Courtesy N.M. Burkhead, NBS

Fishes

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Overview

The inescapable conclusion from the data presented in this section is that within historical time, native fish communities have undergone significant and adverse changes. These changes generally tend toward reduced distributions, lowered diversity, and increased numbers of species considered rare. These changes have been more inclusive and more dramatic in the arid western regions where there are primarily endemic (native) species, but similar, though more subtle changes, have occurred throughout the country. These trends are the same whether one focuses on faunas (Johnson; Starnes; and Walsh et al., this section) or on populations or genetic variation within a single species (Marnell; Miller et al.; and Philipp and Claussen, this section). Changes in fish communities may be indicative of the overall health of an aquatic system; some species have narrow habitat requirements.

The fact that fish populations have changed over historical time should not come as any great surprise. We have massively modified fish habitat through the very water demands that define our society (domestic, agricultural, and industrial water supplies; waste disposal; power generation; transportation; and flood protection). All of these activities have resulted in controlling or modifying the flow or degrading the quality of natural waters. In addition, almost all contaminants ultimately find their way into the aquatic system. Species of fishes that have evolved under the selection pressures imposed by natural cycles have often been unable to adapt to the changes imposed on them as a result of human activities.

Physical and chemical changes in their habitats are not the only stresses that fishes have encountered over time. Through fish management programs, the aquarium trade, and accidental releases, many aquatic species have been introduced to new areas far beyond their native ranges. Although these introductions were often done with the best of intentions, they have sometimes subjected native fish species to new competitors, predators, and disease agents that they were ill-equipped to withstand.

The data presented by Philipp and Claussen (this section) further suggest that managed fish populations (hatchery-stocked populations) have a lower genetic diversity than unmanaged populations. In other words, theoretically, the smaller the gene pool, the less likely a species may be able to adapt to changing environmental conditions.

It appears unlikely that the forces that have led to these changes in our fish fauna will lessen

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significantly in the immediate future. Therefore, if we are to preserve the diversity and adaptive potential of our fishes, we must understand much more of their ecology. Vague generalizations about habitat requirements or the results of biotic interactions are no longer enough. We must know quantitatively and exactly how fishes use habitat and how that use changes in the face of biotic pressures. Only when armed with such information are we likely to reduce the current trends among our native fishes.

Imperiled Freshwater Fishes

by
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The United States is blessed with perhaps 800 species of native freshwater fishes (Lee et al. 1980; Moyle and Cech 1988; Warren and Burr 1994). These fishes range from old, primitive forms such as paddlefish, bowfin, gar, and sturgeon, to younger, more advanced fishes, such as minnows, darters, and sunfishes. They are not equally distributed across the nation, but tend to concentrate in larger, more diverse environments such as the Mississippi River drainage (375 species; Robison 1986; Warren and Burr 1994). Drainages that have not undergone recent geological change, such as the Tennessee and Cumberland rivers, are also rich in native freshwater fishes (250 species; Starnes and Etnier 1986). Fewer native fishes are found in isolated drainages such as the Colorado River (36 species; Carlson and Muth 1989). More arid states west of the 100th meridian average about 44 native fish species per state, while states east of that boundary average more than three times that amount (138 native species; Figure).

Extinction, dispersal, and evolution are naturally occurring processes that influence the kinds and numbers of fishes inhabiting our streams and lakes. More recent human-related impacts to aquatic ecosystems, such as damming of rivers, pumping of aquifers, addition of pollutants, and introductions of non-native species, also affect native fishes, but at a more rapid rate than natural processes. Some fishes are better able to withstand these rapid changes to their environments or are able to find temporary refuge in adjacent habitats; fishes that lack tolerance or are unable to retreat face extinction.

In 1979 the Endangered Species Committee of the American Fisheries Society (AFS) developed a list of 251 freshwater fishes of North America judged in danger of disappearing (Deacon et al. 1979), 198 of which are found in the United States. A decade later, AFS updated the list (Williams et al. 1989), noting 364 taxa of fishes in some degree of danger, 254 of which are native to the United States. Both AFS lists used the same endangered and threatened categories defined in the Endangered Species Act of 1973, and added a special concern category to include fishes that could become threatened or endangered with relatively minor disturbances to their habitat. These imperiled native fishes are the first to indicate changes in our surface waters; thus their status provides us with a method of judging the health of our streams and lakes. This article compares the two AFS data sets to assess the trends in the status of freshwater fishes in the United States over the past decade.

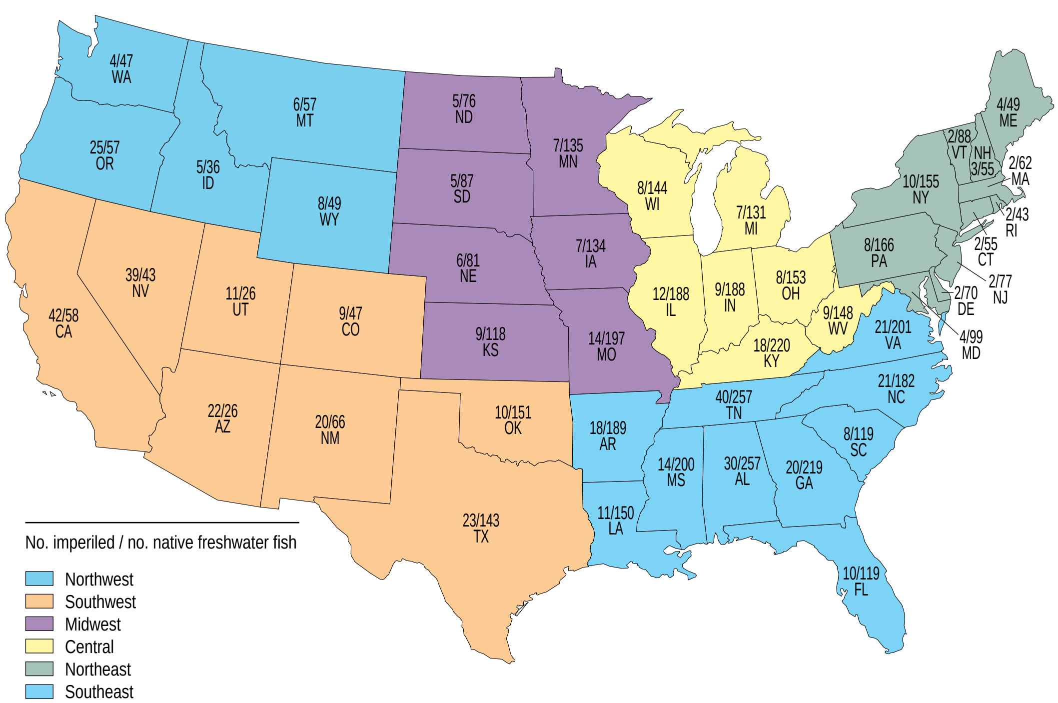

Figure. Number of fishes considered imperiled and number of native freshwater fishes of the contiguous United States by state (redrawn from Warren and Burr 1994).

Basis of the American Fisheries Society Listings

The 1979 and 1989 AFS listings were based entirely on biological considerations throughout the geographic range of the taxon and ignored jurisdictional or political considerations. For example, the johnny darter (*Etheostoma nigrum*) is a small darter found in clear streams from the East coast to the Continental Divide; the species reaches the western periphery of its range in Colorado. Johnny darters are rare in Colorado, which recognizes the species' rarity (Johnson 1987). Throughout most of its range, however, the johnny darter is common and thus was not included in the AFS listing. Only those taxa that appear imperiled are included in the lists; populations were not considered unless they were distinct enough to be recognized as subspecies.

The preliminary 1979 AFS listing was obtained by asking knowledgeable fishery scientists which fishes should be included. Those taxa were added to a 1972 listing of protected fishes (Miller 1972) that was then sent out to every state and to selected federal agencies for review.

The native fish faunas of some areas of the country are better studied than others and may therefore be better represented in the listing. The 1989 listing used knowledgeable biologists but not extensive agency review to build upon the 1979 listing. These two data bases provide the best information presently available on rare native fishes of the United States.

Changes in the Status of Native Freshwater Fishes, 1979-89

Analysis of the 1989 list provides some basic information on the status and trends of the native fishes of the United States. About one-fourth of our native freshwater fishes are perceived to be imperiled. Ninety-three percent of imperiled species are in trouble because of the deteriorating quality of the aquatic habitats on which they depend; this deterioration results from physical, chemical, and biological effects to our surface waters and underground aquifers. Overuse, introduction of non-native species, disease, and other problems that also affect our native fishes cause much less endangerment than habitat destruction.

