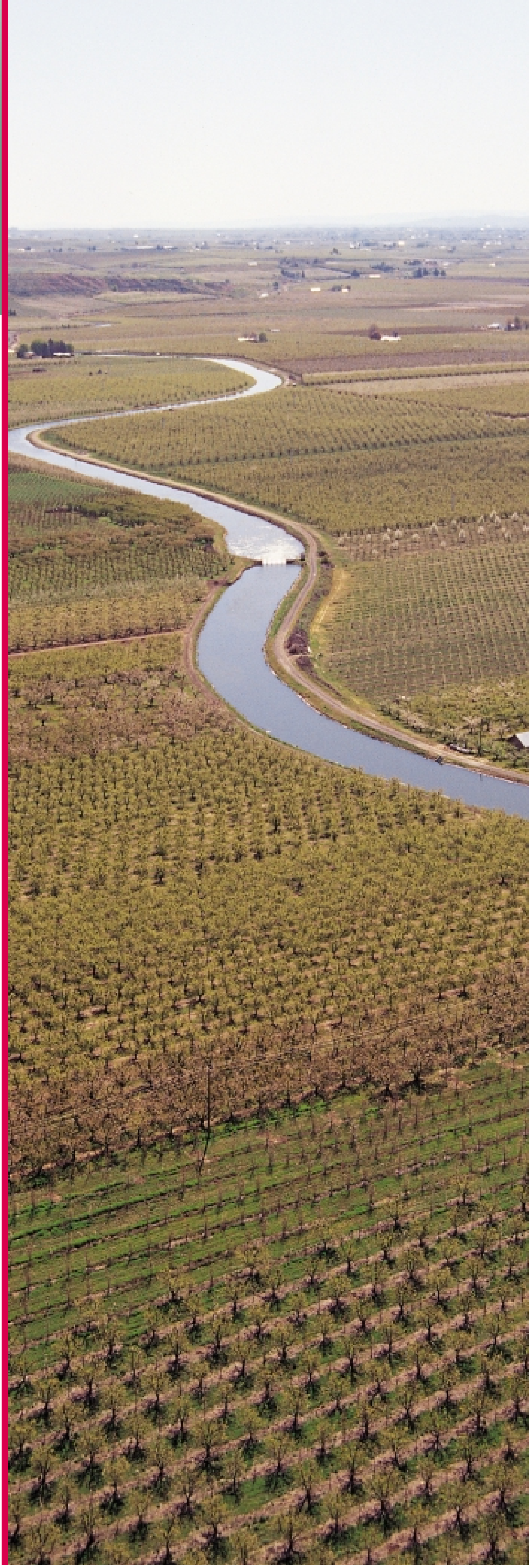
Water Use

Fresh water is vital to life and to habitat preservation. It is a major component of climate and a determining influence on economic growth and human population patterns. Modern competition for freshwater resources directly relates to the complex history of land use as humans evolved from hunter-gatherers to farmers and to modern agriculturists, industrialists, and members of modern urban communities. Water developments that proliferated during the twentieth century have greatly benefited most aspects of modern life, including public health and safety, agriculture, recreation, and commerce, but at the same time, these water developments have caused many environmental changes (Schilling et al. 1987; McDonald and Kay 1988; Waterstone and Burt 1988; Johnson and Viessman 1989; Moore 1989). The need to balance water use and development with environmental change will continue to increase as the human population grows and associated changes in land use accelerate.

Trends in Freshwater Use

From 1950 to 1990, both the population and domestic water use in the United States increased steadily. Withdrawals of fresh and salt waters increased to a peak of 1.7 billion cubic meters per day in 1980, and by 1990 daily freshwater withdrawals were 1.5 billion cubic meters (Fig. 1). Rural use of water for households and livestock increased from 1960 to 1990 (Table 1). Irrigation increased from 1950 to 1980, to a maximum of 570 million cubic meters per day, while per capita water use in the United States decreased from 6.8 million cubic meters per day in 1970 to 5.9 million cubic meters per day in 1990. Commercial and industrial uses of water, including self-supplied industrial use and withdrawals of water for mining, increased to a plateau in 1975–1980 before declining by 14%. The estimated use of fresh groundwater—fresh water drawn from below the ground—was 130 million cubic meters per day in 1950. Use of groundwater increased to 310 million cubic meters per day by 1975, decreased during the 1980's to 280 million cubic meters per day, and then increased again to 300 million cubic meters per day in 1990 (Table 1). The use of fresh surface water peaked in 1980 at 1.1 billion cubic meters per day and declined to 980 million cubic meters per day by 1990 (Table 1). Consumptive use—water that is withdrawn from a water source and does not eventually return to the water source—of fresh water followed the same patterns as withdrawals (Table 1). The reduction of withdrawals during 1980–1985 reflected conservation but could also relate to climate or the economic slowdown (van der Leeden 1975; Solley and Pierce 1988; Solley et al. 1993).

Fresh water is now a limited ecological (physical and biological) and economical resource. The trend in the present use of water reflects its limited availability. Krusé (1969) estimated that by 1965, withdrawals of 1.3 billion cubic meters per day were exceeding the available dependable water supply by 13%. The deficit reflected the need for reusing water, the increased use of salt water, and the lack of new water development opportunities. How did we reach this point?



Courtesy Agricultural Service, USDA

Table 1. Water use in the United States, 1950–1990[a] (in million cubic meters per day; modified from Solley et al. 1993).

	1950	1955	1960	1965	1970	1975	1980	1985	1990
Offstream use									
Total withdrawals[b]	680	910	1,000	1,200	1,400	1,600	1,700	1,500	1,500
Public supply	53	64	79	91	100	110	130	140	150
Rural domestic livestock	14	14	14	15	17	19	21	29	30
Irrigation	340	420	420	450	490	530	570	520	520
Industrial									
Thermoelectric power	150	270	380	490	640	760	790	710	740
Other	140	150	140	170	180	170	170	120	110
Source of water									
Groundwater									
Fresh	130	180	190	230	260	310	310	280	300
Saline		2.3	1.5	1.9	3.8	3.8	3.4	2.5	4.6
Surface water									
Fresh	530	680	720	790	950	980	1,100	1,000	980
Saline	38	68	120	160	200	260	270	230	260
Reclaimed water		0.8	2.3	2.6	1.9	1.9	1.9	2.2	2.8
Consumptive use			230	290	330	360	380	350	360
Instream use									
Hydroelectric power	4,200	5,700	7,600	8,700	10,600	12,500	12,500	11,500	12,500

[a]For years before 1960, data include the contiguous United States; 1960–1965, the 50 states; 1970, the 50 states and Puerto Rico; and after 1970, the 50 states, Puerto Rico, and the Virgin Islands. For 1970 and after, consumptive use values are for fresh water only.

[b]The numbers in this column are not column totals because of rounding.

History of Water Use in the United States

The beginning of water development in North America can be traced to between A.D. 600 and A.D. 800, when the Hohokam Indians of southern Arizona dug irrigation canals for cornfields (Josephy 1968; Hurt 1987). Modern water development for irrigation began in Utah in 1847 (Lea 1985), and the first dam for municipal water was completed in 1916 (van der Leeden et al. 1990). Today, an estimated 75,000 dams and an untold number of canals, levees, locks, power plants, and pipelines exist (Parfit 1993). The U.S. Army Corps of Engineers reported that in 1982, each of 2,654 large dams stored more than 6 million cubic meters of water, 50,000 smaller dams stored 60,000 to 6 million cubic meters, and more than 2 million small dams and farm ponds stored an undisclosed amount of water (van der Leeden et al. 1990). By 1988, 91% of the river lengths in the lower United States had been developed (Hunt 1988). Water storage in reservoirs increased to 549 billion cubic meters in 1990 (Fig. 2).

The national emphasis of water development in the United States has shifted back and forth among flood control, water supply and quality, and conservation (Table 2). Likewise, the responsibility for water management in the United States has also shifted greatly over time (Josephy 1968; Hurt 1987; Hunt 1988; Johnson and Viessman 1989; van der Leeden et al. 1990; Tyler 1992; Wilkinson 1992). Until 1850 the responsibility for water use remained primarily with individuals or corporate entrepreneurs. During the next century, however, the management of the waters of the United States in public trust was the joint responsibility of the states and the federal government. Federal interest in interstate water-use conflicts on major rivers resulted in agreements such as the Pick-Sloan Plan for the Missouri River (Schmulbach et al. 1992) and the Colorado River Compact (Trelease 1967). The states have determined how water uses are administered and who has the right to use water primarily through two different approaches. The first approach is by the riparian common law doctrine in the humid eastern United States, where streamside owners are entitled to the natural flow of the streams

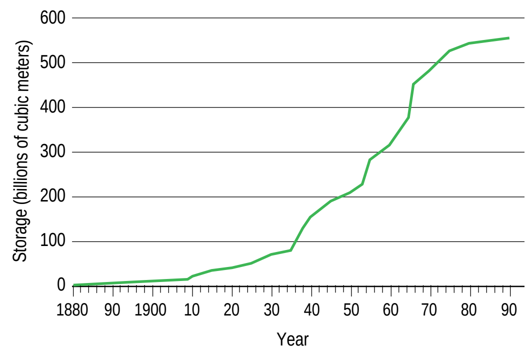


Fig. 2. Reservoir storage in the United States (© W. B. Solley, unpublished table of reservoir data, 1994).

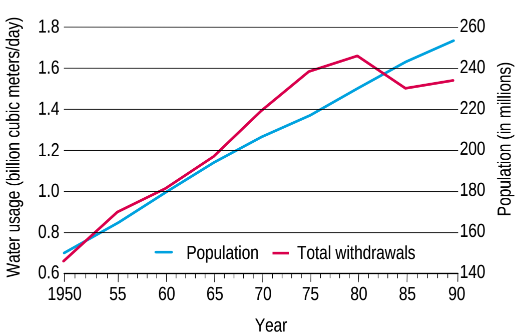


Fig. 1. Population and water use in the United States, 1950–1990 (Solley et al. 1993).

past their land, substantially undiminished in quantity and unimpaired in quality (that is, a reasonable use of water consistent with like use by others). The second approach is by the prior appropriation doctrine in the arid western United States, where first in time is first in right, and beneficial use of water (not landownership) and priority of use (not equality of right) are the basis for division when there is not enough water for everyone (Trelease 1967). These legal approaches reflect the public interest in the availability of water for beneficial uses. Major federal projects that support national purposes and respond to national emergencies and regional needs also had a significant influence on the nature of water development during the nineteenth century (Table 2).

Effects of Water Use on Watersheds

The withdrawal of water or the alteration of water quality elicits responses in watersheds—the area drained by a stream or river. These alterations occur even in the most remote places, and responses include changes in biological diversity and ultimately in the entire landscape (Ward and Stanford 1979; Becker and Neitzel 1992; Pederson 1994). In fact, few wild rivers are completely wild, and few native populations are not affected by humans. Benke (1990) estimated that during the past century, 98% of the 5.2 million kilometers of streams in the contiguous 48 states were altered sufficiently by human activities so that they did not meet the more stringent requirements for protection under the Federal Wild and Scenic River provisions (Fig. 3). For example, as human population and water use increased, the species diversity of fish communities decreased (Moyle and Leidy 1992). Thus, by 1989, in spite of conservation and restoration, over 100 species of freshwater fishes were added to the threatened or endangered list and more than 250 freshwater fish species were in danger of disappearing (Deacon et al. 1979; Williams et al. 1989; Johnson 1995). The endangerment of freshwater fishes in several regions of the United States has been linked to dams, the straightening of channels of large rivers, the building of cities, the expansion of agriculture, the logging and clearing of headwaters, the erosion of river channels, the pollution of water, and the introduction of nonindigenous species. The total effect of these developments is the alteration of stream ecology as evidenced by changes in the migration patterns of fishes, in stream water temperature and nutrient levels, in water chemistry, and in biological diversity (Warren and Burr 1994).

