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Aquatic Ecosystems

by
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Overview

Aquatic ecosystems have been especially subjected to the environmental degradation that has occurred over the last century in this country. Nearly every activity that occurs on land ultimately affects the receiving waters in that drainage. Whether it's pesticides and herbicides applied to crops, silt washed away because of vegetation removal, or even atmospheric deposition, aquatic ecosystems are a product of all local disturbances regardless of where they occur. In addition, waterways have been used for numerous activities other than providing habitat to aquatic organisms. They have been altered for transportation, diverted for agricultural and municipal needs, dammed for energy, borrowed as an industrial coolant, and straightened for convenience. These abuses have taken their toll as evidenced by worldwide declines in fisheries, monumental floods, an ever-growing list of endangered aquatic species, and communities trying to deal with finite water supplies.

The traits that make aquatic ecosystems particularly vulnerable also make them useful for monitoring environmental quality. Water serves to integrate these impacts by distributing them among the elements within aquatic ecosystems. Although dilution is occurring, subtle changes can be detected in habitats or organisms over a much larger area that may be the result of a single point source. A clean aquatic ecosystem with a healthy biological community will be indicative of the condition of the terrestrial habitat in the watershed, whereas the reverse may not necessarily be true.

This section features accounts of the alterations of aquatic habitats and their impacts on the biota. Evidence is presented documenting habitat destroyed by dams or channelization (*see* this section, Bogan et al.; Wlosinski et al.; and Wiener et al.), contaminants affecting organism health (*see* Hesselberg and Gannon; Lerczak and Sparks), wetlands affected by water-level control (*see* Wilcox and Meeker), reduced water quality (*see* Charles and Kociolek), and introductions of exotic species (*see* Hansen and Peck; Wiener et al.). These kinds of changes have caused declining biodiversity in many groups of aquatic species ranging from freshwater mussels to waterfowl.

Some encouraging trends are emerging. Persistent organic contaminants in the Great Lakes have declined (*see* Hesselberg and Gannon), and marginal water-quality improvement has been accompanied by increased diversity of the fish community (*see* Lerczak and Sparks). Despite these achievements, much needs to be done to effectively manage and conserve aquatic resources. As is evident from the reports on diatoms (*see* Charles and Kociolek),

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Habitat loss on a stretch of the Mississippi River modified for navigation (shown here) contrasts with a diverse complex of habitats on less developed areas of the Upper Mississippi River (*see* plate previous page).

algae (*see* Moe), and protozoa (*see* Lipscomb), little is known of the national trends in their populations, diversity, or biomass. Our knowledge of these groups is poor even though they provide basic functions of photosynthesis, production, and decomposition critical to the normal functioning of aquatic ecosystems.

Without increased monitoring, some very basic attributes of aquatic systems may be unknowingly lost or severely degraded. Groups of species that seem insignificant actually are critical parts of a food web that supports valuable commercial and sport species. Subtle changes such as losses of island habitat and constant water depth or level may lead to drastic declines in productivity or diversity (*see* Wlosinski et al.; Wilcox and Meeker). The loss of some of these integral pieces of ecosystems may be impossible to restore. The unsuccessful attempt to restore self-sustaining lake trout populations in the Great Lakes, despite massive efforts, exemplifies this (*see* Hansen and Peck).

Habitat Changes in the Upper Mississippi River Floodplain

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The U.S. Congress recognized the Upper Mississippi River (UMR) as a nationally significant ecosystem in 1986. The UMR extends northward from the confluence of the Mississippi and Ohio rivers to the Twin Cities, Minnesota, a distance of more than 1,360 km (850 mi). The floodplain (area between the bluffs) of the UMR includes 854,000 ha (2,110,000 acres) of land and water. The Mississippi River is a major migration corridor for waterfowl and provides habitat for more than 150 fish and 40 freshwater mussel species.

Since 1824 the federal government has implemented numerous changes on the UMR. The river was first modified by removing snags and then sandbars, with changes progressing to rock excavation, elimination of rapids, closing of side channels, and the construction of hundreds of wing dams, 27 navigation dams, and hundreds of kilometers of levees. Reservoirs formed by the navigation dams are known locally as pools (Fig. 1), which are numbered from north to south. Construction of the dams (mostly during the 1930's) significantly altered the northern 1,040 km (650 m) of the UMR (north of St. Louis, Missouri) by increasing the amount of open water and marsh areas. Wing dams and levees have altered aquatic habitats south of St. Louis (the open river) by reducing open-water habitats and isolating the river from much of the floodplain. Most of the changes to the river ecosystem were either designed for navigational improvements or to control the movement of river water. Here we investigate some of the habitat changes at various levels of resolution.

Spatial data were analyzed by using a geographic information system (GIS). Floodplain areas (bluff to bluff) and systemic land-cover/land-use data were obtained from Landsat Thematic Mapper data collected in 1989. Land-cover/land-use data from 1891 were created

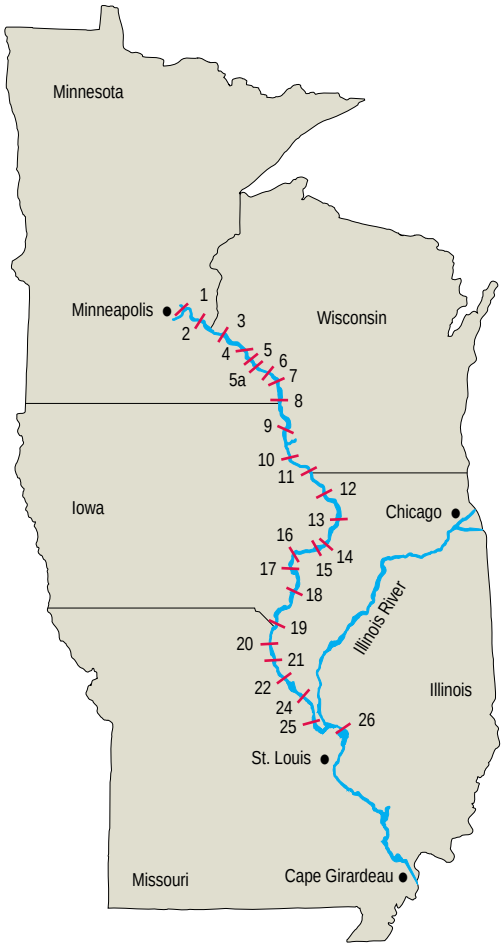

Fig. 1. The Upper Mississippi River. Numbers indicate reservoirs formed by navigation dams and known locally as pools.

from ground surveys conducted by the Mississippi River Commission. High-resolution land-cover/land-use data were created from 1:15,000 (scale) color infrared aerial photographs taken in 1989. Data for 1891 and 1989 were compared for Pools 4, 5, 8, 13, 26, and for a 64-km (40-mi) stretch of river, near Cape Girardeau, Missouri, which is not affected by navigation dams. Historical aerial photographs from 1939, 1954, 1967, and 1989 were used to measure island loss in an area just upriver of the dam in Pool 8.

Long-term daily data at three stations on the open-river portion of the UMR were analyzed to evaluate changes in the relationship between discharges and water-surface elevations.

Status and Trends

Comparison of the land-cover/land-use data between 1891 and 1989 in the dammed portion of the UMR showed that open water and marsh habitats generally increased, mostly at the expense of grass/forb, woody terrestrial, and agricultural classes. For example, the combined classes of open water and marsh in Pool 8 have increased from 3,600 ha (8,900 acres) in 1891 to 9,500 ha (23,430 acres) in 1989 (Figs. 2a, b). Similar increases in these two classes were found at Pools 5 and 13. In Pools 4 and 26 increases were less significant.

In many pools inundation created an impounded area with a mosaic of islands, open water, and marsh, which, in general, increased aquatic habitat for fish and wildlife. Although dam construction has benefited aquatic habitat in many pools, the reservoir aging process has reduced these benefits, especially in areas just upriver of dams. For example, island areas have been steadily eroding upriver of the dam in Pool 8 (Fig. 3). The dam that forms Pool 8 began operating in 1937, and photographs taken 2 years later showed 253 ha (624 acres) of islands. By 1989 the island area in the same location was reduced by 79% to 52 ha (129 acres).

Sedimentation is also a major concern on the UMR; rates of 1 to 3 cm/yr (0.4-1.2 in/yr) have been measured (McHenry et al. 1984). Erosion and sedimentation were both detected in comparisons between present elevation data and surveys before dam construction. Erosion was more prevalent in shallow areas and sedimentation more prevalent at greater depths. Erosion and sedimentation converge at depths of between 0.9 and 1.5 m (3 to 5 ft). This has resulted in a more homogeneous distribution of depth, which is dominated by areas 0.9 to 1.5 m (3 to 5 ft) in depth. Similar frequency distributions of water depth were observed for lower portions of Pools 8 and 13. Comparison of historical and present bottom geometry revealed the loss of elevational diversity.

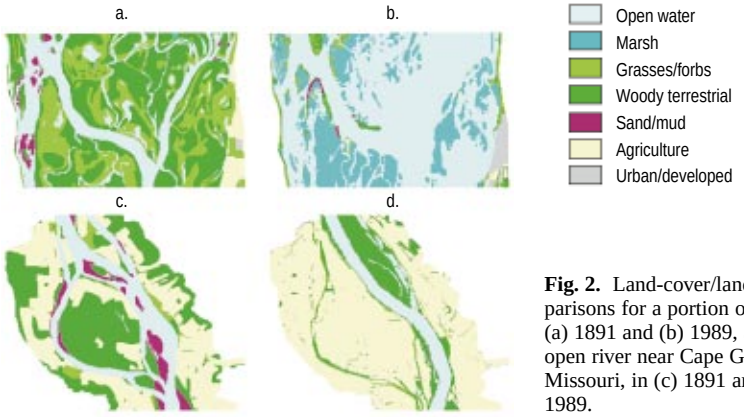

Fig. 2. Land-cover/land-use comparisons for a portion of Pool 8 in (a) 1891 and (b) 1989, and the open river near Cape Girardeau, Missouri, in (c) 1891 and (d) 1989.

In areas of the UMR unaffected by navigation dams (the 40-mi stretch of river near Cape Girardeau), there was a 28% reduction in open water and a 38% reduction in woody and terrestrial habitat between 1891 and 1989 (Figs. 2c, d). Agricultural areas increased by 6,360 ha (15,700 acres). The 1,900-ha (4,710-acre) reduction of open water can be explained by the construction of levees and wing dams (also known as pile dikes). One large side channel that existed in 1891 was cut off by construction of a levee, reducing the area of water by 550 ha (1,350 acres). In all, nearly 2,000 km (1,240 mi) of levees now isolate more than 400,000 ha (988,000 acres) from the river during all but the highest discharge rates.

Wing dams and levees, along with other changes to the watershed, have also had a major effect on habitats by changing the relationship between discharge and water-surface elevations. Wing dams have narrowed and deepened the main channel so that water elevations at low discharges are now lower than they were historically. Levees restrict flows and result in higher

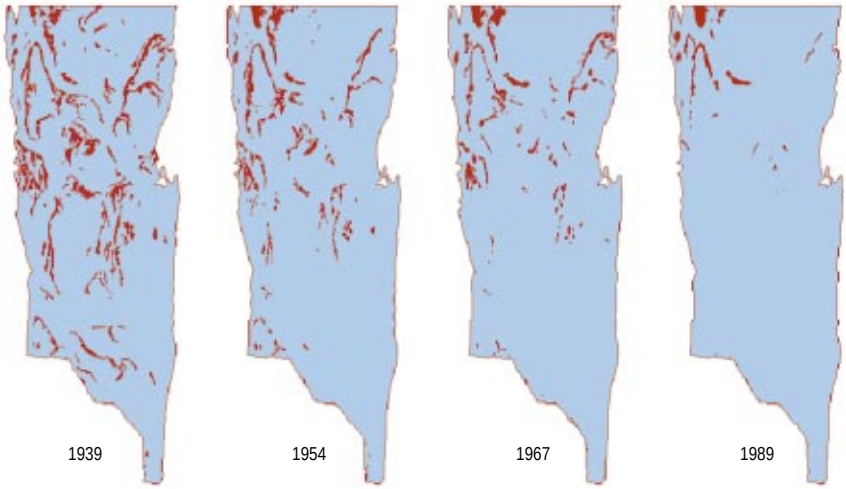

Fig. 3. Island loss that has occurred in Pool 8, in the area just upriver of the dam, since construction of the lock and dam system.

water elevations during high discharges. Water-surface elevations at relatively low discharges (60,000 cfs) have dropped about 2.4 m (8.0 ft) over the record 133-year period at St. Louis, Missouri, 0.5 m (1.5 ft) over the 52-year record at Chester, Illinois, and 1.5 m (5.0 ft) over the 60-year record at Thebes, Illinois. Water-surface elevations at relatively high discharges (780,000 cfs), however, have risen about 2.7 m (9 ft) over the record period at St. Louis, 1.5 m (5.0 ft) at Chester, and 1.1 m (3.6 ft) at Thebes.

