9.08
Worldwide Eutrophication of Water Bodies: Causes, Concerns, Controls

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NOMENCLATURE

Glossary of Limnological Terms (from Ruttner, 1952; Crosby et al., 1990)

biomass	Weight of living matter.
chlorophyll a	Primary photosynthetic pigment. The concentration of chlorophyll *a* in water is an indicator of phytoplankton biomass. See *biomass*.
cyanobacteria	Prokaryotic organisms in the phytoplankton community. Cyanobacteria are typified by cells without a nucleus or organelles and with photosensitive pigments dispersed throughout the cell. Some species are capable of obtaining nitrogen for metabolism from the atmosphere.

drainage basin   The land area that contributes surface runoff to a water body. See "runoff."

epilimnion   The warm uppermost layer in a thermally stratified water body that is subject to mixing by wind.

eutrophic   Waters with a good supply of nutrients and hence a rich organic production.

freshwater   Water with concentration of total dissolved solids below 500 mg L^{-1}.

hypolimnion   The cool bottom layer in a thermally stratified water body that is separated from surface influences by a thermocline (see *thermocline*).

macrophytes   Large aquatic plants.

mesotrophic   Waters with a moderate supply of nutrients and organic production.

nutrient limitation   Limitation of phytoplankton biomass by an insufficient nutrient supply compared to the demand.

oligotrophic   Waters with a poor supply of nutrients and organic production.

phytoplankton   The photosynthesizing portion of the plankton. See also *plankton*.

plankton   The community of the free water.

primary production   The production of organic matter from inorganic materials within a certain period of time by autotrophic organisms.

runoff   The water reaching a lake, stream, or ocean after flow over land or through the surficial layers of the land.

Secchi disk depth   The depth in water to which a Secchi disk (a 20 cm diameter disk with alternating black and white quadrants) can be seen from the surface. Secchi disk depth is an easy measurement of water transparency.

stratified   Divided into layers. In stratified water bodies, there may be mixing within a layer but little mixing occurs between layers. Layers have different densities, which may be determined either by temperature and/or salinity.

thermocline   The layer of water in a water body between the epilimnion and hypolimnion in which the temperature gradient is greatest and exceeds a change of 1 °C m^{-1} of depth.

trophic state   The degree of fertility of a lake. See also *eutrophic*, *mesotrophic*, and *oligotrophic*.

zooplankton   The animal portion of the plankton.

9.08.1   INTRODUCTION

9.08.1.1   Aspects of Worldwide Concern over Eutrophication

Eutrophication is the nutrient enrichment of waters that stimulates an array of symptomatic changes, that can include increased phytoplankton and rooted aquatic plant (macrophyte) production, fisheries and water quality deterioration, and other undesirable changes that interfere with water uses (Bartsch, 1972). The trophic state, or degree of fertility, of water bodies ranges from oligotrophic to mesotrophic to eutrophic with increasing supply of nutrients and organic matter (Table 1). Eutrophication is most often the result of an elevated supply of nutrients, particularly nitrogen and phosphorus, to surface waters that results in enhanced production of primary producers, particularly phytoplankton and aquatic plants.

Phytoplankton are unpleasant at high densities. The sight and smell of clots or masses of decaying phytoplankton decreases the recreational value of most waters and usually generates concerns among the public. Furthermore, blooms of toxin-producing phytoplankton can cause widespread illness. A bloom is a conspicuous concentration of phytoplankton, often concentrated at or near

Table 1   Mean annual values for the trophic classification system.

	Total phosphorus (μg L^{-1})	Chlorophyll a (μg L^{-1})	Secchi disk depth (m)
Ultra-oligotrophic	<4	<1	>12
Oligotrophic	<10	<2.5	>6
Mesotrophic	10–35	2.5–8	6–3
Eutrophic	35–100	8–25	3–1.5
Hypertrophic	>100	>25	<1.5

Source: OECD (1982).