Table 2. Events that characterize water use in the United States.

Date	Event
A.D. 600–800	Diversion of water for irrigation by the Hohokam Indians
1824	General Survey Act: the U.S. Army Corps of Engineers given responsibility for navigation and flood control of the Mississippi River and other rivers
1847	First modern U.S. irrigation project, Salt Lake City, Utah
1848, 1850	Swamp Lands Acts: first federal activity in water resources management
1870	Irrigation in Greeley, Colorado
1899	Rivers and Harbors Act as amended through 1977
1902	Reclamation Act: established the Reclamation Service—responsible for irrigation of arid land
1916	Ashokan Dam on Esopus Creek near Olive Bridge, New York, completed. First high dam in United States (over 75 meters high and 1.9 million cubic meters in volume content)
1917, 1928, 1936, 1938	Flood Control Acts: emphasized flood control, not water resources management
1920	Federal Water Power Act: created Federal Power Commission to regulate water resources
1922	Colorado River Compact: divided the states' rights to use the Colorado River
1933	Bureau of Reclamation joined Public Works Administration
1933	Tennessee Valley Authority created, ultimately changing the lower Ohio and Tennessee River valleys
1935	Soil Conservation Service created
1936	Flood Control Act: had the first benefit–cost analysis for water development
1936	Hoover Dam–Lake Mead started operation: supplied power to the urban West
1938	Soil Conservation Service given responsibility for flood and soil erosion control
1938-1956	Colorado–Big Thompson Project: largest transfer across the Continental Divide
1948	Upper Colorado River Compact
1956	Federal Water Pollution Control Act (as amended P.L. 92-500, 1972)
1965	Water Resources Planning Act: mandated river basin planning
1968	Wild and Scenic Rivers Act: effort to protect the remaining undeveloped U.S. rivers
1969	National Environmental Policy Act
1970	Rivers and Harbors Act
1973	Endangered Species Act
1977	Safe Drinking Water Act
1977	Clean Water Act, with the Safe Drinking Water Act, provided safe, clean water to the general public

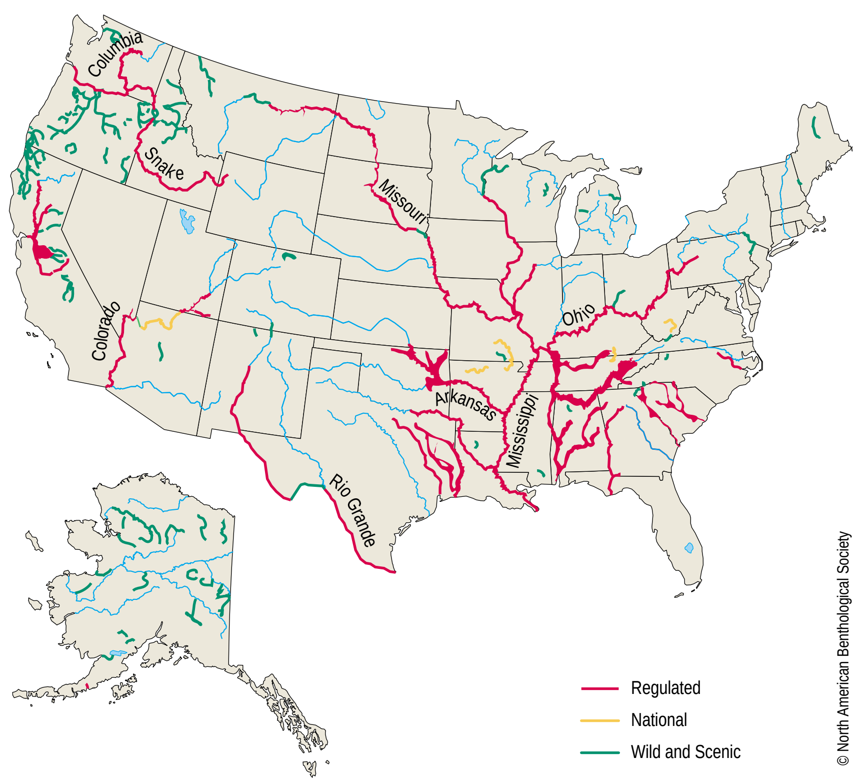


Fig. 3. Regulated, National, and Wild and Scenic Rivers in the United States (Benke 1990).

The terrestrial part of the watershed ecosystem is also threatened (Table 3). Before European settlement, the estimated amount of riparian land in the 100-year floodplains of the lower 48 states was 49 million hectares. By the 1980's it was reduced by 81%, to 9.3 million

Table 3. Endangered and threatened species in the contiguous United States (U.S. Fish and Wildlife Service 1994).

Species	Total species endangered	Aquatic and riparian species endangered	Total species threatened	Aquatic and riparian species threatened
Mammals[a]	53	18	10	4
Birds[b]	30	15	12	7
Reptiles[c]	8	3	18	6
Amphibians	6	6	3	3
Fishes	64	64	36	36
Clams	50	50	6	6
Snails	13	12	7	4
Insects	19	9	9	5
Arachnids	4	4	0	0
Crustaceans	11	11	2	2
Flowering plants[d]	187	55	74	29
Conifers and cycads	2	1	0	0
Ferns and allies	4	3	2	1
Lichens	1	0	0	0
Total	452	251 (56%)	179	103 (58%)

[a]Mammals that occur only in saline or marine environments; 12 endangered and 2 threatened species.
[b]Birds that occur only in saline or marine environments; 6 endangered and 2 threatened species.
[c]Reptiles that occur only in saline or marine environments; 4 endangered and 4 threatened species.
[d]Flowering plants that occur only in saline or marine environments; 2 endangered species.

hectares (Brinson et al. 1981); 22 states had lost more than 50% of their wetlands (Fig. 4). Although the rate of change in wetland areas slowed between the mid-1970's and mid-1980's, there was still a net loss (Table 4), which created a major shift and reduction in the variety of plants and animals in riparian lands (Johnson and McCormick 1979; Petts 1984; U.S. Office of Technology Assessment 1984; Mathias and Moyle 1992).

The total effects of human activities in aquatic and riparian lands are not nearly understood. The change in biological diversity, however, can be linked to habitat change and to the loss of species (Hunt 1988). From alpine and mountain streams to estuaries and deltas, anthropogenic changes have accumulated, and many of the nation's watershed ecosystems have been drastically altered by these changes.

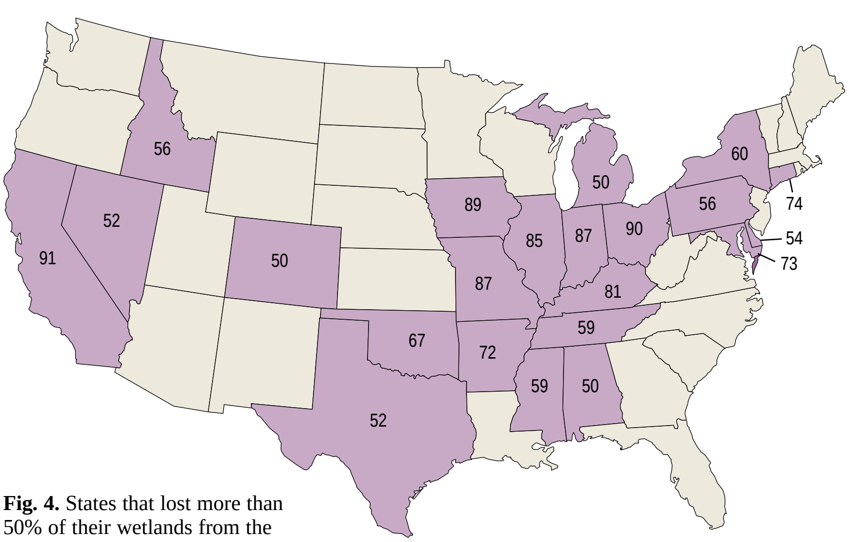


Fig. 4. States that lost more than 50% of their wetlands from the 1780's to the mid-1980's (Dahl and Johnson 1991).

Nature of Water Development and Use

The development of freshwater resources for human use has many consequences for aquatic biota and for riparian and terrestrial species that depend on aquatic ecosystems for food or habitat. Direct human effects include changes in stream and river flows and lake water levels from dams and irrigation (Mesa 1994), the introduction of pollutants (Crowder and Bristow 1988), both intentional and inadvertent introductions of nonindigenous species by providing access pathways (Kitchell 1990; Cloern and Alpine 1991; Mackie 1991), and the overexploitation of selected species, especially fishes and mussels (Hedgecock et al. 1994). Indirect effects on aquatic biota include introductions of extensive atmospheric contaminants (Schindler et al. 1985), widespread use of salts on roads (Likens 1985), change in aquatic species composition from UV-B radiation, change in water nutrient content and temperature from livestock grazing in the riparian zone (Armour et al. 1991), and change in water quality from human development in upstream watersheds (Byron and Goldman 1989; Fisher 1994; O'Dell 1994).

Water developments have single or multiple purposes. For example, stored water may be withdrawn for cooling of electrical power plants, or it may be released for the generation of electric power. A development may provide water for municipal, agricultural, and industrial withdrawals, as well as for recreational uses (boating, fishing, swimming). The American Rivers group (1995) attributed the most frequent threats to the 30 endangered and threatened rivers on their list to dams (13), agricultural (10) and urban (10) runoff, mining (6), and flood-control or navigation demands (6). Other problems include overgrazing, logging, overuse, and sewage. Water projects often must balance competitive uses that can have different direct or indirect effects on aspects of the biological, physical, or chemical environment.

Table 4. Estimated gains and losses of freshwater wetlands in the United States from the mid-1970's to the mid-1980's (in million hectares; modified from Dahl and Johnson 1991).

Wetland type	Mid-1970's	Mid-1980's	Change
Palustrine emergent	9.839	9.929	+0.090
Palustrine forest	22.320	20.942	-1.378
Palustrine scrub	6.275	6.210	-0.065
Palustrine nonvegetated[a]	2.165	2.485	+0.320
Lacustrine	23.327	23.409	+0.082
Riverine	2.073	2.101	+0.028
Total freshwater wetland	65.999	65.076	-0.923
Total intertidal estuarine	2.239	2.215	-0.024

[a]Palustrine nonvegetated wetlands include aquatic beds and unconsolidated bottoms and shores.

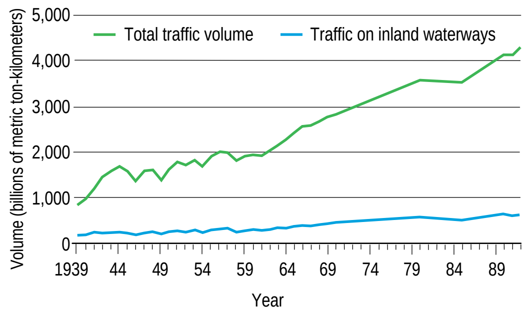


Fig. 5. Volume of domestic intercity freight carried in the United States (modified from U.S. Department of Commerce Bureau of the Census 1975 and 1994).

