Mississippi River

The Mississippi River is one of the world's major river systems in size, habitat diversity, and biological productivity. It is the longest and largest river in North America, flowing 3,705 kilometers from its source at Lake Itasca in the Minnesota North Woods, through the midcontinental United States, the Gulf of Mexico Coastal Plain, and its subtropical Louisiana Delta. "Mississippi" is an Ojibwa (Chippewa) Indian word meaning *great river* or *gathering of waters*—an appropriate name because the river basin, or watershed, extends from the Allegheny Mountains in the eastern United States to the Rocky Mountains, including all or parts of 31 states (Fig. 1) and 2 Canadian provinces. The river basin measures 4.76 million square kilometers, covering about 40% of the United States and about one-eighth of North America. Of the world's rivers, the Mississippi ranks third in length, third in watershed area, and seventh in average discharge.

The Mississippi River and its adjacent forests and wetlands provide important habitat for fish and wildlife and include the largest continuous system of wetlands in North America. The river supports a diverse array of wetland, open-water, and floodplain habitats, including extensive habitats on national wildlife refuges. Yet human activities have greatly altered this river ecosystem. Most of the river and its floodplain (defined as the adjacent, generally flat surface that is periodically inundated by floodwaters overflowing the river's natural banks) have been extensively modified for commercial navigation and other human developments. Much of the watershed is intensively cultivated, and many tributaries deliver substantial amounts of sediment, nutrients, and pesticides into the river. Pollutants also enter the river from metropolitan and industrial areas.

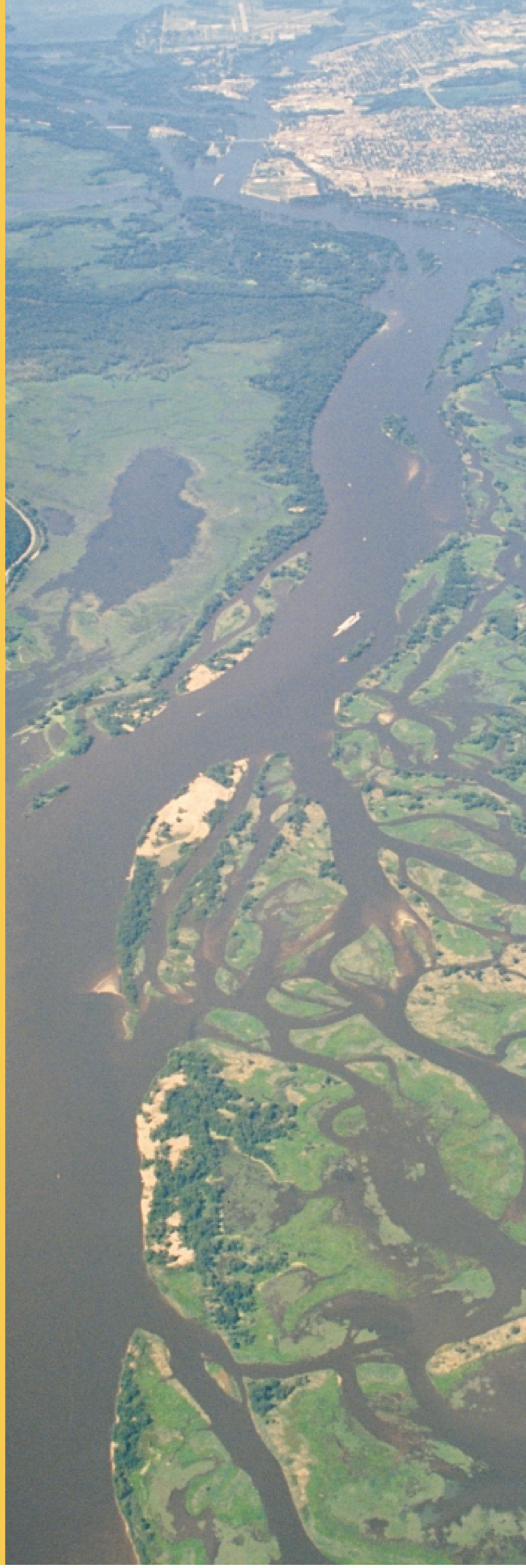
We examine the recent status and temporal trends in the abundance of several key groups of organisms in the Mississippi River. Our analysis shows that certain native flora and fauna, across many trophic levels, have declined along substantial portions of the river. These declines may signal a deterioration in the health of this ecosystem.

Definitions and General Information on River Reaches

The Mississippi River is divided into three segments: the Headwaters, the Upper Mississippi River, and the Lower Mississippi River. The Headwaters is the reach from the source (Lake Itasca) downstream to St. Anthony Falls in Minneapolis, Minnesota, whereas the Upper Mississippi River extends from St. Anthony Falls downstream to the mouth of the Ohio River at Cairo, Illinois. The Lower Mississippi River flows from Cairo to Head-of-Passes in the Gulf of Mexico. The 314-kilometer segment of the Upper Mississippi River extending from the mouth of the Missouri River (near St. Louis, Missouri) to the mouth of the Ohio is often termed the Middle Mississippi River.

Location along the main channel of the river is denoted by *river miles*, starting with mile 0.0 at Head-of-Passes and proceeding 953.8 river miles upstream to the mouth of the Ohio River. Numbering of river miles starts at 0.0 again at the mouth of the Ohio and continues up the Mississippi to Lake Itasca (Fremling et al. 1989).

Commercially, the Mississippi is one of the world's most important and intensively regulated rivers; the term *regulated* applies to rivers that



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are dammed and constrained. The river is navigable by ocean vessels upstream as far as Baton Rouge, Louisiana, and by commercial craft with a 9-foot (2.7-meters) draft as far as Minneapolis. The Headwaters segment is not used for commercial navigation. The U.S. Army Corps of Engineers maintains the commercial navigation channel, federal locks and dams, and federal levees, and the U.S. Coast Guard maintains navigation markers.

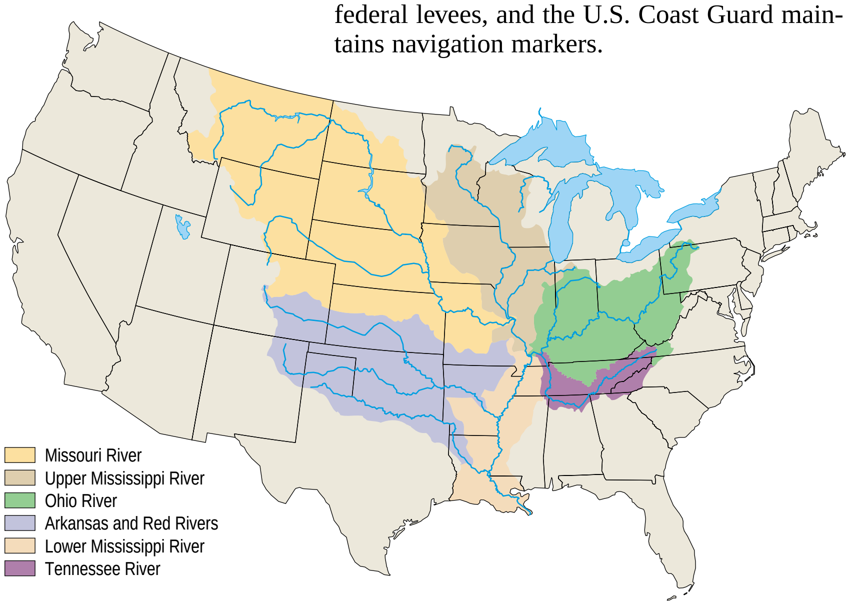

Fig. 1. Drainage basin of the Mississippi River and its major tributaries. The basins for the Headwaters (in Minnesota) and the Upper Mississippi River reaches are combined on the map.

The coastal zone of Louisiana contains about 25% of our nation's wetlands and 41% of our coastal wetlands (see chapter on Coastal Louisiana). These coastal wetlands form one of the world's largest and richest estuaries, essential to the reproduction of marine fishes, oysters, crabs, and shrimp. The coastal wetlands are also critical wintering areas for migratory birds, especially waterfowl. Unfortunately, each year about 100–120 square kilometers of Louisiana's coastal wetlands are lost to open-water or nonwetland habitats because of natural and human causes (Johnston et al. 1995). Especially critical is the erosion of the barrier islands that serve as the first line of defense against hurricanes and storms and which help prevent destruction of freshwater wetlands by saltwater intrusion.

Geography, Geologic History, and Human Development

Headwaters

The Headwaters' reach, located entirely within Minnesota (Fig. 2), has the steepest gradient, exiting Lake Itasca at 440 meters above mean sea level and dropping 204 meters along its 794-kilometer path to St. Anthony Falls, Minnesota (Fremling et al. 1989). This reach passes through spruce swamps, wildrice beds, natural lakes, beds of extinct glacial lakes, artificial impoundments, rapids, and dams (Waters 1977; MacGregor 1995). Beginning as a small, clear stream, the Headwaters gradually are stained a reddish-brown by natural organic acids leached from bog vegetation in the basin. The Mississippi River is a remote stream from Lake Itasca to Bemidji, meandering through corridors flanked by high, pine-topped sand banks, swampy lowlands, alder thickets, wildrice beds, and cattail marshes.

Approaching Bemidji, the wilderness character of the river is gradually lost because of human development. At Bemidji, the river flows through Lakes Irving and Bemidji and then through Stump, Big Wolf, Andrusia, Cass, Winnibigoshish, and Pokegama lakes (Fremling et al. 1989). The outlets of Lakes Winnibigoshish and Pokegama were dammed in 1891 and 1884 as part of a U.S. Army Corps of Engineers navigation and flood-control system that included four other dammed reservoir lakes on Mississippi River tributaries. The Headwaters' dams are now used mainly for flood control, recreation, conservation, and related uses. None of the 11 dams between Lake Itasca and St. Anthony Falls have navigation locks.

In Aitkin County, Minnesota, the river channel meanders for 200 kilometers over a straight-line distance of 64 kilometers. Many meander loops have been cut off, forming crescent-shaped oxbow lakes that are reconnected with the river during high water. Between the cities of Aitkin and Brainerd, the river passes through the Cayuna Iron Range, winding through hilly, heavily forested moraine systems, as well as glacial outwash plains, sand dunes, and swamps. Coniferous forests once covered most of the area, but logging, burning, and farming have produced a mixture of coniferous and hardwood forests. From Brainerd to St. Anthony Falls the landscape is dominated by an extensive system of glacial outwash and alluvial deposits that have been entrenched by the Mississippi River from St. Cloud to Minneapolis. The Headwaters' reach has been described in detail by Waters (1977), Fremling et al. (1989), and MacGregor (1995).

Upper Mississippi River

The Upper Mississippi River flows 1,462 kilometers from St. Anthony Falls in Minneapolis, Minnesota, to the mouth of the Ohio River at Cairo, Illinois (Fig. 2). The major period of valley scouring began about 15,000 years ago when the Wisconsin Glacier began to melt, increasing river flow (Schwartz and Thiel 1963; Wright 1972, 1989; Matsch 1976; Ojakangas and Matsch 1982). About 12,700 years ago, the retreating Wisconsin Glacier blocked the northward drainage routes of its meltwaters toward Hudson Bay, forming glacial Lake Agassiz. This huge lake spilled over its southern rim for about 2,700 years, forming the glacial River Warren and carving the large valley now occupied by the Minnesota River. The River Warren was much larger than the present Minnesota River but carried little sediment. The glacial St. Croix River provided additional sediment-free overflow from Lake Duluth (glacial Lake Superior). The combined flow of the two rivers greatly increased the erosive capacity of the Upper Mississippi River, enabling the river to remove sediments from its bed and to deepen its channel by as much as 90 meters. The Upper Mississippi River must have been spectacular at that time—a massive, torrential river in a gorge that was eventually scoured more than 250 meters deep. As the Wisconsin Glacier retreated into Canada about 9,200 years ago, however, inflows of meltwater to the Upper Mississippi River ceased. The Upper Mississippi River valley then began filling with glacial outwash, mainly sand and gravel, a process that is still under way.

The Mississippi River valley widens considerably where it joins the Minnesota River, 13 kilometers downstream from St. Anthony Falls. Below its junction with the Minnesota River, the Upper Mississippi River flows through a deep valley carved into the surrounding sedimentary rocks. Steep tributary streams have dissected the plateau, creating a complex dendritic drainage pattern and a rugged landscape with high bluffs bordering the river valley. This reach flows through a driftless area (not glaciated during the most recent glacial advance) that includes parts of southeastern Minnesota, west-central Wisconsin, northeastern Iowa, and northwestern Illinois (Hallberg et al. 1984).

Lake Pepin was formed about 9,500 years ago when a delta of sand from the tributary Chippewa River partially blocked the Mississippi River valley, creating a natural impoundment. The Chippewa River, with its steep gradient, is the primary sand source for the reach of the Upper Mississippi River downstream from Lake Pepin (Nielsen et al. 1984). The constant influx of sand from the Chippewa River where it joins the Mississippi River necessitates intensive dredging in order to maintain sufficient channel depth for commercial navigation.

Modern Lake Pepin in Pool 4 begins about 75 kilometers downstream from Minneapolis–St. Paul and extends 35 kilometers downstream. Ranging from 1.5 to 4 kilometers wide, Lake Pepin has a mean depth of about 5 meters and a mean water-retention time of 19 days (Minnesota Pollution Control Agency 1993). The hydrological effect of Lake Pepin has greatly enhanced the quality of the reach of river farther downstream. The lake traps sediment and associated contaminants (Rada et al. 1990; Maurer et al. 1995), greatly reducing the transport of pollutants from the Minneapolis–St. Paul metropolitan area, the Minnesota River basin, and other sources to the riverine ecosystem downstream. Recent sedimentation rates in Lake Pepin range from 3 centimeters per year or greater in upstream reaches to about 0.5 centimeters per year in downstream reaches (McHenry et al. 1980); 21% of the

Fig. 2. Map of the Upper Mississippi River and Headwaters, showing locks and dams, selected cities, and other features. The numbers 1–26 represent the navigation pools in this section of the river. (Note: there is no pool 23.)

lake's volume was lost between 1897 and 1986 (Maurer et al. 1995). The sediment-trapping ability of Lake Pepin substantially reduces contamination of hexagenia mayflies and sediment downstream from toxic substances, such as polychlorinated biphenyls (PCB's; Steingraeber et al. 1994) and cadmium (Beauvais et al. 1995). The lake's sediment-trapping ability, however, will diminish as it fills with sediment and its volume declines.

The reach of river below Lake Pepin is part of the Upper Mississippi River National Wildlife and Fish Refuge, which extends from Wabasha, Minnesota, to near Rock Island, Illinois, a channel distance of almost 500 kilometers (U.S. Fish and Wildlife Service 1987). The refuge is an important wildlife and outdoor recreation area, covering about 80,000 hectares of the river floodplain in Minnesota, Wisconsin, Iowa, and Illinois. The refuge includes a diverse array of habitats, including marshes, backwater sloughs and lakes, myriad flowing channels, floodplain forest, sand beaches, and bluffs.

Just upstream from St. Louis, Missouri, the Missouri River joins the Upper Mississippi River from the west. Most tributaries to the Missouri River flow through highly erodible soils, which means that the Missouri River has always been the principal supplier of sediment to the Mississippi. Construction of a series of large dams in the Missouri River basin in the 1950's and 1960's created deep, cold-water reservoirs that trap sediment, reducing the Missouri River's total contribution of sediment to the Mississippi by about 70% (Keown et al. 1986; Meade et al. 1990).

About 160 kilometers downstream from St. Louis, the Mississippi River flows through Thebes Gap, which resembles the stem of an inverted funnel. Where it exits the gap, the constricted river widens as it enters an ancient sediment-filled lobe of the Gulf of Mexico called the *Mississippi Embayment*. The Mississippi River valley expands to a width of about 80 kilometers where it meets the mouth of the Ohio River.

Human development has greatly altered the Upper Mississippi River and its floodplain. The natural river and its tributaries flowed and meandered freely across the floodplain. Flooding, erosion, and sedimentation were powerful natural processes that shaped and maintained the floodplain and its biotic communities (Sparks 1992). Humans have changed the hydrological regime of the river, altering these processes (Fremling and Claflin 1984; Grubaugh and Anderson 1988; Sparks 1992).

Intensive channelization for navigation began in 1878 (Fremling and Claflin 1984). A series of 29 navigation dams with locks (Fig. 2) was constructed, mostly during the 1930's, to create a 2.7-meters-deep navigation channel between St. Louis and Minneapolis (Fremling et al. 1989). The upstream portions of many navigation pools (the reaches between two consecutive dams; see Fig. 2 for pool locations) are similar to the river before impoundment, whereas the downstream portions resemble shallow reservoirs.

The river floodplain has also been extensively modified with levees to accommodate agriculture and to protect human developments from flooding (Grubaugh and Anderson 1988; Interagency Floodplain Management Review Committee 1994). Agriculture has displaced much of the prairie wetlands and floodplain forests that dominated the presettlement floodplain.