the surface. It is difficult to quantify what constitutes a "bloom," but a rough estimate places it as a chlorophyll *a* concentration over $30 \mu g L^{-1}$. Toxins produced by dinoflagellates such as *Pfiesteria* in marine environments of the northeastern US and red tides in tropical waters have caused massive fish kills, millions of dollars in losses to seafood-related industries, human memory loss, paralysis, and even death (Van den Hoeck *et al.*, 1995; Silbergeld *et al.*, 2000). Bloom-forming species of cyanobacteria can produce potent hepato-(liver) toxins termed microcystins that have been implicated in poisonings of domestic livestock, pets, wildlife, and susceptible humans (Codd, 1995; Dunn, 1996). In addition, an accumulation of dead phytoplankton in bottom waters of eutrophic systems can lead to high decomposition rates by bacteria. Dissolved oxygen consumption by decomposers, combined with a barrier to gas exchange (thermocline or ice cover), can reduce (hypoxia) or eliminate (anoxia) dissolved oxygen in bottom waters. (A *thermocline* is the junction between an upper layer of warm, less dense water (the epilimnion) and a deeper layer of cold water (the hypolimnion). When this stratification is in place, the typically oxygen-rich waters of the epilimnion do not mix with the waters of the hypolimnion.) Oxygen depletion is one of the most harmful side effects of eutrophication because it can cause catastrophic fish kills, devastating local fisheries.

The accumulation of plant biomass depends on the addition of factors that stimulate plant growth. On average, the macronutrients nitrogen and phosphorus are present in marine phytoplankton at an atomic ratio 16 : 1 (Redfield, 1958). The ratio of nitrogen to phosphorus in freshwaters tends to be greater than the ratio in phytoplankton; therefore, phosphorus most often limits the growth of phytoplankton. As a result, phosphorus enrichment of freshwater often causes its eutrophication (Schindler, 1977). In lakes, nitrogen is usually present in concentrations equal to or beyond what is required for aquatic plant growth because, unlike phosphorus, it has an atmospheric source. In marine systems, nitrogen concentrations are often limiting because bacterial nitrogen fixation, while a considerable source of nitrogen in lakes, not as important in marine waters. A wide variety of prokaryotic organisms (i.e., certain cyanobacteria, heterotrophic, and chemoautotrophic bacteria) can use nitrogen gas directly and incorporate it into organic compounds through a process called nitrogen fixation. Nitrogen fixation is an enzyme-catalyzed process that reduces nitrogen gas (N_2) to ammonia (NH_3). Nitrogen-fixing cyanobacteria make up less than 1% of the total biomass of phytoplankton in estuaries of the Atlantic coast of North America, whereas in lakes they often make up more than 50% of
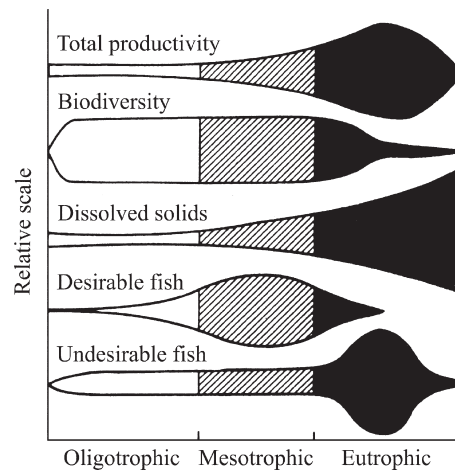

Figure 1 Suggested changes in various characteristics of lakes with eutrophication (reproduced by permission of Cambridge University Press from *Ecological Effects of Waste Water*, **1980**).

phytoplankton biomass (reviewed in Howarth, 1988). An increase in water clarity can also spur the growth of aquatic vegetation in systems where the clarity of water is poor from high concentrations of suspended particles.

The biodiversity of most aquatic systems decreases with eutrophication (Figure 1). Phytoplankton species diversity is reduced in highly productive systems. Cyanobacteria are usually dominant in eutrophic systems because these organisms are better adapted to conditions of high nutrients (Smith, 1986; Trimbee and Prepas, 1987; Watson *et al.*, 1997). In addition, fish and macro-invertebrate species diversity can decrease with eutrophication. Depletion of dissolved oxygen in deep water is associated with eutrophication and can lead to a loss or displacement of species intolerant of such conditions (Ludsin *et al.*, 2001). In eutrophic lakes of North America, characteristic fish types are surface-dwelling, warm water fishes such as pike, perch, and bass, as compared to deep-dwelling, cold-water fishes like salmon, trout, and cisco (Ryding and Rast, 1989).