Flood Control

Flood-control structures (dams, levees, and diversions) may hold back excess runoff (upstream dams), speed runoff (channelization), confine runoff (levees), or do all three for large river basins such as the Missouri–Mississippi River basins. Flood-control structures do not consume water but remove water from rivers and aquatic ecosystems. When flood-control systems fail, the consequences are often catastrophic for the farmlands and cities for which the flood control was provided. The effects of flood control on ecosystems of the Red River, a tributary to the Mississippi River, include shifts from river to lacustrine aquatic habitats, change from river forests to open land from conversion to agriculture, and loss of species richness in habitats (Hardaway and Yakupzak 1981). For example, in the Yazoo Basin on the Mississippi River, flood control caused a significant decline in the quality of the aquatic ecosystem. Only 20% of the stream length now supports a fishery, and even fewer kilometers support a sport fishery (U.S. Fish and Wildlife Service 1979). Since 1870 in south-central Oklahoma, 87% of the riparian forests and 17% of the channel length have been eliminated by flood control (Barclay 1978).

Bank stabilization and navigation structures created erosion and deposition in the natural channel of the Missouri River. Structures reduced the channel from 120,000 to 50,000 hectares (U.S. Army Corps of Engineers 1980; Hunt 1988). Between 1879 and 1954, channelization decreased the total water-surface area of the Missouri River from 49,000 to 16,000 hectares. Modification of the river caused the disappearance of the river otter by 1935. Commercial fish catches declined by 80% between 1947 and 1963. Lake sturgeon, paddlefish, and blue catfish populations greatly declined (Funk and Robinson 1974; Hunt 1988).

Navigation and Transportation

Rivers are important to commerce (Fig. 5). For example, human manipulation of the Upper Mississippi River has a long history. The construction of a lock and dam system in the 1930's to aid commercial transportation not only created a diversity of lentic habitats in the upper basin but also changed the water level and the amount of transported sediment (Holland-Bartels 1992).

Alterations of the Ohio River for navigation since 1800 have been extensive, but the construction of navigation dams from 1900 to 1927 has had especially widespread effects. The biological effects of siltation after clearing of forests in the nineteenth century, combined with modern pollution in the upper third of the river, reduced biological diversity. For example, although 159 fish species were identified between 1819 and 1988, since 1970, 13 of these species have not been found (Pearson 1992).

In Florida, the Jim Woodruff Dam, which opened in 1957, now traps almost all silt and debris from the Chattahoochee and Flint rivers, reducing the flow of nutrients and sediments to the Apalachicola Bay. Dams on the Chattahoochee and Flint rivers eliminated the striped bass fishery and caused a shift to rough and forage fish species (Livingston 1984; Hunt 1988).

Similarly, in the major rivers of the Mobile Bay basin, dams eliminated many aquatic species that still occur in the smaller tributaries. Construction of the Tennessee–Tombigbee Waterway, for example, eliminated 17 aquatic species from the main channel. In the basin, 99 species of snails and 24 species of freshwater clams and mussels are now endangered or threatened, and 14 species of freshwater pearlymussels were lost from the Tennessee River (Ward et al. 1992).

Hydroelectric Power, Irrigation, and Municipal Use

Diversions

When water is diverted to crops, air is humidified and cooled in irrigated areas. Reservoirs behind dams decrease daily air temperature variation over stored waters and increase evaporation. Little change in the local climate is noted unless the reservoir is located in arid lands (Budyko 1982). The ultimate effects of diversions and irrigation on the hydrological cycle are not fully understood and are presently impossible to completely separate from natural factors. Large-scale effects, however, are suggested (Pielke et al. 1992). The quantity and quality of available water, soil moisture, and frequency of extremes such as droughts are affected by precipitation and evaporation changes from whatever source. For example, climate change—the altering of the amount and timing of rainfall, carbon dioxide levels, and temperature—could limit available water and therefore adversely affect agriculture. How soil temperature and moisture are affected by the combination of increased precipitation, carbon dioxide, and temperature is not clear. Therefore, the effects on water caused by anthropogenic change, such as water diversions, cannot be separated easily from processes such as climate change in any given location (MacCracken et al. 1990).

Fourteen major dams that were constructed on the Columbia River for hydroelectric production of power, transportation, and agriculture provide many economic benefits to the region but changed a rapid-flowing river ecosystem to a warmer water, slow-flowing series of impoundments (Strober and Nakatani 1992). The dams inhibit or block migrating fishes and, by flooding spawning grounds, cause changes in competition between species, changes in predator–prey relations, and a decline in the variety and numbers of native fish species. In 1911 the commercial fish harvest on the river was 24,400 metric tons, but by the early 1970's it had declined to 6,800 metric tons. In 1949 the Lower Columbia River Development Program was funded for fish restoration. By 1962, 50% of the harvested coho and chinook salmon were raised in hatcheries. Since then, other innovations, such as protection devices, fish passages, and timing of flows and construction, have been used (Trefethen 1972). Even so, hydrological regulation, especially by dams, is probably still the principal factor in placing 75% of the native Pacific salmon stocks at moderate to high risk of extinction (Nehlsen 1994).

The effects of hydrological regulation by dams have been well-studied. Findings from the Columbia River point out the ecological complexity of dam effects on the declining salmon populations, especially juveniles. Juvenile salmon incur multiple stresses from water agitation below dams and become lethargic and disoriented, which heightens their vulnerability to predation (Mesa 1994). Dams also reduce the biological diversity of aquatic primary producers and their immediate predators, which are prey for fishes. For example, the Columbia pebblesnail population experienced major declines because of the creation of inhospitable habitat along the Columbia River drainage (Neitzel and Frest 1992).

Water development in the Columbia River basin, not including the Snake River, has also been responsible for the loss of more than 108,000 hectares of fish and wildlife habitat (Hunt 1988; Fig 6). The flooding of backwaters, bays, canyons, riparian forests, and river banks eliminated habitat and reduced populations of many animals. Changing water levels also affected nest sites on islands and reduced the nesting success of birds in rookeries. Because of the loss of prey, the abundance of many predators is also declining.


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Fig. 6. The Columbia River in Oregon below the Bonneville Dam. Most of the Columbia River is now controlled by water-development projects.

The Sacramento River drainage has also been affected by human activities and water development. During the past 20 years, the size of the river's salmon population has declined 50% in the Sacramento–San Joaquin watershed because dams and upstream developments eliminated rearing and spawning habitat. The river was greatly modified during the last 200 years to provide more than half of the surface water used in California (Mongan and Miller 1992). More recent modification of diversion and irrigation processes demonstrated that changing the way water is used and treated is possible. For example, California rice farmers flood fields after harvest to get rid of stubble through natural decay instead of through burning; this flooding creates seasonal wetlands for migrating waterfowl. Later, a timed release of the water assists with the migration of salmon downstream to the open ocean (Hunt 1988; Conniff 1993). Such modifications—although they do not address the loss of natural habitats or biological diversity—require cooperation rather than competition in water use.

Irrigation

Irrigation is critical to modern life and to agriculture in the dry western United States. In addition, eastern farmers are beginning to use supplemental irrigation to increase crop yields

Impounded River Systems

River basin development projects, including the construction of dams and irrigation diversions, were led by state and federal government partnerships during the 1950's and 1960's. Human demands for flood protection, water for irrigation, hydroelectric power, navigation, and bank stabilization resulted in large public works undertaken by the U.S. Army Corps of Engineers and the U.S. Bureau of Reclamation. These projects pose pervasive threats to midcontinent aquatic ecosystems because they have significantly altered the physical characteristics of most of the region's major river systems, which has resulted in numerous adverse effects, including the loss or decline of many native plant and animal species, especially fishes.

In fact, The Nature Conservancy's *Troubled Waters: Aquatic Ecosystems at Risk* (Flack and Chipley 1996) states that nationally 67% of freshwater mussels, 64% of crayfish, 37% of freshwater fishes, and 29% of amphibians are at risk. Alterations of river habitats have been instrumental in causing the decline in abundance and diversity of many species (Figure).

Unimpounded River Corridors

The large-river systems of the midcontinent are characterized by native riverine species of fish, mussels, and crayfish. These systems course through an arid to semiarid environment where often the only forested habitat is the extensive riparian corridor that lines these streams. This riparian zone provides important habitat for wetland species and serves as a migratory corridor for waterfowl, shorebirds, mammals, and other animals. The native river fishes require flowing water habitats for either all of their life stages (for example, the darters) or for only a portion of their life history (such as paddlefish, sturgeons, and other migratory species).

Melting snowpack in the distant mountains and high-intensity summer rainstorms are the dominant forces that shape the river channels as they cut through alluvial materials. The rivers are composed of shallow, often braided channel habitats, with warm, turbid water interspersed with deep pools along the outside of meander bends. The annual scour and fill cycle deposits sediments on the floodplain and replenishes the bars with the fresh sand and soil necessary for cottonwood regeneration.

After Impoundment

The impacts of damming and flow regulation can be classified as immediate or delayed (Holden 1979) or as first-, second-, and third-order effects (Becker and Gorton 1995). The immediate or first-order effects are obvious blockage of upstream and downstream migration of fishes and alteration of the downstream habitat by dewatering and releasing cool (or cold) and clear (free of suspended fine sediments) water through the low-flow portion of the channel. Many kilometers of upstream river corridors are inundated and converted to lakelike habitats. The cold, clear water tends to pick up sediment from the riverbed or banks, which causes the bed to gradually lower or widen. Water releases from midcontinent reservoirs generally produce lower water temperatures in summer and higher water temperatures in winter. Native riverine fishes adapted to the natural temperature regime are displaced downstream and may be unsuccessful in reproduction because of changes in timing of physiological processes keyed to temperature cues. Consequently, these native riverine species may be (and often are) gradually replaced by generalist species, which are often nonindigenous species introduced by humans and adapted to cool or cold water. Basses, sunfish, and northern pike often escape from reservoir stocking programs into the river below, where these predatory nonindigenous fishes not only compete for habitat but also prey on the young native fishes.

The longer-term effects of dams include a degraded and widened channel that can carry a higher volume of flow. This, along with a decrease in the magnitude of peak flows and the trapping of sediments by reservoirs, results in a much lower frequency of overbank flooding. The lack of overbank floods, which deposit sediments, and the erosion of the bed and banks by the sediment-starved reservoir releases result in the loss of sandbars and cause bankwater habitats to be replaced by steep, raw banks along the channel.