Erosion caused by human activities in the watershed, such as agriculture and construction, has increased the rate of sediment delivery to receiving waters. Runoff has also increased because water storage in the watershed has been reduced by drainage of wetlands, urbanization, and other factors (Interagency Floodplain Management Review Committee 1994). Impoundment of the river for navigation has increased the retention of sediment; the valley floor is now blanketed by sediments from the postsettlement era. More detailed descriptions of the Upper Mississippi River are given by Waters (1977), Jackson et al. (1981), Nielsen et al. (1984), and Fremling et al. (1989).

Lower Mississippi River

Below Cairo, Illinois, the Lower Mississippi River and its tributaries follow the broad gulfward-sloping lowlands of the Lower Mississippi Alluvial Valley, which extends to the Gulf of Mexico. The Lower Mississippi River valley's present surface, the Mississippi Alluvial Plain, is characterized by the meandering, silt-laden Mississippi River and its southerly flowing tributaries, including the Black, Tensas, Yazoo, Big Sunflower, White, and Saint Francis rivers. It is an area of broad, nearly level to gently sloping floodplains and low terraces on unconsolidated alluvial material. Relief is generally less than 15 meters, although terraces and natural levees may rise several meters above the adjacent bottomlands. Swamps and bottomland hardwood forests cover large areas, even though much of the floodplain has been cleared for agriculture. There are many sloughs and oxbow lakes, and streams meander widely. The area also contains Yazoo River-type tributaries, which flow along the base of a natural Mississippi River levee, paralleling the Mississippi River for a considerable distance before joining it. The Tensas, White, Arkansas, and Saint Francis rivers all display Yazoo-type characteristics in their lower reaches (Beccasio et al. 1983).

The Lower Mississippi River channel is wide and generally shallow in the northern part of its alluvial valley, where coarse bed materials delivered by tributaries allow the river to meander. In the southern part of the valley, the sediment delivered to the Mississippi River delta is mainly silt and clay (Autin et al. 1991). This thick upper stratum of fine sediment prevents meandering of the channel, which deepens to a maximum of about 60 meters just upstream from New Orleans, where the channel is less than 1 kilometer wide. At Head-of-Passes, the river splits like the toes of a bird's foot into several outlet channels, called *passes*, that empty into the Gulf of Mexico.

The lower end of the alluvial valley includes the modern Mississippi River delta, a low-lying triangular tract of land formed by sediments deposited at the river mouth (Fig. 3). The modern Mississippi River delta lies within southern Louisiana, extending from the head of the Atchafalaya River south to the Gulf of Mexico. The delta is spatially limited by the Mississippi's outermost distributaries, streams that conduct water away from the river. Through historical time, the Mississippi River has had several distributaries or flood outlets within the delta region. The modern delta is composed of smaller delta complexes formed during the past 8,000 years by a delta-switching process, whereby the river successively abandoned one delta for another as it found shorter, steeper paths to the Gulf of Mexico.

The upper surface of the delta, called the Mississippi Deltaic Plain, includes the coastal wetlands of Louisiana and covers 28,568 square kilometers (Coleman 1988). The deltaic plain is dominated by a complex network of distributary channels and natural levees that radiate outward from the Mississippi River main-stem near Baton Rouge and extend southward into the Gulf of Mexico (Frazier 1967; Penland and Boyd 1985). The deltaic plain includes natural levee ridges rising slightly above surrounding land levels, forested swamps, and coastal marshes vegetated chiefly by sedges and grasses.

Six delta lobes or delta complexes have been identified in coastal Louisiana: the Maringouin, Teche, St. Bernard, Lafourche, Plaquemines (or Balize), and Atchafalaya (Fisk 1944; Frazier 1967; Penland and Boyd 1985; Autin et al. 1991). During the past 5,000–6,000 years, formation of a new delta lobe has begun roughly once every 1,000 years in response to major changes in the Mississippi River's course to the Gulf of Mexico. The formation and aging of delta lobes are accompanied by changes in habitat types and plant communities (Neill and Deegan 1986).

Although the modern Lower Mississippi River has not been dammed, it has been greatly modified (Baker et al. 1991) by being channelized and shortened by 230 kilometers. In addition, its natural floodplain has been reduced about 90% in area by levee construction, which began in 1727. The Lower Mississippi River valley contains about 2,700 kilometers of levees along both sides of the river. Below Baton Rouge, where the levees are most susceptible to damage by river currents and waves, the levee faces are paved with concrete. Throughout Louisiana, the flood levees that normally protect the valley and its cities are augmented by a complex series of diversion projects that divert Mississippi River floodwaters into the Gulf of Mexico via the Atchafalaya River or Lake Pontchartrain, thereby diverting as much as two-thirds of flood flows around Baton Rouge and New Orleans.

Such navigation and flood-control activities have changed the Lower Mississippi River from its natural state (Beckett and Pennington 1986; Baker et al. 1991). Levees have reduced the area of seasonally flooded wetlands along the river,

Fig. 3. Map of the Lower Mississippi River and its major tributaries, showing selected cities and other features.

and dikes and revetments used to entrain the channel prevent the river from creating new habitats. The failure to form new habitats, which historically occurred as the river meandered, is undesirable because floodplain lakes on the Lower Mississippi River (oxbow lakes and former channels) are rapidly filling with sediment (Gagliano and Howard 1984; Cooper and McHenry 1989).

Because of seasonal variations in discharge, aquatic habitats of the Lower Mississippi River are characterized by pronounced temporal variations in surface area, volume, depth, and current velocity (Baker et al. 1991). On an areal basis, the main channel is the primary habitat type during low flow, whereas at medium flow both the main channel and sandbars predominate. During overbank flow conditions, sandbars and inundated floodplains predominate. Few such backwater habitats remain on the Lower Mississippi River, though. Aquatic macrohabitats of the channel environment include the main river channel, secondary channels, sandbars, gyres below bars, tributary mouths, natural banks, and areas associated with dike systems and revetted banks (Cobb and Clark 1981).

Degradation of the Delta and Coastal Wetlands

Louisiana's coastal zone contains 41% of U.S. coastal wetlands and 25% of all U.S. wetlands, making it one of the Earth's largest and richest estuarine areas. Natural and human-induced forces, though, have been converting the state's coastal wetlands to open-water or nonwetland habitats at a rate of over 100–120 square kilometers per year during the past four decades (Johnston et al. 1995).

Recent Physical Changes in the Mississippi River Delta

The loss of coastal wetlands has been accelerated by human causes, including inland movement of saltwater via the intracoastal waterway, interception of alongshore sediment transport by jetties and sea walls, weakening of the barrier island profile by oil and gas and access canals, and pollution. The erosion of Louisiana's barrier islands is of critical concern because these islands help ameliorate the effects of hurricanes and tropical storms and prevent saltwater intrusion from destroying freshwater swamps and marshes (Fremling et al. 1989).

Hydrological Regime

Under natural conditions, the Mississippi River would probably have switched its course to the Gulf of Mexico via the Atchafalaya distributary between 1965 and 1975. The river has been prevented from doing so by artificial levees and control structures. The Atchafalaya previously captured the Red River, and in the past 20 years the Atchafalaya delta complex has emerged and is rapidly filling Atchafalaya Bay (Autin et al. 1991). Natural diversion of Mississippi River flow to the Atchafalaya has been imminent because the Atchafalaya is both steeper than the Mississippi (3:1 ratio in bed slope) and shorter (225 kilometers to the Gulf of Mexico from the Red River entrance versus 480 kilometers for the Mississippi). The Atchafalaya now drains about 30% of the combined flows of the Mississippi and Red rivers to the Gulf of Mexico (Fig. 3).

If the Mississippi's flow switched to the Atchafalaya, the supply of fresh water to the cities of Baton Rouge and New Orleans could be reduced, and river transportation would be curtailed during periods of low flow. Moreover, flood-control and navigation structures could be lost (Lower Mississippi Region Comprehensive Study Coordinating Committee 1974; Keown et al. 1981; Fremling et al. 1989). In addition, increased flows in the Atchafalaya River could be detrimental to the Atchafalaya basin, North America's largest bottomland hardwood swamp.

Delivery and Deposition of Sediment

Human influences on sediment transport in the Mississippi River system have significantly affected the process of delta formation. The sediment load of the Mississippi River has decreased markedly in the last half-century because of sediment storage in reservoirs constructed on the Missouri River in the 1950's and 1960's and because of other human modifications and influences (Keown et al. 1981, 1986; Fremling 1987; Kesel 1989; Meade et al. 1990). About one-fourth of the suspended sediment load of the Mississippi River is diverted to the Atchafalaya River (Keown et al. 1986). In addition, the input of sediment to shallow-water deltas has been curtailed by the closing of distributary channels; for example, the La Fourche River in 1904. Only the delta of the Atchafalaya River is now growing; all other deltas are degrading because of insufficient sediment input (Penland and Boyd 1985). Additional sediment has been lost to the depths of the Gulf of Mexico by directing flow beyond the Continental Shelf via the Lower Mississippi River passes.

Glacial Eustatic Changes and Subsidence

Mean sea levels are rising as much as 1.2 to 4.3 centimeters per year along the Mississippi

Deltaic Plain, apparently increasing the rate of landward migration of the shoreline into the delta complexes (Penland and Boyd 1985). About 20% of the rise in sea level may be attributed to eustatic processes, such as melting of polar ice. The remaining 80% of the sea-level change could be caused by subsidence (localized sinking of the Earth's crust), partly due to extraction of water, oil, and natural gas.

The entire Louisiana delta, with the possible exception of some natural levees, has subsided since it was formed (Autin et al. 1991). Estimated annual rates of subsidence in the deltaic plain range from about 1 millimeter to more than 15 millimeters (Trahan 1986). The average elevation of the Mississippi Deltaic Plain is less than 10 meters at its northern limit and is near sea level in much of the southern part.

Biological Responses

The Lower Mississippi River and Atchafalaya River estuarine environments have been important production areas for marine fishes and invertebrates. Commercial landings in Louisiana, produced largely in these estuaries, recently accounted for 30% of the U.S. harvest (U.S. Department of Commerce 1986).

Many saltwater fishes and crustaceans harvested in Louisiana's commercial and sport fisheries depend on coastal wetlands during part or all of their life cycle (Herke and Rogers 1989). Adults of many such species spawn in the Gulf of Mexico, and their young migrate to the coastal wetlands, which provide nursery habitat for their early life stages. After a few weeks or months, the juveniles or subadults migrate back to the gulf, where the survivors eventually spawn. Examples of important species that annually complete this cycle are brown shrimp, white shrimp, gulf menhaden, and Atlantic croaker. Other species, such as the red drum and spotted seatrout, make similar migrations but may take longer to complete the cycle.

Decreased sedimentation in the delta area is causing the loss of nursery habitats and preventing the creation of new ones. In addition, the recent use of levees and water-control structures in attempts to slow the loss of coastal wetlands, to improve habitat, and to reduce saltwater intrusion has also created migration barriers for many fishes and crustaceans (Herke et al. 1992; Rogers et al. 1994).

Riverine and coastal wetlands in the Mississippi River delta also support large numbers of wintering waterfowl (Chabreck et al. 1989). More than two-thirds of the entire Mississippi Flyway waterfowl population winter in the coastal wetlands of Louisiana (Bellrose 1980). These wetlands support more than half of the continental mottled duck populations and 38% of wintering canvasback populations. The Delacroix marshes east of the Mississippi River and south of the Mississippi River gulf outlet were once considered to be southeastern Louisiana's most productive waterfowl marsh habitats. Their productivity has declined, however, because of land subsidence, hurricane damage, and increased salinity and associated vegetational changes following construction of the Mississippi River gulf outlet (Chabreck et al. 1989).

Status and Trends of Plants and Animals

We examined the recent status and temporal trends in the abundances of several key groups of organisms in the Mississippi River. Our analysis included an ecologically diverse array of organisms representative of many trophic levels in the ecosystem, but it was neither spatially nor taxonomically comprehensive. The extent or abundance of many key native biotic communities and organisms has decreased along substantial reaches of the river in recent years or decades; these communities include floodplain forests, submersed plants, pearlymussels, fingernailclams and other bottom-dwelling invertebrates, certain fishes, migratory waterfowl, colonial waterbirds, songbirds, and mink. Abundances of certain nonindigenous plants and animals have increased recently.

Floodplain Forest

Historical Decline

Floodplain forests in the Upper Mississippi River valley (Fig. 4) are now confined to a riparian zone a few kilometers wide at most. By 1989 the proportion of the Upper Mississippi River valley covered by forest had decreased spatially from upstream to downstream as follows: 18.9% between Minneapolis, Minnesota,

Courtesy Y. Yin, USGS

Fig. 4. Floodplain forest in Navigation Pool 13 (near Bellevue, Iowa) on the Upper Mississippi River.

and Bellevue, Iowa; 13.5% between Bellevue and Alton, Illinois; and 7.3% downstream from Alton. In many reaches, especially downstream from Bettendorf, Iowa, most of the remaining floodplain forest occurs on islands.

Agricultural and urban developments have been leading causes of floodplain forest loss along the Upper Mississippi River (Fig. 5). By 1929 farmland and urban areas covered 22% of the floodplain, and forest had declined to 29% of its former extent (Peck and Smart 1986). In 1989 forests covered 1,233 square kilometers (14.3%) of the Upper Mississippi River valley (Laustrup and Lowenberg 1994). The loss of floodplain forests in the Upper Mississippi River valley, although considerable, has been less than that in many other large North American floodplain rivers, such as the Missouri, the Illinois, the Ohio, and the Lower Mississippi. This lower level of loss is attributed to acquisitions of land for navigation pools and national wildlife and fish refuges, which placed more than 800 square kilometers of the Upper Mississippi River valley into public trust.

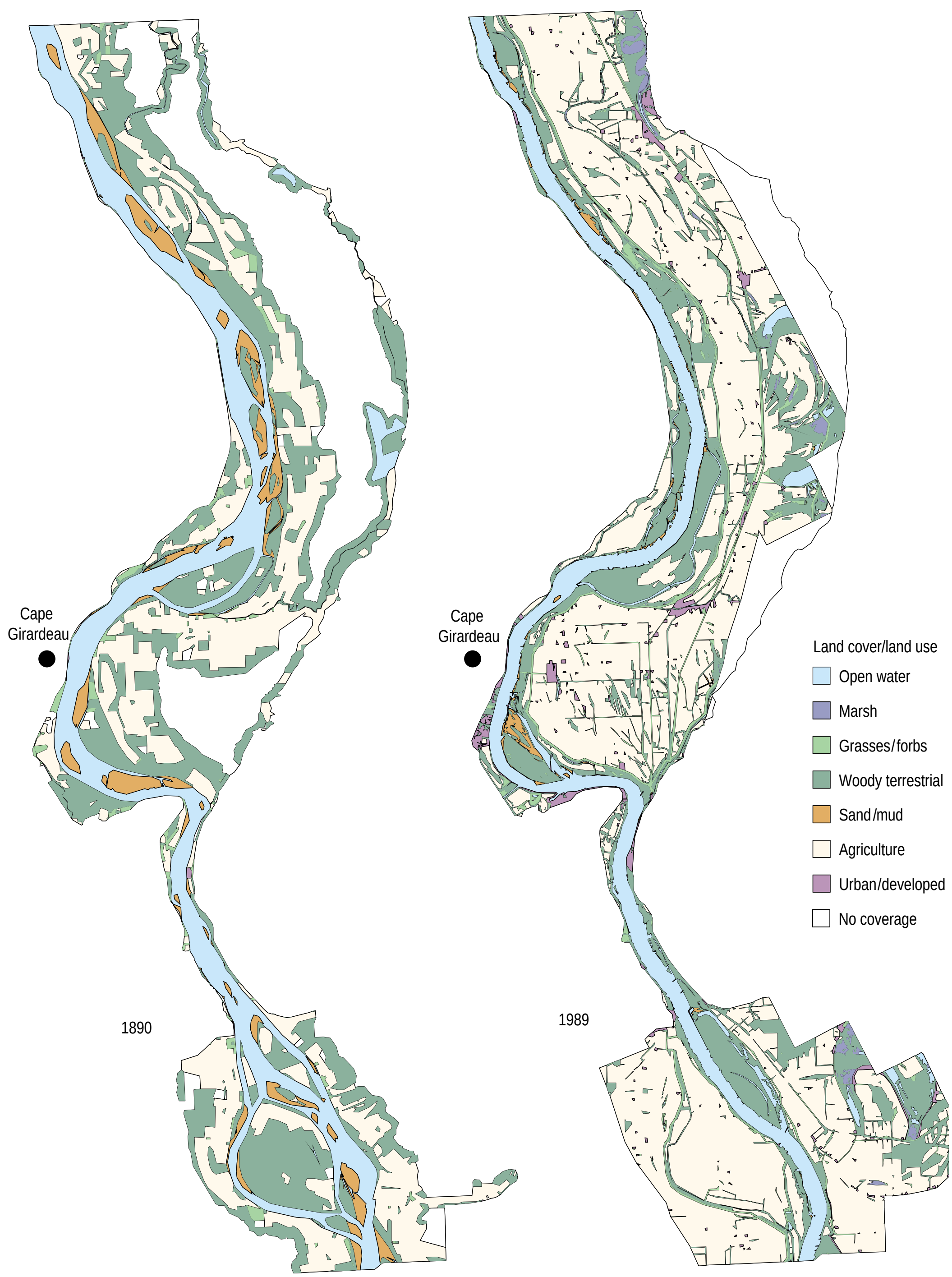

Fig. 5. Comparison of floodplain landscapes in 1890 with 1989 in an 80-kilometer reach of the Mississippi River near Cape Girardeau, Missouri.