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Biota of the Upper Mississippi River Ecosystem

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The Mississippi River is one of the world's major river systems in size, habitat and biotic diversity, and biotic productivity. The navigable Upper Mississippi River, extending 1,370 km (850 mi) from St. Anthony Falls (Minnesota) to the confluence with the Ohio River, has been impounded by 27 locks and dams to enhance commercial navigation. The reach between two consecutive locks and dams is termed a "pool." The upstream portions of many pools are similar to the unimpounded river, whereas the downstream reaches are similar to reservoirs.

The Upper Mississippi contains a diverse array of wetland, open-water, and floodplain habitats, including extensive national wildlife and fish refuges. Human activities, though, have greatly altered this river ecosystem; much of the watershed is intensively cultivated, and many tributary streams deliver substantial loads of nutrients, pesticides, and sediment from farmland. Pollutants also enter the river from point sources.

We examine recent temporal trends in the abundance of several key groups of organisms in the Upper Mississippi River and show that certain flora and fauna have declined along substantial reaches of the river. Our analysis is spatially constrained by available data to the reach of river extending from Pool 2 (near Minneapolis-St. Paul, Minnesota) to Pool 19 (near Keokuk, Iowa).

Information on the abundance of selected riverine biota was obtained by compiling historical data and by censusing or sampling. Bottom-dwelling invertebrates, collectively termed benthic macroinvertebrates, were extensively studied. Some of these organisms are important in the diets of fish and wildlife and are useful as biological indicators of toxic pollution. Data on densities of fingernail clams (*Musculium transversum*) from 1973 to 1992 were obtained from regional scientists, published literature, and sampling (Wilson et al. 1994) in selected pools (pools are numbered consecutively from upstream to downstream).

In 1992 benthic macroinvertebrates were sampled in soft sediments of five reaches of the Upper Mississippi and one reach of the Illinois River to estimate densities of fingernail clams and the burrowing mayfly (*Hexagenia*). In 1975 and 1990, benthic macroinvertebrates were extensively sampled in five habitats (marsh, bay, open water, side channel, and dredged side channel) in Pool 8 of the Upper Mississippi (near La Crosse, Wisconsin) to examine changes in abundance, biomass, and community structure (Brewer 1992). Data on the unionid mussel fauna in the river were obtained from the literature and other 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).

Wildcelery plants (*Vallisneria americana*) were sampled each August during 1980-84 and 1989-93 in quadrants along 12 0.8-km (0.5-mi) transects in Lake Onalaska, a backwater lake in Pool 7. Numbers of canvasback ducks (*Aythya valisineria*) in Pool 7 during fall migration were determined by aerial surveys. Trends in the abundance of mink (*Mustela vison*) were assessed by examining indices of mink harvest per unit of trapping effort (total harvest/total number of trappers) on the Upper Mississippi River National Wildlife and Fish Refuge and in states along the river corridor (Dahlgren 1990).

Status and Trends

Benthic Macroinvertebrates

Densities of fingernail clams declined significantly ($P \leq 0.05$) in five of eight pools examined (declines in Pools 2, 5, 7, 9, and 19; Figs. 1 and 2) along 700 km (435 mi) of river from Hastings, Minnesota, to Keokuk, Iowa. Densities in Pool 19, which had the longest historical record on fingernail clams, averaged 30,000/m^2 (2,800/ft^2) in 1985 and decreased to zero in 1990 (Fig. 2). In 1992 densities of fingernail clams were still low in sampled areas on the Upper Mississippi and Illinois rivers, averaging 5-94 individuals/m^2 (0.5-8.7/ft^2). Only

8% of 721 samples taken in 1992 had densities exceeding 100 fingernail clams/m^2 (9.3/ft^2). Corresponding mean densities of burrowing mayflies in these areas ranged from 10 to 99/m^2 (0.9-9.2/ft^2).

Wilson et al. (1994) hypothesized that the declines in fingernail clams in Pools 2 to 9 were linked to point-source pollution, and that the declines in Pool 19 were linked to low-flow conditions during drought. The causal mechanisms by which low flow influences fingernail clam abundance may involve unfavorable changes in the chemistry of sediment pore water.

In Pool 8, the structure of benthic macroinvertebrate communities changed between 1975 and 1990 in all five habitats studied. Standing crop of the benthos decreased significantly in both open-water and bay habitats, and diversity and abundance decreased in open-water habitat (Brewer 1992). These declines suggest that the standing crop of invertebrates has decreased substantially in Pool 8 because open-water habitat was 45% of the total area of the pool.

The biodiversity of the unionid mussel 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. Many of these species are commercially important; others are threatened or endangered. Unionid mussels are further imperiled by the zebra mussel (*Dreissena polymorpha*), which recently invaded the Illinois and Upper Mississippi rivers.

Rooted Aquatic Plants

The abundance of submersed aquatic plants—including wildcelery, which produces a vegetative tuber important as food for certain migratory waterfowl—declined along extensive reaches of the Upper Mississippi River in the late 1980's. This decline has been attributed to changing environmental conditions caused by the severe midwestern drought of 1988-89. In Pool 7, the abundance of wildcelery was fairly stable during 1980-84, but declined greatly after the dry summer of 1988. In Pools 5 through 9, more than 4,000 ha (10,000 acres) of wildcelery beds were lost (C.E. Korschgen, Upper Mississippi Science Center, unpublished data). Overall, the abundance of wildcelery and many other submersed plants declined along 600 km (375 mi) of river from Pool 5 to Pool 19. Coincidentally, the abundance of the exotic plant Eurasian watermilfoil (*Myriophyllum spicatum*) has seemingly increased, particularly in locations formerly occupied by wildcelery or other native submersed plants.

Migratory Birds

Millions of migratory birds use the Mississippi River corridor during fall and spring migration. The river is critical in the life cycle of many migratory birds because of its north-to-south orientation and its nearly contiguous habitat. Diving ducks, swans, pelicans, and cormorants use the river's open waters. Dabbling ducks, geese, herons, egrets, terns, bitterns, rails, and many resident and Neotropical songbirds use the shallow riverine wetlands. Bottomland forests support migrating and nesting songbirds, and nesting raptors, herons, egrets, and waterfowl.

The primary factor affecting the use of the river ecosystem by birds is the production of food by various plants and animals. The number of birds in riverine habitats decreases rapidly if preferred food resources are unavailable. The use of Lake Onalaska (Pool 7) by canvasback ducks (*Aythya valisineria*), for example, decreased greatly when the abundances of their preferred foods, wildcelery and benthic invertebrates (Korschgen 1989), declined in the late 1980's (Fig. 3). A gradual increase in foods in 1992 resulted in increased use by canvasbacks (C.E. Korschgen, unpublished data).

Numbers of other migratory waterfowl have also decreased along the river corridor, reflecting deterioration of habitat on the breeding grounds and the river. The decrease in the abundance of fingernail clams has adversely affected waterfowl that feed heavily on the small mollusk, particularly lesser scaup (*Aythya affinis*). The peak number of lesser scaup on Pool 19 during fall migrations, for example, has decreased from 300,000-500,000 in the 1970's to fewer than 25,000 in 1993.

Mink

The abundance of mink (*Mustela vison*) on the Upper Mississippi River Refuge declined precipitously during 1959-65, remained low until about 1970, and then began to slowly increase to numbers that are now less than half those of the 1950's (Dahlgren 1990). In contrast, mink populations in the adjoining states of Iowa, Minnesota, and Wisconsin were relatively stable during this period and did not exhibit the pattern of decline and partial recovery seen in populations on the refuge. These patterns indicate that some factor unique to the river corridor, not present in the mostly agricultural watersheds of the adjoining states, caused the decline of mink populations on the refuge.

The survival and reproduction of mink are adversely affected by dietary exposure to small doses of polychlorinated biphenyls, (PCBs; Aulerich and Ringer 1977; Wren 1991). The

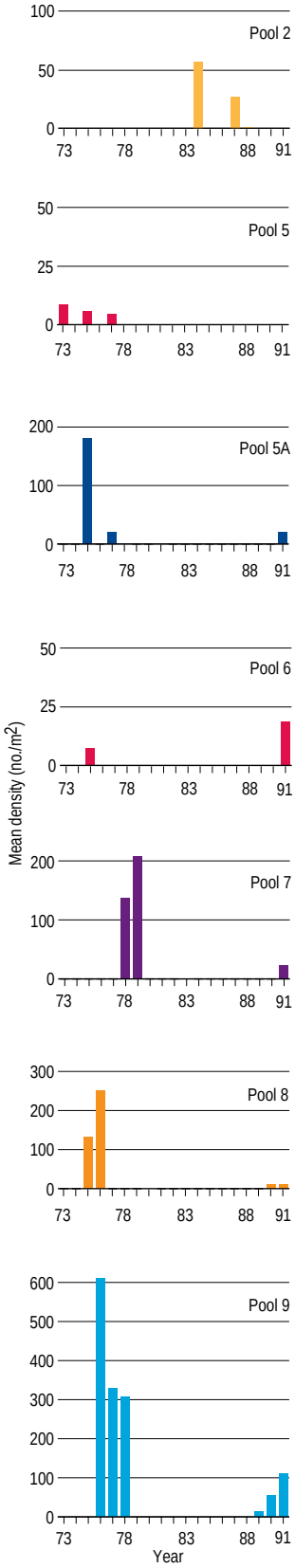

Fig. 1. Mean density of the fingernail clam *Musculium transversum* in Pools 2, 5, 5A, 6, 7, 8, and 9 of the Upper Mississippi River during 1973-91.

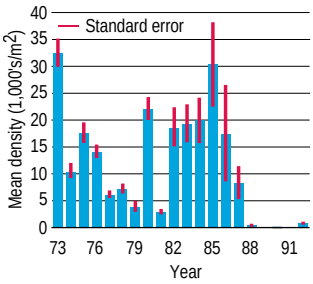

Fig. 2. Mean densities (±1 standard error) of the fingernail clam *Musculium transversum* in Pool 19 of the Upper Mississippi River during 1973-91. Mean densities that were too low to appear in the chart are 1989 (17 clams/m^2), 1990 (0), and 1991 (18 clams/m^2).

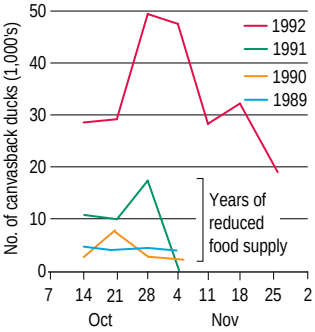

Fig. 3. Numbers of canvasback ducks (*Aythya valisineria*) in Pool 7 of the Upper Mississippi River during four consecutive fall migrations (1989-92) varied in relation to the abundance of plant and animal food in the pool.

decline of mink on the refuge coincided with the probable period of most severe PCB contamination in the river. Conversely, the partial recovery of mink populations that began in the late 1970's coincided with a period of declining PCB levels in riverine fishes (Hora 1984). R.B. Dahlgren and K.L. Ensor (U.S. Fish and Wildlife Service, personal communication) estimate that a diet containing 33% fish, having PCB concentrations similar to those in the early 1970's, would contain enough PCB to prevent reproduction in mink, based on experimental toxicity studies (Platonow and Karstad 1973). In 1989-91, PCB concentrations in mink from the Upper Mississippi River in Minnesota exceeded those in mink from all other areas of the state except Lake Superior (Ensor et al. 1993). Recent studies show that PCBs continue to enter or cycle within the riverine ecosystem and that they are transferred from the sediment to higher trophic levels via the benthic food chain (Steingraeber et al. 1994).

Ecosystem Health

The declines in these riverine flora and fauna signal a deterioration in the health of this ecosystem. In recent decades, populations of fingernail clams, unionid mussels, certain other invertebrates, submersed vegetation, migratory waterfowl, and mink have decreased along extensive reaches of the river. The Upper Mississippi is often heralded as a multiple-use resource, and human use of the river for navigation, hydropower, discharge of wastes, and other purposes may increase while inputs of sediment, nutrients, and chemicals from the watershed continue. Yet the cumulative impacts of humans may already exceed the assimilative capacity of this ecosystem.

Many complex questions concerning environmental degradation, declining flora and fauna, and human impact on this ecosystem need objective analysis and resolution. It is suspected that mink populations declined in response to PCB contamination and that fingernail clams declined in response to sediment toxicity, perhaps linked to low-flow conditions during droughts (Wilson et al. 1994). The factors causing most of the observed biotic declines are largely unknown, however, hampering the application of corrective measures. Several factors, for example, are suspected of contributing to declines in the unionid mussel fauna, including habitat modification and degradation, contaminants, overharvest, commercial and recreational navigation, and poor water quality (Williams et al. 1993). The need for scientifically based, integrated resource management of the Upper Mississippi is illustrated by the economic and ecological effects of the flood of 1993 on the river floodplain and its inhabitants. Federal and state agencies involved with resource management need integrated, proactive policies based on an understanding of the ecological structure and functioning of this complex ecosystem.

