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The Sustainability of Freshwater Species and Water Resources Development Policy of the Army Corps of Engineers

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CONTENTS

List of Figures

List of Tables

EXECUTIVE SUMMARY

ISSUES AND PURPOSE

Accelerated species extinction and associated loss of biodiversity are among the leading environmental concerns worldwide. At congressional hearings in 2002, the same year that the U.S. Army Corps of Engineers (Corps) first declared in its Environmental Operating Principles (EOP) that it will strive to achieve environmental sustainability, critics testified that the Corps Civil Works projects have been and continue to be "among the leading reasons" for freshwater species disappearance in North America. The critics implied that the Corps was a leading cause of all extinction and imperilment in the United States based on scientific evidence that freshwater loss rates were 5 times greater than terrestrial loss rates, growing faster and rivaled rainforest rates. They also indicated that "agency-wide biases, institutional barriers, and faulty analyses are all contributing to the continued degradation of the nation's rivers and wetlands." If these and other claims are accurate, Corps actions since the National Environmental Policy Act, the Endangered Species Act (ESA) and other Federal environmental laws were passed have been inconsistent with long-established policy that commits the Corps to pursuit of beneficial national economic development (NED) while protecting the environment.

These claims raised concerns about Corps policy effectiveness, an issue that was the major impetus for this study. But an equally important motivation for the study was identification of areas of freshwater species decline where the Corps could use its environmental mission and ecosystem restoration authority to address environmental problems of national significance. The objectives of this study were: 1) to estimate past and projected future rates of species extinction in the freshwaters of the United States; 2) to compare freshwater species extinction rate to terrestrial extinction rate in the United States and to published estimates of worldwide rainforest extinction rate; 3) to examine the historical, geographical and ecological context of extinction and imperilment in the United States for indications of cause; and 4) to discuss the issues that emerge and issue-management difficulties and 5) to identify potential restoration opportunities for the Corps. The results are compared with published analyses used to support claims that the Corps has contributed and continues to contribute largely to conditions that threaten the sustainability of freshwater species.

METHODS

The review of existing methodology revealed shortcomings in all methods and databases used to estimate past and future extinction rates because of ecological, taxonomic and historic uncertainties. They are described in detail in the main text. However, growing claims of increasingly high rates of extinction and imperilment are troubling and conservation biologists cautiously accept the best methods available, despite their imperfections, because of pressing needs to inform decision makers and natural resources managers. The resulting uncertainties leave substantial room for different interpretation of extinction and imperilment rates, and their causes. This study nonetheless produced some reasonably confident conclusions because of recent advances in information availability.

The availability of information about the conservation status of species native to the United States has much improved because of the efforts of NatureServe, a conservation organization that organizes information on species conservation status for the network of international natural heritage programs and other users of the database, NatureServe Explorer. The results presented here relied on NatureServe Explorer data and, secondarily, on other published reviews of species extinction and imperilment status and causes.

The analysis was limited to the 50 states of the United States to assure inclusion of the most reliable datasets consistent with the geographical area most influenced by Federal water resources development. Future rates of extinction were estimated from the imperilment status of species indicated in NatureServe Explorer data and from the endangered species listed under ESA protection, the assumption being that all are ultimately doomed to extinction. Future rates were also estimated from trend forecasts of past species extinctions based on dates of last observation. The evaluation of past extinction cause is based on a chronological analysis of the last dates of species observation determined from the NatureServe record between 1880 and 1989, published understanding of species ecology and physiology, and the chronological and geographical alignment of environmental change with the known ranges of extinct and imperiled species. In many cases, specific potential stressors, such as the actions of the Corps of Engineers, could be eliminated because they did not geographically and chronologically align with extinction patterns. Alignment analysis cannot provide proof of cause, but does indicate the possibility. The possible causes of extinction were rarely reducible to a single source because of the multiplicity of probable stressors in most locations and insufficient data.

The extinction results were compared to those of Ricciardi and Rasmussen (1999), which have been used to support claims of high freshwater biodiversity loss. Ricciardi and Rasmussen made no claims about specific cause, however. Consistent with their study, this study focused on vertebrate, freshwater mollusk and crayfish species. It also examined the effect of including other taxonomic groups and different levels of extinction confidence on estimated extinction rates. The effects of including or excluding Hawaii were also analyzed to assess continental status and trends.

RESULTS

Estimated recent extinction rates for freshwater species in the United States are at least 1000 times background rates of extinction (before European colonization). This study estimated the freshwater extinction rate in the United States to be about 3 times the terrestrial extinction rate for the same groups and categories of extinction used by Ricciardi and Rasmussen. While less than their estimate of 5 times, the difference is substantial and their point is confirmed. Confidences in the results varied widely depending on what taxonomic groups were included; being substantially greater for the vertebrates and crayfish than for mollusks, however. The ratio dropped to 1.2 when the least known group, the mollusks, was excluded from the analysis and to 0.9 when fish species with questionable full species status were also removed. When Hawaii was excluded, the ratio increased to between 2.8 to 1 and 20 to 1, depending on the taxonomic groups included. Thus the general concerns raised by Ricciardi and Rasmussen proved to be confirmed for the continental United States. Continental rates of extinction estimated in this study were similar to the lowest estimates of rainforest rates of extinction, which vary widely in the literature.

Forecasts of future rates of extinction vary with method used. Rates based on imperilment status (similar to the methods used by Ricciardi and Rasmussen) and ESA endangerment status were substantially greater than extrapolations from past extinction rates, much as Ricciardi and Rasmussen reported. Long-term trend extrapolation from extinction records predicted a leveling of extinction rates over the next few decades with an increasing proportion of freshwater species extinctions in the continental United States. Rates of extinction for freshwater vertebrate and invertebrate species were predicted to remain stable; at close to recent past rates of loss and the invertebrate extinction rate was predicted to be greater than the vertebrate rate. Based on past extinction history, mollusks are by far the most vulnerable group. Based on imperilment status, however, amphibians and crayfish are also quite vulnerable. Whereas predictions of extinction based on the assumption that imperiled and endangered species are doomed indicates a worse-case scenario, the estimates based on trend extrapolation over recent decades are more consistent with public environmental awareness and dramatic development of environmental law since the 1960s.

Life history data indicate that many freshwater species inhabiting small, isolated ecosystems and mollusks that inhabit shoals and riffles of river ecosystems are especially vulnerable to extinction. These species comprise the majority of those in the continental United States that are now considered globally extinct. Many potentially stressful changes have occurred in their aquatic environments. Environmental impacts began to accumulate in some freshwater ecosystems before the 19th century. Specific causes of extinction and imperilment are best documented for vertebrates, including some amphibians and fish. Most extinction is associated with agricultural/urban development (more specifically with accelerated erosion and sediment deposition, groundwater withdrawal, and spring alterations) and with habitat invasion by non-native species (often intentionally introduced by humans).

Water resources development or operations have been linked to extinctions based on geographical and chronological proximity to the extinctions and to understanding of species ecology. About one-fourth of freshwater vertebrate extinctions recorded for the United States are linked to large water resources development, but only 15% are linked to Federal projects. Corps projects may have played some role in three freshwater vertebrate extinctions (about 11% of the total) but the evidence is tenuous. No terrestrial species extinctions are linked to water resources development activities. Thus, no more than 6% of all aquatic and terrestrial vertebrate extinctions recorded in the United States can be linked to Corps projects. Similar to extinctions, about one quarter of the imperilment of freshwater vertebrate species can be linked to large water resources projects based largely on geographical proximity of impoundment and other water resources development impacts to the original ranges of the species. Also similar to past extinctions, Federal projects make up a smaller fraction and Corps projects a smaller fraction still. The numbers are significant, however, because of the large number of freshwater vertebrate species that are now imperiled.

Among invertebrates, crayfish extinction and imperilment is linked most usually to nonnative species introduction and to pollutants. The history of mollusk status change, especially among snails, is less certainly known than for other groups. Early freshwater mollusk decline appears to be most linked to changes in hydrology, erosion and sedimentation associated with deforestation and agricultural development and to domestic, industrial and mine pollution. Nearly half of all mussel extinctions and most snail extinctions are linked to more recent large-scale water resources development because of the major hydraulic and erosion-depositional changes the development has brought about in their ranges. The large majority of extinct species once lived in riffle and shoal habitat and required flowing water over generally silt-free bottoms. One-fourth to one-third of all freshwater mollusk extinctions are linked to causes geographically and chronologically associated with Corps water resources development projects. Similar fractions of imperiled and endangered species are associated with water resources projects in total and with Corps projects specifically. Molluscan imperilment from water resources development is particularly concentrated in warmwater tributaries of the Mississippi River and the Gulf of Mexico. Because the number of freshwater species that are now vulnerable to extinction is so large, the number of imperiled species affected by Corps projects is in the hundreds, and higher still for all water resources development projects.

Future threats to species continue to be associated largely with nonnative, invasive species and with agricultural and urban development. Even so, future protection and recovery of threatened and endangered species may emphasize Federal water resources management disproportionate to its threat level because water resources projects are somewhat more manageable in the short run than many invasive species and land-use threats. Water resources management may be one of the more effective ways to manage some threats from land use and invasive species.

DISCUSSION AND CONCLUSIONS

Documentation of species losses and their causes has been uneven at best and often uncertain, but some conclusions can be made with reasonable confidence. The evidence indicates that the claims of some critics—that the Corps is among the primary causes of freshwater biodiversity loss—are exaggerated. However, strong circumstantial evidence suggests that Corps projects have contributed to some past extinction (mostly before the environmental laws of the 1970s) and continue to contribute to present imperilment of numerous species. Freshwater extinction rates in the United States have accelerated, are higher than terrestrial rates and are likely to remain higher than desired by the public, as stated in the goals of the ESA, if present conservation effectiveness does not improve. Estimates of future extinction rate based on present imperilment are higher than estimates based on trend extrapolation. Estimates based on imperilment probably are overly pessimistic given the apparent effectiveness of law and education in slowing the extinction rate in recent decades. Even so, the list of potential candidates for ESA protection continues to grow without commensurate increases in funding. The water resources development agencies, including the Corps, may be able to do more to achieve environmental sustainability by contributing to the security of presently imperiled species consistent with their authorities and responsibilities.

An outstanding opportunity exists for the Corps to apply its unique aquatic ecosystem restoration authority to reversing the decline of freshwater biodiversity in the United States in collaboration with other agencies and with non-government organizations. There is little doubt that the most consistently important recovery need for imperiled freshwater species is habitat improvement.

The results of this study affirmed that management needs are greatest in island-like habitats of exceptional extinction vulnerability, including the freshwater "islands" in the continental United States, which include the medium to large rivers where the Corps Civil Works Program is most active. Many of the species occupying those rivers are exceptionally vulnerable to pervasive impacts of habitat change from various chemical and physical causes, including water resources development. However, most freshwater species extinction and imperilment are attributed to urban-agricultural development and invasive species. Widespread alteration and fragmentation of river habitats from all sources has contributed greatly to the imperilment of the remaining species populations.

Most past extinctions in freshwater ecosystems had their basis in land and water changes before science and the Nation recognized the threats. With few exceptions, extinctions and the present imperilment of species occurred before the environmental laws of the 1960s and 1970s. The Corps was probably involved in the extinction of a small fraction of fish species and more than half of the extinctions of invertebrate species. The causes of present imperilment are similar to the causes of extinction, many of which originated before the environmental laws of the 1960s and 1970s. Since then, Corps policy and history indicate that it has been and continues to be as serious about and responsive to statutory authority and regulation as other agencies. The Corps declared a commitment to striving for environmental sustainability in the EOP of 2002. The Corps has established a strategic plan with goals that highlight sustainable development, environmental repair and environmental sustainability. However, increasing pressure to serve all of its authorized purposes with a limited budget encourages the Corps (much like other agencies) to do little more than required by law.

Limited budgets also contribute to the mixed effectiveness of the ESA based on various assessments reviewed here. While the ESA appears to have slowed species decline once species are listed as threatened and endangered, it has been much less successful recovering species to a secure enough status to delist them. No freshwater aquatic species has been delisted as a consequence of recovery. The failure is systemic, however, and in general not attributable to individual agencies. The Corps has participated in many endangered species recovery plans and has contributed significantly to relevant research and management efforts consistent with budget constraints. The problems faced by the ESA program and by government commitment to species sustainability in general are basically the same as for all discretionary Federal programs. The discretionary Federal budget has been in decline since the 1960s as the budget needs of mandated programs have increased.

In addition, the listing process of the ESA has failed to keep up with the large number of species now considered imperiled by conservation biologists. While the agencies that administer the ESA have promoted reversing the decline of species before they need to be listed under ESA protection, the law itself has no influence. The need to reverse the decline and restore imperiled species not yet listed to a secure status has to be addressed through other means. A large effort by non-government biodiversity conservancies to protect remaining habitat has been underway for decades, but the challenges are daunting and protection of remaining habitat may not be enough for many species, especially in freshwater ecosystems. The conservancy nongovernmental organizations (NGOs) realize that all sources of funding, expertise, and effort have to be better coordinated and collaboratively applied to be cost-effective. The Nature

Conservancy recently signed an agreement with the Corps with this in mind and has partnered in numerous new projects and operational adjustments with biodiversity recovery in mind.

The efficiency and effectiveness with which Corps authorities and funding are used to restore and protect vulnerable species can improve, however. Both inside and outside the Corps, evidence indicates that efforts are less integrated, strategic and systems-based than desirable for cost effective protection and recovery. To further improve, the Corps and other agencies need to be more vigilant in their analysis of traditional project effects, more thoughtfully involved in systems approaches to integrated resource management and more aggressive in seeking restoration opportunities promoting restoration of habitats in “hotspots” of species vulnerability. Civil Works policy needs to be reviewed, refined and clarified to meet the need, especially with respect to ecosystem restoration guidance and to systems-oriented planning at a regional scale. Visionary leadership and inter-organization collaboration are the keys to more effective action. Just as it has not stood alone among the causes of past species extinction and imperilment, the Corps cannot stand alone in the solution of one of the most challenging environmental problems in the United States today—the progressive loss of the Nation’s freshwater biodiversity.

INTRODUCTION

ISSUES

Global Biodiversity Concerns

Documentation of increasing rates of species extinction and imperilment has caused widespread concerns over future global loss of biodiversity (e.g., Wilson and Peters 1988, Lubchenco et al. 1991). The genesis of species conservation concerns goes back more than a century in the United States and elsewhere to a time when the major recognized threat to species viability in the United States was unregulated hunting and fishing. Before 1900, the problem was addressed primarily by state law and local ordinances. At the Federal level, interstate transport of illegally killed bird species was prohibited in 1900 and migratory bird species were more fully protected in 1916.

The importance of habitat loss displaced hunting and fishing as major causes of species endangerment in the United States, especially after World War II, when the Department of Interior, Fish and Wildlife Service (FWS) began to catalog species in decline. In 1966, Congress passed the Endangered Species Protection Act, which authorized the FWS to identify endangered species and use revenues collected under the Land and Water Conservation Act to purchase and protect needed habitat. Protections were strengthened with passage of the Endangered Species Act (ESA), which authorized the FWS and National Marine Fishery Service (NMFS) to list threatened and endangered species, identify critical habitat, enforce protection of listed species and oversee species recovery to a viability status that would allow delisting.

The ESA is among the strongest laws of its kind, especially with respect to critical habitat protection. The ESA motivated much-improved documentation of the extent and depth of species imperilment in the United States and provided measures of last resort (NRC 1995) to prevent extinction. Particularly relevant to Corps policy, it directs all Federal agencies to use their authorities to promote conservation of listed species, including recovery to a secure status. Despite chronically controversial aspects, several amendments over 30 years and slow progress in recovering listed species, it remains among the most effective national laws for preventing extinction of globally imperiled species and subspecies.

The growing threat of species extinction from excessive harvest and habitat destruction grew rapidly in developing nations after World War II and topped lists of global conservation issues by the 1980s, especially for tropical rainforests and other areas hosting high or unique terrestrial biodiversity (Myers 1979, Wilson 1988, Wilson 1989). By the early 1990s, the Ecological Society of America had placed biodiversity loss among the trends of greatest ecological concern, requiring serious research attention (Lubchenco et al. 1991). Wilson (1992), Pimm et al. (1995), and Reid (1997) warned that species losses could reach 20 to 65% or more of all species on earth during the 21st century if past trends continued.

Substantial variation is evident in the forecasts of different rainforest studies. More specifically, assuming a 1% per year loss of tropical forest, Wilson (1989) estimated that 2 to 3% of all species could be lost per decade. Reid and Miller (1989) estimated 2 to 11% per decade depending on deforestation rates and different assumptions for species-area curves. Reid (1997) more recently lowered the estimate to 1 to 8% per decade.

More recent analyses and reviews include substantially more conservative estimates of extinction increase. They reveal that substantial variation exists depending on taxonomic groups and factors operating in and among habitats (e.g., May et al. 1995). Among the more careful and recent analyses, May et al. (1995) consistently estimated extinction of half of the birds and mammals in the world within 200 to 400 years, an average loss rate of about 0.12 to 0.25% per year (15 to 30 species per year) or 1.2 to 2.5% per decade. The estimates were made only for the best-documented species—the birds and mammals—using a species-area model and two IUCN (World Conservation Union) based methods, each with “serious shortcomings” admitted by May et al. (1995).

Lomborg (2001) estimated a world extinction rate of about 0.1% per decade (most of it tropical rainforest), which is two orders of magnitude lower than the high estimate of 11% per decade made by Reid and Miller (1989). But even the more conservative analysts conclude that human actions have increased the world-wide extinction rate of certain taxonomic groups by at least 1,000 times the natural background rate (Lomborg 2001). This is more consistent with the worldwide increase in extinction rates for birds and mammals estimated by May et al. (1995), which are thought to be more vulnerable to extinction than most species.

During the last three decades, conservation biology has coalesced and matured to become a strong organizational force for building more advanced conservation science, more informed publics and more effective conservation policies. Having both social and political dimensions, conservation biology is much broader than its name implies. It has strong ties with the non-government biodiversity conservancies and, together, they dominate global strategic thinking about sustaining biodiversity (Dinerstein et al. 2000, Groves et al. 2000, Groves 2003).

The approach of conservation biology now most advocated emphasizes protection of biodiversity at ecoregion, ecosystem, community and species population levels because of the interdependent interactions that occur across all scales of organization (Groves 2003). A key objective is to sustain ecological and evolutionary processes that continuously regenerate diversity. Because evolution occurs at the level of individuals and populations, the objective is conceptually population centered, but achieved through strategic protection of entire communities, ecosystems and underlying ecological processes. Biodiversity in all of its forms is assumed to be sustained if representative examples of each population can be sustained in a naturally adaptive state where populations interact with one another and their physical environment.

With respect to global changes, the main concern has been the accelerating rate at which natural landscapes in undeveloped nations are being converted to human use primarily for forest products and agriculture, but also for human habitation and transportation-communication networks. Tropical rainforest diversity dominated the concerns of the earliest conservation biologists (e.g., Myers 1979) because most species are located there and the rate of ecosystem conversion to intense forest and agricultural use was rapidly accelerating. Biodiversity loss was projected primarily from what could happen more than from what had already happened.

Except for urban expansion, terrestrial ecosystem conversion in the United States has largely stabilized. Agricultural use of lands expands and contracts with changes in agricultural policy and economic conditions. Species loss, both past and future, is linked less with what remaining natural ecosystems could be converted to intense human use than with what has already happened. The issue is now more about effective implementation of environmental policies and environmental changes set in motion by past conversion of terrestrial ecosystems and present land use practices.

Freshwater Species Concerns

Among the important effects of land use in the United States are changes in watershed hydrology, erosion, and sediment, nutrient and other materials output that have had major impacts on aquatic ecosystems. More or less complete protection of lands in wilderness areas and natural parks exists in less than 10% of the land area in most large river basins of the United States (Revenga et al. 1998). Profoundly important links between watershed condition and the geophysical and ecological condition of freshwaters have long been recognized, as has the utility of watershed-based planning and management to improve degraded flow, water quality, substrate, and channel conditions (e.g., Hynes 1975, Hasler 1975, NRC 1999a, Naiman 1992, Williams et al. 1997, Wissmar and Bisson 2003, Cole et al. 2005).

Similarly, the importance of stream-flow connectivity in sustaining flowing water structure and function has been well established in theory and in many specific analyses of impacts by dams and other structural alterations and their management (Ward and Stanford 1979, Vannote et al. 1980, Petts 1984, Poff et al. 1997, Minshall et al.1985, Stanford et al.1996, Hart and Poff 2002, Palmer et al. 2005). The large extent of impoundment and other structural alterations of rivers in the United States and elsewhere is also well documented (Dynesius and Nilsson 1994, Graf 1999). While often eclipsed by rainforest and other terrestrial concerns, freshwater species decline in the United States has been documented for many years, especially for fish (Stiassny 1996) and for waters of the western and southeastern United States (Benke 1990, Minckley and Deacon 1991, Lydeard and Mayden 1995, Neves et al.1997). However, the recognition of loss and how it might be connected to land and water transformation did not develop much before mid-20th century when concepts of ecosystems, genetics, evolutionary ecology and the process of extinction began to advance rapidly. A new awareness spread quite quickly from science to institutional actions, most prominently in the unprecedented environmental legislation of the 1960s and 1970s.

Species listed under ESA protections have included a significant fraction of freshwater species since the first list was published nearly four decades ago. The American Fisheries Society (AFS) published two influential series of articles on the conservation status of fish (Williams et al. 1989, Miller et al.1989). Concern continued to grow during the 1990s (e.g., Allan and Flecker 1993, Richter et al. 1997) and the AFS continued its review of aquatic species conservation status in publications on aquatic mollusks (Williams et al. 1993, Turgeon et al. 1998) and crayfish (Taylor et al. 1996). Freshwater species and supporting ecosystems achieved greater prominence in the agendas of biodiversity conservancies (e.g., Abell et al. 1998, Stein et al. 2000).

Despite the increased research and conservation activity done in the 1990s, it was the view of some that the community of conservation biologists placed more emphasis on terrestrial conservation than the status of freshwater and terrestrial species actually justified. Ricciardi and Rasmussen (1999) investigated this premise through an analysis of AFS publications and other assessments of freshwater conservation status and compared it to existing terrestrial data. They estimated the North American freshwater extinction rate to be about five times the terrestrial extinction rate in well-documented taxonomic groups (mammals, birds, reptiles, amphibians, fish, crayfish and freshwater mussels) and similar to estimates of rainforest extinction rates (which are widely accepted as among the highest rates in the world). They also concluded that the ratio of freshwater to terrestrial extinction probably would increase based on the fractions of species considered to be seriously threatened with extinction (imperiled) in the scientific literature they reviewed. Ricciardi and Rasmussen summarized the many causes of extinction and imperilment generically from their reading of the scientific literature, but did not analyze it in detail nor rank relative importance of each cause.

There is little doubt that the geophysical and ecological conditions of freshwater ecosystems and their watersheds in the United States (and elsewhere in the developed world) have been largely transformed by humans and that the relative abundances of many species has changed. That in itself does not necessarily indicate cause of extinction or threat of extinction if enough intact ecosystem remains to sustain species viability. In the broadest sense, however, there is little doubt that the transformation has had, and is continuing to have, an impact on the sustainability of freshwater and terrestrial species. Yet the specific connections between ecosystem alterations and species extinction and imperilment are less well documented than might be expected.

Corps of Engineers Policy Concerns

In congressional hearings held in 2002, Civil Works projects of the U.S. Army Corps of Engineers (the Corps) were lumped "among the leading reasons" for documented disappearance and imperilment of freshwater species in North America. In that same year the Corps declared that it will strive to achieve environmental sustainability in its statement of Environmental Operating Principles (EOP). That was, in many respects, the culmination of a long history of integrating environmental policy into Corps planning and operations policy, including the procedural requirements of the National Environmental Policy Act and the ESA. In 1986 the Corps received environmental improvement authority, which evolved into ecosystem restoration authority. These new authorities reinforced a sense of responsibility for the environment and the criticisms were a cause for some concern about how well policy was being carried out.

Because the fauna of the United States—especially the freshwater species—adds significantly to global biodiversity (e.g., LaRoe et al. 1995, Stein et al. 2000), these claims imply much about the leadership of the United States government in reversing world-wide trends in biodiversity loss. If the claims are accurate, Corps actions are inconsistent with the EOP and other Corps policy, which commits the Corps to pursuit of beneficial national economic development while protecting the environment consistent with law, including the Endangered Species Act (ESA) of 1973. Although there is less concern about Corps contribution to cause before the problem of species loss was recognized in science and law, the knowledge that could be found about past causes could contribute information useful for actions.

With respect to the problem, water resources projects in general, and Corps projects in particular, have been placed among the leading causes for past species disappearance and present imperilment. In testimony before The Water Resources and Environment Subcommittee of the Transportation and Infrastructure Committee of the House of Representatives, the Senior Director of Water Resources at American Rivers stated:

> "The transformation of the nation's rivers brought about by Corps levees, dams and dredging projects are among the leading reasons that North America's freshwater species are disappearing five times faster than land-based species, and as quickly as rainforest species….Despite an explicit environmental protection mission, and specific environmental restoration programs and projects, the Corps' traditional flood control and navigation projects do not appear to be doing any better for the environment. To the contrary, agency-wide biases, institutional barriers, and faulty analyses are all contributing to the continued degradation of the nation's rivers and wetlands." (American Rivers testimony, April 10, 2002)

The validity of such assertions about the Corps and freshwater biodiversity deserve serious consideration if the Corps Civil Works program is to effectively follow its own environmental policy and strive to achieve environmental sustainability. The claim that freshwater loss rates of species are much greater than terrestrial species also implies that the Corps and other water resources agencies have been especially egregious contributors to the total species disappearance in the United States.

At least as important as correcting possible policy implementation deficiencies, this study was motivated by the potential to identify opportunities for Corps contribution to the improvement of the condition of freshwater biodiversity in aquatic ecosystems. Especially relevant to such improvements is the Corps ecosystem restoration mission.

REPORT PURPOSE

Before this review, the Corps had no assessment of Civil Works program impact on past species extinction rate, present species imperilment or projected extinction rate. While some of the assertions may seem exaggerated, the lack of program inventory of Corps impact on freshwater biodiversity leaves it open to criticism without sound defense and it impedes identification of opportunities for its ecosystem restoration mission. This review is intended to contribute initially to what could become a continuing process of environmental inventory for assessment of progress in achieving environmental sustainability for the Corps Civil Works Program.

The purpose of this analysis is to analyze assertions about rates of past and future loss of freshwater species through human-caused extinction in the United States, the role of water resources development in general and the role of the Corps of Engineers in particular. This study is consistent with the need for an inventory of water resources development and management impacts on environmental sustainability, specifically the viability of species in freshwater ecosystems. In keeping with that need, past and future rates of species extinction in the freshwaters of the United States were estimated and evaluated for probable cause using widely available data. The estimates are contrasted with data from Ricciardi and Rasmussen (1999) used to support claims that the Corps and other Federal water resources management agencies contribute to unsustainable conditions in freshwater ecosystems.

The claims were recast as hypotheses for examination with existing information that is reported in the results. They include:

- Freshwater species are disappearing at a significantly more rapid rate than land-based species in the United States and at rate similar to rainforest rates.
- Rates of freshwater extinction are increasing significantly faster than rates of terrestrial extinction.
- Federal water resources projects in general, and existing Corps flood control and navigation projects in particular, are among the leading causes of past freshwater species extinction, present freshwater species imperilment and, by implication, total species imperilment.

The objectives of this study include evaluation of these hypotheses using existing information and to address in discussion the most damaging claim made with respect to Corps policy; that there has been no improvement in the ways that the Corps plans, implements and operates its Civil Works. Perhaps more importantly, the objectives also include initiation of a regional inventory of ecosystem restoration opportunities for the Corps. It is not the intent of this paper to address all possible aspects of environmental sustainability associated with water resources development or to describe the extent to which freshwater ecosystems have been altered from their natural state by humans, except as it applies to the sustainability of freshwater species. Nor is there an intent to provide a conservation plan. These are important needs too and are being addressed separately by the Corps.

METHODS

ESTIMATING ANIMAL EXTINCTION RATES

Estimating past and future animal extinction rates is an imprecise process limited by the incomplete and often uncertain state of existing knowledge (May et al.1995, Ricciardi and Rasmussen 1999, Lomborg 2001). Despite methodological shortcomings, the serious implications of increased extinction rate (Barbier et al. 1995) warrant its estimation. All calculations of extinction rate require an estimate of the total number of extant and extinct species in the group considered and the total number estimated to become extinct during a specified period.

For this study, only total species extinction was considered in estimates of extinction rate because sub-specific classification is typically less complete in all taxonomic categories. The total number of subspecies is not well known in any taxonomic group, making estimation of extinction rates highly uncertain and too high if based on existing taxonomy. For example, numerous pacific salmon populations (demes) have been listed as threatened or endangered under the ESA (e.g., NMFS 2005), but the total number of distinct demes is unknown and large (NRC 1996). Even if the status of all salmon demes were known, there would be no way to determine a rate of loss at the deme level across all species.

For this study, the numbers of extant, extinct and imperiled species were estimated from the searchable database (NatureServe Explorer) maintained by NatureServe (2007)) as it was in June 2005. The geography of species extinction and imperilment identified in the NatureServe Explorer database is a primary source of information about where Corps and other water resources development projects have caused past impacts and are most likely to cause future impacts. In NatureServe Explorer, each species is classified according to conservation status at global (G), national (N) and subnational (S) scales. The status categories include presumed extinct (X), possibly extinct (H), critically imperiled (1), imperiled (2), vulnerable (3), generally secure (4) and secure (5). A G2 designation, for example, indicates imperilment at global scale. The status designation is based primarily on the rarity of species, as indicated by the number of viable populations, but is also influenced by population distribution throughout the species range, trends in the size and number of populations, threats, narrowness of environmental requirements, degree of habitat and other protection, and other factors. Global designation was used in this study because global extinction was the basis for comparing past and predicted future rates of biodiversity loss.

The NatureServe Explorer database is widely accepted as the most comprehensive record of species status for the taxonomic groups included in this study. It is updated regularly as new data become available. NatureServe is a non-profit conservation organization that serves a network of 50 state natural heritage programs and other programs outside the United States. Standardized methods are used to summarize data at regional, national and international scales. The data are compiled and maintained to provide information on the status, locations and levels of protection provided each species. Other studies of the status of imperiled species, including threats and corrective actions, were also consulted.

Although the approach is now standardized, the actual search effort and conditions surrounding species determination of status have varied over the period of record used to estimate trends to determine status. In general, vertebrates have received more consistent documentation of status than invertebrates. Only taxonomic groups considered to be “comprehensively covered” by NatureServe Explorer were included in the analysis. Extant species are categorized as critically imperiled; imperiled; vulnerable to extirpation or extinction; apparently secure; and demonstrably widespread, abundant, and secure. Extinction status is split into two categories: presumed extinct is more confident than possibly extinct.

The criteria used to separate freshwater from terrestrial and marine species is the source of oxygen for respiration and the salinity of the water occupied at some time in their life cycle. Only those species that respired dissolved oxygen from water at some time in their life after birth were classified as aquatic. Only those aquatic species that require freshwater to reproduce (generally less than 2 g/liter of salt) were considered freshwater species. Marine mammals, birds, and reptiles were included among the terrestrial species, consistent with their dependency on atmospheric oxygen throughout their life cycle.

Only a few taxonomic groups of the many that exist are documented well enough to consider in a quantitative analysis. Ricciardi and Rasmussen (1999) limited their study to terrestrial (including marine mammals addressed separately) and freshwater vertebrates, freshwater snails and mussels, and crayfish species. They included amphibians under a freshwater designation. These same groups were included in this study, with the exception that amphibians were split into terrestrial and freshwater species. Terrestrial snails were added to this study to provide more invertebrate and vertebrate balance in both terrestrial and freshwater categories (these data were not available to Ricciardi and Rasmussen (1999). The aquatic and terrestrial insects comprehensively covered by NatureServe Explorer were also considered in an additional analysis of the effect of adding new taxonomic groups to the estimate of past and future extinction rates and ratios. These included among the freshwater groups all of the members of the dragonfly, mayfly and stonefly orders, and, for the terrestrial groups, the butterflies, skippers, and giant silkworm, royal, notodontid, underwing and *Papaiperma* moths.

A one-century period from 1889 through 1988 was selected to estimate and compare extinction rates. This was the best-documented century over the period of record and had the highest total number of estimated extinctions of species native to the United States in any 100-year period since 1825. It leaves out seven species last observed before 1889, including six birds and one crayfish species last observed between 1825 and 1884 (Stein et al. 2000).

Prehistoric Background Rates of Animal Extinction

Relevant to the issues investigated, fundamental questions are whether or not background rates are actually exceeded by recent extinction rates and by how much. May et al. (1995) provide a good summary of the methodology used to estimate background levels of extinction. The concept is simple. The data are extracted from the fossil record and include estimates of the average length of time species persisted over defined periods of time and the total number of recorded species. From that data, a species extinction rate is estimated for whichever rate measure is desired, such as number per century.

While simple in concept, there are numerous complications. The fossil record is "very incomplete" for many taxonomic groups (Jablonski 1995) and not dependable below the family level. Species extinctions are estimated from family-level extinctions based on the estimated average number of species per family now living. The average time that species persist in the record is estimated for each of the major taxonomic groups (e.g., insects, birds) and used to weight the estimate of average species persistence time. Several episodes of rapid extinction have occurred over the past several hundred million years of fossil record (Jablonksi 1995). Whether or not to include those episodes makes a difference of less than 10%, however.

Estimates of average length of species existence range from 1 million years for mammals to 11 million years for invertebrates (May et al. 1995). There is no clear indication of consistent trends in this average through geological time so the average is assumed to apply to present conditions. Because invertebrates in general, and insects in particular, have dominated the total number of animal species, the average time of animal species existence is typically estimated to be closer to 10 million years than to 1 million years. May et al. (1995) estimated that the average animal species persists between 5 and 10 million years before becoming extinct.

Dividing the estimated average species persistence time into the fossil record results in an estimate of about 100 species per century naturally lost worldwide to extinction before humans influenced the tally (May et al. 1995). Divided by the estimated 5 to 10 million species now in existence worldwide, the resulting estimate of background extinction rate is 0.001 to 0.002% per century. This estimate might easily be larger or smaller by a factor of at least two according to May et al. (1995). Birds and mammals have undergone background extinction rates roughly 10 times higher. For the United States that is about 0.1 to 0.2 birds and 0.05 to 0.1 mammals per century.

To account for the uncertainty, estimates of current extinction rates need to exceed the estimated background rate by 10 times or more to confidently indicate a higher rate. In the United States, they would have to exceed one to two birds per century and one to two mammals every two centuries. Because the background rate of extinction is lower for most invertebrates, even an extinction rate of 0.1 to 0.2 mollusks per century could be a significant rise in extinction rate.