Dominant Tree Species

Flooding, erosion, and sedimentation are powerful natural processes that shape floodplain landscapes and affect succession and species composition of floodplain forests (Shelford 1954; Wistendahl 1958; Bedinger 1978; Hupp and Osterkamp 1985). However, these hydrological and geomorphic processes have been constrained for several decades by navigation and flood-protection structures in the Upper Mississippi River.

Individual forest stands in the Upper Mississippi River floodplain can be dominated by any or a few of several species, including (but not limited to) black willow, eastern cottonwood, sycamore, boxelder, silver maple, river birch, green ash, American elm, hackberry, pin oak, bur oak, and swamp white oak. Silver maple is the predominant species in all reaches. In 1993 silver maple was the dominant species in 23% to 49% of the forest stands surveyed at Pools 4, 8, 13, 17, 22, 26, and in an 80-kilometer unimpounded reach about 50 kilometers upstream from the mouth of the Ohio River (Table 1); the mean diameter of trees ranged from 28 to 32 centimeters at breast height, and their mean density varied from 301 to 512 stems per hectare.

The composition of dominant tree species in floodplain forests of the Upper Mississippi River has changed considerably in the last 200 years. American elm declined markedly during the 1900's because of Dutch elm disease. Eastern cottonwood, green ash, and oaks (mainly pin, swamp white, and bur oaks) have become less abundant, compared with silver maple. In the floodplain forest between Minneapolis and St. Paul, for example, George (1924) outlined a successional sequence involving a pioneering cottonwood–willow community, a transitional cottonwood–silver maple community, and a mature community of silver maple, elm, and ash. During early European settlement, the floodplain forests at the tri-state border of Iowa, Minnesota, and Wisconsin were codominated by green ash and silver maple (Moore 1988). Floodplain forests at the confluence of the Mississippi and Illinois rivers, codominated by hackberry, elm, pecan, willows, and eastern cottonwood during early European settlement, are now dominated by silver maple (Nelson et al. 1994). Similarly, floodplain forests along an 80-kilometer unimpounded reach of the Upper Mississippi (starting 21 kilometers upstream from the mouth of the Ohio River) were dominated by eastern cottonwood and sycamore during early settlement times but are now dominated by silver maple and willow (Yin and Nelson 1995). The amount of floodplain forest in pioneering and transitional successional stages has decreased greatly, and much of the present floodplain forest in the Upper Mississippi River valley is mature.

Table 1. Dominant tree species in floodplain forests in the Upper Mississippi River valley at seven river reaches, 1993. Trees were defined as woody stems 10 centimeters or greater in diameter (Y. Yin, U.S. Geological Survey, Environmental Management Technical Center, Onalaska, Wisconsin; and C. E. Korschgen, U.S Geological Survey, Upper Mississippi Science Center, La Crosse, Wisconsin, unpublished data).

	No. 1 dominant		No. 2 dominant				
River reach	Species	Stands dominated (%)	Species	Stands dominated (%)	Mean diameter (centimeters)	Mean density (stems/hectare)	Mean basal area (square meters/hectare)
Pool 4	Silver maple	49	Green ash	16	30	485	41
Pool 8	Silver maple	36	Green ash	15	32	470	45
Pool 13	Silver maple	38	American elm	23	29	483	38
Pool 17	Silver maple	36	American elm	18	28	376	29
Pool 22	Silver maple	35	American elm	19	30	427	39
Pool 26	Silver maple	23	American elm	16	28	512	36
Unimpounded (river miles 30 – 80)	Silver maple	26	Black willow	22	30	301	27

Floods and Forests

Extreme flooding during a single growing season can severely disturb floodplain forests. Such disturbance through flooding is illustrated by the effects of the Flood of 1993, a year when unusually heavy, persistent rainfall caused extreme flooding that lasted from early spring through much of the growing season along a significant portion of the Upper Mississippi River (Parrett et al. 1993; Wahl et al. 1993).

The Flood of 1993 caused substantial tree mortality in the floodplain forests, particularly in the lower reaches of the Upper Mississippi River (Yin et al. 1994), where the flood persisted the longest (Fig. 6). Mortality was positively correlated with flood amplitude and duration, and negatively correlated with tree size. More trees were killed in areas where the flood was most intense and lasted the longest. Smaller

Courtesy Y. Yin, USGS

Fig. 6. A floodplain forest community near St. Louis, Missouri, showing the effects of high tree mortality caused by the Flood of 1993.

trees were most likely to have been killed by flooding. Overall tree death rates ranged from 1% to 4% in Pools 4, 8, and 13, and from 18% to 37% in Pools 17, 22, 26, and in the open river. For saplings, overall mortality rates were higher, ranging from 2% to 9% in Pools 4, 8, and 13 and from 48% to 80% in Pools 17, 22, 26, and in the open river reach between St. Louis, Missouri, and Cairo, Illinois (Yin et al. 1994).

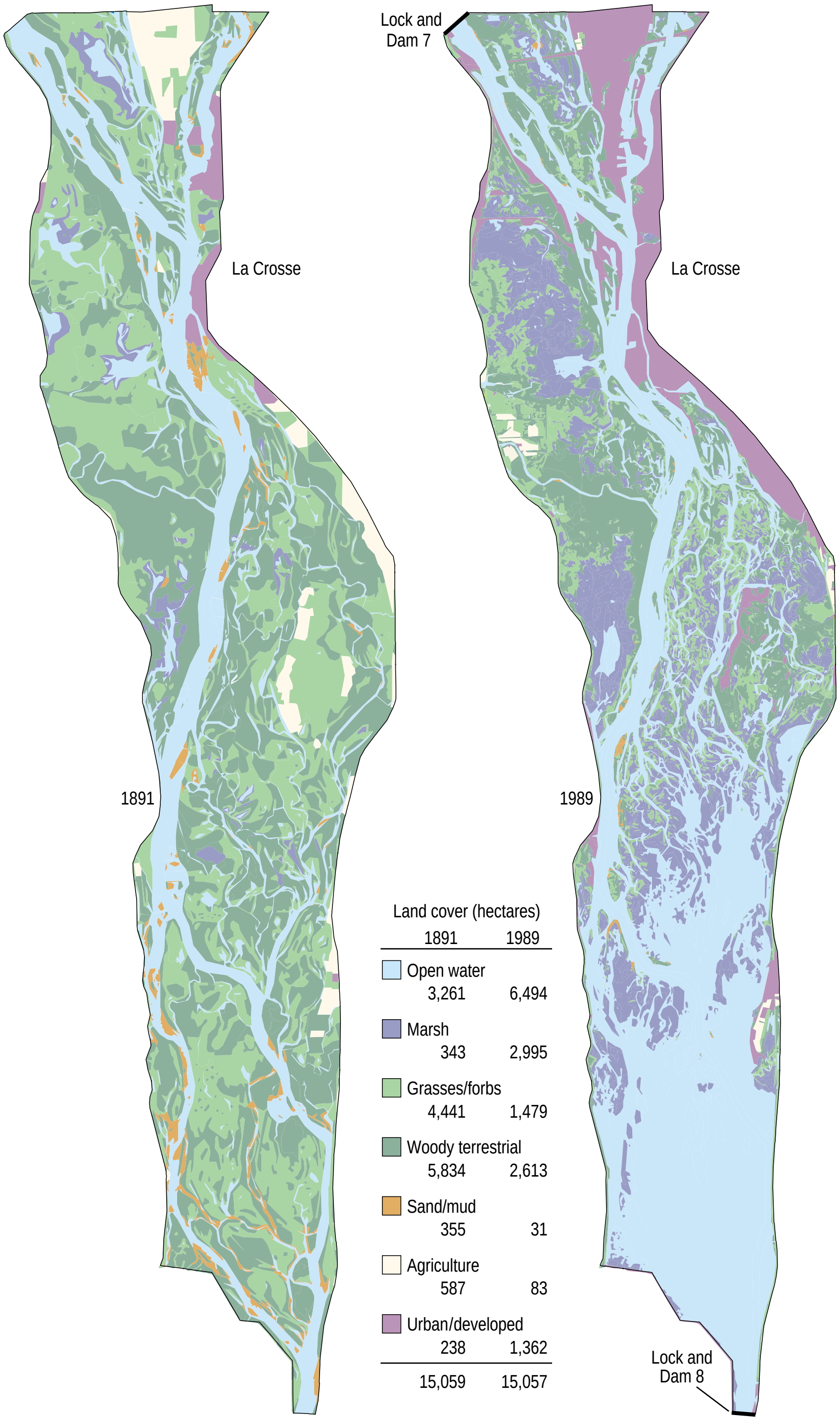

Fig. 7. Changes in land cover over a 98-year period in Pool 8 of the Upper Mississippi River (near La Crosse, Wisconsin). Numbers in the two columns are not equal because of rounding.

The mortality of trees and saplings also varied greatly among species (Yin et al. 1994). The least flood-tolerant trees were hackberry, Kentucky coffeetree, sugarberry, river birch, and white mulberry. Pin oak, silver maple, American elm, and slippery elm were moderately tolerant, and sycamore, hawthorne, green ash, black willow, swamp white oak, and eastern cottonwood were most tolerant. The effects of the Flood of 1993 on floodplain forests along the Upper Mississippi River are expected to persist for decades (Yin et al. 1994).

Aquatic Vegetation

Effects of Impoundment

Emergent and submersed aquatic plants were present but not abundant in the Upper Mississippi River before the locks and dams contructed during the 1930's flooded thousands of hectares of former agricultural areas, lowland hardwood forests, and shallow marshes (Fig. 7). The creation of navigation pools abruptly altered the hydrology of the river; similarly, the diversity, abundance, and distribution of aquatic plant species changed markedly in the decades after impoundment (Peck and Smart 1986). The downstream reaches of the newly created pools provided stable habitat for aquatic plant species. In midpool regions, conditions after impoundment were also favorable to marsh vegetation (Olson and Meyer 1976). Upstream reaches, in contrast, remained similar to their preimpoundment conditions.

Extensive, dense beds of water smartweed developed in the year after impoundment and remained productive for about 5 years (Green 1947). Thereafter, remnant water smartweed was sterile and reproduced only vegetatively. Eventually, water smartweed was replaced by various species of pondweeds, mostly long-leaf pondweed and sago pondweed.

The abundance of submersed plants changed notably after drawdowns of water in several pools during the winters of the early 1940's. Pool 8, for example, was drained from 1 January to 15 February 1944, and from 10 January to 15 March 1945 (U.S. Army Corps of Engineers, St. Paul District, St. Paul, Minnesota, unpublished records). Although Congress ended this practice by the passage of an Anti-Drawdown Law in 1948, the lower water levels apparently stimulated the germination of seeds (Green et al. 1964). The most common submersed plants to become established during this period were long-leaf pondweed, sago pondweed, narrow-leaf pondweed, flat-stem pondweed, curly leaf pondweed, coontail, elodea, water star-grass, and wildcelery. Of these, long-leaf pondweed was most abundant and most widely distributed, occurring in habitats ranging from shallow water to deep, flowing channels (Green 1947).

Wildcelery, which produces a vegetative tuber important as food for certain migratory waterfowl, became the dominant submersed plant around 1960 in much of the 675-kilometer reach of the river between Pools 4 and 19. Minor et al. (1977), who delineated vegetative cover types for Pools 1–12 in the 1970’s, found that the aquatic cover class containing wildcelery occupied the greatest area. No stands of water smartweed were identified by Minor et al. (1977), indicating a marked change in species composition since the 1940’s. In lower Pool 8, wildcelery contributed nearly 50% of the relative biomass of submersed plant species in 1975 (Sefton 1976); most of the remaining 50% of biomass was collectively contributed by coontail, long-leaf pondweed, water star-grass, sago pondweed, and elodea.

Until the late 1980’s, a submersed plant community dominated by wildcelery covered half of Lake Onalaska in Pool 7. Wildcelery covered more than 650 hectares of lower Pool 8 by 1975, and more than 1,275 hectares of Lake Onalaska (Pool 7) by 1982 (Fig. 8). The wildcelery beds were maintained by production of overwintering buds that emerged each spring. By early summer, wildcelery beds were well established and so dense that they significantly affected the hydrology and water quality of the lake. The perimeters of the beds functioned as a sediment screen, making the water inside the beds normally quite clear. Submersed plants grew in all areas of the lake where water was less than 2 meters deep (C. E. Korschgen, U.S. Geological Survey, Upper Mississippi Science Center, La Crosse, Wisconsin, unpublished data). Several other submersed plants were common in these beds, including water star-grass, sago pondweed, Richardson pondweed, narrow-leaf pondweed, flat-stem pondweed, curly pondweed, and Eurasian watermilfoil.

Recent Declines and Status

The abundance of many submersed plants, including wildcelery, declined markedly in much of the Upper Mississippi River in the late 1980’s. Information from surveys supported by LandSat photography of Pools 5, 7, 8, 9, 11, and 19 shows that the abundance of wildcelery and other submersed aquatic plants declined greatly between 1987 and 1989 (Fig. 9) and continued to decline through 1994 (Fig. 10) (R. V. Anderson, Western Illinois University, Macomb, Illinois, personal communication; Korschgen, unpublished data; J. Lyons, U.S. Fish and Wildlife Service, McGregor, Iowa, personal communication; E. Nelson, U.S. Fish and Wildlife Service, Winona, Minnesota, unpublished data; W. Thrune, U.S. Fish and Wildlife Service, La Crosse, Wisconsin, personal communication). This decline coincided with the severe midwestern drought of 1987–1989, which affected water quality in the Upper Mississippi River.

In Lake Onalaska (Pool 7), the abundance of wildcelery changed little during 1980–1984 but declined greatly after the extremely dry, hot summer of 1988 (Fig. 10). More than 1,200 hectares of submersed vegetation, mainly wildcelery, disappeared in Lake Onalaska during 1988 and 1989 after the plants failed to produce winter buds during the late summer and fall of 1988 (Korschgen, unpublished data).

Nearly 500 hectares of submersed vegetation disappeared in the lower half of Pool 19, where plant beds dominated by wildcelery, water star-grass, sago pondweed, and coontail had generally been expanding since the 1960’s (Anderson, personal communication). In early September 1990, small patches of Eurasian watermilfoil were the only submersed vegetation found in the lower half of Pool 19 (S. Rogers, U.S. Geological Survey, Onalaska, Wisconsin, unpublished data).

Fischer and Claflin (1995) compared the biomass and occurrence of aquatic plants in Pool 8 in 1975 with 1991. Overall biomass and frequency of occurrence of aquatic plants were lower in 1991 than in 1975. According to their study, declines in both frequency of occurrence and biomass were greater for submersed plants than for emergent plants. They discovered declines within several habitats and species, notably wildcelery, in the shallow open-water areas. Many other submersed aquatic plants also declined.

Much of the area formerly occupied by wildcelery remains unvegetated, although Eurasian watermilfoil, a nuisance nonindigenous species, now occupies some of the shallower sites. The abundance of Eurasian watermilfoil has seemingly increased since the mid 1980’s. In Pools 8 and 13, monotypic beds of Eurasian watermilfoil have been found near areas where wildcelery had occurred. In Pools 4–8, 13, and 26, Eurasian watermilfoil is occasionally found near or with other submersed plants, including sago pondweed, wildcelery, and coontail (U.S. Geological Survey, Long Term Resource Monitoring Program, Onalaska, Wisconsin, unpublished data; Rogers, personal observation).

