>in practice</u>, the direction of interference or control is opposite … regulatory outputs tend to correspond to the interests of the regulated party…." (Mantic 1980, p. 14). This is the familiar problem of capture of a regulatory agency by the interests it is supposed to regulate (Davis and Pierce 1994), a phenomenon first systematically analyzed by Nobel Prize winning economist George Stigler.

Management on USFS Lands

Forest Service scientists have admitted that they do not conduct site-specific analyses of the impacts of management actions on wildlife – the demands of assessing impacts "preclude site-specific study" of those impacts (Toth and Baglien 1986, p. 255). The reasoning is circular, and the violations of NEPA seem clear. Such issues may also apply to BLM management actions. Under regulations promulgated pursuant to NFMA, the USFS is required to assure the viability of vertebrates on USFS lands (36 C.F.R.§ 219.19). The Forest Service is a known scofflaw – one judge noted the "deliberate and systematic refusal by the Forest Service and FWS to comply with the laws protecting wildlife." <u>Seattle Audubon Society v. Evans</u>, 771 F. Supp. 1081, 1090 (W.D. Wash. 1991).

Management by the US Fish and Wildlife Service

It is not merely state and other federal agencies whose mis-management threatens species preservation. The Service itself has mismanaged its ESA duties, including its listing responsibilities under section 4, as well as the lands it administers directly via the National Refuge System.

Because these issues are raised in a rule-making petition pursuant to the Administrative Procedures Act, the Service must consider the degree to which its own legal violations constitute a threat to the species.

Management of Listing and other ESA Duties

A variety of entities, including scientists, legal commentators, and federal judges (see below) have heavily criticized the Service for mismanagement of its duties under the ESA. For example, law professors such as Rohlf (1991, 1992), Doremus (1997), and Parenteau (1998) have all criticized the Service as have other scholars, e.g. Burgess (2001), Bonnett and Zimmerman (1991), Yaffee (1982), and Lieben (1997). Governmental and scientific organizations have also criticized the Service and its handling of mandatory duties (National Research Council 1995b), as have scientists such as Wilcove, et al. (1993) and Sidle (1998a, 1998b). The Service has been criticized by the scientific society of professional ecologists, the Ecological Society of America, for delaying recovery planning, for only producing recovery plans for about half the species listed, and for formulating recovery plans that have such "weak goals" that over half the vertebrates with such plans" would remain in serious risk of extinction" after reaching the goals in the plans (Carroll, et al. 1996). The Service's recovery plans often "manage for extinction" rather than for survival (Tear, et al. 1993). Effects of various factors on population viability and extinction risk can be difficult to assess, and Doak, et al. (2008) note that prudent management requires a precautionary approach, but the Service has not adopted such an approach.

Protection of subspecies and distinct population segments is particularly important in implementing the goals of the ESA. As Green (2005) explained: species status assessment and the conservation of biological diversity requires defining units that are smaller than the species level to portray probabilities of extinction accurately and to help set priorities for conservation efforts. Even in the 1980s, experts widely agreed that the subspecies is a useful taxonomic category (Wiens 1982; Class Aves). Later, Phillimore and Owens conducted a comprehensive analysis of subspecies, and found no empirical support for the idea that the increase in designated taxonomic units is primarily epistemological rather than data-driven. Instead, their study showed that previous claims about the causes and effects of taxonomic inflation lack empirical support (Phillimore and Owens 2006). These studies point up that no only are subspecies real, they are important in conserving biodiversity at higher (e.g. species) taxonomic levels. However, the Service has no official policy on subspecies, despite being ordered by the Ninth Circuit Court of Appeals to do so.

Since the inception of the ESA, the Service has managed to recover and delist only a handful of species – 8 species according to the Congressional Research Service (Corn 1997); 5 species according to Schwartz (1999, p. 86 <u>citing</u> 50 C.F.R.§ 17.11, 17.12). In the 10 year period from 1984-1994, seven species or subspecies became extinct while listed under the ESA (Langner and Flather 1994; FWS 1992). Worse, the Service never considered over 200 former candidates and subspecies for listing because they were believed to have gone extinct, again due to inaction by the Service (FWS 1989, GAO 1990).