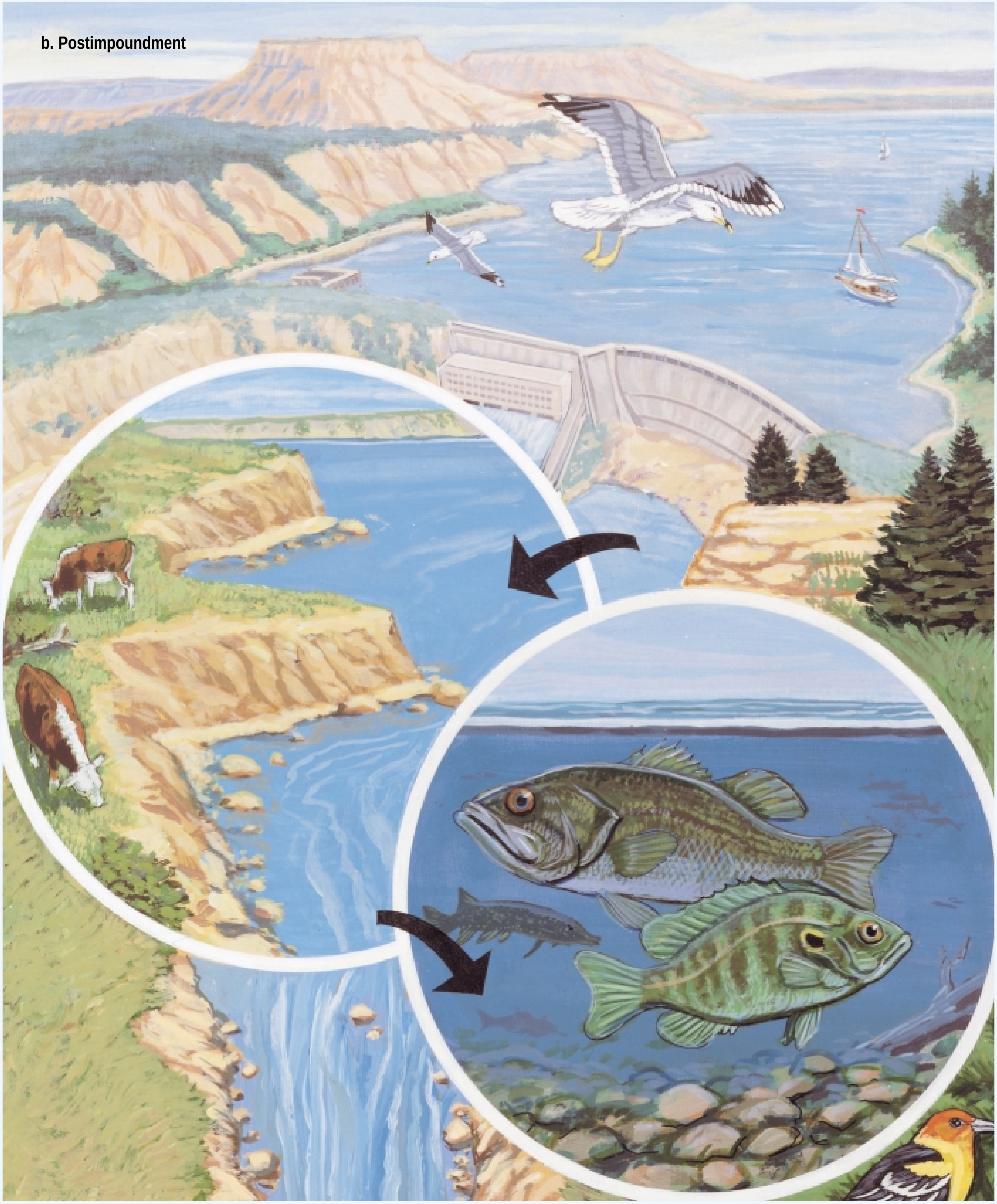
Over a period of decades, the cessation of the annual scour and fill cycle that replenishes the bars with fresh sand and soil causes riparian cottonwood stands to gradually become open; these stands are eventually replaced by nonindigenous plant species. Thus, biological productivity is reduced as the number and diversity of wetland and riparian communities along the river corridors decline. In addition, the reproductive success of the native riverine fishes is greatly reduced due to temperature changes and loss of spawning areas. Those fish that do spawn have fewer of the remaining backwater habitats they need to feed and grow. As previously noted, after nonindigenous species such as pike and centrarchids invade, they compete with native species for habitat and they prey on the young native fishes.

Rehabilitation, Monitoring, and Research

Restoration of these declining large-river ecosystems is being discussed widely among resource agencies, conservation groups, and the public. Restoration of these areas to their natural predevelopment condition is almost impossible, however; most restoration efforts are simply attempts to rehabilitate selected segments of river to some predetermined structure and function (Gore and Shields 1995). Dramatic examples include the Kissimmee River in Florida, which biologists are trying to reroute to its original channel, and the March 1996 "test flooding" by the Department of Interior in the Grand Canyon, Arizona. Federal listing of several native large-river species as threatened or endangered and declining biodiversity of aquatic and riparian communities throughout the midcontinent have prompted a reevaluation of how the U.S. Army Corps of Engineers and the U.S. Bureau of Reclamation operate many large federal reservoirs.

Two major studies are ongoing in the Colorado River basin: the Recovery Implementation Program for Endangered Fish Species in the Upper Colorado River basin and the Grand Canyon studies. Extensive monitoring studies have recently begun in the Missouri and Yellowstone rivers. The U.S. Army Corps of Engineers has funded wetland, riparian, and fisheries studies as part of the Missouri River Master Manual Review and Update Study, and the U.S. Bureau of Reclamation is funding fisheries and geomorphological studies in the upper Missouri and Yellowstone rivers. The objective of these studies is to build an analysis and decision support system to allow water managers to better understand the trade-offs associated with various operating scenarios. Biological monitoring is being designed and coordinated through the Missouri River Natural Resources Committee, a group of scientists from each

Figure. Depiction of a segment of the Missouri River with a) pre- and b) postimpoundment views of the river channel and biota, including close-up views (see circles) of a benthic (bottom) habitat and a marsh habitat before and after impoundment. The preimpoundment river corridor provides for a rich and complex array of habitats, including a thriving riparian and wetland community and a diverse instream fauna of benthic (bottom-dwelling) insects, mollusks, and native riverine fishes such as paddlefish, pallid sturgeon, and sauger (see bottom circle in a). In contrast, the postimpoundment river channel, though it may appear



visually attractive at first glance, has eroded banks, is devoid of the native riparian habitats that are so invaluable in this region, and has an impoverished instream benthic fauna, including a fish fauna that is now dominated by nonindigenous centrarchid species (basses and sunfish; see bottom circle in b), which gradually replace native species.

of the basin states, the U.S. Army Corps of Engineers, U.S. Fish and Wildlife Service, U.S. Geological Survey, U.S. Bureau of Reclamation, and the Western Area Power Authority.

Much remains to be learned about the ecology and life-history requirements of the biological communities of the midcontinent river corridor before rehabilitation schemes can be designed by ecologists and natural resource scientists. The opportunities for designing and carrying out scientific studies on the physical and biological processes of large warmwater environments appear good for the next decade. By perfecting large-river sampling techniques and intensive monitoring in large-scale experiments (such as the Grand Canyon), we will have more science-based information about how to manage these systems, which in turn will allow us to more accurately assess the status of these biological communities.

See end of chapter for references

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and as crop insurance to guard against drought. About 86% of the irrigated land in the United States is in the West, where irrigation waters are delivered by furrows or ditches, flooding, sprinkler systems, or underground systems. Future growth in irrigated agriculture is limited by water supply and cost, high energy and operating costs, water laws, water pollution from agricultural salts and chemicals in runoff, and competition for land and water from urban areas. Because urban growth competes for water supplies (Lea 1985), it is important to agricultural water users. In 1990, 517 million cubic meters of water per day were withdrawn for irrigation, and 288 million cubic meters per day were consumed. During that year, water use for governments, businesses, and households was 148 million cubic meters per day and consumption was 25.6 million cubic meters per day (Solley et al. 1993).

Interbasin Transfer

Ten major interbasin transfers, which move water from one river basin into another by canal, aqueduct, or pipeline, occur in the western United States (van der Leeden et al. 1990). The movement of water across river divides is not unique to the West, however. Since 1967, New York City has met about 50% (2.83 million cubic meters per day) of its daily demand for water by transferring water from the Delaware River (Major 1992). Another example of interbasin transfer was the construction of the Colorado–Big Thompson diversion, which began in 1938 and was completed in 1956; this diversion, which transfers water through the Alva D. Adams Tunnel (Fig. 7), was supported by the federal government under the 1902 Reclamation Act. Recent growth of cities and suburbs has placed additional demand on this water. Although the Colorado–Big Thompson diversion transfers the greatest volume of water in Colorado, 37 other large and small transmountain diversions contribute to the annual transfer of more than 802 million cubic meters of water from the basins of the Gunnison, San Juan, and Colorado rivers on the Western Slope across the Continental Divide to the Eastern Slope of Colorado, where 80% of the state's human population resides (Tyler 1992). These diversions changed the ecological character of both slopes by elevating salinity (chemical salts) in Colorado River waters and by reducing the amount of water in the basin and in the river that, according to treaty obligation, should flow to Mexico. The projects limit future growth and development on the Western Slope and encourage continued urban growth on the Eastern Slope.

Point and Nonpoint Pollution

Public concern about the quality of the nation's waters was demonstrated by the passage of the Water Pollution Control Act of 1956 and its amendments in 1972 and by the passage of the Clean Water and Safe Drinking Water acts of 1977. These acts developed a clear statement of the national goal to restore and maintain the chemical, physical, and biological integrity of the nation's waters (Federal Water Pollution Control Act 1972). In spite of substantial cleanup, however, harmful substances are still discharged into the nation's natural waters, and their effects are often not obvious. The following few examples demonstrate the physical and biological nature and effects of these pollutants (also see chapter on Environmental Contaminants).

Pollutant Sources

Many studies have been conducted on the effects of specific water or airborne pollutants on aquatic biota. Many of these studies, though, have been narrowly focused; assessments of the magnitude of effects on ecosystems are incomplete or nonexistent. Research on the effects of atmospheric contaminants, such as hydrogen, nitrogen, and sulfur, on small lake or stream ecosystems is an exception and is considered in a separate chapter.

Irrigation runoff is a major source of nonpoint pollution, which occurs over broad land areas often at low levels. Contaminants in such runoff include sediments, salts, fertilizers, pesticides, and bacteria (Lea 1985). Airborne contaminants are another widespread source of nonpoint pollution. Whenever contaminants such as oxides are released into the atmosphere—for example, by burning fossil fuels—they can be transported considerable distances before falling to the ground. Not only can such contaminants cause changes in surface-water quality, but animals and plants can also be harmed by such pollution. One airborne pollutant, nitrate, is of particular concern to the environment because elevated nitrate levels in lakes, rivers, marshes, and other water bodies can contribute to increased plant productivity or potentially to increased acid levels. Nitrate in rain or snow can lead to acidification of the upper parts of some watersheds.

Release of Atmospheric Contaminants into Lakes and Streams

Beginning in the 1970's, national attention focused on the release of atmospheric hydrogen, sulfur, and nitrogen into small, sensitive lakes and streams. During this period, many studies of aquatic ecosystems began, and the results of these studies improved our understanding of the structure and function of these systems.