The increase of taxa of fishes between the 1979 (189 taxa) and 1989 (254 taxa) AFS listings does not include 19 taxa that were removed from the 1989 listing because of extinction, taxonomic revisions, or better information on status. Seventy-five imperiled taxa that did not appear in the 1979 AFS listing were added to the 1989 AFS listing, an increase of 38% in a single decade. In addition, the status of 39 fishes was changed: 7 taxa improved (e.g., changed from threatened to special concern), 22 taxa declined, and 10 taxa were recognized as extinct (Table). No fish was removed from the 1989 AFS listing because of successful recovery efforts, indicating that our freshwater fishes continue to decline overall, and factors causing those changes appear difficult to reverse.

Species	Population trend
Pallid sturgeon (*Scaphirhynchus albus*)	Declined
Longjaw cisco (*Coregonus alpenae*)	Extinct
Deepwater cisco (*C. johannae*)	Extinct
Blackfin cisco (*C. nigripinnis*)	Extinct
Alvord cutthroat trout (*Oncorhynchus clarki* ssp.)	Extinct
Fish Creek Springs tui chub (*Gila bicolor euchila*)	Improved
Independence Valley tui chub (*G.b. isolata*)	Extinct
Thicktail chub (*G. crassicauda*)	Extinct
Chihuahua chub (*G. nigrescens*)	Improved
Least chub (*Iotichthys phlegethontis*)	Declined
White River spinedace (*Lepidomeda albivallis*)	Declined
Cape Fear shiner (*Notropis mekistocholas*)	Declined
Blackmouth shiner (*N. melanostomus*)	Declined
Oregon chub (*Oregonichthys crameri*)	Declined
Blackside dace (*Phoxinus cumberlandensis*)	Declined
Loach minnow (*Rhinichthys cobitis*)	Declined
White River sucker (*Catostomus clarki intermedius*)	Declined
Zuni bluehead sucker (*C. discobolus yarrowi*)	Improved
Shortnose sucker (*Chasmistes brevirostris*)	Declined
June sucker (*C. liorus mictus*)	Declined
Lost River sucker (*Deltistes luxatus*)	Declined
Razorback sucker (*Xyrauchen texanus*)	Declined
Pygmy madtom (*Noturus stanauli*)	Declined
Alabama cavefish (*Speoplatyrhinus poulsoni*)	Declined
Preston springfish (*Crenichthys baileyi albivallis*)	Declined
White River springfish (*C.b. baileyi*)	Declined
Moorman springfish (*C.b. thermophilus*)	Declined
Railroad Valley springfish (*C. nevadae*)	Declined
Devils Hole pupfish (*Cyprinodon diabolis*)	Improved
Desert pupfish (*C. macularius*)	Declined
Amistad gambusia (*Gambusia amistadensis*)	Extinct
San Marcos gambusia (*G. georgei*)	Extinct
Gila topminnow (*Poeciliopsis occidentalis*)	Improved
Spring pygmy sunfish (*Elassoma* sp.)	Improved
Sharphead darter (*Etheostoma acuticeps*)	Improved
Amber darter (*Percina antesella*)	Declined
Blue pike (*Stizostedion vitreum glaucum*)	Extinct
Utah Lake sculpin (*Cottus echinatus*)	Extinct
Shoshone sculpin (*C. greenei*)	Declined

Table. Population trends for endangered, threatened, and special concern freshwater fishes of the United States whose status changed between 1979 and 1989 (Williams et al. 1989).

The relation between declining aquatic habitats and fishes facing extinction is not as simple as might be expected. Species with limited distributions are more likely to be jeopardized by changes in their local aquatic habitats than are species with extensive ranges. Many fishes on the lists have local distributions, and a few, such as the Clear Creek gambusia (*Gambusia heterochir*) and Devils Hole pupfish (*Cyprinodon diabolis*), are limited to a single spring. These unique fishes could be lost by a single, isolated event. Some of the widespread species included in the listings—such as paddlefish (*Polyodon spathula*) and six taxa of sturgeons—depend on large rivers, and their inclusion indicates widespread threats to these extensive habitats.

States with the most listed (imperiled) species include California (42), Tennessee (40), and Nevada (39). Somewhat fewer listed fishes are found in Alabama (30), Oregon (25), Texas (23), Arizona (22), Virginia (21), North

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Carolina (21), New Mexico (20), and Georgia (20; Figure). Regionally, the Southwest has the highest mean number of fish species listed per state (22.5), closely followed by the Southeast (19.3); the northeastern states have the lowest mean number of native fish species in trouble (3.7). Nearly half (48%) of the southwestern native fishes are jeopardized, followed by fishes of the Northwest (19%), the Southeast (10%), the Midwest (6.4%), the central states (5.9%), and the Northeast (4.3%; Warren and Burr 1994).

The AFS will likely update its listing of native fishes in peril toward the end of this decade, thus providing us with more than 20 years of information on the status of these fishes, a short time in the overall life of a species but a good data base upon which to evaluate the environmental health of our streams and lakes. If the trend over the last decade continues, we can expect a further decline in the richness of our native fishes. In addition, as aquatic habitat deterioration becomes more extensive, we can expect to see an increase in the listing of widespread fishes.

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Southeastern Freshwater Fishes

by
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North America has the richest fauna of temperate freshwater fishes in the world, with about 800 native species in the waters of Canada and the United States. The center of this diversity is in the southeastern United States, where as many as 500 species may exist (62% of the continental fauna north of Mexico). Many coastal marine species also enter fresh waters of the Southeast, and at least 34 foreign fish species are established in the region.

Although freshwater fishes of the United States are better studied than any fish fauna of comparable scope in the world (Lee et al. 1980; Hocutt and Wiley 1986; Matthews and Heins 1987; Page and Burr 1991; Mayden 1992), large gaps exist in scientific knowledge about the biology and ecology of most species. New species are still being discovered, and the taxonomy of other species is being refined.

Seriously declining populations of freshwater fishes in the United States concern the scientific community (Deacon et al. 1979; Williams et al. 1989; Moyle and Leidy 1992; Warren and Burr 1994). This article briefly summarizes the current conservation status of southeastern freshwater fishes; the Southeast is emphasized because of its important fish biodiversity and to focus attention on the growing

Courtesy N.M. Burkhead, NBS

Principal causes of declining fish resources in the Southeast are due to habitat perturbations, such as loss of forested stream cover, mining activities, and impoundments, as at this site in northern Georgia.

problem of adverse human impacts on the region's aquatic habitats (Mount 1986; Burkhead and Jenkins 1991; Etnier and Starnes 1991; Warren and Burr 1994).

Hydrologic Regions

The southeastern United States as defined here is delimited on the north and west by the Ohio and Mississippi rivers. The following hydrologic regions (Fig. 1) are defined on the basis of common geophysical characteristics and similar fish faunas of the drainages within

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each region (Hocutt and Wiley 1986): (a) Atlantic Slope—coastal waters from the Roanoke River (Virginia) southward to the Altamaha River (Georgia); (b) Peninsular—waters from the Satilla River (Georgia) to the Ochlockonee River (Florida); (c) Lower Apalachicola Basin—waters from the Apalachicola River (Florida) westward to the Perdido River (Alabama); (d) Lower Mobile Basin—lowland portions of the Tombigbee and Alabama rivers and tributaries (Alabama and Mississippi); (e) Lower Mississippi—the Mississippi River and its eastern tributaries below the Ohio River (Mississippi, Tennessee, and Kentucky); (f) Interior Plateau—upland waters of the middle and lower Ohio River and southern tributaries, including the lower Cumberland and Tennessee rivers (Kentucky and Tennessee); and (g) Southern Appalachians—upland waters of the mountains in the geological provinces known as the Cumberland Plateau, Valley and Ridge, Blue Ridge, and Piedmont, south of the Kanawha (West Virginia) and Roanoke rivers. Many fishes are widely distributed in the Southeast and occur in two or more hydrologic regions.

Imperiled Freshwater Fishes

The Southeast has about 485 known species of native freshwater fishes, representing 27 families. Most of the diversity of the southeastern fish fauna is in five families: the darters and perches (family Percidae; 31.3%); the minnows (family Cyprinidae; 29.7%); the madtoms and bullhead catfishes (family Ictaluridae; 6.8%); the suckers (family Catostomidae; 6.6%); and the sunfishes and basses (family Centrarchidae; 5.8%). The greatest diversity is in the Appalachian Mountains and Interior Plateau (Fig. 1), but other regions of the Southeast also harbor many more species than do similar-sized geographic areas elsewhere in the United States.

As of January 1994 the U.S. Fish and Wildlife Service (USFWS) had designated 15 southeastern fish species as endangered and 12 as threatened, representing 6% of the entire regional fish fauna. Ninety-three fish taxa (19%) are imperiled (endangered, threatened, or of special concern) in the Southeast, including proposed listings and those recognized by other authors (Williams et al. 1989). During the past 25 years, only seven species were upgraded by the USFWS, mainly because of discovery of new populations, inadequate knowledge at the time of listing, or invalid taxonomy. No endangered or threatened species have been delisted. A steady upward trend in designation of imperiled southeastern fishes has occurred in the last 20 years (Fig. 2); the number of species considered imperiled by the USFWS increased from 3 (less than 1%) in 1974 to 84 (17%) in 1994 (USFWS listings only). During the 10-year period from 1979 to 1989, the number of species considered imperiled by the American Fisheries Society increased from 63 (13%) to 81 (17%; Fig. 2).