Recent Animal Extinction Rates

Extinction trends can be analyzed from records of last observation where records are reasonably accurate and complete. Historic evidence of recent extinctions is found primarily in the last dates recorded for a species in museum records and field notes. Past records have rarely been assembled under consistently ideal conditions, however. Much of the data from which determinations of extinction are based have been gathered opportunistically rather than systematically (May et al. 1995), and the biases that may have been introduced are not always well understood. The trends that emerge need to be interpreted with these shortcomings in mind, including the possibility that the number of unidentified species that have been lost to extinction has, if anything, decreased as a more complete inventory of extant species has been assembled.

Determining when a species becomes extinct remains a matter of professional judgment; at best approximate and never entirely certain. The most recent case in point is the controversial discovery of a living ivory-billed woodpecker (*Campephilus principalis*), which was considered by many conservation biologists to be extinct after 60 years of unconfirmed reports (Fitzpatrick et al. 2005). Primary considerations for concluding a species is extinct include the completeness of knowledge about the range and natural history of the species, the intensity of search in the known range, the certainty of documentary evidence and the time lapse without a confirmed encounter.

The frequency of assessment and documentation rates for species occurrence has varied among taxonomic groups and geographic areas. Study intensity in the past depended on the social importance attached to taxonomic groups (e.g., commodity and recreational use) and the number and distribution of taxonomic specialists (May et al. 1995). Easily recognized, highly visible species are more certainly extinct after a long period without observation than more cryptic species of little public interest. It follows that the confidence in the conservation status of a species tends to increase with species size.

The certainty of extinction-rate estimates is greatly increased when estimates are limited to taxonomic groups and geographic areas that have had the most intense study. In general, the confidence expressed in the data used to confirm extinction is greatest for mammals, birds and reptiles, and decreases through amphibians and fish to invertebrates. Among the invertebrates, more confidence exists for freshwater mollusks (clams, mussels, snails) and large crustaceans, such as crayfish, than for other taxonomic groups. Of course, the extinction history of these groups may not be indicative of extinction rates in general.

The geography included in the analysis can significantly influence the results. Ricciardi and Rasmussen (1999) analyzed North American extinction rates consistent with reports available to them, which may not be altogether relevant to issues discussed here. Because of the interest in the activities of the United States Government in this study, a North American analysis is less relevant than one limited to the 50 states in which Federal water resources development has occurred. For those summaries that include only Canada and the United States in North America, the difference from the United States alone is small because few species are uniquely Canadian. However, Miller et al. (1989) included Mexico in their analysis of fish extinctions, which counts substantially toward a total North American count. A political part of the United States, Hawaii is not physiographically part of the North American continent and there are few freshwater species to include in the analysis. However, Hawaii contributes significantly to the extinction record for terrestrial species. How these issues were addressed by Ricciardi and Rasmussen (1999) is not always clear, but may have contributed to some of the differences in results observed between our two studies.

The data considered were limited to the 50 United States for this analysis. It does not include protectorates, territories and other lands without state status. The presumption of extinction is reasonably confident for species of the United States in the aggregate although a few species may be rediscovered based on recent history. Vertebrate species number and extinctions are best documented. Mollusks and large crustaceans are not as well documented, but add a taxonomic dimension that is likely to more completely reveal the degree of influence that water resources development and other ecological stress has had on species extinction rate.

The NatureServe Explorer database that provided the conservation status data for this study was not available to Ricciardi and Rasmussen (1999), who were limited to original literature. These included, especially, the summaries in Miller et al. (1989), Williams et al. (1989), Williams et al. (1993), Taylor et al. (1996), Neves et al. (1997), and Turgeon et al. (1998). The NatureServe Explorer database is continuously being updated so some differences were expected from that source alone (NatureServe 2007).

Future Animal Extinction Rates

Estimates Based on Habitat Loss

Most global and rainforest estimates of future extinction rate are based on modeled relationships between species number and geographical area anticipated to undergo habitat loss (Shafer 1990, May et al. 1995). Because most of the world's diversity is in the undeveloped tropical forests, predicted global rates of extinction are based on predicted rates of habitat conversion. This method derives conceptually from the theories of McArthur and Wilson (1967) based on studies of species on oceanic islands. They found that the number of species resulting from the interaction between new species colonization and established species extinction increased with island size for islands a similar distance from the mainland (the species equilibrium theory). Log-log plots of species (S) verses geographical area (A) often result in a straight line, indicating the relationship: $S = cA^z$. A broadly approximate "rule" observed from actual data sets indicates that a 90% reduction in habitat area halves the number of extant native species.

This concept has been transferred from oceanic islands to other islands of habitat, such as isolated habitats in ponds (Hubbard 1973) and at the tops of high mountains (Brown 1978). Simberloff (1986), Wilson (1989) and others (see, for example, the reviews by Whitmore and Sayer 1992 and Rosenzweig 1995) added to the theory and extended the concept of species-area relationship to predict the extinction rate of mainland terrestrial fauna as habitat shrank in size. They assumed that species in unique rainforest and other habitats behaved as islands in a matrix of other habitats.

This method is the one of choice when: 1) most of the geographical area under consideration is in its natural state and has yet to be affected by human use, 2) major threats to vulnerable species operate through habitat alteration, 3) variations in models developed under various conditions are appropriately applied to the conditions under study, and 4) the species chosen for the analysis are generally representative of all species. This method does not work well for landscapes that have already undergone most of the anticipated habitat conversion to agriculture, urban use, water resources development, or other change or alteration—such as most landscapes in the U.S. Therefore, comparisons to rainforest estimates of results from this study, and the study of Ricciardi and Rasmussen (1999), are based in different estimation techniques, which could lend to differences in observed results. The method has gained the attention of many scientists. Mann and Plummer (1995) reviewed some of the major uncertainties associated with its application.

Estimates Based on Species Status Change

A common approach to evaluating future extinction rates is to use their present conservation status, or change in conservation status, to indicate future extinction. Smith et al. (1993) examined changes in species status on the IUCN (World Conservation Union) “red list” of threatened species, determining from the list which of the species met strict criteria for extinction. They also ranked rates of change in species placed in status categories based on relative vulnerability to extinction, including unthreatened, vulnerable, endangered, probably extinct and extinct categories. The median time to extinction is estimated from the number of status changes needed to become extinct for the entire group studied divided by the median change in status for each species each year.

A weakness of the approach used by Smith et al. (1993) is the non-systematic, opportunistic way in which species have been added to the IUCN red list (May et al. 1995) and categorized according to relative vulnerability. Mace (1995) describes a variation of this approach and the pitfalls associated with the use of red lists, including ESA listed species. An important unknown is the extent that a threatened species, once identified as such, is made less vulnerable to extinction because of protection.

A variation of this method was used in this study to indicate possible future extinction rates, similar to the approach of Ricciardi and Rasmussen (1999), who used imperilment status summarized in published “red lists” (e.g., Miller et al. 1989, Taylor et al. 1996) to estimate probable extinction rates and a ratio for freshwater and terrestrial species during the next century. Two sources of data were used in this study to evaluate future vulnerability of freshwater and terrestrial species to extinction: 1) species imperilment data stored in NatureServe Explorer, including both imperiled (G2) and highly imperiled (G1) species, and 2) ESA listings of threatened and endangered species. Indicated future extinction rates were estimated, like past extinction rates, by dividing the number of imperiled or threatened and endangered species by the total number of extant and extinct species. In the analysis of the ESA list, listed species with only partial range protection were included. No attempt was made to estimate rates of change in species status in either database.

Estimates Based on Trend Analysis

Past trends in the rates of last recording of species were analyzed in this study to estimate the extent present and future extinction rates may be increasing. This method is relatively new and consistent with recent improvements in information availability. The information comes from museum and other scientific documentation for those species judged to be extinct (GX) and possibly extinct (GH), as summarized in the NatureServe Explorer database. The dates were pooled by decade because dates were sometimes bracketed in a range (e.g., 1940s). This practice also is more consistent with the variation in species persistence that is likely to follow the last observation.

This method has several advantages over others. It requires no assumption about relationships between extinction and cause (e.g., habitat conversion). It documents the patterns of presumed extinction events. And, it has potential for use in analyzing relationships with suspected causal events, stressor contributions to extinction, and effectiveness of past policy and management. Its primary weakness lies in the substantial uncertainty surrounding the date of last observation as an indicator of extinction time. Common to other methods, factors influencing past events are likely to change in uncertain ways, including the attention paid to management. The probability of trend-shifting change increases with extrapolation of the trend into the future. Additional uncertainty accompanies the assumption that the taxonomic groups included in the analysis are representative of those groups that are not included. The species included in this study comprise a small fraction of the total number of species documented for the United States.

The continuity of tracking conservation status is much more complete for some species than for others. Not all of the last observation dates for species listed as presumed and possibly extinct were identified or dated accurately enough to include in the forecast analysis. To estimate the future number of extinctions, the total number of estimated extinctions based on the extrapolation of dated last observations was divided by the fraction of all presumed and possibly extinct species (an average of 0.41 for all species). This divisor varied widely among the taxonomic groups included in the study.

The fraction dated and reliability of dating to decade of last observation are greater for large, high profile, easily identified and widely known species than for smaller, more cryptic and less studied species. In general, vertebrate status was much more closely tracked than invertebrate status. The last dates of observation are identified for all land and freshwater vertebrates. For invertebrates, 56% of the terrestrial snails, 50% of the mussels, 50% of the crayfish and 20% of the freshwater snails were assignable to decade of last observation. Because they comprise 61% of the presumed or possibly extinct freshwater species, the freshwater snails disproportionately contribute to the uncertainty of trend in freshwater extinction.

EVALUATING CAUSES OF ANIMAL SPECIES DISAPPEARANCE

In this study, evaluation of cause for species decline are based in part on summaries of knowledge; especially in Matthews (1990,1992,1994), Miller et al. (1989), Federal Register statements and the NatureServe Explorer database. These summaries often offer little more than lists of factors that could have contributed to decline given proximity of potentially stressful events to species habitats and typically do not quantify relative contribution of impact to the decline. They are important, however, for indicating possible causes of species and subspecies extinction and imperilment/endangerment. The information needed to analyze for the contribution of environmental stressors to past species decline and extinction include species environmental requirements, tolerances and stressors. A large quantity of relevant information has been published and much of it has been compiled in various recent treatments including Smith (2001), Thorp and Covich (2001), and Diana (2003). The intent here is to distill the main themes of this literature from such reviews as it applies to the causes of extinction and imperilment.

Many declines of freshwater species toward extinction have occurred with little notice and without careful evaluation for cause. In general, vertebrate declines are better documented and recognized earlier than invertebrate declines. The scientific record for a number of species now regarded as extinct amounts to little more than a few museum specimens and minimal location data collected many decades ago. Little is known of the original population status of some vertebrate species and many of the invertebrate species. More recently, direct evidence for cause of decline was most frequently gathered late in the decline when it was difficult to quantify contributions to decline experimentally.

In most assessments, much is inferred about the causes of species decline from general knowledge about species life histories and stressors and recent distribution data. More typical than not, imperiled populations are exposed to a variety of stresses that are difficult to sort using distribution data. Only rarely is a single cause clearly discerned. Frequently, the documented sensitivities of physiologically similar, extant species are the primary information available about the stresses that may have contributed to the decline of species now considered extinct. In a few instances, knowledge about a well-documented species can be extended to species known to have similar environmental sensitivities.

In addition, a historical analysis of environmental change was done (Wissmarr 1997). Chronological and geographic records of environmental changes and species decline were required to reveal coincidence, if any, with stressful changes in home ecosystems. Insight into cause of extinction can sometimes be developed from a historical analysis of environmental change and the pattern of species decline leading up to recent conditions. A rich, if often general, history exists about the transformation of the American landscape by agriculture, urbanization, and transportation developments, but less is known about the early history of change in freshwater habitats. Much can be deduced about changes in aquatic habitat, however, by integrating what is known from the history of landscape change with scientific understanding of the relationships between landscape condition and freshwater habitats. One of the better regional treatments using this approach was developed to explain changes in fish habitats and fish abundance by Trautman (1981).

Completely sorting out causes of decline through the sequence of changes in stressors and species abundance is typically thwarted by incomplete knowledge, especially for invertebrate species. Some potential causes for decline can be quickly eliminated based on the location and timing of past extinctions indicated by date of last observation and other information about species abundance. For example, extinctions in small isolated waters are rarely associated with Federal water resources developments, which occurred mostly in large river settings. More probable causes in isolated settings are associated with local development of water supply and purposeful introduction of nonnative species.

EVALUATING THE CORPS' RECENT ROLE

Corps Civil Works history and policy was reviewed as it pertains to the loss, protection, and recovery of freshwater biodiversity. Present policy documentation is available at the Corps home page Internet address (USACE 2007). Historic information pertaining to policy and planning process and to project implementation are maintained at the Institute for Water

Resources, USACE and at Internet addresses of individual Civil Works districts. Changes in the status of ESA listed species and Corps reports about ESA involvement were the primary sources of information used to judge Corps effort expended toward and contribution toward effectiveness in improving the status of threatened and endangered species.

Evidence gathered to judge Corps action came from summary reports of Corps activities at operating projects, involvement in species recovery plans and ecosystem restoration projects and Corps investments in research directed at vulnerable species. Other information is associated with rates of species status change, as indicated in ESA and other assessments. Effectiveness was evaluated by stabilization of vulnerable populations and by recovery of vulnerable populations to improved status. A preliminary analysis was conducted for this report based on readily available information**.**

RESULTS

RATES OF ANIMAL SPECIES EXTINCTION IN THE UNITED STATES

Past Extinction Rates

Comparison to Background Rates

Results from this analysis support the conclusion that recent extinction rates substantially exceed background rates for both freshwater and terrestrial species included in the study and that the freshwater extinction rate was substantially higher than the terrestrial extinction rate. Table 1 summarizes extinction rate estimates for the century spanning 1889 to 1988. A total of 59 freshwater amphibians, fish, mussels and snails were presumed extinct (the most certain designation of extinction in the NatureServe Explorer) for some time during that period. This translates to a rate of 2.5% per century for all of the freshwater species included in the analysis, which is about 3 orders of magnitude greater than the background rate of extinction estimated from information provided by May et al. (1995). Including possibly extinct species, as Ricciardi and Rasmussen (1999) did, increases the recent estimate of freshwater extinction to 138 species and more than doubles the estimated extinction rate of the freshwater species included in this study.

During that same century, 30 (1.14%) of the terrestrial mammals, birds, reptiles, amphibians and snails native to the United States are presumed to have become extinct. Similar to freshwater species, the estimated recent extinction rate more than doubles when possibly extinct species are included (71 species). Over 80% of that loss is contributed by land snails. The loss rate is less than for freshwater species, yet more than 100 times greater than the estimated background rate of extinction. Including the possibly extinct species with the presumed extinct species reduced the ratio of freshwater to terrestrial extinction from 2.7 to 1 to 1.9 to 1.

Comparison of Taxonomic Groups

The ratio of freshwater to terrestrial extinction for the taxonomic groupings most like that of Ricciardi and Rasmussen (1999) was 3.1, which is less than the 5.0 they estimated but substantially greater than unity. While less dramatic, their main point held up in this analysis for the data set they used. Adding land snails to the analysis as done in this study was a significant departure from their data set. At the time of this report, land snails made up nearly two thirds of the presumed and possibly extinct terrestrial species (Table 1). The presumed extinction of terrestrial species was split evenly between birds and land snails, but nearly six times as many snails are less confidently assigned a possibly extinct status. Only one mammal was presumed to have become extinct and another possibly became extinct during the century examined. One terrestrial amphibian possibly became extinct. No reptiles are believed to have become extinct in the United States over the past two hundred years. Most of the early high-profile extinctions on the continent—such as the passenger pigeon (*Ectopistes migratorius*) and Carolina parakeet (*Conuropsis carolinensis*)—were clearly connected with over-hunting.

Table 1. The estimated total number of species, species presumed extinct or possibly extinct, and the extinction rate over the century spanning 1889-1988 for native vertebrates and invertebrate groups in freshwater and terrestrial habitats.

Taxonomic Group	Total Species	Presumed Extinct Species		Possibly Extinct Species		Extinction Rate[2]	
		Number	Percent	Number	Percent	Presumed	Presumed &Possible
Mammals	423	1	3.3	1	1.4	0.0024	0.0047
Birds	784	14	46.7	10	14.1	0.0179	0.0306
Reptiles	240	0	0	0	0	0.0000	0.0000
Amphibians-terrestrial[1]	78	0	0	1	1.4	0.0000	0.0128
Snails-terrestrial	1854	15	50.0	59	83.1	0.0081	0.0399
Total Terrestrial	**3,379**	**30**	**100.0**	**71**	**100.0**	**0.0089**	**0.0299**
Amphibians-aquatic	193	1	1.7	0	0	0.0052	0.0052
Fish	799	16	27.1	4	5.1	0.0200	0.0250
Freshwater mussels	307	19	32.2	16	20.3	0.0619	0.1140
Snails-freshwater	744	23	39.0	58	73.3	0.0309	0.1089
Crayfish	345	0	0	1	1.3	0.0000	0.0029
Total Freshwater	**2,388**	**59**	**100.0**	**79**	**100.0**	**0.0247**	**0.0578**

1. In listing aquatic amphibians, the following salamanders were excluded for not reproducing in water: all species in the genera *Aneides, Batrachoseps, Ensatina, Hydromantes, Phaeognathus, Plethodon* and two species of *Desmognathus*.
2. Rate of extinction is the estimated number of species lost over 100 years (1889-1988) divided by the estimated total number of extinct and extant species in the taxonomic group.

Among freshwater species, fish, snails and freshwater mussels share most of the presumed extinctions from 1889 through 1988. Adding possibly extinct species more than doubles the contribution of mussels and snails to the list, which together comprise 93.6% of the possibly extinct freshwater species and 84.1% of all presumed and possible extinctions of freshwater species. Possibly extinct snails out-number possibly extinct fish by over 14 to 1. Crayfish and aquatic amphibians underwent relatively low rates of extinction during the century studied.

As a measure of extinction uncertainty, the ratio of possible to presumed extinctions indicates that terrestrial species, at 2.37 to 1, are more uncertainly classified as extinct than freshwater species (1.34 to1). Much of this uncertainty is associated with snail extinction status. The certainty of extinction, as indicated by the percent presumed extinct, is relatively high for freshwater vertebrates (81.0%); intermediate for birds (58.3), mussels (54.3%) and mammals (50.0%); and least certain for land and freshwater snails (20.3 to 28.4%). Over one-third of the terrestrial snails have not been assigned a rank and some could be extinct. Thus the estimated terrestrial extinction rate with snails included is, if anything, an underestimate indicating that the freshwater to terrestrial extinction ratio is more likely to be overestimated than underestimated.

Birds are among the better studied and protected of all groups, yet substantial change has occurred in their extinction status since Stein et al. (2000) summarized NatureServe data in the late 1990s. Since then, one bird species has been rediscovered and two species have been reclassified from presumed to possibly extinct. On the other hand, six imperiled bird species were added to the list of possibly extinct species.

Effects of Taxonomic Groups and Conservation Status

Another measure of uncertainty is the variation that exists in extinction rate among the groups included in the analysis. The included groups make up less than 4% of all animal species in the United States, the vast majority of which are invertebrate. Variation in extinction among the groups in the sample set is a crude measure of the variation that might be expected among all of the animal species present. Mollusks dominate the extinction that has occurred in the sample set. Mollusk extinction amounts to 84% of the total presumed and possibly freshwater extinction and 73% of the terrestrial extinction estimated for all groups in the study. Fish and birds make up most of the remainder. The relatively low extinction rate of insects, indicated in Table 2, as well as the low crayfish extinction rate (Table 1), suggests that mollusk extinction rate is exceptionally high and not a reliable indicator of other invertebrate extinction rates.

As Table 1 shows, the range in presumed and possible extinction percentage across taxonomic groups is substantial, from over 11% for freshwater mussels to 0 for reptiles. The mussels are a small taxonomic group composed of a single, somewhat consistently specialized family, compared to the reptiles, which, while fewer in number in the United States, are more taxonomically and ecologically diverse. However, if specialization within taxonomic groups is a factor in determining extinction vulnerability, it does not seem to predominate across all groups. The aquatic amphibians and crayfish have sustained relatively little extinction and are not much more diverse at the family level than the snails, which are going extinct at nearly the same rate as the mussels (and probably more so given the high numbers of uncertain status).

Table 2 shows the freshwater to terrestrial extinction ratios for three taxonomic groups that have members in both freshwater and terrestrial habitats. Of the three groups, the mollusks had the highest ratio. The ratio for vertebrates and insects is too low to conclude that a difference exists between terrestrial and freshwater extinction rates in those taxonomic groups. Whether the freshwater mollusks are calculated to be at greater risk of extinction depends greatly on the ultimate conservation status assigned to presently unranked terrestrial snails. Taken separately or all together, the groups have a lower estimated extinction ratio than the 5:1 ratio estimated by Ricciardi and Rasmussen (1999) from earlier data summaries. In general, the data do not consistently support the conclusion that extinction responses to stresses in freshwater ecosystems over all 50 states exceed those in terrestrial ecosystems. They also indicate great effect on the estimated ratio by the mollusks, which have many species and a high incidence of extinction. Some uncertainty derives from the assignment of extinction status to species. Whether or not both presumed and possibly extinct species are included has a measurable effect on the calculated ratio of freshwater to terrestrial extinction rate. Ricciardi and Rasmussen (1999) included both categories of extinction. This study used the same taxonomic groups and extinction designations as they and calculated the ratio of freshwater extinction to terrestrial extinction to be about 3 to1. The ratio changes only to 2.3 to1 when the analysis is limited to the more certainly presumed extinctions. While less than a ratio of 5 to1, this estimated freshwater extinction rate remains greater than the terrestrial rate, regardless of the certainty of extinction status and generally consistent with the conclusions of Ricciardi and Rasmussen (1999).

Table 2. For the United States, the percentages of presumed and possibly extinct species and the freshwater to terrestrial extinction ratio for the three major taxonomic groups that include both freshwater and terrestrial species.

Taxonomic Group	Total Species Terrestrial	Total Species Freshwater	Percent Extinct Terrestrial	Percent Extinct Freshwater	FW: T Extinction Ratio
Vertebrates	1,525	992	1.77	2.12	1.2:1
Mollusks[1]	1,854	1,051	3.99	10.75	2.7:1
Insects[2]	1,594	1,598	1.81	2.01	1.1:1
All Groups	4,973	3,641	2.67	4.59	1.7:1

1. Includes the snails and mussels listed in NatureServe Explorer for the 50 United States.
2. Only groups considered comprehensively covered by NatureServe Explorer were included. These are grasshoppers, butterflies, skippers and giant silkworm, royal, sphinx, notodontid, underwing and *Papaipema* moths among terrestrial insects, and stoneflies, mayflies and dragonflies among the freshwater insects

However, uncertainties in taxonomic and conservation status suggest that these values may not be large enough to be certain that the ratio is larger than unity. The estimate is expected to change, even in the near future, as it has since Ricciardi and Rasmussen (1999) did their study, because of status reconsiderations and increased knowledge of other taxonomic groups. This may explain some of the difference between my results and theirs. Reclassifications of conservation status are common in NatureServe Explorer. In the few years between the analyses of Stein et al. (2000) and the 2005 version of NatureServe Explorer upon which this study was based, one mammal had been added to the possibly extinct category, doubling the estimated extinction rate. Bird listings increased from 20 to 24 species. Numerous snail species were moved from possibly extinct to presumed extinct classification and five other snail species once believed to be extinct were discovered and reclassified among the imperiled. Among the vertebrates believed extinct, one fish species and one bird species have been discovered alive.

Confidence in the extinction ratio also is affected by the uncertainty in taxonomic status of some freshwater species, which is most clearly described for the fish. At least five extinct species are of somewhat questionable taxonomic status. The phantom shiner was once included in *Notropis simus*, with which it is known to have hybridized. It could be an ecophenotype of *N. simus* (Sublette et al. 1990). The Clear Lake splittail, *Pogonichthys ciscoides*, could be a lake-inhabiting variation of a more widely distributed species (*Pogonichthys macrolepidotus*) (Lee et al. 1980). The species classification of the Snake River sucker, *Chasmistes muriei*, is based on a single specimen, which Lee et al. (1980) included with the June sucker, *Chasmistes liorus* (NatureServe Explorer does not question the taxonomy). Harris and Mayden (2001) concluded that sucker classification needed revision. The blackfin cisco, *Coregonus nigripinnis*, is quite likely not a distinct species. Less likely, the deepwater cisco, *Coregonus johannae,* may also be included with an extant cisco (Lee et al. 1980). There appear to be no sub-specific fish extinctions that are suspected of having full species status.

Several mollusk groups are undergoing closer taxonomic scrutiny to refine their taxonomic status. While this may either enlarge or contract the number of extinct species, progress so far suggests that fewer species extinctions will result. Taxonomic uncertainties appear to be less of an issue among the terrestrial vertebrates than among fish and mollusks, but do occur among a few Hawaiian birds last observed in the 19th century.

In summary, if only the most confidently determined species extinctions were included in the analysis, the ratio of freshwater to terrestrial vertebrate extinctions for all 50 states would decrease to less than 1.0 (0.9). Clearly, the degree to which freshwater and terrestrial extinction rates differ for the United States depends on the combination of taxonomic and conservation status categories included in the evaluation. If not greater, past rates of freshwater extinction are at least as high as for terrestrial extinction in the United States and are substantially greater than background rates for both land and aquatic species.

Geographical Effect of Hawaii

The Corps does no significant work in the freshwaters of Hawaii. Excluding Hawaii from the analysis has a profound effect on the ratio of freshwater to terrestrial extinction and on conclusions about relative rates of continental extinction and the possible role of water resources development. The geographic distribution of terrestrial extinctions in the United States is greatly influenced by the fate of Hawaii's unique fauna. Among terrestrial taxonomic groups included in this study, as of summer 2005, 133 birds, 2 mammals, 6 reptiles, and 775 terrestrial snails occurred only in Hawaii. From 1889 to 1988, 9.5% of those species have become listed as extinct (including both presumed and possibly extinct species). Hawaii's rate of extinction is more than three times the rate in the entire United States and approaches twice the extinction rate of freshwater species in the United States.

About 88% of the recently extinct terrestrial vertebrates and snails in the United States are birds and snails that once lived in Hawaii. Twenty of 24 birds and 20 of 27 terrestrial vertebrates listed as becoming extinct from 1889 through1988 are Hawaiian natives. Sixty-seven of 74 terrestrial snails (90.5%) listed as extinct also lived in Hawaii. Of all terrestrial extinctions during the study period, 85.2% occurred in Hawaii.

In contrast, few native freshwater species are found in Hawaii (no amphibians, five fish, nine snails, no mussels, no crayfish, and 30 freshwater insects in the dragonfly, mayfly and stonefly orders) and none are presumed extinct. However, two freshwater snail species and four dragonfly species were recently listed among the possibly extinct. They raise the fraction of freshwater extinctions in Hawaii to 13.6%, and raise the ratio of freshwater to terrestrial extinction in Hawaii to about 1.4 to1 for the groups included in this study.

The extinction ratio for freshwater and terrestrial species in the continental United States (without Hawaii) is 11.8 to1 for species presumed extinct among the vertebrates, mollusks and crayfish. Adding the possibly extinct species reduces the estimate to 9.1 to 1. Including the insect groups confidently ranked in NatureServe Explorer increases the ratio to 20 to1. Excluding all but the vertebrates and crayfish decreases the ratio to a low estimate of about 2.8 to1. While the effects of including or excluding taxonomic groups are strong, the evidence indicates that the recent rate of continental freshwater extinction is consistently and significantly greater than the continental terrestrial extinction rate and is greatest when all groups for which there is reliable data are included in the analysis. This confirms the general concerns raised by Ricciardi and Rasmussen (1999) for the freshwater species inhabiting continental freshwaters of the United States.

Comparison to World Rates of Extinction

Estimated extinction rates in the United States are high compared to estimates for the world over the past 500 years. Table 3 contrasts estimates of extinction number and rate for the United States and the world based on data maintained by the IUCN (Baillie et al. 2004). The extinction rate estimated for species categories included in this analysis was twice as high in the United States as for the world (including the United States). Birds, amphibians, fish, crustaceans and mollusks had higher extinction rates in the United States than in the entire world.

The great apparent difference between freshwater and terrestrial extinction rates in the world and the United States is the result of poor documentation of freshwater extinction rate in most of the world (Baillie et al. 2004). Just as in the United States, the terrestrial vertebrates are much better known worldwide than the aquatic vertebrates and invertebrate extinctions are more poorly documented. This difference in documentation does not diminish the importance of freshwater extinction in the United States so much as it emphasizes the probability that world freshwater extinction rates have been higher than estimated (Baillie et al. 2004).

Table 3. Estimated species extinction number and rate since the year 1500 for the United States and the world in major taxonomic groups. The data are from Baillie et al. (2004) and NatureServe Explorer (2005).

Species Group	United States (NatureServe)		World (Baillie et al. 2004)	
	Number Extinct	% of the group	Number Extinct	% of the group
Mammals	2	0.47	73	1.6
Birds	29	3.70	129	1.3
Reptiles	0	0	21	0.3
Amphibians	2	0.74	34	0.7
FW Fish	20	2.50	81	0.9
Crustaceans	3	0.03	7	0.02
Mollusks	190	6.54	291	0.6
Total	246	0.43	636	0.2

For the world data, Baillie et al. (2004) are most confident for mammal, bird and amphibian extinction estimates. Assuming those data are reasonable, about 14% of the world's mammal, bird and amphibian extinctions have occurred in the United States at about twice the world rate of loss (2.2% vs. 1.2%). Most of the difference is associated with bird extinctions. Once again, an important variable is Hawaiian extinction. Continental extinction of birds, mammals and reptiles is substantially below the world rate because such a high fraction of bird extinction occurred on Hawaii. The Hawaiian losses are not unique for oceanic islands. Past terrestrial extinctions world-wide have most commonly occurred there (Baillie et al. 2004).

Existing evidence indicates that past extinctions of terrestrial vertebrates documented for the United States exceed past extinctions for tropical rainforests, which make up a fraction of the extinctions documented for the rest of the world. The history of freshwater extinction is poorly documented in much of the world, however. Making a point about how high freshwater extinction rates are in North America, Ricciardi and Rasmussen (1999) compared them to extinction rates of terrestrial species in rainforests where predicted extinction rates are high (e.g., Simberloff 1986, Wilson 1988, Reid 1992) based on numbers of species per unit area and rates of landscape conversion from the wild state. Those rates vary substantially, however. The rates of freshwater extinction estimated in this study for the continental United States were similar to some of the lower estimates of rainforest rate.

Future Extinction Rates

NatureServe Species Imperilment Status

Species ranked in imperiled categories of conservation status are assumed to be the least secure and the most likely to become extinct by the end of this century if conditions do not improve. Nearly half (45.1%) of all freshwater species in the groups studied are listed as imperiled or critically imperiled (Table 4). The imperilment fraction of freshwater invertebrates is over twice that of freshwater vertebrates. Nearly two thirds of the freshwater snails are imperiled. The crayfish, which had lost only one species to possible extinction, have had a third of their species ranked as imperiled based on the criteria described by NatureServe.

In contrast, less than 7% of the mammals, birds and reptiles are imperiled (about 5% of the continental species). Nearly half of the terrestrial amphibians and over one-third of the terrestrial snails are imperiled. Because the snails make up a large fraction of all terrestrial species in the study, they are the dominant contributor to the 25% that are considered imperiled. Without snails, only 9% would be listed as imperiled.

Based on present imperilment status alone, the freshwater species included among the vertebrates, mollusks and crayfish of the United States are 1.6 times more vulnerable to extinction than the terrestrial species (Table 4). This ratio is slightly less than the ratio determined for past extinctions in these same groups (1.9), indicating stability or even some decline in the extinction ratio for freshwater and terrestrial species. In contrast, Ricciardi and Rasmussen (1999) predicted an increase in the relative vulnerability of freshwater species based on a 60% increase in the extinction ratio forecast from the imperilment ratio for freshwater and terrestrial species. An important reason for the difference from this study is the high imperilment fraction for terrestrial snails, which were not included in the Ricciardi and Rasmussen analysis and comprised a large fraction in this study.

Table 5 includes a summary of the imperiled fractions of species and estimated future extinction ratios for freshwater and terrestrial species in three major taxonomic groups with both freshwater and terrestrial representation. Imperilment of freshwater vertebrates is much greater than imperilment of terrestrial vertebrates, which is a relatively low value Ratios for mollusks,

Table 4. Estimated total number of extant species, species that are critically imperiled and imperiled, and future vulnerability to extinction in the United States based on the fraction listed as imperiled and critically imperiled in NatureServe Explorer.

Taxonomic Group	Extant Species	Critically Imperiled[2]	Imperiled[3]	Total Imperiled	Extinction Vulnerability[4]
Mammals	421	13	16	29	0.0689
Birds	764	27	21	48	0.0628
Reptiles	240	6	14	20	0.0833
Amphibians-terrestrial[1]	77	12	22	34	0.4416
Snails- terrestrial	1,780	448	236	684	0.3843
Total Terrestrial	**3,282**	**506**	**309**	**815**	**0.2483**
Amphibians-aquatic[1]	192	23	16	39	0.2031
Fish	779	112	76	188	0.2413
Freshwater mussels	304	82	45	127	0.4178
Snails-freshwater	663	333	90	423	0.6380
Crayfish	344	61	55	116	0.3372
Total Freshwater	**2,282**	**611**	**282**	**893**	**0.3913**

1. In listing aquatic and terrestrial amphibians, the following salamanders were considered terrestrial based on the absence of an aquatic life stage: *Aneides, Batrachoseps, Ensatina, Hydromantes, Phaeognathus, Plethodon* and two species of *Desmognathus*.
2. Equivalent to G1 status in the NatureServe Explorer; includes G1-G2 rank.
3. Equivalent of G2 status in NatureServe Explorer; includes G2-G3 rank.
4. Vulnerability to future extinction is the fraction that the sum of critically imperiled (G1) and imperiled (G2) species (total imperiled) make up of the total extant species (excludes all presumed and possibly extinct species). For example, for mammals the future extinction vulnerability would be 29/421 = 0.0689.

insects and all groups are half as large as the vertebrate ratio. Except for the vertebrates, the ratios are not great enough to conclude that important differences exist, given uncertainties in ranking the species and the relationships between imperilment status and future extinction. The higher ratio evident in the species groups examined by Ricciardi and Rasmussen (1999) is influenced substantially by a relatively high imperilment of crayfish. The estimated future extinction from imperilment indicates that crayfish extinction rate would increase by over 100 times past rates. In comparison, terrestrial vertebrates would increase by only 2 times, freshwater mollusks by 5 times, freshwater vertebrates by 8 times, terrestrial mollusks by 10 times, and aquatic amphibians by 40 times.