The Service rarely considers species for protection on its own initiative – most species are listed as a result of petitions (GAO 1992b), and the Service has not listed any species as a result of its own review during the year 2001. Consequently, the Service's own data show that more species are declining than are improving (National Research Council 1995b), and that as of 1993, more listed species have gone extinct than have been recovered (Mann and Plummer 1995,

p. 240). An audit of the Service's ESA program by the Inspector General found that from 1980 to 1990 alone, 34 species became extinct without ever receiving the protections of the ESA (Inspector General – USDI 1990). The reason is that the Service has "not effectively implemented a domestic endangered species program" (Inspector General – USDI 1990).

Only part of the delays in listings are caused by funding shortfalls, and these funding shortfalls are, in turn, the result of deficient funding requests from the Service. For example, The Interior Department requested only $1.3 million for species protection efforts for fiscal year 2002, even though it acknowledges that $120 million is required just to eliminate the species listing backlog and respond to court orders (Seelye 2001, p. A14). The Service has also improperly delayed listing actions through the use of its listing priority systems. The ESA requires the Service to use a "scientifically based priority system" to list and delist taxa, and this system must be based on the "degree of threat" to the taxa (H.R. Conf. Report. No 97-835, at 21, reprinted in 1982 USCCAN 2862). The Service has not done this. Instead, it has created a series of listing priority guidances (LPG) and petition management guidances (PMG) that prioritize species actions based on other factors (e.g. 1983 Guidelines, 48 Fed. Reg. 43,102-43,103; 1996 Guidelines, 61 Fed. Reg. 24,722). As a former FWS biologist noted, instead of using its funds to protect species as Congress directed, the FWS has "seemingly unrestricted use of public funds to carry on litigation and other actions to thwart or delay appropriate classification and regulation…" (Nowak 1997).

The result is to delay listing taxa that are in great danger of extinction. For example, the 1996 guidance allows the Service to ignore a candidate species with a high priority for listing while it is processing species that it has already proposed for listing – even if their priority is lower. The temptation for the Service to propose species for listing in order to delay action on species with greater needs but also a higher political profile is great. And, typically, the Service only proposes species for listing if they will not cause any political controversy. Congress was concerned about exactly such "Don't Make Waves" delays in listing by the Service, and established provisions in the Act for judicial review of the Service so that courts could separate real delays from those caused by "the foot-dragging efforts of a delinquent agency" (H.R. Conf. Rep. No. 97-835 at 22, reprinted in 1982 USCCAN 2860, 2863). Delays further endanger species. The National Research Council (1995b, p. 159-160) has pointed out that decisions must often be made on incomplete data which might be bolstered after a delay – but, that the delay itself "carries a risk" to the species (Ludwig 1999, p. 307).

Easter-Pilcher (1996), Tear, et al. (1995) and Wilcove, et al. (1993) have all criticized the arbitrariness of listing decisions. Moreover, the Service typically waits too late in the slide of a species towards extinction to list it under the Act, often precluding successful recovery and engendering additional economic and societal costs (Wilcove 1993; Wilcove, et al. 1993). The Service has delayed action under the ESA until too late, at the "crash and burn stage" of a slide toward extinction (Houck 1993 at 292; Orians 1993). As the Ecological Society of America points out, delays in listing of species, in designating critical habitat, and in recovering species "bring these species even closer to extinction," "restrict the options available for achieving recovery," and "increase the eventual cost of the recovery process" (Carroll, et al. 1996).

The Service's own scientists and other employees have been forced to criticize the Service for its illegal actions, inactions, and the politicization of its duties. The Public Employees for Environmental Responsibility has termed the Service's behavior towards the statute it is sworn to uphold as a "war of attrition" (PEER 2000). As another FWS employee has noted, the FWS has "consistently missed statutory deadlines for listing and deliberately allowed politics to influence listing decisions" (Sidle 1998a). Indeed, the FWS "is often working against" its function to classify and protect wildlife pursuant to the ESA (Nowak 1997).

The Service has most frequently attempted to excuse its shirking of these duties by claiming that it lacks the funding and resources to fulfill the mandatory requirements of the ESA. However, the Service has routinely misallocated its time, monies, and resources. For example, employees have wasted time on tasks that are not mandatory, instead of focusing on required tasks. Moreover, the mismanagement of its listing responsibilities and other mandatory duties is pervasive throughout the Service from biologists in the field to the highest administrative levels.