The Flood of 1993 also affected the river’s submersed aquatic plant communities. During the 1993 growing season, most species of submersed plants decreased in frequency of occurrence at monitoring sites in Pools 4, 8, 13, and 26 (Spink and Rogers 1997). The decreases

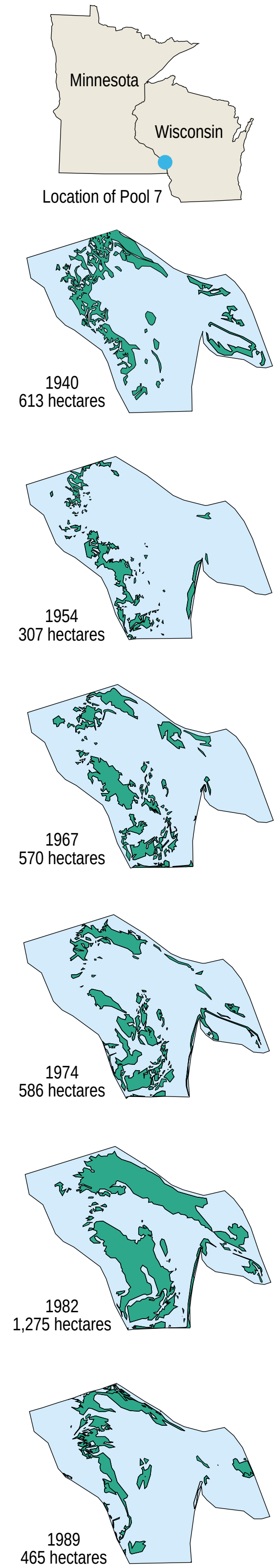

Fig. 8. Spatial distribution of submersed aquatic vegetation (dominated by wildcelery) in Pool 7 (Lake Onalaska) of the Upper Mississippi River in 1940, 1954, 1967, 1974, 1982, and 1989.

were greatest in Pools 13 and 26, which had more severe flooding than Pools 4 and 8. In 1994 submersed aquatic plants had recovered to preflood frequencies in Pools 8 and 13, but not in Pool 26, where the duration and magnitude of the flood were greatest. Interestingly, the distribution and abundance of wildcelery in Pools 8 and 13 were greater after the flood year than before the flood (U.S. Geological Survey, Long Term Resource Monitoring Program, unpublished data).

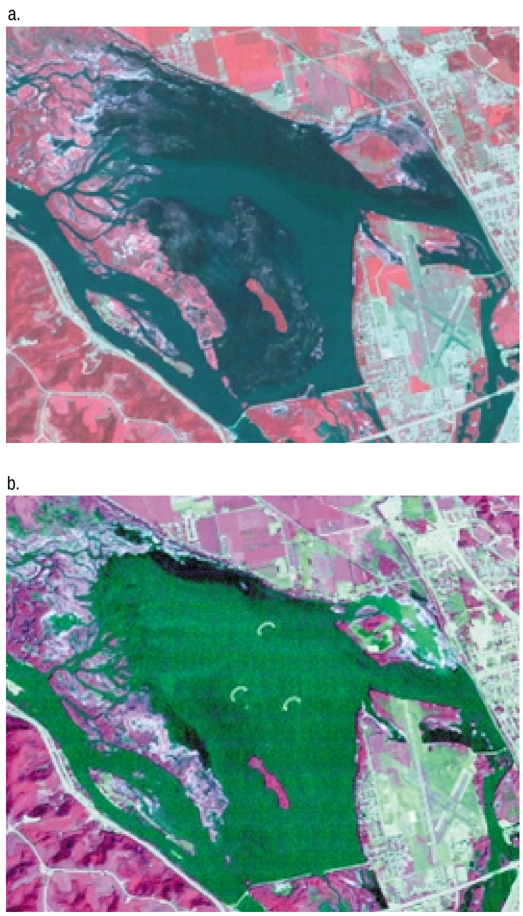

Fig. 9. LandSat image of Pool 7 (Lake Onalaska) of the Upper Mississippi River, showing abundant and widely distributed aquatic plant beds in a) 1987, and greatly reduced abundance and distribution of plant beds in b) 1989. Beds of submersed vegetation appear as dark green areas within the lake.

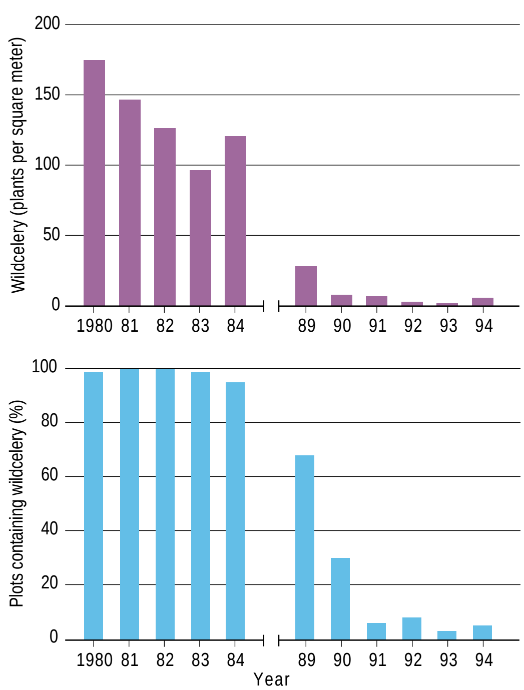

Fig. 10. Abundance of wildcelery, a submersed aquatic plant, in Lake Onalaska (Pool 7) of the Upper Mississippi River, 1980–1994.

Potential Causes of Declines in Aquatic Vegetation

As stated previously, the recent decline in submersed plants in the Upper Mississippi River coincided with the severe drought of 1987–1989. Although information on drought-related conditions in the river is limited, a number of potential causes have been identified.

Blooms of planktonic or attached algae during the drought, particularly in the summer of 1988, may have severely limited the depth to which sufficient light—the source of energy for photosynthesis—penetrated the water column to support the growth of rooted aquatic plants. High concentrations of nutrients in water (retained in backwaters because of extremely low flows) and abnormally high solar radiation during the drought may have stimulated the production of epiphytes or planktonic algae, thereby reducing light penetration in the water column (J. Lennartson, U.S. Fish and Wildlife Service, Winona, Minnesota, and J. Wetzel, Wisconsin Department of Natural Resources, La Crosse; Korschgen, memorandum dated 3 October 1989). Concentrations of orthophosphorus at several main-channel sites were high during the summer of 1988, possibly contributing to the prolific bloom of the blue-green alga *Aphanizomenon*; the bloom extended from Lake Pepin (Pool 4) to Pool 11 (J. Sullivan, Wisconsin Department of Natural Resources, La Crosse, personal communication; see Fig. 2 for location).

This potential mechanism of submersed plant decline is similar to the model devised by Phillips et al. (1978), which illustrates the relationship between nutrient loading, increasing algal growth, and declines of rooted aquatic plants. In some systems, biologically produced turbidity caused by nutrient enrichment has led to the disappearance of rooted plants (Phillips et al. 1978; Hough et al. 1989). Clearly, the reestablishment and recovery of aquatic vegetation have been hindered by limited light availability in the turbid backwaters (Kimber et al. 1995; Owens and Crumpton 1995).

Conversely, there is evidence that submersed aquatic plants may benefit from conditions caused by moderate drought. During summer 1985, for example, water clarity markedly increased in Pool 8 in apparent response to reduced runoff caused by a summer drought, and the mean depth of the light zone during that growing season increased to 1.3 meters. That summer, the distribution of submersed plants, including wildcelery and Eurasian watermilfoil, increased in Pool 8 in apparent response to the increased availability of light (Korschgen, unpublished data). Similar increases in submersed aquatic plants occurred in 1977 in Pool 19, coincident with a period of increased water clarity, low flow, and stable water levels during spring and summer (Sparks 1980; Steffeck et al. 1985).

Rogers et al. (1995) hypothesized that the availability of sediment nutrients may have been reduced by low flows during the drought. Input of sediments may provide nutrients important to the maintenance of submersed plant beds (Barko et al. 1991). The possible depletion of sediment nutrients, particularly nitrogen, during the low flows of 1987, 1988, and 1989, in combination with above-normal water temperatures, may have reduced plant growth and reproduction in some areas of the river.

The reestablishment of submersed aquatic plants in the river may be inhibited by grazing fish, particularly common carp, whose abundance has increased significantly in Pools 8 and 13 since 1991 (S. Gutreuter, U.S. Geological Survey, Onalaska, Wisconsin, unpublished data). Common carp often forage in beds of submersed plants (Lubinski et al. 1986), where they resuspend bottom sediments, increase turbidity, and uproot some submersed plants, particularly species with shallow root systems. Peitzmeier-Romano et al. (1992) found that fish grazing on unprotected wildcelery plants reduced leaf growth and affected tuber production in backwaters of the Illinois River. Grazing apparently foiled attempts to grow wildcelery in a backwater lake of the Upper Mississippi River, where much of the wildcelery planted into unprotected, suspended buckets was damaged by grazers (Korschgen, unpublished data).

Purple Loosestrife

Purple loosestrife, a nonindigenous wetland plant introduced to North America from Europe in the early 1800's, was probably introduced into the Upper Mississippi River basin in the early 1900's (Thompson et al. 1987). This perennial plant forms dense monotypic stands in wetlands, replacing many native wetland plants (Thompson et al. 1987; Edsall et al. 1995). Purple loosestrife has no food value for wildlife, and its replacement of native emergent plants such as cattail makes wetlands less suitable as wildlife habitat (Thompson et al. 1987; Malecki 1995).

By 1985 purple loosestrife had become established throughout much of the Upper Mississippi River basin (Thompson et al. 1987). In the early 1980's, this nuisance plant had become notably abundant on the Upper Mississippi River National Wildlife and Fish Refuge, and it had infested wetlands of Pools 4 through 14 by the late 1980's (Nelson, personal communication).

Refuge managers have been combating this nonindigenous plant since the mid 1980's with hand-weeding and herbicide application on localized stands. These traditional control methods have met with little success, probably because the plant's seed reservoir is so extensive (Malecki 1995). Biological-control methods are now being attempted through the release of natural enemies such as root-boring and leaf-eating insects (Skinner et al. 1994). Now that we recognize that eradication of purple loosestrife is no longer feasible, our goal is to achieve modest control of the plant (Malecki 1995).

Bottom-Dwelling Invertebrates

Bottom-dwelling invertebrate organisms, collectively termed *benthic invertebrates*, are ecologically important in riverine ecosystems. The Upper Mississippi River has supported a diversity of benthic invertebrates. In 1977, for example, Trapp (1979) found 54 taxonomic groups of benthic invertebrates in Lake Pepin (Pool 4), where flies and segmented worms made up 96% of the total. Elstad (1986), who studied Pool 7 (Lake Onalaska) and Pool 8 in 1976–1977, found at least 144 taxa of benthic invertebrates; eight major groups (segmented worms, leeches, isopods, amphipods, moths, flies, snails, and bivalve mollusks) accounted for more than 90% of the total number sampled.

In 1992–1994, benthic invertebrates were sampled in Pools 4, 8, 13, and 26, and in an open-river reach near Cape Girardeau, Missouri. Initial data from this sampling are available for hexagenia mayflies and fingernailclams, both of which inhabit soft sediments and are important in the diet of certain fish and waterfowl (Carlander et al. 1967; Jude 1973; Thompson 1973). The 3-year period encompassed by these monitoring data is too short to assess temporal trends; however, the abundances of fingernailclams and hexagenia mayflies varied considerably among the five reaches sampled (Table 2). Densities of both organisms, but particularly fingernailclams, were consistently highest in Pool 13 and lowest in Pool 26 and in the open reach of river. Densities of both mayflies and fingernailclams also varied among habitat types; areas classified as contiguous backwater, impounded, and tributary delta lake had much higher mean densities than main-channel border and side-channel habitats (Table 3).

In the Lower Mississippi River, the distribution of benthic invertebrates is strongly influenced by current velocity and substrate composition, physical variables that vary spatially and seasonally in many habitat types (Beckett et al. 1983; Beckett and Pennington 1986). The shifting, coarse sand and gravel substrates of main and secondary channel habitats in the Lower Mississippi River support few large invertebrates, which are termed *macroinvertebrates* (Wright 1982; Beckett et al. 1983; Beckett and Pennington 1986). Sand substrate in channel habitat, however, contains a recently discovered, unique assemblage of very small invertebrates consisting of midges, worms, and microturbellarians (Baker et al. 1991). Natural clay banks of the Lower Mississippi River contain abundant numbers of burrowing mayflies (*Tortopus incertus* and *Pentagenia vittigera*) and hydropsychid caddisflies (Beckett et al. 1983; Beckett and Pennington 1986). Abandoned channel habitats, characterized by slack currents and silty substrates, support high densities of invertebrates, including phantom midges, segmented worms, and fingernailclams (Mathis et al. 1981; Beckett et al. 1983; Beckett and Pennington 1986).

Table 2. Mean densities of hexagenia mayflies and fingernailclams, weighted by strata (aquatic area), at reaches being monitored by the U.S. Geological Survey's Long Term Resource Monitoring Program. The open river reach is near Cape Girardeau, Missouri. ND = no data available.

Organism	Density of organism (number/square meter)				
Year	Pool 4	Pool 8	Pool 13	Pool 26	Open river
Mayfly					
1992	57	85	124	29	21
1993	125	111	154	10	ND
1994	199	86	208	21	20
Fingernailclam					
1992	47	22	90	13	4
1993	75	22	2,463	1	ND
1994	85	13	583	4	0

Hard substrates provided by revetments, stone dikes, and articulated concrete mattresses (interconnected concrete blocks used to control bank erosion) support significant numbers of invertebrates in the Lower Mississippi River (Mathis et al. 1982; Beckett and Pennington 1986; Way et al. 1995). The invertebrate communities inhabiting hard substrates are commonly dominated numerically by a few species that are tolerant of moderate to high current velocity (Mathis et al. 1982; Beckett and Pennington 1986; Way et al. 1995). Way et al. (1995), for example, found that three species—a caddisfly, a midge, and an amphipod—represented more than 98% of the invertebrates on articulated concrete mattresses after 3 months of colonization.

Table 3. Densities of hexagenia mayflies and fingernailclams (numbers per square meter) in five habitats on the Upper Mississippi River, sampled as part of the U.S. Geological Survey's Long Term Resource Monitoring Program in Pools 4, 8, 13, and 26, and in the open river near Cape Girardeau, Missouri.

Habitat	Mayfly		Fingernailclam	
	Mean density	Standard error	Mean density	Standard error
Contiguous backwater	105	9	42	5
Impounded	127	11	975	181
Tributary delta lake	158	27	93	9
Main-channel border	16	3	18	5
Side channel	46	6	28	5

Rocky substrates associated with dike structures on the Lower Mississippi River support higher total densities of aquatic invertebrates than abandoned channels, natural river banks, dike fields, temporary secondary channels, sandbars, revetted banks, main channel, and permanent secondary channels (habitats listed in order of decreasing invertebrate density; Mathis et al. 1981, 1982; Wright 1982). Aquatic insects were the most abundant invertebrates in rocky substrates of dikes, accounting for more than 97% of the organisms sampled; the net-spinning caddisfly represented 60% of the total number of invertebrates sampled, two species of tube-building midges represented 24%, another species of net-spinning caddisfly 8.4%, and an isopod 2% (Mathis et al. 1982). Dike fields also contain soft mud substrates that support significant densities of burrowing hexagenia mayflies (Beckett and Pennington 1986).

Recovery from Pollution Caused by Sewage

For decades, benthic invertebrates were absent or scarce in reaches where water quality was degraded by pollution caused by sewage. The 100-kilometer reach of river downstream from the Minneapolis–St. Paul metropolitan area, for example, suffered severe oxygen depletion caused by sewage discharged into the river (Wiebe 1927; Fremling 1964, 1989), and pollution-sensitive organisms, such as burrowing hexagenia mayflies, were absent or scarce in Pools 2, 3, and 4. Burrowing mayflies began recolonizing riverine reaches downstream from the metropolitan area in the early 1980's, when dissolved-oxygen concentrations increased in response to improved wastewater treatment (Johnson and Aasen 1989; Fremling and Johnson 1990).

Recent Declines

Populations of fingernailclams have declined in certain reaches during recent decades. Wilson et al. (1995) examined trends in densities of the fingernailclam from 1973 to 1992 by compiling existing data and by sampling. Significant declines of fingernailclams were evident in five of eight pools examined (declines in Pools 2, 5, 7, 9, and 19) along a 700-kilometer reach of river from Hastings, Minnesota, to Keokuk, Iowa. Densities in Pool 19, which had the longest historical record on fingernailclams, averaged 30,000 per square meter in 1985 and decreased to zero in 1990 (Fig. 11). The declines of fingernailclams occurred chiefly during low-flow periods associated with drought (Sparks 1980; Wilson et al. 1995).

Fingernailclam population declines do not seem to be directly linked to the periodic depletion of dissolved oxygen that occurs in backwater areas. Although fingernailclams are much more tolerant of low dissolved oxygen concentrations than are burrowing hexagenia mayflies, they have not readily recolonized the reaches recolonized by hexagenia mayflies (Wilson et al. 1995). The observed declines of fingernailclams, as well as their subsequent slow rate of recolonization, were seemingly caused by the uninhabitability of bottom sediments—perhaps due to the presence of one or more toxic substances. Fingernailclams are sensitive to many toxicants, including un-ionized ammonia (Sparks 1984; Arthur et al. 1987).