The continental ratio (when Hawaii is excluded) for freshwater and terrestrial imperilment is 1.4 to 1, which is substantially lower than study estimates of past continental extinction ratio (2.8 to 1 to 20 to 1 depending on the groups included). This also suggests that the difference between freshwater and terrestrial extinction rate is decreasing. The result is greatly influenced by the highly imperiled terrestrial snails, many of which are Hawaiian species. Freshwater snails also have a major influence on the ratio too, making up 47% of the imperiled freshwater species. Greater than average uncertainty about snail conservation status adds substantial uncertainty to the estimate of the imperilment and future extinction ratio for freshwater and terrestrial species in all 50 states.

Table 5. The numbers and percentages of imperiled and critically imperiled species of the 50 United States and the future freshwater to terrestrial extinction ratio assuming that all such classified species face extinction during the next century.

	Total species		Percent Imperiled[3]		Future Extinction Ratio
	Terrestrial	Freshwater	Terrestrial	Freshwater	
Vertebrates	1,525	992	8.52	22.88	2.7:1
Mollusks[1]	1,854	1,051	38.43	52.33	1.4:1
Insects[2]	1,594	1,598	16.71	19.95	1.2:1
All Groups	4,973	3,641	22.76	29.69	1.3:1

1. Includes only the snails and mussels listed in NatureServe Explorer.
2. Only groups considered comprehensively covered by NatureServe Explorer were included. These are grasshoppers, butterflies, skippers and giant silkworm, royal, sphinx, notodontid, underwing and *Papaipema* moths among terrestrial insects, and stoneflies, mayflies and dragonflies among the freshwater insects.
3. Includes all critically imperiled (G1) and imperiled (G2) species identified in NatureServe Explorer.

Leaving out snails more than doubles the estimated future freshwater to terrestrial extinction ratio from 1.9 to1 to 4.2 to1. Terrestrial snail conservation status is most uncertain. Leaving them out alone, much as Ricciardi and Rasmussen (1999) did (because the data were not available at the time), increases the ratio of freshwater to terrestrial extinction for the 50 states to 5.3 to1. Leaving out the unranked snail species increases the imperilment fraction for ranked snails to over 60% and the terrestrial imperilment fraction to over 31%. That would result in an extinction ratio closer to 1.0. Thus, the imperilment fraction estimated for terrestrial snails should be viewed as a minimum and the estimate of the future extinction ratio is likely to increase as the status of unranked snail species is further determined.

Although the ratio of freshwater to terrestrial extinction appears to be stable when terrestrial snails are included in the analysis, estimates of future vertebrate and invertebrate extinction rates differ. Imperilment indicates that future extinction of freshwater vertebrates will increase more than extinction of terrestrial vertebrates (2.7 to1 compared to 1.3 to 1). In contrast, the extinction ratios for invertebrates indicate a worsening of terrestrial species extinction status relative to freshwater species. The uncertainty of terrestrial snail conservation status (one-third of the species have as yet to be ranked) suggests that the estimate of future terrestrial extinction is a conservative one because many of the unranked species are likely to be imperiled. However, others may already be extinct and the uncertainty of those determinations adds greatly to the uncertainty of forecasting relative rates of future terrestrial and freshwater extinction from imperilment data.

Because critically imperiled species (G1 species) are more likely to become extinct than imperiled species (G2 species), the fraction of all imperiled species (G1 plus G2 species) that are critically imperiled is also an indicator of relative vulnerability. The slightly higher fraction of imperiled freshwater species that are critically imperiled in the United States—68 versus 62% for terrestrial species (Table 4)— indicates little change in the relative vulnerability of freshwater and terrestrial species and is consistent with the stability indicated by the imperilment ratio compared to the past extinction ratio. Regardless of habitat, however, the invertebrates included in this study are more critically imperiled than the vertebrates—68 versus 54%— indicating, similar to the imperilment ratio, that invertebrate status is worsening compared to vertebrates. These trends are dominated by the high criticality of molluscan imperilment, regardless of terrestrial or freshwater status.

ESA Species Endangerment Status

Threatened and endangered species comprise 3.8% of all extant terrestrial species and 9.6% of all extant freshwater species native to the United States (Table 6). The terrestrial taxonomic categories have relatively low percentages of species listed under the ESA, varying between 4.3% of the terrestrial snails and 7.4% of the reptiles listed as threatened or endangered. This is consistent with the low fraction of imperiled and critically imperiled mammal, bird and reptile species (Table 4), but is only 6% of the number of imperiled snails and 12% of the number of imperiled amphibians. Most species listed under ESA protections are considered imperiled to critically imperiled in the NatureServe database, but there are important exceptions, especially among birds and mammals.

Table 6. Species and subspecies listed under ESA protection as threatened or endangered and occurring in one or more of the 50 United States (as of January 2003).

	Endangered			Threatened					
Animal Group	**Species**	**Sub-species**	**Total**	**Species**	**Sub-species**	**Total**	**Total[1] T&E Species**	**Total Extant U.S. Species**	**% T&E species**
Terrestrial									
Mammals[2]	16	43	59	5	8	13	21	421	5.0
Birds	42	27	69	4	9	13	46	764	6.0
Reptiles	7	2	9	14	7	21	21	240	8.8
Amphibians	2	0	2	2	0	2	4	77	5.2
Snails[4]	48	1	49	3	2	5	51	1,780	2.3
Total	115	73	166	35	21	56	127	3,382	3.8
Freshwater									
Amphibians	6	3	9	4	1	5	10	192	4.1
Fish[5]	58	15	73	30	11	41	88	779	11.3
Mussels	55	8	63	7	0	7	62	304	21.8
Snails	17	0	17	5	0	5	22	663	3.0
Crayfish	4	0	4	0	0	0	4	344	1.2
Total	140	26	166	46	12	58	186	1,938	9.6
Grand Total	255	99	332	81	33	114	313	5,320	5.9

1. The sum of all threatened and endangered species native to the United States; excludes subspecies.
2. Includes marine mammals.
3. Includes 44 extant species in one Hawaiian genus (*Achatinella*) as indicated by NatureServe Explorer.
4. Includes salmon and trout stocks as 1 subspecies for each species listed.

The percentages listed under the ESA for the freshwater groups in Table 6 average higher and range between 1.2% of the crayfish and 21.8% of the mussels listed. Unlike terrestrial groups, none of the percentages of freshwater groups listed under the ESA approach the percentages listed as imperiled and critically imperiled by NatureServe Explorer. Mussels have the greatest protection at 52% of the imperiled numbers, but less than 10% of the imperiled snails and crayfish are ESA listed.

Within the vertebrates, the percentage of ESA listed freshwater fish is nearly twice the collective average for mammals, birds and reptiles, but is less than half of the imperiled and critically imperiled percentage of fish. In contrast, aquatic amphibian species are better protected under ESA listing than terrestrial amphibian species. All of the terrestrial amphibians are cryptozoic salamanders. In comparison, a higher fraction of the imperiled and critically imperiled freshwater invertebrate species is protected, but only because of protections extended to freshwater mussels.

In general, the vertebrates classified as imperiled and critically imperiled in NatureServe Explorer are about 5 times better protected than the imperiled and critically imperiled invertebrates, regardless of habitats occupied. Whereas 44% of the invertebrate species in Tables 5 and 6 are listed as imperiled and critically imperiled, only 4.5% of these are protected under the ESA. This compares to total listed imperilment of 13.9% of the vertebrates compared to ESA protection of 6.8%.

In addition, terrestrial and vertebrate subspecies are more likely to be listed than aquatic invertebrate subspecies. Total ESA listings across all taxa are now nearly equal for freshwater and terrestrial species, but a much larger proportion of the terrestrial species are protected at the subspecies level (Table 6). The proportion of listed subspecies is 2.5 times greater for terrestrial species than for aquatic species. This apparent bias is in part a consequence of greater taxonomic discrimination at the subspecies level for vertebrates. Similarly, greater knowledge of the status of subspecific “evolutionary units” among high profile species of recreational or commercial interest is reflected in the listing of population stocks for the salmonid genus *Oncorhynchus.*

The proportions of endangered and threatened species listed under ESA protection provides some insight into the degree of threat perceived for different taxonomic groups and how far decline had occurred before species were listed under ESA protection. The fraction of listed species that are categorized as endangered is nearly the same for terrestrial and freshwater species, but only because 94% of the listed terrestrial snails are endangered. Of the listed invertebrate species, 89% are endangered compared to 69% of the listed vertebrate species. The later listing of many invertebrates in an endangered status reflects the later recognition of their decline toward extinction. History suggests that imperiled invertebrates are more likely to become extinct because they are tracked less closely than imperiled vertebrates and are less likely to be listed under ESA protection before it is too late to reverse their decline (Flather et al. 1999).

However, listing trends "suggest a taxonomic shift away from highly visible and charismatic species"; mostly vertebrates, which dominated the earliest species lists (Flather et al. 1999). From 1980 to 2002, the listing of freshwater species (indicated by fish, clams and crustaceans) increased by nearly 3 times that of terrestrial species (indicated by mammals, birds and reptiles). Much of the difference between listing rates of terrestrial and aquatic species is due to more recent acceleration of new invertebrate listings, which increased over 4 times as fast as new vertebrate listings. Listing of aquatic species increased markedly during the middle1980s and continued until 1997 when a moratorium was placed on new listing (Flather et al. 1999). The number of new listings have varied from year to year more as a consequence of amendments to the ESA and administrative actions (e.g., moratoriums reflecting political pressures) than as a consequence of imperilment recognition (Flather et al. 1999, Greenwald et al. 2006). If listing ever occurs at rates indicated as needed by imperilment data, the difference between invertebrates and vertebrates should diminish.

A review of candidate species should show a substantially higher fraction of the less protected taxonomic groups—such as snails, crayfish and amphibians—if listing trends are continuing to include more of the lower-profile imperiled species. In addition to the 313 species listed as threatened or endangered among the groups included in Table 6, 73 species are candidates for listing. Of those, 38 are invertebrates and 35 are vertebrates. While this is a higher fraction of invertebrates than in the past, the proportion would need to be much greater to reflect the relatively high number of imperiled invertebrates that remain unprotected. Despite the high fraction of imperiled mammals, birds and reptiles that are already protected, another 14 species and 17 subspecies are among the candidates for listing. Ten species and 2 subspecies of fish are listed. With 9 species and 2 subspecies, amphibians are proportionately gaining the attention that their imperilment indicates is needed (all are aquatic species). However, only ten terrestrial snails (out of 684 listed as imperiled and 13 freshwater snails (out of 423 imperiled) are listed as candidates. No crayfish are on the list despite 116 listed as imperiled.

Whereas a large fraction of the ESA-listed species are categorized as critically imperiled or imperiled in the NatureServe database, 32 are placed in a more secure status (higher than the imperiled G2 status) in NatureServe Explorer. Most of these species are listed as vulnerable (G3) and all are vertebrates. A few vertebrates are listed as secure (G4 and G5). This difference is consistent with the bias toward recognizing the vulnerability of high-profile vertebrate species for protection before other species are recognized.

The recovery status of species, while not very impressive in general, is consistent with the greater length of protection time and knowledge associated with terrestrial and vertebrate species. A small fraction of the ESA listed species has recovered more than 50% as of the 2000-2001 report to Congress (10% of the listed species). Among the species included in Table 6, 23 terrestrial species and 8 freshwater species have recovered by at least 50%. All of those freshwater species were fish, however. Four species of terrestrial snail were the only invertebrates recovered to at least 50% of a delisting status.

Overall, freshwater species and invertebrates are gaining protection more so than they once did, but, as a fraction of the imperiled species, ESA listing of invertebrates still lags far behind the vertebrates. Relative to their low imperilment status, mammals, birds and reptiles continue to be

favored in the listing process. Some of the lag may be associated with only recent recognition of the imperilment status of invertebrate species, especially the snails and terrestrial amphibians. According to Yaffee (1982), discrepancies are significantly affected by differential advocacy for listing by staff biologists, which have collectively favored vertebrates over mollusks except during a short period when an "assertive" malacologist was on the staff.
Other explanations have to do with differences in the way imperilment and endangerment are judged. Regardless of present threat level to the species, rarity is the primary reason species are listed as imperiled in the NatureServe Explorer database (NatureServe 2007). Many imperiled species are localized in springs, caves, mountain tops and other such isolated habitats where threats may not rise to the same level as for other imperiled species. Demonstrated threat is a much more important reason for listing species under ESA protections and for differentiating between threatened and endangered status.

For whatever reason, the more complete protection of imperiled vertebrates, and especially mammals, birds and reptiles, indicates greater likelihood that imperiled vertebrates are more likely to survive the stresses of the next century than imperiled invertebrates. Vertebrate distribution, biology and threats are better known in general, and their protection is more likely to be effective, once listed (Scott et al. 2006, Greenwald et al. 2006).

Based on the assumption that listed species are much more vulnerable to extinction than unlisted species, the ratio of freshwater to terrestrial vulnerability to extinction based on ESA listing is another way to estimate future extinction rate differences between terrestrial and freshwater species. The freshwater to terrestrial ratio is 2.5, which is somewhat greater than the 1.9 determined from imperilment and past extinction identified in NatureServe Explorer data. The primary reason for this difference is the comparatively low ESA listing of terrestrial snails and the comparatively high ESA listing of fish and mussels. The ratio would be substantially lower (2.0) if either terrestrial snails alone or all snails were excluded from the analysis.

Trend Analysis For Future Extinction Rates

The number of extinct and possibly extinct animal species last observed in each decade from 1880 through 1989 is summarized in Table 7. The decadal rate of species loss appeared to double from 1930 to 1940 largely because of terrestrial snail loss in Hawaii. Thus the trends indicated in Figure 1 are largely determined by the extinction history of Hawaiian land snails. The data were fit with linear, logarithmic and exponential models to assess the range of forecasts predicted from different trend interpretations of the extinction record. These result in forecasts that range from the pessimistic to the optimistic and widely bracket some probable outcome.

The dates of last record for all extinct species almost quadrupled over the period of record whether the trend is fit to a linear or logarithmic model. Extrapolation of a linear model for the record of last observations fit to the data in Figure 1 results in a forecast of about 250 last dates of observation over a 100 year period from 1990 to 2089. Accounting for the fraction of the total loss that was not dated (53%), the forecast loss totals to about 53 species per decade and 540 in total by about 2089 among the indicator groups. This model also implies that about 85 species among the indicator groups have been observed for the last time since 1990.

Table 7. Decade of last observation, by taxonomic category, of species presumed extinct and possibly extinct as summarized in Stein et al. (2000).

Taxonomic Category	Before 1880	Initial Year of Decade Last Observed (1880-1980)										
		1880	**1890**	**1900**	**1910**	**1920**	**1930**	**1940**	**1950**	**1960**	**1970**	**1980**
Mammals			1						1			
Birds	5	1	7	3	3			1		2		7
Amphibians										1		
Snails			1	5	0	1	2	14	11	8	5	3
Terrestrial	5	1	9	8	3	1	2	15	12	11	5	10
Amphibians								1				
Fish		1	1			2	2	2	3	2	4	3
Mussels	1		1			1	4	2	2	6	3	2
Snails		1		1		6	1				2	1
Crayfish	2						1					
Freshwater	3	2	2	1	0	9	8	5	5	8	9	6
Grand Total	8	3	11	9	3	10	10	20	17	19	14	16

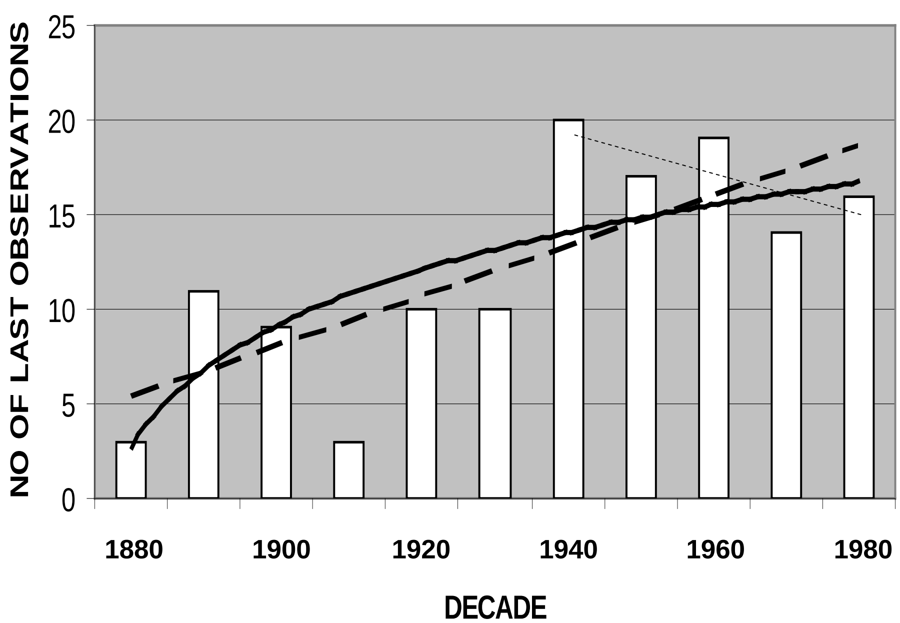

Figure 1 Three recent extinction trends indicated by linear and logarithmic models fit to the number of last species observations per decade for all presumed and possibly extinct species included in Table 1. Dated species comprise 47% of the total.

The linear forecast assumes that increased awareness and protections in the last half of the 20th century will have little affect on a growing rate of species loss. Yet the forecast amounts to about 30% of the imperiled and critically imperiled species, thus is substantially less pessimistic than a forecast based on the assumption that all imperiled and critically imperiled species are doomed. The linear model is more pessimistic than the forecast of a logarithmic model, which fits the data better than the linear model. It estimates a loss of about 385 species over the period from 1990 through 2089, and about 23% of all imperiled and critically imperiled species.

A more optimistic model ignores the increasing loss recorded in the early history of recent extinction and assumes that a decreasing linear trend observed since the 1940s will continue (Figure 1), which is consistent with changes in human awareness and legislated protections that emerged after World War II. It forecasts a loss of 180 species from 1990 through 2089, which is 3.5% of the extant species and 10.5% of all critically imperiled and imperiled species. This estimate of future loss is consistent with the conclusion that numerous invertebrate species are already functionally extinct, i.e., no longer reproducing for reasons that cannot be explained or managed (e.g., Neves et al. 1997). More optimistic forecasts are based on the less likely assumption that research and protections can be greatly accelerated and that all but a few imperiled species can be made secure.

The most pessimistic forecasters might attempt to fit an exponentially increasing function (not shown) to the data, predicting much higher extinction rates than the models applied here. Forecasts based on an exponentially increasing model would result in the extinction of most species. It is the only model evaluated that predicts more species disappearance than estimated by the sum of imperiled and critically imperiled species. This model is the least likely fit to the observed data, however, and is the least consistent with changes in events that have been implicated with past extinctions.

A logarithmic model predicted a leveling of loss rates for and nearly equal species loss from terrestrial and freshwater groups; about 195 terrestrial and 190 freshwater species from 1990 through 2080 (Figure 2). A less optimistic linear model predicts about 265 terrestrial and 245 freshwater species extinctions over the same time frame. Both models forecast a freshwater to terrestrial extinction ratio of just over 1.0, which is more similar to the ratio estimated from present imperilment status than the 8 to1 ratio estimated by Ricciardi and Rasmussen (1997). The timing of last observations is quite different for terrestrial and freshwater species. About two-thirds (65%) of the extinct terrestrial vertebrate species were last observed before 1920 (Table 7). In contrast, only 10% (two species) of recently extinct freshwater vertebrates were last observed before 1920. A secondary upswing in vertebrate extinctions occurred in the 1980s when seven bird species disappeared from observation records, all but one of which are Hawaiian.

More optimistic projections of extinction are based on the linear models fit to data obtained after the 1920s for freshwater species and after the 1940s for terrestrial species (Figure 2). Slowly decreasing trends in freshwater species forecast a loss of 130 species from 1990 to 2090, which differs little from a slightly higher loss estimated by a logarithmic model fit to the same data. A linear model of the recent trend for terrestrial species forecasts a loss of only 20 more terrestrial species from 1990 to 2090. A less optimistic logarithmic model for the terrestrial extinction

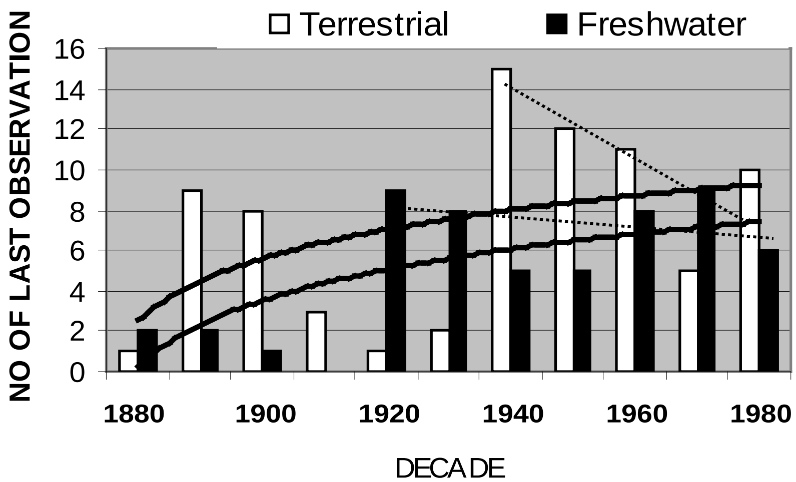

Figure 2. Extinction trends based on decades of last observation for terrestrial and freshwater species in the United States for the 1880s-1980s, for the 1920s-1980s for freshwater species, and for the 1940s-1980s for terrestrial species.

trend since the 1940s responds to the recent increase in last observations recorded in the 1980s and forecasts a loss of about 115 terrestrial species. Both linear and logarithmic models forecast greater losses of freshwater species than terrestrial losses, with the ratio of freshwater to terrestrial extinction varying between 2 to1 and 12 to1. This range spans the differences observed in the estimates of this and the Ricciardi and Rasmussen (1999) study based on imperilment ratios and reflects the substantial uncertainty associated with any projection.

Limiting the analysis to the continental United States (Figure 3) produces much different forecasts of extinction ratios because all but a few terrestrial extinctions occurred on Hawaii. Extrapolation of a linear model for continental extinction rates forecasts an estimate of about 225 species lost in total from the 1990s through the 2080s. This is less than half of the estimate that includes Hawaii because all but a few terrestrial species became extinct there. Extrapolation of the terrestrial loss on the continent forecasts only one to two more species lost from1990 through 2089 and forecasts a very high freshwater to terrestrial extinction ratio (over 100 to1) for continental species. Estimates from logarithmic models reduce the ratio to17 to1, but it remains high compared to estimates from imperilment data.

Invertebrates make up the large majority of predicted disappearances. Their past pattern of disappearance is similar to freshwater vertebrates in the early years of record. However, the record of last observation is substantially less complete. About 88% of the presumed and possibly extinct invertebrates were last observed since the 1920s (Table 7) and reached a peak disappearance rate in the 1920s when several freshwater river snails were last observed in the

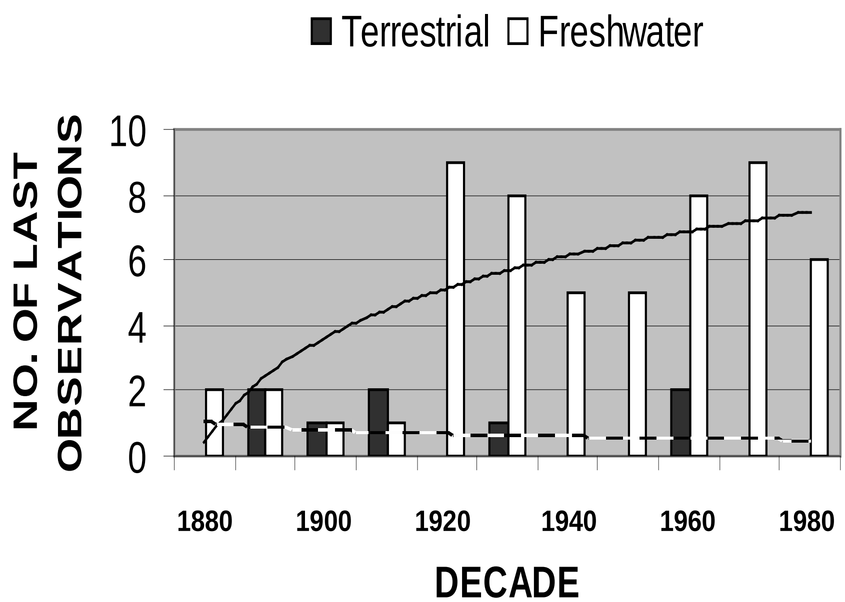

Figure 3. Extinction trends for continental freshwater and terrestrial species estimated from decade of last observation for the 1880s-1980s.

Coosa River system. Last records for mussels increased substantially in the 1930s but did not peak until the 1960s. The trends based on logarithm models of loss rates since the 1920s (Figure 4) indicate that extinction rates of both vertebrates and invertebrates in freshwaters will level off over the next century. Linear models, in contrast, forecast substantially higher vertebrate extinction rates for the same period. Both projections differ from imperilment data, which indicate twice the extinction rate for invertebrates.

Correcting for the fractions of extinct species with last dates of extinction of extant freshwater species and the average rate of loss per decade, Figure 4 indicates that about 95 more freshwater invertebrate species and 30 more freshwater vertebrate species could be lost over the next 100 years, assuming stability in the rate of loss. Rates of species loss since the 1970s, when the ESA was passed, appear to be declining even more rapidly, based on very limited data. Continuation of the rate of drop from the 1970s to the 1980s suggests a much lower expected loss of species and a decrease toward an unlikely low loss rate, approaching zero, within a few decades.

In summary, the trend analysis indicate very different future extinction rates from those indicated by imperilment and endangered species data, both with respect to total rates of loss and to the proportions of freshwater and terrestrial species. While logarithmic trends appear to reflect the probable positive impacts of the ESA and other environmental legislation more realistically than projections based on imperilment and endangerment, the lower projected loss rates based on these trends also assumes that environmental protection commitments will remain strong.

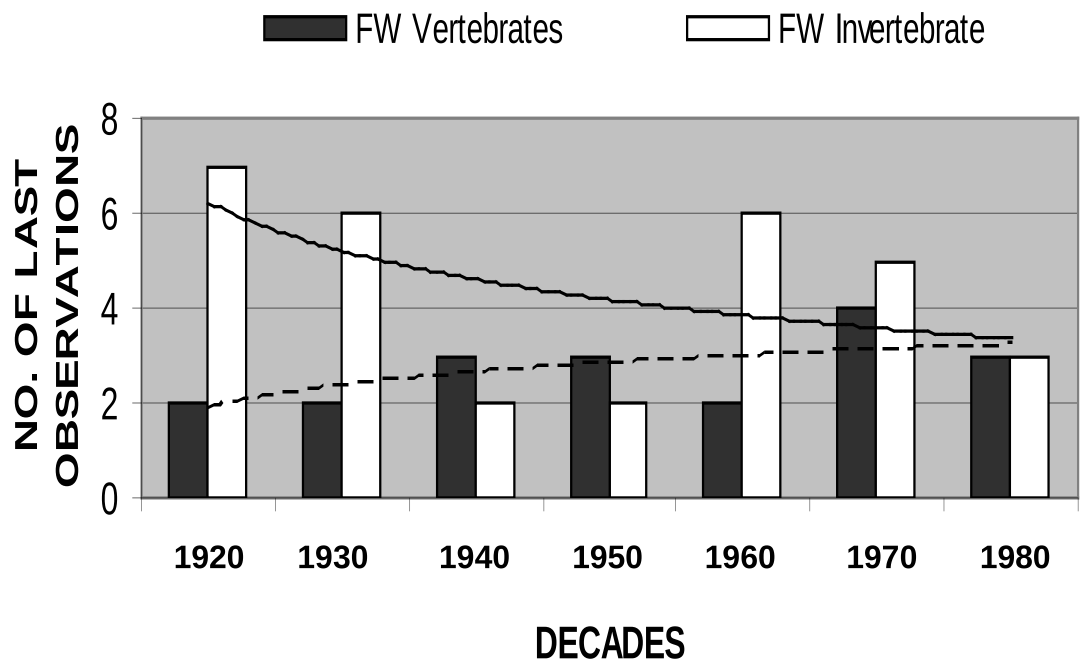

Figure 4. Logarithmic extinction trends for freshwater vertebrates and invertebrates since the 1920s.

CAUSES OF FRESHWATER ANIMAL SPECIES DISAPPEARANCE

Biological Attributes Associated with Freshwater Species Decline

General

Freshwater animal population numbers increase, disperse and decrease in response to habitat variation through time and with interactions with other species. The susceptibilities of freshwater animal species are generally described in a number of overviews including Ono (1983), Diamond (1984), Minckley and Deacon (1991), Moyle and Leidy (1992), Matthews (1990, 1992, 1994), Allan and Flecher (1993), Bogan (1993), Stiassny (1996), Richter et al. (1997), Neves et al. (1997), Taylor et al. (1996), Parmalee and Bogan (1998), Stein et al.(2000) and Lannoo (2005). Others review the general biological implications of natural and human caused variation in the physical and chemical habitat (e.g., Bryan and Rutherford 1993; Waters 1995).

Habitat change is usually key because of its direct effects on recruitment, dispersal and mortality, and its indirect effects on species interactions such as predation, competition, parasitism, disease and hybridization. All freshwater animal species require enough fresh water to respire normally through all fully aquatic stages of their life cycle. Most desiccate quickly once exposed to air although some may persist for weeks after habitat is dewatered. All rely on dissolved oxygen and suffocate once it is reduced to intolerable levels (which vary among species). Most have limited tolerance to elevated temperature, which also reduces the amount of oxygen dissolved in water. Most respire through gills, which must be kept free of coating by very fine sediments (clay and silt) to perform optimally. Through gill and other soft tissue uptake, most are sensitive to extremes in acidity/alkalinity, high concentrations of metals, various constituents of petroleum, and synthetic organic compounds developed for pesticides and other uses.

Most species are most narrowly adapted to their habitats during the earliest life stages when eggs and larval forms are often unavoidably exposed to environmental threats. Many species fail to reproduce normally without temperature, discharge, water-level or other seasonal cues. The capacity for juvenile and adult dispersal to suitable habitat well beyond the home habitat is often critical to species sustainability. Broadly adapted species with high dispersal capacity typically are less threatened by habitat and other change than narrowly adapted species prevented from dispersal by environmental barriers. These narrowly adapted species often occur in freshwater springs, cave waters, or a single lake or stream system isolated by land or by unsuitable aquatic habitat, where they are vulnerable to any pervasive stressor that enters the habitat.

Small ecosystems are more likely than larger rivers and lakes to be thoroughly damaged by dewatering, filling and pollutants loading that threaten inhabitant survival. Species that are sustained in particular habitats by high dispersal rates usually decline as ecosystems become fragmented. Cumulative alterations of larger, well-connected habitats, as occur in naturally free-flowing rivers, have often reduced and fragmented suitable freshwater habitats in the United States. The fragmentation results in smaller, more isolated subsystems that prevent species dispersal (Angermeier 1995, Warrant et al. 1997) and increase population vulnerability to pervasive stressors within each of the isolated subsystems.

Even larger ecosystems can be thoroughly degraded by the invasion of nonnative species or by widespread physical and chemical impacts. Species locked within single uniform ecosystems, regardless of size, are especially vulnerable to any pervasive threat to their survival, such as a new predator, competitor, or closely related species with which it might hybridize, and various pollutants. Isolated ecosystems typically have relatively low biodiversity. The native inhabitants are typically more vulnerable to the effects of invasive species probably because they have evolved with less intense competition and predation.

Amphibians

Aquatic amphibians have complex life cycles requiring appropriate connectivity among suitable habitats for each life stage (Duellman and Trueb 1994 and Lannoo 2005). North American amphibian reproduction is typically most successful in isolated waters generally free of fish and other large aquatic predators. Some large salamander and frog species are exceptions to this general rule, however. Many salamander species have small ranges in isolated habitats and are sometimes limited to a single freshwater spring.

Few aquatic salamanders live in the arid west (*Ambystoma tigrinum* is the outstanding exception), but a number of western anuran species depend on scattered remnant habitats persisting since prehistorically wetter times. Predaceous fish and frogs (especially bullfrogs, *Rana catesbeiana*) have been widely introduced into once isolated perennial habitats, especially in the western United States.

The eggs of many anuran species and some salamander species are laid in small stream and wetland pools that dry up seasonally. The annual population recruitment of these species is especially impacted by drought. Eggs and larvae are fully exposed to predators and diseases (fungi are a common cause of egg mortality) that manage to penetrate the isolation. Many of these temporary wetland habitats have been drained, filled, or modified for urban and agricultural purposes, which decreases population dispersion success and increases the probability of local extirpation by droughts. Most aquatic salamanders lay eggs adjacent to or in small seeps and springs where larvae develop free of fish and other large predators.

The eggs and skin surface of amphibian larvae and adults are highly permeable to dissolved materials, making them especially sensitive to many pollutants. Freshwater amphibians are vulnerable to bacterial infections commonly associated with pollution. While the skin is an active surface for respiration for most species, most larval salamanders also have gills, the size of which varies inversely with the oxygen content of their habitats. Many larval frogs have rudimentary lungs which are used when oxygen concentrations fall. Most amphibian larvae cannot tolerate much salinity in their aquatic habitats.

Little appears to be known about the tolerance of larval amphibians to inorganic suspended solids and sediments, although declines are often associated with agriculture, logging and urban development, and associated pollution and siltation of aquatic habitats (Lannoo 2005). Sedimentation may interfere with both feeding and respiration as well, totally filling habitats or reducing their utility by filling bottom interstices. With respect to food, amphibians tend to be generalists. Salamander larvae select a wide variety of small invertebrate foods, probably using visual, olfactory and tactile cues. Many frog and toad larvae filter suspended matter from the water for food and appear to be adapted especially well to plankton-rich eutrophic waters. The extent to which feeding and respiration are directly impacted by suspended solids and sediment appears not to have been well researched.

Freshwater Fish

The life histories of fish are among the more diverse of the freshwater taxonomic groups included in this study. Freshwater fish usually grow to be among the largest fully aquatic species in their ecosystems and as adults rely largely on size and mobility to avoid environmental threats. Other than over-harvest, fish are quite consistently most vulnerable to environmental stress in their earliest life stages. Consistent with egg and larval sensitivities, fish typically undergo high mortality at an early age (Diana 2004), more so than most terrestrial vertebrate species. Mortalities vary, but generally exceed 99% in the first few weeks of life.

Most mortality occurs in the egg and after hatching just after the yolk sac is absorbed when the larval fish must feed for themselves for the first time. Mortality at that time is related to egg development, which uses up yolk in stressful conditions leaving less than needed by larvae before the alimentary canal fully develops. Oxygen depletion, pollutants, low pH and other environmental variables can contribute to that stress. The larger a larva is at hatching, the greater is its prospects for survival because loss to predation decreases with increased size. Increasing egg incubation temperature above optimum causes decreased size at hatching. Some small species largely reduce this early mortality by retaining the eggs and giving birth directly to late larval or early juvenile stages. Older fish vary widely in growth rate, but usually exhibit more stable mortality rates than eggs and larvae.