The Service has repeatedly requested less funding from Congress than it needs to perform its mandatory duties. Most recently, the Service admitted that "the listing program is not proposed at a level that would allow the Service to meet all of the Act's requirements and deadlines" (Trezise 2000). The Secretary blames this inadequate request on "other compelling needs" such as "habitat conservation planning, candidate conservation agreements, and species recovery plans." However, these activities, while important, are either not required by the ESA or do not have mandatory deadlines under the ESA as do listing and critical habitat designation (and likewise do not carry the mandatory protections of listing and critical habitat), and for the Secretary to request funds for discretionary activities while not requesting sufficient funds for his mandatory duties compounds the illegality of the Service's behavior. FWS biologists spend large amounts of time discussing endangered species issues with ranchers, and other resource users (Webb, personal communication). Although this may be useful to deflect concerns by economic interests, such activities are not mandatory. It is illegal to promote them over working on listing determinations. FWS biologists also spend large amounts of time attending public meetings, when they could be working on listings.

But it is not merely law professors, scientists and the agency's own employees who have criticized the Service's execrable record in enforcing the Endangered Species Act – numerous federal judges have also harshly criticized the Service. In a case involving the Service's refusal to list the Canadian lynx, Judge Gladys Kessler stated that the Service applied an "incorrect legal standard, in clear violation of the plain wording" of the ESA, "relied on glaringly faulty factual premises," and "ignored the views of its own experts." Defenders of Wildlife v. Babbitt, 958 F.Supp. 670, 685 (D.D.C. 1992). Judge Bilby found that the Service rejected a listing petition first "because the goshawks are too homogenous… and then rejected [a second time] because there are too many variations of goshawks." Southwest Center for Biological Diversity v. Babbitt, 980 F.Supp. 1080, 1083 (D. Az. 1997). In another case, Judge Bilby evaluated the Service's policy on distinct population segments, and found the Service's "attitude incredulous." Southwest Center for Biodiversity v. Babbitt, 926 F.Supp. 920, 927 (D. Az. 1996). Another court rejected the Service's habitat conservation plan for the Alabama beach mouse, observing that the plan "was devoid of any rational basis," and noting that it relied on "insufficient, inadequate, and out of date data." Sierra Club v. Babbitt, 15 F.Supp.2d 1274, 1283-84 (S.D. Ala. 1998). When the Service rejected the listing of the Canada Lynx, Judge Kessler found that the Service had based its decision on "glaringly faulty factual premises" that fly "in the face of the overwhelming evidence gathered … by the [Service's own] biologists." Defenders of Wildlife v. Babbitt, 958 F.Supp. 670, 681 (D.D.C. 1997). Judge Bunton reviewed the Service's the decision not to list the Barton Springs Salamander and found that the Secretary "failed to follow proper procedures…, failed to allow comment on issues that were fundamental …missed virtually every statutory deadline provided by the ESA … and considered factors other than those contemplated by the ESA." Save Our Springs v. Babbitt, 27 F.Supp.2d 739, 748 (W.D. Tex. 1997). There are numerous other such cases where federal judges have excoriated the Service (e.g. "Service disregarded all expert opinion on population viability, including that of its own expert" Northern Spotted Owl v. Hodel, 716 F.Supp. 479 (W.D. Wash. 1988). Another judge noted the "deliberate and systematic refusal by the Forest Service and FWS to comply with the laws protecting wildlife." Seattle Audubon Society v. Evans, 771 F. Supp. 1081, 1090 (W.D. Wash. 1991). If individual citizens were guilty of such behavior, they would likely be jailed for contempt.

Congress has also criticized the Service – when it created the "warranted but precluded" category, it referred to the Service and cautioned against "the foot-dragging efforts of a delinquent agency." H. Conf. Rep. No. 97-835 at 22 (1982), reprinted in 1982 USCCAN 2860, 2863; as noted by the court in Center for Biological Diversity v. Norton, 254 F.3d 833, 838 (9th Cir. 2001).

As of 1996, over 3,000 species were listed as "candidates" under the Endangered Species Act (Carroll, et al. 1996). The Service's use of candidate designations and the claim that species are "warranted [for listing], but precluded" by other work has been termed a "black hole" by Prof. Oliver Houck (1993, p. 286), perhaps the most prestigious legal commentator on the Endangered Species Act. The Nature Conservancy estimated that 165 unlisted species, including 30 vertebrates, 84 invertebrates, and 51 plants have become extinct while the FWS focused its attention on "higher priority" matters (Houck 1993, p. 286, n. 65. The Ecological Society of America criticized the Service for allowing over 114 species to languish in the "warranted but precluded" category for two or more year, and allowing 56 species to remain in that black hole for 8 years or more (Carroll, et al. 1996).