9.08.1.2 Indicators of Eutrophication and Sampling Methods

Trophic state can be measured directly via indicators of phytoplankton biomass and nutrient concentrations or indirectly by measuring water transparency (Table 1). Water transparency can be an accurate indicator of phytoplankton productivity, even though it is affected by nonbiological particles suspended in the water column. Secchi disk depth is the oldest, simplest, and quickest quantitative measure of water clarity. A Secchi disk is a black and white disk that is

lowered down through the water column until it can no longer be seen. Secchi disk depth is the midpoint between the depth at which it disappears when lowered and reappears when it is pulled up again. A more objective method of measuring transparency is with a light meter, generally an upward-directed quantum sensor that determines the intensity of light (often measured in $\mu E\,m^{-2}\,s^{-1}$). A light meter can be used to measure the euphotic zone, or the portion of the water column receiving over 1% surface irradiation. In the euphotic zone, net photosynthesis takes place and phytoplankton and rooted aquatic plants (macrophytes) can grow. A third approach to determine water transparency involves measuring turbidity in water samples. Turbidity is a measure of the scattering of light caused by suspended particles (e.g., mud, silt, and phytoplankton) in the water column and is recorded in standard international Nephelometric Turbidity Units (NTU). Water transparency is a good indicator of eutrophication and can easily be measured by members of the general public.

Because eutrophication is associated with high phytoplankton numbers, the biomass of these organisms is a useful indicator of trophic state. A good descriptive measure of eutrophication and the general state of water is the presence of phytoplankton blooms. Phytoplankton blooms can cause visible coloration of surface waters or dense mats and clumps, usually accompanied by an unpleasant rotting smell. Cyanobacteria are the dominant type of phytoplankton in most freshwater blooms (Crosby *et al.*, 1990). Many species of cyanobacteria contain gas vacuoles in their cells that make them buoyant. As such, in highly eutrophic lakes, cyanobacterial mats and clumps can form and be transported by wind into foul smelling, decaying masses around the perimeter of a lake. In the sea, blooms are caused by species of nonbacterial forms of phytoplankton (e.g., dinoflagellates, haptophytes). An effective indicator of changes in phytoplankton biomass, and the most widely used, is the concentration of chlorophyll *a*, the primary photosynthetic pigment in all phytoplankton. Chlorophyll *a* is measured by filtering water, extracting chlorophyll from phytoplankton cells captured on a fine filter and measuring the color density of the extract with spectrophotometry.

Because an increased supply of nitrogen in marine waters and/or phosphorus in freshwaters is typically responsible for eutrophication, directly measuring the concentration of these nutrients can indicate trophic state. Nutrients in water are either dissolved or bound to particulates produced *in situ* and deposited into water bodies from groundwater, atmospheric deposition, and drainage basin runoff. To measure trophic state, nutrient concentrations are measured in samples taken from a water body when surface water nutrient concentrations are at their maximum after replenishment from bottom waters. Sampling is carried out during spring overturn (i.e., complete mixing of the water column) where nutrient concentrations in surface waters vary seasonally (i.e., systems with alternating thermal stratification and overturn with the seasons). In systems with less routine thermal dynamics (i.e., those with multiple breakdown of thermal stratification), additional sampling may be required to take account of changes in nutrient concentrations from multiple mixing events. Nutrient concentrations in surface water are relatively low (i.e., often few ppb) and are measured in raw water samples. The principal method of determining the total concentration of phosphorus and nitrogen in a water sample is by digestion or oxidation of organic matter to release bio-available inorganic nutrients (phosphate, nitrate, and ammonium). The addition of a mixed reagent creates color and its density is measured with spectrophotometry (Prepas and Rigler, 1982; Crumpton *et al.*, 1992).