Fish eggs are particularly sensitive to conditions that inhibit passage of oxygen into the egg and passage of carbon dioxide and other waste out of the egg, such as coating with fine sediments. Tolerance to sediment and turbidity varies among species according to the habitats they occupy. Even the adults of many imperiled fish in the eastern United States are adapted to conditions generally free of prolonged turbidity and fine sediments except during flood events. They typically depend largely on sight to feed and escape predation and are closely associated with bottom habitats in small to medium size streams where they typically reproduce on stony bottoms, or in bottom gravel, submerged plants, and recesses (Angermeier 1995, Warren et al. 1997). Those fish species that have low fecundity, large eggs, and a dependency on gravel/cobble substrates for reproduction are especially susceptible to sedimentation and sediment embedded spawning substrate. They include many minnows, darters, stonecats, sculpins, and salmonids, which make up a large fraction of the imperiled fish species.

Some small species are found in isolated habitats free of intense predation, competition and closely related species capable of genetic introgression (hybridization). They are prominent in arid regions that were once much wetter (Minckley and Deacon 1991), but also occur less prominently in freshwater springs elsewhere (Warren et al. 2000). These include, most prominently, minnow, cyprinodontid (i.e., pupfish, killifish) and poecilid species (e.g., mosquito fishes). Many of them are imperiled.

At the other extreme in size and mobility, some of the larger freshwater fish species migrate long distances between habitats suitable for reproduction and other aspects of their life cycle. Most of them migrate between ocean and inland habitats of very different salinities. Blocking migrations between required habitats stifles reproduction and often increases migrant mortality. These predominantly include members of the sturgeon, salmonid and clupeid families. Some of these are vulnerable. Whereas fish species that migrate between freshwater and the ocean are most notoriously impacted by dams, other freshwater fish require access to different habitats within their freshwater ecosystems, such as migration between lake and tributary habitats, and between river and wetland backwaters made accessible during annual flooding.

Crayfish

Many crayfish species are widespread and broadly tolerant, especially to respiratory conditions (Hobbs 2001). They have radiated into virtually all types of aquatic ecosystems,. They feed omnivorously, are mobile and can avoid temporary desiccation and freezing by burrowing into sediments. A large fraction have become quite specialized inhabitants of caves and associated spring systems and are much more narrowly adapted to those specific conditions. Crayfish are particularly diverse in the midwestern and southeastern United States. That few have suffered extinction is testament to the resilience of the adaptable and typically widespread species and to the somewhat protective isolation of many more specialized species inhabiting small ranges in springs and caves.

Widely ranging crayfish species are adaptable to many of the habitat changes that have occurred in the United States including turbidity and sedimentation; accentuated fluctuation in river discharge, water levels, and temperature; organic loading, including the consequences of eutrophication; and river impoundment. Crayfish have relatively low fecundity and carry their eggs with them, which reduces vulnerability to environmental stressors during the earliest life stages. However, they are sensitive to contamination, especially by metals and pesticides (Buikema and Benfield 1979, Naqvi and Leung 1983, Thorp and Gloss 1986) . Crayfish are especially vulnerable to environmental stress during periodic molting of the exoskeleton, which is necessary for growth and maturation.

More specialized species, which are often restricted naturally to small ranges, are highly vulnerable to any intolerable habitat change that pervades their home ecosystem, including the widespread introduction of fish and more aggressive crayfish species (Clancy 1997). Because cave species tend to live longer they may accumulate lethal doses of contaminants over long periods of time (Hobbs 2001) as well as be susceptible to a single massive exposure (e.g., Bechler 1983). The progression through ground waters of chemical contaminants from agricultural and urban sources is a troublesome concern for isolated spring, cave and stream species. A less important concern now, only because it is more readily controlled, is inundation of small river habitats by construction and operation of large impoundments.

Freshwater Mussels

The habitat needs of freshwater mussels are generally understood (McMahon and Bogan 2001, Smith 2001, Strayer et al. 2004), if not precisely known for many species. The susceptibility of species in this group to habitat change has been recognized for many decades (e.g., Stansbery 1970) and is reflected in the high fraction of species that are now listed as imperiled and endangered. McMahon and Bogan (2001) summarized some attributes of many freshwater species that may explain this vulnerability:

> "Extended life spans, delayed maturity, low effective fecundities, reduced dispersal, high habitat selectivity, poor juvenile survival and extraordinarily long turnover times make unionacean populations highly susceptible to human perturbations."

These attributes limit recovery rates once populations are decimated, making them especially vulnerable to the cumulative impacts of different stresses even after stress has eased.

Acknowledging that some species show mixed characteristics, unionacean freshwater mussels tend to fit into two ecological categories. Species that are broadly adapted to a range of velocities and bottom types, including lake habitats, tend toward thin, light, smooth and often elongate shells that are more likely to be crushed by rock and other debris during floods than species adapted specifically to river environments. The broadly adapted species are more likely to thrive in depositional environments of ponds, lakes and rivers. Some are able to move through, rest on, and even thrive on deep muddy sediment and flourish in impoundments constructed in their native rivers (Howells et al. 1996, Parmalee and Bogan 1998).

Morphologically these species grade into more specialized species found most commonly in river riffles and shoals.

Mussels adapted to river riffle and shoal environments commonly have thick, heavy, ridged shells that are especially resistant to breaking. They are found most commonly at moderate velocities in riffles with bottoms of coarse sand, gravel and small cobble washed free of fine sediments. Shell thickness and shape protects them from physical impact by stones dislodged during moderate flood events. Deep shifting sand, boulder and bedrock are the least tolerated bottom types for most species (Howells et al. 1996). They decline in species number and abundance wherever high concentrations of inorganic suspended solids are chronic and fine sediments accumulate in riffle habitats. The thick-shelled species tend to grow slower, mature slower and live longer than thin-shelled species (Howells et al. 1996). The history of mussel imperilment and extinction since the 19th century indicates that riffle species are substantially more vulnerable to extinction than more broadly adapted species found over a range of habitats, including ones that are now impounded.

Adult mussels and many of their fish hosts require seasonal change in temperature to initiate successful spawning and some may respond to seasonal changes in river discharge. Changes that reduce the seasonal variation, such as sometimes occur below large dams, may obscure or eliminate spawning and other life cycle cues.

The life cycle completion of most species typically depends on the availability of specific host vertebrates, usually fish, for survival and development of the parasitic larvae, or glochidia (Hoggarth 1992, Parmalee and Bogan 1998). The extent to which the parasitic stage is skipped is not completely documented, but is generally thought to be exceptional (Strayer et al. 2004). In addition to nutrition, host species are important vectors for mussel dispersal, especially upstream from parent populations. Local elimination of host species has been associated with local extirpation of freshwater mussels (Fuller 1974, Neves et al. 1997), and restoration of host populations has led to the recovery of some imperiled populations (McMahon and Bogan 2001, Neves et al. 1997). Among most mussel species so far investigated, one to a few fish species are uniquely suitable as hosts; a smaller fraction of mussel species accept a wide variety of host species (e.g., Fuller 1974, Watters 1994, Parmalee and Bogan 1998, Strayer et al. 2004).

The fish hosts for riffle mussels typically spawn in riffles while the hosts of the more broadly adapted mussel species more typically spawn in the shallows of lake, ponds and river backwaters. Both residential and anadromous fish species have been identified as hosts for specific mussel species. Host fish declines in mussel habitat are commonly listed among the suspected causes of mussel declines, but are much more rarely certainly identified as the cause. Detailed information about host mussel relationships is rarely available for extinct species, but is an active and important area of research needed for improved management of imperiled mussels.

As glochidia metamorphose to the juvenile form, they detach from the fins and gills of their hosts and drop to bottom where they may live as juveniles in sediment for years before maturing to adults (Strayer et al. 2004). The juvenile form is much like that of the adults; both life stages have limited mobility and are slow to avoid rapid, harmful changes. However, mussels can sense and exclude harmful metals, pesticides and sediment concentrations in water or exposure to air by tightly closing the shells and relying on anaerobic respiration, for several weeks in some species. Because they are immobile under such conditions, longer durations of exposure eventually become lethal. Some species can move fast enough to avoid desiccation from slowly decreasing water levels, but prolonged drought concentrates survivors where they may be more uniformly exposed to other stressful changes in water quality, temperature and sedimentation. The tolerance of emerged individuals to freezing is in need of research (McMahon and Bogan 2001).

The optimal water velocity for many of the thick-walled species is fast enough to suspend and carry off fine clays and silts, but slow enough to leave stable shoals of sand and gravel where the mussels can maintain proper position for respiration and feeding. Scouring velocities can be as harmful as velocities that are too slow, especially for thin-walled species. Even thick-shelled species are vulnerable to crushing in extreme events where gravel beds are mobilized and gouged by logs and other debris. Valoverta (1990) described the damage done by log rafting in rivers of Finland to a European mussel species. Species richness is reduced among streams as flow variability increases, causing exposure to air or low oxygen during low flow periods and to scouring during high flow periods (Strayer 1983).

Mussels filter organic suspensions from the water using cilia and filaments which form a “stiffened grid” closely associated with their gills (McMahon and Bogan 2001). In addition, juveniles may feed directly from bottom deposits (Strayer et al. 2004). Because they consume large quantities of organisms and organic detritus they are prone to accumulate associated contaminants, such as heavy metals and pesticides. High concentrations of suspended silt and clay impede both respiration and filter-feeding, and, coupled with significant bottom siltation, can cause massive mortality of riffle species.

Mussel gills must be kept moist and generally free of silts and clay to respire effectively. But some species can tolerate weeks of suspended sediment and emergence by tightly shutting, thereby protecting the gills and other organs from sediment and desiccation. Because of smaller size, young riffle mussels are especially vulnerable to loading of river beds with fine sediment, localized oxygen depletion and habitat dewatering. For similar reasons they may be more sensitive to the desiccation, freezing and overheating associated with habitat dewatering. Smith (2001) indicated that glochidia and juvenile mussels are among “the least tolerant” of any freshwater organism to toxic byproducts of water treatment.

In his review of the early literature, Fuller (1974) indicated that the riffle species were more sensitive to oxygen depletion, but none of the species investigated can persist in low oxygen concentrations for more than a few weeks. Gunning and Suttkus (1985) documented the recovery of 5 mussel species in a southern river once sewage was treated and oxygen restored. Most shelled mollusks do best in alkaline waters above pH 7.0 and do not tolerate extreme acidity below a pH of 4.7 to 5.0 (Fuller 1974, McMahon 1991). Recent studies reviewed by McMahon and Bogan (2001) indicate that mussels exposed to sewage, industrial waste, pesticides, acid discharges and mine silts were “severely depauperate or totally extirpated.”

Using the above characteristics as a guide, a habitat model for predicting the number of mussel species would indicate maximum number in generally stable, well-aerated flows of moderate velocity and low turbidity over sand-gravel bottoms. A number of studies confirm this habitat model (McMahon and Bogan 2001). The optimum habitat was found in many of the medium to large rivers of the midwestern United States before substantial human settlement. The species number generally decreases as stream size and the reliability of flow decreases.

One complication for assessing the causes of mussel species decline is the long life expectancy of many of the riffle species and the ability of at least some species to persist as adults without reproducing successfully. Most species live for 50 years or more, and some may live more than a century (Strayer et al. 2004). “Disturbance-induced lack of juvenile recruitment raises the specter of many North American unionacean populations being composed of slowly dwindling numbers of long-lived adults destined for extirpation as disturbance of some kind prevents juvenile recruitment to aging populations”(McMahon and Bogan 2001). Reproductive failure and population decline may have originated from stressful events that occurred decades ago or even a century or more, if recent evidence that past life expectancies may be substantially underestimated (Strayer et al. 2004) proves correct.

The frequency of reproduction under natural conditions is not well studied. While some if not most species reproduce annually (Strayer et al. 2004), reproductive success may vary widely with good recruitment occurring much less frequently than annually, but enough to sustain healthy populations (e.g., Payne and Miller 2000). Naturally infrequent reproductive success may complicate determination of a diminished reproductive condition for an unknown number of species. Such species require especially long periods of time to recover population abundance, once depleted. Yet reliable estimates of rates of loss and recovery for mussel species are now well documented (Strayer et al. 2004).

Freshwater Snails

Much like the mussels, the snails fall into two ecologically distinct groups that generally indicate their relative vulnerability to environmental threats. The preponderance of extinct species are members of the gilled prosobranchs. The other major group, the lunged pulmonates, is less vulnerable. General information on snail taxonomy, ecology and distribution is provided by Burch (1989), Neves et al. (1997) and Brown (2001).

Prosobranchs have some traits in common with riffle mussels. They are most diverse in flowing waters, also occur in well-aerated lakes, and are more rarely encountered in ponds where oxygen concentrations often vary. The gilled snails do not tolerate oxygen depletion, nor do they tolerate thermal extremes or extended dewatering. They are in general more mobile than mussels, but have operculum shells that allow them to close tight to retain moisture and to avoid water quality threats for short periods, much like mussels. They are typically thick shelled, which may be an adaptation to somewhat turbulent habitats as well as to predation in exposed habitats. It is not clear whether they require moderate to high velocities separate from the needs of their periphytic foods and other requirements. Johnson and Brown (1997), for example, found that adult snails of one prosobranch species were most common at lower velocities, possibly because higher flows impede their movements. In general, the number of snail species decreases with increased stream velocity (Smith 2001).

Unlike the glochidia of freshwater mussels, which require specific fish hosts, snails tend to be more generalized feeders on diverse species within broad categories of food types. Most feed on either periphytic algae attached to substrates or detritus. Most gilled snails scrape rocky substrates for the attached algae, which requires well-illuminated habitats free of persistent fine sediment or frequent scouring by mobilized sand and gravel. Based on distributions, gilled snails probably are vulnerable to exceptional scour associated with high velocities and to fine sediment accumulation associated with low velocities, although research verifying cause and effect has been sparse. Snails in general do not tolerate acidity (pH below 6.0) and the associated low concentrations of calcium needed for shell maintenance and growth (Smith 2001). Their high gill surface area and permeable epidermis makes them especially vulnerable to uptake of contaminants in low concentrations. The similarly high rates of extinction and imperilment for gilled snails and riffle mussels in rivers are consistent with the similarity of their habitat requirements.

The gilled freshwater snails have diversified most in southeastern river systems. Of the presumed and possibly extinct species, most are river species that lived in the Mobile River basin of Alabama. Secondary diversification of gilled snails has occurred among spring snails (e.g., *Pyrgulopsis*) found in isolated streams and lakes of the western United States where several species extinctions have possibly occurred. In general, gilled snails disperse less successfully than pulmonate snails and are more likely to be limited to ranges within river basins.

Pulmonate snails appear to more widely tolerate environmental variation including oxygen, temperature and sedimentation (Brown 2001). They are more widely distributed among habitat types and are much more likely to be found in small, productive ponds with fluctuating oxygen concentrations. Pulmonate snails are more likely to be thin-shelled and less well adapted to life in river riffles. Like the gilled snails, they are sensitive to low concentrations of metal and other contaminants. Their greatest vulnerability to extinction appears to lie with those species that have become isolated within springs or other small habitats susceptible to destruction. Relatively few pulmonate snail species are listed among the extinct species.

Freshwater Habitat Changes in the United States

Freshwater habitats have undergone a long succession of changes in the United States. Few habitats are pristine in the lower 48 states and Hawaii because of the wide-ranging impacts of human development on water quality. In one form or another, agriculture has pervaded the highest, driest, and seasonally wettest landscapes. Even public lands, comprising more than one-third of the national landscape, were over grazed, over logged and otherwise degraded well into the 20th century. Urban-industrial emissions spread contaminants through the atmosphere to remote locations. Few freshwater habitats have not been hunted, fished, trapped or invaded, with human assistance, by nonnative species. These changes have profoundly influenced the distribution and composition of native species inhabiting freshwater ecosystems

Changed Hydrology, Erosion, and Sedimentation

The pristine watersheds of colonial America west and south of the Appalachian were covered by hardwood forests. Early travelers remarked on the clarity of streams and rivers, even during periods of high runoff (Trimble 1974, Troutman 1981). Human-caused deforestation accelerated in the eastern Mississippi River watershed as Americans migrated westward after the American Revolution. Wood fueled the new settlement and provided most of its building material. Deforestation typically started along river shores and spread rapidly through river floodplains then into the uplands (Williams 1989). It also progressed northward from gulf coastal settlements up the Mississippi and other gulf tributaries as steam navigation became established and expanded early in the19th century. By 1824, the Corps of Engineers was authorized by Congress to remove snags and floating debris to improve river navigation (Shallat 1992), which opened the river floodplains to agricultural development and export of agricultural products to market, and fostered growth of market and manufacturing centers. All of the lower reaches of the major Gulf and Mississippi River tributaries were served by shallow draft steam boats by the 1830s.

Invention of the steel-plow in 1837 advanced the rate and extent of agricultural development. By mid-century, most of the forested watersheds of the upper midwest had been cut or burned and converted to agriculture (Williams 1989). Many rivers were used to float logs to milling centers and log rafts were a probable source of molluscan impact, as they now are in Finland (Valovirta 1990). Cutting progressed more slowly in the southern states, but three-fourths of the forest area was cut for timber, fuel and agricultural development by the early 20th century. As the land was converted to human use, travelers and settlers began to note lower groundwater tables, lower river base flows and increased flood flows, river turbidity and sedimentation (Trimble 1974, Trautman 1981, Sublette et al. 1990, Jackson 1995).

Contemporary understanding of relationships between watershed condition and aquatic habitats explain the changes observed following land conversion to human use. Forest removal along the rivers, including snag removal by the Corps of Engineers, caused river channel erosion to accelerate and contribute more fine sediment to stream and river habitats. But the major new source of fine sediment came from agricultural development and soil erosion following invention of the deep plow in 1847. The proportion of precipitation that infiltrated to groundwater decreased on converted lands and surface runoff increased, carrying eroded soil with it. Rivers became continuously turbid with clay and silt, and graveled stream bottoms were covered by shifting fine sediment, which filled many mill-dam ponds by the end of the 19th century. Areas that had been perpetually wet or flooded dried out as groundwater levels fell and many perennial streams became intermittent (Trautman 1981). The loss rate of timber resources, increased flooding and decreased perennial flow in many areas east of the Mississippi River caused leading scientists to campaign for forest protection and restoration (Dana and Fairfax 1980). This led to creation of the first national forests late in the 19th century.

A similar progression in aquatic habitat change occurred in the floodplains and watersheds of the Rio Grande and central Texas, but was influenced most extensively by rapid expansion of livestock grazing after the Civil War. Runoff from these naturally fragile watersheds became

Chronology of Change in Freshwater Habitats

- **Floodplains and watersheds were deforested throughout the 1800s.**
- **Runoff flashiness increased following deforestation and prairie conversion to agriculture.**
- **Deep-plow agriculture led to faster soil erosion and more stream turbidity in the mid 1800s.**
- **Many small mill dams were built on tributaries where some remained until the 1930s.**
- **Starting in the 1820s, river navigation was improved by removing snags and dredging shoals.**
- **Diversion dams were built in southwestern rivers for irrigation starting early in the 1800s.**
- **Grazing-caused erosion and water diversion were extensive in the West by 1900.**
- **River sediment loads greatly increased from the 1860s to 1950s.**
- **High- and low-flow discharges of rivers reached extremes by the late 1800s.**
- **Many small springs were developed for agriculture and urban use by the early 1900s.**
- **Oxygen-depleting pollution was extensive by the early 1900s and peaked in the 1950s.**
- **Surface mining was extensive in the upper watersheds by the early 1900s.**
- **Early lock and dam development to 6-foot minimum began on some large rivers in the 1880s.**
- **Mussel shell harvest expanded from the late 1800s to 1930s, then resurged in the 1960s.**
- **Severe drought accentuated environmental stress in the 1880s, 1930s, and 1950s.**
- **Petrochemical contamination was common by the 1920s.**
- **Many of the large rivers were "canalled" to 9-foot minimum depth after 1920.**
- **Most large multipurpose dams were completed from 1935 to 1975.**
- **Cumulative environmental stress peaked in the 1950 to 1970s.**
- **Point-source pollution was much reduced after the 1970s.**
- **Endangered species were protected after the 1973 ESA.**
- **Non-point sediment and pollutant loads had resisted management in many locations.**
- **Negative impacts from dams accumulated but some are managed for positive habitat effect.**

flashier and the soils more erodible as livestock density increased (Sublette et al. 1990). Streams became shallower and wider throughout much of the Southwest (Sublette et al.1990). Combined with harsh winters and drought in the late 1800s, overgrazing caused catastrophic losses of livestock, which stimulated the first studies of widespread grazing impacts on watershed conditions (Smith 1899).

Deforested land area peaked in the 1930s. Agricultural land abandonment and natural reforestation had begun in the east early in the history of the Nation, but was more than counterbalanced by deforestation further west. Many of the areas on Appalachian slopes returned to forest cover during the 19th century and restored watershed stability by mid-20th century (Trimble 1974), disguising earlier erosion and sedimentation stress. Intense and extended droughts in the 1880s, 1930s, and 1950s added to the stresses of changed hydrology, water quality and bottom structure.

The impacts of 19th century land use may have persisted in the stream communities of farmed watersheds. Based on the work of Harding et al. (1998) in watersheds of North Carolina, agricultural and other land use in the 1950s was linked more clearly to depressed community biodiversity than land use in riparian and watershed areas during the 1990s. They concluded that land use changes, especially from agriculture, could result in long-term reductions in stream biodiversity despite reforestation.

Pollution from Urban Development and Mining

Urban impacts on midwestern and southern rivers grew rapidly during the 19th century. Domestic wastes had become a serious condition in many river sections by the early 1900s (Merritt 1984), which typically resulted in reduced dissolved oxygen content. Municipalities began to build sewage systems that carried raw organic wastes directly into rivers as urban and industrial development expanded in the late 1800s. Lowered base-flow discharge and warmer water temperature contributed to the extent and degree of oxygen depletion, especially during sustained droughts. In addition to domestic wastes, slaughter houses, dairies, food processing industries, saw mills, paper mills and other agro-industrial wastes contributed to biological and chemical oxygen demands.

Gas, oil and coal mine development occurred in many areas of mussel habitat from Pennsylvania, Ohio, and the Wabash region of Indiana through eastern Kentucky, Tennessee, West Virginia and western Virginia into northern Alabama and Georgia. Trautman (1981) noted that many streams in southeastern Ohio became fishless following development of coal mines in the late 1800s, which probably was true of less documented regions as well. Contaminated runoff from oil and gas development contributed to chemical oxygen demand and to toxicity. Erosion of fine sediment from extensive surface coal and other mining not only contributed to stream sedimentation, but also carried toxic materials with it into the upper Tennessee, Cumberland, Ohio and Mobile river systems. Mine acid drainage increased acidity to intolerable levels for mollusks in many upper tributaries.

Fisheries

Outside of some eastern states, harvest of freshwater species was, for the most part unregulated, until the late 19th century. Sportfishing was increasing in intensity, but commercial exploitation was a bigger concern until market fishing was outlawed or regulated in the 20th century. Even so, the extent of commercial fishing impact was not recognized in the Great Lakes until the 1960s (Smith 1968). Some mussel species were intensively harvested for mother of pearl buttons from the late 1800s through the 1930s before plastic buttons supplanted them. Anthony and Downing (2001) believed commercial harvest was the major cause of mussel loss early in the history of mussel decline. Shell harvesting declined sharply for several decades and resumed in the 1960s to provide shell for Japanese pearl culture (Parmalee and Bogan 1998). Unlike the early years, mussel harvest is now closely regulated by state agencies (Neves et al. 1997).

Nonnative species introduction into new waters began in earnest during the late 19th century (Fuller et al. 1999), usually with the intent to develop fisheries. The common carp (*Cyprinus carpio*), a native of Eurasia, was a popular food fish widely stocked under auspices of the Federal government. Another Eurasian native, the brown trout (*Salmo trutta*), also was transplanted to many locations in cooler waters. Native fish and crayfish were widely moved about and introduced into waters they had not previously inhabited both by private parties and by government agencies. Numerous isolated springs, streams and lakes were stocked, often on top of “useless” native species. Nonnative species invasion also occurred as an accidental consequence of canal construction by state and provincial governments. Building of the Erie and Welland Canals in the early 19th century allowed eventual access to alewives (*Alosa pseudoharengus*) and sea lampreys (*Petromyzon marinus*), which invaded the upper Great Lakes from the east coast.

Widespread introduction of native species for fisheries purposes by government agencies continued well into the 20th century but has slowed since the ESA was passed in 1973. Fisheries related introduction continues through accidental means, however, mostly as a consequence of live bait use for recreational fishing (Fuller et al. 1999).

Invasion of aquatic habitats by foreign species accelerated throughout the 20th century as international trade increased in volume. Hundreds of freshwater species have established a foothold in the United States. Most remain uncommon and, while their impacts on native biodiversity are suspected, they are, in general, not well documented (Cole 2006). One of the most prominent is the Asian clam (*Corbicula fluminea*), which invaded many of the mussel habitats after World War II. It has been suspected of negative impact on some native mussel species through competition for space and food, but confirming data are scarce. Observed co-occurrence of the Asian clam with healthy native species indicates that this is not a critical stress in high quality habitat (Miller and Payne 1994). In the late 1980s, several species of the zebra mussel genus, *Dreissena*, became established in the Great Lakes where they decimated otherwise common mussel species in parts of the lakes (Schloesser et al. 1996).

Early Water Resources Development

Many small dams were built by local interests during the 19th century. Most were built to power small mills. Low-head dams and locks were built to augment navigation on the Kentucky and Green rivers in the early 1800s, followed by many others later in the century. Other low-head dams farther west were built to divert irrigation water to floodplain agriculture. Many small dams were built on spring heads to develop agricultural and urban water supplies. Small dams contributed to bottom sedimentation in the impounded reach and scour immediately downriver. Because impoundment decreases turbulent aeration, it contributed with organic pollution to oxygen depletion in many small- to medium-size rivers. Most small mill dams and early hydropower dams were abandoned for alternative sources of power by the early 20th century (Trautman 1981). Most of the early dams washed out or were breached, allowing subsequent habitat improvement for stream species in unpolluted areas.

Few permanent dams were built on large rivers before the 20th century when the Corps of Engineers was the only Federal agency involved in water resources development. Before the late 19th century, river navigation was improved by the Corps mostly by removing snags, dredging sand and gravel bars, and building wing dams, which concentrated river flow. Dredging was usually limited to a channel less than 100-feet wide, usually in river channels more than 5 to10 times as wide. Occasionally, the Corps built canals around rapids, such as on the Ohio at Louisville and around Mussel Shoals on the Tennessee. The direct physical impact on the large rivers in these early days usually amounted to less than 10% of the shoal and riffle area and much less than that overall. While waterway improvement during the 19th century contributed to habitat degradation, especially for shoal and riffle fauna, it was not as extensive as the degradation caused by other environmental stress. Federal waterway improvement stopped short of many tributaries that underwent extensive habitat alteration from hydrologic and geomorphic changes associated mostly with watershed deforestation and agricultural development.

Pollution from mines and urban development added to the more widespread accentuation of chronic turbidity, bottom sedimentation, flooding and low-flow extremes, and water warming during the 19th century. Locally intense and collectively extensive stress also resulted from oxygen depletion, mine acid, metal and petrochemical contamination, nonnative species introduction, unregulated fish and shellfish harvest, and private dam construction. Most of these stresses were accentuated by extreme drought, such as occurred over several years around 1890 (drought records are anecdotal before the late 1800s). Subsequent severe droughts in the 1930s and 1950s continued to aggravate worsening habitat conditions in many locations. These stresses in combination continued to grow until pollution controls began to be applied, mostly after World War II and especially after the amendments to the Federal Water Pollution Control Act of 1972.

Large Scale Water resources Development

Major changes in the scale of water resources development and Federal involvement began with the 20th century when the Reclamation Act was passed by Congress in 1902. The Reclamation Service—the predecessor of the Bureau of Reclamation (BOR)—immediately began planning for irrigation water supply and delivery systems and initiated construction of Roosevelt Dam, its first, in 1905. Other early BOR projects included Jackson Lake Dam on the Snake River, which inundated several small lakes at Jackson Hole, Wyoming, and Elephant Butte Reservoir on the middle Rio Grande in New Mexico.

The main-stem Ohio, Cumberland, Tennessee, Alabama, Apalachicola, Flint, Chattahoochee, and Ochlockonee rivers were transformed to contemporary waterway dimensions starting in the late 1920s. Waterway development to a 9-foot depth continued into the 1970s (generally ending with development of the Tombigbee Waterway) and that footprint has, in general, been maintained ever since. Contemporary locks and dams substantially increased water depth, slowed average river velocity, accumulated fine sediments, and probably contributed to local extirpation of fish species that may have served as suitable reproductive hosts for some of the extinct river mussels (Neves et al. 1997, Parmalee and Bogan 1998). Because of the impact on water depth, water velocity and fine sediment accumulation, large lock and dam structures contributed substantially to changes in riffle mussel habitat and complemented the changes caused by multipurpose impoundments built subsequently in upstream tributaries.

The era of large, multipurpose dams began early in the 20th century in the western United States, but relatively few were completed in the habitats of what are now extinct and imperiled freshwater species until after World War II. Combined flood control and hydropower purposes were common to most of these reservoirs in combination with water supply in the interior west and navigation elsewhere. While authorization of multipurpose dam construction began in earnest with authorizations of Wilson Dam in 1926 and Hoover Dam in 1928, the peak period of construction occurred after World War II, ending in the 1970s. Multipurpose reservoirs transformed thousands of river miles into elongated lakes and often profoundly altered habitat a similar distance downstream in reservoir tailwaters.

By the early 1920s, private agricultural development in the Central Valley of California, had caused harmful salinity increase in the lower San Joaquin-Sacramento River delta and depressed summer flows in the rivers. In anticipation of worsening conditions, the California state government examined various plans and initiated project funding by sale of bonds in 1933. It soon ran into financial problems. After a brief involvement with the Corps, Congress authorized the BOR to assume control in 1935. Over the next several decades, it and the Corps built 20 reservoirs and several hundred miles of irrigation delivery canals and drainage canals, built in part to freshen the delta and upper San Francisco Bay. Many of the original wetlands and backwaters in the Central Valley were pumped dry, drained and filled in the process.

Several large multipurpose dams were built on the Colorado River by the BOR starting with Hoover Dam in the early 1930s. Over the next four decades much of the Colorado River was altered from its naturally dynamic flood prone and turbid warmwater state to large lakes with regulated releases of clear, cold water. With much of its sediment load trapped in large reservoirs, the remaining river channel degraded, deepened and simplified, loosing backwater habitats important to native fish.

Many large multipurpose impoundments were built by the Corps of Engineers in the Cumberland, Red, Ouachita, Arkansas, White, Missouri, Yazoo and Apalachicola, Savannah and Columbia river basins during this period. The Tennessee Valley Authority (TVA), acting on plans developed by the Corps, completed numerous multipurpose projects in the Tennessee River basin. Most dams in the Coosa and Tallapoosa rivers of the upper Alabama, the Santee, the Pee Dee and other east coast basins were built primarily for hydropower by private utilities starting in the 1920s. Some in Alabama incorporated a flood control purpose managed by the Corps of Engineers. By the 1970s the majority of these river systems were either inundated or greatly influenced by a combination of waterway impoundments in the main stems and multipurpose dams in the larger tributaries. While relatively few large multipurpose reservoirs were built in the large rivers of the Ohio basin, numerous smaller reservoirs were built in the upper watersheds by a mix of government and private agencies, including the Corps of Engineers.

After Elephant Butte Reservoir was completed, several reservoirs were built for a combination of flood control and irrigation storage in the upper and middle reaches of the Rio Grande, which included both BOR and Corps projects. The first of these, El Vado Reservoir, was built during the mid 1930s on an upper Rio Grande tributary to sustain irrigation flow in the middle Rio Grande valley. Two more reservoirs were completed at Abiquiu and Cochiti by the Corps of Engineers in 1963 and 1975. Two large multipurpose impoundments were built by the International Boundary Commission on the Rio Grande, but much of the river remained uninfluenced by dams other than for agricultural diversion for irrigation purposes.

Between the 1930s and the 1970s, stress from sediments and flow variability began to diminish somewhat as stresses from water pollution and impoundments increased. Secondary forest succession, soil conservation practices and trapping of sediment in large impoundments moderated the stresses from turbidity, stream sedimentation, and some water flow variability and pollutants in remaining unimpounded reaches. Some upper watershed impoundments were built during this period in part to sustain stream flow during summer low flow periods.

Post World War II Development and Environmental Regulation

After World War II, new developments and use of pesticides, heavy metals, radioactive materials, fertilizers, and petrochemicals added to the total load and extent of river pollution and to burgeoning environmental concern. Responding to that concern in the 1960s and 1970s, environmental legislation was passed that better regulated pollutant release from point sources into water courses, but was less effective regulating agriculture and other non-point sources. Coincident with increased multipurpose dam construction in response to agricultural and urban demands, groundwater development greatly expanded, threatening the continued existence of ground-water dependent habitats. The soil conservation techniques developed after the 1930s gave way to the green revolution after World War II, which increased soil erosion. Based as it was in heavy use of inorganic fertilizer, pesticides, and large farm machinery, the green revolution encouraged tilling of larger expanses and the elimination of untilled areas, including wetlands and small drainages. Later conservation legislation, starting in the 1980s, encouraged set-asides of wetlands on agricultural lands.

The 1960s was a period of rapid growth in environmental awareness and many state advances in water quality improvement, which culminated in the Federal Water Pollution Control Act of 1972. Point source pollutants from municipal and industrial origins were quite quickly treated under the administration of the Environmental Protection Agency, formed just two years earlier. By the 1980s the once extensive reaches of severely oxygen depleted waters contaminated by industrial waste were rare. Mine reclamation and superfund legislation in the late 1970s had made some progress controlling mine pollution. Non-point sources of nutrients, sediment, and pesticides from agricultural and urban runoff had made much slower progress despite adoption of an approach based on limiting the total maximum daily load of potential pollutants from the watershed. The resulting eutrophication continues to be the major source of diurnal oxygen depletion. Soil erosion rates have declined from their peaks before World War II, but they remain a major source of habitat alteration throughout much of the agricultural midwest and south (USDA survey data displayed in Holechek et al. 2002), and more locally from urban development.

Passage of the Wild and Scenic Rivers Act in 1968, the ESA in 1973 and associated state laws and public sentiment resulted in the exclusion of most remaining undeveloped rivers from water resources development. Few major navigation, flood control, or irrigation water supply projects have been authorized for new construction on wild rivers since then. Most water resources development now occurs within a regional context that has already been substantially developed. That “footprint” extends throughout the major rivers of the United States, however, and is a critical consideration in the future management for sustainable freshwater biodiversity.

At present, the influences of agricultural and urban land and water use, nonnative species, and reservoirs interact to determine most of the threat faced by imperiled species in fresh waters of the United States. Harding et al. (1998) identified land use in the 1950s, especially agriculture, probably had more to do with decreased biodiversity in 24 watersheds in western North Carolina than riparian and watershed land use in the 1990s. This not only indicates the present-day effects of land use over half a century ago, but also indicates the long-term effects that major changes in land conditions during the 19th century probably had in stream ecosystems.