Recent studies by the U.S. Geological Survey have shown that surficial sediments add considerable amounts of nitrogen to the reach of the Upper Mississippi where populations of fingernailclams have declined. The production of ammonia by microbial decomposition in the sediments would presumably be increased by the conditions of high temperature and nutrient enrichment associated with low-flow, drought periods. High microbial activity (decomposition), stimulated by high temperature and an abundant supply of organic matter, would greatly increase the concentration of toxic ammonia in the sediments (Frazier et al. 1996), possibly causing episodic toxicity in fine-grained sediments during periods of drought and low flow.

Brewer et al. (1995) compared the abundance, standing crop (biomass per unit area), and community composition of benthic invertebrates in Pool 8 of the Upper Mississippi River near La Crosse, Wisconsin, between midsummer 1975 and midsummer 1990. The taxonomic composition of the communities changed in all four habitats studied; similarity of species composition between 1975 and 1990 was 19% in open-water habitat, 44% in bays, 50% in side channels, and 62% in marshes. Between 1975 and 1990, overall standing crop of the benthos decreased significantly in all four habitat types, and overall abundance decreased in all habitats except bays.

In the Lower Mississippi River, the distribution and abundance of benthic invertebrates have been examined in relation to habitat type and its characteristics. We found no information on temporal trends of invertebrates in the Lower Mississippi River.

Native Pearlymussels

The Upper Mississippi River is one of a few large rivers that still has a substantial pearlymussel fauna. The abundance and species richness of pearlymussels in the Upper Mississippi exceed that of many other midsize to large North American rivers (Miller et al. 1987, 1993), although the species richness of native pearlymussels in the Mississippi River has clearly declined since the early 1900's. Thiel (1992) estimated that the number of pearlymussel species had decreased 30% in Pools 4–6 and 26% in Pools 8–10 since the 1920's. Trend analysis of historical data from several mussel surveys (Wiener et al. 1995) showed that overall, the pearlymussel fauna in the Upper Mississippi River drainage has declined from about 50 to 60 species in the early 1920's to about 30 species in the mid-1980's (Fig. 12).

The decline of pearlymussel species richness in the Upper Mississippi River mirrors a broader continental pattern. Almost half of the 292 pearlymussel species in North America are either extinct or at serious risk of extinction (Bogan 1993; Neves 1993; Williams et al. 1993). Several factors are suspected of contributing to declines in North America's native freshwater mussel fauna, including habitat modification and degradation, pollution, overharvest, and commercial and recreational navigation (Williams et al. 1993).

From 1982 to 1986, a notable die-off of pearlymussels occurred in Pools 8 through 25, a 720-kilometer reach of the Upper Mississippi (Blodgett and Sparks 1987; Thiel 1987). Analyses of healthy, moribund, and dead mussels for parasites, disease, and selected contaminants failed to reveal the cause of the die-offs (Thiel 1987). The status of the remaining pearlymussel species varies widely, ranging from abundant to endangered (Wisconsin

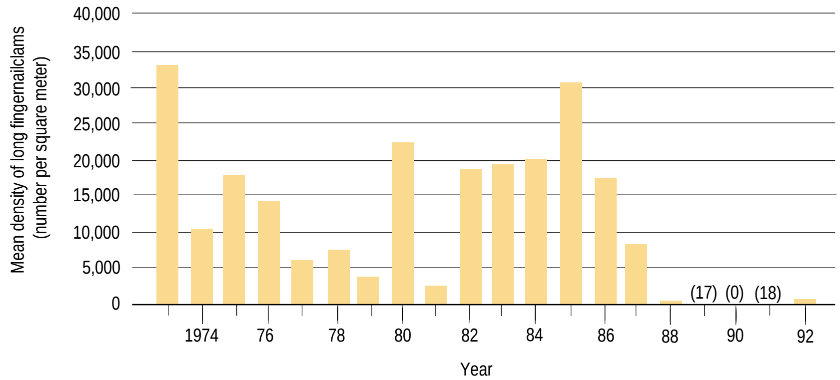

Fig. 11. Mean densities (±1 standard error) of the long fingernailclam in Pool 19 of the Upper Mississippi River during 1973–1992. Mean densities in 1989, 1990, and 1991 were too low to readily appear on the chart and are given in parentheses.

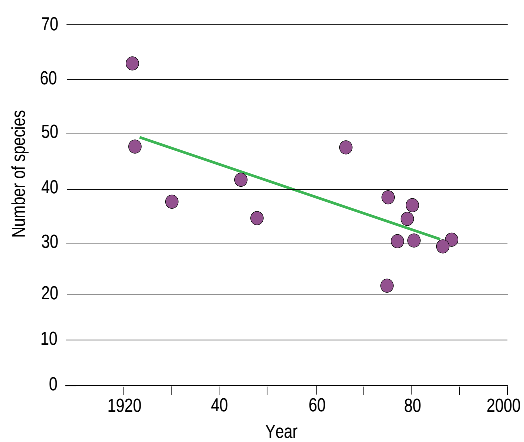

Fig. 12. Species richness of pearlymussels in the Upper Mississippi River drainage has declined substantially. Data compiled from the following sources: Shimek 1921; Grier and Mueller 1922; Ellis 1931a,b; Dawley 1947; Finke 1966; Coon et al. 1977; Fuller 1978, 1979; Mathiak 1979; Perry 1979; Thiel et al. 1979; Ecological Analysts Inc. 1981; Thiel 1981; Duncan and Thiel 1983; Holland-Bartels 1990.

Department of Natural Resources 1985). Three species—the Higgins eye, the fat pocketbook, and the winged mapleleaf—are federally listed as endangered. One of these, the fat pocketbook, may now be locally extirpated in the Upper Mississippi River (Wisconsin Department of Natural Resources 1985). The winged mapleleaf is now restricted to 16 kilometers of the St. Croix River, a tributary of the Upper Mississippi (Mueller 1993).

Effects of Zebra Mussels

By 1991 the zebra mussel, a nonindigenous species from eastern Europe, had entered the Upper Mississippi River via the tributary Illinois River (Sparks et al. 1994; Benson and Boydstun 1995). Zebra mussel populations expanded rapidly, and by mid-1993 zebra mussels were found throughout most of the Upper and Lower Mississippi River (Benson and Boydstun 1995). By mid-August 1993, average densities of zebra mussels in the lower Illinois River had increased to more than 50,000 per square meter of river bottom. Subsequent high mortality reduced densities there to about 4,000 per square meter by August 1994.

High densities of zebra mussels can degrade water quality, indirectly harming other organisms. Zebra mussels have caused major depletion of dissolved oxygen in affected reaches of the Seneca River in New York (Effler and Siegfried 1994) and in the Illinois River (Sparks et al. 1994); both rivers had populations exceeding 30,000 mussels per square meter. The concentration of dissolved oxygen in the Illinois River declined to 1.5 milligrams per liter, which is insufficient for the survival of many native aquatic animals.

The zebra mussel can directly harm certain native benthic invertebrates, particularly pearlymussels. Zebra mussels attach to hard surfaces, including the shells of pearlymussels (Fig. 13), by means of a byssal thread. Zebra mussel infestation on pearlymussels may interfere with pearlymussel feeding, reproduction, and movement. In Lakes Erie and St. Clair, pearlymussels suffered as much as a 100% mortality within 1 to 2 years in areas heavily infested with zebra mussels (Nalepa 1994; Schloesser and Nalepa 1994).

Since 1993 most of the native pearlymussels in the Illinois River have been infested with zebra mussels, and mortality of the native mollusks has increased markedly (Whitney et al. 1995). In the Upper Mississippi River, the number of zebra mussels has continued to increase, and the mussels are expected to reach high densities. Artificial substrate samplers deployed in Pool 8 from spring through midautumn, for example, had mean densities of 3.5 zebra mussels per square meter in 1992, 6.1 in 1993, 480 in 1994, and 4,240 in 1995 (W. G. Cope, U.S. Geological Survey, Upper Mississippi Science Center, La Crosse, Wisconsin, unpublished data).

D. Blodgett, © Illinois Natural History Survey

Fig. 13. Pearlymussel encrusted with zebra mussels, a nonindigenous species that recently invaded the Upper Mississippi River via the Illinois River, a tributary.

Pearlymussels in the Upper Mississippi River are being increasingly infested with zebra mussels (Tucker et al. 1993; Tucker 1994). At a site in Pool 26, for example, infestation rates increased from 27% in 1992 to 99.7% in 1993, and the average number of zebra mussels per pearlymussel increased concomitantly from 2 to 37 (Tucker 1994). Ricciardi et al. (1995) estimate that pearlymussel mortality will exceed 90% when the density of zebra mussels reaches 100 attached individuals per pearlymussel or 6,000 zebra mussels per square meter of habitat. Thus, the native pearlymussel fauna in the river could rapidly and severely decline unless methods for protecting pearlymussels from zebra mussels can be developed. Perhaps no other group of freshwater organisms is more seriously threatened with extinction than our native pearlymussels (Neves 1993).

Zebra mussels could also alter the invertebrate communities inhabiting the rock substrates of wing dams on the Mississippi River. Colonization of wing dams by zebra mussels will probably affect certain invertebrate species adversely while benefiting others (Wisenden and Bailey 1995).

Fishes

Distribution of Fishes

The fossil record suggests that the Mississippi River has long provided suitable habitat for many fishes; most present families and many genera of Mississippi River fishes date to the Miocene (5 to 25 million years ago) or earlier (Baker et al. 1991). During the last several million years, the larger rivers of the Mississippi River drainage system have presumably fluctuated between two very different channel patterns (braided and meandering), which provide different types and abundances of aquatic habitats. Although major changes in climate, including the Pleistocene glaciations, have also occurred, there have been few fish extinctions. Many fishes probably retreated ahead of southward-moving glaciers and repopulated northern reaches of the basin as the glaciers receded (Hynes 1970).

The 195 species of freshwater fishes in the main-stem of the Mississippi and Atchafalaya rivers represent almost one-third of the freshwater fish species in North America (Fremling et al. 1989). An additional 46 or more marine fishes may occur as far upstream as St. Francisville, Louisiana, during low-flow periods when saltwater intrudes into the river. Fremling et al. (1989) estimated that 67 fish species inhabit the Headwaters, 132 species inhabit the Upper Mississippi River, and about 150 species inhabit the Lower Mississippi and Atchafalaya rivers. Baker et al. (1991) estimated that 91 species of freshwater fishes inhabit the Lower Mississippi, with 30 or more other species present intermittently. Reviews have summarized the recent distribution and status of fishes in the Headwaters reach (Fremling et al. 1989), the Upper Mississippi River (Rasmussen 1979; Van Vooren 1983; Fremling et al. 1989; Pitlo et al. 1995), and the Lower Mississippi River (Fremling et al. 1989; Baker et al. 1991).

The fishes in swift-current habitats of the main-stem Lower Mississippi River have not been studied much because of the sampling difficulties posed by the channel's great size, depth, and strong current (Baker et al. 1991). Moreover, the abundance of fishes in swift-current habitats of the main-stem Lower Mississippi has probably been underestimated by traditional sampling methods (Baker et al. 1991).

The distribution of fishes in the Mississippi River has been influenced by natural and human barriers to migration. Until modern times, St. Anthony Falls at Minneapolis prevented the colonization of the Headwaters by 59 species that originally occurred in the Upper Mississippi. In 1963 completion of the locks at St. Anthony Falls provided access for all species previously excluded from the Headwaters above the falls. The dam at Coon Rapids, Minnesota, completed in 1906, is now the principal barrier to fish migration and maintains the distinct assemblage of fishes in the Headwaters upstream. Lock and Dam 19 at Keokuk, Iowa, may have impeded fish migration, particularly of the skipjack herring (Coker 1914, 1930). Work by Holland et al. (1984) supports this view but shows that some species do pass through Lock and Dam 19 and other navigation dams. Heavily polluted zones within the river have also impeded the movement of fishes. In August 1927, for example, organic (sewage) pollution caused about 75 kilometers of river below Minneapolis–St. Paul to lack sufficient oxygen to sustain any fishes. This situation may have persisted seasonally for decades.

Fish Habitats

Most fishes require several different habitats to complete a life cycle. The quantity and quality of certain habitats, however, have diminished in many reaches. In the Upper Mississippi, the navigation pools are aging, and overwintering habitats for fish have declined as sedimentation reduces water depth (McHenry et al. 1984; Bhowmik and Adams 1989; Holland-Bartels 1992; Gent et al. 1995). Recent die-offs of aquatic vegetation have reduced the suitability of many areas as nursery habitats for fishes. In many places, declines of invertebrate prey organisms associated with soft bottom sediments (Brewer et al. 1995; Wilson et al. 1995) and aquatic vegetation (Chilton 1990) have diminished food resources for fishes.

The Upper Mississippi River provides many aquatic habitats, including main channel, tailwater, main-channel border, side channel, navigation pool, floodplain lake or pond, slough, and tributary mouth (Littlejohn et al. 1985; Fremling et al. 1989). These habitats can differ markedly in current velocity, depth, temperature, water quality, bottom substrate, vegetative structure, food resources, and other characteristics.

The main channel has a swift current, coarse-sand or gravel substrate, and deep water. Tailwaters, which extend about 0.8 kilometers below each dam, have well-oxygenated water, rapid currents, and coarse substrates. Walleye, sauger, white bass, freshwater drum, and catfishes concentrate in these tailwaters. Dike fields along the main-channel border provide rocky substrates where walleye, sauger, channel catfish, smallmouth bass, white bass, black crappie, bluegill, redhorse, freshwater drum, and smallmouth buffalo concentrate (Holzer 1980; Pierce 1980; Pitlo 1981). Main-channel borders have multiple substrates, including silt, sand, wing dikes, snags, and riprap. Abundance of fishes in main-channel borders varies with season and river stage. The flow of side channels links them to other habitats during most of the year; these channels are used by many species. Nearshore zones in main-channel borders, side channels, and pools provide important nursery areas for many fish species, especially sunfishes (including bluegill, crappie, and largemouth bass); these same zones are important areas where certain sport-fish species, particularly sunfishes, are caught by anglers.

Pools upstream from dams are lakelike, shallow, and have bottoms composed of fine sediments; structure and hard substrates are limited to stumps, snags, wing dikes, and riprap. Vegetated areas in these navigation pools provide important spawning and nursery habitats. Many floodplain lakes and ponds are connected to the river only when flow is high. Sloughs, lakelike habitats that receive inflows from the river only when water levels are high, are much warmer than the main channel, may stratify thermally, and provide spawning and rearing habitat for many fishes.

In the northern reaches of the Upper Mississippi River, many fish populations seem limited by a lack of suitable winter habitat. During ice cover, dissolved oxygen is depleted in many shallow backwater lakes, reducing the suitability of these lakes for fishes (Bodensteiner and Lewis 1992; Gent et al. 1995; Knights et al. 1995). Even though flowing areas, such as the main channel and side channels, typically contain sufficient dissolved oxygen in winter for fishes, the high current velocities and cold temperatures (as low as 0°C) of flowing waters are stressful, and often lethal, to many fishes that inhabit the Upper Mississippi River (Sheehan et al. 1990; Bodensteiner and Lewis 1992). Consequently, the distribution of fishes in winter in the northern reaches of the river is strongly influenced by dissolved oxygen, temperature, and current velocity. Knights et al. (1995), in their study of a series of backwater lakes in Pool 5, found that bluegills and black crappies select areas where water temperature exceeds 1°C and the current is undetectable when dissolved oxygen is above 2 milligrams per liter. When dissolved oxygen falls below this level, fish move to areas with higher oxygen concentration and may tolerate water temperatures below 1°C and current velocity of 1 centimeter per second. Fish avoid areas with water temperatures lower than 1°C if the current velocity exceeds 1 centimeter per second (Knights et al. 1995).

In the main-stem Lower Mississippi River, swift-current habitats include the river channel, natural steep banks, revetted banks (covered with protective materials, mostly limestone rock, to prevent erosion), and flowing sandbars (Baker et al. 1991). Channel habitat has deep water, a swift current (1 to 5 meters per second), constantly shifting coarse-sand or gravel substrates, high suspended solids, and low primary productivity. The Lower Mississippi River provides plentiful habitat for fishes that thrive in swiftly flowing water. Few species, however, can tolerate the high current velocities of the upper and middle water column of channel areas for very long (Fremling et al. 1989; Baker et al. 1991). Most fishes probably inhabit areas near the banks (Pennington et al. 1983a) and the channel bottom where the current is slower (Baker et al. 1991).

Dike fields in the Lower Mississippi River often contain many fish species (Pennington et al. 1983b). Constructed of rock and wood pilings, dikes are used to constrain flow to the river's navigation channel. Some dike fields on the Lower Mississippi fill with sand, however, forming terrestrial habitats (Beckett and Pennington 1986).