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Fish Populations in the Illinois River

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The Illinois River is formed by the confluence of the Des Plaines and Kankakee rivers, about 80 km (50 mi) southwest of Chicago, Illinois. It then flows 439 km (273 mi) to join with the Mississippi River about 50 km (31 mi) northwest of St. Louis, Missouri (Fig. 1). The Illinois River has been extensively modified and degraded by industrial and municipal pollution for most of this century (Mills et al. 1966). The upper river reaches above the Starved Rock Dam (Fig. 1) became the most degraded because most of this pollution originated in the densely populated and heavily industrialized Chicago metropolitan area. In fact, by the late 1920's, the upper river was thought devoid of fish (Thompson 1928). Soon after this period, as pollution-control efforts began to have an effect, fish gradually returned.

Changes in the composition of a fish community in a polluted environment can be a useful index for assessing environmental health and the effectiveness of pollution control because different fish species vary in their ability to tolerate effects of pollution. In 1957 the Illinois Natural History Survey (INHS) initiated an annual electrofishing survey of the Illinois River to monitor fish populations. A central purpose of the survey was to relate changes in fish populations to environmental conditions. This article summarizes trends in fish populations of the upper Illinois River as determined from electrofishing catches from 1959 to 1993.

Status and Trends

Fish sampling was conducted at five stations in the upper Illinois River and at two stations in the Des Plaines River (Fig. 1) from late August through October. Data from these seven fixed stations were combined for analyses. At each station, fish were sampled by electrofishing for 1 hr; thus, catches are expressed as number of

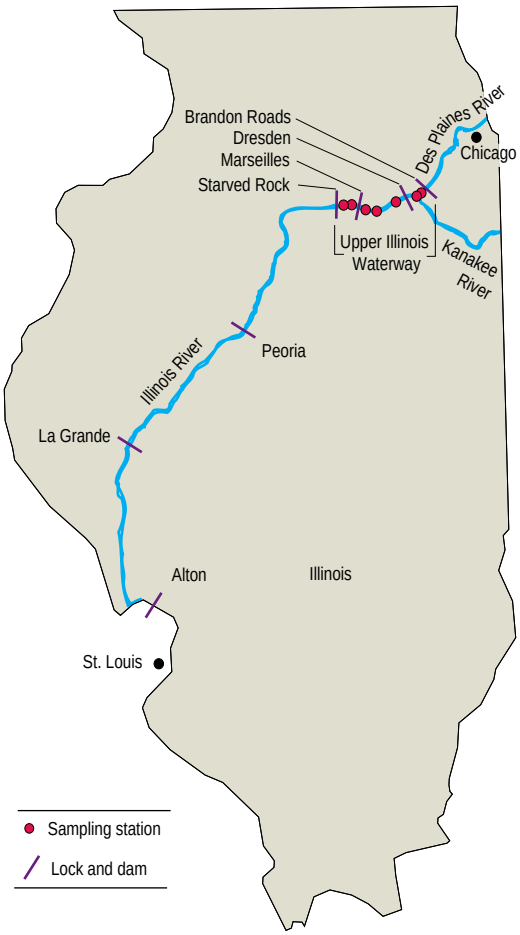

Fig. 1. The Illinois River with locations of navigation locks and dams. Locations of the Illinois Natural History Survey's upper Illinois Waterway electrofishing stations are also shown.

fish obtained per hour of sampling. Fish were stunned in an electric field, gathered with a net, measured, checked for externally visible abnormalities (sores, eroded fins, etc.), and returned to the water. The same methods and similar equipment have been used for all years of the survey to allow comparability of data among years.

Fishes of the family Centrarchidae (e.g., largemouth bass [*Micropterus salmoides*], bluegill [*Lepomis macrochirus*]) were treated as a group to simplify data analysis because they have very similar habitat requirements and are generally considered intolerant of polluted conditions. (The green sunfish [*Lepomis cyanellus*], however, is usually indicative of a stressed environment [Karr et al. 1986].) Also, because many of these fishes are piscivorous, their presence or absence will have a direct impact on overall fish community composition. Catches of common carp (*Cyprinus carpio*) and goldfish (*Carassius auratus*), both non-natives to North America, and their hydrids were analyzed separately. These two species are omnivorous habitat generalists that are tolerant of polluted waters.

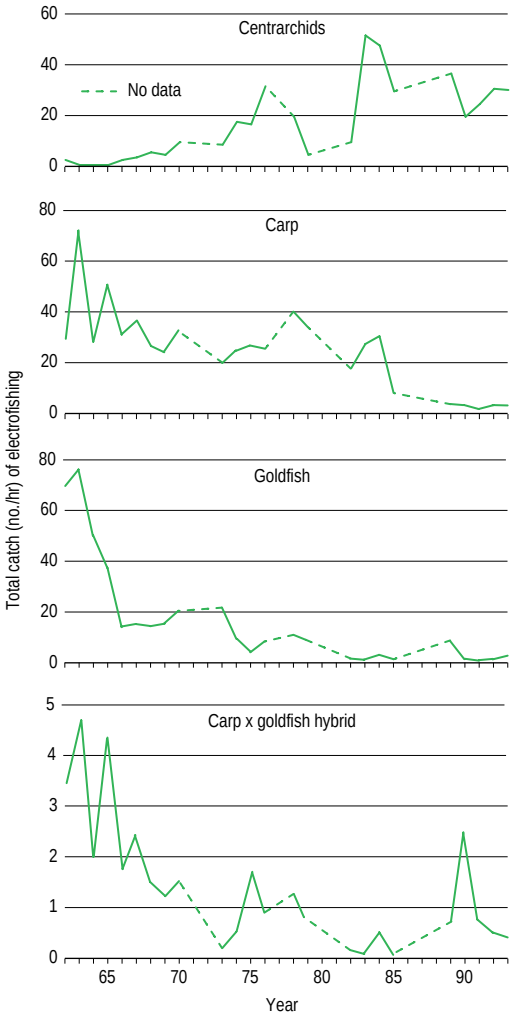

Fig. 2. Number of individual fish obtained per hour of electrofishing from the upper Illinois Waterway for fish species identified as pollution indicators. Dashed lines connect data points for years between which electrofishing was not conducted.

Sediments of the upper Illinois River contain varying amounts of toxic substances (IEPA 1992), which are thought to contribute to the incidence of abnormalities on fishes that forage in sediments while minimally affecting fishes that forage in the water column. To test this hypothesis, all fishes were assigned to one of two groups: benthic species that frequently forage in bottom sediments (e.g., common carp) and pelagic species that usually inhabit the water column (e.g., bluegill).

Substantial changes have occurred in catch rates from the seven upper Illinois Waterway stations between 1962 and 1993 (Fig. 2). Catches of centrarchids have increased (D = 327, $P < 0.001$) since the early 1960's, peaking at 52/hr in 1983. Catches of centrarchids appear to have stabilized during the last 5 years (Fig. 2), indicating their populations may have reached carrying capacity. Catches have decreased for carp ($P < 0.001$), goldfish ($P < 0.001$), and carp x goldfish hybrids ($P < 0.001$) since the early 1960's. Carp were able to maintain their numbers until the mid-1980's (Fig. 2), as larger, older individuals probably died off and smaller, younger individuals were more vulnerable to predation by piscivores. Catches of goldfish declined rather precipitously from 1963 to 1966 for an unknown reason before substantial increases in centrarchids.

Data from 1963 and 1992 were chosen for more detailed examination, those years being representative of catches from early and recent years of the electrofishing survey. In 1963, goldfish accounted for almost one-third of all fish collected per hour, followed by carp, emerald shiner (*Notropis atherinoides*), and gizzard shad (*Dorosoma cepedianum*); together these four species dominated the catch, accounting for 95.8% of all individuals collected per hour (Fig. 3). In 1992, 13 species accounted for 95.4% of all fish collected per hour: emerald shiners were most abundant followed by centrarchids; carp and goldfish were reduced to a minor component (Fig. 3). The increase in centrarchids and decrease in carp and goldfish since the early 1960's (Fig. 2) reflect a more diverse fish community in recent years (Fig. 3).

For all years when data were collected from 1959 to 1993, the percentages of fish with external abnormalities were higher on benthic fishes than on pelagic fishes, suggesting that sediments may contain significant amounts of contaminants. In fact, the IEPA (1992) identified several locations near our electrofishing stations on the upper Illinois and Des Plaines rivers as having sediments that contained elevated levels of toxicants, including mercury, lead, and PCBs. Brown et al. (1973) reported, however, that benthic fishes had a higher frequency of

tumors than pelagic fishes (1.7% and 1.0%, respectively) even when collected from a relatively unpolluted Canadian watershed. Both groups of fishes, though, had higher rates of tumors in the polluted Fox River of northeastern Illinois (benthic fishes, 7.0%; pelagic fishes, 3.0%) than in the Canadian system (Brown et al. 1973). Hughes and Gammon (1987) noted that increasing pollution seems correlated with an increase in the incidence of abnormalities on fishes of the Willamette River in Oregon. Likewise, Tyler and Everett (1993) reported that bottom-dwelling barbel (*Barbus barbus*) collected from polluted rivers in England had a higher incidence of abnormalities than those collected from a clean river. Therefore, the relationship between a high incidence of abnormalities on fish and polluted waters has been well established. On the upper Illinois River, there was a marginal trend of decreasing incidence of abnormalities against years for pelagic fishes since the early 1960's (D = 3,156; $P < 0.05$), coincident with known improvements in water quality over the same period (Butts 1987), but not for benthic fishes (D = 1,937; P = 0.23).

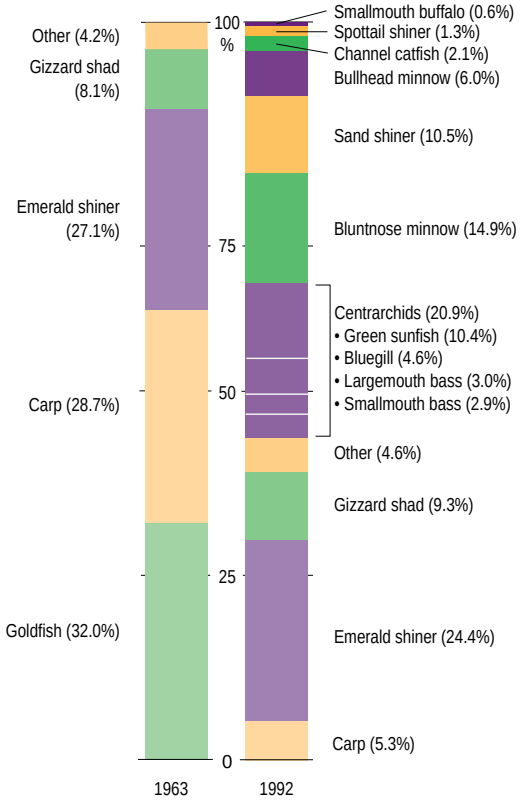

Fig. 3. Composition of catches (%) for the upper Illinois Waterway for 1963 and 1992, based on number of individuals collected per hour of electrofishing.

Conclusions

Long-term trends of fish populations in the upper Illinois River reflect improved water quality in recent years as compared with the early 1960's. This trend is consistent with data presented in other studies that showed improved water quality in the upper Illinois River (Butts 1987; Lerczak et al. 1992). The increased incidence of external abnormalities between bottom-foraging fishes compared with pelagic fishes suggests contaminated sediments (Essig 1991; IEPA 1992).

Because recovery of fish populations in the upper Illinois Waterway appears to be a response to pollution-control efforts, definite restoration goals should be identified to help guide further recovery and to determine expectations. In addition, the specific causes for the high incidence of abnormalities in benthic fishes need to be explicitly identified.

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Contaminant Trends in Great Lakes Fish

by
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The Great Lakes region is home to many large industrialized cities and extensive agricultural areas that produce and use an array of potentially toxic chemicals. Some of these chemicals entering the lakes' food chain have been related to environmental health problems including poor egg-hatching success, reproductive abnormalities, and birth defects in fish, fish-eating birds, and mammals. Tumors and other deformities in some fish and wildlife species are also attributed to exposure to toxic contaminants. In addition, fish consumption advisories are issued annually by the Great Lakes' states and the Province of Ontario for certain fish species and larger sizes of Great Lakes fish that accumulate toxic contaminants.

To measure progress in reducing chemicals in the Great Lakes ecosystem, the National Biological Service's (NBS's) Great Lakes Science Center began a contaminant trend-monitoring program in Lake Michigan in 1969. The program was expanded in 1977 to include all of the Great Lakes and additional species of fish through a cooperative agreement between the NBS Great Lakes Science Center and the U.S. Environmental Protection Agency (USEPA), Great Lakes National Program Office. Fish are sampled for this program from 12 sites. All sites were sampled annually through 1982 and thereafter were divided into odd- and even-year sampling regimes. Results from these long-term monitoring programs are extremely valuable in understanding the dynamics of contaminants, developing predictive models for contaminant trends, and determining the effectiveness of regulatory programs.