The government agencies responsible for water and land management and environmental resource protection have subscribed to more of an integrated systems approach to resource management through a watershed perspective and planning process (Cole et al. 2005). How that manifests in future action could have much to do with the fate of freshwater biodiversity in the United States.

Causes of Animal Extinction and Imperilment

Amphibians

The Las Vegas leopard frog is the only freshwater amphibian considered extinct. It lived in isolated spring habitats at Las Vegas, Nevada before disappearing sometime after 1942, apparently by habitat alteration associated with spring-water development in an urbanized environment. Federal water resources development is not known to be involved.

Review of the NatureServe Explorer database indicates that many of the amphibian species now listed as imperiled exist in freshwater springs and other hydrologically isolated habitats (Table 8). Most causes for imperilment are unspecified and primarily associated with small numbers in small natural ranges. None of these locations is clearly threatened by Federal water resources development. Of 24 critically imperiled species reviewed, 15 are salamanders with aquatic larvae. These aquatic salamanders provide an excellent example of “island species” in freshwater habitats. All but one of the 15 species appear to live in single spring and cave systems. The one exception also is relatively isolated, limited to a few higher-elevation streams on a single mountain.

In addition, 5 toads and 3 frogs are listed as critically imperiled. The taxonomic status of 3 toads is under review. Most of the imperiled frogs and toads occur in a few scattered aquatic habitats typically occupying single basins or mountain ranges in the arid west. Two other species occupied larger ranges on the Gulf Coast plain, which has been highly modified by a succession of agricultural and urban land developments that began 150 years ago

A variety of specific threats are of concern, including, most commonly, groundwater draw-down, introduction of aquatic predators (e.g., fish, bullfrogs), grazing impacts, physical obliteration by filling or other development, and watershed conveyance of sediments, toxins and oxygen-depleting organic matter. Of a more general nature, changes in atmospheric temperature, ultraviolet light, and acid precipitation are of concern because they are suspected as causes of amphibian decline separate from or in combination with increased incidence of disease (mostly fungal infection), increased habitat salinity, habitat degradation by grazing and exposure to environmental contaminants of various kinds (Stebbins and Cohen 1995, Lannoo 2005).

Most fundamentally, a small range is the main reason for listing taxa as imperiled for most amphibian species. Any pervasive and lethal agent that gains access to the habitat is a critical threat, and there appear to be many candidates. Contamination of surface and groundwater is a big concern, especially from agriculture and contaminants transport systems (e.g., roads, railroads, barges, pipelines). Global warming and other atmospheric changes are of concern because their effects are so widespread and could interact in additive or synergistic ways with disease and contaminants. Even among the amphibians with large ranges, the habitats in which they complete their amphibious life cycles tend to be fragmented and susceptible to population loss and species decline (Stebbin and Cohen 1995). A small fraction is associated with local development of water supply impoundments. Many species are located in habitats located on or surrounded by private land. None of the recorded imperilment of amphibians seems to be associated with major water resources development.

Despite extensive imperilment, only nine freshwater amphibian species are listed under protection of the ESA. Reasons given for the listing of 6 species are summarized in Table 9

Table 8. The number of times specific factors were identified in professional judgment as primary, secondary and tertiary causes of critical imperilment in NatureServe Explorer (2005).

Indicated Cause of Imperilment	Amphibians			Fish			Mussels			Snails			Crayfish		
	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3
Degraded habitat		2		3	13	5		4		2			1		
None identified	2			4	4	1				34			3		
Small Range	19			34			4			116			50		
Unspecified Cause	21	2	0	41	17	6	4	4	0	152	0	0	54	0	0
Land development	2	4		4	5	1	1			2	1			4	
Water depletion		2		1	6	2					1			1	
Contamination			3	5	6	10	2	8	1	2	3	1		1	
Sedimentation			1	1	3		4		1		2				
Small impoundments		1		1	2	1		1							
Over-fishing				1			23								
Keystone species loss			1				31								
Nonnative Species				1	6	9		1			1			1	
Collecting									1						
Natural causes				3	2										
Other than large-scale water Development	2	7	5	17	30	23	61	10	3	4	8	1	0	7	0
Large impoundments				6	3	1	6	3	1	27		1		1	
Channelization															
Dredging															
Hydroregime change				5				1							
Water Diversion				2	3										
Large-scale Water development	0	0	0	13	6	1	6	4	1	27	0	1	0	1	0
Total	23	9	5	71	53	30	71	18	4	183	8	2	54	8	0

based on recovery plans and other sources summarized in Matthews (1990,1992, 1994). Freshwater amphibians are more typically threatened most by local water depletion, pollution and other alteration of their aquatic environments. No large-scale water developments have been identified as threats.

Fish

Fish are the most thoroughly studied of the taxonomic groups adapted to freshwater in the United States. Miller et al. (1989) summarized contemporary knowledge of extinction causes for 19 species and 11 subspecies of fish that once lived in the United States. Since then, extant populations of one species (Miller Lake lamprey, *Lampetra minima*) and one subspecies (*Gila bicolor isolata*) have been rediscovered. Both taxa were found at sites that had not been among suspected habitats. The longjaw cisco (*Coregonus alpenae*), also listed as extinct by Miller et al. (1989), has since been reclassified as indistinguishable from another extant species in the genus (Robbins et al. 1991). More than making up for these 3 taxa, however, 5 taxa have been added to the list (Table 9), including the Santa Cruz pupfish (*Cyprinodon arcuatus*), shortnose cisco.

Table 9. Summary of the stated causes for threatened and endangered status of species listed under ESA protection (from Matthews 1990, 1992, 1994). More than one threat may be identified for a single species.

Cause of Endangerment	Amphibians	Fish	Mussels	Snails	Crayfish	Total
Land Development						
Water depletion[1]	2	18		5	1	26
Pollution/Contamination	3	35	50	9	4	101
Sedimentation[2]		21	45	3		69
Small dams/channels		16		2	1	19
Other Habitat causes[3]		8				8
Over-fishing/bait use			15		1	16
Keystone species loss					1	1
Nonnative Species[4]		49	5	4	1	59
Collecting				1		1
Natural causes	1		15			16
Not Large-scale Water Development	6	147	130	24	9	316
Large impoundments[4]		43	51	6		100
Channelization		1	24			25
Dredging[6]		1	27	1		29
Water Diversion		10	2	5		17
Large-scale Water Development	0	55	104	12	0	171
Total	6	202	234	36	9	487

1. Water depletion is caused by groundwater draw-down or pumping from spring sources.
2. Sedimentation includes sources from coal mines, agriculture, livestock and forest cutting.
3. Other habitat causes include fragmentation, mining, land conversions and recreation.
4. Nonnative species causes include hybridization, predation, competition and grass carp.
5. Large impoundment effects include downstream impacts on hydroregime and water quality.
6. Dredging includes navigation maintenance.

(*Coregonus reighardi*), Siskwit Lake cisco (*Coregonus bartlettii)*, Maryland darter (*Etheostomus sellare*), and the High Rock Springs tui chub (*Gila bicolor spp.).* Table 10 summarizes the most probable habitat and causes of species decline for the fish taxa now believed to be extinct**.**

Fish Extinction in Isolation. Much like the extinction of the Las Vegas leopard frog, the majority of extinct fish lived in isolated habitats that were destroyed or invaded by destructive predators and competitors, and by closely related species that formed hybrids with them.

The only record of extant whiteline topminnows was in 1889, in its only known habitat, a developed spring in Huntsville, Alabama. The spring pool was routinely drawn down to clean it of debris and otherwise maintain it (Ono 1983, Miller et al. 1989). The last of the topminnows probably was killed during one of these events, perhaps decades after the topminnow was last observed. Nonnative species may have played some role in the species' decline. Carp and goldfish (*Crassius auratus*) were stocked widely by 1889 and were present in the spring in subsequent surveys.

A large fraction of the extinct taxa was found in western freshwater springs, watersheds or lakes generally isolated from invasion by other fish taxa and thereby protected from hybridization, competition and predation. Eight taxa appeared to have been driven to extinction primarily by competition from, predation by and/or hybridization with introduced nonnative taxa. These include the yellowfin cutthroat trout (*Oncorhynchus clarkii macdonaldi*), Alvord cutthroat trout (*Oncorhynchus clarkii alvordensis*), Ash Meadows poolfish (*Empetrichthys merriami*), Maravillas red shiner (*Cyprinella lutrensis blairi*), Pahranagat spinedace (*Lepidomeda altivelis*), Santa Cruz pupfish (*Cyprinodon arcuatus*) and Tecopa pupfish (*Cyprinodon nevadensis calidae*), the San Marcos gambusia (*Gambusia georgei*) and the High Rock Springs tui chub (*Gila bicolor spp.*). The two trout species probably hybridized with stocked rainbow trout (*Oncorhynchus mykiss*). The gambusia hybridized with the western mosquito fish (*Gambusia affinis*). Warmwater sportfish in the sunfish family were responsible for some extinctions.

Other taxa were extirpated by a combination of biological and physical changes. The High Rock Spring tui chub (*Gila bicolor ssp.*) lived in isolated springs where they apparently succumbed to invasion by a nonnative species in one spring and groundwater drawdown for agricultural purposes in two other springs. The invading species was a tilapia (*Oreochromis mossambica*) that escaped from aquaculture permitted in the same limited drainage system. The Clear Lake splittail (*Pogonichthys ciscoides*), Utah Lake sculpin (*Cottus echinatus*) and Utah Lake June sucker (*Chasmistes liorus liorus*) occurred in isolated lake habitats and disappeared following nonnative species introduction and water quality changes from agricultural runoff. Agricultural pollution (eutrophication, toxic contaminants and sediments) and competition probably caused the June sucker to fall to such low abundance it hybridized with another native sucker subspecies, eliminating the old subspecies and forming a new one.

The role of nonnative taxa is uncertain for several other species that were most clearly impacted by habitat destruction, including groundwater withdrawal, from agricultural and/or urban development. These included the Pahrump Ranch poolfish (*Empetrichthys latos pahrump*), Grass Valley speckled dace (*Rhinichthys osculus reliquus*), Las Vegas dace (*Rhinichthys deaconi*) and Raycraft Ranch poolfish (*Empetrichthys latos concavus*).

Table 10. Last date observed and causes attributed to extinction of fish species (sp) and subspecies (spp) (data from NatureServe Explorer 2005).

Species/ Subspecies	Date Last Seen	Location	Habitat	Associated Extinction Causes: Fishing	Public Project	Nonnative fish	Agriculture/ urban
whiteline topminnow (sp)	1889	AL springs	isolated springs	no	city	Competition?	no
harelip sucker (sp)	1893	Midwest rivers	pools	no	no	No	sediment
yellowfin cutthroat trout (ssp)	1910	CO lakes	isolated small lakes	?	no	hybridization	no
Snake River sucker (sp)	1927	WY river	River/lakes?	no	BOR	No	no
Utah Lake sculpin (sp)	1928	UT lake	isolated lake	no	no	Predation	pollution
Utah Lake June sucker (ssp)	1930	UT lake	large isolated lake	no	no	Predation/ hybridization	pollution
Silver trout (sp)	1930	NH lake	Isolated lake	sport	no	hybridization	no
Pahranagat spinedace (sp)	1938	NV closed basin	isolated spring and small lake	no	no	competition & predation	no
Alvord cutthroat trout (ssp)	1940	NV & OR	Isolated streams	?	no	hybridization	no
Las Vegas dace (sp)	1940	NV	isolated spring	no	no	?	physical alteration
Grass Valley speckled dace (ssp)	1950	NV	isolated spring	no	No	?	physical alteration
Ash Meadows poolfish (sp)	1953	NV	isolated springs	no	no	Predation	physical alteration
Maravillas red shiner (ssp)	1954	South TX	isolated springs	no	no	Competition	no
Deepwater cisco (sp)	1955	Great Lakes	large lakes	food	NY, Can.	competition/ hybridization	no
Scioto madtom (sp)	1957	Big Darby, OH	medium river	no	no	?	Sediment/ pollution?
Thicktail chub (sp)	1957	Central CA	lakes, wetlands and backwaters	no	BOR, COE	competition & predation	drainage & filling
Pahrump Ranch poolfish (ssp)	1958	NV basin	isolated springs	no	no	competition & predation	water pumping
Raycraft Ranch poolfish (ssp)	1960	NV basin	isolated spring	no	no	?	filling & pumping
Bluntnose Shiner (ssp)	1964	Rio Grande NM & TX	riffles and pools	no	BOR	predation & competition	Irrigation diversion
Siskiwit Lake cisco (sp?)	1966	Isle Royal	isolated lake	?	no	?	?
Lake Ontario kiwi (ssp)	1967	Lake Ontario	large lake	food	no	competition	pollution
Blackfin Cisco (sp?)	1969	Great Lakes	large lakes	Food	NY, Can	competition	no
Clear Lake splittail (sp)	1970	CA lake	isolated lake	No	no	predation & competition	pollution & diversion
Blue pike (ssp)	1971	Lake Erie	large lake	food	no	hybridization	pollution
Santa Cruz pupfish (sp)	1971	AZ Spring	isolated spring	No	no	Predation	no
Tecopa pupfish (ssp)	1971	CA spring	isolated spring	No	no	hybridizing	no
Amistad gambusia (sp)	1973	TX spring	dam flooded it	No	IBWC	hybridization	no
Phantom shiner (sp?)	1975	Rio Grande	riffles and pools	no	BOR. COE	competition, hybridization	irrigation & pollution
Diamond Valley speckled dace (ssp)	1978	NV springs	isolated springs	no	no	?	?
San Marcos gambusia (sp)	1983	TX spring	isolated spring	no	no	hybridization	pollution
Shortnose cisco (sp)	1985	Great Lakes	large lakes	Food	NY, Can	competition/ hybridization	no
Maryland darter (sp)	1988	MD streams	riffles	No	utility	?	pollution
High Rock Springs tui chub (spp)	1990s	NV/CA springs	isolated springs	No	no	predation & competition	aquaculture

The Diamond Valley speckled dace, *Rhinichthys osculus ssp,* was restricted to a few Nevada springs. The cause of its extinction is unclear. Similar to other western fish taxa restricted to small spring-fed stream habitats in the Great Basin region, it may have succumbed to introduced species or to habitat changes caused by agriculture development.

The ranges of two northeastern species believed to be extinct were each restricted to a single cold-water lake. The silver trout, *Salvelinus agassizi*, lived in New Hampshire. The suspected causes for extinction include sportfishing, competition with introduced yellow perch (*Perca flavescens),* and competition and hybridization with stocked brook trout (*Salvelinus fontinalis*). The Siskiwit cisco (*Coregonus bartlettii*) occurred only in Siskiwit Lake on Isle Royale in the middle of Lake Superior. The cause of extinction is a mystery. Siskiwit Lake is nearly pristine in a wilderness watershed protected by National Park status since 1940. It gets limited recreational use. The species has not been observed since 1966, when 18 individuals were netted. More effort is needed to verify its extinction status and validate its taxonomic distinction.

The extinction of four cisco taxa and a walleye subspecies occurred in the Great Lakes. While their ranges were relatively large, in many respects these species had been just as isolated for thousands of years as those restricted to much smaller habitats. These include the Deepwater cisco, (*Coregonus johannae*), the blackfin cisco (*Coregonus nigripinnis*), the shortnose cisco (*Coregonus reighardi*), the Lake Ontario kiyi (*Coregonus kiyi orientalis*) and the blue pike (*Sander vitreus glaucus*). Excessive commercial fishing probably contributed to the decline of all 5 taxa and their ultimate loss as a consequence of hybridization with more common, closely related taxa.

For several of the ciscoes, habitat invasion by the sea lamprey and the alewife also contributed to the decline (Smith 1968). The lakes had remained isolated from invasion by east coast species since they formed after the last glacial retreat. Construction of the Erie and Welland canals allowed the alewife and the sea lamprey to invade the Great Lakes from the east coast. The alewife is a formidable cisco competitor and the sea lamprey probably preyed on them as larger native salmonid species declined. The blue pike became extinct after commercial fishing and eutrophication reduced the population abundance enough to cause hybridization with the much more common walleye (*Sander vitreus vitreus*) (Miller et al. 1989).

Fish Extinction in Open Systems. The harelip sucker, *Moxostoma lacerum*, was a widespread and common species last observed alive in 1893. It apparently succumbed to chronic suspensions of fine clay, bottom sedimentation and, possibly, oxygen depletion and pollution effects (Trautman 1981,Ono et al. 1983, Miller et al. 1989). It was widespread and abundant in pools of the Ohio, White, and Maumee river systems as late as 1876 (Lee et al. 1980, Ono 1983), then declined rapidly over the next few decades. It was captured in clear, warmwater rivers of medium to large size with substrates free of fine sediments (Trautman 1981, Ono et al. 1983), where it fed on snails and other invertebrates.

The sucker declined after widespread deforestation and agricultural development increased variation in river flow and accelerated sheet and channel erosion. Trautman (1981) believed that the small mouth and “closely bound gill covers must have been particularly susceptible to asphyxiation” by fine particles in suspension. Trautman (1981) observed many massive kills of other sucker species in the genus *Moxostomus* that he attributed to silt runoff from agricultural fields. The harelip sucker probably depended on sight to find its prey (it had exceptionally large eyes) and probably could not tolerate chronic turbidity (Troutman 1981, Ono et al. 1983). Increasing municipal and industrial pollution may have contributed to its demise. Extreme drought in the 1880s/90s probably exacerbated the impacts of these habitat changes and may have been a key factor in the apparent rapidity of decline toward extinction.

Two other open-system species became extinct more recently. Both species were much less widespread than the harelip sucker; known only from a few riffles of medium-size, warmwater streams. Whether or not they were more widespread before European settlement is unknown. They include the Scioto madtom (*Noturus trautmani*), which is documented only for the lower Big Darby Creek in south central Ohio, and the Maryland darter (*Etheostoma sellare*), which was documented in three small coastal streams tributary to the lower Susquehanna River and Chesapeake Bay.

The Scioto madtom has not been observed since 1957. Only 18 specimens were collected, between 1943 and 1957, all in one small section of Big Darby Creek (Trautman 1981). Nothing is known about the original range or how common it may have been, but was probably uncommon well before its discovery. Its possible extinction has not been linked clearly to any cause. Other fish species are known to be sensitive to accumulations of fine sediments in Ohio (Trautman 1981), as well as other members of this riffle inhabiting genus. The fauna of Big Darby Creek is now relatively diverse, but, much like other midwestern streams, it experienced significant turbidity and sedimentation from erosion starting in the 19th century when much of the watershed was converted to agricultural use. Agricultural use of the watershed has declined. At least some fine sediment had accumulated in the river in the 1940s (Trautman 1981) and it remains the most evident source of habitat alteration.

The Maryland darter seemed already scarce in 1912 when it was first collected from riffles in a coastal plain stream draining into Chesapeake Bay near the mouth of the Susquehanna River (Ono et al. 1983). Despite subsequent sampling, it was not rediscovered until 1962 and was last observed in 1988. The range may have been limited by rising saline waters in Chesapeake Bay following the last glacial retreat. Two of the streams where it was found drained directly into the Bay, blocking the species' movement. However, the last known stream inhabited by the darter (Ono et al. 1983), Deer Creek, entered directly into the fresher waters of the Susquehanna River just upstream from the Bay. The darter's range may have been restricted since1928 to a few downstream tributaries of the Susquehanna River by Conowingo Dam (built and operated by a Philadelphia power company) but there is no evidence that it once lived anywhere upstream.

For over a century, the three known habitats of the darter have been extensively exposed to agricultural and urban impacts, including a number of small, locally constructed impoundments. Summing up threats to the remnant population, Ono et al. (1983) stated: "Prolonged periods of high turbidity, silt, impoundments, pesticide and herbicide-runoffs, reduction of stream flow for consumptive uses, construction projects, and waste from sewage treatment plants all pose problems for the Maryland darter population."

Fish Extinction Associated with Public Water Resources Development. Nine extinctions of fish species and subspecies of the 33 (27.2%) now listed as extinct by NatureServe Explorer can be linked in part to large water-development projects financed by state, provincial and Federal agencies (Table 10). Four of those taxa are Great Lakes ciscoes. Canal development sponsored by state and provincial governments and private sources opened the lakes to invasion of alewife and sea lamprey. Neither of the projects were sponsored by the Federal government of the United States.

Federal water resources development has been linked to extinctions of five fish taxa native to western waters (15.1%). Four of the extinctions have been linked to BOR projects and to associated agriculture-caused water diversion and wetland destruction. Nonnative species introductions and invasions are believed to have contributed to their decline.

The causes of the Snake River Sucker (*Chasmistes muriei*) decline are conjectural. Little is known about the distribution of the sucker, which was found only once, in 1927, in the Snake River below Jackson Lake Dam (Lee et al. 1980, Miller 1989). When completed in 1916, the dam raised the level of water by 17 feet over a series of natural lakes on the Snake River. Because other members of this genus are lake inhabitants, Scoppettone and Vinyard (1991) hypothesized that the Snake River sucker inhabited the natural lakes before they were inundated by Jackson Lake Dam. The dam may have allowed fish to move downstream from the reservoir, but blocked return to reproductively suitable habitat. It was not identified as a distinct species until 1981.

Data presented in Sublette et al. (1990) indicate that a number of fish species have been locally extirpated in the main-stem middle Rio Grande of New Mexico and two endemic species are now globally extinct. The locally extinct species include the grey sucker (*Moxostoma congestum*), blue sucker (*Cycleptus elongatus*), and shortnose sturgeon (*Acipenser brevirostrum)*, which apparently declined sharply before the era of big dam development. Sublette et al. (1990) indicate that much of the middle Rio Grande of New Mexico was completely dewatered as a consequence of irrigation diversion in the mid to late 1800s, decades before the first Federal dam was built at Elephant Butte in 1916.

The Rio Grande bluntnose shiner (*Notropis simus simus*) and closely related phantom shiner (*Notropis orca*) were last observed alive in 1964 and 1975 after more than a century of extensive water resources development for irrigation on the middle Rio Grande of New Mexico (Sublette et al. 1990). Water diversion for irrigation, starting in the 1800s and augmented by BOR in the 20th century, probably contributed most to these declines throughout the main Rio Grande and over 1000 miles of habitat. Based on survey results depicted in Sublette et al. (1990), the bluntnose shiner was widely dispersed in the middle Rio Grande above and below Elephant Butte Reservoir between 1901 and 1950, and rapidly declined thereafter. The two taxa declined after completion of Elephant Butte, Caballo and El Vado reservoirs were completed by the BOR between 1916 and 1935 and increased diversion of water from the main river channel to agricultural use thereafter. These species also faced additional predation following introduction of large mouth bass (*Micropterus salmoides*), Channel catfish (*Ictalurus punctatus*) and Walleye, starting early in the 20th century.

BOR activities also impacted habitat of the thicktail chub (*Gila crassicauda*), which once lived throughout much of the Sacramento and San Joaquin rivers, mostly in backwaters, lakes and wetlands at lower elevations in California's Central Valley. Last seen alive in 1957, the loss of this species is linked to agricultural filling and draining of small lakes and wetlands, and water channeling and diversions facilitated by the BOR and state agencies. Introduced nonnative species also competed with and preyed on the chub (Moyle 2002).

Extinction of a fifth species is associated with a project planned and built under the authority of the International Boundary and Water Commission (IBWC). Impoundment of water behind Amistad Dam was the root cause of extinction of wild Amistad gambusia (*Gambusia amistadensis*). This gambusia was not identified as a unique species until after its only habitat, a small spring-fed tributary of the Rio Grande, was inundated by reservoir impoundment in 1969. Habitat inundation probably dispersed gambusia population members and exposed them to intolerable predation, competition and/or hybridization with western mosquito fish (*Gambusia affinis*). A fortuitous collection of live fish from the habitat just before it was inundated provided a captive population for maintenance by the Fish and Wildlife Service at Dexter, New Mexico and the University of Texas. Both captive populations became contaminated with the western mosquito fish, which hybridized with the Amistad gambusia, causing its total extinction (Miller et al. 1989).

The Corps of Engineers may have played an incidental role in the extinction of three of these fish species (9% of extinct freshwater vertebrates). The Corps dredged the lower Sacramento-San Joaquin rivers for navigation improvement, which may have negatively impacted a small fraction of thicktail chub habitat. It also completed two impoundments in the upper Rio Grande in 1963 and 1975 that may have contributed to the final demise of the Rio Grande bluntnose shiner after1964 and the phantom shiner after 1975. Implication of the Corps with respect to the Amistad gambusia at Amistad Reservoir appears to be the result of confusion over agency authority. A Federal Register (Vol. 45, No. 85:28721) notice of intent to list the Amistad gambusia as endangered referred twice to the "Corps of Engineers Amistad dam" even though it was built and operated under the auspices of the International Boundary and Water Commission.

Present imperilment of fish is believed to be caused mostly by the same agents as those implicated with extinction (Table 8). Consistent with past extinctions, the primary reason for identifying nearly half of the critical imperilment of fish as "small range" is linked to tenuous protections against threats to fish in small habitats, such as small springs surrounded by private development, groundwater depletion, climate change or other changes that pervade the limited range. Also consistent with past water resources impacts, 18% of the imperilment was associated with large reservoir, hydroregime, and water diversion impacts. Federal and nonfederal projects were not separated. When secondary and tertiary factors are included, large water resources projects are indicated 12% of the time. However, for those species threatened by specified causes, about 22% were associated with large water resources projects. These are slightly lower percentages than the contribution of water resources development to past extinction.

Based on geographical setting of species ranges, more imperilment from water resources development is associated with projects built and operated by TVA, BOR or private utilities than by the Corps. Imprecise specification of cause in the reviewed information may result in an underestimate of Corps involvement, however. Over 13% of listed causes were under the general category of habitat degradation, which could include some navigation and flood control dredging and channelization, as well as scouring, bank erosion, sedimentation and other results of intense and widespread land development.

Many imperiled species are small riffle inhabitants in three genera (*Etheostoma, Percina, Noturus*). The combined effect of river impoundment has greatly reduced the available habitat for many of these species. Impoundment also impedes recolonization of habitats that have become depauperate. Including some influence from channelization and dredging, the Corps may be contributing to the imperilment of a significant number of these species in medium to large rivers, but the majority of threats appear to be associated with agricultural water diversion, sedimentation, and eutrophication; urban pollution; and nonnative species predation and competition.

Of ESA listed species, about 27% of the causes listed for threatened and endangered status involve large-scale water resources development (Table 9). This fraction is slightly larger but similar to the fractions implicated with past extinctions and with the critical imperilment indicated by NatureServe Explorer. Many of these projects are Federal impoundments, especially where western fish species are threatened, but exact figures are not available. A number of the listed eastern species are influenced by Corps project operations. The greatest number of fish species are threatened by nonnative species and by pollution, sedimentation, small dams, channels and other habitat changes associated with agriculture and, to a lesser extent, with urban development. A small fraction of the threat is associated with sportfishery management.

Crayfish

Taylor et al. (1996) considered three crayfish species extinct since the 1860s, one of which has since been discovered alive. One extinct species burrowed to groundwater for its refuge and reproduction. Its disappearance is most associated with agricultural and other land development. Another crayfish species was last observed in the 1860s and probably died out as a consequence of urban development and introductions of nonnative crayfish. Neither extinction is linked to Federal water resources development.

Stein et al. (2000) identified 107 species as critically imperiled or imperiled species, identified as such mostly because of limited range. Many of the critically imperiled cave and spring species do not appear to be immediately threatened at this time. The generic concern is degradation of groundwater quality from agriculture and other sources and introduction of nonnative species, especially other crayfish (Lodge 1993). The security of several less vulnerable species may be loosely linked to Corps impoundments, which are believed to fragment existing ranges and reduce genetic flow. The effect of fragmentation is speculative in many specific situations, however, because of insufficient research attention.

At least in part because many imperiled species seem not to be immediately threatened, only 4 crayfish species are listed as endangered under the ESA. The leading threats include water contamination from agriculture and urban development. Other threats include exotic species (especially more aggressive crayfish species and crayfish-eating fish), localized physical alteration of habitats and local extirpation of bats, which provide guano-based food sources in caves.

Freshwater Mussels

The freshwater mussels in the family Unionidae are among the most threatened of major taxonomic groups (Neves et al. 1997, Strayer et al. 2004). Of the 36 presumed or possibly extinct species listed by NatureServe Explorer in 2005 for the United States, 83% are in 5 genera and 61% in 2 genera; more specifically, 14 in *Epioblasma*, 8 in *Pleurobema*, 3 each in *Alasmidonta* and *Quadrula*, and 2 in *Lampsilis*. One extinction each apparently occurred in 6 other genera, including one species that probably died out early in the 19th century. Even more so than fish, questions remain about the taxonomic status of a number of extinct mussel species, some of which may prove to be subspecies or even ecological phenotypes. *Epioblasma* and *Pleurobema* are among the most species diversified genera (they include nearly 20% of all extant species in the United States), but also are among the more difficult to differentiate taxonomically.

Mussel losses have clustered geographically in 6 river systems. Of 13 extinct species that lived in the Mobile River system, 12 occurred only there (36% of the total). Thirteen extinct species once lived in the Ohio River system (36% of the total). Of those species, one lived in disjunct populations in the Cumberland and White (Arkansas) rivers, six apparently were limited to the Tennessee and Cumberland rivers, and three occurred north of those rivers in the Ohio River and northern tributaries (e.g., Wabash River). Three species occurred only in the Apalachicola River system. Three other species were found in the Rio Grande (one of the species was also found in central Texas rivers). One species occurred in the Pee Dee River of the Carolinas, another lived in the Ochlockonee River of Georgia and Florida, and the last—for which information is particularly sparse—lived in the Colorado River of California and Arizona. The large majority of species losses occurred in rivers with sources in the Appalachian mountains (81%) and in a belt extending between the mountains and the Mississippi River from the Great Lakes to the Gulf of Mexico (83%). As already described, the history of habitat change in these rivers is complex and began early in the 19th century.

Bogan (1993) summarized the complexity of causes for extinction among the freshwater mussels:

> "Extinction of the 18 taxa in North America is not clearly due to a single cause but is the long term combination of a suite of detrimental factors. The cumulative effects of impoundment, of dumping of municipal and industrial pollutants into the rivers, deforestation, channel modification, and over-harvesting have contributed to the extinction."

Bogan's listing of impoundments as first among causes of extinction is typical of most evaluations of mussel decline and extinction, which places substantial emphasis on impoundment impacts. For the mussels of Tennessee, Parmalee and Bogan (1998) concluded that: "The greatest overall detrimental impact on mussel populations probably can be attributed, directly or indirectly, to dam construction—especially those built in the 1930s, 1940s, and 1950s." In one of the most critical evaluations of impoundments, Watters (1999) concluded that "perhaps several dozen mussel species...were driven to extinction wholly or in large part by the construction of dams." That assertion seems to account for all of the 36 mussel species now presumed and thought possibly extinct as listed in NatureServe Explorer.

Williams et al. (1993), however, started their list of influential changes in mussel habitats with habitat loss from sediment and Bogan (1997) most definitely identified habitat sedimentation from poor agricultural and timbering practices as a "major leading factor." With respect to cumulative effects, starting with deforestation, erosion and sedimentation better fits the chronology of change in mussel habitats than major impoundments, which came much later (and were more obvious). The differences in accounts and emphasis on professional judgment in lieu of exhaustive data reflects the difficulty in separating contributing causes and the chronology of extinction. With that caveat in mind, Table 11 summarizes the extinction record and the most probable primary causes of species decline for 25 mussel taxa for which there is a reasonably precise record of last observation reported in Stein et al. (2000) and in NatureServe Explorer.

Widespread forest removal for wood and agricultural development accelerated erosion in the 19th century, including the upland piedmont regions described in detail by Trimble (1974). Sediment loading also accelerated in the Rio Grande basin with the rapid growth of cattle ranching and serious overgrazing after the Civil War (Sublette et al. 1990). Erosion rates increased dramatically in the piedmont region and adjacent areas before the Civil War and peaked at a high and continuous rate over the period after the Civil War until World War I. Some of this area had since reverted to secondary forest growth and erosion rates had returned to relatively low levels by the 1960s (Trimble 1974). Many sediment-choked rivers began to recover old channel configurations through degradation. Construction of large impoundments contributed to these changes (Trimble 1974), in the process improving downstream habitats for species sensitive to sedimentation. Even these relatively low rates of loss remain a chronic concern to state conservation agencies responsible for water quality and management of risks to sensitive species.

At least seven mussel species appeared to have declined to low abundance during the 19th century before the era of large impoundment construction. One species found in the Apalachicola River apparently died out sometime early in the 19th century from undocumented cause (Table 11). Intensive cotton farming began to accelerate erosion in this period. Based on early accounts, stream turbidity and sedimentation markedly increased between the 1820s and 1850s (Trimble 1974). One species was last seen in the Rio Grande in 1898 (it may survive in Mexico). Much of that river was impacted by irrigation diversion and sediment loading by then.

Five species of once widely distributed mussels in the genus *Epioblasma* apparently were rare by 1900 even though they persisted for sometime thereafter, based on dates of last observation reported in Stein et al. (2000) and NatureServe Explorer. In early reviews of mussel conservation status, Stansbery (1970, 1976) noted that most of these species had not been observed alive since the early 1900s or before and were most likely rare by the turn of the century. It is possible that these species were always rare, making them that much more vulnerable to environmental changes. Isolated populations of mussels may have persisted for several decades, into the 1930s and 1940s, without significant reproduction consistent with their long life expectancies. Early waterway development also occurred in these rivers and could

Table 11. Summary of events associated with mussel loss for which the last date of observation is recorded in NatureServe Explorer (2005) or Stein et al. (2000).