Several fish species forage in the floodplain of the Lower Mississippi River when it is inundated by high water levels (Baker et al. 1991); these include gars, bowfin, common carp, buffalos, river carpsucker, channel catfish, blue catfish, white bass, crappies, and freshwater drum. Many fishes also use the inundated floodplain for spawning. Densities of larval fishes in the Lower Mississippi River are highest in backwaters, which are important nurseries for fishes and which contain a larval fish assemblage differing from that of the main-stem river (Beckett and Pennington 1986). The Lower Mississippi River floodplain includes many artificial floodplain ponds (borrow pits excavated during the construction of levees), which may also serve as important nursery areas for fishes (Sabo and Kelso 1991; Sabo et al. 1991).

Fish Harvest and Recent Trends

Lakes on the Headwaters reach are readily accessible and fished heavily in summer and winter for walleye, northern pike, muskellunge, basses, and other species (Fremling et al. 1989; MacGregor 1995). In the Upper Mississippi River, the catch of sport fishes has been dominated by bluegill and crappie (Farabee 1979). Other sport fishes, in approximate order of importance, include white bass, freshwater drum, sauger, channel catfish, yellow perch, walleye, and largemouth bass (Farabee 1979).

No commercial fishery exists in the Headwaters. For the Upper Mississippi River, commercial catch data are provided voluntarily and may be underreported. In the Lower Mississippi and Atchafalaya rivers, the commercial harvest of fishes is difficult to assess because of inconsistencies in methods of gathering and reporting data (Fremling et al. 1989).

The commercial harvest in the Upper Mississippi River is dominated by four groups of fishes: common carp, buffalos (bigmouth buffalo and smallmouth buffalo), catfishes (channel catfish and flathead catfish), and freshwater drum (Fremling et al. 1989). Between 1953 and 1977, these fishes made up 95% of the total catch and 99% of the monetary value. The common carp has ranked first among species in commercial catch for decades.

Although the commercial harvest on the Upper Mississippi has not changed greatly in the last century—totaling 6,200 metric tons in 1894, 3,200 tons in 1931, and 5,200 tons in 1987—the abundance of several species in the catch has changed greatly. The common carp, an introduced nonindigenous species first reported in the Mississippi River in 1883 (Cole 1905), has increased the most in abundance. In 1894, 206 metric tons of common carp (3% of the total harvest) were taken from the river; by 1899, the catch had risen to 1,400 metric tons. During 1953–1977, an average of 2,400 metric tons of common carp (47% of the average total annual harvest) were harvested each year (Kline and Golden 1979).

The grass carp, another nonindigenous species, first appeared in Pool 25 in 1975 and has since expanded upstream to Pool 5A (Rasmussen 1979; Fremling et al. 1989). Commercial harvest of grass carp was 6.1 metric tons in 1983 and 7.7 metric tons in 1991. Grass carps seem to be reproducing successfully in the Lower Mississippi River, in lower reaches of the Upper Mississippi River, and in tributaries as far upstream as the Illinois River (Raibley et al. 1995).

A decline in the harvest of buffalo fishes coincided with the increase in catch of common carp. Buffalo fishes made up 43% of the total catch in 1894 and averaged 22% of the total harvest during 1953–1977 (Kline and Golden 1979). The decline in buffalo fishes may have resulted from competition with common carp and from destruction of their spawning habitat (Coker 1930).

During 1978–1991, the total annual commercial fish harvest varied little, ranging from 3,900 metric tons in 1982 to 5,200 metric tons in 1987 (Duyvejonck 1997). Throughout this period the common carp ranked first in catch, about 30% of the total annual harvest. Buffalo fishes were second in biomass harvested, followed by freshwater drums and catfishes in roughly equal amounts. During this 14-year period, the harvest of American eel declined from 1.2 metric tons in 1978 to 0.3 metric tons in 1991. In the past 30 years, the reported commercial harvest of channel catfish has also declined.

Some fishes that inhabit swift-current habitats have apparently declined in the Upper Mississippi River since the construction of navigation dams but have not declined in the unimpounded Lower Mississippi (Pflieger 1975; Baker et al. 1991). These include the shovelnose sturgeon, blue sucker, and blue catfish (Pennington et al. 1983a; Beckett and Pennington 1986). The reproduction of paddlefish in the Upper Mississippi may be adversely affected by dams, which could impede paddlefish access to suitable spawning habitat. The decline of the federally endangered pallid sturgeon may be attributable to channelization of the open river below St. Louis, Missouri.

In Pool 2 of the Upper Mississippi River, the abundance and diversity of fishes have increased markedly in recent years in response to improved water quality. For decades, Pools 2 through 4 suffered severe oxygen depletion caused by sewage pollution from the Minneapolis–St. Paul metropolitan area (Wiebe 1927; Fremling 1964, 1989). In 1964, Pool 2 was badly polluted, oxygen depletion was common on summer nights, and only three fish species (common carp, gizzard shad, and white bass) were found during electrofishing surveys. Dissolved oxygen levels and overall water quality in this reach subsequently became much more favorable for fishes because of improved treatment of wastewaters, including advanced secondary treatment, nitrification, dechlorination, and a pretreatment program for industrial wastes (Fremling 1989; Johnson and Aasen 1989). In 1980 sampling through electrofishing produced a count of 16 fish species—13 more than in 1964—including walleye, sauger, smallmouth bass, largemouth bass, and channel catfish. In short, the pool has been transformed from a polluted water with few or no game fishes to one supporting a significant walleye and sauger fishery (D. Zappetillo, Minnesota Department of Natural Resources, St. Paul, personal communication). Clearly, a riverine fishery affected by pollution can be rejuvenated if water quality is improved.

Birds

The Mississippi River is a major bird migration corridor within North America. Millions of migratory birds use the Mississippi River corridor each year during fall and spring migration. The river's north-to-south orientation and nearly contiguous habitat make it critical to the life cycle of many migratory birds. Diving ducks, swans, pelicans, and cormorants use the river's large open-water pools, and dabbling ducks, geese, herons, egrets, black terns, bitterns, rails, and numerous resident and Neotropical migrant songbirds use shallow backwater riverine wetlands. Bottomland forests support migrating and nesting populations of songbirds, bald eagles, ospreys, herons, egrets, hooded mergansers, mallards, and wood ducks.

Riverine floodplain habitats are vital to the life cycles of many migratory birds. The modern landscape along the Mississippi River has been altered by an expanding human population for agriculture, industry, and urbanization and therefore contains far fewer wetlands, forests, and prairies than were present before the arrival of European settlers. Concerns regarding the long-term viability of bird populations that require these habitats relate directly to the adverse effects of sedimentation, operation and maintenance of the 2.7-meter channel navigation project, navigation-induced developments (including the impoundment of water), industrial and sewer effluent, urban and agricultural runoff, recreation, and other human-induced influences.

Colonial Waterbirds

Eight species of colonial waterbirds nest within Mississippi River habitats (Thompson 1977; U.S. Fish and Wildlife Service, Winona, Minnesota, unpublished data). Populations of great blue herons, great egrets, and double-crested cormorants appear to have declined on the Upper Mississippi River (Thompson 1977, 1978; Graber et al. 1978; Kirsch 1995; U.S. Fish and Wildlife Service, Winona, Minnesota, unpublished data). The range of cattle egrets has expanded to include areas in and near the river floodplain as far north as Pool 13 (U.S. Fish and Wildlife Service, Winona, Minnesota, unpublished data). Populations of least terns, which occur on the lower portion of the Mississippi River, appear stable. Information is not sufficient to estimate trends of black terns specific to the river, although the Breeding Bird Survey data (1966–1987) indicate 4% annual declines of black terns in Iowa, Minnesota, and Wisconsin (Hands et al. 1989). Data are not sufficient to examine abundance trends of Forster's terns, black-crowned night-herons, and yellow-crowned night-herons.

The Upper Mississippi River is an important nesting and feeding area for great blue herons and great egrets because its extensive bottomland forests and diverse aquatic areas provide suitable nesting and foraging habitats (Thompson 1978). Possible causes for apparent population declines in great blue herons and great egrets include poor water quality, loss of nesting trees and foraging areas, and toxic contaminants (Thompson 1978). Thompson (1977) reported that in 1977, 31 heron and egret colonies (18 colonies contained both species) occurred within the Upper Mississippi River. Thompson (1977) also reported average nesting success of 3.0 young herons per nest (518 nests examined) and 2.5 young egrets per nest (73 nests). Little reliable data on heron and egret productivity have been obtained since Thompson's study, though by 1978 the number of colonies had decreased to 27, again with 18 colonies containing both species (Thompson 1978). Several state and federal agencies have censused colonies since Thompson's study, but the years, methods, and reaches examined differed among surveys. The Upper Mississippi River National Wildlife and Fish Refuge began standardized surveys of great blue herons and great egrets in the refuge in 1993, but the Flood of 1993 hampered the initial survey. Refuge personnel reported that 6 of 18 colony sites that were active in 1992 were abandoned after nest initiation or were not colonized in 1993.

The Illinois Department of Conservation has aerially surveyed heron and egret rookeries since 1983, and other states along the Upper Mississippi River have conducted intermittent surveys. The Illinois surveys revealed that the number of active heron nests along the Illinois stretch of the Mississippi River substantially increased—from 2,111 nests in 21 colonies in 1987 to 5,045 nests in 20 colonies in 1991. Active egret nests also increased, from 351 nests in 14 colonies in 1987 to 1,099 nests in 18 colonies in 1991.

Double-crested cormorants were common breeders and abundant migrants on the Upper Mississippi River from St. Paul, Minnesota, to St. Louis, Missouri, during the 1940's and 1950's. The effects of contaminants and human disturbance caused their numbers to decline in the 1960's and 1970's; their numbers remained low for several years but slowly increased in the late 1980's (U.S. Fish and Wildlife Service, Winona, Minnesota, unpublished data). Current numbers of breeding and migrating cormorants remain much lower than historical levels (Figs. 14 and 15). A minimum of 418 cormorant pairs nested in 4 colonies in 1992 (Kirsch 1995), and 504 pairs nested in 9 colonies in 1993 (Kirsch 1997). Only 500 to 2,000 cormorants were seen during spring 1992–1993, and 5,000 to 7,000 were seen during the fall migration of 1991–1992, much lower numbers than the counts of 20,000 to 50,000 cormorants in the 1940's (U.S. Fish and Wildlife Service, Winona, Minnesota, unpublished data; Fig. 15).

The least tern, which breeds along the Mississippi River between Cape Girardeau, Missouri, and Vicksburg, Mississippi, has been monitored since 1985 (Rumancik 1985, 1986, 1987, 1988, 1989, 1990, 1991, 1992). A 100% increase occurred in tern numbers along the Lower Mississippi River (2,503 to 5,038) between 1989 and 1990; this increase cannot be explained by increased survey efforts or methods. Since 1990, least tern numbers have decreased slightly (Fig. 16).

Waterfowl

Waterfowl are the most prominent and economically important group of migratory birds on the Mississippi River. Numbers of waterfowl on the river often reflect national population trends and habitat conditions elsewhere. Four major groups of waterfowl use the Mississippi River during migration, and a few species also breed on the river. During fall and spring migrations, ducks tend to occupy areas with submersed and emergent aquatic vegetation or seasonally flooded areas rich in plant foods.

The most numerous diving ducks using the Mississippi River are canvasback, lesser scaup, redhead, and ring-necked duck. Other diving ducks, such as greater scaup, bufflehead, common goldeneye, hooded merganser, common merganser, and ruddy duck, also use the river, but their peak numbers during migration are relatively low. The most important areas for migrating diving ducks are Pools 5, 7, 8, 9, and 19. Together, Pools 5–9 extend for about 150 kilometers and include large open-water areas and shallow marsh zones that have supported luxuriant communities of submersed and emergent aquatic vegetation. Pool 19 extends from Keokuk, Iowa, to Oquawka, Illinois, but the most important area for diving ducks is the 32-kilometer reach from Keokuk to Fort Madison, Iowa. Tundra swan and Canada geese are also common migrants on the river. Most populations of canvasback and tundra swan that migrate in the Atlantic and Mississippi flyways now stage on the Mississippi River. Concentrations of these species are associated with the availability of food resources, particularly wildcelery and arrowhead tubers (Korschgen et al. 1988).

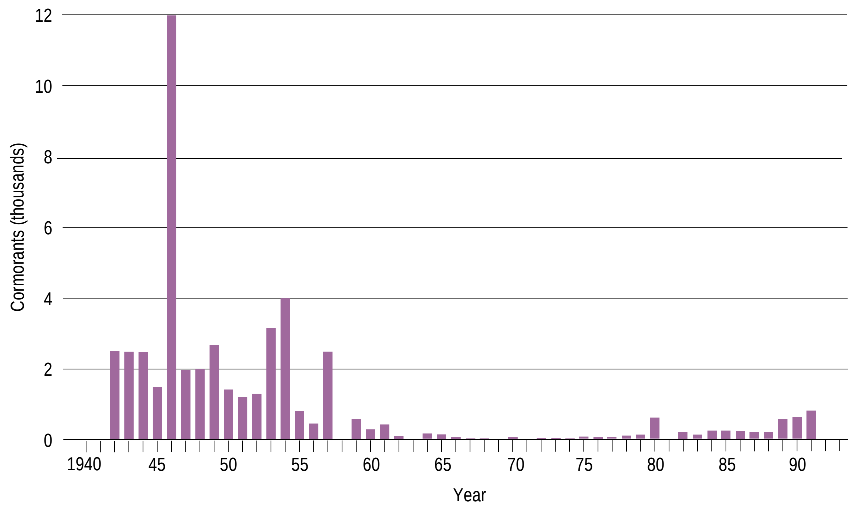

Fig. 14. Abundance of double-crested cormorants during the breeding season on the Upper Mississippi River National Wildlife and Fish Refuge from 1942 to 1992.

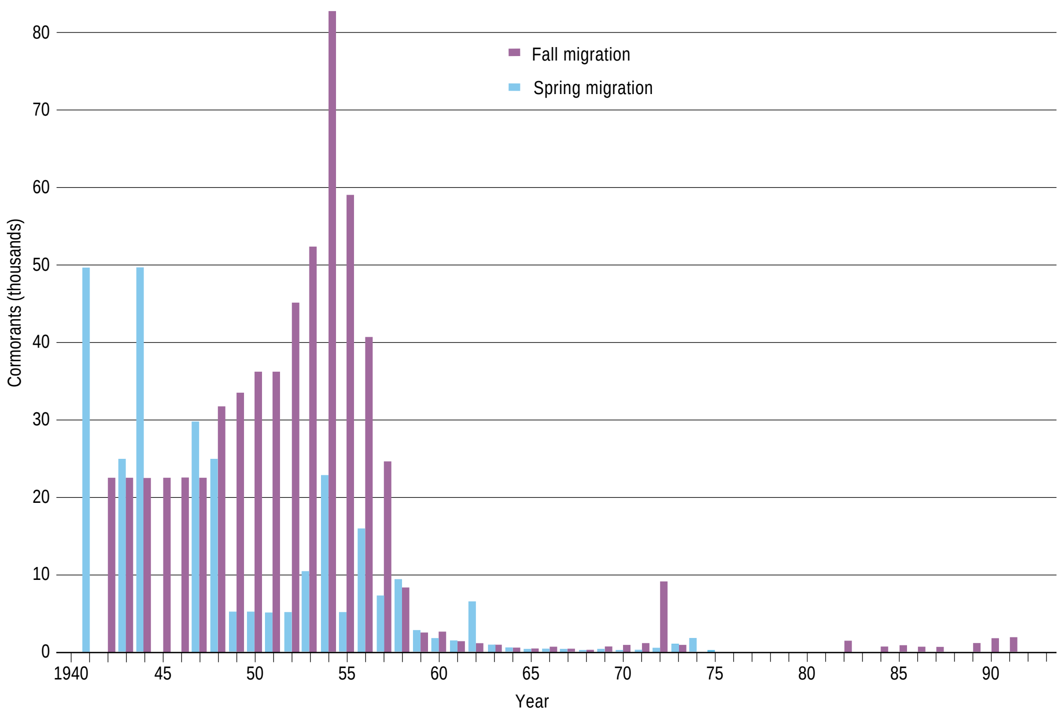

Fig. 15. Abundance of migrating double-crested cormorants on the Upper Mississippi River National Wildlife and Fish Refuge from 1942 to 1992.

Most continental populations of diving ducks depend upon large lakes and riverine impoundments in the upper portions of the Mississippi Flyway of the United States for feeding and resting areas during fall migration (Korschgen 1989). Historically, wetlands were a dominant feature of the landscape in Minnesota, Wisconsin, Iowa, and Illinois, totaling several million hectares. Wetland losses now total 6.7 million hectares in these states (Tiner 1984). Most of the riverine and deepwater wetlands remain, although few are used by waterfowl because of human-caused and natural changes; still, these wetlands may have future management potential.