9.08.2 NATURAL EUTROPHICATION

Because of the high retention of nutrients in most undisturbed land, very few aquatic systems are naturally eutrophic. However, nutrient loading, and thus trophic state, is influenced first by natural processes. Nutrient composition of soil and its exposure to draining water are important determinants of the chemistry of water exported from the drainage basin. Soil fertility is related to the chemistry of its parent rock material. Soils that lie on sedimentary rock tend to be more fertile than soils on hard igneous rock. For example, the classic prairie soils, famous for the production of corn, are of sedimentary origin (Jenny, 1980). Lakes on igneous geology can be naturally more nutrient-poor than lakes on sedimentary geology (Rawson, 1960). Further, there is more potential for leaching or biological decay of soils in drainage basins with longer exposure to water. Therefore, drainage basins that are relatively flat (i.e., low slopes), and those that are located in areas of low precipitation, are more likely to export water with high nutrient concentrations (D'Arcy and Carignan, 1997). Water bodies are closely linked to the characteristics of their drainage basins.

The drainage ratio ("surface area of drainage basin" to "water body") can represent the contribution of the drainage basin to the nutrient budget of receiving waters, relative to the atmosphere. The nutrient budget of a water body with a relatively small drainage ratio will be controlled primarily by atmospheric nutrient

inputs because drainage basin contributions are relatively minor. The relative drainage basin size ("surface area of drainage basin" to "water body volume") can indicate how long it takes for the entire volume of water (and nutrients) in a water body to be flushed out of a system. Rapidly flushed systems with large relative drainage basins, such as streams, are more closely connected to their drainage basin than those that are flushed slowly, such as large lakes (Søballe and Kimmel, 1987). A large relative drainage basin, all else being equal, is typically associated with higher nutrient content in surface waters, because nutrient inputs per unit of time are greater. Within one geological setting, the relative drainage basin size of water bodies can account for most of the variability in phosphorus concentrations and phytoplankton biomass in aquatic systems (D'Arcy and Carignan, 1997; Prepas *et al.*, 2001b). The quantity of water and nutrients exported from a drainage basin also depends on evapotranspiration. Studies comparing drainage basins that vary in forest composition indicate that the quantity of water exported is a function of the effect of vegetation differences on transpiration rate (Bosch and Hewlett, 1982). Therefore, the amount (e.g., Hobbie and Likens, 1973) and type (e.g., Cronan and Aiken, 1985) of vegetation, in addition to relative drainage basin size, can influence water and nutrient export to water bodies.

Summer phosphorus concentrations in nutrient-rich water bodies can often depend on *in situ* rather than external loading. Internal phosphorus loading, or the recycling of phosphorus from bottom sediments, usually occurs and/or is enhanced after a long history of eutrophication and phosphorus enrichment of sediments due to high rates of organic sedimentation. In culturally eutrophic systems with a long history of external phosphorus enrichment, recovery can be delayed due to the release of biologically available phosphorus from bottom sediments. Sediment phosphorus release is regulated by the intensity of oxidizing or reducing conditions within an aquatic system as measured by the reduction–oxidation, or redox, potential. A reducing environment produced by the oxidation of sediment organic matter during bacterial decomposition causes dissolution of $FeOOHPO_4$ complexes and release of highly mobile and bio-available PO_4^{3-}. In productive systems, anoxic conditions can develop at the sediment–water interface after an extended period of thermal stratification that prevents atmospheric oxygen from reaching deeper waters. Oxygen depletion, and therefore internal loading, is most likely in relatively shallow lakes, because the hypolimnion (i.e., deep layer of cold water), no matter how transient, has a reduced water volume and thus dissolved

oxygen pool. In shallow and fertile systems, external loading can be insignificant compared to internal loading on an annual basis (Riley and Prepas, 1984). All else being equal, most lakes become shallower over a long timescale through accumulation of bottom sediments from particles that have settled from the water column. Therefore, lakes tend to evolve towards eutrophy (Hutchinson, 1973).