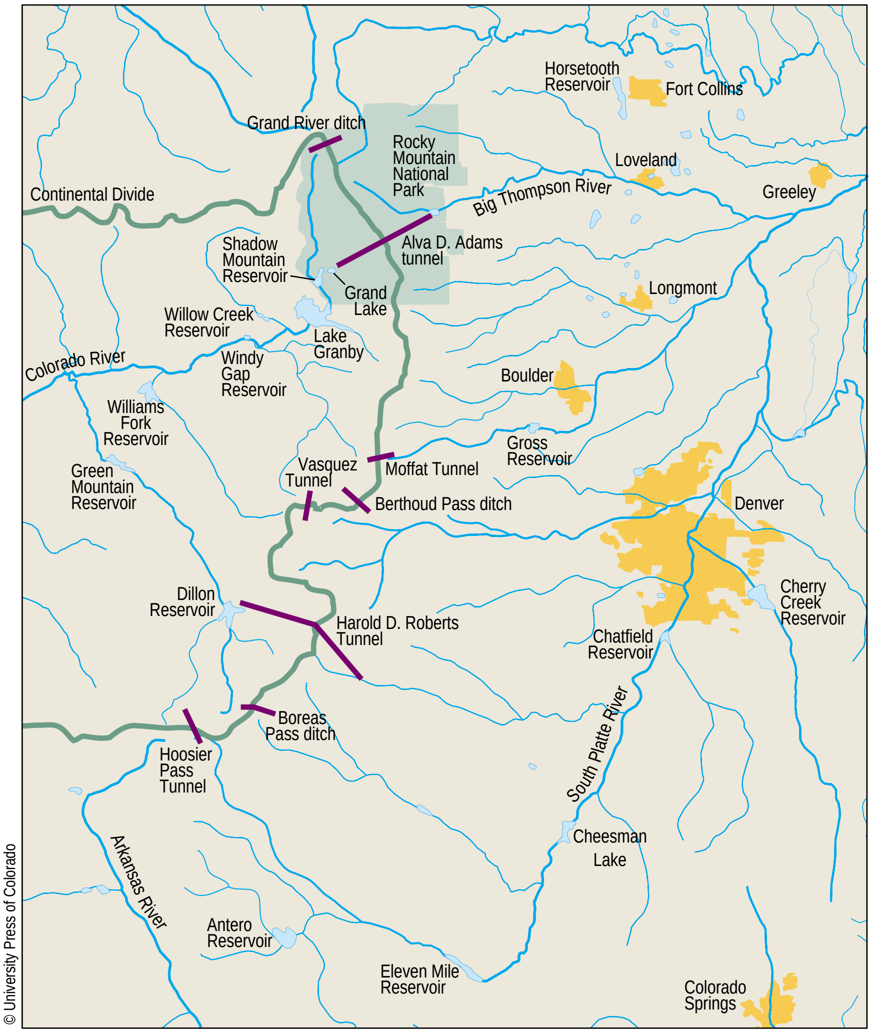
One of the better long-term studies on small lakes was conducted in the Experimental Lakes Area in southwestern Ontario (Schindler et al. 1985; Schindler 1987). This study showed that the earliest response to stress from experimental acidification was a change in species composition of phytoplankton. Other research revealed morphological change in benthic invertebrates such as crustaceans, which are sensitive early indicators of stress from pollution. It is important to note that aquatic ecosystem processes such as productivity, respiration, and nutrient cycling are relatively less sensitive to stress.

Effects of Multiple Pollutants on Lakes and Streams

Point sources of pollutants in surface waters affect local areas but often can be eliminated more easily than more widespread nonpoint source pollutants. Examples of point-source pollutants include nitrogen and phosphorus, which increase algal growth in surface waters and estuaries.

Many studies have revealed ecosystem-level responses to air- and waterborne pollutants in the Great Lakes. The addition of limiting nutrients such as nitrogen and phosphorus to the lakes by nonpoint drainage waters from developed and agricultural areas creates eutrophication and locally high levels of algal biomass (Conley et al. 1993; Schindler et al. 1993). This rapid increase of biomass limits other nutrients such as silica, essential for exoskeleton development of primary producers such as microscopic diatoms. The subsequent fluctuation in species composition of producers alters the cycling of nutrients and changes populations of zooplankton and their predators. Rapid increases in aquatic plant biomass can then drastically change water nutrient levels and lower oxygen concentrations.

This, in part, is what happened during the 1960's and 1970's in Lake Erie (Nalepa et al. 1991) and is what initiated the cleaning of the lake. Such chemical change can directly affect benthic species such as mussels, which are especially sensitive to increased turbidity and low dissolved oxygen levels. Manufacturing in the Great Lakes region historically released high levels of contaminants into the lakes. Organic pollutants in particular may concentrate in benthic macroinvertebrates, reducing the size of their populations and promoting the incorporation of contaminants into the food chain (Nalepa 1991). Increased organic



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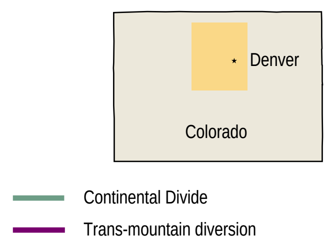

Fig. 7. Major trans-mountain diversions in Colorado (Tyler 1992).

contaminant concentrations in predatory fishes limited human consumption of those fishes, greatly reduced the value of a historically important fishery, and reduced fish population sizes and community composition by limiting reproduction.

Long-term studies of point and nonpoint pollutants have been conducted in the Great Lakes and their tributaries (Smith et al. 1992). Concentrations of toxic materials such as arsenic, chlordane, dieldrin, dichlorodiphenyl-trichloroethane (DDT) and related compounds, and polychlorinated biphenyls (PCB's) recently declined, but mercury levels remained constant. These data represent the success of major efforts to clean up point-source discharges.

Other regions have not improved. Streams in agricultural areas of the Midwest have high herbicide concentrations. In the Mississippi River and its major tributaries, alachlor and atrazine occasionally exceed the maximum contaminant levels set by the U.S. Environmental Protection Agency (Smith et al. 1992). These contaminants adhere to the high loads of suspended sediments and are carried long distances, which increases their potential harm to aquatic organisms and humans (Ellis 1993).

Most aquatic ecosystems suffer from multiple anthropogenic stresses (Karr et al. 1985). The Illinois River in the Midwest exemplifies a river in which increased city populations and agricultural uses in its watershed, hydrological regulation, discharge of oxygen-demanding wastes, overuse of water, and introduced nonindigenous species have interacted to drastically alter the aquatic ecosystems. The river, with a drainage area of 73,000 square kilometers, starts southwest of Chicago and flows to the Mississippi River. Lands surrounding the river were cleared, and bottomlands, including lakes, were drained for agriculture. Sediment, fertilizer, and pesticide runoff from agriculture now enter the river, adding to urban runoff. Elevated and rising nitrate levels were noticed from 1975 to 1994 by Schideman and Blanchard (1994). The number of phytoplankton is high in reaches with nutrient enrichment from farmland runoff and low in reaches with turbidity and toxic metals. The poor water quality of the Illinois River is exemplified by high turbidity and low dissolved oxygen levels (Anderson et al. 1991), which have resulted in an increase in abundance of aquatic species that favor cloudy water (bottom-feeding fish). More recent studies (Lerczak and Sparks 1995) have indicated that water-quality improvements on the Illinois River have resulted in a return of a more diverse fish community.

Sedimentation is a factor in the loss of aquatic plants in the Illinois River; sedimentation and the loss of aquatic plants contributed to the disappearance of 20 fish species and the loss of the commercial fishery. Loss of aquatic plants also adversely affects the waterfowl that feed on the plants. In addition, the draining of bottomland lakes and marshes further decreased the abundance of fishes and the habitat of migratory waterfowl along the river (Starrett 1972; Holland-Bartels 1992). In sum, contaminants that are released into the nation's waters greatly restrict the usefulness of receiving waters for human water supply, fish habitat, other wildlife habitat, and water-contact activities.

Sediment

Sediment that erodes from the land into streams and rivers is transported to reservoirs and deposited behind dams, where flow velocity decreases. Data compiled by Crowder (1987) indicate that this process has decreased the storage volumes of large reservoirs in the United States by 0.22% annually (van der Leeden et al. 1990; Table 5). Hunt (1988) cited losses as high as 73% over 30 years (1942–1972) from the Ocoee Dam Number 3 in North Carolina.

Trapped sediment alters river channel characteristics, which, in turn, affect water-table elevation and adjacent riparian vegetation. With a reduced downstream sediment load, rivers below dams erode their beds and banks. This decreases floodplain width, reduces riparian habitat area, and alters river channel character and level. In response to these changes, people often line and stabilize the banks of the main

Table 5. Water storage capacity that is lost annually (in million cubic meters) from reservoirs because of sedimentation; data are for reservoirs in the contiguous United States with total capacities of 6.17 million cubic meters (modified from Crowder 1987).

Farm region	Total water storage capacity	Usable water storage capacity	Water storage capacity lost (est.)	Percent water storage capacity lost (est.)	Reservoir sedimentation from cropland (est.)	Percent stream sediment originating on cropland (est.)[a]
Northeast	45,000	31,100	34.7	0.08	10.1	29
Appalachian	73,400	37,800	93.2	0.13	27.0	29
Southeast	90,800	58,700	157.1	0.17	51.8	33
Lake states	36,200	24,100	97.6	0.27	62.4	64
Corn Belt	49,000	18,800	129.3	0.26	81.4	63
Delta states	52,700	24,800	108.0	0.20	44.3	41
Northern plains	97,400	67,100	227.8	0.23	82.1	36
Southern plains	136,100	57,500	255.9	0.19	48.6	19
Mountain	206,200	170,400	373.3	0.18	29.9	8
Pacific	111,900	92,200	544.9	0.49	49.0	9
Total	898,700	582,500	2,021.8	0.22	486.6	24

[a] The percentage of the sediment in the stream that originated on cropland was calculated by dividing the estimated sedimentation from cropland in the reservoir by the estimated lost water storage capacity.

channel and affected tributaries to protect riparian land and adjacent developments. Such bank stabilization, though, further alters the natural channels (Simons 1979; Hunt 1988). An example of such processes can be seen downstream of the Gavins Point Dam on the Missouri River upstream of Yankton, South Dakota. Between the Gavins Point Dam and Ponca State Park, 0.88 hectares of land per kilometer of river have been lost each year since the dam was built in 1956. Similar data from four Missouri River dams show erosion losses below the dams of 53 hectares per year (Fort Peck), 30 hectares per year (Garrison), 121 hectares per year (Fort Randall), and 81 hectares per year (Gavins Point; Hunt 1988). This combination of erosion and siltation from river development and stabilization has caused the loss of 40,600 hectares of aquatic habitat, 26,400 hectares of island and sandbar habitat, and 125,000 hectares of associated riparian habitat along the Missouri River (Hunt 1988).

Habitat modification by sedimentation and siltation has caused extensive separation of fish populations in the Southeast. The most imperiled aquatic species live in creeks and small rivers, where they depend on clean stream-bottom substrates, especially for reproduction. Poor land-use patterns, though, have eliminated much suitable habitat for bottom-dwelling species (Walsh et al. 1995). In addition, removal of riparian vegetation has increased soil erosion and siltation because water cuts into banks that are no longer protected by plant roots. Loss of riparian vegetation also has increased soil temperature and soil water evaporation and has elevated stream water temperatures, which have increased algal growth and water stagnation. The end result is a great loss of plants and animals (Campbell 1970; Hunt 1988).

Effects of Grazing on Aquatic Ecosystems

The deterioration of riparian zones is a major factor in the loss of the integrity of aquatic ecosystems (Cummins et al. 1989; Armour et al. 1991; Gregory et al. 1991). Cattle grazing, especially in the West, has caused major effects on riparian zones (Fig. 8). Because riparian aquatic zones are often the most productive, they are grazed more heavily (Rinne and Medina 1988; Armour et al. 1991), which increases siltation. This increased siltation can cause the loss of fish-spawning areas and can reduce food for fishes by lowering the number of invertebrates, destroying streamside and instream cover, increasing water temperatures and velocities, decreasing organic matter, and reducing the number of species that prefer to inhabit cold, clear water (Behnke and Raleigh 1978).

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Fig. 8. Cattle grazing causes deterioration of riparian vegetation, soil erosion, and siltation of streams.

Grazing also degrades aquatic ecosystems by adding nitrogen and reducing leaf litter necessary for stream benthic invertebrates such as shredders. Although the riparian zone contains ecologically important plant communities, wildlife, and fisheries, it has too often been considered strictly a component of the terrestrial-aquatic boundary instead of a zone with significant influence on both terrestrial and aquatic ecosystems.

Mining

Stream siltation from mining is significant in some regions of the nation. Mining of coal, oil shale, oil sands, sand, and gravel is a particularly important source of elevated stream sediment loads. Mining degrades water quality and channel integrity and withdraws large quantities of water for processing. Runoff from mines may also contain elevated concentrations of heavy and trace metals (Boyles et al. 1974; Hardie et al. 1974) or suspended sediments with metals. The direct effect of mining on surface waters is significant. By 1965, 9,300 kilometers of streams (about 23,000 hectares) and 12,000 surface hectares of impoundments and reservoirs in the United States were affected by surface coal mining, including acid drainage from mines (U.S. Department of the Interior 1973).