Mussel Species	Date	Comments
Fusconia apalachicola	early 1800s	Apalachicola River. Found in archeological sites. Extinction may have occurred sometime before or as agricultural development began.
Rio Grande monkeyface *Quadrula couchiana*	1898	Rio Grande. Sedimentation & irrigation water diversion accompanied by major drought before it disappeared. Validity of record is uncertain.
round combshell *Epioblasma personata*	1925	Ohio and Tennessee rivers. Rarely collected after 1900, it probably succumbed to sediment, shelling, and pollutants before deep canalization.
Tennessee Riffleshell *Epioblasma propinqua*	1930	Ohio, Wabash and Tennessee rivers. Its early decline included undammed areas implicating sediment, pollutants, shelling, and early canalization.
Cumberland leafshell *Epioblasma stewardsoni*	1930	Cumberland and upper Tennessee tributaries. Rarely collected after 1900, shelling, sediment and pollutants impacted before habitat was impounded.
Ochlockonee arcmussel *Alasmidonta wrightiana*	1931	Ochlocknee River. Dam obstructed loss of an anadromous fish host is conjectured and agricultural impacts were extensive.
Acornshell *Epioblasma haysiana*	1937	Tennessee & Cumberland. It declined before the big dams were built and was likely impacted by sediment, pollution, mine acid and canalization.
Leafshell *Epioblasma flexuosa*	1940s	Tennessee, Cumberland and Ohio. Rarely collected after 1900, it may have succumbed to sediments, pollution, shelling and early canalization.
sugarspoon *Epioblasma arcaeformis*	1940s	Tennessee and Cumberland. It probably succumbed to the combined effects of canalization, flood control/ hydro dams, sediment and pollutants.
winged spike *Elliptio nigella*	1958	Apalachicola. Was always scarce in collections, including two years after a Corps dam was built in 1952. Loss of fish host conjectured.
Wabash riffleshell *Epioblasma sampsonii*	1950s / 60s	Wabash River, which was never very canalized or dammed, but heavily sedimented. Probably succumbed to sediment, shelling and pollutants.
Forkshell *Epioblasma lewisii*	early 1960s	Tennessee and Cumberland. Rare by the 1940s, it probably succumbed to sediment, pollutants, canalization, and, last, flood control/ hydro dams.
angled riffleshell *Epioblasma biemarginata*	1960s	Tennessee and Cumberland. Canalization and flood control/ hydro dams contributed, but it also disappeared from areas free of those impacts.
narrow catspaw *Epioblasma lenior*	1965	Tennessee and Cumberland tributaries. Dams may have contributed, but sediment, pollutants and drought were major stresses in tributary habitats.
lined pocketbook *Lampsilis binominata*	1967	Apalachicola. Scarce since 1800s. Last seen in the undammed Flint river where sediments/ pollutants were the probable causes.
Haddleton's lampmussel *Lampsilis haddletoni*	Mid 1960s	Black Worrior, Pascagoula, Choctowhatchee. Siltation and other agricultural pollutants are best general explanation for complete loss.
turgid blossom *Epioblasma turgidula*	Mid 1960s	Tennessee system. In small tributaries where sediment, drought and pollutants were more likely widespread factors than large dams.
tubercled blossom Epioblasma torulosa torulosa	1969	Tennessee and Ohio system. Widespread, big river form last observed in the Kanawha River.
southern acornshell *Epioblasma othcaloogenis*	1974	Upper Coosa and Cahaba rivers. In its remaining habitats, dams played a role secondary to sediments and pollutants.
Mexican fawnsfoot *Truncilla cognata*	1975	Rio Grande. Pollution, sediment and water diversion are widespread stresses over the more local impacts of two Federal dams.
False spike *Quincuncina mitchelli*	Mid 1970s	Central Texas rivers and Rio Grande. Once common. Numerous dams built on central Texas rivers.
stirrupshell *Quadrula stapes*	1978	Tombigbee, Black Warrior and Alabama rivers. Its last populations were much impacted by Tombigbee waterway development.
Flat pigtoe *Pleurobema marshalli*	1980	Tombigbee River. Tombigbee Waterway development (Corps) was the most obvious stress.
green blossom *Epioblasma torulosa gubernaculum*	1984	Upper tributaries in the Tennessee River. Sensitive to water quality changes. Recently quite abundant in the Clinch River.
upland combshell *Epioblasma metastriata*	1988	Upper Coosa & Cahaba Rivers. Recent losses associated with sediment, mine acid and other pollutants. Utility impoundments contributed earlier.

have interacted with other perturbations to secondarily influence declines to extinction, especially after 1920 when nine feet was widely targeted for waterway development depth.

Rare or not, the 19th century changes in stream and river habitat in the Ohio, Mobile, Rio Grande and Apalachicola river basins are as likely to have had widespread impact on mussels as they had on the harelip sucker, which was common in the Ohio River basin before it became extinct. Extensive changes occurred in those river basins before major water resources development took place in the 20th century. Higgens (1858), as referenced in Trautman (1981) noted the extirpation of 6 mussel species and near extirpation of 10 others by 1858 in the Scioto River of central Ohio. Trautman (1981) associated these earliest mussel losses with the effects of widespread forest cutting and water-powered sawmills on soil erosion and organic loading from sawdust. In an early experimental analysis, Ellis (1936) tested a perceived relationship between widespread sedimentation in mussel habitats and mussel decline on 18 species. He observed mussel mortality in an accumulation of as little as 0.6 cm of sediment and almost 90% mortality in sediment up to 2.5 cm deep. Log rafting most likely contributed as well.

McMahon and Bogan (2001) reviewed the literature pertaining to sediment-mussel relationships and how suspended sediment and sediment accumulation on the bottom exerts its effects on mussel survival. Juvenile riffle species may be most vulnerable when they settle into fine sediments. The low pH, hypoxia and elevated ammonia concentrations associated with fine sediments is correlated with juvenile mussel mortality. Heavy species sink into the soft sediments and cannot move away from the sediments or maintain the necessary position for respiration or feeding at the sediment surface (Howells et al. 1996). All mussel species are adapted for suspension feeding and temporarily high levels of inorganic suspended sediments, but prolonged exposure to suspended solids can interfere with mussel feeding and respiration (Aldridge et al. 1997). As a consequence of these and other effects, the number of freshwater mussel species typically increases as bottom particle size increases from fine silts and clays to sand and gravel mixes.

Because the riffle species settle most successfully into bottom interstices with enough well-aerated flow-through to sustain nourishment and respiration, they are exceptionally vulnerable to sedimentation that fills those interstices. Juveniles are least mobile and least able to persist by shell closure through emersion and other stressful events or to survive scouring flows. Accelerated loading by fine sand, as most clearly happened in the Rio Grande, would embed the gravel, closing out suitable habitat for juvenile clams as well as adults. Entire generations may fail to be recruited as a consequence of juvenile mortality soon after settling to the bottom (McMahon and Bogan 2001).

In addition to sedimentation, toxic metals, petrochemicals, mine acid and oxygen depletion were implicated with mussel decline by the early 1900s in the Ohio River basin. Ortmann (1909) linked complete extirpation of mussels and other aquatic species in the upper Ohio River watershed in Pennsylvania to mine acid. In that same region, Rhoad (1899) had associated local mussel extirpation in the Monongahela River at Pittsburgh with severe pollution (probably from a combination of domestic, industrial, and coal mine inputs) and early lock and dam structures, which slowed reaeration and trapped polluted sediment. Ortmann (1918) cited damage to mussels from paper mills in Tennessee, which added toxicants in addition to chemical oxygen demand. Subsequent numerous studies confirm the impact that low oxygen concentrations often have on many invertebrates, and especially of riffle inhabitants (Hart and Fuller 1974, Brown 2001, McMahon and Bogan 2001). Wilson and Clark (1912) identified the damage caused to mussel populations by oil and gas development. They also noted early mine acid effects on mussels in the Cumberland River (Wilson and Clark 1914). High acidity (pH below 5.0) reduces the concentration of calcium in water below that necessary for shell development and maintenance (McMahon and Bogan 2001). Ortmann (1924) associated sharp decreases in mussel populations with iron and phosphate mines in the Duck River watershed. Similar forms of pollution occurred in upper tributaries of the Mobile system, especially in the Cahaba and Black Warrior rivers.

Snag removal, shoal dredging and other early water resources development contributed to erosion, sediment movement and sedimentation, but was a lesser disturbance for most habitats compared to widespread land-use impacts that increased flood flows, sheet and channel erosion, and sediment loading. Dredging no doubt killed some mussels outright in a fraction of the total riffle habitat. But the thick-shelled riffle species were able to tolerate some physical disturbance and dredging typically impacted only a portion of the channel width. Shoal dredging at any one location was often infrequent, allowing time for settlement of juvenile mussels on dredged areas as long as host species were present.

Other habitat impacts probably interacted with dredging in complex ways, which may have, to some extent, counteracted negative effects, given increased sedimentation and decreased low-flow. Dredging concentrated river flow and redistributed suitable habitat (much as natural discharge changes might do). Concentrating flow during low-flow periods may have increased the suitability of bottom habitat for mussel settlement in some locations. Wing dam construction had similar complex effects.

The early lock and dam structures undoubtedly slowed average velocity as they increased depth, but generally retained riverine attributes. Low-head lock and dam structures slowed average velocity through the impoundment and accelerated sedimentation there, but eroded it from the tailwater. Depending on how much sediment loading occurred in the river and where it originated in the basin, the net change in suitable substrate may have been small in some locations. Lock and dam structures probably increased fine sediment storage in the system, but primarily redistributed fine sediments causing some areas to accumulate sediment while eroding other areas clean. In some situations, they may have stabilized substrates too unstable to support mussel populations. Many of the now extinct mussel species ranged well beyond the upstream limits of navigation improvement into tributaries where they also disappeared.

Seven species later disappeared from rivers that drained watersheds extensively impacted by poor land-use practices and had relatively little development of large impoundments. These included species last seen in the Wabash River in the 1950s, the Flint River of Georgia in 1967, the Cahaba river of Alabama in 1974 and 1988, the Pascagoula and Choctowhatchee of Mississippi and Alabama in the 1960s, and the Texas Rio Grande in the mid 1970s (two species). All of these rivers were impacted by increased sediment loading and/or urban-industrial pollutants. These rivers were not, for the most part, improved for navigation after the paddle-wheel era. Two large reservoirs had been built along the Rio Grande in Texas by the

International Boundary and Water Commission in the 1960s, but hundreds of miles of river habitat were minimally affected by them. One of the mussel species that died out along the Rio Grande also disappeared from other rivers in southern Texas, some with extensive dam construction on them. While large impoundments occurred within the ranges of most of these species, each species also disappeared from river reaches without much reservoir influence.

Several studies compared the number of species in early surveys with later surveys in rivers with and without intense water resources development. In the Illinois River, Starrett (1971) noted a drop in mussel species number from 45 in 1906-1912 to 24 species in 1966-1969. This river was transformed to a deep waterway during that period, but also underwent severe pollution (Starrett 1971). Other mussel declines occurred in rivers that, at the time, were without significant water resources development. For example, mussels underwent "drastic decline" in the Big Vermillion River (Matteson and Dexter 1966) and declined in the Kankakee River from 29 species in 1906 to 15 in 1980 (Suloway 1981). Isom and Yokley (1968) reported 48 species in the Duck River compared to 63 species reported earlier in the century by Ortmann.

A species last seen in the Ochlockonee River in 1931, may have been the first loss to be linked to the construction of a specific dam. The mussel lived upriver from an impoundment built a few years beforehand by local government. The dam may have disrupted a host species' migration, which stopped mussel reproduction. More definitely, the impoundment flooded the river area harboring the last known populations, probably contributing to their demise. But the already limited distribution of the species at the time of flooding suggests other limiting stresses existed at the time the dam was built. Like other Gulf Coast river floodplains and watersheds, forest cutting and agriculture became widespread starting in the 19th century. Watershed erosion and sedimentation remain significant and chronic in the basin.

After an abrupt increase in dam construction following World War II, four mussel species and one subspecies in the genus *Epioblasma* disappeared from the upper Tennessee and Cumberland rivers during the 1950s and 1960s. The four species were relatively rarely encountered in surveys (Parmalee and Bogan 1998), but the subspecies was widespread and common in larger rivers. The four species probably died out from a combination of stresses including sedimentation, pollution, canalization and, lastly, impacts by Federal multipurpose dams built on the last tributary streams harboring these species, such as J. Percy Priest and Center Hill reservoirs built by the Corps in the Cumberland River basin, and Normandy Dam built by TVA. These large dams probably were the final impact added to the cumulative impact on suitable habitat.

Two other species disappeared from habitats where impoundments probably contributed to the environmental stress, but the connections to extinction are less clear. One species had been scarce for decades before it disappeared from the Apalachicola River sometime after 1958, after decades of dredging and agricultural development before that. It persisted in a silt-free, spring-fed tributary, consistent with possible sediment impacts, but disappeared after completion of Jim Woodruff Dam (a large lock and dam structure completed by the Corps in 1952). The dam is hypothesized to have blocked migration of a critical anadromous host species. Another mussel species once occurred in the Coosa River where a string of private hydropower dams probably contributed to its extinction, but pollution and sedimentation in the upper Coosa River may have been the most immediate cause. Several of the impoundments are secondarily managed by the Corps for flood control purposes.

Two mussel disappearances after 1978 were associated with development of the Tombigbee Waterway by the Corps of Engineers in Alabama and Mississippi during the early 1970s. This was the last major waterway development to take place in the United States. It confirmed the degree of impact that the last phase of waterway development can have on riffle mussels with distributions mostly limited to the impacted river habitats.

One other species disappeared relatively recently from upper tributaries of the Tennessee River. While Federal multipurpose reservoirs destroyed much habitat in the upper Tennessee river, the subspecies inhabiting those small to medium rivers more likely succumbed to sediment and pollution stress from mines and urban-industrial centers.

Numerous studies document the impacts of reservoirs on mussels (Watters 1999). Several studies of mussel communities before and after impoundment confirm that impoundments could locally reduce the number of mussel species by half or more (Neves et al. 1997), just as combined pollution and sediment impacts did in studied river reaches before many impoundments were built (e.g., Troutman 1981). In this analysis, large reservoirs probably contributed to the immediate extinction of 8 out of 25 taxa (32%), having impacted the last suitable habitat. The role of reservoirs in the extinction of three other species (12%) is probable but less clear because of insufficient data. A total of 14 species (56%) died out before major impoundment development or substantially independent of it in undeveloped river reaches. Corps dams were important in 7 extinctions (28%). TVA dams were implicated in three (12%) extinctions, and private utilities in one extinction (4%). Both Federal and private impoundments contributed indirect impacts by reducing the range of species that died out primarily from other causes. This included at least 7 species (28%).

McMahon and Bogan (2001) indicate there is little substantiation of impact by invasive species on scarce native species so far. However, the findings of Schloesser et al. (1996) indicate the potential for mussels in the zebra mussel family *Dreissenidae* to decimate lake species. The recent rapid spread of nonnative zebra mussels into the Mississippi River system is a new threat of uncertain effect (Ricciardi et al. 1998), although this exotic mussel appears to be adapted more to lake habitats (McMahon and Bogan 2001) than to the riffle habitats where the most vulnerable mollusk species now live (Neves et al. 1997). They have yet to become abundant enough in the riffle habitats of rare mussels to be viewed as an important threat, but may become a threat to some of the species tolerant of reservoirs and other slack-water conditions that dreissenids favor, especially in the more northerly areas where waters do not warm beyond their tolerance.

The cumulative impact of all environmental stress on mussels may have peaked just before actions were taken to comply with the Federal Water Pollution Control Act of 1972, the ESA of 1973, and other environmental laws passed during that period. Early sedimentation, hydrologic changes and pollution contributed importantly to the losses of most extinct mussel species. These were the predominant stresses likely most responsible for the loss of about 60% of the species and subspecies with last dates of observation (Table 9), many of which were already scarce by the early 20th century. Reservoir construction and waterway canalization contributed prominently to the extinction of about 40% of the extinct mussels. In the context of uncertain information, these estimates must be considered approximate.

Also uncertain is the extent to which the extinction history of taxa with last dates of observation is representative of all extinctions. Many of the mussel taxa without dates of last observation are in the genus *Pleurobema*, native to the Mobile River basin. Only one species in this genus is among those with recorded dates of last observation. This genus is under taxonomic review and some may prove to be clinal variations of a smaller number of extinct species or extant species (NatureServe Explorer 2005). Of those believed to be extinct, none have as yet been confirmed to have persisted as discrete taxonomic units much beyond the 19th century. Because the Mobile River system seems to have suffered from severe hydrologic and sediment loading stresses in the 19th century as a consequence of dramatic watershed changes (e.g., Jackson 1995), some of these species may have succumbed before the extensive waterway development by the Corps and hydropower development by private utilities, most of which occurred in the mid 20th century.

On the other hand, many of the *Pleurobema* mussels without records of last observation may have persisted until more recently, implicating water resources development (mostly private utility development of hydropower) as an important cause for their loss. The same might be said about the loss of the Coosa elktoe (*Alasmidonta mccordi*) from the Mobile system. Losses of species in California (*Anodonta dejecta*) and the Carolinas (*Alasmidonta robusta*) are much more likely to be associated with activities of the BOR in the west and private utilities in the east than the activities of the Corps of Engineers. However, the loss of *Epioblasma cincinnatiensis* from the Ohio River basin may have been impacted by Corps waterway development, if it did not succumb earlier from sedimentation and pollution.

Regardless of past causes for decline and extinction, dams built for navigation, flood control, hydropower and water supply purposes now play a pivotal role in the management of imperiled mussel species. Their effects dominate the riverine habitats of many imperiled mussel species (e.g., Neves et al. 1997, Watters 1999). While they are implicated in imperilment of only 14% of the identified causes in Table 8, this low percentage may be misleading because the absence of proper fish hosts is implicated 33% of the time and dams are the primary reason given for the absence of fish hosts.

There can be little doubt that a combination of impacts from multipurpose dams and waterways developed for commercial navigation now contribute largely to the exclusion of many imperiled species from once suitable river habitat. The Corps is the primary Federal agency involved in past reservoir development and present management in mussel habitats, mostly through navigation impoundments but also through flood control and multipurpose reservoirs. The TVA plays a dominant role in the Tennessee River. The BOR is the Federal presence in the Colorado River basin. The Natural Resources Conservation Service (previously the Soil Conservation Service) maintains many small dams and channels many small rivers to reduce agricultural flooding. Numerous private impoundments have been constructed primarily for hydropower generation in the Alabama, Santee and Pee Dee river basins. The limiting effect of impoundments and potential for improvement, combined with continued dredging and canalization, places the Corps prominently among the Federal agencies contributing to the fate of imperiled mussel species.

Navigation pools behind lock and dam structures have affected the greatest total length of river habitat once suitable for mussels. Navigation pools typically cause sedimentation of required riffle and shoal habitat, and elimination of intolerant mussel species. However, the water exchange rate in most navigation pools is too high for significant alteration of water temperature or oxygen concentration in river discharges downstream. Discharges tend to erode and transport very fine sediments downstream (often to the next impoundment) thereby creating suitable habitat for some rare mussels in some locations (Parmalee and Bogan 1998). Other impoundments may have less extensive impacts overall, but deep ones with low water exchange rate often reduce oxygen and summer temperature significantly for miles downstream (Thornton et al. 1990). Deep reservoirs often serve a hydropower purpose that causes water fluctuations, which often reduce suitable habitat to the low-flow width of the channel.

Since the 1930s, development of deep navigation waterways undoubtedly eliminated much suitable habitat for riffle and shoal mussels in the Ohio, Mississippi, Illinois, Green, Cumberland, Tennessee, Mobile, Arkansas, Ouachita, Apalachicola and other river systems. Single-purpose flood control reservoirs are widespread, but typically are smaller and have less impact downstream than large multipurpose reservoirs managed to maintain deep storage. Multipurpose reservoirs often maintain cold summer temperatures and oxygen-depleted waters a significant distance downstream in tailwaters. Concentrations of Corps-operated multipurpose reservoirs occur on the White, Arkansas, Missouri, Apalachicola, Savannah and Cumberland rivers, and scattered over much of the remainder of the eastern U.S. TVA operates a number of large multipurpose reservoirs in the Tennessee River system. TVA has done much to improve oxygen and hydroregime in the tailwaters of its reservoirs (Parmalee and Bogan 1998).

The impacts of most mussel habitat alteration are accentuated by drought, which can be moderated by impoundments. Some dam construction during the mid 20^{th} century was justified in part to supply low-flow supplements for water quality improvement and to maintain flows during low-flow periods. Most of the environmental factors least tolerated by the extinct and imperiled mussels were intensified by drought. Prolonged drought has occurred at roughly twenty-year-intervals since the late 1880s in the midwestern United States. Droughts caused normally perennial stream habitats to dry up for extended periods and significantly reduced the capacity of rivers to moderate the effects of sediment loading, oxygen depletion, toxins, summer warming and winter freezing.

Sixty-one freshwater mussel species are now listed as endangered or threatened under protection of the ESA. The reasons given for listing (Table 9) include 44% associated with large-scale water development. The effects of large impoundments are the most commonly identified, but channelization and dredging are also identified as significant threats. More than half of the identified threats are independent of water resources development. Most commonly, impoundment, water contamination and sedimentation are identified together as threats without much indication or knowledge of the relative contributions of each. Because impoundments can both exacerbate and mitigate for the effects of sedimentation and pollution, specific understanding of those interactions is critical for each species.

Not all of the environmental changes described here interacted additively to cause negative cumulative impact on mussel habitats. A few important mussel habitats have been sustained by dams. Many reservoirs were built in part to counteract the effects of hydrologic variability to sustain water supplies for navigation or other purposes, which sometimes sustain mussel habitats downstream. Large impoundments often are effective traps for sediments, eutrophying nutrients and other pollutants, but also often counteract these positive features by releasing oxygen deficient water of invariably cold temperature that impede mussel survival and reproduction in otherwise suitable habitat. These attributes sometimes provide opportunities to manage presently harmful effects of dams more positively for endangered mussel recovery (Parmalee and Bogan 1998).

Invasive nonnative species have been among the more recent of environmental stresses layered upon freshwater mussels and are difficult to implicate with any past extinction. They may contribute to present and future imperilment, however. There is little evidence that Asian clams are a threat in unmodified native habitats (McMahon 1991). The equally invasive zebra mussel (*Dreissena polymorpha*) is spreading rapidly, which is worrisome for the future of numerous mussel species (Ricciardi et al. 1998), especially in marginally acceptable habitats. However, zebra mussels appear to be best adapted to locations other than those favored by riffle species, most typically on hard substrates in the deeper waters of lakes and large rivers (McMahon 1991). These habitats are now occupied primarily by the more common native mussels, some of which may now come under increasing stress.

Glochidial host extinction seems to have played a limited role in past mussel extinction, if any. There is no known host link between extinct fish and extinct freshwater mussels, but a few extinct fish species are possible candidates. The harelip sucker could have been a host species. It was once widespread in rivers where many mussels became extinct. While it spent its days in deep pools, it may have fed in shallower mussel habitat at night and spawned there as well. Extinction of bluntnose and phantom shiners in the Rio Grande was roughly coincident with extinction of 3 mussel species in the Rio Grande. One of the mussel species was last observed more than 65 years before the fish were last observed, however.

Freshwater Snails

Of the 81 presumed or possibly extinct species of freshwater snails, two-thirds lived in Alabama, mostly in the Mobile River basin, and about three-fourths lived in fast-flowing river riffles. The Mobile and other gulf rivers underwent extensive change that began in the 19th century, including increased hydrologic variability, turbidity and sedimentation. A combination of waterway development and private hydropower dams inundated much of what probably was snail habitat between 1930 and 1980, and probably contributed, with pollution and sedimentation, to snail disappearance. The sequence of impact is unclear, however, because snail collections over the past century have been less frequent than for fish and mussels. Some species have not been observed since they were first collected. A gap in collections exists between the 1920s and the 1970s for many locations and less information exists about snail relative abundance in the early 1900s than exists about freshwater mussels.

Many assessments of decline and present imperilment implicate dams because many of the original habitats are now impounded (Neves et al. 1997). Much of the snail loss appears to result from the cumulative effects of dam construction and widespread sedimentation and turbidity from land erosion, which remains a major water quality issue in Alabama. Impoundments now inundate much of the known habitat area. Combined with high turbidity caused by poor land management practices, these impoundments eliminate much of the food supply of extinct snails. Impoundments also provide habitat for a different array of snail predators, some of which may be more effective than the original river inhabitants. However, the thick shells of most of these species deters much potential predation.

The potential for impact from sources other than impoundments is indicated by snail life history and the history of habitat change in snail habitats. Most of the Mobile system was exposed to extreme water level fluctuation and widespread sedimentation resulting from land use changes before impoundment construction (Jackson 1995). Increased chronic turbidity and sedimentation probably limited the distribution of attached algal foods more to shallow riffles once deforestation occurred and agriculture became established in floodplains and watersheds. While much of the river snail habitat has been impounded, the primary agents of snail loss may be chronic turbidity, sedimentation, and other pollution, more than altered river velocity.

Parts of the Mobile basin became densely populated and industrialized starting in the late 19th century, especially along the Black Warrior and upper Cahaba River. Some parts of the Mobile River basin have lost snails to extinction even though they have remained generally free of impoundments. The Cahaba River, for example, has remained largely free of dams, but lost at least 2 snail species. One of the most endangered of the river snails, *Leptoxis formani*, is absent from 95% of its original range (Johnson 2004), only 50% of which is impounded (NatureServe Explorer 2007).

Snail species extinction and imperilment outside the southeastern rivers is scattered over the United States. One species native to a Michigan lake was last seen in 1907. Two stream species once lived in Hawaii. The causes of these extinctions are poorly documented, but are most likely associated with localized alteration of habitats, where such changes have been dramatic. Federal water resources development is not likely to be an important factor.

The Tombigbee, Black Warrior and Alabama rivers have been developed as waterways by the Corps. Little other water development in Alabama is Corps associated, being connected with either private hydropower, especially in the Coosa River, or with TVA projects in the Tennessee River. However, the Corps does manage a flood control purpose in some of these reservoirs. The extent of Corps impact is difficult to sort out in Alabama, but because much of the snail species loss is linked most clearly to water resources development in either the Coosa or the Tennessee river, where private utilities and TVA are the primary agents of development, the Corps' role in development and impact appears to be secondary.

The remaining losses are difficult to ascribe to cause based on existing knowledge of the species. The Corps is most certainly implicated in the loss of one species from the Tombigbee River and another in the Ouachita River, which has been most clearly impacted by Corps flood control reservoirs and lock and dam structures installed from the 1950s through 1980s. One extinct species in Arkansas is known only from a fossil and may have been extinct before human impacts accrued. Another species may have been extirpated by Corps reservoir construction, although it has not been observed since it was first recorded in the 19th century. In summary, as much as a third of the snail extinctions may have been linked to Corps projects, but connections are relatively clear for less than 5%.

Less is known about snail declines than any of the other taxonomic groups included in this study. The conservation status of extant snails continues to be in flux. A large fraction is considered to be critically imperiled or imperiled in the NatureServe Explorer database. One species identified as possibly extinct was recently rediscovered. On the other hand, numerous species recently were transferred from the possibly extinct to the presumed extinct category in NatureServe Explorer.

Most snails are considered imperiled because their range is small, and not necessarily because they are known to be in decline. Some species seem to be in decline, but specific causes are obscure. Abundance trends are in general not well documented. Where identified, two-thirds implicate large reservoirs, many of which are privately owned and operated. Other identified causes include the general effects of land development, water contamination, sedimentation and water depletion.

Most of the imperiled and endangered snails are gilled snails ecologically much like most of the species thought to be extinct. Habitat deterioration in rivers has been caused by sedimentation, hydrologic changes, oxygen-depleting pollutants, eutrophication and other change that may have contributed to declines of gastropods before extensive impoundment, but the history is poorly known. Dredging for navigation purposes may threaten snails in some habitats, but does not explain losses in tributaries where there was no navigation development. Other imperiled snails occur in isolated springs, mostly in the western U.S., where Federal water development is rarely involved.

Impoundments probably reduce habitat suitability for imperiled species in ways similar to their effects on freshwater mussels. The impoundment of water causes sedimentation over algal food sources. Chronically turbid water in reservoirs also reduces light transmission necessary for algal production. Deep reservoirs are prone to intolerable oxygen depletion in deep waters and their tailwaters. Impoundments also fragment habitat and may block upstream movement of snails into suitable habitat and trapping downstream drift of larval snails in unsuitable habitat. However, the primary impacts may derive from poor land management and, similar to some freshwater mussels, improved reservoir management may provide protection of imperiled species in some settings.

Alabama is a center of snail diversity and imperilment where threatening impoundments are often privately operated for hydropower. Navigation impoundments and dredging managed by the Corps in Arkansas, Missouri and Alabama rivers probably contribute to species imperilment. Of these, 82% are gilled snails (prosobranchs) in 3 genera (*Elimia, Leptoxis, Somatogyrus*) centered in the Mobile River system of Alabama and its major tributaries. Some species occur in the Alabama reaches of the Tennessee River. Many are suspected of being extinct because their known habitats are now impounded (Bogan and Pierson 1993). Many species were described long before significant impoundment occurred, but there is little information about the stability of their populations before impoundment. While least well documented among the taxonomic groups studied, Corps operations probably contribute to snail imperilment at impoundments and during dredging and channelization. Centers of such impact occur in the Mobile, Arkansas and Ouachita rivers.

Twenty-one freshwater snail species are listed under ESA protection. Many of these are western species associated with freshwater springs. A few are members of genera inhabiting eastern rivers. More species might qualify for ESA listing if survivors can be found. The number of ESA listed species is substantially less than the number of imperiled snails listed and a smaller fraction is located in waters where the Corps is active. Only one-third of the identified threats to species listed under ESA protection are attributed to large-scale water resources development (Table 9). Most are threatened by contaminants, water depletion, and nonnative species. Only 6 are clearly threatened by impoundments.

Corps-Influenced "Hotspots" of Species Imperilment

Consistent with the location of most freshwater species in the United States, a large fraction of the existing threats to endangered freshwater species occur in the Corps districts of the South Atlantic, Ohio River, and Mississippi Valley divisions (Table 12). The potential impact of the Civil Works program depends very generally on where the vulnerable species are located and existing and future Corps projects are located. Of these, the present location of vulnerable species and projects is best known. Future actions are less certain, but are best indicated by recent activity.

Most hotspots of imperiled species occur in the Southeastern United States. Schute et al. (1997) indicate that 50% of the fish, 90% of the freshwater mussels, 73% of the freshwater snails and about 90% of the crayfish species occur in the southeastern United States. Based on distribution data presented in Chaplin et al. (2000), the greatest concentrations of imperiled fish and mussel species occur in the Tennessee-Cumberland river system (104 species). Watersheds with 50 to 99 imperiled fish and mussel species include the Mobile River system of Alabama and the middle Red and Arkansas drainages. Watersheds with 25 to 49 imperiled fish and mussel species include the lower Mississippi, Apalachicola, Savannah, Santee, Pee Dee, Cape Fear, upper Ohio, Wabash, and lower Missouri River systems. The remaining watersheds of the Nation have less than 25 species of fish and mussels of concern. In order of their freshwater biodiversity indicated to be at risk, the Districts are led by Nashville, Mobile, Little Rock, Tulsa, and Kansas City (Table 12). No districts are free of extinction-related concerns, but those with the least concern occur in the Western United States.

The relative low concern expressed in Stein et al. (2000) for the Columbia, Sacramento and other river assemblages results from the relatively low number of full species at risk in those river systems. A higher fraction of stocks and subspecies are at risk there than at other locations, however, much because the west coast salmon and trout species are better known than other species at that level of genetic distinction. Other evaluators of Corps activities would place higher emphasis on the activities of West Coast districts, based on endangered stock concentrations of salmonid fish. Internal estimates of Corps operations expenditures on ESA

Table 12. Rank of species imperilment concern based on the numbers of fish and freshwater mussel species listed as imperiled by Chaplin et al. (2000).

Rank (species #) of Corps Districts Based on Number of Imperiled Fish & Mussel Species (Conterminous U.S.)			
1 (0-9)	**2** (10-24)	**3** (25-49)	**4** (50-104)
Albuquerque	Baltimore	Charleston	Kansas City
Anchorage	Buffalo	Cincinnati	Little Rock
Honolulu	Chicago	Huntington	Mobile
New England	Dallas-Ft Worth	Louisville	Nashville
New York	Detroit	Memphis	Tulsa
Omaha	Galveston	New Orleans	
Los Angeles	Jacksonville	Pittsburgh	
Seattle	Norfolk	Savannah	
Walla-Walla	Philadelphia	Wilmington	
	Portland	Vicksburg	
	Rock Island		
	Sacramento		
	Saint Louis		
	Saint Paul		
	San Francisco		

compliance indicate that the Columbia and Missouri have risen to high levels of freshwater biodiversity concern. Much of this concern has been motivated by nongovernmental organizations (NGOs) and other non-government stakeholders in the values attached to endangered taxa in these rivers.

Instead, Stein et al. (2000) identify 327 watersheds covering 15% of the United States total that would conserve freshwater fish and mussels (and many other invertebrate species) at risk of extinction based on the numbers of full species. Basing priority on full species designation avoids many debates about genetic distinction and it is consistent with emphasizing evolutionary uniqueness (May et al. 1995) in making conservation decisions when resources are limited. "Protecting and restoring" much of the Tennessee, Mobile, Apalachicola, Altamaha, lower Red River watershed including the upper Ouachita River, and Black (lower Arkansas tributary) rivers of the southeastern and southcentral U.S. would have major positive impact according to the Stein et al. (2000) analysis. Smaller key watersheds are scattered throughout the United States. Dobson, et al. (1997) found that half of the endangered species were concentrated in a small fraction of the United States landscape, which, for mussels, was mostly in the southeastern United States. Flather et al. (1998) found nearly 25% of all endangered fish species and over 90% of endangered mussel species in the Tennessee and upper Mobile watersheds.

Corps project operation also tends to concentrate in the districts and river basins associated with hotspots of species vulnerability in the top two ranks of Table 12. Navigation projects are especially common in these areas, but flood control and hydropower dams also are common in some areas. This is consistent with the relatively high association of imperiled freshwater species with warmwater rivers in the East.

DISCUSSION

RATES OF EXTINCTION

Ricciardi and Rasmussen Were Correct in General

The results of this study vary from those of Ricciardi and Rasmussen (1999) in the details, but the general conclusions are similar. Despite many uncertainties associated with the data of species extinction, the evidence points to significantly accelerated extinction rates in the United States for vertebrates and mollusks at least since the late 1800s. It also indicates greater past rates of freshwater species extinction than terrestrial species extinction in the continental United States (without Hawaii). Some of this study's higher estimates of past freshwater extinction rate are as high as lower estimates of rainforest rates of extinction. Patterns of extinction emerge that inform about species vulnerability and the likely contribution of different environmental stressors, including water resources development, to species extinction and imperilment. The data and analyses brought together here provide a new resource for informing conservation activities of water resources development programs in the Corps of Engineers and elsewhere.

Representativeness of the Data

The extent to which estimates of freshwater extinction rates exceed background rates and terrestrial extinction, and rival rainforest, rates is greatly influenced by the status of the least confidently known species, especially the mollusks, and by conditions in Hawaii. Limiting the analysis to the continental United States much more clearly indicates the differential loss of terrestrial and freshwater species and the potential for the Federal water resources development agencies to contribute to reversing past trends. Because mollusks make up 84% of all freshwater species that are listed as presumed and possibly extinct, the relatively high uncertainty associated with their taxonomic and conservation status is a dominant factor in the confidence placed in estimates of past extinction rates.

The degree to which mollusks in particular, and fish and birds secondarily, are exceptional in freshwater extinction history is most relevant to assessments of ecological impacts and the potential for ecosystem restoration, which is a Civil Works mission of the Corps. The 8,614 animal species included in this study of extinction comprise only 5.8% of the 148,800 animals that inhabit the United States, as estimated in Stein et al. (2000). Only a few more species would be added to the list of presumed extinct compiled by Stein et al.(2000) if all animal species were included in the analysis completed for this study. It is uncertain, however, whether extinction in other groups has largely gone unnoticed or the taxonomic groups included in this analysis are exceptionally vulnerable to extinction.