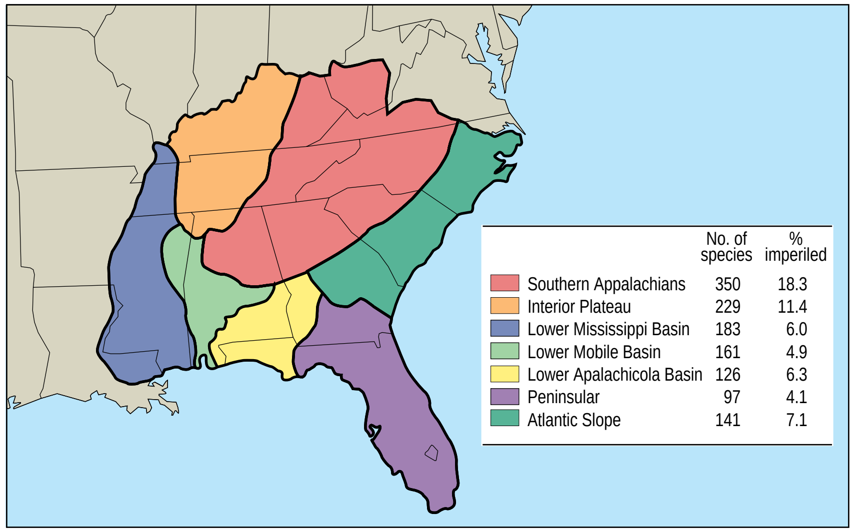

Fig. 1. Total numbers of freshwater fishes and percentage imperiled by hydrographic region of the southeastern United States.

An alarming 21% of the nearly 300 species of minnows and darters are imperiled in the Southeast. Considered alone, more than 30% of the 150 species of darters are in trouble, representing the highest total number of species in any one family. Madtom catfishes (genus *Noturus*) are also disproportionally imperiled among large families of more than 30 species (Etnier and Starnes 1991; Warren and Burr 1994). Among smaller groups of fishes, the most severe status is among the sturgeons and paddlefish, where seven of the eight (86%) southeastern species are in jeopardy. In terms of ecological requirements, most imperiled species are those that live in small to large creeks and small rivers, are closely associated with clean stream-bottom substrates, or are isolated in spring and cave environments.

On a regional scale, the greatest number of imperiled species occurs in the highland areas of the Appalachians and Interior Plateau,

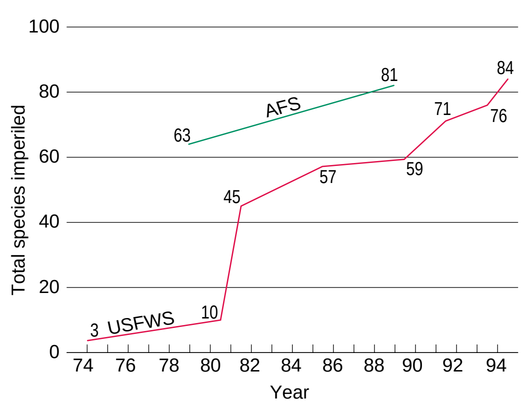

Fig. 2. Total numbers of imperiled fishes in the Southeast during the last 20 years, as recognized by the American Fisheries Society (AFS) and the U.S. Fish and Wildlife Service (USFWS). Numbers represent imperiled species during years of listing activity.

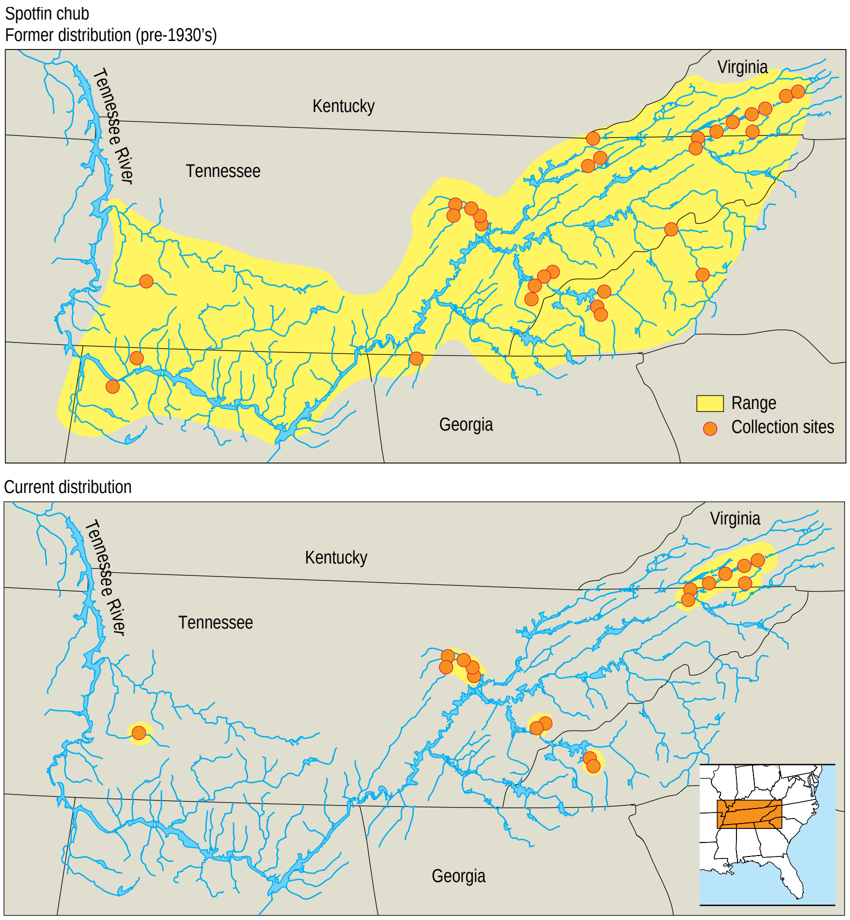

Fig. 3. An example of habitat fragmentation, decline, and isolation of populations of a southeastern freshwater fish, the endangered spotfin chub (*Cyprinella monacha*). Former (pre-1930's) and present range in yellow.

Courtesy N.M. Burkhead, NBS

Tangerine darter (*Percina aurantiaca*).

Courtesy N.M. Burkhead, NBS

Mountain redbelly dace (*Phoxinus oreas*).

followed by the Coastal Plain subregions (Fig. 1). This geographic trend is correlated with both a high level of diversity in the respective hydrologic regions and the quite localized or endemic distributions of many species. Especially important are a number of watersheds that harbor many species confined within those drainages; these watersheds include the Tennessee River, the Mobile Basin, the Cumberland River, and the Roanoke and James rivers (Warren and Burr 1994). Most jeopardized species have restricted distributions, but the number of more geographically widespread species that are disappearing from large portions of their ranges is increasing.

Two species of southeastern fishes have become extinct in the last century: the harelip sucker (*Moxostoma lacerum*) and the whiteline topminnow (*Fundulus albolineatus*). At least one other species, the least darter (*Etheostoma microperca*), has disappeared from the southern portion of its range that falls within the region covered here. The slender chub (*Erimystax cahni*) has not been seen since 1987 and may be near extinction. Two other species peripheral to the Southeast are feared extinct: the Scioto madtom (*Noturus trautmani*) and the Maryland darter (*Etheostoma sellare*; Etnier 1994).

The declining status of freshwater fishes among divergent taxonomic groups and across broad habitat types and geographic areas is interpreted as evidence for widespread and pervasive threats to the entire North American fish fauna (Moyle and Leidy 1992; Warren and Burr 1994). In the Southeast, fish declines are the result of the same factors that cause global deterioration of aquatic resources, primarily habitat loss and degraded environmental conditions. The principal causes of freshwater fish imperilment in the Southeast and other areas of the United States are dams and channelization of large rivers, urbanization, agriculture, deforestation, erosion, pollution, introduced species, and the cumulative effects of all these factors (Moyle and Leidy 1992; Warren and Burr 1994). The most insidious threat to southeastern fishes is sedimentation and siltation resulting from poor land-use patterns that eliminate suitable habitat required by many bottom-dwelling species. Cumulative effects of physical habitat modifications have caused widespread fragmentation of many fish populations in the Southeast (Fig. 3), presenting difficult challenges for those trying to reverse and restore diminished fish stocks.