Population survey data in the upper Midwest have been collected most consistently by the U.S. Fish and Wildlife Service and the Illinois Natural History Survey. Serie et al. (1983) compiled data from 1961 to 1977 on canvasbacks in Mississippi River Pools 7 and 8 (combined) and Pool 19. A peak of 147,000 canvasbacks was estimated in 1975 in Pools 7 and 8 and 169,000 in 1970 in Pool 19. During 1978–1994, a peak of 195,000 canvasbacks was estimated on Pools 7, 8, and 9 (Korschgen, unpublished data); large numbers of ring-necked ducks and lesser scaups also used these pools during this period

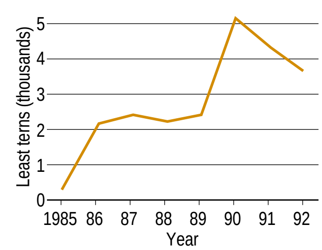

Fig. 16. Numbers of least terns counted during a small boat survey (Rumancik 1985, 1986, 1987, 1988, 1989, 1990, 1991, 1992) on the Mississippi River between Cape Girardeau, Missouri, and Vicksburg, Mississippi, from 1985 to 1992. The low numbers reported for 1985 reflect an incomplete and late survey.

(Fig. 17). In 1969 a peak of 875,000 diving ducks was estimated in Pool 19 in 1969. In Pool 19, fall waterfowl censuses between 1948 and 1984 revealed a mean yearly peak of 345,000 diving ducks (F. Bellrose and R. Crompton, Illinois Natural History Survey, Havana, unpublished data). The percent composition of peak numbers was 71% for lesser scaups, 18% for canvasbacks, 10% for ring-necked ducks, and 1% for redheads. Peak counts of these four species in Pool 19 have been much lower during the past 10 years (S. Havera and M. Georgi, Illinois Natural History Survey, Havana, unpublished data; Fig. 18).

The numbers of migrating waterfowl on the river have fluctuated greatly among years because of variations in waterfowl production on the breeding grounds, in food resources, and in weather. Censuses made by Bellrose in the Pool 19 area in fall 1941 revealed only moderate numbers of diving ducks. By 1946 lesser scaup numbers had greatly increased, whereas canvasback populations were low, peaking at only 5,500. Fall canvasback numbers increased steadily to 168,000 in 1970. Peak counts of canvasbacks, lesser scaups, and ring-necked ducks in Pools 7, 8, and 9 were relatively low during 1989–1991 (Fig. 17), probably because of the effects of drought on both continental reproductive success and on food resources in the Mississippi River. Drought conditions in the late 1980's also may have contributed to the decline in numbers of lesser scaups, ring-necked ducks, and canvasbacks in Pool 19 (Fig. 18). Submersed aquatic vegetation and fingernailclams, the principal food resources of diving ducks in Pool 19, have dwindled as a consequence of stresses incurred during low-flow periods resulting from drought (Wilson et al. 1995). The use of Pool 19 by migrating lesser and greater scaups, which feed heavily on fingernailclams, decreased substantially after the recent decline in this invertebrate food resource (Fig. 19).

Most of the changes in the distribution of migrating diving ducks in the Upper Midwest over the last several decades are directly attributable to habitat alteration caused by changes in land and water use. The Illinois River illustrates the potential severity of human modifications on such habitat (Mills et al. 1966; Havera and Bellrose 1985). Drainage and levee districts drained almost half of the existing bottomland lakes between 1909 and 1922, increasing flood heights and the deposition of sediments on the remaining lakes and floodplain (Bellrose et al. 1983). Erosion from uplands and tributary stream banks has caused high sedimentation rates and rapid filling of bottomland lakes. Loss of the invertebrate and plant-food components of the Illinois River habitat between 1946 and 1964 seriously reduced the numbers and distribution of lesser scaups, ring-necked ducks, and canvasbacks. Factors directly affecting the species composition and abundance of the wetland plants were fluctuating water level, water turbidity, water depth, and competition among plant species. The primary food item for diving ducks at Pool 19 has been fingernailclams (Thompson 1973), which may not have reached high densities until the early 1950's.

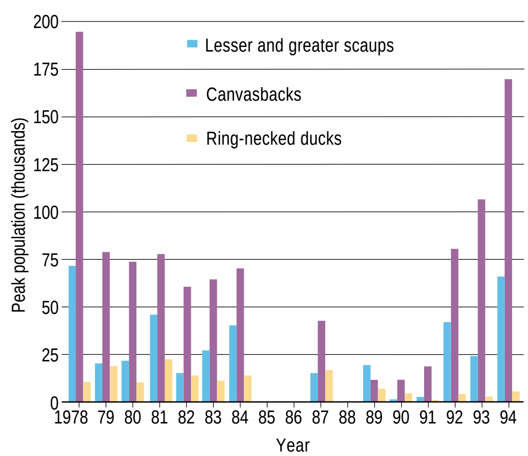

Fig. 17. Peak numbers of migrating lesser and greater scaups, canvasbacks, and ring-necked ducks from 1978 to 1994 in Pools 7, 8, and 9 of the Upper Mississippi River. No data are available for 1985, 1986, and 1988.

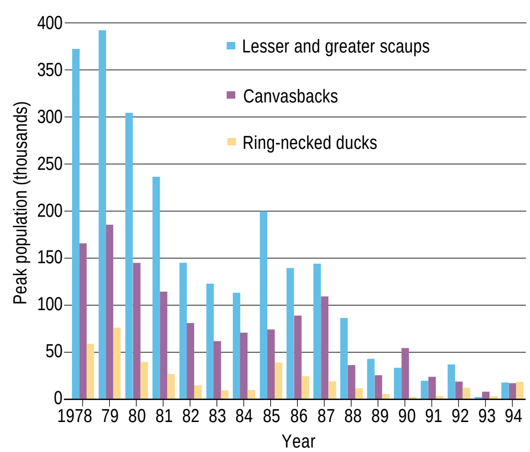

Fig. 18. Peak numbers of migrating lesser and greater scaups, canvasbacks, and ring-necked ducks from 1978 to 1994 in Pool 19 of the Upper Mississippi River.

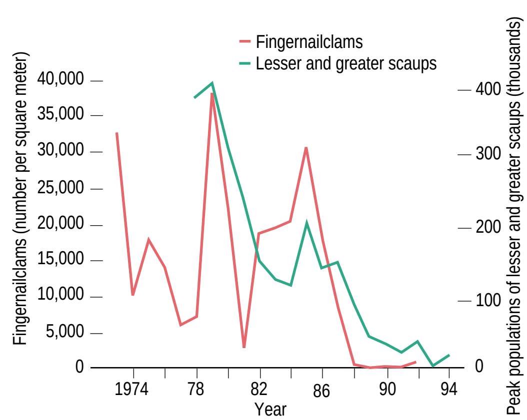

Fig. 19. Peak numbers of migrating lesser and greater scaups on Pool 19 during fall migrations are strongly linked to the abundance of fingernailclams, small mollusks important in the diet of scaup.

Mallards, wood ducks, blue-winged teals, hooded mergansers, and Canada geese nest in the Upper Mississippi River. Islands in Pools 7, 8, 9, and 10 may have high densities of nesting mallards (J. Wetzel, Wisconsin Department of Natural Resources, La Crosse, and R. Dahlgren, U.S. Fish and Wildlife Service, La Crosse, Wisconsin, unpublished data). Canada geese and, on occasion, blue-winged teals nest on islands that lack predators. Wood ducks and hooded mergansers, both cavity nesters, are common in bottomland hardwood forests. Mallard nesting densities as high as 70 nests per hectare, with nest success of 86%, have been recorded on islands managed for ground-nesting birds. From 10% to 40% of mallard ducklings typically survive to fledgings in places where adequate brood habitat is available near nesting islands (Korschgen and Kenow, unpublished data).

The floodplain of the Lower Mississippi River also includes critical wintering habitat for waterfowl (Reid et al. 1989). As much as 40% of the continental populations of mallards and American black ducks, as well as much of the continental wood duck population, overwinters in the Mississippi Alluvial Valley (Reinecke et al. 1989). Winter mallard populations formerly averaged about 1.5 million (Bellrose 1980; Bartonek et al. 1984) but have declined recently because of poor recruitment in prairie and parkland nesting areas in the northern United States and Canada. Habitat-protection efforts in the Mississippi Alluvial Valley now attempt, through acquisition and management of public lands, to offset habitat losses on private lands.

Raptors

The Upper Mississippi River is a major migration route and wintering area for bald eagles. During winter, more than 100 bald eagles roost at several traditional sites depending on ice conditions on the river. Peak numbers of bald eagles seen on informal roadside surveys between Winona and Red Wing, Minnesota, were 677 in the winter of 1990–1991, 540 in the winter of 1991–1992, 302 in the winter of 1992–1993, 363 in the winter of 1993–1994, and 206 in the winter of 1994–1995. Between 1963 and 1967 bald eagles were surveyed annually from Cairo, Illinois, to Minneapolis, Minnesota; the minimum number counted on these surveys was 397 in 1964, and the maximum was 885 in 1965.

Numbers of breeding bald eagles along the Upper Mississippi River have increased from 2 to 5 pairs in the 1970's to 43–44 pairs in 1993 and 1994 (Fig. 20). Productivity of young per occupied nest varied little between 1986 and 1993, with 0.95 to 1.5 young per nest (U.S. Fish and Wildlife Service, Winona, Minnesota, unpublished data).

Red-shouldered hawks are listed as endangered in Iowa and Illinois, threatened in Wisconsin, and of special concern in Minnesota. The Upper Mississippi River floodplain includes most of the forested habitat in Iowa and Illinois. Consequently, the floodplain is important for maintaining red-shouldered hawk populations in these states and for linking the habitats of northern and southern populations. The ecology of red-shouldered hawks has been studied along the Upper Mississippi River since 1983, and surveys have recently been expanded to cover more of the river (Pools 9–11 and 16–19). Thirty-two breeding territories were confirmed in 1992, and 37 territories in 1993 (J. W. Stravers, Midwest Raptor Fund, Pella, Iowa, unpublished data).

Songbirds

The U.S. Geological Survey Breeding Bird Survey is the only long-term data set for assessing population trends of breeding birds (migratory songbirds, as well as certain other migratory birds, and residents) along the Mississippi River corridor (Peterjohn 1994). Unfortunately, too few survey routes include portions of the Mississippi River floodplain to estimate trends specific to the river. Trends have been estimated, however, from routes within physiographic strata of the United States and Canada. Two of these strata (17 and 5) lie mainly along the Mississippi River but also include large areas removed from the river. Stratum 17 encompasses portions of Minnesota, Wisconsin, Illinois, and Iowa along the Upper Mississippi River from near St. Croix Falls, Wisconsin, to Clinton, Iowa, as well as up the Wisconsin River to Stevens Point, Wisconsin. Stratum 5 lies along the Lower Mississippi River valley and includes eastern Louisiana, eastern Arkansas, southern Missouri, western Tennessee, and western Mississippi. All of the other strata through which the Mississippi River flows primarily encompass upland areas away from the river.

In the Upper Mississippi River (stratum 17), 35 of the 119 species for which trends could be calculated from Breeding Bird Survey data had significant trends during 1966–1994. Twenty, or 59%, of these significant trends were positive, indicating increasing populations, whereas 15 (41%) were negative, indicating decreasing populations (Table 4). The continental trends for 27 of these species (80%) were in the same direction as the regional trends and were also significant. The continental trend for the American redstart was negative, but was positive for stratum 17. These trends may indicate that habitats in this region are not influencing

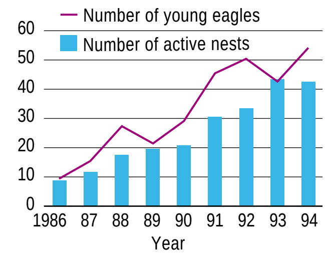

Fig. 20. Numbers of active bald eagle nests and young eagles produced between Pools 4 and 14 of the Upper Mississippi River.

populations of these bird species differently than elsewhere in North America. There were no significant continental trends for 6 species that had significant trends within the stratum (5 positive, 1 negative). For the 6 species, factors influencing populations may differ between the Upper Mississippi River and the rest of the continent.

In the Lower Mississippi River valley (stratum 5), 47 of the 105 species for which trends could be calculated had significant trends during 1966–1994. Thirteen of these significant trends were positive (population increasing), and 34 were negative (population decreasing) (Table 4). The continental trends for 24 of these species were significant and in the same direction as those for the region, which may indicate habitats in the Lower Mississippi River valley are not influencing populations of these bird species differently than elsewhere in North America. However, for 24 other species, the factors influencing populations may differ between the region and the rest of the continent. Continental trends for 6 species were significant and in the opposite direction of the trends on the Lower Mississippi River valley. Continental trends were not significant for 17 species with significant trends (2 positive, 15 negative) in the Lower Mississippi River.

Habitat-specific data on the occurrence and relative abundance of bird species are not yet available for most areas along the Mississippi River. Furthermore, the breeding success of most species along the Mississippi River is unknown. Finally, there are few site-specific data concerning songbird use of the river corridor during migration. Preliminary species lists and counts are available for a few areas within the refuges managed by the U.S. Fish and Wildlife Service.

Resident Birds

The only other nationwide, long-term survey of birds is the Christmas Bird Count. The methods used in this survey are far less standardized than the methods used in the Breeding Bird Survey, although certain methods of analysis can be used to assess trends in wintering and resident bird species. We used the methods of Morrison and Morrison (1983) to examine population trends for the ruffed grouse, red-bellied woodpecker, pileated woodpecker, downy woodpecker, hairy woodpecker, house finch, northern cardinal, great horned owl, and barred owl from 11 Christmas Bird Count locations centered on the Upper Mississippi River (Table 5). The house finch, which has been expanding its range and numbers throughout the upper Midwest, was the only species exhibiting a significant change in abundance in the Christmas Bird Count data.

Table 4. Bird species with significant trends in one or both of the two physiographic strata that lie primarily along the Mississippi River (data from the U.S. Geological Survey Breeding Bird Survey). Stratum 17 encompasses portions of Minnesota, Wisconsin, Illinois, and Iowa along the Upper Mississippi River (see text). Stratum 5 lies along the Lower Mississippi River valley and includes eastern Louisiana, eastern Arkansas, southern Missouri, western Tennessee, and western Mississippi.

Species	Stratum 17	Stratum 5	Continental
Pied-billed grebe	–	▼	■
Anhinga	–	▼	■
Mallard	▲	▲	▲
Canada goose	■	▲	▲
Great blue heron	▲	■	▲
Little blue heron	–	▼	■
Cattle egret	–	▲	▲
Black tern	▼	–	▼
Killdeer	▲	■	■
Northern bobwhite	■	▼	▼
Wild turkey	▲	▲	▲
Mourning dove	■	▼	■
Turkey vulture	▲	■	▲
Mississippi kite	–	▼	■
Red-tailed hawk	▲	■	▲
Broad-winged hawk	–	▼	■
American kestrel	■	▲	■
Barred owl	▲	■	▲
Great horned owl	▲	■	▲
Black-billed cuckoo	▼	–	▼
Yellow-billed cuckoo	■	▼	▼
Belted kingfisher	▼	▲	▼
Red-headed woodpecker	▼	■	▼
Red-bellied woodpecker	▲	■	▲
Yellow-shafted flicker	▼	▼	▼
Common nighthawk	■	▼	■
Ruby-throated hummingbird	■	▼	▲
Eastern kingbird	■	▼	■
Eastern wood-pewee	■	▼	▼
American crow	▲	■	▲
Fish crow	–	▼	■
Bobolink	▼	■	▼
Red-winged blackbird	■	▲	▼
Orchard oriole	■	▼	▼
Baltimore oriole	■	▼	■
Western meadowlark	▼	–	▼
Brown-headed cowbird	▼	■	▼
Common grackle	■	▼	▼
House finch	▲	–	■
Northern cardinal	▲	■	■
Rose-breasted grosbeak	▲	–	■
American goldfinch	■	▼	▼
Grasshopper sparrow	▼	▼	▼
Lark sparrow	■	▲	▼
Chipping sparrow	■	▼	■
Dickcissel	▼	■	▼
Field sparrow	▼	▼	▼
Vesper sparrow	▼	–	▼
Indigo bunting	■	▼	▼
Painted bunting	–	▼	▼
Purple martin	■	▼	■
Barn swallow	■	▲	▲
Tree swallow	▲	▲	▲
Northern rough-winged swallow	■	▲	■
Cedar waxwing	▲	–	▲
Yellow-throated vireo	▲	■	▲
White-eyed vireo	–	▼	■
Bell's vireo	▼	–	▼
Marsh wren	▼	–	■
Prothonotary warbler	–	▼	▼
Prairie warbler	–	▼	▼
Blue-winged warbler	▲	–	■
Common yellowthroat	■	▼	■
Yellow-breasted chat	–	▼	▼
American redstart	▲	▲	▼
House sparrow	■	▼	▼
Northern mockingbird	–	▼	▼
Grey catbird	■	▼	■
Carolina chickadee	–	▼	▼
Blue-gray gnatcatcher	▲	▼	▲
American robin	▲	▲	▲

▲ = Positive trend (increase in abundance).
▼ = Negative trend (decrease in abundance).
■ = There was no significant change.
– = The trend was not estimated.