This article presents data from the top predators sampled during even years for the NBS/USEPA monitoring program, lake trout (*Salvelinus namaycush*) or walleye (*Stizostedion vitreum vitreum*, Lake Erie only). In addition, information is presented on locations in the Great Lakes where tumors and other deformities in fish have been observed, indicating potentially contaminated sediments.

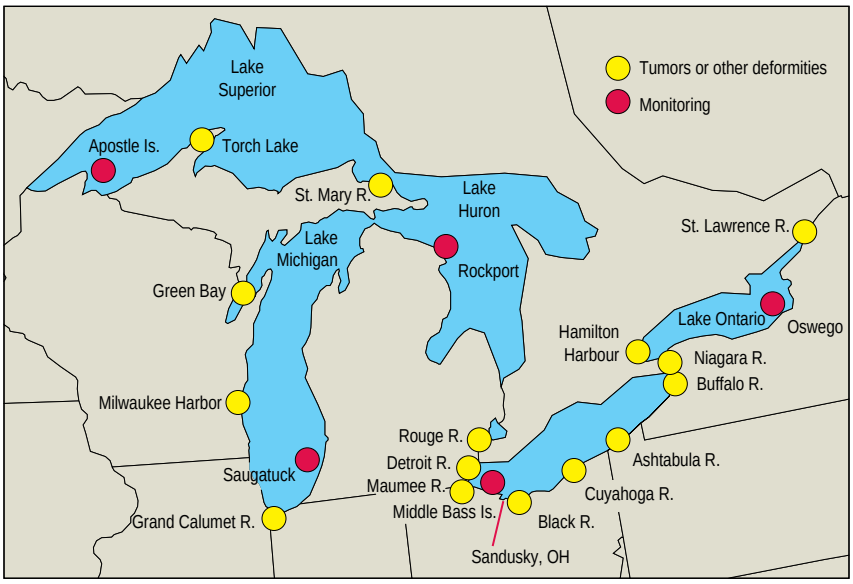

Fig. 1. Sampling sites for the NBS/USEPA Fish Contaminant Monitoring Program and "hot spots" of sediment contamination where tumors and other deformities have been detected in fish.

Methods

Lake trout from 600 to 700 mm (23.6-27.6 in) total length were collected from Lakes Superior, Michigan, Huron, and Ontario from even-year sampling sites by using gill nets (Fig. 1). Walleye from 400 to 500 mm (15.8-19.7 in) total length were collected from Lake Erie near Sandusky, Ohio, by using gill nets (Fig. 1). All fish were stored frozen until analyzed. Fish were prepared for analysis by thawing, compositing fish into five samples, and homogenizing. Contaminants were extracted and separated into nonpolar (polychlorinated biphenyls [PCBs]) and polar (DDT [sum of DDT, DDE, and DDD] and dieldrin) fractions and analyzed by a gas chromatograph equipped with an electron capture detector. Contaminants were reported as total DDT, total PCBs, and dieldrin.

Tumor surveys were conducted by the NBS Great Lakes Science Center and other agencies in highly industrialized rivers and harbors. Most of the work focused on the brown bullhead (*Ameiurus nebulosus*), a bottom-feeding fish especially exposed to tumor-causing chemicals in contaminated sediments.

Contaminant Trends

Results of DDT, PCB, and dieldrin trends during an approximately two-decade period are presented in Figs. 2-6. Data are from DeVault et al. 1985; Hesselberg et al. 1990; and DeVault and Hesselberg, in press. In general, concentrations of contaminants in fish consistently declined until the mid-1980's, but since then the downward trend has leveled off. Similar trends have been observed in fish in Canadian waters of the Great Lakes (Baumann and Whittle 1988).

Lake Michigan

Contaminants were higher in Lake Michigan lake trout than in fish of any of the other Great Lakes. Both total DDT and PCBs declined (Fig. 2), yet total PCBs did not decline after the voluntary control in 1972 but did after the mandatory ban in 1976.

In lake trout dieldrin reached a high in 1978 and a low in 1987 (Fig. 2). Dieldrin is higher in Lake Michigan fish than in fish from the other

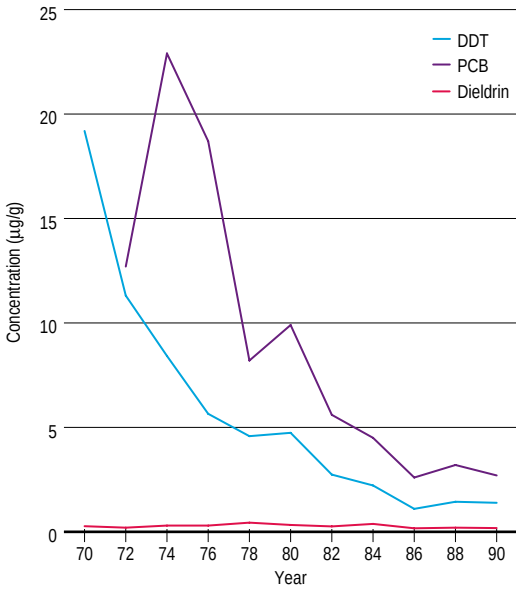

Fig. 2. Contaminant results from Lake Michigan lake trout, 1970-90.

Great Lakes, and changes in fish tissue concentrations do not follow use patterns for reasons that are not well understood.

Lake Superior

Total DDT and PCB concentrations in lake trout from Lake Superior were the lowest of all the Great Lakes and generally declined from 1977 to 1990 (Fig. 3). Dieldrin was always low and varied little from 1977 to 1990. Contaminant concentrations are lowest in Lake Superior because of the low density of agriculture and industry in the lake basin.

Lake Huron

Concentrations of total DDT and PCBs in lake trout from Lake Huron were intermediate between Lake Michigan and Lake Superior. Similar trends of declining concentrations of these chemicals were observed in Lake Huron (Fig. 4). Dieldrin concentrations were similar to Lake Superior but declined from a high in 1979 to a low by 1988. With the exception of the Saginaw Valley, both agriculture and industry are much less developed surrounding Lake Huron than Lake Michigan, thereby resulting in lower contaminants in Lake Huron fish.

Lake Ontario

The contaminants in Lake Ontario fish are relatively high (Fig. 5), second only to Lake Michigan. Trends in total DDT concentrations in lake trout from Lake Ontario were fairly constant from 1977 to 1990. Total PCBs in lake trout declined significantly from a high in 1977 to a low in 1990, a slower decline than in Lake Michigan. The relatively high contaminant concentrations in Lake Ontario fish are a result of the highly urbanized, industrial, and agricultural basin. In addition, it is the lowermost of the Great Lakes, receiving pollutants from upstream through the Niagara River. Dieldrin concentrations in lake trout from Lake Ontario reached a high in 1979 and then declined to a low by 1988.

Lake Erie

Total DDT, PCB, and dieldrin concentrations in Lake Erie walleye (Fig. 6) were lower and more similar to concentrations in lake trout in Lake Superior than those of other Great Lakes. Total DDT and PCBs peaked in 1977 and declined to a low in 1982; no consistent trend was noted for dieldrin. Low concentrations of contaminants in Lake Erie were similar to those in Lake Superior even though Lake Erie is surrounded by the largest urbanized, industrial, and agricultural basin of all the Great Lakes. Lake Erie, however, is the shallowest of all the Great Lakes and contains the highest amount of particulate matter. Contaminants flush more quickly through the shallow lake and are removed from the water column as they adhere to particulate matter and settle to the bottom. These factors work together in reducing the amount of contaminants available to fish in Lake Erie.

Contaminant Effects

Reduced reproductive success in fish-eating birds has been linked with DDT and PCBs (Giesy et al. 1994). As the concentrations of these contaminants have declined, populations of fish-eating birds such as the bald eagle (*Haliaeetus leucocephalus*) are beginning to recover in the Great Lakes basin. In lake trout, PCBs are also linked to reduced egg hatchability and may also be responsible for fry deformities and mortality (Mac et al. 1993). In spite of reductions in PCBs in lake trout in all of the Great Lakes, substantial natural reproduction occurs only in Lake Superior (Mac and Edsall 1991). The role of contaminants and other factors in lake trout reproductive problems in the other four Great Lakes is still under investigation.

Another fish health problem associated with toxic chemicals is found in Great Lakes harbors and tributaries where heavy industry was located (Baumann et al. 1991). Bottom sediments in these areas are heavily contaminated with polycyclic aromatic hydrocarbons (PAHs). Presence of liver tumors and other deformities such as lip papillomas, stubbed barbels, or skin discolorations in bottom-feeding fishes, such as the brown bullhead, have been linked to the

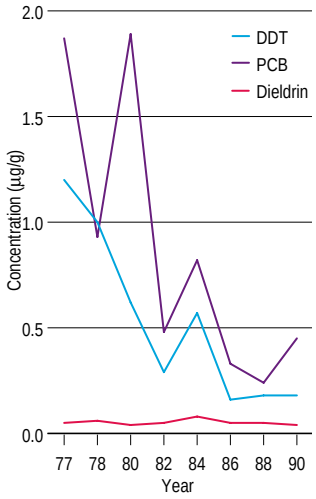

Fig. 3. Contaminant results from Lake Superior lake trout, 1977-90.

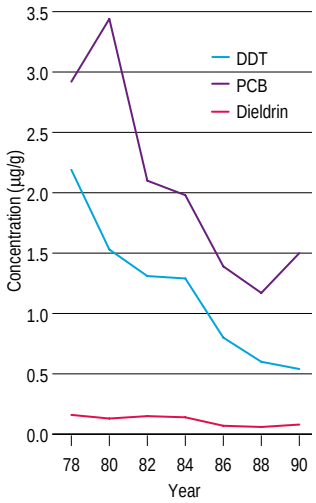

Fig. 4. Contaminant results from Lake Huron lake trout, 1977-90.

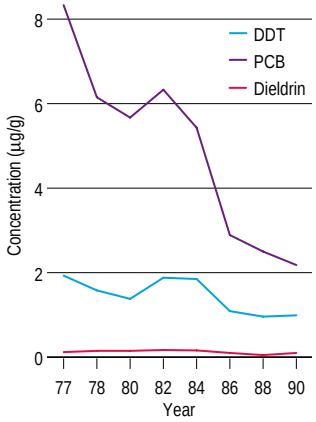

Fig. 5. Contaminant results from Lake Ontario lake trout, 1977-90.

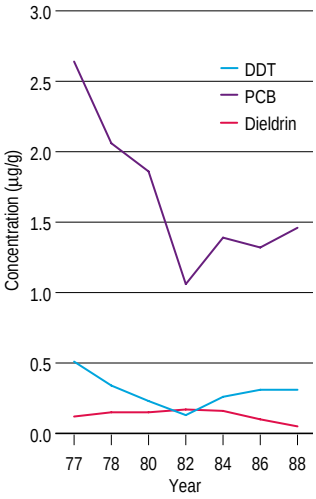

Fig. 6. Contaminant results from Lake Erie walleye, 1977-90.

presence of PAHs in the sediment (Baumann et al. 1991; Smith et al. 1994; Fig. 7). Tumors and other deformities have been detected in 15 locations (Hartig and Mikol 1992; Fig. 1).

Conclusions

The monitoring program for contaminants in Great Lakes fish has documented successful reduction of contaminants in response to usage bans for DDT and PCBs. Trends in dieldrin are less clear and concentrations of this pesticide remain especially high in Lake Michigan in comparison to the other Great Lakes. Fish communities are rebounding in some Great Lakes harbors, tributaries, embayments, and connecting channels that formerly were so contaminated that only the most pollution-tolerant organisms could survive. More reductions in contaminants are required, however. Monitoring results clearly indicate that the downward trend in contaminants leveled off in the mid-1980's, and resource-management agencies and research institutions are investigating the potential to further reduce sources of contamination in Great Lakes fish.

Courtesy Marc Blouin, NBS/GLSC

Fig. 7. Lip tumor and stubbed barbels on a brown bullhead.

Reproductive problems, tumors, and other deformities are still being detected in certain fish and wildlife populations in most of the Great Lakes. Similarly, consumption advisories recommending restrictions on eating certain species and sizes of Great Lakes fish still remain. The United States and Canada have agreed upon a virtual elimination policy for toxic contaminants under the auspices of the Great Lakes Water Quality Agreement. Remedial action plans are being developed by federal and state agencies in cooperation with local municipalities and local citizens to eliminate beneficial use impairments in the most contaminated rivers, harbors, and bays in the Great Lakes. Continued long-term monitoring of contamination in fish is required to determine the success of these programs and to guide where further corrective actions may be necessary.