Aquatic resources are often resilient and capable of recovery, given favorable conditions. Conservation of southeastern fishes will require significant changes in land management and socioeconomic factors (Moyle and Leidy 1992; Warren and Burr 1994), but such changes are necessary to stem future losses of biodiversity. The first step required is to improve public education on the value and status of native aquatic organisms. For resource managers and policy makers, increased efforts must be made to assume proactive management of entire watersheds and ecosystems; establish networks of aquatic preserves; restore degraded habitats; establish long-term research, inventory, and monitoring programs on fishes; and adopt improved environmental ethics concerning aquatic ecosystems (Warren and Burr 1994). The southeastern fish fauna is a national treasure of biodiversity that is imminently threatened. If this precious heritage is to be passed on, its stewardship must be improved through cooperative actions of all public and private sectors within the region.

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Loss of Genetic Diversity Among Managed Populations

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Species are composed of genetically divergent units usually interconnected by some (albeit low) level of gene flow (Soulé 1987). Because of this restriction in gene flow, natural selection can genetically tailor populations to their environments through the process of local adaptation (Wright 1931).

Because freshwater and anadromous (i.e., adults travel upriver from the sea to spawn) fishes are restricted by the boundaries of their aquatic habitats, genetic subdivisions may be more pronounced for these vertebrates than for others. Consequently, managers of programs for these species must realize that the stock (i.e., local discrete populations), and not the species as a whole, must be the units of primary management concern (Kutkuhn 1981).

Genetic variability in a species occurs both among individuals *within* populations as well as *among* populations (Wright 1978). Variation *within* populations is lost through genetic drift (*see* glossary; Allendorf et al. 1987), a process increased when population size becomes small. Variation *among* populations is lost when previously restricted gene flow between populations is increased for some reason (e.g., stocking, removal of natural barriers such as waterfalls); differentiation between populations is lost as a result of the homogenization of two previously distinct entities (Altukhov and Salmenkova 1987; Campton 1987).

Beyond this loss of genetic variation, mixing two groups can result in outbreeding depression, which is the loss of fitness in offspring that results from the mating of two individuals that are too distantly related (Templeton 1987). This loss in fitness is caused by the disruption of the process that produced advantageous local adaptations through natural selection. Inbreeding depression, on the other hand, is the loss of fitness produced by the repeated crossing of related organisms. The area of optimal relatedness occurs between inbreeding depression and outbreeding depression.

Loss of Genetic Integrity Through Stocking

Many sportfish populations are managed by using a combination of harvest regulation, habitat manipulation, and stocking. Jurisdiction for these activities falls to federal, state, tribal, and local governments, as well as private citizens. Many resource managers in the past were unaware of the long-term consequences that stocking efforts would have on the genetic integrity of local populations (Philipp et al. 1993).

Fish introductions can be classified into three types: non-native introductions, in which a given species of fish is introduced into a body of water outside its native range (regardless of any political boundaries); stock transfers, in which fish from one stock are introduced into a water body in a different geographic region inhabited by a different stock of that same species, yet are still within their native range; and genetically compatible introductions, in which fish are removed from a given water body and they, or more often their offspring, are introduced back into that water body or another water body that is still within the boundaries of the genetic stock serving as the hatchery brood source (Philipp et al. 1993).

Although non-native introductions may often cause ecological problems for the environments in which they are introduced, they can also cause genetic problems if they hybridize with closely related native species. Examples of this are the hybridization of introduced smallmouth bass (*Micropterus dolomieu*) and spotted bass (*M. punctulatus*) with native Guadalupe bass (*M. treculi*) in Texas (Morizot et al. 1991), and the hybridization of introduced rainbow trout (*Oncorhynchus mykiss*) with native Apache trout (*O. apache*; Carmichael et al. 1993). The greatest degree of genetic damage, that is, the loss of genetic variation among populations, is caused by stock transfers, a common

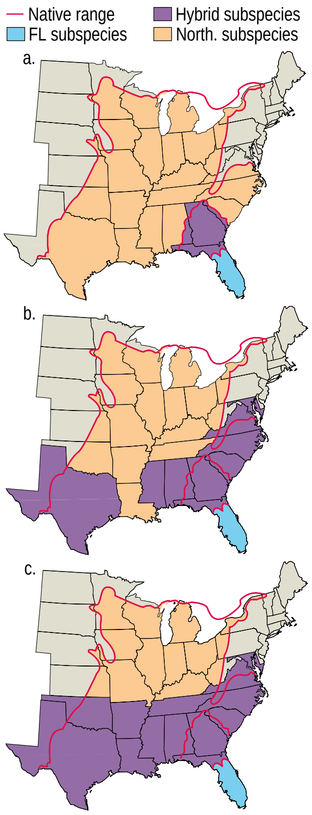

Figure. Loss of genetic variation among largemouth bass populations. **a.** The native range of the largemouth bass (*Micropterus salmoides*) is delineated by the red lines (MacCrimmon and Robbins 1975). As first described by Bailey and Hubbs (1949), the Florida subspecies, *M.s. floridanus*, was restricted to peninsular Florida (blue); the northern subspecies, *M.s. salmoides*, covered most of the rest of the range of the species; and there was a relatively small intergrade zone between the two resulting from some indeterminable combination of natural hybridization and human-caused mixing of stocks. **b.** The expansion of the intergrade by 1980 was described by Philipp et al. (1983). Because detailed ranges were not explored in all states, and because this intergrade zone expansion was likely caused by state stocking programs, entire states are classified according to whether the intergrade zone was expanded. **c.** The current intergrade zone is now even larger because of the addition of more states in which largemouth bass containing at least some *M.s. floridanus* genes are being introduced either by the state fish and game agencies themselves or by private groups. Notice that the entire southern and eastern portion of the original range of the northern subspecies, *M.s. salmoides*, is at risk of being inundated with *M.s. floridanus* genes.

practice among fisheries management agencies and the private sector.

Largemouth Bass

Largemouth bass (*Micropterus salmoides*) exemplify how introduction programs cause the loss of genetic diversity. The original range of the largemouth bass was restricted to parts of the central and southeastern United States (Figure), extending northward into some of southern Ontario (MacCrimmon and Robbins 1975). Bailey and Hubbs (1949), however, described two subspecies. The Florida subspecies, *M.s. floridanus*, was formerly restricted to much of peninsular Florida (Figure, a), whereas the range of the northern subspecies, *M.s. salmoides*, extended north and west of an intergrade zone that included parts of South Carolina, Georgia, Alabama, and northern Florida. It is likely, though, that the intergrade zone had already been expanded from the original natural hybrid zone as a result of early fish stocking programs.

Since 1949, however, much more serious stocking efforts have extended this intergrade zone. A survey of largemouth bass populations conducted in the late 1970's (Philipp et al. 1983) revealed that the intergrade zone had grown considerably larger through the deliberate stocking efforts of the involved state agencies (Figure, b). Additional introductions of *M.s. floridanus* since that genetic survey have now spread the genes of that subspecies across the entire southern range of *M.s. salmoides* (Figure, c).

This introduction of the Florida largemouth has compromised the genetic integrity of all the populations of the northern largemouth bass into which the species has been introduced (populations in Texas, Oklahoma, Arkansas, Louisiana, Mississippi, Tennessee, Alabama, Georgia, South Carolina, North Carolina, Virginia, and Maryland, at a minimum). Those now-genetically mixed populations have lost much of their distinctness because of the loss of among-population genetic variation that accompanies this type of homogenization. Populations other than those in the water bodies actually stocked will be affected as well because of inevitable gene flow into and between other connected populations. As a result, genetic integrity is now at risk for all populations of this important sportfish species throughout the southern and eastern portions of its native range.

In addition, because the two subspecies have quite different characteristics (Cichra et al. 1982; Fields et al. 1987; Kleinsasser et al. 1990), these massive stock transfers will likely result in outbreeding depression. More specifically, the Florida subspecies exhibits significantly poorer survival, growth, and reproductive success in Illinois than does the northern subspecies (Philipp 1991; Philipp and Whitt 1991). Also, the offspring resulting from crossing the two subspecies (in either direction) are less fit in Illinois than are the offspring of the pure northern subspecies (Philipp 1991). These results extend to populations of the northern subspecies across its range from Texas to Minnesota (unpublished data).

Conclusions

The genetic integrity of largemouth bass stocks, and likely of many other managed fish species as well, is eroding as a result of management programs that inadvertently permit or deliberately promote stock transfers. This causes not only the loss of genetic variation among populations, but through outbreeding depression it is also probably negatively affecting the fitness of many native stocks involved. We need to address genetic integrity when restoring native populations.

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Colorado River Basin Fishes

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The Colorado River and its tributaries have undergone drastic alterations from their natural states over the past 125 years. These alterations include both physical change or elimination of aquatic habitats and the introductions of numerous non-native species, particularly fish. Ironically, several more species occur at most localities today than were historically present before these alterations. This situation complicates the use of biodiversity as a litmus test for monitoring trends of either the deterioration or the health of an aquatic ecosystem.