Extinction is the inevitable result from the Service's "delinquent foot-dragging." On September 19, 1997, the USFWS deleted five species from the list of "candidates" for listing as threatened or endangered. The reason for the action is that the five species, which had been in candidate status for years without any steps taken toward listing, had gone extinct in the meantime. The prioritization process that the Service uses to consider which species to consider for listing first does not take into account the protective effects that the listing of one species could have on other species and on ecosystems, nor does it consider the ecological role or importance of a species in its ecosystem (Carroll, et al. 1996).

The Service has failed to explicitly define risk thresholds for endangered or threatened species listings, most likely in order to maximize discretion in individual listing and recovery decisions for political purposes (Carroll, et al. 2012). The Service has attempted to limit the protection due species at high risk of extinction by restricting the "significant portion of a species range" language in the ESA to only the current species range, not the historic range, and only to those areas where extirpation is most imminent. Greenwald (2009).

The Service also appears to be abusing its emergency listing authority. The Service emergency listed 10 species from 1980 through 1988 (Lieben 1997, p. 1351 and n. 219), and emergency listed 6 species from 1988 through 1995 (Lieben 1997, p. 1351 and n. 220), yet has not exercised its emergency listing authority since 1995 – not even once (Lieben 1997). The Service has been requested to use its emergency listing authority by numerous petitioners, and the "duty to consider emergency listings" is "part of the petitioning process." Friends of the Wild Swan v. US FWS, 945 F.Supp. 1388, 1395 (D.Or. 1996). Congress mandated that the

Service exercise its emergency authority "to prevent a significant risk to the well-being" of species (Endangered Species Act Amendments of 1988, Sen. Rep. No. 100-240 at 7 (1987), reprinted in 1988 USCCAN 2700, 2707). The emergencies faced by species have not declined since 1995; instead, the Service has illegally avoided its duties under the Act.

One example of the Service's failures is illegal lack of actions to uphold the ESA with respect to sage grouse. First, the Service affirmatively sought to undermine potential petitions and the listing procedure: "Mr. Ireland [a FWS employee based in Grand Junction, Colo.] is trying to keep all the players in the loop so a petition is not started" and is "trying to slow things down" *(Summary of North Park Working Group Meeting 1999)*. Such actions are per se arbitrary and capricious and implicate violations of the Service's duties and trust responsibilities. Later, Mr. Ireland stated that conservation plans could provide some "relief or relaxation from [ESA] listing" *(id., p. 2)* and that after a listing "grazing will not be shut down at all" *(id., p. 3)*. Service employees also discussed the significant impacts of a listing petition and the advantage of "being pro-active" to "divert potential litigation" *(Parker 1998, attachment)*. The ESA mandates various affirmative duties on the FWS to evaluate threats to the species and to use the best available science in its evaluation, but employees have contemplated abrogating those duties. Personnel noted that if the sage grouse were petitioned range-wide, "US FWS could conclude [that the listing was] 'not warranted' if [the] petition [was] not good" *(Anonymous FWS document 1999, p. 2, Rich Howard speaking)*. Almost immediately after the American Lands Alliance held a public meeting to discuss the plight of sage grouse, the Service held a conference call *(Anonymous FWS document 1999)*. FWS employees discussed how to evaluate the danger to sage grouse but without moving towards protection for the species. As early as Jan. 1999, FWS intended to ensure that a sage grouse petition would be "going to [the] bottom of the pile" (*Anonymous FWS document 1999, p. 3, Rich Howard speaking)*. FWS decided they would "have to call it [the status review] something else" to "keep [it] out of listing" procedures *(Anonymous FWS document 1999, p. 3, Chuck Davis speaking)*. The FWS admitted that they already "have data" but "need synthesis" *(Anonymous FWS document 1999, p. 3, Chuck Davis speaking)*. Although the Service has the data, it has taken no steps on its own to consider protection for sage grouse. The Service is not allowed to consider economic or any non-scientific issues when deciding on listing a species. Yet, the Service constructed a Regulatory Action Alert prior to the listing of the Washington population of sage grouse that discussed the "potential importance to industry" and which Congressional Districts were involved in the area covered by a listing *(FWS 2001c)*.

Yet, it is rarely the biologists in the Service who delay listings, craft inadequate critical habitat designations and recovery plans, or otherwise break the law. Instead, political appointees and administrators have frequently interfered with the biological findings by agency employees. It was just such politically motivated actions that caused Judge Jones to remand the bull trout listing case to the Service, that led to the series or remands in the lynx listing cases, and that led to formation of an organization dedicated to ethical and professional service to the public, the Public Employees for Environmental Responsibility, PEER, (2001 S St. NW, Washington, DC 20009; www.peer.org). As Yaffee (1982, p. 89) noted in his critique of the Service's carrying out of its ESA duties, "political considerations are increasingly incorporated at higher levels of the bureaucracy."