Acid mine drainage has been well publicized historically and remains a common, seasonal problem in much of the eastern and southeastern United States. A National Stream Survey by the U.S. Environmental Protection Agency revealed that 10% of the streams in the northern Appalachians were acidic because of receiving mine drainage during spring baseflow. Throughout the survey area, almost 5,000 kilometers of stream, or 2% of the total survey stream length, were acidic because of acid drainage from mines; another 6,000 kilometers were severely affected (Herlihy et al. 1990).

Although the effects of gravel mining on aquatic ecosystems have been little publicized, the removal of gravel from one area of a channel affects areas upstream and downstream as bedload materials move to establish a new equilibrium in the stream bottom (Kondolf 1994). Following mining, gravel necessary for spawning by fishes is lost, and suspended sediments can travel far downstream, eliminating even more spawning area and altering the environment for benthic invertebrates. Stream instability from such mining also eliminates sandbars where migratory shorebirds often nest. For example, colonial waterbirds, such as the endangered least tern and the threatened piping plover, must use areas such as sand pits in place of lost sandbar habitat along the Platte River in Nebraska (Sidle and Kirsch 1993).

Fig. 9. The Colorado River basin (Tyler 1992).

Ecological Ramifications of Water Use

Colorado River

Water regulation has greatly altered the aquatic and riparian ecosystems in the Colorado River basin (Johnson 1977; Carothers and Brown 1991; Fig. 9). A century ago, few Americans knew much about the vast 632,000 square kilometers that make up the basin of the Colorado River, which originates as the Green River in the Wind River Range of Wyoming and as the Colorado River at Grand Lake, Colorado. After the scientific explorations of the river by Joseph Ives in 1857 and by John Wesley Powell after the Civil War, a period of mining began late in the nineteenth century. But the real effect on the drainage of the Colorado River did not occur until 1922, when the Colorado River Compact was signed. This agreement among the states in the river basin not only inadvertently divided too much river water among the basin states but, more importantly, also created the basis for major construction to put the basin's water to what was perceived as beneficial use. Thus, major alteration of the river to the benefit of the lower basin began with the completion of the Hoover Dam (Lake Mead) in 1935. In the upper basin, the most significant projects are the Glen Canyon Dam (Lake Powell), completed in 1963 (Figs. 9 and 10), and the Flaming Gorge Dam (Fig. 9) and reservoir on the Green River, completed in 1964.

The effects of water regulation through dams and through regulated water use on aquatic and riparian ecosystems were not understood at the time of the 1922 compact. Some initial studies were made before the construction of dams, but a systematic study of the affected ecosystems did not begin until 1971 (Johnson 1977); this systematic study was mostly confined to the river inside Grand Canyon National Park. By then, the drainage had significantly altered the river and had affected federal lands, numerous national parks and equivalent reserves, and Native American lands. The effects on the environment were profound and complex.

Control of the river eliminated naturally silt-laden floodwaters. Before construction of the dams, floods were much greater than today and brought much-needed sediments that provided nutrients, supplied new substrates for shorter-lived native riparian vegetation, and replenished the eroding shoreline (Fig. 11). With the downstream sediment loss, water clarity and light

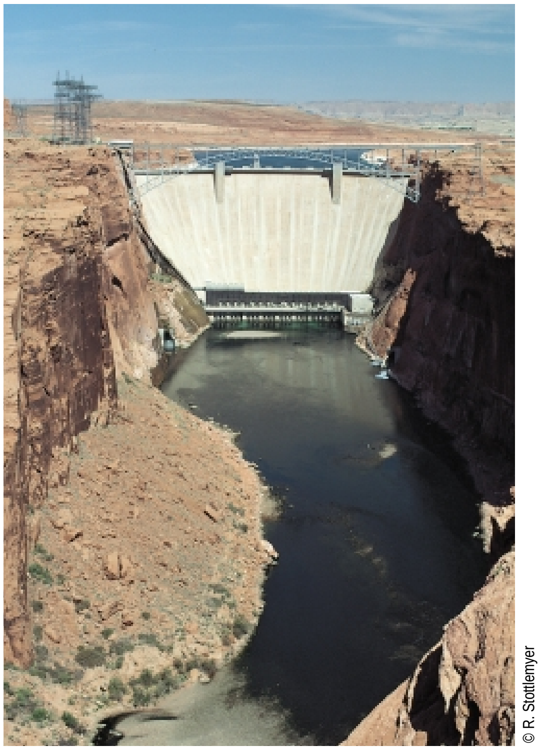

Fig. 10. Glen Canyon Dam, near the Arizona–Utah border.

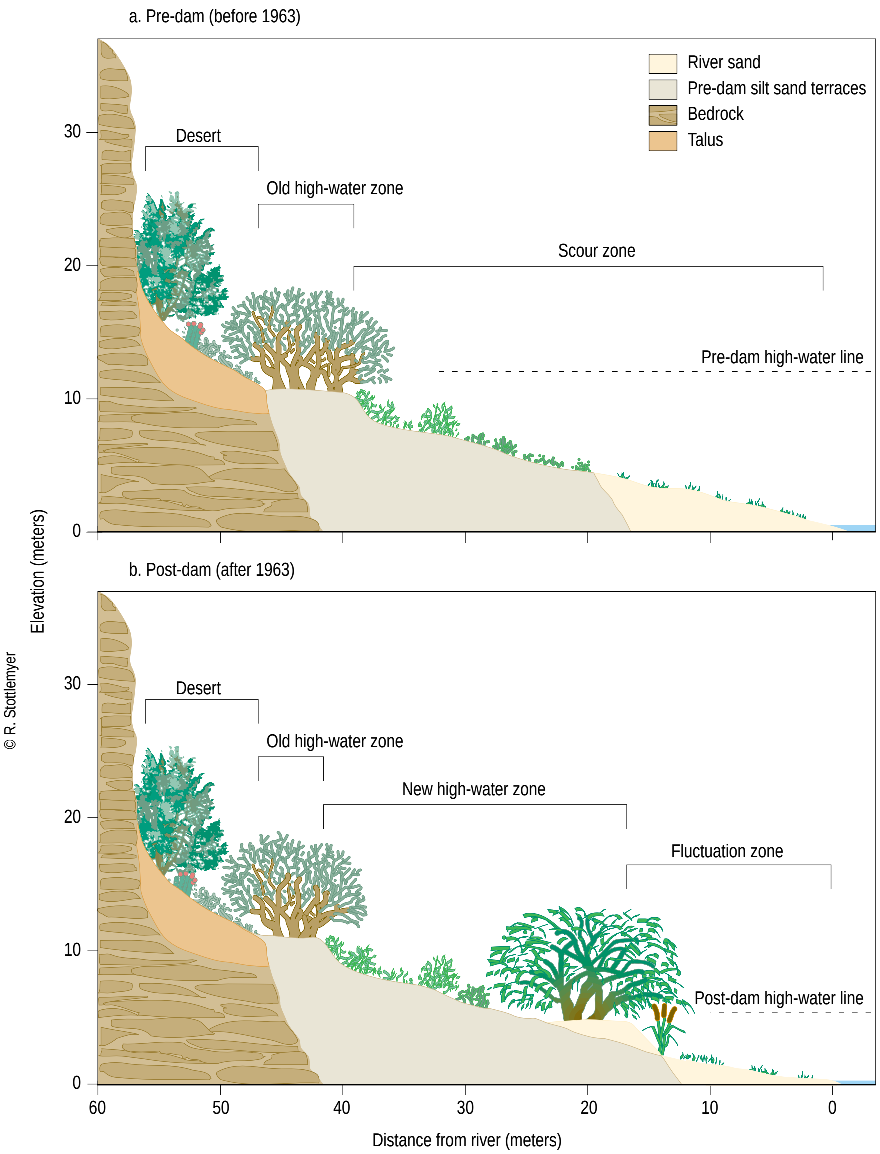

Fig. 11. Changes in downstream riparian zones caused by the construction of the Glen Canyon Dam.

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penetration increased and the environment for plankton and benthic algal growth improved, but phosphorus in stream waters—a limiting nutrient for freshwater algae—declined because of sediment loss (Carothers and Brown 1991; Fig. 12).

The reduction of water temperature and the variation in temperature in the rivers are other significant effects of hydrological regulation. Behind larger dams, water is released from considerable depths where temperatures are colder. In the Grand Canyon, for example, the average water temperature typically was about 28°C before the construction of the dams, but has been 11°C since the construction of the dams (Carothers and Brown 1991). Temperature reduction stresses native aquatic organisms and confines them to small, unregulated, and warmer tributaries. Conversely, improved water clarity, light, sustained flow, and reduced water temperature provide suitable conditions for the invasion of nonindigenous fishes such as trout and salmon, and their invertebrate prey. Such a change in important species has altered the entire aquatic community.

Flow regulation altered the natural predator–prey relations by benefiting nonindigenous predators of fishes. A well-studied example of this process is the effect of such species on native fish species below Glen Canyon Dam and above Lake Mead (Johnson 1977; Carothers and Brown 1991). Historically, the fish community in the Colorado River was dominated by Colorado squawfish, a native chub species, and flannelmouth and razorback suckers. Common carp and catfishes were introduced in the late nineteenth century and dominated much of the river before the major dams were completed in the twentieth century. After damming, flooding and the river's sediment load were greatly reduced; also, the water cleared and averaged 14°C cooler. These physical changes forced most of the remaining native fish species into the small but warmer tributaries and completely altered or eliminated their food sources on the river bottom. The lakes behind the dams became reservoirs that support additional nonindigenous fishes such as rainbow trout. In the clearer and cooler river, rainbow trout moved freely upstream and became significant predators on the remaining native fish species. Thus, the prognosis is poor for native fishes in much of the Colorado River today, as it is for native fishes throughout the Southwest. Razorback suckers no longer

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Fig. 12. Control of the Colorado River by dams upstream eliminated naturally silt-laden floodwaters, increasing water clarity and light penetration in the river in the Grand Canyon, thereby reducing habitat for native fishes.

reproduce in the Grand Canyon below the Glen Canyon Dam, and their reproductive success is poor in Colorado River reservoirs; the species will probably be eradicated by early in the twenty-first century. The Colorado squawfish is considered extirpated from the Colorado River below Glen Canyon Dam. The humpback chub is the only remaining chub species, but for spawning it requires water temperatures warmer than those now found in this stretch of the Colorado River. In addition, native fish species were deliberately eradicated by poisoning because nonindigenous trout provide a more economical sport fishery (Carothers and Brown 1991). As with so many studies of species that were lost during major human alteration of an aquatic ecosystem, the nearly complete absence of biological and ecological research on the Colorado River before 1960 means the role of the human effect in the decline and extinction of these species cannot be fully quantified. In addition to the effect from hydrological regulation of the Colorado River, major portions of the riparian habitat were destroyed by grazing, land-use practices, and polluted surface water.

Reservoirs created by damming can also create their own set of internal problems, such as the effect of fluctuating water levels on shoreline erosion and developments (Lorang et al. 1993), high fecal coliform levels (Doyle et al. 1992), invasions of nonindigenous plants such as hydrilla (Bain 1989), and high nonpoint contaminant levels in reservoir sediments (Novotny and Chesters 1989).