An Altered Ecosystem

Over its entire basin (Figure), the Colorado River has been changed from its natural state perhaps as much as any river system in the world. The demands for water and power in the arid West have drastically altered the system by impoundments, irrigation diversions, diking, channelization, pollutants, and destruction of bank habitats by cattle grazing and other practices. Some reaches, ranging from desert spring runs to main rivers, have been completely dewatered or, seasonally, their flows consist almost entirely of irrigation return laden with silt and chemical pollutants. The Gila River of Arizona, one of the Colorado's largest tributaries, has not flowed over its lower 400 km (248 mi) since the early 1900's. These alterations and their effects on the fish fauna have been discussed by several authors (Miller 1961; Minckley and Deacon 1968; Stalnaker and Holden 1973; Carlson and Muth 1989; Minckley and Deacon 1991). Only a few small tributaries, mostly at higher elevations, retain most of their natural characteristics.

Courtesy J.N. Rinne

Captive bonytail (*Gila elegans*), rarest of the larger river species in the Colorado River Basin.

Native Fish Fauna

Despite the expansive drainage basin (631,960 km^2 [243,937 mi^2]) of the Colorado River, the system supported only a relatively small number of native fish species compared with basins of much smaller size east of the Continental Divide. The Colorado Basin's native fauna, however, was nearly unique. If two former marine invaders are removed from the 51 native taxa known from the system (Table 1), 42 of the 49 that remain (86%) are considered endemic to the system. The greatest diversity of taxa (44) was distributed in the Lower Basin downstream of the Arizona-Utah border, in a variety of habitats that include mainstem rivers, smaller tributaries, and isolated springs. The Upper Basin was much less diverse, containing 14 species, including a subset of the Lower Basin fauna plus 4 headwater species that occur in cooler water and a warm spring endemic. Basinwide, about 5 species occurred mostly in mainstem river or larger tributary habitats, 37 were restricted to smaller, in some cases isolated, habitats, and 7 were more generally distributed among different habitat types.

Trends

As a consequence of habitat alterations, the prevailing trend among native fish populations in the Colorado River Basin has been drastic

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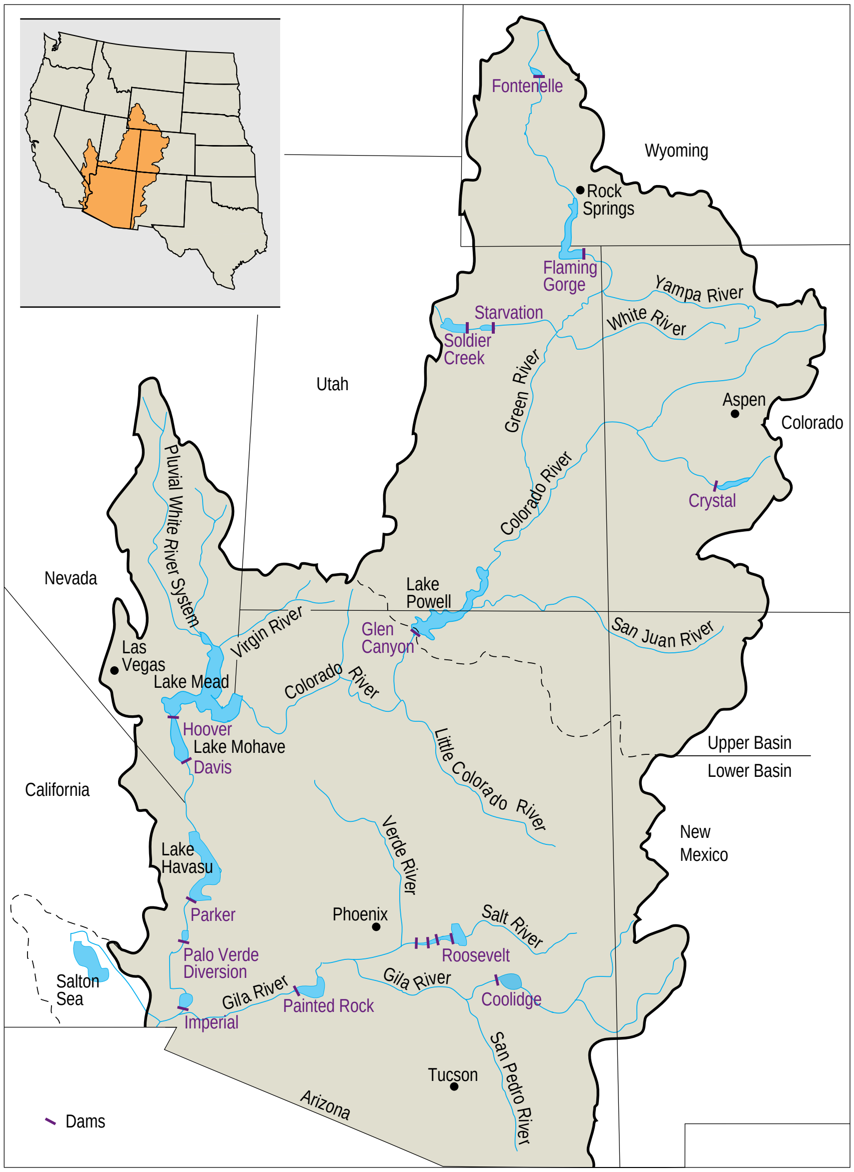

Figure. Colorado River Basin.

reductions that include decreased abundance in all or part of their ranges, overall range reductions, or virtual or actual extinctions (Tables 1 and 2). Presently, 40 of the 49 strictly freshwater, native species are considered either possibly or actually jeopardized or are extinct (Table 1). Of the 40, 12 are of special concern, 25 are considered endangered or threatened, and 3 are believed extinct.

In the Lower Basin, only 3 of the 10 native species that inhabited the mainstem of the lower Colorado River remained by the 1940's but by the 1960's, none remained. In the lower Salt River portion of the Gila River system, the original complement of 14 taxa was also reduced to 3 by the 1940's and to 2 by the 1960's; today, they are probably extirpated. In the early 1900's, the isolated springs of the Pluvial White River system in southern Nevada harbored 17 endemic taxa; today, 1 of those taxa is extinct, 9 endangered, 3 threatened, and the remainder of

Table 1. Native fish taxa of the Colorado River Basin including currently recognized subspecies. Taxa denoted by * may eventually prove genetically distinct from populations outside the Colorado River Basin. Those denoted "(m)" are marine invaders. Status of jeopardized and extinct species appears in parentheses: E = endangered; T = threatened; SC = special concern; X = extinct (based, in part, on Carlson and Muth 1989; Williams et al. 1989; and the National Biological Service's Category 2 list). Common names bracketed with quotation marks indicate that those species are undescribed and not officially named.

Scientific name	Common name
Family Elopidae	
Elops affinis (m)	Machete
Family Cyprinidae	
Agosia chrysogaster	Longfin dace
Gila cypha (E)	Humpback chub
G. elegans (E)	Bonytail
G. intermedia (SC)	Gila chub
G. robusta jordani (E)	Pahranagat chub
G. robusta robusta (SC)	Roundtail chub
G. seminuda (E)	Virgin chub
Lepidomeda albivallis (E)	White River spinedace
L. altivelis (X)	Pahranagat spinedace
L. mollispinis mollispinis (T)	Virgin spinedace
L.m. pratensis (E)	Big Spring spinedace
L. vittata (T)	Little Colorado spinedace
Meda fulgida (T)	Spikedace
Moapa coriacea (E)	Moapa dace
Plagopterus argentissimus (E)	Woundfin
Ptychocheilus lucius (E)	Colorado squawfish
Rhinichthys cobitis (T)	Loach minnow
R. deaconi (X)	Las Vegas dace
R. osculus osculus	Speckled dace
R. osculus ssp. (SC)	"Preston speckled dace"
R. osculus ssp. (SC)	"Meadow Valleys speckled dace"
R. osculus ssp. (SC)	"White River speckled dace"
R.o. thermalis (SC)	Kendall Warm Springs dace
R.o. velifer (SC)	Pahranagat speckled dace
Family Catostomidae	
Catostomus clarki clarki	Desert sucker
C.c. intermedius (E)	White River sucker
C. clarki ssp. (E)	"Meadow Valley sucker"
C. discobolus discobolus	Bluehead sucker
C.d. yarrowi (SC)	Zuni sucker
C. insignis	Sonora sucker
C. latipinnis (SC)	Flannelmouth sucker
C. platyrhynchus	Mountain sucker
C. sp.(SC)	"Little Colorado sucker"
Xyrauchen texanus (E)	Razorback sucker
Family Salmonidae	
Oncorhynchus apache (T)	Apache trout
O. clarki pleuriticus (SC)	Colorado cutthroat trout
O. gilae (T)	Gila trout
*Prosopium williamsoni**	Mountain whitefish
Family Goodeidae	
Crenichthys baileyi albivallis (E)	Preston springfish
C.b. baileyi (E)	White River springfish
C.b. grandis (E)	Hiko springfish
C.b. moapae (T)	Moapa springfish
C.b. thermophilus (T)	Moorman springfish
C. nevadae (T)	Railroad Valley springfish
Family Cyprinodontidae	
Cyprinodon macularius macularius (E)	Desert pupfish
C. sp. (X)	"Monkey Springs pupfish"
Family Poeciliidae	
Poeciliopsis occidentalis (SC)	Gila topminnow
Family Cottidae	
*Cottus bairdi**	Mottled sculpin
*C. beldingi**	Paiute sculpin
Family Mugilidae	
Mugil cephalus (m)	Striped mullet

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special concern. On the other hand, a few small tributaries, by virtue of their isolation, rare intermittent flows in lower reaches, and physical barriers, have been spared significant alterations or invasions by non-native species and retain an intact native fauna (e.g., Redfield Canyon, Arizona, Table 2).