Based on the assumption that past extinction rates are accurately represented for all animal species, the total estimated extinction estimate is two orders of magnitude lower than the one based on the better known, mostly larger, species, which may be naturally more vulnerable to extinction. If the animal species that have gone extinct have been fully accounted, the rate of past animal extinction overall is too close to background rates to judge that there has been a significant effect on the total composition of animal species, at least so far. If that is the case, the groups in this study, and especially the freshwater mollusks, are ecologically analogous to the canaries that warned with their death of lethal gas accumulation in coal mines. If so, providing for their needs may provide for the viability for all species, at least in those ecosystems where they occur. Improved knowledge of freshwater mollusk ecology is especially relevant to establishing more effective management of water resources in keeping with a goal of environmental sustainability. More probably, however, some acceleration of extinction has also occurred among the lesser known invertebrate groups, but not necessarily as great as that of the mollusks and the vertebrates.

Consistent with this thinking, Lomborg (2001) criticized the use of vertebrate data alone (a practice in many analyses of extinction rate), because vertebrates are not necessarily representative of other species and may bias the estimated increase in loss rate against background estimates for all species. The mollusks are even more vulnerable than vertebrates. Insects, however, may be much less vulnerable. Because insects comprise a large fraction of the world's species, Lomborg estimated a loss of only 0.7% of all species over 50 years, mostly in the tropics (0.14% per decade) compared to rates an order of magnitude higher based on vertebrates. Because insects and their close relatives make up 65% of the known animal species in the United States, full inclusion might reduce the estimated extinction rate substantially if the data were complete.

The Impacts of Past Freshwater Extinction

The impacts of past extinction can be framed in both economic and ecological terms. A few of the extinct freshwater species were among groups that could now have economic value, mostly because of recreational use. Several of the extinct fish were trout and cisco species, which collectively are valued for their recreational and food value. However, all of the trout had very limited range, which lent to their extinction and greatly limited their total economic potential. For the cisco species, close relatives remain abundant and might serve as substitutes except that commercial fishing has greatly declined in their native habitats. Some extinct mussels may have been valued for their shell, but others always were rare and many substitute species remain common. In general, the known economic loss has been small, if not negligible. The unknown economic potential is impossible to estimate. In that sense, the losses irreversibly reduced possible resource opportunities for future generations.

Depending on how exceptional extinction is in the groups investigated, the past ecological consequences of accelerated extinction with respect to ecosystem sustainability is likely, for the most part, to be small as well. The taxonomic groups selected for this analysis of extinction and imperilment rates are exceptional in a number of ways. Perhaps most important, they are all consumers, and do not include primary producers and decomposers, which likely would maintain basic production and nutrient cycling alone in freshwater ecosystems. While there are numerous plants listed under ESA protection, few are aquatic and virtually all are vascular. Most freshwater production is sustained by non-vascular organisms (e.g. Wetzel 2001) that are not listed among the imperiled. Consumers, however, increase rates of mineral cycling and other ecosystem function by breaking down organic detritus into much smaller particles that are more rapidly decomposed mostly as fecal material. In addition, they contribute stability and functional diversity to community dynamics by maintaining controls on population growth of potentially dominant species.

Relative size is an important factor in the functional importance of species. Many ecological attributes of species are related to size (Peters 1983). Life expectancy is directly correlated with species size over the full range of sizes that occur in nature. The ability to rebound from environmental stress is related to reproductive potential, which generally decreases as the average size and life expectancy increase. Larger species are more vulnerable to human exploitation and may be more vulnerable to environmental impacts in general, and to fragmentation of the larger and more complex habitat arrangements they often require to complete their life cycles. This fits the pattern of observed extinction, which tends to be greater for larger freshwater taxa.

With respect to freshwater species, most of the extinct fish and amphibians are predators that feed relatively high in the food web. Many of them were abundant in their native habitats and sometimes very important in determining the relative abundance and function of other species in the ecosystem. Because many selected their foods based on relative abundance, they may have contributed substantially to maintaining diversity and stability among the species they consumed. This is more likely in small ecosystems where many of the extinct species considered in this study once occurred uniquely in their functional group. In larger ecosystems, such as the Great Lakes and large rivers, even the loss of several predator species may be functionally assimilated by the remaining species because of functional redundancy.

Most of the extinct invertebrate species feed lower in the food web. Riffle mussels filter-feed on decomposing organic detritus in the stream current and, in the high abundances often typical of pristine ecosystems, could significantly reduce the concentration of organic matter in habitats, maintaining water clarity and low oxygen demand. Gilled snails are herbivores that graze on attached periphyton and stimulate its productivity by removing less productive old growth. Crayfish are omnivorous opportunists that can feed to a large extent on dead and decaying plant and animal matter.

Collectively, mussels, snails and crayfish contribute largely to regulating interactions among primary producers and decomposers, controlling the buildup of oxygen demanding organic matter and maintaining oxygen-saturated environments. However, many of the invertebrate extinctions occurred in communities composed of numerous species where functional impact was mitigated by the functional redundancy in surviving communities, as, for example, in many warmwater river ecosystems. Ecosystem functions are likely to erode, however, as more species are lost. In contrast, major changes in ecosystem function are more likely to result from loss of a few species in small ecosystems with low biodiversity.

While science is poorly informed about the specific functions lost in the extinction of most species, functional loss in ecosystems appears overall to have been small. Where it is most likely to have been locally large, as in small spring systems of low diversity and functional redundancy, the geographical extent of the effect is small. Where it is most likely to have extensive geographical impact, as in the Great Lakes and large southeastern rivers, high functional redundancy has mitigated impacts.

FUTURE RATES OF EXTINCTION

Alternative Futures for Extinction in the United States

Based on worst-case projections of future extinction in freshwater ecosystems, greater impacts on resource development potential and ecosystem function can be expected as more species are lost. Thus, recent acceleration in extinction rate is of concern primarily because of what it portends. The great uncertainty associated with forecasts of future extinction gives pause to its utility, except to inform risk management strategies against a backdrop of alternative possible future conditions. We have to be satisfied with what might be and to manage adaptively (Hollings 1978, Walters 1986, Shaffer et al. 2000) as the future unfolds. The future becomes increasingly cloudy beyond the next few decades. Even in the near term, the future is uncertain because numerous species may be already functionally extinct Neves et al. (1997), creating an "extinction debt" (Tilman et al. 1994) that will eventually be paid, especially among the long-lived mollusks. It is important to determine which are beyond help in order to more cost-effectively protect and restore others.

Extrapolation of past extinction trends to estimate future extinction rates is perhaps less certain than present estimates of species imperilment, but the assumption that either imperilment or listing under the ESA imply future doom may be even more uncertain. Indeed, the listing of threatened and endangered species has either slowed extinction, in response to environmental protections stimulated by legislation and education, or the slowed extinction rate has been coincidental. The NRC (1995) and Scott et al. (2006) believe the Act has been effective in slowing the extinction rate, if not in recovering species to a viable status. Unlike imperilment data, trends picks up the recent leveling and possible downturn in last decadal observations of species in logistic trends and indicate what might have been in linear trends. The differences indicate that decadal rates of extinction have been reduced, and that makes sense given the rise in environmental awareness and environmental legislation of the 1960s and 1970s.

Many rare species listed as imperiled are now less exposed to threat than they might have been. By the very nature of the intent associated with assessing conservation status, future risk is more likely to be managed than in the past and future extinction rates overestimated based on present imperilment. Therefore, predictions of extinction based on imperilment data are most likely to be worse-case scenarios. Extrapolation of past extinction trends in this study support this conclusion. Even so, the threats of agricultural and transport-system contamination and invasion by nonnative species remain large for naturally rare species with limited ranges or species with ranges limited and fragmented by dams and other structural and chemical alteration. In the latter case, the water resources agencies have the opportunity to contribute to recovery through reversal of past habitat fragmentation trends, especially in river ecosystems.

Much has been done to frame the outer dimensions of the biodiversity "problem" in worst-case scenarios, mostly with the intent of raising the profile of the potential problem into public awareness and corrective policy action. Less has been done to examine the probabilities of various future projections playing out, given what we know. The needs are daunting, however. The most cost-effective adaptive management requires planning and management across the spectrum of possibilities, including education, research, impact mitigation, ecosystem restoration and other management. It requires collaboration of unprecedented scale and complexity, which, while facilitated by major technological advances, depends most fundamentally on personal values and skills and how government agencies, educational institutions and other NGOs nourish them.

Federal agencies are expected to lead in this endeavor through execution of the ESA, but face their own limitations rooted in limited public tolerance of government spending. Success is threatened by the many government agencies and NGOs that perceive their roles as too small to make much difference and focus, to the extent the law allows, almost exclusively on more immediately achievable objectives. Short-sighted public interest groups encourage this. Agency leadership, especially vision, is key to success.

The trend data suggest that future extinction rates depend much more on commitments to species conservation than on the forces in history that have caused extinction and present imperilment. The costs of reversing the trends will be high. The mollusks and fish are not so exceptional among the lists of imperiled species that they should dominate future water resources management concerns. Crayfish and aquatic amphibians now approach the same levels of imperilment. The imperilment of those species, however, occurs largely outside the river habitats that are most affected by Federal water resources development, where the main concerns are mussels, snails, many of the smaller fish species in the minnow, perch and catfish families, as well as large anadromous fish species.

Commitment to protection, at least at past and present funding rates, appears to be clear. Legislative attempts to significantly weaken the ESA have, so far, consistently met with majority disapproval. However, commitment to protection and recovery of all imperiled species to a secure status is less clear. While the ESA seems to have slowed extinction remarkably, it has had much less success recovering species (Mann and Plummer 1995, NRC 1995, Scott et al. 2006). Historical funding has limited gains in species recovery and Congress has slowed the further listing of species, in part to encourage recovery of already listed species, but *de facto* to control costs. A high-profile extinction or two might roil the fiscal stagnation, but for now the public appears to be satisfied with sustaining endangered species in precarious status over spending more to recover them to securely viable status.

Public acceptance of the status quo explains in part the failed attempts to legislate biodiversity protection before species need listing. However, NGOs led by The Nature Conservancy have taken up this challenge generally because they get significant public support and have made significant privately funded gains. These NGOs cannot meet the challenge alone and emphasize the need for leveraging public funds and for collaboration across all organizations, private and public (Groves 2003). What transpires over the next century depends more on future socio-economic and political trends than on existing conservation status and ecological conditions. For the issues raised here, this understanding is particularly relevant to how government agencies collaborate with each other and with NGOs to use authorities more effectively. This especially applies to the ecosystem restoration mission of the Corps of Engineers. The Corps has a tremendous opportunity to use its ecosystem restoration mission more effectively to collaborate with organizations and agencies intent on reversing the decline in biodiversity.

What plays out, both in the near and far future, depends not only on continued dedication to conservation goals, but also the extent to which global changes in climate, already set in motion, continue to erode the remaining habitat of vulnerable species without much management recourse. It also assumes that the public will remain otherwise satisfied with its general welfare. Aging trends and present public commitments to mandated financing of entitlement programs that primarily benefit older people will continue to stress discretionary program funding, including environmental programs. Any significant long-term decline in public welfare could reduce public commitment significantly.

Among ecologists, concern about future rates of biodiversity loss relates to possible loss of ecosystem functions that sustain natural services (e.g., Wilson 1988, Lubchenco et al. 1991), which may be irreversible once extinction has occurred. If the species included in this study are representative of all species, a high future rate of extinction of all species groups is forecast by imperilment data and with it a large impact on ecosystem function, natural services and the benefits they provide to humanity. Loss of those species would foreclose any restoration of functions dependent upon them and eliminate options for possible resource development.

On the other hand, if relatively few extinctions occur because of a sustained public commitment, the overall effect on sustaining ecosystem function and service in the amounts desired may be less troubling, especially if extinction trends stabilize or decline. While present scientific understanding of extinction points more toward this second prognosis, more research is needed on taxonomic groups that have not received as much attention as the vertebrates, crayfish, and freshwater mussels. It seems clear, however, that in freshwater ecosystems, imperilment is exceptionally high, which indicates a need to restore ecosystems to an extent necessary to restore species viability. Because the Corps is the only Federal agency with an ecosystem restoration authority that extends beyond the lands it owns it has a unique opportunity to work with other nonfederal and Federal agencies to restore species viability in freshwater systems.

Comparison of Freshwater Extinction to Terrestrial and Rainforest Extinction

Ricciardi and Rasmussen (1999) raised concern about future freshwater biodiversity loss rate in North America to a new level with their conclusion that it equaled projected rates of biodiversity loss in rainforests. The progressive loss of rainforest and its associated biodiversity has been widely used as an indicator of unsustainable environmental practices (e.g., Wilson 1989). The result was used to bolster allegations that the Corps of Engineers was continuing in its old destructive ways similar to the management practices in less developed nations. This conclusion, however, based on the results of this study, is the least dependable of Ricciardi and Rasmussen (1999) because of the substantial uncertainty that exists in both rainforest and freshwater loss rates, depending largely on which taxonomic groups are used in the study.

Imperilment data used in this study produced results at the high end of the range and were similar to those of Ricciardi and Rasmussen (1999), which were also based on the conservation status of species and the assumption that the imperiled species were largely doomed to extinction. They indicated a potential loss of 4.5% per decade, which is somewhat higher than the midpoint (about 1%) of estimates for rainforest and world (mostly rainforest) rates. Estimates based on ESA listing are substantially lower. Lower still is the recent past freshwater extinction rate in the United States (at most, 0.6% per decade). It is more than an order of magnitude lower than the high estimates yet higher than the low estimate of Lomborg (2001), which is 0.1% per decade for the world, most of it in rainforests. Continued commitment to the long-term protection of endangered species would make the 0.6% estimate based on trends more probable than the high estimate based on imperilment. That estimate is lower than the low range of estimates (1% per decade) reported before the Ricciardi and Rasmussen study, but technically remains within the range of rainforest and world rates because of Lomborg's low estimate made in 2001.

Like the rainforest rates, comparison of freshwater extinction with terrestrial extinction in the United States is useful for making a point about freshwater extinction rates because even informed conservation biologists have become much more acquainted with high-profile terrestrial extinctions. Birds especially interest both the public and conservation scientists, and extinction has visited them most of all the terrestrial species in the United States. The perception of the relative loss rate is greatly influenced by whether the geography compared is based on political or physiographic boundaries. Leaving Hawaii out of the calculations substantially increases the relative rate of freshwater extinction compared to terrestrial extinction estimated on the continent. More interesting than the ratio effects, however, are explanations for the effects, because they provide valuable insights to managing future risks of species loss.

THE CAUSES OF EXTINCTION AND IMPERILMENT

The Geography of Extinction

Recent extinctions have not been geographically random and their distribution often reveals insight into cause. Extinction has occurred in geographical hotspots, which should to be carefully considered in terrestrial and freshwater comparisons or, for that matter, in any other comparison of extinction rates and examination of causes. The influence of Hawaii on the ratio of freshwater to terrestrial extinction in the United States is a case in point. More important for management, the characteristics of these hotspots also inform generally about the strategies required to reverse species declines and decrease extinction rates.

Stein et al. (2000) devote several chapters to the geography of extinction and imperilment. In general, both past extinction and future vulnerability of extinction (as indicated by imperilment) increase along gradients from north to south and from east to west. This pattern is related both to increased environmental threats along these gradients and to increased number of species. Southern and western states support greater species diversity and more endemic species than northern and eastern states and have higher percentages of species at risk of extinction. The general pattern most clearly reflects temperature gradients.

For freshwater diversity alone, the impact of aridity is quite apparent in the much higher diversity of the eastern states and the much higher percentage that are imperiled in the southwestern states. A secondary factor is the extent and retreat of past continental glaciation and associated changes in sea level. Both the Great Lakes and Florida waters have lower than expected natural freshwater diversity because they did not exist until relatively recently in geological time and have remained isolated from species rich waters by oceans and land barriers.

The Mississippi River system is the center of freshwater diversity in large part because of its north-south connectedness and the moderating effect of water on temperature extremes.

The pattern of endemism reflects the pattern of locally isolated habitats forming ecosystem islands. Oceanic islands have long been recognized as locations of unique terrestrial fauna that are especially vulnerable to extinction (Diamond 1984, King 1985, Richman et al. 1988, Olson 1989). Reid and Miller (1989) indicated that 61percent of all mammal extinctions, 81% of all bird extinctions and 95% of all reptile extinctions have occurred on oceanic islands. Native terrestrial species on islands often occur no where else, having evolved from a few colonizers.

On continental land masses, mountain ranges and basins both isolate populations and provide islands of unique habitat at different elevations. Similar concentrations of unique species have evolved in freshwater springs, lakes and rivers, and, if high rates of imperilment are a reliable indicator, those species are similarly vulnerable to extinction because they too are islands in a much larger land mass.

Freshwater isolation is most evident in the isolated waters of the mountain west where closed basins and aridity are common causes for isolation following much wetter periods during the last continental glaciation. In the midwestern and eastern states, freshwater isolation is more often caused by natural differences in aquatic habitats. The oceans are downstream barriers to species movement of all rivers entering the sea. In the Mississippi River drainage, major habitat differences caused by erosion-deposition changes occur in the mainstem river and tributaries, which especially cause barriers to molluscan dispersal. Major western tributaries with sources in the Rocky Mountains and Great Plains naturally carried much higher sediment loads than eastern tributaries with sources in forested watersheds. Associated with that condition, the Missouri River and its tributaries support much lower freshwater diversity than the Ohio and its tributaries.

Concepts organized into the theory of island biogeography by MacArthur and Wilson (1967) have continued to be a focus of much research in evolutionary biology and ecology. The theory has been extended to any habitat type that forms an "island" isolated by different surrounding habitat types (Simberloff 1986). Whitmore and Sayer (1992) reviewed the extension of island biogeography from ocean islands to other insular habitats. An important hypothesis emerging as increasingly valid from the studies of Diamond (1984) and others is that the risk of natural extinction in island habitats decreases as the island-habitat size increases, and it increases with fragmentation of large habitat into smaller habitats. Despite continuing controversy over the completeness of testing and uncertainty in the validity of concepts, the theory of island biogeography has had a major influence in thinking about extinction prevention through habitat protection and reversal of habitat fragmentation through restoration (May 1975, Lovejoy et al. 1984, Harris 1984, Shafer 1990).

Whether freshwater in continents or land in oceans, island fauna are more vulnerable to extinction than their continental and marine environments because their habitat area is easily pervaded by threats to their survival, such as generally adapted predators, competitors, pathogens, toxic agents and physical conversion. Island size may be less important than the total extent of impact penetration. Even though they are large, the Great Lakes do not have great habitat diversity offshore and, once they gained access, were easily penetrated by nonnative predators (including fishermen) and competitors that contributed to the extinction of several fish taxa (Smith 1968). Habitat diversity within islands is an important secondary variable. The main refuges from human-induced change in terrestrial landscapes have been provided by rugged terrain and extreme climates, which are more likely to occur on large islands and continents.

Thus in freshwater ecosystems, the distribution of species diversity, past extinction and present imperilment is largely predicted by winter temperature, aridity, the natural erosion of watersheds, and the size and isolation of species habitats. The districts having the greatest concentrations of imperiled species are located in the southcentral and southeastern United States where freshwater species diversity is especially high in warmwater rivers. Ecosystem restoration for ecosystem sustainability is likely to be most effective in the imperiled ecosystems of these same districts.

Connectivity and Pervasive Change

The high relative vulnerability of species in continental freshwaters owes much to pervasive ecosystem changes that have occurred in freshwater habitats. Certain types of stressors can quickly pervade freshwater ecosystems once they are introduced or invade them. This is much less true of ocean and continental ecosystems, which, because of their very large size and variation, resist totally pervasive impacts. Oceans, especially, have sustained few verified extinctions as a consequence of pervasive change other than commercial fishing, which has grown to that scale for many species (e.g., NRC 1999b, Pauly and Maclean 2003).

The large majority of freshwater extinctions has resulted from pervasive, cumulative change in the hydrology, geomorphology, water quality and species composition of aquatic ecosystems. Nonnative invasive species are the major biological agent, which often has the capability of multiplying in abundance and intensity of impact. Changes in water chemistry also can be pervasive, but are more likely to be diluted in larger waters when they originate from isolated and relatively small point sources. Even large aquatic systems can be completely altered by very large point sources or widespread sources of pollutants, however, such as from watersheds largely converted to agriculture. Because of gravity-generated flow, materials entering by way of upper watersheds and floodplains can disperse rapidly throughout river systems uninterrupted by lakes or wetlands, which trap some materials. Eutrophying nutrients and pesticides are among the most common contaminants associated with agriculture, which is common on the floodplains and watersheds of medium to large rivers and lakes of the United States.

The most extensive physical changes have most typically taken the form of hydrologic and geomorphic changes resulting from increased discharge fluctuation and sediment load originating from disturbed watersheds. These typically result from intense land use (Trautman 1981) and are often intensified by extreme climate variation. Especially destructive practices include pumping small spring habitats entirely dry or enclosing them in concrete or other structures for local water supply purposes. Modifications due to impoundment and dredging also have become the last in a series of pervasive changes to occur in medium to large rivers. Impoundments alter river hydraulics and contribute variously to increased sedimentation at the upper ends and to altered oxygen and temperature in tailwater discharges from deep reservoirs.

The results of this study indicate that most freshwater extinctions in flowing waters are largely associated with the pervasive impacts of altered discharge, sediment and water quality carried through highly connected systems. Analysts typically emphasize the cumulative effect of diverse changes in large river basins (Parmalee and Bogan 1998, Neves et al. 1997). However, early changes in hydrology, sediment dynamics and water quality were widespread (Troutman 1981, Jackson 1995, Trimble 1974) before much impoundment happened and seem to have contributed largely to the early extinctions of several river species, possibly coupled with over-harvest. Early dredging and lock and dam construction by the Corps probably contributed in a small way through the small fraction of the total habitat that was altered. The widespread direct impact of agriculture, nonnative species and fishing on vertebrate extinction revealed in this study was most typically associated with small spring, stream and lake habitats.

Certainly, the cumulative effects include large impoundments that are at once nearly pervasive and fragmenting in many medium-large rivers. Large impoundments contribute importantly to the imperilment of river species but probably less so than is implied by some critics (e.g., Watters 1999). In general, other studies consistently confirmed the results of this study, which indicate that water resources development has played a smaller role in species imperilment than land use and nonnative species impacts, but nonetheless a significant one. From a review of the literature, Strayer et al. (2004) found that water quality degradation and other habitat alteration most threatened freshwater mussels in 47% of the 45 reviewed articles compared to 33% caused by impoundments. NatureServe Explorer (2007) also identified water resources development as a cause of imperilment in about one-third of the cases elaborated, as did Stein et al. (2000). Richter et al. (1997) revealed that the three leading threat categories recognized by surveyed biologists were agricultural sources of sediment and nutrients, nonnative species and hydrologic regimes altered by impoundment operations for agricultural and hydropower purposes. Much of the hydropower threat comes from private impoundments, but Federal projects contribute.

A third of the 900 species now identified as imperiled among the group of indicator species studied is still a troubling number for water resources management. The fraction is about the same as the fraction of extinct species connected to large water resources projects, which to some implies little improvement. Some Corps multipurpose impoundments may contribute to species imperilment because of hydroregime and water quality impacts, but other agencies and private utilities collectively contribute more. While less than 5% of the threats to species were perceived to be associated with waterways navigation in the Richter et al. (1997) study, that amounts to nearly 45 freshwater species of particular interest to the Corps. A more detailed inventory of species impacted at Corps projects is needed to more precisely determine the dimensions of imperilment contributed to by Corps projects.

INTEGRATING DAMS INTO MANAGEMENT FOR SPECIES RECOVERY

Of the water resources development projects implicated in extinction and imperilment, impoundment by dams leads the list. The results of this study uncovered many ways in which dams and their impoundments have altered habitats that may have contributed to species loss. Whether that influence has been as great as implied by some (e.g. Watters 1999, Parmalee and Bogan 1998) is, as shown here, debatable. Less debatable, however, is the positive role that Federal water resources management could have in the protection and recovery of many vulnerable freshwater species because they are more immediately manageable than many other threats originating from more dispersed sources. Because of its unique ecosystem restoration authority, which extends to all habitat degradation regardless of the cause, the Corps is especially endowed with the opportunity to reverse trends in declining freshwater biodiversity.

Federal agencies are expected (but not obligated) to do what they can within their authorities to recover endangered species and the ESA has provided a platform for cooperation primarily through the Federal agencies (Clark and Wallace 2006). Because 80% of the species listed under the ESA and a similarly large fraction of imperiled species depend primarily on private land use practices, the Federal land management agencies have an important but limited recovery effectiveness. In contrast, Federal water resources agencies extensively affect nonfederal properties in and underlying river waters and they have borne a large burden with respect to species recovery (Yaffee 2006). While the need for nonfederal land owner collaboration is well recognized, so are the challenges (Clark and Wallace 2006). There will continue to be an outsized expectation of contributions from the Federal water resources agencies to recover species and prevent the need for future ESA listing.

Because impoundments also can act to reduce the pervasiveness of stressors, the cumulative effects of river changes have not been entirely negative. Some impoundment tailwaters provide the last refuges for some mollusk species (e.g., Parmalee and Bogan 1998) where they are isolated from watershed sources of sediment and pollutants. Some species may not have survived without the more dependable water supply provided by upstream reservoirs during low-flow periods and droughts. Understanding of these complex interactions has advanced, but needs more attention to more effectively adapt water resources management to future recovery of vulnerable species.

Whereas many studies emphasized dire aspects, some (e.g., Parmalee and Bogan 1998, Strayer et al. 2004) indicate that the future extinction rate of freshwater species, and mollusks in particular, can be substantially reduced, if not entirely prevented, with proper risk management, which often involves Federal water resources projects. This optimistic view is contingent on the effort placed on understanding the stresses on imperiled species and acting swiftly to alleviate them enough to assure species survival, if not full recovery. Total prevention of non-source pollution from agriculture and other sources and the spread of nonnative species has proved difficult. Based on a 20% sample, EPA (2007) reports that the designated uses of 45% of streams and rivers in the United States remain impaired by pollution from sediment, pathogens and habitat alterations caused mostly by agricultural practices, water diversion and channelization (much related to agriculture). Often overlooked in assessments of imperilment are the indirect impacts of watershed use on water, sediment and other habitat quality and the interaction of impoundments and other hydrologic changes with those impacts. These are, however, important considerations for future management.

Past patterns of ecological change and species decline need to be considered in future approaches to management for a more sustainable freshwater ecosystem supporting viable species populations. A systems approach is essential for understanding the processes linking ecosystem change to the conservation status of individual species so as to manage for both ecosystem and species viability. Restoration of freshwater species security might best be achieved through a regional ecosystems approach, including assessment methods that track impacts from indirect sources in the watershed through chains of effect transmitted to freshwater habitats. More effective restoration also would benefit from use of contemporary planning methods that seek out the most cost-effective restoration for species viability. Because needed information is often scarce and so many species are now imperiled, a practical approach is adaptive management that integrates research into carefully considered conservation actions (Walters 1986, Strayer et al. 2004).

Water resources management can play prominently in this strategy, especially with respect to reservoir management for more dependable flows with acceptable water quality. Given that the main sources of stress on imperiled species and their supporting ecosystems is from agricultural runoff and invasive species, existing impoundments need to be more carefully assessed for possible positive mitigation effects as well as the intensively criticized negative effects. At this time neither the positive nor negative effects of impoundments have been thoroughly researched and modeled systematically to more rigorously assess net effects of different management scenarios in a watershed context.

Only rarely are impoundments clearly the sole threat to a species or the sole factor limiting recovery, even as they standout among other factors. Systematically sorting out the ecological pathways of threat and endangerment often is difficult, costly and controversial, however, and has been achieved for relatively few threatened and endangered species. Short-sighted actions taken to reduce the apparent impacts of impoundments may simultaneously open avenues for other limiting factors, possibly resulting in a greater total loss of benefits to all stakeholders. Providing the necessary habitat for recovery requires carefully integrated planning at the watershed level.

Regardless of original cause, the extensive transformation of many river habitats by impoundments and channelization may inhibit recovery of numerous imperiled species to secure status. However, other threats to species and supporting ecosystems may interact with impoundments in ways that need to be better understood before decisions are made to change river resources management. Parmalee and Bogan (1998), among others, acknowledge positive roles for impoundments in management planned to restore imperiled mussels to secure status. Some downstream impacts of impoundments can be mitigated, as the TVA has shown for several hundred miles of reservoir-altered tailwaters (Parmalee and Bogan 1998). An important objective is increasing the mussel reproduction and colonization rates. In addition to a wider choice of habitat being made suitable for colonization, the glochidial life history of mussels and other factors that contribute to reproductive failures need to be better researched.

Some impoundments can be managed or modified to provide more suitable habitat needs, as TVA has done at a number of its dams (Parmalee and Bogan 1998) to improve oxygen concentration and hydroregime. Impoundments at times contribute to maintenance of habitat conditions for some freshwater species by maintaining appropriate substrate conditions downstream from the dam. Some species of freshwater mussels in the Tennessee River are sustained downstream from navigation impoundments, where the water temperature and oxygen are adequate and the bottom is kept clean of fine sediment (Parmalee and Bogan 1998). More research is needed to determine the needs of species and communities in reservoir tailwaters and how well changing tailwater flow regimes comply with needs (e.g., Strayer et al. 2004).

Impoundments can sometimes trap nutrients that might otherwise cause oxygen deprivation from eutrophication downstream. Some can also trap potentially destructive sediments and sustain more appropriate bottom conditions for miles downstream. Some impoundments can sustain flows that would otherwise dry up in droughts. Some impoundments may provide a barrier between imperiled species and threatening invasive species. However, other management is required to prevent introductions of nonnative species by way of colonized boats, bait buckets and other avenues around the dams. Positive management is challenging, however. Even temporary and “minor” failures in maintenance of mollusk habitat can eliminate populations that take decades to reestablish.

Reservoir draw down or even dam removal may be a logical choice in some situations. It may be cost-effective to draw down certain impoundments enough to make significant improvements in habitat without totally eliminating the other benefits they provide. Dam bypass or removal may be the only measures left to recover endangered species when a dam acts as a direct impediment to essential species movements, including host-fish populations for glochidia (Neves et al. 1997). None of these options are attractive if other limiting factors are likely to remain and preclude recovery anyway. A thorough systems approach to analysis and collaborative management are the most effective ways to prevent bad decisions and incomplete implementation of all needs.

A careful inventory of all imperiled species within reach of impoundment management effects is a prerequisite to more effective incorporation of water resources management in recovery of river species and ecosystems. Especially where there may be significant impacts, the purposes served by dam operation should be identified and a determination made of whether the benefits of operation continue to justify the costs (including environmental costs). Reference to inventories of freshwater species diversity hotspots , such as those of Abel et al. (1998) and Stein et al. (2000), can help to direct attention to marginally beneficial dam locations for consideration of ecosystem restoration potential.

CORPS PERFORMANCE WITH RESPECT TO BIODIVERSITY

Claims that “traditional” Corps Civil Works projects have not reduced their impacts on vulnerable species can be evaluated through review of Corps policy guidance, planning and implementation process and the changes in species status that have occurred at Civil Works projects. The Corps has several avenues available for more positively promoting recovery of the ecosystem viability needed to sustain species now in decline. These start with clarification of Civil Works project planning and operations guidance when necessary. The Corps has the authority to restore ecosystems that provide the needs of imperiled species either at Corps projects or other locations as long as it has a willing nonfederal funding partner. Past Corps effectiveness can be assessed through evidence of species protection from further decline and species recovery to a secure status.

Corps Policy Guidance

Policy guidance indicates that the Corps is committed to environmental protection and improvement. The Civil Works program never had an organic authority defining program purpose. The water resources development purposes served by the Corps have instead accumulated from many individual project and small program authorities. Bit by bit, the Corps accumulated navigation, flood and storm protection, hydropower, water supply and recreational development authorities, all of which, by policy declaration, contribute to national economic development (NED) measurable in monetary terms. Policy requires that NED projects be evaluated based on economic net benefits and environment protection consistent with existing environmental law. Detailed environmental protection protocols have been developed for project planning, implementation and operations. Corps lands are to be managed based on ecosystem management principles for sustainable outcomes.

In 1986, Congress authorized the Corps to plan and implement environmental improvement projects, which evolved into programmatic ecosystem restoration authorities in 1996 and a Civil Works program mission in 1999. Planning policy guidance excludes NED purposes from qualifying as ecosystem restoration projects and indicates that contributions to national ecosystem restoration (NER) are improvements in ecological resources that are a function of habitat improvement; that is, the resources associated with species and community inhabitants of the restored ecosystems. Indicators of success according to planning policy include increased abundances of “biologically desirable species,” improved ecosystem support for “desired outputs,” and a high fraction of native species. Imperiled, threatened, and endangered species qualify as biologically desirable species consistent with the goal of the ESA to restore and sustain the viability for all but a few pest species based only on their vulnerable status. Species valued for their utility do not qualify because their value can be measured as economic net benefits (Cole 2009a).

Thus the Corps has both water resources development and restoration authorities and is challenged with determining an appropriate balance of the two consistent with the constitutional mandate for government to improve the general welfare. The issue is complicated by a complex collaborative environment. The complexity derives from the many geophysical and ecological interactions and public service effects that can occur as a consequence of any substantial change in the system, such as a new project or major rehabilitation. The complexity increased after 1986, when congressional legislation determined that most new project planning and construction would be funded in part from nonfederal sources and operated and maintained by the nonfederal sponsors. The sustainability of environmental improvements following project plan implementation depends on the nonfederal commitment to sustaining project outputs consistent with the ecosystem restoration objective.

In 2002, the USACE formally stated its program commitment to the environment in its Environmental Operating Principles (EOP), which started with a commitment to “strive to achieve environmental sustainability.” The EOP reflects the balance expected of a natural resource development agency sensitive to environmental needs as well as to economic and social needs. The EOP also reiterates the importance of agency responsibility and accountability consistent with compliance with the law. Because of a tightening budget and the demands of nonfederal sponsors to hold the line on spending, the Corps has little ability to volunteer time and funding for environmental protection beyond what is explicitly required in congressional legislation.

Similar to other Federal agencies, the Corps is directed by Congress through the Government Performance and Results Act (GPRA) to consider future agency performance more strategically in a program strategic plan and to prioritize its annual budget activities with respect to strategic goals and objectives that reflect satisfaction of public welfare improvement. The last strategic plan was developed for 2004-2009. The first two goals in the plan commit the agency to the practice of sustainable development and the repair of past environmental damage.

Much because of its complex mix of authorities and frequent criticism from both development and environment advocates, the Corps has consistently linked its corporate sense of integrity to strict compliance with the law. Obligations continue to be honored as set forth in the many authorities under which projects are now maintained and operated while also complying with environmental law. This sometimes proves challenging. Any significant change from meeting the intent of the authorized purposes requires legislation. However, when there is latitude, Corps operations policy promotes management consistent with environmental stewardship principles and ecosystem sustainability, including species viability.