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Lake Trout in the Great Lakes

by
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Lake trout (*Salvelinus namaycush*) populations in the Great Lakes collapsed catastrophically during the 1940's and 1950's because of excessive predation by the sea lamprey (*Petromyzon marinus*) and exploitation by fisheries. The lake trout was the top-level predator in most of the Great Lakes as well as an important species harvested by commercial fisheries. Interagency efforts to restore lake trout into the Great Lakes included comprehensive control of sea lamprey populations (Smith 1971), regulation of commercial and recreational fisheries, and stocking (Eschmeyer 1968). It was hoped that without sea lamprey predation and fishery exploitation, stocked lake trout could reproduce and eventually restore wild lake trout populations in each of the Great Lakes. Lake trout restoration began during the 1950's in Lake Superior (Hansen et al. 1995), the 1960's in Lake Michigan (Holey et al. 1995), the 1970's in Lake Huron (Eshenroder et al. 1995) and Lake Ontario (Elrod et al. 1995), and the 1980's in Lake Erie (Cornelius et al. 1995).

Long-term monitoring of lake trout populations relied on catch records of commercial fisheries before the populations collapsed. Later monitoring of lake trout populations relied on assessment fisheries to measure the increase in abundance of stocked fish and, subsequently, of naturally produced fish. At present, natural reproduction by lake trout has been widespread only in Lake Superior. In contrast, lake trout reproduced in only limited areas of Lakes Huron, Michigan, and Ontario, and only in Lake Huron have progeny survived to adulthood. We describe the historical collapse and subsequent restoration of lake trout populations in U.S. waters of Lake Superior. We also describe the limited natural reproduction that has occurred in the other Great Lakes.

We compiled data describing abundance trends of lake trout in Lake Superior during 1929-93, expressed as the number of fish caught in a specified length of gill net. Data sources were for Michigan during 1929-49 (Hile et al. 1951), Michigan and Wisconsin during 1950-70 (Pycha and King 1975), and Michigan and Wisconsin during 1970-93 (Hansen et al. 1995).

Fishing was by commercial fishers during 1929-58 and by commercial-fisher contractors or state agencies during 1959-93. Lake trout populations in Lake Superior during 1929-43 sustained stable yields in commercial fisheries, providing a benchmark for judging restoration status. Therefore, lake trout abundance, expressed as a percentage of the 1929-43 average, directly compares lake trout abundance during the various phases of population collapse and recovery. Hatchery lake trout were all marked by removal of one or more fins before stocking. Thus, we show the abundance of stocked lake trout (one or more fins missing) separately from that of wild-origin lake trout (no fins missing). Comparable data are not available for Canadian waters of Lake Superior.

Status and Trends

Lake Superior

Abundance of wild lake trout in Michigan declined from stable levels in the 1930's to nearly zero in the late 1960's (Figure; Hansen et al. 1995). In the 1970's and 1980's, abundance of wild lake trout increased steadily, but in the late 1980's and early 1990's decreased slowly because of increased commercial fishing and sea lamprey predation. The abundance of stocked fish increased in the late 1960's well beyond the 1929-43 average and remained there during most of the 1970's.

Abundance of stocked lake trout declined rapidly in the late 1970's and 1980's and has remained less than 10% of the 1929-43 average since 1988. Stocked lake trout reproduced in the late 1960's and produced an increased abundance of wild fish in the 1970's and 1980's.

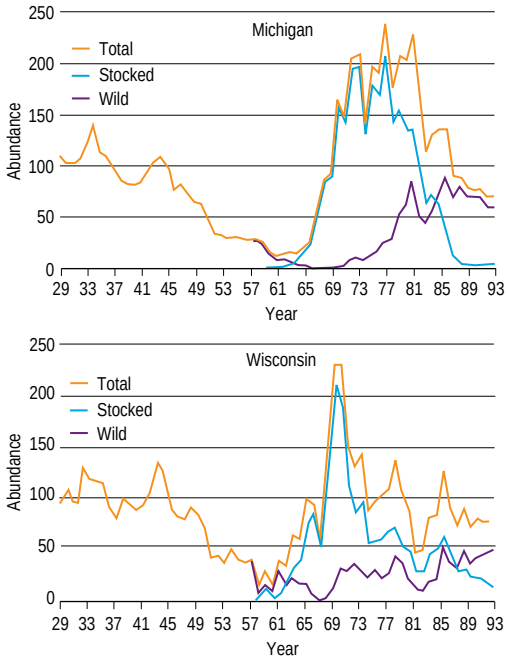

Figure. Abundance of stocked and wild lake trout (*Salvelinus namaycush*) in Michigan and Wisconsin waters of Lake Superior in 1929-93, expressed as a percentage of the 1929-43 mean (from Hansen et al. 1995).

The key to this successful natural reproduction was the presence of abundant inshore spawning grounds that inexperienced stocked lake trout easily located. Also, the decline in abundance of wild lake trout in the late 1970's and 1980's was evidently due to the earlier decline in stocked lake trout. The decline was less severe, however, because of reproduction by wild fish, the progeny of the first stocked spawners.

By 1993, 80%-90% of the lake trout in Michigan were wild, but abundance of wild lake trout was only 61% of the 1929-43 average. Fishery management agencies deferred lake trout restoration in eastern Michigan (Whitefish Bay) so that court-affirmed Native American fisheries could maximize their harvest in that area.

In Wisconsin, abundance of wild lake trout declined irregularly through 1968 and increased after that (Figure; Hansen et al. 1995). The abundance of wild lake trout in Wisconsin, even at its lowest point, remained higher than in Michigan in the late 1960's. Increased abundance in the 1970's was mostly of stocked lake trout, as in Michigan, and peak abundance also greatly exceeded the 1929-43 average. The abundance of stocked lake trout declined earlier than in Michigan, though not as much, and remained at 19% of the 1929-43 average.

Abundance of wild fish in Wisconsin increased irregularly from the 1970's through the early 1990's, but remained lower in 1993 than in Michigan and was only 53% of the

1929-43 average. Stocked lake trout were less important in the restoration of wild lake trout in Wisconsin than in Michigan. Because most spawning reefs in Wisconsin were farther offshore than in Michigan, they were not found by inexperienced stocked spawners. The increased abundance of wild lake trout in Wisconsin was largely due to reproduction by surviving wild fish in the 1960's and 1970's.

Direct measures of historical abundance do not exist for Minnesota or Ontario. The current abundance of lake trout in Minnesota is below that in Michigan and Wisconsin, but in Ontario it is probably similar to Michigan. Lake trout restoration is progressing in Minnesota but is behind that in Michigan or Wisconsin. Patterns of abundance in Minnesota since 1963 are similar to those in Michigan since 1959. Reproduction by stocked lake trout produced increased abundance of wild lake trout in Minnesota, as in Michigan (Hansen et al. 1995). Progress in lake trout restoration in Ontario is sufficient to eliminate stocking in most areas. Excessive fishery exploitation in the Michigan side of Whitefish Bay caused the deferral of lake trout stocking in the Ontario side. This deferral of lake trout restoration will continue until fishery management agencies in Michigan better regulate fishery exploitation.

Lake trout reestablished self-sustaining populations in much of Lake Superior, though few have reached former levels of abundance. Still, most of these populations are sufficiently large to support limited commercial and sport fishing. Current or proposed strategies for restoring wild lake trout in Lake Superior include controlling fishery exploitation, reducing sea lamprey populations, and reducing or eliminating stocking where self-sustaining populations exist.

Lake Michigan

Wild lake trout populations collapsed in Lake Michigan during the 1940's and the species became extirpated in the 1950's (Holey et al. 1995). Stocking began in the 1960's. The abundance of stocked lake trout increased in the late 1970's, then decreased in the northern part of the lake because of excessive fishery exploitation. Scattered evidence of lake trout reproduction, including eggs deposited on spawning grounds and newly hatched juvenile lake trout, has been found since the 1970's, although the only production of wild lake trout more than 1 year old was in Grand Traverse Bay during the late 1970's and early 1980's. Unfortunately, excessive fishery exploitation destroyed the wild lake trout produced in Grand Traverse Bay, preventing the establishment of a self-sustaining population (Holey et al. 1995). Current efforts to restore lake trout to Lake Michigan focus on stocking a variety of lake trout strains in offshore refuges that may afford protection from fishery exploitation, allowing restoration of wild populations to occur.

Lake Huron

Wild lake trout populations collapsed in Lake Huron in the 1940's and the species became extirpated in the main basin in the 1950's (Eshenroder et al. 1995). Stocking began in the 1970's. Abundance of stocked fish in southern Michigan waters increased steadily during the 1970's and 1980's, then decreased in response to reduced stocking. Abundance in northern Michigan waters increased briefly during the late 1970's and early 1980's, but decreased slowly after that because of excessive sea lamprey predation and fishery exploitation.

Natural reproduction occurred in Thunder Bay, Michigan, and South Bay, Ontario, but self-sustaining populations have not developed at either location. Restoration efforts now focus on reducing the number of sea lampreys and stocking a variety of lake trout strains on offshore reefs and in a refuge. The refuge, located in the northern part of the lake, may provide protection from fishery exploitation, and thereby may allow a self-sustaining population to become established.

Lake Erie

Wild lake trout populations collapsed in Lake Erie during the 1920's (Cornelius et al. 1995). Stocking began in the 1980's. Abundance of stocked lake trout increased steadily following initial chemical treatment of sea lampreys in 1986-87, although abundance of stocked lake trout decreased after 1990 for unexplained reasons. Current restoration efforts focus on controlling sea lampreys and stocking yearling lake trout. Research efforts focus on identifying causes of declining abundance of stocked fish and determining whether adult lake trout will aggregate at suitable spawning locations and reproduce successfully.

Lake Ontario

Wild lake trout populations collapsed in Lake Ontario between 1930 and 1960 (Elrod et al. 1995). Stocking began in the 1970's. Stocked lake trout subsequently survived to maturity, spawned, and deposited eggs that hatched into juveniles. These juveniles, however, evidently did not survive to later ages because fishery biologists have not yet discovered any older, wild-origin lake trout. Current restoration efforts focus on stocking strains of lake trout that reproduce most successfully. Research focuses on evaluating factors that limit survival of the fry, such as predation and contaminants.

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Wetlands in Regulated Great Lakes

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Water levels in the Great Lakes are affected by variations in precipitation, evaporation, ice build up, internal waves (seiches), and human alterations that include modifying the connecting channels between lakes and regulating the water levels of Lake Superior and Lake Ontario. Fluctuations in water level promote the interaction of aquatic and terrestrial systems, thereby resulting in higher quality habitat and increased productivity. When the fluctuations in water levels are reduced through stabilization, shifting of vegetation types decreases, more stable plant communities develop, and species diversity and habitat value decrease (Wilcox and Meeker 1991, 1992). Although water levels in Lake Superior are regulated by structures at the outlet, water-level cycles and patterns remain fairly similar to natural conditions. Lake Ontario water levels are also regulated, but high and low water extremes have been eliminated since the mid-1970's. The effects of water-level history on wetland plant communities under the two regulation regimes were investigated by studying wetlands on each lake.

Seventeen sites on Lake Ontario and 18 on Lake Superior were sampled. Vegetation was mapped and then sampled along transects that followed elevation contours with specific water-level histories (number of years since last flooded or last dry). The histories and elevations differed between lakes. Correlations between specific elevations and accompanying plant communities were assessed across all wetlands sampled in each lake to determine the range of elevations in which the most diverse plant communities occur; these data were used to create schematic cross-sections depicting the structural habitat provided by the plant communities characteristic of each lake.

Vegetation and Water Level

At study sites on both Lakes Ontario and Superior, wetland plant communities differed at different elevations; these plant communities developed as a result of the water-level history of each elevation that was sampled. In general, plant communities at elevations that had not been flooded for many years were dominated by shrubs, grasses, and old-field plants. If flooding was more recent, small shrubs that became established after flooding were present, as were grasses, sedges, and other nonwoody plants.

The plant communities at elevations that were flooded periodically at 10- to 20-year intervals and dewatered for successive years between floods had the greatest diversity of

Courtesy J. Meeker, Northland College

Small unnamed bay near Bete Grise, Lake Superior, August 1991. Scattered lilies (*Nuphar variegata*) with submersed plants adjacent to a floating bog mat.

wetland vegetation. Dominants included grasses, sedges, rushes, short emergent plants, and submersed aquatic vegetation. At elevations that were rarely or never dewatered, submersed and floating plants were dominant, with emergent plants also occurring at some sites.