Management of Critical Habitat

Critical habitat designation under the US Endangered Species Act (ESA) is not working as Congress intended, and problems include the use of science during designation or avoiding the use of science thereof, and the extraordinary and illegal delays in designation that imperil species protections. Hodges and Eldera (2008).

Although the Service routinely violates Congressionally mandated deadlines under the Endangered Species Act it dislike being brought to heel by the judiciary, operating under the citizens suits provisions enacted by Congress as a safeguard for just such contingencies. Yet, scientists who have analyzed the outcomes of such citizen initiated law suits have found critical habitat designation is actually improved when the Service is forced into action by lawsuits. When the US Fish and Wildlife Service faces a Court Order to designate critical habitat, the habitat designated uses more of the biologically-based designation criteria than when the Fish and Wildlife Service is left alone and operates without facing a court injunction. Hodges and Eldera (2008).

The US Fish and Wildlife Service is not consistent with its use of its own biologically-based designation criteria with respect to taxon and region. Hodges and Eldera (2008). The US Fish and Wildlife Service is more likely to use designation criteria that focus on discrete elements, such as nest sites or locations where required food species occurred, than to use criteria that addressed broader ecological needs such as space for normal behavior or representation of historic range conditions. Hodges and Eldera (2008). Space for critical behaviors and representation of historic range and habitat conditions are important are important for both recovery and for population persistence, and the Service cannot legally, and as a practical matter, should not neglect these considerations.

Management of Recovery Planning

The goal of the Endangered Species Act is to recover listed species so that the economic, esthetic, religious, and environmental benefits of their existence may be enjoyed. Congrss, therefore, mandated certain requirements for recovery plans. However, studies show that the Services recovery planning is deficient.

In an analysis of 41 Recovery Plans, Elphick, et al. (2001) found that recovery plan goals were not based on biological traits of the species for which the recovery plan was written. Moreover, taxonomic bias in recovery plans is a problem. Clark, et al. 2002. Recovery and spending on recovery plans is important. A 2007 analysis of the ESA's effectiveness found that endangered terrestrial and freshwater vertebrates became less endangered only if substantial funding was spent on their recovery (Ferraro, et al. 2007). Worse, many recovery plans are out of date and do not address current scientific understanding of either population mechanisms nor of current broad ranging threats, such as climate change (Povilitis and Suckling 2010).

To promote more effective recovery planning for species listed under the U.S. Endangered Species Act (ESA), the Society for Conservation Biology (SCB) sponsored a systematic review of a large sample of ESA recovery plans. The review was conducted in collaboration with the U.S. Fish and Wildlife Service, the National Center for Ecological Analysis and Synthesis, and 19 universities. The scientists and agencies found several problems with recovery plans: (1) threats to species should be a primary focus of recovery plans, (2) adequate monitoring tasks for species status and recovery should be specified in the recovery plans, and (3) recovery plans should ensure that species trend data are current, quantitative, and documented. Clark, et al. 2002.

Suborning of Agency Personnel and Administrative Capture

Another factor affecting the inadequacy of existing regulatory mechanisms is the suborning of both state and federal agencies by powerful extractive interests. The Service must therefore consider the degree to which proper management regimes will, in fact, be carried out by these agencies, as well as whether scientists have the independence to perform adequate studies and to make unbiased recommendations.

The objective nature of scientific inquiry has been a maxim for decades. Only recently have critics claimed that the self-correcting nature of modern science lacks objectivity. Unfortunately, several studies bear out this problem. University faculty "with industry ties are more likely to report research results that are favorable to a corporate sponsor" and "are more likely to conduct research that is of lower quality" than comparator researchers (Cho, et al. 2000 – reviewing results of 6 studies). Notably, the studies Cho reviewed are all experimental or clinical in nature – these are the types of studies that ought to be least influenced by a lack of objectivity. Studies that are non-experimental in nature, and rely instead on observational or descriptive techniques (as do much of the literature on wildlife and related issues) are less likely to be self-correcting precisely because they lack experimental controls. Although bias is not inherent in such studies, it is likely to be more common. The person doing the research can also be a source of bias. University scientists, who often possess tenure, ought to be more reliable than government scientists who can be transferred at the whim of a politically motivated bureaucrat. However, no area of inquiry is too abstract or arcane to escape political attack. National Science Foundation and university grants were denied to a politically active mathematician (Rorabaugh 1989, p. 104-105). Moreover, such transfers and demotions have happened many times to scientists and managers in both federal and state governments. On other occasions, transfers or demotions have been threatened, but opposition from public interest organizations has prevented them.