9.08.3 CULTURAL EUTROPHICATION: CASE STUDIES

Water pollution shadows human population growth and development and is caused by diffuse (nonpoint) and concentrated (point) nutrient enrichment. Some of the most severe cases of anthropogenic eutrophication occurred soon after the Second World War, when rapid population growth and development followed. At that time, population growth and the concentration of nutrient-rich sewage enhanced the point-source nutrient loading. Deforestation, or clearing of native vegetation, is one of the most basic human alterations of the environment and can be an early and important nonpoint source of eutrophication. With the removal of actively transpiring vegetation, an excess of water in the ground is available for soil weathering (Roby and Azuma, 1995). Therefore, vegetation removal can increase the transfer of nutrients from the drainage basin to receiving waters. The largest improvements in water quality have often been associated with reduction or elimination of point-source inputs. Not surprisingly, the most important challenges for the future preservation of water quality lie in controlling nonpoint nutrient sources from intense activity such as agriculture and associated use of fertilizer, concentration of animal wastes, and vehicle emissions. Drainage basins with intensive agriculture can export five or more times the phosphorus and nitrogen of forested drainage basins (Dillon and Kirchner, 1975). Although eutrophication abatement has focused on enhanced nitrogen and phosphorus content of drainage basin runoff, concern over atmospheric nitrogen pollution has developed with increasing industrialization. Our case studies take us from the west coast of North America (Lake Tahoe and Lake Washington) across to Lake Erie and then to Chesapeake Bay on the east coast. Two further examples come from southern Europe (Lago Maggiore) and Africa (Lake Victoria; Figures 2 and 3). These six case studies highlight varying degrees of eutrophication and restoration of systems impacted by numerous sources of nitrogen and phosphorus pollution (Table 2).
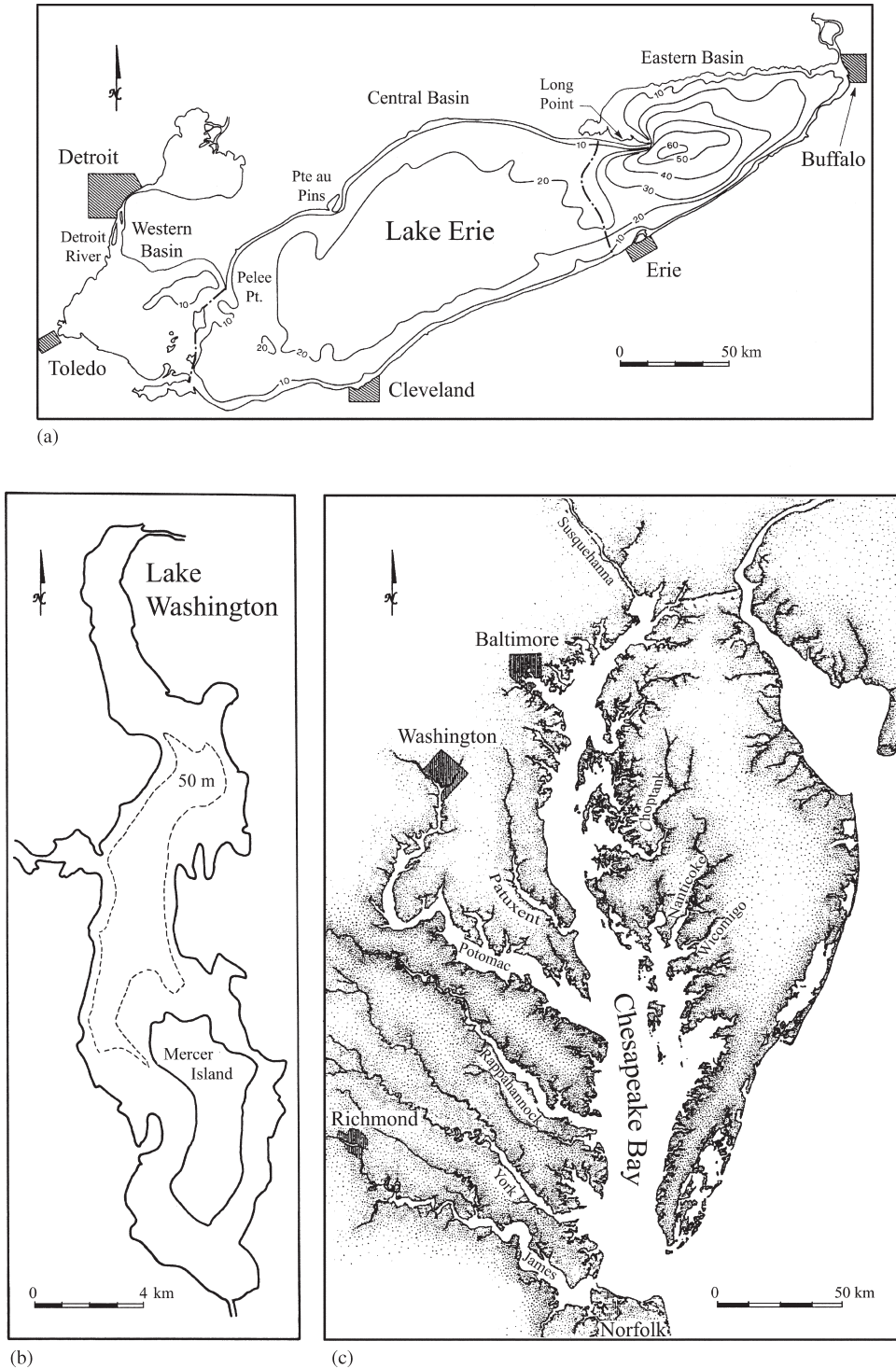
Figure 2   (a) Lake Erie, (b) Lake Washington, and (c) Chesapeake Bay ((a) reproduced by permission of National Research Council Canada from *J. Fish. Res. Board Can.*, **1976**, *33*, 355–370; (b) reproduced by permission of The American Society of Limnology and Oceanography, from *Limnol. Oceanogr.*, **1991**, *36*, 1031–1044; and (c) reproduced by permission of Inter-Research Science Publishers from *Mar. Ecol. Prog. Ser.*, **1994**, *104*, 267–291).