Lake Superior

Water levels on Lake Superior have been regulated for much of this century, although the range of fluctuations and the cyclic nature of high and low lake levels have not been altered

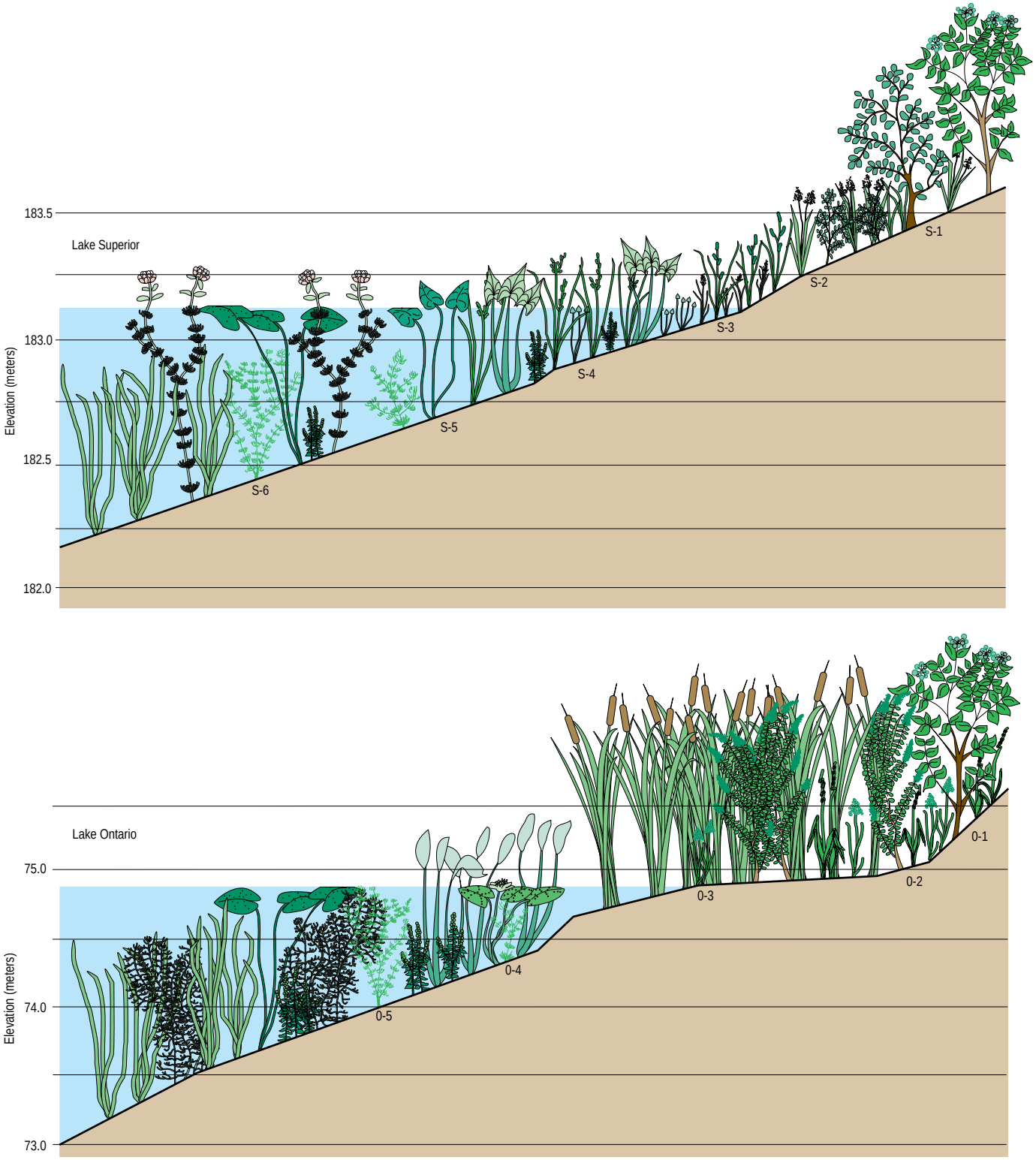

Figure. Schematic cross-sections depicting the structural habitat provided by plant communities characteristic of regulated Lakes Superior and Ontario. Elevations at which vegetation sampling was conducted are shown beneath each cross-section (benchmark: International Great Lakes Datum 1955).

substantially. More than 275 taxa were recorded in a sampling of 18 wetlands along the U.S. shoreline, 216 of which were obligate (*see* glossary) or facultative (*see* glossary) wetland species. Vegetation mapping showed the most prevalent vegetation types to be those dominated by submersed aquatic vegetation or shrubs, both of which were present in all sites and averaged about 25% of the cover. Vegetation types dominated by cattails (*Typha* sp. or other taxa plus cattails) occurred in about half the sites but averaged only about 6% of the cover. Across all sites, 27 different vegetation types were mapped.

Lake Ontario

Water levels on Lake Ontario have been regulated since 1960, when the St. Lawrence Seaway began operation. Before regulation, the range of fluctuations during the 20th century was about 2 m (6.6 ft). After regulation, the range was reduced slightly between 1960 and 1976, but low water-supply conditions in the mid-1960's and high supplies in the mid-1970's maintained much of the range. Regulation reduced the range to about 0.9 m (2.9 ft) in the years after 1976.

The lack of alternating flooded and dewatered conditions at the upper and lower edges of the wetlands resulted in establishment of extensive stands of cattail (*Typha* sp.) and domination of other areas by purple loosestrife (*Lythrum salicaria*), reed canary grass (*Phalaris arundinacea*), and various shrubs. Although more than 250 taxa were recorded in a sampling of 17 wetlands along the U.S. shoreline, only 151 were obligate or facultative wetland plants. Vegetation mapping showed the cattail-dominated vegetation type to be most prevalent, occurring at all sites and averaging about 32% of the cover. The submersed aquatic vegetation type occurred at 75% of the sites and averaged about 30% of the cover. Across all sites, 20 different vegetation types were mapped.

Habitat Structure

Differences in the species and structural types of plants at different elevations in wetlands of regulated Lakes Superior and Ontario result in different habitats for faunal organisms because the greater diversity of taxa and vegetation types in Lake Superior wetlands provides more niches for fauna than in Lake Ontario wetlands (Figure; Engel 1985; Wilcox and Meeker 1992). The prevalence of dominant cattail stands in Lake Ontario wetlands reduces habitat value there (Weller and Spatcher 1965).

Periodic high waters are necessary to reduce dominant emergent vegetation in Great Lakes wetlands; low waters are necessary to reduce dominant submersed vegetation. High waters followed by low-water years allow a diversity of plants to grow from seed on the exposed sediments, reproduce, and replenish the seed bank. Although competitive species such as cattails will again become dominant, the next high-water year will eliminate them again. When water-level fluctuations are reduced by regulation, the processes for rejuvenating wetland plant communities are lost and habitat values decrease.

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Decline in the Freshwater Gastropod Fauna in the Mobile Bay Basin

The historical freshwater gastropod fauna of the Mobile Bay basin in Alabama, Georgia, Mississippi, and Tennessee was the most diverse in the world, comparable only to the diversity reported for the Mekong River in Southeast Asia. This fauna was represented by 9 families and about 118 species. Several families have genera endemic to the Mobile Bay basin: Viviparidae: *Tulotoma*; Hydrobiidae: *Clappia*, *Lepyrium*; Pleuroceridae: *Gyrotoma*; and Planorbidae: *Amphigyra* and *Neoplanorbis*. The greatest described species diversity was in the Pleuroceridae (76 species). The pleurocerid genera *Pleurocera*, *Leptoxis*, and *Elimia* had their greatest radiation in the Coosa River drainage.

Although this extremely diverse aquatic gastropod fauna received little attention in the past 50 years, it was actively studied during the second quarter of this century (Goodrich 1922, 1924, 1936, 1944a, 1944b). During the last 60 years, this unique gastropod fauna has declined precipitously (Table 1; Athearn 1970; Heard 1970; Stansbery 1971). More recent documentation of the decimation of this fauna was presented by Stein (1976) and Palmer (1986). The endemic genus *Tulotoma* (Figs. 1 and 2), formerly widespread in the main channel of the Alabama and Coosa rivers, was presumed extinct until recently rediscovered (Hershler et al. 1990). The pleurocerid genus *Gyrotoma*, restricted primarily to the shoals of the Coosa

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River, contained six recognized species, all of which are presumed extinct (Table 2; Fig. 3).

Status and Trends

Literature records were compiled to document the gastropod species present historically. Recent surveys of the aquatic gastropod fauna of the Coosa and Cahaba river drainages in Alabama have been conducted by using standard field techniques (Bogan and Pierson 1993 a, b). Additional unpublished data (Bogan and Hartfield) are included.

Recent surveys of the aquatic gastropod fauna at about 800 sites (Table 1) have documented population declines, decreases in species' ranges, and the loss of a major portion of the gastropod diversity, especially in the Coosa River. The Coosa River drainage had at least 82 species historically (Table 1); today 26 species are presumed extinct in six genera, and four genera (*Clappia* [2 species], *Gyrotoma* [6 species], *Amphigyra* [1 species], and *Neoplanorbis* [4 species]) are presumed extinct (Tables 1 and 2). The genus *Leptoxis* has been reduced to a single species restricted to three creek tributary systems in the Coosa River.

The fauna of the Cahaba River drainage has fared much better (Table 1). Although the Cahaba River drainage does not suffer from the numerous dams and the siltation problems of the Coosa River drainage, it is heavily affected by nonpoint-source runoff, siltation, acid mine drainage, pollution from wastewater treatment

Courtesy J.M. Pierson

Fig. 1. Live specimens of the endangered tulotoma, *Tulotoma magnifica*, from Kelly Creek, Elmore County, Alabama, 1993.

Table 1. Summary of the aquatic gastropod fauna of the river systems in the Mobile Bay basin.

Data*	Alabama River	Tombigbee R. drainage	Black Warrior R. drainage	Cahaba R. drainage	Coosa R. drainage	Talapoosa R. drainage	Mobile Bay basin total
Approximate total historical gastropod species diversity	19	8	17	36	82	8	118
Number of species found in recent surveys	3	3	7	24	30	4	80
Federally listed endangered species	1	0	0	0	1	0	1
Federal candidate species	4	1	6	16	43	2	70
Number of species presumed extinct	?	0	2	4	26	?	38
Percent decline in gastropod fauna	84%	62%	58%	33%	63%	50%	32%

* Data from Bogan and Pierson (1993 a,b), Burch (1989), and A.E. Bogan and P. Hartfield (unpublished data).

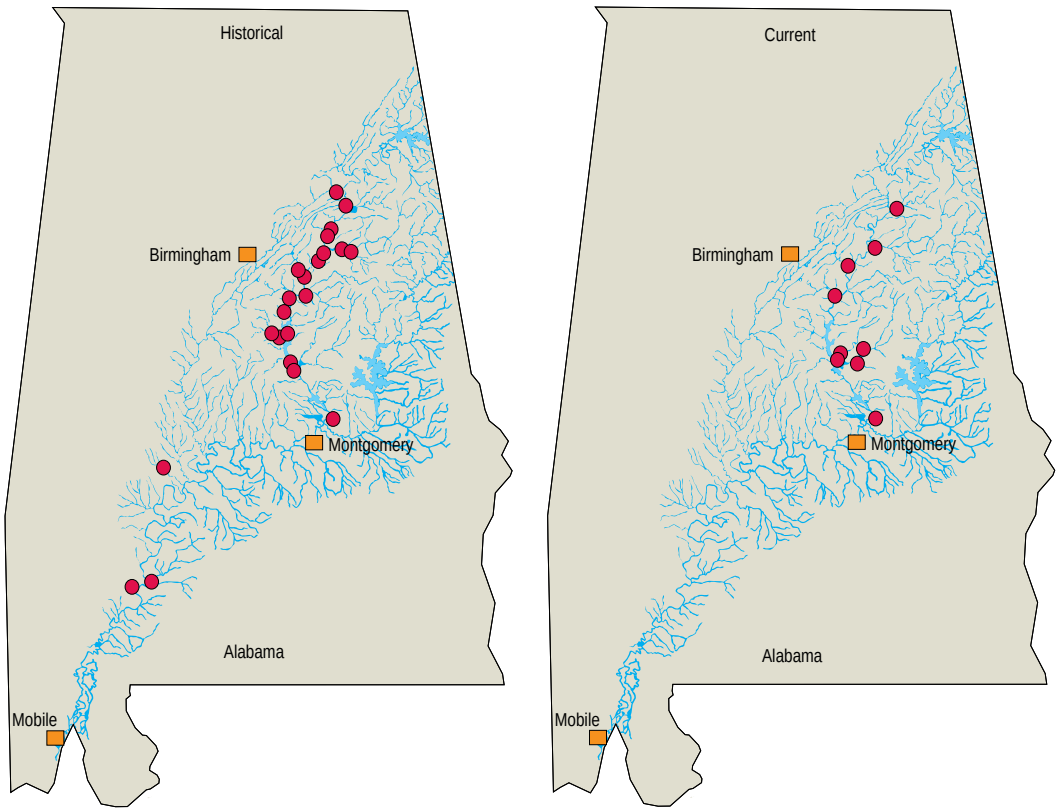

Fig. 2. Historical and current distribution of *Tulotoma magnifica*. Filled circles represent a single or two closely located collection sites (after Hershler et al. 1990). Map modified from the U.S. Geological Survey 1:500,000 scale—State of Alabama sheet (1970 ed.).