In the larger rivers of the Upper Basin, such as the Green, lower Yampa, and most of the upper Colorado, most native taxa are extant but one or two (razorback sucker [*Xyrauchen texanus*], possibly bonytail [*Gila elegans*]), are represented by very rare individuals that may not be reproducing; all native fishes are greatly exceeded in numbers and kind by non-native taxa. In smaller tributaries of that region, varied numbers of native taxa persist; in the worst affected streams (e.g., most Green River tributaries in Utah), most taxa have been replaced by non-native taxa (author's observation).

Case studies of two endangered Colorado River species, which are hallmarks to conservationists, further elucidate patterns of decline among these fishes. They are large, long-lived (20-50 years) species that inhabit larger streams. The Colorado squawfish (*Ptychocheilus lucius*) is a highly migratory (Tyus 1990) predatory minnow. Perhaps because of fragmentation or impediment of migratory routes, its original extensive range has been reduced by roughly two-thirds, and it is uncommon where it remains. The last confirmed report in the Gila River was in 1950 and the last in the Lower Basin in 1975 (Miller 1961; Minckley 1973; Maddux et al. 1993).

The fourth species, the humpback chub *(Gila cypha*), is strictly a denizen of turbulent canyon reaches so difficult to sample that it was not discovered until 1946; it ranged from Boulder Canyon on the lower Colorado throughout canyon reaches of the Upper Basin well into Wyoming. Today, it occurs only in Grand Canyon, Arizona (Maddux et al. 1993), near the confluence of the Colorado and Little Colorado rivers, and in five Upper Basin canyon areas (rare in three), although the genetic "purity" of the Upper Basin populations is questioned. Recovery plans are in place for these fish as well as the bonytail and the razorback sucker. These fish are all easily propagated in captivity. It is otherwise difficult to find anything positive in the history of these or other Colorado Basin native fishes over the past several decades.

Non-native Species

Concomitant with the pervasive physical alteration of the Colorado River ecosystem has been both purposeful and accidental introductions of at least 72 non-native fish taxa (Maddux et al. 1993), including those indigenous to other North American basins and more exotic species. Alterations of the ecosystem's natural characteristics have apparently tipped the ecologic balance in favor of many of the non-native species that now vastly outnumber natives in numbers of species (Table 2), population density, and often biomass at most localities. There is evidence that some, such as the extremely pervasive red shiner (*Cyprinella lutrensis*), displace native taxa (Douglas et al. 1994) while others, such as channel and flathead catfish (*Ictalurus punctatus* and *Pylodictis olivaris*), are known predators on larval and juvenile native species (several references in Maddux et al. 1993). The introduced white sucker (*Catostomus commersoni*) is hybridizing extensively with native suckers throughout much of the Upper Basin (author's observation), possibly threatening the genetic integrity of those taxa. These and other interactions between non-native and native taxa may have significant negative effects on native fishes. The dominance held by non-native fishes may be symptomatic of the overall degree of alteration of the Colorado River ecosystem and could potentially confound future studies of biodiversity.

Table 2. Overall and relative abundance of native and non-native fishes from various localities in the Colorado River Basin. Numbers for 1800's represent original complements of native taxa. For subsequent years, total abundance is followed by ratio of non-native to native taxa in parentheses. Sources: Miller 1961; Taba et al. 1965; Vanicek et al. 1970; Stalnaker and Holden 1973; Cross 1975; Holden and Stalnaker 1975a,b; Suttkus et al. 1976; Carlson et al. 1979; Miller et al. 1982; Valdez et al. 1982; Valdez 1984,1990; Wick et al. 1985; Platania and Bestgen 1988; Griffith and Tiersch 1989.

Locality	Survey date				
	1800's	1940's	ca. 1965	ca. 1975	ca.1985
Yampa-Green River area, CO-UT	10	-	21(12/9)	22(13/9)	24(15/9)
White River, CO-UT	9	-	-	13(7/6)	12(5/7)
Dolores River, CO	9	-	-	11(7/4)	16(12/4)
Colorado River, Lake Powell,UT to Gunnison River, CO	10	-	15(9/6)	29(19/10)	31(23/8)
San Juan River, NM	9	-	-	-	18(12/6)
Colorado River, Grand Canyon, AZ	10	-	-	19(15/4)	-
Virgin River, AZ-UT	6	-	-	19(13/6)	-
Lower Colorado River, AZ-CA	10	12(9/3)	11(11/0)	-	-
Salt River near Phoenix, AZ	14	9(6/3)	22(20/2)	-	-
Redfield Canyon, San Pedro River system, AZ	5	-	-	-	5(0/5)

Altered Species Diversity and Biodiversity Studies

While native taxa have declined, there have actually been two- to threefold increases in the number of species at most localities in the Colorado Basin because of the success of introduced taxa (Table 2). If future biodiversity monitoring is to truly gauge positive and negative shifts in the health of the Colorado River ecosystems, then an accurate baseline is necessary. A baseline describing unaltered native fauna might be an ideal but unattainable goal. That line could be approached, however, by divesting faunal lists of all non-native taxa and determining, as much as possible, the true extent of diversity of that which remains. In fish, it is practical to do so to the level of distinctive populations through studies of genetic variability. With luck, it is even possible to include extirpated populations through DNA studies of museum specimens if historic material is available.

Once a baseline is determined, researchers and managers can know where to try to "hold the line" in maintaining diversity through management and protection. Of course, on a systemwide basis, the baseline diversity of a pristine system can never be reattained because genetically unique populations have already been lost. On a more local basis, however, positive increments and recovery of the habitat are indicated if monitoring reveals increased diversity resulting from the successful reestablishment of taxa which were conserved in other, less altered, portions of the system.

For monitoring purposes, when non-native species are added to biodiversity determinations, we must carefully tease out the cause of shifts toward or from the "desired baseline" which, in the case of the Colorado River, is probably a value far less than the present overall number of species. Thus, "desirable" outcomes may be indicated by overall decreases in diversity caused by the disappearance of non-native taxa as an indicator of habitat "healing," but not so by the loss of native taxa. Conversely, actual increases may yet be positive if caused by reestablishment of native taxa, but may be an indicator of further degradation if caused by success of additional non-natives. Realistically, monitoring will have to include, in addition to determinations of diversity, attention to shifts in dominance among native and non-native species, which can be indicative of both positive and negative trends.

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Cutthroat Trout in Glacier National Park, Montana

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The indigenous fishery of Glacier National Park has been radically altered from its pristine condition during the past half-century through introductions of non-native fishes and the entry of non-native species from waters outside the park. These introductions have adversely affected the native westslope cutthroat trout (*Oncorhynchus clarki lewisi*; Fig. 1) throughout much of its park range.

The effects of non-native fishes on indigenous fisheries have been reviewed by Taylor et al. (1984), Marnell (1986), and Moyle et al. (1986). Effects of fish introductions in Glacier National Park include establishment of non-native trout populations in historically fishless waters, genetic contamination (i.e., hybridization) of some native westslope cutthroat trout stocks, and ecological interferences with various life-history stages of native trout.

Research conducted in the park during the 1980's addressed the genetic effects of fish introductions on native trout. Of 47 lakes known or suspected to contain cutthroat trout or trout hybrids, 32 lakes contained viable populations of cutthroat trout, rainbow trout (*O. mykiss*), or hybrids. Trout introduced in the other waters were evidently unable to sustain themselves through natural reproduction.

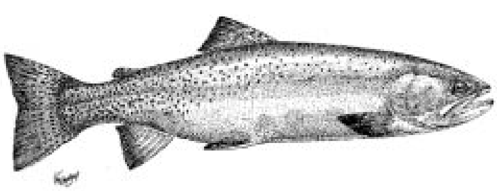

Fig. 1. Westslope cutthroat trout (*Oncorhynchus clarki lewisi*).