Before the construction of Glen Canyon Dam, annual flood flows near the Grand Canyon exceeded 2,400 cubic meters per second (Carothers and Brown 1991), and frequent flooding subjected the riparian community to unpredictable change. Such natural flooding created a roughly 9-meter-wide zone along the river where only short-lived native species, such as grasses, could exist in small numbers (Figs. 11, 12, and 13). Longer-lived vegetation, such as mesquite and cat-claw acacia, occur farther than 4.5 meters above river level where larger, periodic floods do not reach but from where the plants' roots can still reach water. Adjacent to this upper zone is desert vegetation. Overall, the riparian zone is one of the most productive vegetation associations and is especially important in drier regions. This zone represents a transition between different ecosystems, and it is a major pathway for plant and animal migration. Hydrological regulation, though, has caused the invasion of the riparian zone by nonindigenous vascular plants, particularly saltcedar or tamarisk (Fig. 14). Some native vegetation, especially coyote willow, also became established in what once was the flood zone. Overall, hydrological regulation increased the riparian

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Fig. 13. Lack of beach replenishment by silt-laden floodwaters means natural erosion in the Grand Canyon significantly erodes beaches and changes the riparian zone.

biomass by allowing longer-lived shrubs and trees and animal species to exist in that zone—biological diversity increased in part through species substitution. After hydrological regulation, restoration of equilibrium in the new riparian community takes more than 10 years, and the number, frequency, and balance among native and nonindigenous species are not well understood.

More than 12 million hectares of riparian and associated habitat were flooded in the Colorado River basin by water development projects. This represents a gain of narrow riparian corridors in steep-walled canyons but a large loss of broad riparian flats in other stretches. Although the lost flats made up only a small percentage of the landscape, they included the important cottonwood and cottonwood–willow forests along the river; the loss of these forests is attributed to inundation and to dehydration caused by the interruption of seasonal streamflow or reduction of streamflow in unflooded areas. This, in turn, has caused many bird populations to decline (Hunt 1988). Farming and ranching communities that moved into the basin do not support bird populations as large as those in native riparian communities (Conine et al. 1978).

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Fig. 14. Tamarisk (the taller, darker green shrubs) is a nonindigenous plant that has invaded the riparian zone of the Colorado River in the Grand Canyon since hydrological regulation of the river.

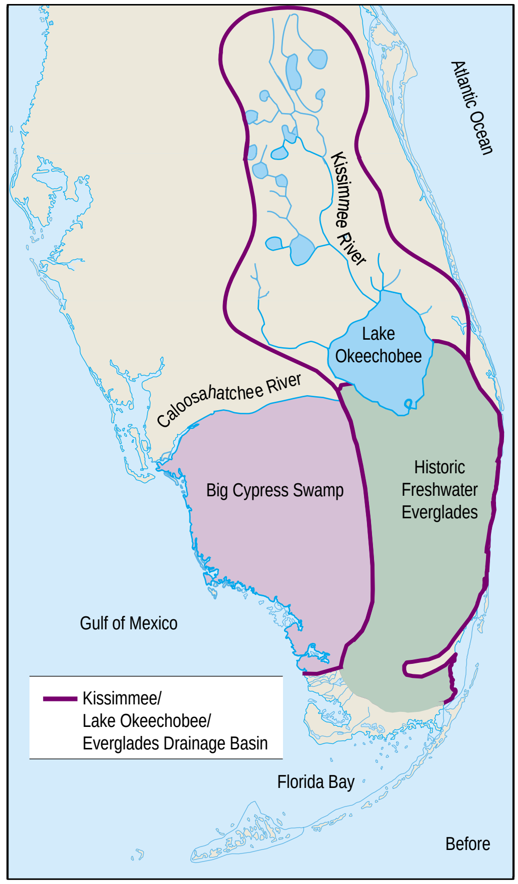
After three decades, the hydrological regulation of the Colorado River in the Grand Canyon has fundamentally changed the structure and function of the aquatic and riparian ecosystems. Changes in aquatic chemical and physical properties do not favor native species of plants and animals but do support some nonindigenous species, which is most apparent in the few native fish species that must now compete with 20 nonindigenous fish species. In addition, the invasion of what used to be the riparian scour zone (area of the riverbank that is swept clean of vegetation by fast-flowing water) has not yet stabilized, and the eventual biological diversity, biomass, and relative mix of native and nonindigenous species is not yet known. New species, especially invertebrates, continue to invade. The insect community in particular is dynamic (Carothers and Brown 1991).

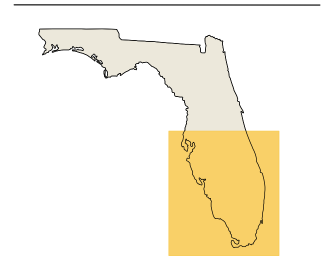
The Everglades of South Florida

In southern Florida, flood-control structures, canals, and conservation areas drain lands for agriculture, flood natural habitat, and rapidly remove floodwaters. These projects reduce flooding in the greater Miami urban area and protect the Biscayne Aquifer, a subsurface water-bearing layer and the source of Miami's fresh water. A history of use, overuse, and conservation, beginning in 1896 and continuing to the present, has ensured sufficient fresh water to the metropolitan area and lowered water levels for agriculture (Fig. 15). But the cost has been massive, and the change of the Everglades ecosystem is long term (Davis and Ogden 1994).

Factors that regulate the amount and diversity of living and dead biomass in time and space include moisture, temperature, high wind, fire, and nutrient availability (Gunderson and Snyder 1994). Long-term cycles, such as shifts in sea level and climate, episodic events such as hurricanes and fire, and annual variation such as wet and dry seasons must be understood to measure potential effects of human use on an ecosystem like the Everglades. In this diverse ecosystem, species adapt to change by being mobile or, as is true for most plants, by having a wide tolerance for change. Taken together, these physical forces drive the biological diversity in the Everglades.

The 5,000-year-old Everglades are an ecosystem with rich biological diversity that is dominated by sawgrass, wet prairie, tree islands, sloughs, ponds and creeks, and the highly productive mangroves (Fig. 16). The ecosystem's diversity and productivity result from the complex interaction of seasonally high and low water levels that bring in needed inorganic nutrients and promote the decomposition





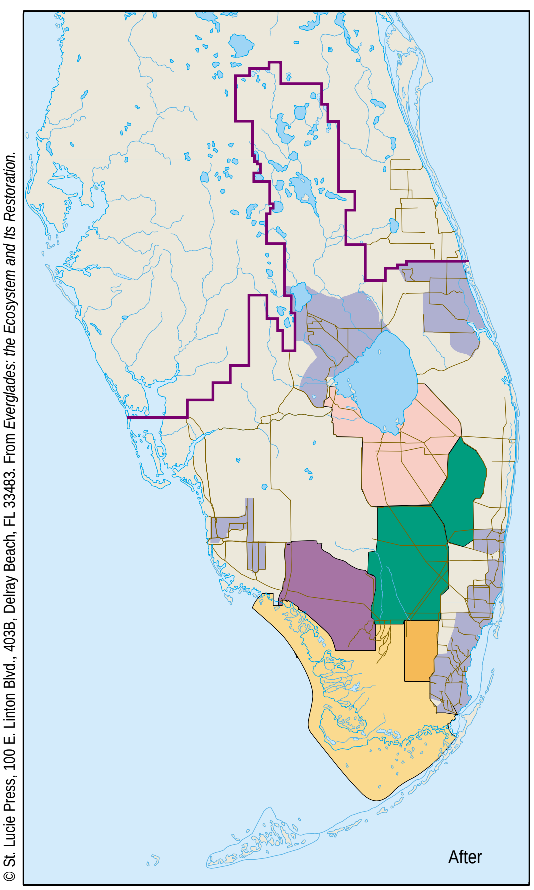


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Fig. 15. The Florida Everglades, before and after development (from Davis and Ogden 1994).

of accumulated organic matter. The local change in topographic relief permits tree islands to flourish, and the frequent and widespread fires during the dry season rapidly reduce accumulated dead organic matter and release much-needed nutrients. Historically, hurricanes have also been important for maintaining the biological diversity of the Everglades, especially of the tree islands and perhaps of the mangroves.

Much of the present controversy and research focus on Everglades National Park, a World Heritage Site, and on adjacent reserves. These protected areas encompass only about 20% of what was originally an ecosystem of more than a million hectares. Half the original area was drained for development and agriculture. Much of what remains has been diked for water conservation areas where water levels and flows are controlled to balance environmental concerns and the rapidly expanding human population in south Florida. The control of water to the Everglades is one example of a change in quantity, quality, and seasonal availability of water that exceeds natural variation, to the detriment of native species (Robertson and Frederick 1994; White 1994).


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Fig. 16. A sea of sawgrass around bayheads (tree islands) in Everglades National Park, Florida.

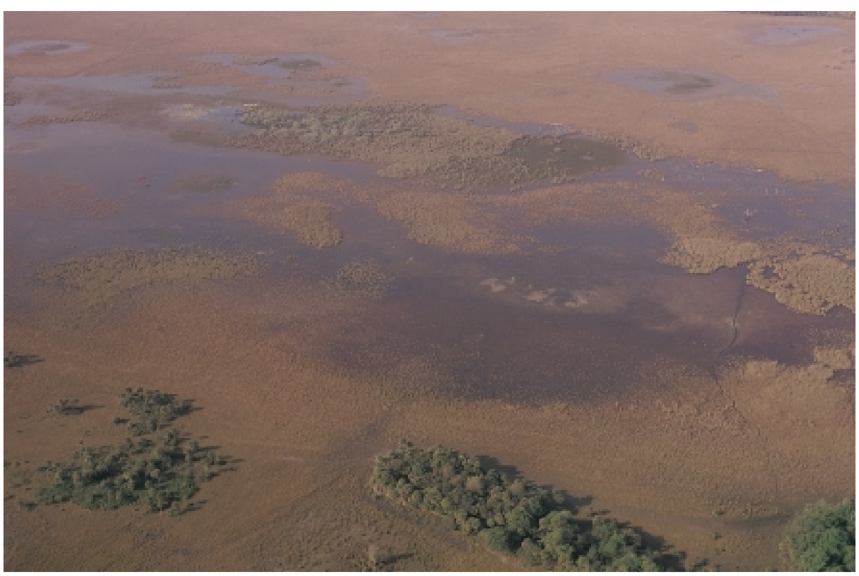
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Fig. 17. Sawgrass glades in Everglades National Park, Florida, under artificially lowered water table conditions in winter.

Research conducted largely since 1970 has revealed that most of the ecosystem is denied sufficient water at the proper time (Fig. 17). This lack of water has permitted the invasion of sawgrass into wet prairies and sloughs. American alligators moved from former wetlands, which are now too dry, to the sloughs, and the estuarine salinity is now too high for mangroves, thereby reducing the productivity of those trees (Lodge 1994). Breeding populations of birds have sharply declined since 1930. The abundance of wood storks decreased by 90%, white ibises by 95%, great egrets by 35%, and small herons by 90% (Fig. 18). Much of the decline of these birds is attributed to poorly timed water releases, which reduce aquatic prey concentrations for nesting birds. Furthermore, hydrological control structures often redistribute the water far from suitable nesting areas (Ogden 1994). Agriculture in the Everglades also began early in this century and intensified with the control of water. Agricultural practices were not compatible with the high water table and wet season and, directly or indirectly, are the major source of nutrient enrichment of water that enters the Everglades (Davis 1994).

Change in water quality promoted the expansion of cattails, which are not suitable aquatic habitat for wading birds. Water concentrations of phosphorus, a limiting nutrient in this aquatic ecosystem, increased considerably in water that enters the park, thereby profoundly changing the primary production and diversity of the aquatic plants and animals on which birds and other animals feed (Frederick and Powell 1994). Today, the remnants of this disrupted ecosystem are not self-sustaining.