Table 2. Freshwater gastropod species presumed extinct in the Mobile Bay basin.

Family and common name	Scientific name
Hydrobiidae	
Cahaba pebblesnail	*Clappia cahabensis* Clench 1965
Umbilicate pebblesnail	*C. umbilicata* (Walker 1904)
Pleuroceridae	
Short-spire elimia	*Elimia brevis* (Reeve 1860)
Closed elimia	*E. clausa* (Lea 1861)
Fusiform elimia	*E. fusiformis* (Lea 1861)
No common name	*E. gibbera* (Goodrich 1922)
High-spired elimia	*E. hartmaniana* (Lea 1861)
Constricted elimia	*E. impressa* (Lea 1841)
Hearty elimia	*E. jonesi* (Goodrich 1936)
No common name	*E. lachryma* (Reeve 1861)
Ribbed elimia	*E. laeta* (Jay 1839)
No common name	*E. macglameriana* (Goodrich 1936)
Rough-lined elimia	*E. pilsbryi* (Goodrich 1927)
Pupa elimia	*E. pupaeformis* (Lea 1864)
Pygmy elimia	*E. pygmaea* (H.H. Smith 1936)
Cobble elimia	*E. vanuxemiana* (Lea 1843)
Excised slitshell	*Gyrotoma excisa* (Lea 1843)
Striate slitshell	*G. lewisii* (Lea 1869)
Pagoda slitshell	*G. pagoda* (Lea 1845)
Ribbed slitshell	*G. pumila* (Lea 1860)
Pyramid slitshell	*G. pyramidata* (Shuttleworth 1845)
Round slitshell	*G. walkeri* (H.H. Smith 1924)
Agate rocksnail	*Leptoxis clipeata* (H.H. Smith 1922)
Oblong rocksnail	*L. compacta* (Anthony 1854)
Interrupted rocksnail	*L. formanii* (Lea 1843)
Maiden rocksnail	*L. formosa* (Lea 1860)
Rotund rocksnail	*L. ligata* (Anthony 1860)
Lirate rocksnail	*L. lirata* (H.H. Smith 1922)
Black mudalia	*L. melanoides* (Conrad 1834)
Bigmouth rocksnail	*L. occultata* (H.H. Smith 1922)
Coosa rocksnail	*L. showalteri* (Lea 1860)
No common name	*L. torrefacta* (Goodrich 1922)
Striped rocksnail	*L. vittata* (Lea 1860)
Planorbidae	
Shoal sprite	*Amphigyra alabamensis* (Pilsbry 1906)
No common name	*Neoplanorbis carinatus* (Walker 1908)
No common name	*N. smithi* (Walker 1908)
No common name	*N. tantillus* (Pilsbry 1906)
No common name	*N. umbilicatus* (Walker 1908)

plants, and water withdrawn for domestic water use. Species such as *Lepyrium showalteri* and *Lioplax cyclostomaformis*, formerly much more widespread in the basin, are now apparently restricted to one or two shoal areas in the Cahaba River main channel. The status of the pebblesnails (Hydrobiidae) is uncertain. The former diversity of the genus *Somatogyrus* in the Coosa River has probably suffered the same fate as most of the main channel shoal-dwelling pleurocerid species—extinction. Detailed information on the distribution of the freshwater limpets (Ancylidae) is not available, but they appear to have suffered similar range restrictions.

The uncertainty expressed in the diversity of the historical gastropod fauna presented in Table 1 is indicative of our lack of information regarding all aspects of the historical gastropod fauna of the Mobile Bay basin. There are a lack of detailed data on the ecology and life history of all of the species, and a paucity of distributional information for most of the families other than the Pleuroceridae, making estimation of gastropod diversity by drainage difficult.

Courtesy A.E. Bogan

Fig. 3. Illustration of a representative species of the extinct slitshell genus *Gyrotoma* from Butting Ram Shoals, Coosa River, Alabama.

Declining species diversity can be directly linked to the inundation of the shoal areas of the rivers of the Mobile Bay basin by impoundment and siltation resulting from a variety of watershed disturbances, including 33 major dams for hydroelectric generation, locks and flood control on the major rivers of the Mobile Bay basin, and numerous smaller impoundments on tributary rivers and streams. Most gastropods inhabiting shoal areas are gill-breathing species typically grazing on the plant life growing on the rock substrate in shallow riffle and shoal areas. They formerly lived on rocks in the shallow shoal areas with highly oxygenated water. The pleurocerid gastropod fauna represented a significant portion of the invertebrate biomass living on these shoal areas.

When this habitat was impounded, the snails were not able to survive the deep, cold, and often oxygen-depleted water. Many areas not impounded have suffered because of the heavy siltation of shoal areas, smothering the plant life that formed the diet of these gastropods. Major sources of siltation include poor agricultural and silvicultural practices, lack of riparian buffer zones, and generally poor land-use practices. The drastic decline in gastropod diversity is especially evident in the Coosa River main channel where numerous species formerly found on shoals have disappeared after the damming of the river (Bogan and Pierson 1993a). Other species have had their ranges fragmented by the damming of the rivers and have become restricted to the unimpounded areas below the dams with clean current-swept gravel and bedrock outcrops.

Tulotoma magnifica (Figs. 1 and 2) is the only aquatic gastropod now federally listed as endangered; none is listed as threatened, although 104 species of aquatic gastropods from Alabama are on the federal candidate list. Most are from the Coosa and the Cahaba rivers (Table 1). Conservation and recovery of the remaining diversity will require immediate action to prevent further declines and extinctions. This will necessitate action to improve water quality across the basin and to decrease the amount of silt entering the streams and rivers. In addition, the survey of the aquatic gastropod fauna of the Mobile Bay basin is not complete, and additional fieldwork in the main channels of the larger rivers is needed, especially on the vertical limestone wall habitats.

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Protozoa

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The diverse assemblage of organisms that carry out all of their life functions within the confines of a single, complex eukaryotic *(see* glossary) cell is called protozoa. *Paramecium*, *Euglena*, and *Amoeba* are well-known examples of these major groups of organisms. Some protozoa are more closely related to animals, others to plants, and still others are relatively unique. Although it is not appropriate to group them together into a single taxonomic category, the research tools used to study any unicellular organism are usually the same, and the field of protozoology has been created to carry out this research. The unicellular photosynthetic protozoa are sometimes also called algae and are addressed elsewhere. This report considers the status of our knowledge of heterotrophic protozoa (protozoa that cannot produce their own food).

Free-living Protozoa

Protozoans are found in all moist habitats within the United States, but we know little about their specific geographic distribution. Because of their small size, production of resistant cysts, and ease of distribution from one place to another, many species appear to be cosmopolitan and may be collected in similar microhabitats worldwide (Cairns and Ruthven 1972). Other species may have relatively narrow limits to their distribution.

Marine ciliates inhabit interstices of sediment and beach sands, surfaces, deep sea and cold Antarctic environments, planktonic habitats, and the algal mats and detritus of estuaries and wetlands. Our actual knowledge of salinity, temperature, and oxygen requirements of marine protozoa is poor (although some groups, such as the foraminifera, are better studied than others), and even the broadest outlines of their biogeographic ranges are usually a mystery. In general, freshwater protozoan communities are similar to marine communities except the specialized interstitial fauna of the sand is largely missing. In freshwater habitats, the foraminifera and radiolaria common in marine environments are absent or low in numbers while testate amoebae exist in greater numbers. Relative abundance of species in the marine versus freshwater habitat is unknown.

Soil-dwelling protozoa have been documented from almost every type of soil and in every kind of environment from the peat-rich soil of bogs to the dry sands of deserts. In general, protozoa are found in greatest abundance near the soil surface, especially in the upper 15 cm (6 in), but occasional isolates can be obtained at depths of a meter (yard) or more. Protozoa do not constitute a major part of soil biomass, but in some highly productive regions such as forest litter, the protozoa are a significant food source for the microinvertebrates, with a biomass that may reach 20 g/m^2 of soil surface area there.

Environmental Quality Indicators

Polluted waters often have a rich and characteristic protozoan fauna. The relative abundance and diversity of protozoa are used as indicators of organic and toxic pollution (Cairns et al. 1972; Foissner 1987; Niederlehner et al. 1990; Curds 1992). Bick (1972), for example, provided a guide to ciliates that are useful as indicators of environmental quality of European freshwater systems, along with their ecological distribution with respect to parameters such as amount of organic material and oxygen levels. Foissner (1988) clarified the taxonomy of European ciliates as part of a system for classifying the state of aquatic habitats according to their faunas.

Symbiotic Protozoa

Parasites

Protozoa are infamous for their role in causing disease, and parasitic species are among the best-known protozoa. Nevertheless, our knowledge has large gaps, especially of normally free-living protozoa that may become pathogenic in immuno-compromised individuals. For example, microsporidia comprise a unique group of obligate, intracellular parasitic protozoa. Microsporidia are amazingly diverse organisms with more than 700 species and 80 genera that are capable of infecting a variety of plant, animal, and even other protist hosts. They are found worldwide and have the ability to thrive in many ecological conditions. Until the past few years, their ubiquity did not cause a threat to human health, and few systematists worked to describe and classify the species. Since 1985, however, physicians have documented an unusual rise in worldwide infections in AIDS patients caused by four different genera of microsporidia (*Encephalitozoon*, *Nosema*, *Pleistophora*, and *Enterocytozoon).* According to the Centers for Disease Control in the United States, difficulties in identifying microsporidian species are impeding diagnosis and effective treatment of AIDS patients.

Protozoan Reservoirs of Disease

The presence of bacteria in the cytoplasm of protozoa is well known whereas that of viruses is less frequently reported. Most of these reports simply record the presence of bacteria or viruses and assume some sort of symbiotic relationship between them and the protozoa. Recently, however, certain human pathogens were shown to not only survive but also to multiply in the cytoplasm of free-living, nonpathogenic protozoa. Indeed, it is now believed that protozoa are the natural habitat for certain pathogenic bacteria. To date, the main focus of attention has been on the bacterium *Legionella pneumophila*, the causative organism of Legionnaires' disease; these bacteria live and reproduce in the cytoplasm of some free-living amoebae (Curds 1992).

Symbionts

Some protozoa are harmless or even beneficial symbionts. A bewildering array of ciliates, for example, inhabit the rumen and reticulum of ruminates and the cecum and colon of equids. Little is known about the relationship of the ciliates to their host, but a few may aid the animal in digesting cellulose.

Data on Protozoa

Bibliography

While our knowledge of recent and fossil foraminifera in the U.S. coastal waterways is systematically growing, other free-living protozoa are poorly known. There are some regional guides and, while some are excellent, many are limited in scope, vague on specifics, or difficult to use. Largely because of these problems, most ecologists who include protozoa in their studies of aquatic habitats do not identify them, even if they do count and measure them for biomass estimates (Taylor and Sanders 1991).

Parasitic protozoa of humans, domestic animals, and wildlife are better known although no attempt has been made to compile this information into a single source. Large gaps in our knowledge exist, especially for haemogregarines, microsporidians, and myxosporidians (*see* Kreier and Baker 1987).

Museum Specimens

For many plant and animal taxa, museums represent a massive information resource. This is not true for protozoa. In the United States, only the National Natural History Museum (Smithsonian Institution) has a reference collection preserved on microscope slides, but it does not have a protozoologist curator and cannot provide species' identification or verification services. The American Type Culture Collection has some protozoa in culture, but its collection includes relatively few kinds of protozoa.

Ecological Role of Protozoa

Although protozoa are frequently overlooked, they play an important role in many communities where they occupy a range of trophic levels. As predators upon unicellular or

filamentous algae, bacteria, and microfungi, protozoa play a role both as herbivores and as consumers in the decomposer link of the food chain. As components of the micro- and meiofauna, protozoa are an important food source for microinvertebrates. Thus, the ecological role of protozoa in the transfer of bacterial and algal production to successive trophic levels is important.

Factors Affecting Growth and Distribution

Most free-living protozoa reproduce by cell division (exchange of genetic material is a separate process and is not involved in reproduction in protozoa). The relative importance for population growth of biotic versus chemical-physical components of the environment is difficult to ascertain from the existing survey data. Protozoa are found living actively in nutrient-poor to organically rich waters and in fresh water varying between 0°C (32°F) and 50°C (122°F). Nonetheless, it appears that rates of population growth increase when food is not constrained and temperature is increased (Lee and Fenchel 1972; Fenchel 1974; Montagnes et al. 1988).

Comparisons of oxygen consumption in various taxonomic groups show wide variation (Laybourn and Finlay 1976), with some aerobic forms able to function at extremely low oxygen tensions and to thereby avoid competition and predation. Many parasitic and a few free-living species are obligatory anaerobes (grow without atmospheric oxygen). Of the free-living forms, the best known are the plagiopylid ciliates that live in the anaerobic sulfide-rich sediments of marine wetlands (Fenchel et al. 1977). The importance of plagiopylids in recycling nutrients to aerobic zones of wetlands is potentially great.