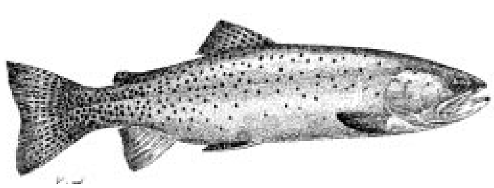

Fig. 2. Yellowstone cutthroat trout (*Oncorhynchus clarki bouvieri*).

About 30 trout sampled from each lake underwent laboratory genetic analyses. Close agreement of the results from two analytical procedures yielded a high degree of confidence in the conclusions (Marnell et al. 1987). Genetic classifications in Tables 1 and 2 reflect the combined results of the analyses.

Fourteen pure strain populations of westslope cutthroat trout persist in 15 lakes (i.e., some interconnected lakes contain a single trout population) in the North and Middle Fork drainages of the Flathead River; the species was historically present in these waters (labeled as "stable" populations in Table 1).

Pure strain native trout also inhabit four other Middle Fork lakes (i.e., Avalanche, Snyder, and Upper and Lower Howe lakes), but it is unclear whether they are indigenous or were transplanted from other park waters. Recent findings from sediment paleolimnology studies suggest that trout have been present in at least one of these lakes for more than 300 years (D. Verschuren, University of Minnesota, and author, unpublished data). Hence, trout populations in these four lakes are tentatively classified as indigenous (Table 1).

Introduced populations of Yellowstone cutthroat trout (*O. clarki bouvieri*; Fig. 2) and trout hybrids including cutthroat-rainbow trout (*O. clarki* spp. x *O. mykiss*) occur in 13 lakes distributed among the three continental drainages

Table 1. Status and trends of cutthroat trout and their hybrids in the North and Middle Fork, Flathead River drainages of Glacier National Park, Montana.

Lake	Area (ha)	Trout classification*	Population status**
North Fork, Flathead R.			
Akokala	9	WCT	Stable
Arrow	23	WCT x YCT	Hybrid
Bowman	691	WCT	Unstable
Camas	8	YCT	Non-native
Cerulean	20	WCT	Stable
Evangeline	28	YCT	Non-native
Grace	32	WCT x YCT	Hybrid
Kintla	688	WCT	Unstable
Logging	444	WCT	Unstable
Quartz	349	WCT	Stable
Lower Quartz	67	WCT	Stable
Middle Quartz	19	WCT	Stable
Trout	86	WCT	Stable
Middle Fork, Flathead R.			
Avalanche	23	WCT	Stable
Fish	3	WCT x YCT	Hybrid
Harrison	101	WCT	Unknown
Hidden	110	YCT	Non-native
Lincoln	14	WCT	Stable
Lower Howe	12	WCT	Stable
Lower Isabel	17	WCT	Stable
McDonald	2,760	WCT	Unstable
Ole	2	WCT	Stable
Snyder	2	WCT	Stable
Upper Howe	3	WCT	Stable
Upper Isabel	6	WCT	Stable

* WCT — pure strain westslope cutthroat trout.
YCT — the introduced Yellowstone cutthroat trout.
x — two or more species have hybridized.
**Stable — native population exists in a pristine environment.
Unstable — declining condition resulting from presence of competing non-native species.
Hybrid and non-native populations — classified without regard to population condition.

Table 2. Status and trends of non-native and hybrid trout populations in the South Saskatchewan and Missouri river drainages of Glacier National Park, Montana.

Lake	Area (ha)	Trout classification*	Population status**
South Saskatchewan River			
Lower Slide	15	YCT x RBT	Hybrid
Otokomi	9	YCT x RBT	Hybrid
Red Eagle	55	YCT x WCT x RBT	Hybrid
Upper Slide	5	YCT x RBT	Hybrid
Upper Missouri River Drainage			
Katoya	4	YCT	Non-native
Morning Star	4	YCT	Non-native
Old Man	17	YCT	Non-native

*YCT— introduced Yellowstone cutthroat trout.
x — two or more species have hybridized.
RBT— rainbow trout.
WCT — westslope cutthroat trout.
**Hybrid and non-native populations are classified without regard to population condition.

that form their headwaters in Glacier National Park (Tables 1 and 2). Native cutthroat trout were not found east of the Continental Divide in the Missouri River or South Saskatchewan River drainages within the park.

In addition to genetic concerns, ecological disturbances associated with the presence of introduced fishes have compromised the native westslope cutthroat fishery. Fish are no longer stocked in park waters; however, several waters, including some that contain undisturbed native fisheries, remain vulnerable to invasion by non-native migratory species. Introduced kokanee salmon (*O. nerka*), a specialized planktivore, are believed to be competing with juvenile stages of native trout in some waters, especially during periods of winter ice cover when plankton may be limited. Predation by introduced lake trout (*Salvelinus namaycush*) has also been implicated in the decline of native cutthroat trout in several large glacial lakes in the North and Middle Fork drainages (Marnell 1988). Native cutthroat trout have been compromised by fish introductions and invasions throughout about 84% of their historic range in Glacier National Park (Marnell 1988).

Although native cutthroat trout have been adversely affected throughout a large portion of their park range, the species has not been lost from any water where it was historically present. Glacier National Park remains one of the last strongholds of genetically pure strains of lacustrine (i.e., lake-adapted) westslope cutthroat trout. This fact could have important implications for reestablishment of this unique subspecies throughout the central Rocky Mountains, where this trout has disappeared from most of its original range.

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Columbia River Basin White Sturgeon

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White sturgeon (*Acipenser transmontanus*), the largest freshwater fish in North America, live along the west coast from the Aleutian Islands to central California (Scott and Crossman 1973). Genetically similar reproducing populations inhabit three major river basins: Sacramento-San Joaquin, Columbia, and Fraser. The greatest number of white sturgeon are in the Columbia River Basin.

Historically, white sturgeon inhabited the Columbia River from the mouth upstream into Canada, the Snake River upstream to Shoshone Falls, and the Kootenai River upstream to Kootenai Falls (Scott and Crossman 1973; Figure). White sturgeon also used the extreme lower reaches of other tributaries, but not extensively. Current populations in the Columbia River Basin can be divided into three groups: fish below the lowest dam, with access to the ocean (the lower Columbia River); fish isolated (functionally but not genetically) between dams; and fish in several large tributaries.

The Columbia River has supported important commercial, treaty, and recreational white sturgeon fisheries. A commercial fishery that began in the 1880's peaked in 1892 when 2.5 million kg (5.5 million lb) were harvested (Craig and Hacker 1940). By 1899 the population had been severely depleted, and annual harvest was very low until the early 1940's, but the population recovered enough by the late 1940's that the commercial fishery expanded. A 1.8-m (6-ft) maximum size restriction was enacted to prevent another population collapse. Total harvest doubled in the 1970's and again in the 1980's because of increased treaty and recreational fisheries. From 1983 to 1994, 15 substantial regulatory changes were implemented on the mainstem Columbia River downstream from McNary Dam as a result of increased fishing. Columbia River white sturgeon are still economically important. Recreational, commercial, and treaty fisheries in the Columbia River downstream from McNary Dam were valued at \$10.1 million in 1992 (Tracy 1993).

Several factors make white sturgeon relatively vulnerable to overexploitation and changes in their environment. The fish may live more than 100 years (Rieman and Beamesderfer 1990), and overexpolitation is well documented for long-lived, slow-growing fish (Ricker 1963). Female white sturgeon are slow to reach sexual maturity; in the Snake River they mature at age 15-32 (Cochnauer 1981). Mature females in the Columbia Basin only spawn every 2-11 years (Stockley 1981; Cochnauer 1983; Welch and Beamesderfer 1993). Sustainable harvest levels vary for impoundments in the Columbia River. Several impoundments are managed as groups, making overexploitation more likely in impoundments with low sustainable harvest levels.

White sturgeon populations in free-flowing and inundated reaches of the Columbia River Basin have been negatively affected by the abundant hydropower dams in most of the mainstem Columbia and Snake rivers (Rieman and Beamesderfer 1990). These dams have altered the magnitude and timing of discharge, water depths, velocities, temperatures, turbidities, and substrates, and have restricted sturgeon movement within the basin. Sturgeons in other river basins have declined in response to dam-induced habitat alterations (Artyukhin et al. 1978).