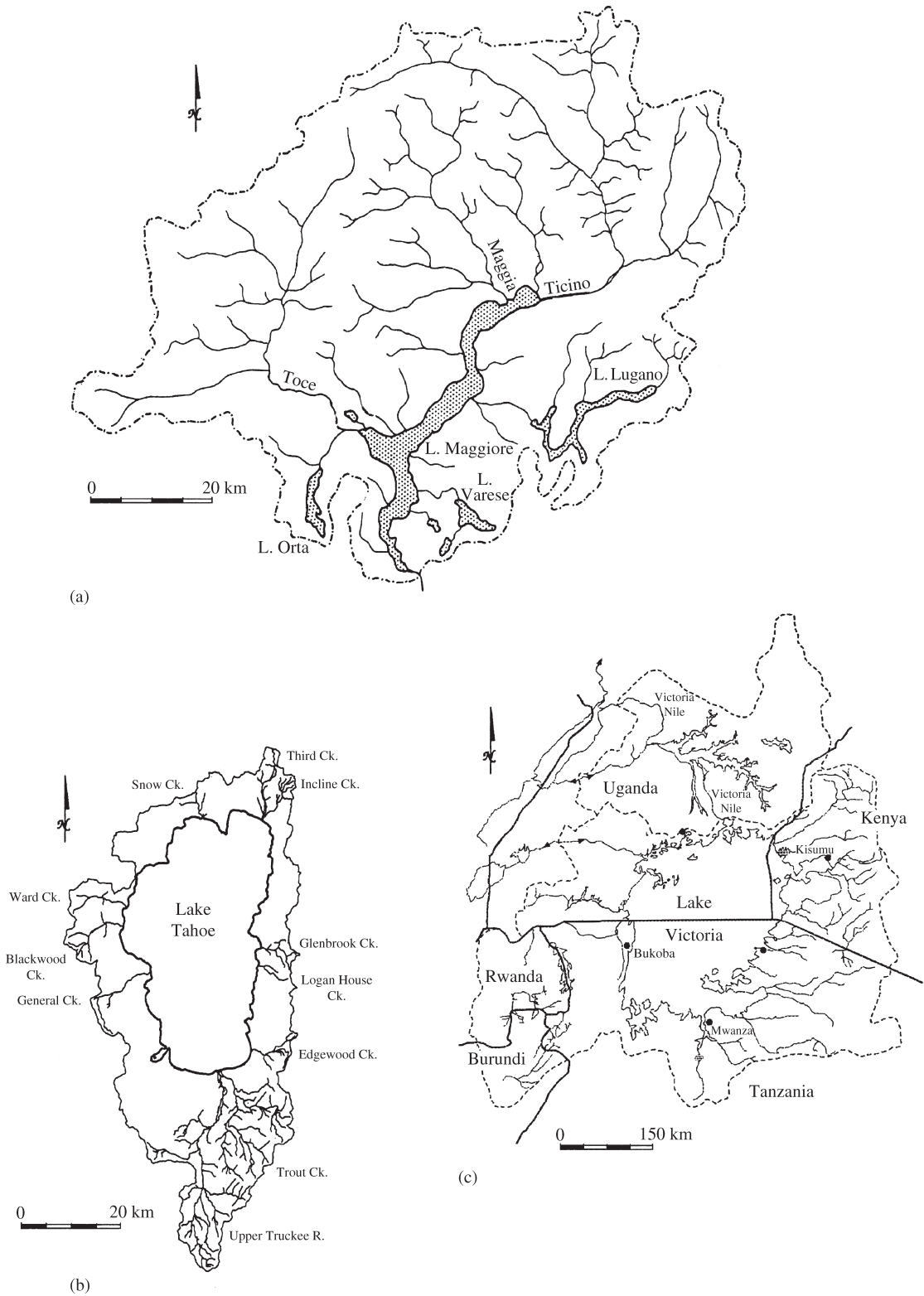**Figure 3** (a) Lago Maggiore, (b) Lake Tahoe, and (c) Lake Victoria and their drainage basins ((a) reproduced by permission of Istituto Italiano di Idrobiologia from *Mem. Ist. Ital. Idrobiol.*, **1995**, *53*, 53–73; (b) reproduced by permission of E. Schweizerbart Science Publishers from *Arch. Hydrobiol.*, **1995**, *135*, 1–21; and (c) reproduced by permission of Elsevier from *J. Environ. Manage.*, **2000**, *58*, 235–248).