Corps environmental policy guidance has undergone tremendous change over the past several decades in response to its need to comply with environmental laws. It has moved beyond environmental quality protection in national economic development to an environmental quality improvement mission expressed in ecosystem restoration project planning and implementation, and a corporate goal to strive for achievement of environmental sustainability. As expected with rapid transitions, policy interpretation issues remain incompletely resolved. Policy guidance continues to evolve in clarity and consistency of language and message. Whether or not the Corps has improved in its actions, however, critical allegations seem not to have much basis in Corps policy guidance.

Corps Project Planning

Incorporating the NEPA Process

The Corps is widely recognized and sometimes criticized for the resources it dedicates to the project planning process. Except for modifications adopted in Civil Works planning regulations for ecosystem restoration contributions to the Federal objective, the Corps continues to follow the 25-year old “Principles and Guidelines” (P&G) developed as project planning guidance for all Federal water resources development (WRC 1983). The P&G provides a framework for analysis, including environmental quality effects as directed for consideration by the NEPA. Review and consideration for all compliance requirements of environmental and other laws are included in the process. This is done in all project planning, including ecosystem restoration planning. Consistent with the NEPA, the planning process is open to public review and comment. Increasing emphasis is placed on including all stakeholders in the planning process at the earliest planning stage and collaborating as needed to assure concerns are raised and addressed.

In a typical planning process, a nonfederal agency or other organization approaches the Corps with a problem that might qualify for Federal partnership and funding. If a Federal interest is served, which is determined in a reconnaissance study, a project feasibility study is initiated contingent upon partner agreement. The Corps follows a clearly defined "six-step" planning process, which starts with problem definition and objective development to guide problem solution. The partners formulate several alternative plans and evaluate them for their net benefits or for cost effectiveness if ecosystem restoration is the project objective. Environmental costs incurred through the NEPA process and compliance with law are included in project plan costs. Environmental costs can determine whether the project goes forward or not if the costs rise to an unjustifiable level based on the anticipated total benefits. Plans are formulated for the purposes authorized, but policy dictates that they be evaluated for all significant benefit and costs, including those incidental to the authorized purposes. Where forecast benefits are positive, the Corps places priority on the plan that best satisfies the evaluation criteria for recommendation to Congress, but also considers the nonfederal sponsor's wishes when differences arise. Environmental protection costs have influenced whether or not project construction is recommended and which plan is selected for recommendation.

Fundamental in the planning process since the ESA was passed is a review of the listed species that might be influenced by project construction and operation. Possibilities are directed informally to the responsible agency, which determines whether a more formal Section 7 consultation is required. The Corps usually abides with the biological opinion that results from the consultation, but has occasionally contested it based on available biological information. Some projects have not been implemented as a consequence of a biological opinion. More often, the additional costs incurred to assure listed species protection are incorporated into the project costs.

Much because of Section 7 consultation requirements, the present project planning environment makes it highly unlikely that any new Corps project, including any major rehabilitation or major operations change, will further jeopardize a threatened or endangered species or a species that is a candidate for listing. Many species have been listed since the ESA was passed. Water resources projects could have contributed to the need to list them in the first place, in part because there was no comprehensive evaluation and publication of species conservation status until the 1990s. This illustrates inherent difficulties facing full assessments of cumulative effects in a knowledge-limited planning environment. Unless there were locally informed proponents among stakeholders, no provisions would be made through the NEPA process to avoid further impact.

With the recent development of the NatureServe Explorer database, this should no longer be a problem. There is no program-level policy directing attention to this database, however, nor indication that protection of imperiled and, perhaps, vulnerable species that are not protected under the ESA should be a required cost in project plans. Even so, the Corps may be cautious about embracing the information and acting upon it for several reasons.

While the database is widely used by Federal agencies, there is no official Federal sanction vouching for its validity—a potential issue for the authority-sensitive Corps. The database may need vetting by an interagency team before the Corps will recognize its technical authority. More fundamental, however, the high fraction of freshwater species now considered vulnerable and imperiled may give pause to development of such policy because of the potentially large costs. While new projects in previously undeveloped waters are rarely studied now, responding to a growing backlog of major project rehabilitation needs will undoubtedly encounter unlisted, but imperiled species. Large programmatic rehabilitation of water resources—such as for the ecosystem restoration, flood control, water supply and recreation purposes planned in South Florida—are likely to encounter large numbers of imperiled species, all of which would need careful impact evaluation and management. Actions other than protection from further decline may seem inconsistent with the Corps Environmental Operating Principles, but they emphasize "balance" between economic development and environmental protection and restoration, consistent with the Corps sensitivities to multiple interests.

Ecosystem Restoration Projects

The Corps ecosystem restoration authority provides a unique opportunity for contributing to the recovery of vulnerable species. But, relatively few ecosystem restoration project feasibility studies have clearly targeted contributions to the recovery of imperiled species in project objectives. This state of affairs owes much to the need for nonfederal sponsorship in all environmental improvement projects and the chronic deficiency of congressional funding for program level analysis and planning.

Some problems derive from unclear mission and objectives. The ecosystem restoration mission has evolved from earlier Federal policy that clearly differentiated environmental quality improvement from economic development by focusing on improving the condition of the natural environment for natural heritage and compatible enjoyment (Cole 2000a). Thus the value of improvement could be measured in terms of securing threatened elements of environmental quality from permanent loss. That focus was replaced with a more obscure concept of "significant resources," which is indicated in evidence of public desire for the resource. Of course, desire is often expressed in economic terms. Policy dealt with that by separating the Federal objective into two subobjectives, one of which addresses national economic development (NED) and the other national ecosystem restoration (NER). Thus NER was defined in large part by what it was not and requires planners to sort out all possible contributions to NED to determine NER —an often complex process. The NER objective might include securing threatened natural heritage, but other possibilities appear to be allowed as long as they clearly are not contributions to NED.

Project feasibility studies frequently lump the objective into a broad, nearly meaningless statement—such as "restoring the ecosystem to a more natural condition"—which says little or nothing about the value expected back from investment and how nationally important it is. The objectives of specific restoration projects often have been unclear and too obscurely stated to separate out economic and environmental benefits, especially for small ecosystem restoration projects (Brandreth and Skaggs 2002). Contributing to the problem is the degree to which plan formulation in ecosystem restoration projects needs to produce "a more natural condition" and non-economic environmental benefits compared to incidental NED benefits. Feasibility study preparation and review often is inefficient as a consequence, driving up expense and sometimes diverting a limited budget away from solution of important national problems.

Because ecosystem restoration projects do not require economic cost-benefit analysis, some potential nonfederal sponsors have been tempted to dress up an economic development proposal as ecosystem restoration for improved environmental quality. Policy confusion has too often allowed interests in recreation and other economically measurable service benefits to use ecosystem restoration to advance economic development without going through a cost-benefit analysis, and without very clear contribution to achievement of the national ecosystem restoration objective. While a headquarters review often screens out such efforts, some questionable projects may be recommended for implementation, possibly diverting limited funds from more clear intents to improve environmental quality.

Many more projects have been recommended than can be funded annually and compliance with the Government Performance and Results Act requires the Corps to rank its ecosystem restoration projects for funding based on beneficial contributions to national welfare. Some projects are ultimately screened out in the process. Ecosystem restoration projects have been allowed to quantify justifying environmental benefits in any number of ways as long as they cannot be acceptably monetized. The number of different metrics has proliferated. Because there was no satisfactory way to sum the different measures of environmental benefits from individual ecosystem restoration projects, an index has been developed by program-level budget planners for the prioritization purpose. It remains unclear how the evaluation of environmental benefits at the project level and the program level of strategic and budget planning are related. This problem can also complicate review and increase costs.

Thus a long-standing problem in restoration planning has gained new urgency—somehow satisfying the need for a widely accepted measure of environmental benefits that is consistent with Corps policy and can be summed project by project to estimate contribution to a national ecosystem restoration objective that maximizes benefits to the Nation. The concept of environmental benefit has its policy roots in the concept of environmental quality with attributes that are valued for other than their recreational, flood control, navigation, commercial or other economically measurable value. For Corps purposes at least, all project purposes are for economic development except ecosystem restoration. Environmental quality is determined in large part by what is not economic development and, by implication, what is natural (does not reveal human effects) or cultural heritage. In Corps ecosystem restoration policy the degradation of environmental quality is more closely linked to the concept of lost benefit than to the concept of a less natural condition, but the difference between the two is not clearly recognized by some planners.

Exactly what it is about a more natural condition or sustainable condition that is beneficial is not clear, except that the outputs are ecological and come from habitat improvement. As a consequence, a mix of measures have been used that cannot be compared for relative contribution to program benefits. In a review of possibilities for improving environmental benefits analysis in restoration projects, Stakhiv et al. (2003) identified the reduction of threats to the sustainability of natural ecosystem integrity, as characterized in its native biodiversity, as a leading contender for measuring the environmental benefits. In concept, biodiversity conservation organizations use this approach to facilitate their decisions. A measure of value based on the security of species and their communities from extinction has recently been proposed (Cole 2009b) for consideration in part because it is universally comparable across projects in the program. If a widely accepted metric of this kind were eventually adopted by the Corps, it would place more focus on restoring threatened species and their support ecosystems.

A small subset of potential nonfederal sponsors have been most interested in recovering species vulnerable to extinction and a new metric based on securing threatened biodiversity is in the conceptual phase of development (Cole 2009b). Modification of Corps authority in 1999 allowed partnering with nongovernmental organizations, such as The Nature Conservancy, which opened up opportunities for more partnerships that target improvement in the conservation status of vulnerable species. Some states spend substantial amounts on protecting and recovering "sensitive species" (Niles and Korth 2006), and are also among the most likely to partner with the Corps for this purpose. These projects are typically planned and implemented under continuing authorities with total project costs not exceeding $7 million and usually less.

There are complications with this approach, which has diminished planner interest in plan objectives that depend on the recovery of endangered and otherwise imperiled species. Planners are often cautious about possible improvement of habitat for endangered species, which may be identified among the incidental benefits from ecosystem restoration that contribute to project justification rationale. Being only incidental to another objective, such as "wetland restoration" to a more natural self-regulating condition, does not tie project success to a single species' recovery, but the possibility can count largely toward project justification despite the risk that the species never returns to the project site. Primarily because of the risk, project planners have been reluctant to target endangered species recovery among ecosystem restoration project objectives. There is little in the way of technical guidance for managing such risk, which is in need of much more attention than it has so far received. By default, downplaying imperiled species in planning objectives elevates reliance on abundant species as indicators of restoration success, which may miss the unique attributes that impart the most non-monetary environmental value to ecosystems and, in the worst cases, can act to impede recovery of the uniquely important species.

The fact that invasive nonnative species are often dominant in ecosystems complicates the restoration process. As shown in the results of this report, nonnative species have played a major role in the decline and loss of native species, and serious attempts at restoration for species made vulnerable by nonnative species have to contend with that limiting factor for indefinitely long periods in the future. The sea lamprey control program in the Great Lakes is one of the few examples of qualified success. Even qualified success is much less likely without the integration and comprehensiveness provided by dedicated programs. Thus invasive species are among the leading factors that need consideration in risk assessment and management, and in determining whether projects remain too risky for investment.

Restoring ecosystems for listed species is also complicated by numerous issues pertaining to authorities and the implementation history of the ESA (Clark et al. 1994). The importance of protecting support ecosystems for listed species is emphasized in the ESA, but the role of ecosystem restoration is not explicitly addressed. The concept of critical habitat, as implemented, is much more relevant to habitat protection than to habitat restoration. Many listed species have recovery plans, but the law and its administering agencies emphasize protection first, which can complicate restoration activities in the vicinity of listed species. Any restoration project that might impact an existing endangered species requires ESA Section 7 consultation with the FWS or the NMFS, depending on species.

The Corps needs to find willing nonfederal partners to share the cost of imperiled species recovery by way of its ecosystem restoration authority. Potential partners are often wary of the ESA and its administration. Whether or not justified, the services have acquired reputations for heavy handedness, unilateralism and exclusivity in decisions pertaining to listed species. Coupled with chronic under funding of the ESA program, this image of ESA involvement has complicated application of ecosystem restoration authorities to habitats of listed species.

For that reason, it may prove more practical for the Corps to concentrate on imperiled species not yet listed under ESA protection, thereby reducing the need for listing. To be most effective, the Corps should programmatically evaluate the need where it can potentially do the most good and seek nonfederal partnerships with compatible agencies and organizations. It might also try to influence the case for 100% Federal funding of ecosystem restoration projects once it has clearly defined the problem, its ecosystem restoration objective and how they relate to national objectives. Regardless of the funding source, however, the complexity of the challenge requires collaboration with other organizations that complement the capabilities of the Corps.

Summary

In summary, the Corps ecosystem restoration presents a unique opportunity to contribute importantly to reversing freshwater biodiversity decline. Corps ecosystem restoration and environmental protection has expanded and improved over the years. There remain many impediments, however, and further improvement is possible. One major improvement is bringing more focus to the ecosystem restoration program, including elimination of threats to a secure natural heritage. Many of the challenges faced by the Corps are based in the complexity of the planning environment, which is required by law to include nonfederal sponsors with substantial say about the process. For this and other reasons, the Corps has emphasized the need to improve collaboration with other organizations. That starts with improved communication of ecosystem restoration purpose and objectives and improved guidance on its execution.

Corps Project Operations

A greater uncertainty in Corps performance centers on Corps project operations. Many Corps-operated projects were built before the environmental legislation enacted since the NEPA was signed into law in 1970. For the most part, project operations and maintenance are decentralized and governed by individual authorities, most of which were passed over fifty years ago. While the large majority clearly comply with environmental law, the extent to which a stewardship principle has been fully embraced varies among projects and districts. More recently, efforts to integrate have intensified, starting with assessments of practices at individual projects.

The Corps manages the lands it holds in trust at the projects it operates under policy last approved in 1996. Policy objectives include managing Corps properties for the conservation and use of natural resources according to ecosystem management principles. The 1996 policy also directed inventories of natural resources, including an inventory of all "special status species." These are species that are listed under the ESA, critical habitats, candidates for listing, state-listed threatened and endangered species and species regulated under the Migratory Bird Act. These species are to be inventoried at frequencies that will determine "significant changes" in population status and managed according to Federal recovery plans and state management plans. Regular inventory provides information for adaptive management of the ecosystem.

Kasul et al. (2000) summarized the results from two mail surveys of Corps projects soliciting information about the inventoried occurrence and distribution of Federal and state listed species at Corps projects and the role they played in project management. They found that project surveys for state and Federal listed species had been completed for 13% of Corps project lands and were planned for a total of 19% completion. Listed species were found on 73% of the surveyed projects. Within the last year, the Corps moved toward a more comprehensive program approach to special status species inventory in the form of a database that field personnel are required to populate with information on listed species.

The Corps has contributed to ESA recovery plans from the beginning. Allred (1996) summarized recovery activities at Corps projects in the 1990s, finding that 68% of 456 Corps project lands and waters were likely to be occupied at times by one or more Federally listed taxa. At that time, the Corps had recovery responsibilities for 76 listed taxa, 24 of which were freshwater taxa. These included 11 species and 1 subspecies of fish, 10 species and 2 subspecies of freshwater mussels, 6 species of listed snails, and 1 species of freshwater shrimp. Recovery responsibilities vary greatly, but are in general most extensive for the birds, aquatic invertebrates, fish and freshwater reptiles.

Cole (2004) more recently reviewed all animal listings with recovery plans published on the Internet (FWS home page). As of May 2004, the Fish and Wildlife Service had identified 411 recovery plans for 448 listed animal species, subspecies, and stocks (all referred to as taxa here) on their ESA website. The needs of 86% of the 519 listed animal taxa were addressed in recovery plans. Of the 282 listed taxa with plans published on the Internet, the Corps is named as an active participant in recovery plans for 41%. If the Internet-published plans are representative, the Corps was involved with recovery of about 180 taxa in total. The extent of Corps involvement varies substantially among species according to plan needs and, to some extent, the enthusiasm of local project management.

In general, the terrestrial vertebrates disproportionately dominate Corps stewardship activities at Corps operated projects. This apparent bias links to the control of land adjacent to many of its Civil Works projects. Among recovery plans, the Corps is involved in a larger fraction of river and oceanic animal species, both aquatic and semi-aquatic. Only 24% of the mammal recovery plans involve the Corps, for example, compared to 78% of the freshwater mussels.

Habitat isolation is an important factor in determining Corps involvement. Many of the species that inhabit ecosystems that are generally isolated from Civil Works activities—such as caves, freshwater springs, vernal pools and small streams—do not involve the Corps in recovery plans. For example, of the plans sampled, the Corps is now involved with only two amphibian plans and none of the crustacean plans, even though all of the species are aquatic for at least part of their life cycle.

The Corps has dedicated significant budget to research and other management associated with species recovery plans and improved compliance with ESA protections. Compliance expenditures vary annually, but totaled at least $100 million in 2003. Much of this funding goes to other agencies for the conduct of research and assessments. In addition, the Corps Engineer Research and Development Center has completed significant research on freshwater mussels and fish. It provides technical notes and other information to facilitate more careful project planning. Recently compiled information, such as that in NatureServe Explorer, adds greatly to the resources available for more environmentally friendly planning and construction.

By comparison, terrestrial management for improved security of species and ecosystems has been less complicated than aquatic management, especially in the rivers where some of the most intense threats to ecosystem sustainability exist. This has less to do with differences between terrestrial and aquatic ecosystems than with property concepts in law. Conservation organizations can and have purchased terrestrial sites to manage them consistent with their biodiversity missions. The Nature Conservancy has been a leader in this concept. They also can buy water rights to restore and protect freshwater ecosystems in western states that allow private ownership of water. Federal land management agencies have similar discretion, consistent with law, on the lands they hold in trust for the public (but Congress has to approve purchase and sale of public lands). This includes the several million acres of land held by the Corps.

Much of the navigable low-flow channel bottoms of the larger freshwaters are public lands held in trust by the states. The commerce clause of the constitution is interpreted as giving the Federal government the right to regulate and manage channel bottoms without explicit permission, including the right to dredge and maintain navigation channels. Ecosystem restoration authority is more complex. Except for threatened and endangered species and certain migratory species, public ownership of fish and wildlife is held in trust by the states. By implication, the Corps must partner with the states with respect to any resource development having an impact on fish and wildlife, including restoration of habitat for fish and wildlife species. For species protected under the ESA they must confer with and probably partner with the FWS or the NMFS.

The Corps has begun to consider sediment differently than it has in the past, more as a resource to be managed for both economic and environmental gain than simply an impediment to navigation and sustained reservoir service to flood control, hydropower and water supply purposes. In this concept, the Corps is a water and sediment resource management agency. Because sediment movement behaves differently than water in managed river systems, the interaction of watershed development with dams and other water resources development has very different environmental implications when examined from the perspective of sediment. Virtually all of the imperiled freshwater species in river systems are as intimately linked to the bottom and the effects of sediment transport and deposition as to the quality, quantity and dynamics of the water. The concept of regional sediment management as applied to Civil Works operations, because of its relevance to traditional project purposes, is a promising means for moving project operations from a strict project focus to more of a program management view.

Greater awareness of environmental improvement opportunities through Corps operations are likely to emerge with that reorientation.

In summary, Corps operations emphasize strict compliance with environmental law, including the ESA, but limit stewardship principles to the lands they hold in trust consistent with congressional authorities and restrictions. Natural resource stewardship policy reflects growing commitment to sustaining species vulnerable to extinction on Corps lands. However, based on its sense of authority, the responsibilities of other agencies and, perhaps most importantly, the costs that are involved, the Corps has been more limited in its approach to project management for aquatic species. In one primary example, The Nature Conservancy has initiated a number of joint ventures with the Corps through its Sustainable Rivers program to find ways to modify dam and other structural operation to restore freshwater biodiversity while maintaining project obligations. Further efforts in that direction would be consistent with more of an ecosystem restoration emphasis on reducing threats to freshwater biodiversity.

Performance of the Corps since the ESA Passage

The apparent success of the ESA in preventing species extinction (NRC 1995, Scott et al. 2006) is, in part, because of the efforts made by agencies, including the Corps, to apply the NEPA process and to comply with Section 7 requirements and other environmental law. While some older Corps projects probably have contributed to decline of some ESA candidate species, there is no clear evidence that Corps projects planned and built in the last two decades have contributed to candidate listing. Many of the candidate species are most threatened by non-point source pollutants from land use practices. The majority of candidate snails are freshwater spring snails and terrestrial snails in habitats well beyond Corps influence. The 10 freshwater mussel candidates follow similar patterns as ones already listed as threatened and endangered. Several older Corps dams are located in the ranges of candidate mussels, fish and a turtle species, but none of those dams have been built recently. Only one Corps project is explicitly linked as a threat to the status of an ESA candidate species, a fish, and it was completed in 1963 at Greer's Ferry Dam. However, many aquatic species thought to be imperiled have yet to be considered as candidates for listing.

While the ESA has prevented extinction, extinction itself has been a common way off of the ESA list for threatened and endangered freshwater species (Clark 1994, Scott et al. 2006). Two terrestrial taxa and five freshwater taxa were delisted because of extinction. Of those, two were bird subspecies, four were fish species and subspecies, and one was a freshwater mussel species. There are small signs of improvement for the plight of listed freshwater and terrestrial species. Of animal taxa that have changed listed status under the ESA, six freshwater species improved in status and one worsened. The latter was the Alabama cavefish, which lives in a single cave system and was exposed to increased threats from land use above the cave. The Corps had no obvious effect on any of the changes in ESA status.

Only those freshwater taxa of recreational interest, namely several subspecies of trout, improved because of active management. Other freshwater taxa improved because previously missed populations were discovered. Two stocks of salmonids in the Columbia River system have also been changed from threatened to endangered status. Although both actions are relatively rare, down-listing to threatened status has been somewhat more common than up-listing to endangered status.

Richter et al. (1997) surveyed a group of conservation professionals who perceived that water resources development and management have become relatively less important among negative impacts on species at risk of extinction. For example, waterways navigation fell from 9.2% of the identified threats to species in historic cases to less than 5% of the identified threats in contemporary cases. Power generation fell from 21.4% to 18.7%. Threats from nonnative exotic species increased from 18.7% to 32.6%. By comparison, the combined impact of agricultural and municipal land use changed little, from 90.8% to 90.2% involvement.

These perceptions are independently consistent with the results of this study. The perceptions of conservation professionals also may reflect a growing appreciation for the complexity of interactions among changes in waterway biology, hydrology, water quality, water quantity and sediment structure in ways that may make focusing on any one category of impacts inefficient in realizing improvement. The combination and sequence of actions taken will be critical for efficient use of public funds and possibly for doing more good than harm. It is therefore difficult to judge the effectiveness of the Corps independent of the institutional context that limits Corps effectiveness. Positive Corps actions may not result in biodiversity improvement if a suite of other factors continues to limit habitat suitability. Such actions may become irresponsible whenever other benefits are forfeited without any resultant improvement in species status.

The decreased relative concern about water resources development impacts does not reflect strongly in measurable contributions to improved habitat conditions for listed species. More evidence of significant species recovery would result if they did. The shift in professional judgment is more apt to reflect a new concern for invasive species added to old concerns rather than replacing them. If species recovery to secure status is the gauge of success, the Federal agencies collectively, including the Corps, cannot show substantial contribution to progress. The status of very few species has recovered enough to delist them over the past 30 years that the ESA has been in effect (Scott et al. 2006).

None of the few animal species that have been recovered are freshwater species. All of the species that have recovered have responded primarily to improved harvest regulation and/or improved regulation of pesticides. This includes the southern Bald Eagle. The Corps has been significantly involved in the Bald Eagle recovery plan because of the importance of Corps reservoirs as Bald Eagle habitat. However, except for reduced pesticide contamination to date, no species has recovered primarily as a consequence of habitat improvements.

Burnham et al. (2006) discuss the lessons learned for species recovery. Reasons given for meager results are most fundamentally linked to insufficient funding. Demands for considering new listings have overwhelmed the FWS and NMFS programs. Some argue that there should be more moratoriums on listing while others argue that ignoring the need for consideration could result in the loss of more species. The scientific information needed to proceed with effective recovery is often lacking. Agencies and private land owners often lack incentives for increasing investment of scarce operations funds into recovery actions. There are no national and regional recovery priorities and responsibilities assigned to participating Federal agencies. Furthermore, secure recovery often requires land acquisition by Federal agencies, which is now politically unpopular. Yet land set-asides for the recovery and protection of terrestrial and aquatic resources may be the biggest single need. Estimates of need are several times the existing network of appropriately managed lands and 20 to 30% of some state's land area (Schaffer et al. 2000).

Because of insufficient funding, the FWS and NMFS have set priorities on attention paid to listed species. Early in the Act's history, Yaffee (1982) described specifically how this was done using 1) a degree of threat rating, 2) a taxonomic factor and 3) an ecological/socioeconomic factor. Of the three, only the threat rating is clearly indicated to be appropriate in the ESA language. Most relevant with respect to the Corps is that the taxonomic factor favored vertebrates over invertebrates and impacts on species of economic value and public "popularity" took priority. The continued bias in listing, although diminishing (Scott et al.2006), is indication that priorities have not completely adjusted to the expressed intent of the ESA. This bias has continued to play out in the interactions with the Corps activities, which have most negatively influenced invertebrates, but spend much more on protection and recovery of large fish, turtles, birds and mammals—consistent with FWS and NMFS emphasis.

There is no evidence that the Corps has performed exceptionally, for better or worse, among Federal agencies. The problems tend to be systemic. Led by the Departments of Interior and Commerce, the Nation seems to be protecting ESA listed species reasonably well, but is failing by and large to recover them to a secure and sustainable status. The net result for the large majority of species is greater stability of population status once they were listed under ESA protection (NRC 1995, Scott et al. 2006), but meager recovery regardless of whether or not the Corps is actively involved. New listing considerations continue to dominate the ESA process and the net number of species listed continues to increase. Many candidate species remain under consideration.

The generally accepted explanation for this predicament is the inadequacy of resources made available by Congress because the public has other priorities for its tax dollars, such as the growing obligation to mandatory social security and related services. It may be possible for agencies to improve their efficiency, however. For example, records of expenditures on threatened and endangered species management are often incomplete and difficult to interpret with respect to activities and effectiveness, and the true costs may be underestimated (Simmons and Frost 2004).

The Potential For Doing More

Budget Limitations

In its policy and practice, the Corps takes the position that it has followed the lead of Congress, the President and authorized regulatory agencies in its respect for environmental law, including compliance with ESA requirements. It is less clear how much more can be done under existing constraints. As for other natural resource agencies, past budget trends are not encouraging. Greater Corps involvement depends on greater budget funding, which in turn depends on which spending is assigned highest priority by Congress. While the Corps has sought and is recommended by NRC (2004a) to continue to seek programmatic funding from Congress for a more integrated approach to its project planning and management, it has yet to be successful. Eventual success may require the Corps to first demonstrate that capability through judicious reallocation of its operations and maintenance budget despite intense demand for those funds for other management purposes.

Projects and Programs

Because the Civil Works program remains primarily project driven, it has yet to invest much in program-level evaluations, including freshwater biodiversity recovery potential. Such evaluation could be used to proactively encourage restoration project partnerships with nonfederal sponsors and to facilitate evaluation of restoration project plans, which must be judged subjectively. Little consistent effort has been made to determine if project operations have had remote impacts on sensitive species off project lands and waters. Neither have any of the studies identified special status species that might be expected to occur on project lands and in project waters but did not show up in inventories. This type of information would be particularly useful for asking how Corps project operations and maintenance might be contributing to the presently restricted distribution pattern and what might be done to improve the status of those species in the most cost-effective way while respecting obligations.

Within the context of project-focused funding, the ecosystem restoration authority holds great potential for contributing to the recovery and sustainability of freshwater biodiversity. Mission investment trends raise questions about how well this potential might be used, however. In the absence of a program plan for prioritizing ecosystem restoration for any specified national objective, such as recovery of biodiversity to secure status, restoration plays out according to the priorities of nonfederal sponsors. Corps policy places high value on involving many such "collaborators" in large projects, and that is consistent with contemporary academic and biodiversity conservancy thinking (Yaffee 2006, Groves 2003).

Partly as a consequence, ecosystem restoration funding is increasingly being directed toward multi-project, multipurpose activities of regional scale that rehabilitate the existing "footprint" of past water resources development projects. The most prominent example to date is the Comprehensive Everglades Restoration Plan (CERP), which will require several decades to implement. While many positive aspects can be cited, future investments in a few very large rehabilitation activities could co-opt investments in restoration projects with higher returns in increased biodiversity security or other high environmental benefit. Without more investment in program planning, the Corps is easily swept into local and regional priorities that may prove less beneficial to the Nation than "might have been."

Perhaps the most promising approach toward more programmatic thinking is through strategic planning. While the strategic planning process is required by GPRA, the Corps has the opportunity to take greater ownership and use the process to encourage a more programmatic watershed or other eco-regional approach to managing its projects, as advocated by the NRC (2004a and b). In its Civil Works strategic plan, project planning policy and stewardship policy, the Corps aspires to a more integrated systems approach to water resources management that fully incorporates environmental concerns in its quest to maximize public benefits. Achievement requires a paradigm shift from project-focused management to a regional program view of the economic, social, political and ecological systems that interact to define the appropriate balance between environment and development. Some changes are underway to start the shift, including a growing emphasis at the highest level of leadership on underlying principles and development of improved management tools at the needed scales.

Policy

The Corps has an important opportunity to contribute to restoring the Nation's threatened freshwater biodiversity through its ecosystem restoration authority and more limited operations authorities. As described already, the Corps has made some progress in this regard. The Corps has produced much useful environmental policy in recent decades, but it has evolved incrementally and remains somewhat fragmented in language and principle. The concept of environmental sustainability and its achievement in the Civil Works program is often implied but not explicitly integrated into existing Civil Works planning documents. Greater clarity with respect to which ecological resources merit the greatest attention in achieving a national ecosystem restoration program objective could also promote greater attention to securing the Nation's freshwater biodiversity.

Any "hotspots" where numerous such special-status species concentrate ought to receive priority protection and recovery attention (e.g., Bibby 1995). "Not all species are equal" (May et al. 1995), however. In addition to level of threat, the evolutionary uniqueness of the threatened taxa is a useful criterion for judging decisions about where to place recovery investments, including Corps ecosystem restoration projects when feasible (Cole 2009b). Other important considerations have to do with species contributions to ecosystem functions and associated services. On the other side of the ledger is consideration of the tradeoffs required for restoring the security of freshwater biodiversity. A detailed assessment of conditions at and influenced by the Corps and other Federal water resources development agencies is needed to ascertain the sustainability status of potentially impacted biodiversity and what might be done to improve it.

The Corps is not likely to fully transcend its past without the support of the social system in which it functions, however. Just as it has not stood alone among the causes of past species extinction and imperilment, the Corps cannot stand alone in the solution of one of the more challenging problems of our day—the progressive loss of the earth's unique global biodiversity. It needs to take more seriously its commitment to collaboration and to redirect its planning and operation policy to the regional scales now embraced by biodiversity conservancies (Groves 2003) and join with them as well as state governments to plan more thoroughly at those levels.

SUMMARY AND CONCLUSIONS

1. Agents of water resources development in the United States, and the U.S. Army Corps of Engineers in particular, have been criticized for being among the primary causes of past and present freshwater species loss, which is claimed to be greater than terrestrial species loss and still growing, despite nearly four decades of Federal policy directing otherwise.

2. Documentation of species losses and their causes has been uneven at best and often uncertain, but some conclusions can be made with reasonable confidence about the validity of criticism.

3. Based on this study of animal indicator species, extinction rate in the continental United States has increased significantly during the past century, especially among freshwater species, which are becoming extinct at about 3 to 20 times the terrestrial rate, depending on the taxonomic groups included in the analysis. The result is, in general, consistent with the conclusions of a past scientific analysis by Ricciardi and Rasmussen published in 1999.

4. Estimates of future extinction rates from trend extrapolation are substantially lower than estimates based on present species imperilment and endangerment, and are more consistent with the increased environmental awareness and protective law that has emerged over the past several decades. A strong bias toward terrestrial and vertebrate species protection under the ESA may contribute to a projected increase in the ratio of freshwater to terrestrial species extinction based on extrapolation of past trends in the continental United States. However, the projected ratios varied widely among different methods used, consistent with the large uncertainty associated with all of the methods.

5. Extinction rate estimates and their uncertainty vary significantly among taxonomic groups and geographical areas. Mollusks have undergone especially high estimated rates of extinction, but are less certainly documented than other groups. Whereas freshwater extinctions rates are significantly greater than terrestrial rates for the continental United States, the difference becomes negligible when Hawaii is included.

6. Freshwater species extinction and imperilment are common in isolated aquatic habitats of semi-arid to arid landscapes, but are more common in warmwater rivers where the water resources have been largely developed for navigation, flood control, water supply and hydropower. Despite this association, however, most freshwater species extinction and imperilment is attributed to urban-agricultural development and invasive nonnative species.

7. Federal water resources development agencies were most certainly involved (probably without knowledge) in a small fraction of fish extinctions (the Corps of Engineers was only peripherally involved). They probably were involved in more than half of the invertebrate extinctions. The Corps is implicated in about one-fourth to one-third of the invertebrate extinctions. The causes of contemporary freshwater species imperilment

identified in conservation databases are similar in general to the causes of past extinction and are reported in similar proportion.

8. There is little evidence that the Corps has been less compliant with environmental laws, including the ESA, than other natural resource agencies. Corps policy, planning and operations have adapted to public expectations for balance between development and environmental sustainability, and continue to evolve with an institutional intent to maximize benefits to present and future generations.

9. Because of its unique ecosystem restoration authority, the Corps has an outstanding opportunity to contribute to the reversal of freshwater species decline. Freshwater species imperilment is largely caused by habitat change, which in many aquatic ecosystems has altered habitat throughout the range of many species. With little natural habitat left to protect, ecosystem restoration is the only option for recovery of many species to a sustainable state.

10. The ecosystem restoration authority of the Corps has not been used as much as it could to recovery the security of the Nation’s freshwater biodiversity for technical, policy and other reasons that need to be broadly addressed to realize the full potential of the restoration authority. A detailed assessment of environmental resources is needed to ascertain the sustainability status of ecosystem biodiversity and what might be done to improve it using restoration and other authorities.

11. Federal water resources projects, including those of the Corps, now interact with water flow, sediment, nutrient, invasive species and other factors in ways that often can be managed for species protection and recovery and for greater overall public benefit. Improvements depend on better understanding of geophysical, ecological, economic, social and political systems; collaborative planning; and management skills applied at scales larger than project areas to achieve more completely integrated water resources management.

12. Like all other government agencies and NGOs, the Corps budget is limited. To be more effective in using scarce funding to protect and restore species security through its ecosystem restoration and other planning and operations, the Corps needs to become more competent, systems oriented and collaborative in its planning and project implementation at large ecoregional scales.

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