Table 5. Locations of Christmas Bird Count (CBC) surveys along the Upper Mississippi River.

Year CBC initiated	City and state	Latitude and longitude
1972	Wabasha, Minnesota	44°16'N 92°02'W
1986	Winona, Minnesota	44°02'N 91°38'W
1972	La Crosse, Wisconsin	43°49'N 91°15'W
1986	Dubuque, Iowa	42°29'N 90°42'W
1957	Clinton, Iowa	41°58'N 90°09'W
1980	Burlington, Iowa	40°52'N 91°06'W
1962	Muscatine, Iowa	41°25'N 91°00'W
1987	Keokuk, Iowa	40°28'N 91°27'W
1972	Quincy, Illinois	39°51'N 91°25'W
1971	Hannibal, Missouri	39°42'N 91°21'W
1985	Pere Marquette, Illinois	39°00'N 90°36'W

Mammals

We are only aware of one comprehensive assessment of the mammalian fauna along the river corridor—the monitoring of annual furbearer harvest on the Upper Mississippi River National Wildlife and Fish Refuge. Dahlgren (1990) recently assessed trends in the abundance of mink and muskrats by examining indices of furbearer harvest per unit of trapping effort (total harvest divided by total number of trappers) on the refuge and in states along the river corridor.

Mink

The abundance of mink on the Upper Mississippi River National Wildlife and Fish Refuge declined precipitously during 1959–1965, remained low until about 1970, and then began to slowly increase to numbers less than half those of the 1950’s (Fig. 21). In contrast, mink populations in the adjoining states of Iowa (Fig. 21), Minnesota, and Wisconsin were relatively stable during this same period and did not exhibit the pattern of decline and partial recovery seen in refuge populations. These patterns indicate that some causal factor was operating in the river corridor but not in the mostly agricultural watersheds of the adjoining states.

Contamination of the riverine food web with polychlorinated biphenyls (PCB’s) was the most probable cause of the decline in mink populations during 1959–1965; mink are one of the organisms most sensitive to PCB’s (Giesy et al. 1994; Leonards et al. 1995), and their survival and reproduction are adversely affected by dietary exposure to small doses of PCB’s (Aulerich and Ringer 1977; Wren 1991; Leonards et al. 1995). The PCB’s biomagnify in aquatic food webs (Rasmussen et al. 1990), and mink, which feed largely on fish, occupy a position at or near the top of the aquatic food web.

Significant amounts of PCB’s have entered the river in Pool 2, which receives treated effluents and urban runoff from the Minneapolis–St. Paul metropolitan area as well as inflow from the Minnesota River (Boyer 1984; Metropolitan Waste Control Commission 1990; Steingraeber et al. 1994). The PCB’s entering the Mississippi River at Pool 2 have been transported downstream, contaminating a reach extending at least 200 to 375 kilometers downstream (Steingraeber and Wiener 1995). Navigation Pool 15 is another significant source area of PCB’s entering the Upper Mississippi River (Steingraeber et al. 1994; Steingraeber and Wiener 1995); however, PCB contamination of the reach downstream from Pool 15 is not readily apparent, in contrast to the area downstream from Pool 2.

The decline of mink on the refuge coincided with the probable period of most severe PCB contamination of the river. Conversely, the partial recovery of mink populations began in the late 1970’s and coincided with a period of declining PCB levels in riverine fishes (Hora 1984; Sullivan 1988). Based on experimental studies (Platonow and Karstad 1973), R. B. Dahlgren and K. L. Ensor (U.S. Fish and Wildlife Service, Bloomington, Minnesota, personal communication) estimated that a diet containing 33% fish that have PCB concentrations similar to those in the early 1970’s would contain enough PCB’s to prevent reproduction in mink. Several recent studies have confirmed the extreme sensitivity of mink to PCB’s (Giesy et al. 1994; Heaton et al. 1995a,b; Leonards et al. 1995). Heaton et al. (1995a) found that female mink fed a diet containing 0.72 parts per million of PCB’s for 2 months before breeding produced kits (young) with low body weight and survival rates. In another study (Hornshaw et al. 1983), female mink fed a diet containing 1.5 parts per million of PCB’s for 7 months before breeding produced no live kits. Average concentrations of PCB’s in common carp from the Upper Mississippi River, for comparison, were 4.0 parts per million during 1975–1976 and 2.1 parts per million during 1979–1980 (Hora 1984).

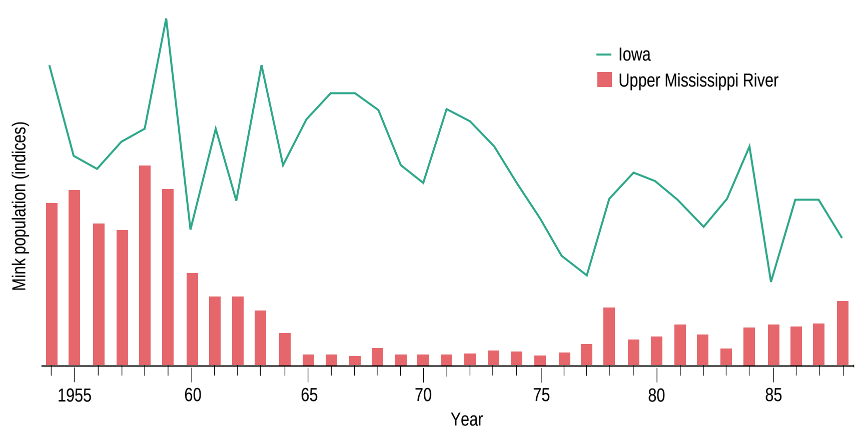

Fig. 21. Trends in the harvest of mink on the Upper Mississippi River National Wildlife and Fish Refuge and in the adjoining state of Iowa during 1954–1988.

In 1989–1991, PCB concentrations in mink carcasses from the Upper Mississippi River in Minnesota averaged 0.26 parts per million, exceeding concentrations in mink from all other areas of the state except Lake Superior (Ensor et al. 1993). Furthermore, recent studies show that PCB's continue to enter or cycle within the riverine ecosystem, suggesting that PCB's are transferred from the sediment to higher trophic levels via the benthic food chain (Steingraeber et al. 1994).

Muskrat

Soon after the lock and dam system was constructed in the 1930's, the abundance of muskrats increased on the Upper Mississippi River, probably because of the increase in water surface area and the development of marsh habitat (Dahlgren 1990). From 1960 to 1988, the harvest of muskrats varied annually but no overall trends in the population were obvious (Fig. 22), particularly given the potential influence of fur prices and the number of trappers on furbearer harvest. In Pool 9, Clay and Clark (1985) found that densities of muskrats were lower in backwater habitats than in open-water habitats, a difference that they attributed to past overharvest and long-term habitat degradation. Muskrat populations on the Upper Mississippi may decline if emergent plant communities continue to decline or change.

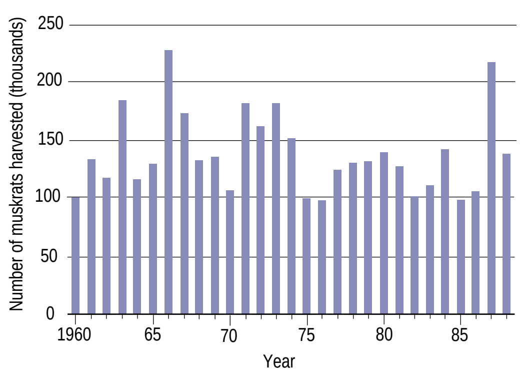

Fig. 22. Harvest of muskrats on the Upper Mississippi River National Wildlife and Fish Refuge during 1960–1988.

Gaps in Knowledge

Information on status and trends is sparse or unavailable for certain species and groups of organisms native to the river. Zooplankton, for example, are clearly important as a food resource during the life cycles of many riverine fishes, yet we are unaware of any systematic studies of the composition, abundance, or production of zooplankton in the river. Similarly, during our review we located only one published report about amphibians and reptiles in the Mississippi River valley (Collins 1991), and we are unaware of any systematic attempts to assess their abundance in the river's floodplain. In Iowa, which borders part of the Upper Mississippi River, most species of amphibians and reptiles declined substantially during the past century (Christiansen 1981).

From the forest manager's perspective, information on the composition and diversity of presettlement and existing floodplain forests is needed for restoration. The relationships among hydrological regimes (including the timing, frequency, intensity, and duration of floods), natural regeneration, and growth of woody species also merit examination. Forest simulation models could be developed to synthesize field data and to predict the effects of different river regulation schemes on forests (Bedinger 1978).

We lack sufficient information to predict long-term trends in aquatic vegetation, a key ecological component of habitat in the Upper Mississippi River ecosystem. Long-term monitoring and research have only recently begun to determine mechanisms of the decline that occurred in native submersed plants in the late 1980's. Elucidation of such cause and effect relationships and of long-term patterns in aquatic vegetation communities could aid in the restoration and maintenance of submersed, native aquatic plants in the pools of the Upper Mississippi.

The assessment of fish abundance in much of the Mississippi River, particularly the Lower Mississippi River, is complicated by swift currents, large fluctuations in river stage and discharge, deep water, wind-driven waves, floating debris, shifting substrates, navigation traffic, and the river's large size. Standardized approaches to fish sampling are hampered by the extreme variations encountered in such environmental conditions, both among habitats and over time within a limited area. There are few precise estimates of standing crops of fishes in the river (Pitlo 1987), and the large uncertainties in what estimates we do have greatly impede the detection of spatial or temporal variation in fish abundance. Gradual changes in riverine fish populations could long remain undetected because of the limited quantitative data available and the spatial and temporal variation in the riverine ecosystem.

Likewise, we encountered many difficulties when we attempted to determine the status and trends of bird populations on the river and its floodplain. In particular, standardized survey procedures are critically needed to assess riverwide trends. Assessment of trends in some colonial waterbirds, for example, was complicated or precluded by methodological differences among surveys and by inadequate spatial coverage of the river corridor in past surveys. To identify the demographic causes of population changes in bird populations, more information is needed on productivity and survival so that we can effectively develop models and guide management programs for any given group of birds.

Riverine resource managers may soon need information on how to manage an ecosystem in which the flows of energy and nutrients have been substantially altered by rapidly increasing populations of the nonindigenous zebra mussel. Studies of the Great Lakes indicate that the potential for food web modification in the backwater habitats of the Mississippi River is considerable. Zebra mussels will probably compete with some native biota for resources such as food and space, and the mussels will probably serve as prey for other species. Zebra mussels may also significantly modify the riverine habitat and its trophic resources by reducing the quantity of suspended solids and increasing water clarity, thereby affecting the distribution and abundance of native organisms. The ecological consequences of zebra mussels on the riverine ecosystem should be critically examined, with particular regard to their effects on food webs, trophic structure, and energy flow.

The biodiversity of the native pearlymussel fauna in the Mississippi River basin could soon be greatly reduced unless refugia from zebra mussels are provided. Examples of potential temporary refugia for native pearlymussels include hydrologically unconnected lakes, artificial ponds, and decommissioned fish hatcheries. Methods for moving native mussels to refugia with minimal handling stress, while ensuring that the native mussels are free of viable zebra mussels, are also needed.

Integrated resource management, based on science, is clearly needed for the Mississippi and other large rivers (Sheehan and Rasmussen 1993). However, the identification of appropriate management actions is often hampered by a lack of information on the ecological structure and function of this and other ecosystems. Information gaps concerning the causes of observed fluctuations in plant and animal communities also hamper the application of corrective measures. Comprehensive, long-term studies of the Mississippi River are surprisingly few, given the size and importance of this ecosystem. Existing long-term studies have focused on a few small segments of the river. Much of the data concerning the status and trends of riverine biota are unpublished or exist only in agency reports and gray literature. Many complex questions concerning environmental degradation, declining flora and fauna, and human effects require objective analysis to prevent continued deterioration of this natural resource.

Ecosystem Status and Health

The declines in key native species across many trophic levels signal a deterioration in the health of this riverine ecosystem. The Mississippi River ecosystem is often heralded as a multiple-use resource, and human use of the river and its floodplain for various purposes is expected to increase while inputs of sediment, nutrients, and potentially harmful chemicals from the watershed continue. Clearly, the greatest challenge on the Mississippi and other large rivers is to maintain ecological integrity while sustaining multiple human uses of the ecosystem (Sheehan and Rasmussen 1993).

Evidence is mounting that the cumulative effects of human activities have already exceeded the ecosystem's assimilative capacity. The abundances of many key native organisms, including submersed plants, native pearlymussels, fingernailclams, certain fishes, migratory waterfowl, colonial waterbirds, songbirds, and mink, have decreased along substantial reaches of the river in recent years or decades. The degradation of the Mississippi River delta represents a severe, nationally significant loss of wetland resources. Sediment deficiency is aiding in habitat destruction in Louisiana's coastal zone while, ironically, sediment deposition is threatening to destroy aquatic habitats in the impounded Upper Mississippi River.

Abundances of undesirable nonindigenous organisms in the river have increased along with these other problems. The common carp, a nonindigenous species present in the river since the late 1800's, has for decades ranked first among fishes in commercial fish harvest on the Upper Mississippi River. There is evidence, however, that common carp have contributed to declines in native submersed aquatic plants and buffalo fishes. The grass carp, another nonindigenous species, has greatly expanded its range in the river in the past two decades and appears to be reproducing in the Lower Mississippi River and in lower reaches of the Upper Mississippi. Dense stands of nonindigenous purple loosestrife are displacing native plants in many wetland areas within the floodplain. The submersed nonindigenous plant Eurasian watermilfoil has also increased in the past decade, particularly in areas of the Upper Mississippi River where native submersed aquatic plants have declined. Populations of the zebra mussel, which entered the Mississippi via the Illinois River around 1990, are expanding rapidly. At high densities, zebra mussels can adversely affect native organisms by direct attachment, by competing for space and planktonic food resources, and by depleting dissolved oxygen in the water column. The zebra mussel poses a particularly severe and immediate threat to the river's native pearlymussel fauna.

Recent declines in benthic invertebrates and submersed aquatic plants constitute a partial, yet significant, collapse in the food web supporting certain key fish and wildlife species. The decline in submersed aquatic plants in the late 1980's, unprecedented in the Upper Mississippi River, has greatly affected migratory canvasbacks, which feed on wildcelery tubers. Effects of the decline of submersed plants on the river's fish fauna have not been assessed but are likely, given that more than 80 resident fish species use aquatic vegetation for some habitat function (Janecek 1988).

The fingernailclam made up a substantial proportion of the biomass of the benthic fauna in the Upper Mississippi in the decades before its decline (Sparks 1980; Elstad 1986). Fingernailclams were important in the diets of migrating lesser scaups (Gale 1973; Thompson 1973) and certain bottom-feeding fishes (Starrett 1972; Jude 1973). In 1967, for example, diving ducks in Pool 19 consumed more than 2,000 metric tons of fingernailclams (Gale 1973). The use of Pool 19 by migrating lesser and greater scaups decreased greatly after the decline in fingernailclams. Similarly, in the mid-1950's, use of the Illinois River by migratory lesser scaups diminished rapidly after fingernailclams declined in response to pollution and acutely toxic conditions in bottom sediments (Sparks 1980, 1984; Schubauer-Berigan and Ankley 1991).

The riverine ecosystem of the Mississippi has undergone many changes. Most of the natural changes have occurred gradually over hundreds or thousands of years, whereas human-induced changes have occurred rapidly and recently. Several factors have apparently contributed to the recent declines in the river's flora and fauna, including habitat loss and degradation, point and nonpoint pollution, toxic substances, commercial and recreational navigation, deterioration of water quality during drought periods, reduced availability of key plant and invertebrate food resources, and invasions of nonindigenous species. The continued accumulation of sediment in the navigation pools on the Upper Mississippi River will eventually destroy or degrade much of the aquatic habitat in the pools (McHenry et al. 1984; Bhowmik and Adams 1989).

Some favorable biological trends are occurring on the Mississippi River. The abundance of bald eagles along the river corridor has increased, paralleling the national trend, in apparent response to the ban on eggshell-thinning insecticides. Mink populations have begun to recover, probably because of the declines in PCB contamination in riverine fishes that followed the ban on production of PCB's as well as other efforts to reduce PCB release into the environment. The fish and benthic invertebrate communities in the reach downstream from the Minneapolis–St. Paul metropolitan area have responded favorably to improved water quality resulting from advanced wastewater treatment.

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