Ecological Interactions

Because of the small size of protozoa, their short generation time, and (for some species) ease of maintaining them in the laboratory, ecologists have used protozoan populations and communities to investigate competition and predation. The result has been an extensive literature on a few species studied primarily under laboratory conditions. Few studies have been extended to natural habitats with the result that we know relatively little about most protozoa and their roles in natural communities. Intraspecific competition for common resources often results in cannibalism, sometimes with dramatic changes in morphology of the cannibals (Giese 1973). Field studies of interspecific competition are few and most evidence for such species interactions is indirect (Cairns and Yongue 1977).

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Marine and Freshwater Algae

by
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Algae are an extremely diverse group of photosynthetic organisms that range from single-celled organisms to complex thalli (e.g., kelps). Benthic algae live attached to the bottom of a water body or to living or nonliving objects on the bottom. Planktonic algae live free-floating in the ocean and in the largest to smallest lakes and streams. Algae also occur in such varied places as the surface layers of soils and porous rocks, on the bark and leaves of trees, in snow, hot springs, and in symbiotic association with fungi to form lichens.

These organisms are important as primary producers (representing the base of the food chain or pyramid), in contributing to the fertility of soil, in providing substrate for other organisms, and in defining aquatic environments such as kelp beds and algal reefs.

The toxicity of certain marine unicellular algae can limit coastal marine fisheries (e.g., dinoflagellates in red tide). In fresh water, blooms tied to nutrient enrichment are often a major nuisance. A few species of macrophytic algae (large enough to be seen by the naked eye) are harvested from the wild for food and industrial purposes.

Knowledge of the algae of the United States is not uniform across various groups or environments. Some modern regional floras, or lists of plants (e.g., California, southeastern coast, gulf coast), are available for marine benthic macroscopic algae (Dawes 1974; Abbott and Hollenberg 1976; Schneider and Searles 1991), of which there are approximately 900 species on the Pacific coast and fewer on the Atlantic and gulf coasts (approximately 450 for the northern Atlantic coast, 350 for the southeastern Atlantic coast, and 300 for the gulf coast). Local floras are available for many places. Few species are shared between the Atlantic and Pacific coasts. Information about marine microalgae is less accessible.

Local and regional floras are available for some groups of freshwater algae (e.g., Hoshaw and McCourt 1988; Dillard 1989; Johansen 1993), but information is absent or has not been compiled for much of the country. Because no attempt has been made to produce a national flora of freshwater algae in this century, it is not possible to estimate the number of such species. Many groups of algae are cosmopolitan, however, and European monographs and floras can be useful.

In general, distribution, status, and trends of algae, even of conspicuous marine algae, are not well established. Floras usually provide ranges, but distribution of many species may be discontinuous, with various causes for the discontinuity. Filling the gaps (or confirming the discontinuities) will require considerable effort.

Although nationwide data on status and trends of North American algal populations are not readily available, scientists do know that a great deal of formerly aquatic habitat has become unavailable for algae because of landfill, reclamation, and water diversion. In addition, other habitat has been altered through farming and municipal and industrial waste discharge. In the case of reservoirs, however, one kind of aquatic habitat has been replaced by another.

Long-term information about phytoplankton is available for the Great Lakes; this information has allowed documentation of water-quality improvement in Lake Erie and analysis of the effect of the invasion of the zebra mussel (*Dreissena polymorpha*; Makarewicz 1993; Nicholls 1993). Much limnological information is available for individual water bodies or catchment basins (e.g., Brock 1985 for Lake Mendota in Wisconsin), but reconciling the different methods used when comparing separate studies is a challenge.

Interpretation of marine baseline and trend data is complicated by differences in communities over time and space (Foster et al. 1988). An example of the utility of marine baseline studies is the census of algae along the coast near Los Angeles (Dawson 1959) that showed how sewage discharge reduced algal diversity. Subsequent resurveys (Widdowson 1971) demonstrated some improvement after stricter environmental regulations were enacted. Long-term studies are available for giant kelp (*Macrocystis pyrifera*), the economically important component of southern California kelp beds. North (1971) and Foster and Schiel (1985) documented the decline of kelp beds after sewage was discharged into the ocean. They also discussed the partly successful attempts at remediation, which involved transplantation and predator control and which led to an appreciation of the complexity of organismal interactions in kelp beds.

Achieving a uniform estimate of the status of algae in North America will take considerable original observation and collection. Furthermore, different research approaches will be necessary for freshwater versus marine algae and for macrophytic algae versus microphytic algae. To determine status and trends of marine macroalgae, published literature must be compiled and analyzed. In addition, unpublished information should be obtained from herbaria and from private collections in the form of specimens, labels, and collectors' notebooks, illustrations, and checklists.

This process has been followed for west coast algae in a project by T. DeCew, the results of which are available at the Herbarium of the University of California. This project condenses

the 100-year record of west coast phycology (study of algae) by using a literature review, compilation of data from specimens at west coast herbaria, and original observations. For each species a tabular representation of geographic and hydrographic range is provided. Presence or absence in different precincts along the coast and details of phenology (relations between climate and periodic biological phenomena), such as reproductive state throughout the year, are indicated. The study gives ecological information such as requirements for substrate and exposure to waves as well as the presence of epiphytes and parasites. In addition, illustrations and references to pertinent taxonomic, chemical, ecological, genetic, and physiological literature are given.

If this kind of project is done on a national scale, workers must have the necessary taxonomic training and herbarium resources must be preserved. About 100 American scientists have algal taxonomy as a principal area of interest (Anonymous 1992). Modern molecular taxonomic methods aid in the study of some groups of algae, but to progress toward a national inventory, traditional taxonomic methods must be supported.

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Freshwater Diatoms: Indicators of Ecosystem Change

by
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Diatoms are photosynthetic unicellular organisms. They are found in almost all aquatic and semi-aquatic habitats and are of great ecological importance because they form an important part of the base of the food web. Although diatoms are widely distributed as a group, most species occur only in habitats with specific physical, chemical, and biological characteristics. Ecologists have long made practical use of this habitat specificity by collecting and analyzing individual species and community data to determine the quality or condition of aquatic habitats. Both long-term monitoring of specific lake and stream habitats and analysis of diatom remains (that become part of the sedimentary record of lakes) allow scientists to obtain a unique long-term historical perspective on these ecosystems. This perspective is especially valuable in assessing the long-term effects of human activities on aquatic and terrestrial ecosystems. Diatoms have been studied throughout the country, but no reasonably complete compilation or summary of these studies exists.

Diatoms are divided into two groups based on overall symmetry of the cell walls; radially symmetrical forms are informally called “centric” diatoms while bilaterally symmetrical forms are referred to as “pennate” diatoms. One remarkable aspect of these organisms is that they have cell walls made of glass (silicon dioxide). The glass cell walls are perforated and ornamented with many holes, which are usually arranged in definite patterns. The nature of these perforations as well as their orientation and densities help in the identification of diatom species. Diatom cell walls come in two pieces that fit together the way a Petri dish or pill box does. When these organisms divide, each half reproduces a “daughter” half that, because of the rigidity of the glass walls, must be smaller than the original half.

Despite the important roles diatoms play in aquatic ecosystems and their utility in evaluating and monitoring environmental change in these systems, intensive floristic or taxonomic studies on freshwater diatoms in North America have been limited. A two-volume work entitled

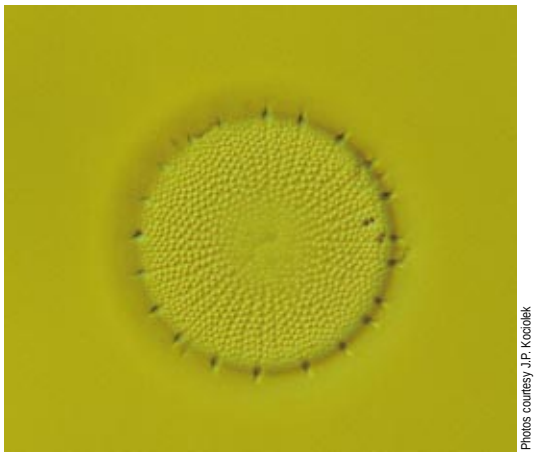
The Diatoms of the United States (Patrick and Reimer 1966, 1975) considered a selected number of genera, and in those genera treated only those species reported from the United States up to 1960. There are only a few regional or statewide taxonomic treatments of diatoms in the United States. The focus has been on specific habitats; areas receiving the most attention have been the Northeast, upper Midwest, the Great Lakes, and isolated areas in the West. Only a few checklists of diatom taxa exist.

Fifteen centric and 63 pennate diatom genera have been reported from fresh water. No exact species counts have been made, but about 4,000 species have been described in the literature. This number is undoubtedly a conservative estimate because in two areas where intensive research has been conducted, in Dickinson County, Iowa (around the Iowa Lakeside Laboratory), and the Laurentian Great Lakes, about 1,200 and 2,000 species, respectively, have been recognized. In the Great Lakes, nearly 10% of those species are new to science. There is still a great need to document the variety and distribution of freshwater diatoms in the United States.

Diatom assemblages provide the basis for many important assessments of trends in the status of freshwater ecosystems. These versatile indicators tell us about the acidification (*see* glossary) of lakes caused by acidic deposition, the eutrophication (*see* glossary) of lakes caused by human impacts and changing land use, improvements and declines in the quality of our rivers and streams, and changes in climate over the past thousands of years. Because diatoms are important components of the biological community and food web and are sensitive to changes in water quality, they provide information on both the biological integrity of the ecosystem and those factors likely to be causing any observed changes. Researchers are rapidly developing new techniques for using diatoms to provide even more quantitative and accurate inferences of ecosystem condition, and diatoms are being included in a growing number of local and regional-scale monitoring programs.

Lake Acidification

The extent, magnitude, timing, and causes of lake acidification in acid-sensitive regions of the country have been inferred from analysis of diatom assemblages in the stratigraphic record of dated lake sediment cores. These paleolimnological studies show, for example, that about 25%-35% of the lakes in the Adirondack Mountains with the lowest ability to neutralize acids (acid neutralizing capacity < 400 μeq/L) have become more acidic since preindustrial



Photos courtesy J.P. Kociolek

Examples of diatoms (top to bottom):

Aulacoseira sp.,
Tabellaria sp.,
Gomphonema sp., and
Stephananodiscus sp.

times (Cumming et al. 1992). Lakes in other regions of the country have also acidified but not to the same extent (Charles et al. 1989). The amount of acidification inferred from diatoms is related to the level of atmospheric loading of strong acids and the ability of watersheds to neutralize those acids. Analysis of diatoms and sedimentary remains of other biological groups (e.g., chrysophytes, chironomids, Cladocera) reveals that acidic deposition has had significant effects on aquatic communities in many lakes. Numbers of taxa are reduced, but some acid-tolerant taxa have significantly increased in abundance.

Lake Eutrophication

Population estimates of the numbers of lakes in New England and New York that are more eutrophic now than in presettlement times are being obtained from analyses of diatom assemblages from recent and preindustrial levels of sediment cores taken as part of the U.S. Environmental Protection Agency's Surface Water component of the Environmental Monitoring and Assessment Program (EMAP; Dixit and Smol 1994). The approach of examining lake eutrophication by using diatom assemblages has been widely applied in North America and throughout the world.

Rivers and Streams

Many long-term diatom data sets exist that can inform us about trends in water quality. The monitoring program conducted by the Federal Water Pollution Control Agency in the 1960's tracked the status of major rivers throughout the country (Williams and Scott 1962). Monitoring of diatom assemblages in rivers and streams is just beginning as part of the U.S. Geological Survey's National Water Quality Assessment (NAWQA) and of the Environmental Monitoring and Assessment Program. The Academy of Natural Sciences of Philadelphia has long-term records for several rivers in the eastern United States. Many of these records show that the quality of water downstream from industrial effluent outfalls and sewage treatment plants has improved markedly, but others show worsening conditions, often due to the increased number of sources of stress along the river or in the watershed. Much more could be learned about trends by simply analyzing the immense data that already exist, especially by using new quantitative techniques developed in the past 5-10 years.

Climate Change

Diatom assemblage composition is sensitive to changes in water level, salinity, ice cover, wind-mixing patterns, and other characteristics directly and indirectly affected by climate. Paleolimnological studies of sediment cores are providing valuable data on climate change over the past hundreds to thousands of years, which are essential for understanding the nature and magnitude of ecosystem change that can be expected in future years.

Conclusions

The ability to infer ecosystem status and trends from diatoms is largely dependent on the availability of ecological data for the species occurring at study sites. The amount of such data is accumulating at an increasingly rapid rate, but it is in many separate data bases. These need to be coordinated so that users will have easier access to the data that already exist.

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