Mainstem Columbia River

Abundance and growth of white sturgeon are greatest in the lower Columbia River (Figure). These fish use estuarine and marine habitats as well as riverine habitats, allowing them to feed on anadromous prey fishes (those fishes traveling upriver from the sea to spawn; Tracy 1993). Although the lower Columbia River population may be the only one in this basin that is abundant and stable, even it is at some risk of collapse (Rieman and Beamesderfer 1990). Of the 11 populations isolated between dams upstream, white sturgeon are known to be relatively abundant in only 3 (Figure). White sturgeon densities in three of the remaining eight populations are much lower than in the abundant populations. Data are sparse for the remaining five populations, although Zinicola and Hoines (1988) reported that in 1988 fewer than 10 white sturgeon were harvested in each of four of these impoundments and only 34 in another.

Although the lower Columbia River population probably declined during the 1980's, adoption of more restrictive harvest regulations appears to have stabilized the population (Tracy 1993). Successful spawning occurs each year in this reach (McCabe and Tracy 1993). Catch-per-unit-effort of most size groups in the three populations for which data are available declined considerably from 1987 to 1991; fisheries there have collapsed and the populations are at risk of collapse (Beamesderfer and Rien 1993). Recruitment in some populations appears limited to years with high river discharges in spring (Miller and Beckman 1993). Although most of the mainstem populations appear unstable, their genetic similarity to the stable lower Columbia River population has excluded them from consideration for listing under the federal Endangered Species Act.

Overexploitation and poaching have reduced population size (Beamesderfer and Rien 1993), and impoundments and altered hydrographs caused by development of the hydropower system have altered critical spawning habitat (Parsley et al. 1993). Because the factors identified as causing declines in other white sturgeon populations are present to varying degrees in each of the other eight upstream impoundments, these populations are likely declining as well.

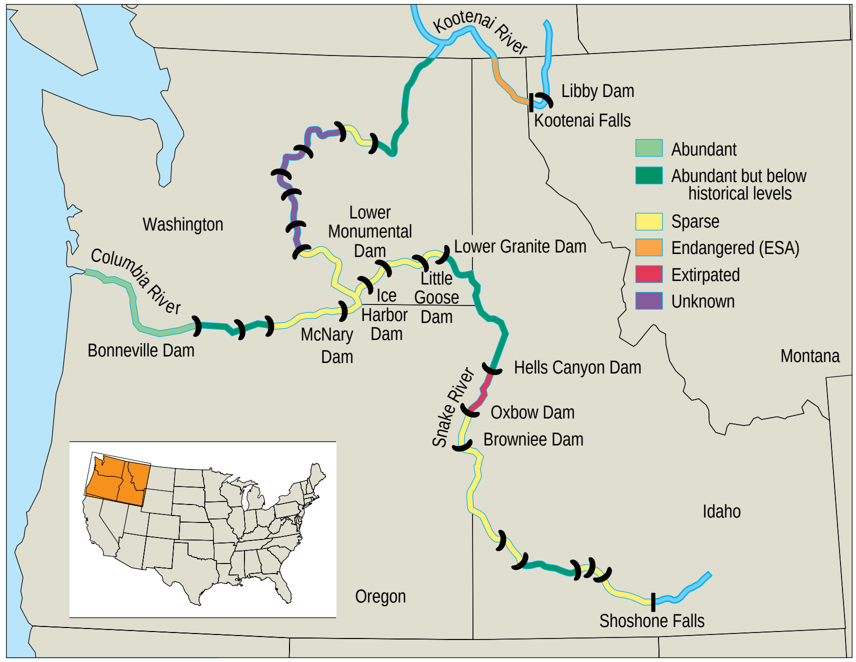


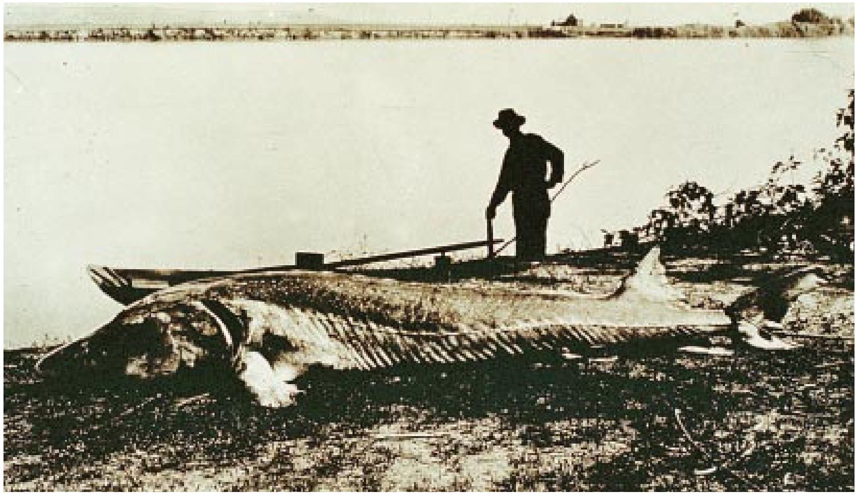
Figure. Distribution and status of white sturgeon in the U.S. portion of the Columbia River Basin.

Kootenai River

Current research on white sturgeon in the Kootenai River indicates that this population is unstable and declining. The U.S. Fish and Wildlife Service listed the Kootenai River population as endangered in 1994.

This population has declined to fewer than 1,000 fish, about 80% of which are more than 20 years old. Apperson and Anders (1990) concluded that virtually no recruitment has occurred since 1974, soon after Libby Dam began regulating flows, thereby altering historical discharge patterns of the river. This altering of discharge patterns is thought to be a major causal factor limiting recruitment into this unique sturgeon population. Research on the Kootenai River is examining the effects of increased discharge on the spawning behavior of white sturgeon. During 1993 increased discharges resulted in the collection of only three white sturgeon eggs despite intensive efforts to collect early lifestages of white sturgeon (Marcuson 1994).

Fishing for white sturgeon in the Kootenai River has been regulated in Idaho since 1944, in Montana since 1957, and in British Columbia since 1952, indicating that overharvesting may have been affecting population size. Fishing for white sturgeon has been closed in Montana since 1979, and catch and release angling



Courtesy Oregon Historical Society

Fifteen-hundred-pound white sturgeon caught near Payette, Idaho, circa 1911.

restrictions have been in place since 1984 in Idaho and 1990 in British Columbia.

Snake River

The Snake River has 12 dams from its mouth upstream to Shoshone Falls in Idaho. White sturgeon are believed to exist in small numbers in the lower three pools on the Snake River formed by Ice Harbor, Lower Monumental, and Little Goose dams (Zinicola and Hoines 1988). Of the nine impoundments upstream from Little Goose Dam, white sturgeon are relatively abundant in two, present at low numbers in six, and are absent in another (PSMFC 1992).

Although little is known about the early life history and spawning habitat requirements of white sturgeon in the Snake River, the construction and operation of the river's dams are likely to have the same effects as the impoundments on the Columbia and Kootenai rivers. White sturgeon appear more abundant in regions of the Snake River where free-flowing river habitat exists (PSMFC 1992), such as between Lower Granite and Hells Canyon dams where 76% of the river is free-flowing. Conversely, white sturgeon are not present in the impoundments created by Hells Canyon Dam and not abundant in the impoundment created by Oxbow Dam, which constitute two continuous slackwater regions (Welsh and Reid 1971).

While free-flowing sections of the Snake River exist in varying proportions between the dams, impoundments upstream of these sections influence both water temperature and the annual discharge pattern. At least 28 sturgeon died during July 1990 because of low dissolved-oxygen levels in Brownlee Pool (PSMFC 1992). Sturgeon production in the Snake River also appears limited by dewatering from irrigation diversions (Lukens 1981) and small spawning populations (Cochnauer et al. 1985).

Harvest of white sturgeon from the Snake River has had a definite negative impact on these populations, but the magnitude of the effect is unknown. Commercial fishing was permitted on the Snake River until 1943; then increasingly restrictive regulations were implemented from 1944 to 1969. In 1970 catch and release regulations were imposed on the entire river. A recommendation has been made that 3 of the 12 reaches of the Snake River discussed in this article be completely closed to fishing (Cochnauer et al. 1985).

Summary

Habitat changes (e.g., decreased discharges resulting in decreased spawning habitat) caused by development of the hydropower system have contributed to white sturgeon population declines in the Columbia River Basin; spawning habitat has been particularly affected by dams.

Overharvest of white sturgeon has caused population declines in several Columbia River Basin populations, both historically and in the past two decades. Recent management changes have helped alleviate overharvest in much of the Columbia River Basin, but refinement of management strategies is still needed in some areas.

The status of the 25 Columbia River Basin white sturgeon populations varies considerably: 1 is stable and abundant; 5 are relatively abundant, but probably at lower levels than in the past; 12 are sparse and many are declining; 5 have unknown status but creel data suggest they are sparse; 1 is sparse, declining, and listed under the Endangered Species Act; and white sturgeon have probably been extirpated from another. Conditions that have contributed to stock declines in other white sturgeon populations are present in populations whose status is unknown, suggesting that populations with unknown status may also be declining.

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