University scientists who have conducted work on wildlife, forestry and other land management issues are usually lodged in Schools of Agriculture, which are closely tied to industrial and agricultural interests. The bias inherent in such close ties has been criticized by journalists (Marston 1993). Marston notes that Colorado legislators have attempted to intimidate professors at the state universities regarding water issues in the state, and suggests that land grant universities have not provided useful information on solving contemporary natural resources problems, because they are either too "cowed" or "too trained" to do so (Marston 1993). Marston notes that during the 1970's, Johanna Wald (attorney for the Natural Resources Defense Council) was told by every range science professor she talked to, that "everything [was] fine" regarding grazing on the public's lands, belying the crisis in grazing reform. The student clinic at the University of Oregon law school was also attacked during the spotted owl campaign and ultimately forced off campus. The Service should take into account the quality of the data and analyses conducted by any scientist whose employer or granting sources are subject to bias. A regulatory mechanism that is based on biased science is *per se* inadequate.

A more pernicious and subtle lack of objectivity relates to bias in the selection of questions to be investigated. Problems may not be researched if funding is unavailable for political reasons, if the likely results may harm the agency doing the research, or if an individual scientist – unconsciously, or not – believes that no problem exists worth investigating. Such beliefs may be based on political or cultural motivations, such as co-optation of an agency scientist by the entity that the agency is supposed to be regulating, a phenomenon so common that it bears its own term "administrative capture." In such cases, the self-correcting schema of experimental hypothesis testing is never given a chance to function. Science, *strictu sensu*, is not done at all.

Ecosystem Protection

Petitioners request that the Service recognize and consider the importance of this species as a keystone species, , and as an umbrella species, which can protect numerous other species in this ecosystem type in this part of the United States. Petitioners request that the Service incorporate such considerations into all aspects of its section 4 responsibilities under the Endangered Species Act. By protecting this species, the Service will also be protecting an entire ecosystem type, and the other species that depend on this ecosystem. This fulfills the statutory purpose of the Endangered Species Act, 'to provide a means whereby the ecosystems upon which endangered or threatened species depend may be conserved…." 16 U.S.C. § 1531(b). The Service recognizes that the conservation of such ecosystems is "a primary purpose of the Act." 59 Fed. Reg. 34273, Friday, July 1, 1994. The Service adopted the "Ecosystem Approach to Fish and Wildlife Conservation" on July 1, 1994, explaining that "species will be conserved best not by a species-by-species approach but by an ecosystem conservation strategy that transcends individual species." 59 Fed. Reg. 34273. By listing this species under the protections of the ESA, the Service has a chance to comply with that policy. Moreover, the Service should take into consideration the importance of ecosystem conservation when assigning priorities in the listing process.

This species functions as a "keystone food resource" (Terborgh 1986; Meffe and Carroll 1997) supporting many predator species, and also serves important ecosystem functions. Extinction or reduction in numbers of this species could be a primary extinction or effect that necessarily causes multiple extinctions of numerous other species in these ecosystems because of food web interactions and other community level effects (Terborgh 1976, Wilcox and Murphy 1985). As the National Biological Service noted, a "significant decline in a once-dominant or keystone species could have profound ecological ramifications," and these "ecosystems-wide effects could occur long before a pivotal species becomes rare enough for listing as endangered" (Noss, et al. 1995).

In the past, the Service has not considered ecosystem effects in its listing priorities because ecosystem "information is seldom available at the time a species is considered" (1983 Guidelines, 48 Fed. Reg. 43098, 43101, Sept. 21, 1983). Here, extensive information is provided to remove that barrier. The Service promised to consider "ecosystem importance" on an ad hoc basis (id.), and petitioners formally request such consideration here.

Request to Apply Trust Doctrines

Petitioners request the Service to apply the legal doctrines known as the Wildlife Trust Doctrine and the Public Trust Doctrine in its consideration of this petition. Petitioners further request the Service to exercise all its trust responsibilities in all matters affecting the species discussed herein, including the Indian Trust Doctrine as respecting the importance of these species as cultural symbols, as religious objects, and as food sources, both currently and in historic times.

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