In the Everglades, the introductions of nonindigenous biota and their spread in the human-modified environment are becoming a major source of stress on native aquatic species. For example, the introduction of Cuban treefrogs and marine toads in the Miami area eliminated other native amphibians such as the squirrel treefrog, green treefrog, and southern toad. These amphibians have been poorly studied, but because of their seasonal abundance, they are certainly important food sources for wading birds and reptiles (Lodge 1994).

Nonindigenous fishes are the most common introduced species, and their complex effects are exemplified by the nonindigenous blue tilapia and its close relatives. Locally, this species eliminated the growth of naiads, important aquatic plants that provide cover for smaller native fishes and food for larger fishes such as shiners, which, in turn, are important prey for many native waterbirds. The greatest threat that an introduced species poses to the integrity of the Everglades, though, seems to be from the Australian melaleuca. After humans drain marsh and swamp, melaleuca invades the stressed sawgrass marsh and cypress swamps (Fig. 19). The tree is fire-tolerant and transpires large amounts of water, thereby further drying out areas it invades.

The water cycle also links the Everglades to the adjacent estuary of Florida Bay. Altered amounts and timing of freshwater flows to and through the Everglades have sharply increased the salt levels in Florida Bay. The increased salinity has caused reduced reproduction and distribution of the aquatic and terrestrial species once typical of the bay (McIvor et al. 1994).

As shown by these examples of widespread decline in aquatic ecosystem function and biological diversity, the absence of baseline research before a region has experienced major alteration by humans makes understanding the exact causes of ecosystem decline difficult, if not impossible. Without a baseline, recent research during and after human manipulation is hampered and may seem only to document ecosystem decline. Even with the limitation of a lack of baseline research, however, the synthesis of recent studies provides a good understanding of how the ecosystem now functions. By using this research as the basis for modeling the future of ecosystems, some attempts at restoration are feasible.

Small Lakes

The conditions of endangered, commercial, and recreationally important aquatic species generally represent long-term changes in habitat quality (Schindler et al. 1985; Schindler 1987; Rinne and Medina 1988). Comprehensive experimental studies of entire lake ecosystems reveal extensive alteration and loss of macro- and microinvertebrate biota before the effect may be seen in species higher in the food chain, such as fishes (Likens 1985; Schindler et al. 1985; Edmondson 1988; Byron and Goldman 1989). The results from such studies strongly suggest that the detection of the early effects of anthropogenic stress on aquatic ecosystems requires an overall knowledge of species composition that is rarely available. Without such information, the restoration of aquatic ecosystems is difficult because of the substantial time it takes a system to recover from such stresses (Kondolf 1993).

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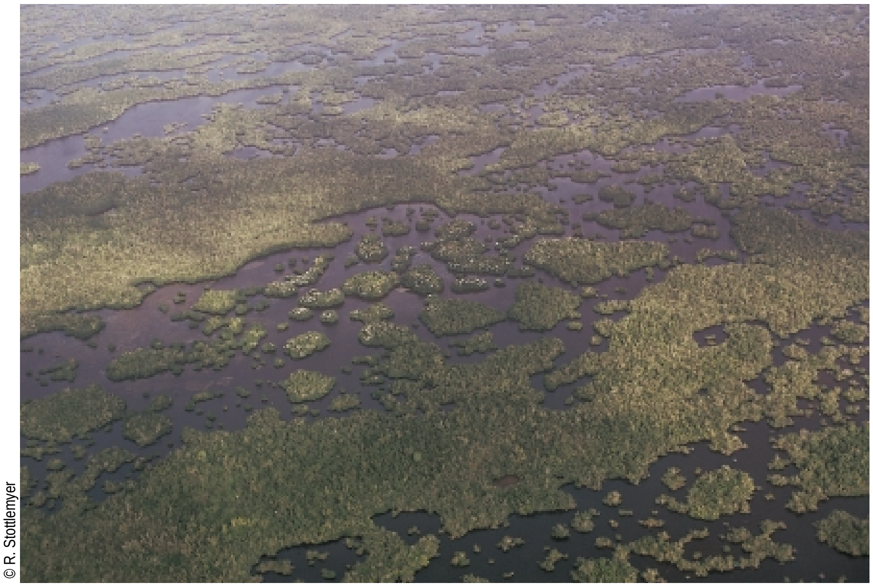

Fig. 18. Egrets on islands and a wilderness waterway in Everglades National Park, Florida. Many breeding bird populations are threatened by human-altered distribution and timing of water in the ecosystem.

Lake Washington in Seattle went through a typical case of eutrophication caused by sewage effluent, experiencing sharp increases in plant biomass and reductions in light transmission. The public took corrective action before the deterioration became serious—in part because ongoing research revealed the early phases of lake eutrophication (Edmondson 1991). This early and unusual corrective action was taken after an unprecedented campaign of public education and the formation of an organization that could issue bonds to finance a cleanup.

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Fig. 19. A baldcypress tree supports a strangler fig in Corkscrew Swamp, Florida. Baldcypress swamps are threatened by the invasion of melaleuca, a tree introduced from Australia.

In sum, aquatic ecosystems, possibly more than any others, have taken the brunt of human activities that are incompatible with the structure and functions of these ecosystems. Humans have derived tremendous short- and occasionally long-term economic benefits from changing aquatic ecosystems but have caused instability, massive losses of integrity that preclude the natural functioning of the systems, and large reductions in species composition. None of these are short-term effects. Restorations have not returned some of these ecosystems to the degree of self-sufficiency and sustainability they possessed before human perturbation.

Conclusions

For most of the last century and at least until the 1950's, the people of the United States largely ignored the true environmental and long-term economic costs of water development and use. Since 1970 a rapid expansion of the human population and per capita consumption of natural resources has revealed limits to past water-use and development practices and to the ecological, societal, and ultimately economic costs of such practices.

The cumulative effects of human activities in aquatic and riparian ecosystems have been dramatic but remain poorly understood. We can link changes in biological diversity to habitat alteration or species loss; the importance of long-term shifts in habitat are also being assessed. There are enough scientifically documented declines of species abundances and extinctions of aquatic species that are direct results of human activity to indicate that present water-use and development practices cannot continue. Although much research is still needed, much better use can be made of existing information in aquatic resource management decisions.

Future Research Needs

The future of water resource management in the United States will be influenced by an increasingly complex set of issues because of expanding human populations and an uncertainty about the United States economy. These forces accelerate the rate of present nonpoint pollution and the problems arising from incompatible land and water uses. We must improve our understanding and ability to deal with these complex problems of land and water interactions, including agricultural runoff, landfill management, urban industrial waste treatment, sludge disposal, and radioactive or hazardous wastes, while balancing strategic needs to retain the wetlands and natural rivers that influence how water is conserved, recycled, and reused (Patrick 1989; Whipple 1989). We must protect our water sources, including groundwater, from contamination and overuse, and commit to maintaining or continuing to restore degraded aquatic systems, riparian forests, and natural rivers.

Continuing emphasis on water quality and pollution control will draw more attention to the importance of instream flow issues, and will increase attention to widespread repercussions of global changes such as the greenhouse effect (Caulfield 1989). Managers of water resources must recognize the benefits of having sufficient pure fresh water for humans and for ecosystems (Fig. 20).

Limited availability of water for withdrawal and use by urban areas will continue to require innovative planning for water use. Traditional expansions of capacity no longer solve the problems of supply and demand for public water. Satisfying uncontrolled future requirements for water is too expensive. Current solutions include increasing the supply from existing storage facilities, managing water demand, reducing loss from distribution systems, providing lower-quality supplies for nondrinking purposes, encouraging reuse, and conservation through pricing (a large increase in costs for exceeding an established level of use) (Rees 1976; Billing and Day 1989). But all competing uses, including the needs of natural ecosystems,

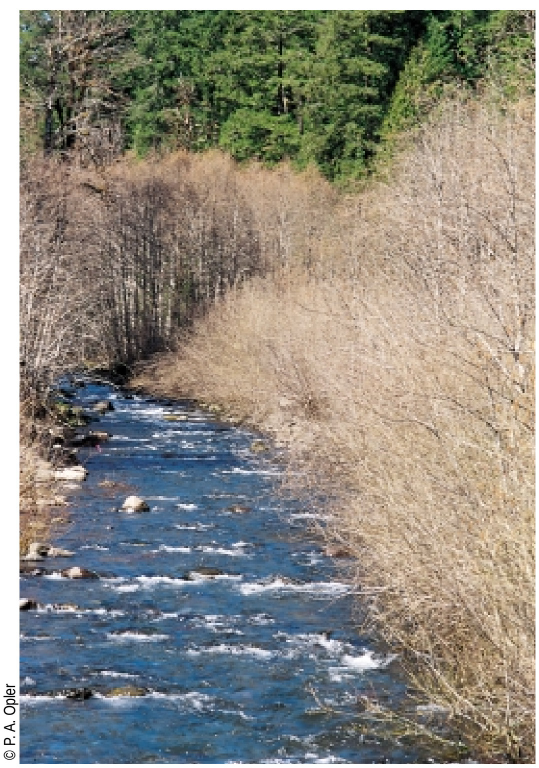
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Fig. 20. Fresh water for humans and for natural ecosystems: an undammed stream in the Pacific Northwest.

must be considered to suitably manage watersheds and river basins.

Research, especially at the ecosystem level, should be a top priority (Likens 1985) so that we can better understand watershed ecosystems. Sustained ecological research is essential for developing good water-use policy and for developing a more comprehensive basis to effectively manage or restore aquatic ecosystems. An understanding of processes over time and of their range of natural variation provides a strong tool for an early assessment of when human activities may force these processes beyond their natural variation. In aquatic ecosystems, we must continue to take a special interest in monitoring changes in species and in community composition.

In addition, we need more replication of long-term watershed studies. Such studies should be initiated primarily in a series of representative systems, not just in systems that are known to be affected or which are only representative of what is defined as sensitive. Too often, past studies that focused on sensitive systems greatly limited broad application of their results.

Most hypothesis testing today is short term, although the importance of long-term monitoring and research to provide a context for developing more meaningful hypotheses cannot be overstated. Paleoecological techniques, when calibrated with ecosystem-level experiments, may help resolve deficiencies of past monitoring. More effort should also be made to use results from current long-term studies for the development and calibration of complex models of aquatic and watershed ecosystems.

Future Management of Watersheds

Human effects on watershed ecosystems are long term. Water-resource management may be limited by past actions, legal or political considerations, conflicting user needs, lack of new water sources, environmental or economic concerns, or global events. For these reasons, we must learn to understand the complex social, ecological, and environmental problems of water management in the United States and acquire more information about the relations of watersheds with climate, the hydrological cycle, chemical processes, and the biota. Demand for existing water supplies continues to increase even though unallocated supplies are rarely available. In the past, humans looked toward engineering for increasing supplies. In many cases, humans may continue to use construction to meet their water needs, but environmental concerns may be leading us away from many large water project choices and toward new models of sustainable uses and watershed management.

To improve the management of watersheds, researchers are striving for a greater understanding of ecological processes in minimally disturbed watersheds and the responses of watersheds to various anthropogenic influences. Management depends on an ability to predict the consequences of human interference. Society's choices are limited by changes—many are known, but many are not yet understood. Decisions about water use must take into account resource alterations that directly and indirectly influence the hydrological cycle and associated lands and biota. Our ability to manage and to sustain water resources is constrained by current limits to our understanding of these cause-and-effect relations.

Water is the most limiting factor to life. For now, the human species has the capacity to alter the amount and quality of water for most of the planet's animals and plants. In the United States and worldwide, however, trends clearly show that our present water-development and use practices cannot continue. We must plan water use as part of a sustainable relation between the environment and society.

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