Table 2 Characteristics of, and lessons from, case studies.

Water body	Characteristics
Lake Washington	Point-source (wastewater) nutrient stress.
	Mitigated via diversion of nutrient source downstream and chance biological control of phytoplankton.
	Rapid recovery.
Lake Erie	Point (wastewater) and nonpoint (agriculture) source stresses.
	Multiple basins illustrate effects of morphometry, stratification, and flushing rates on trophic status.
	Mitigated via legislated wastewater and laundry detergent nutrient targets.
	Partially recovered—depends on future nonpoint-source mitigation.
Lake Tahoe	Atmospheric pollution stress.
	Low drainage ratio = atmosphere important nitrogen source.
	Tertiary sewage treatment limited success because sensitivity to nitrogen rather than phosphorus.
	Mitigation via nonpoint-source controls but atmospheric sources difficult to control = further degradation likely.
Lago Maggiore	Point (wastewater + polluted upstream lakes) and nonpoint (agriculture) source stresses.
	Mitigated via wastewater treatment and recovery of upstream lakes.
	Partially recovered system.
Lake Victoria	Nonpoint-source stresses: atmospheric and land use + internal nutrient loading controlled by wet–dry seasons.
	Low drainage ratio and high basin evapotranspiration = atmosphere important nutrient source.
	Further degradation likely because little mitigation.
Chesapeake Bay	Mostly nonpoint (agriculture), some point (wastewater) source stress.
	Estuarine example where nitrogen-limitation important.
	Mitigation efforts via wastewater treatment (point source) + wetland restoration to serve as nutrient traps (nonpoint source).
	Early restoration phase.
Lessons	Point sources are easy to control.
	Nonpoint sources are difficult to control because of diffuse nature.
	Internal loading is important long-term feedback mechanism.
	P problems associated with land use changes.
	N problems associated with atmospheric pollution.

9.08.3.1 Lake Washington

Lake Washington (87.6 km^2; Figure 2) is a relatively deep lake (maximum depth 65.2 m) in Washington State, USA, with one of the finest histories of limnological observations due to the lifelong dedication of the late W. T. Edmondson. Classified as mesotrophic (water with moderate supply of nutrients and organic production) in recent history (Scheffer and Robinson, 1939), Lake Washington was used as a source of drinking water for some small communities until 1965. At that time, total and bio-available (soluble reactive) phosphorus concentrations were relatively low (16 μg L^{-1} and 8 μg L^{-1} (ppb), respectively), and there was no record of nuisance phytoplankton blooms.

Pollution was associated with water effluent from the city of Seattle. By 1963, water quality in Lake Washington was the poorest on record. At that point, sewage accounted for ~65% of all phosphorus inputs into Lake Washington (Edmondson, 1975). The average annual total phosphorus and winter phosphate concentrations peaked at 3 and 5 times, respectively, that in 1933 (Figure 4). Summer phytoplankton biomass (estimated as chlorophyll *a*) was over 10 times higher in 1963 than only 13 years earlier and Secchi disk depth dropped from 3.7 m to less than 1 m over the same time span (Figure 4). At the peak of water quality decline, as much as 98% of the volume of phytoplankton was made up of cyanobacteria, largely *Oscillatoria* species such a*s O. rubescens* and *O. agardhii*, that are often present in eutrophic systems (Edmondson, 1969).

Public pressure to halt the severe deterioration of Lake Washington resulted in action. Between 1963 and 1968, sewage effluent from the treatment plants was progressively diverted to Puget Sound (Figure 4). The first diversion removed 28% of the effluent, stopping the eutrophication process. Recovery was quick with a sharp (65%) decrease in phytoplankton biomass two years after the beginning of diversion (Figure 4). By the end of the diversion process, water quality had returned to mesotrophic conditions
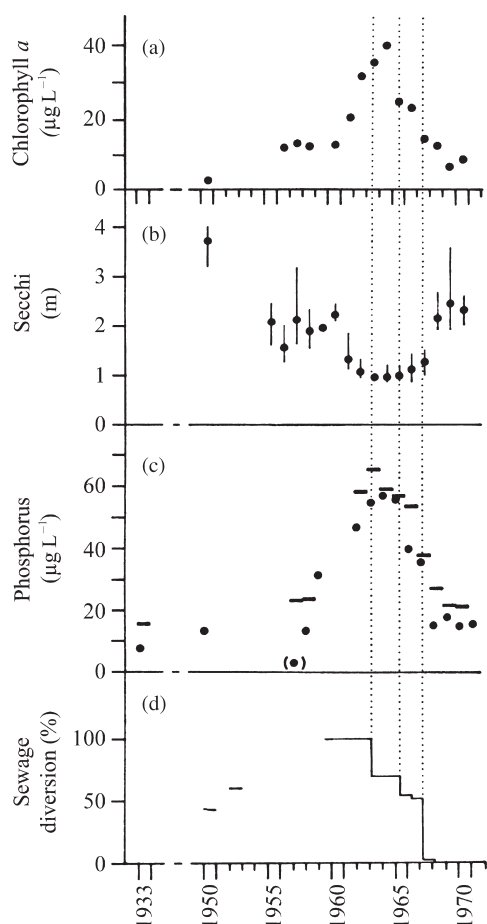
Figure 4 Changes in Lake Washington water quality in relation to sewage diversion. (a) Mean (July–August) lake surface (epilimnetic) chlorophyll *a* concentration. (b) Mean (July–August) Secchi disk depth. The vertical lines show the range. (c) Bars = annual mean of total phosphorus in the epilimnion. Circles = mean (January–March) bio-available phosphorus (phosphate) concentrations in the epilimnion. (d) Relative amount of sewage as indicated by the capacity of the treatment plants emptying into the lake, maximum taken as 100% (reproduced by permission of American Society of Limnology and Oceanography from *Nutrients and Eutrophication*, **1972**, pp. 172–193).

(chlorophyll *a*: 5–10 μg L^{-1}, Secchi disk depth: 2–2.5 m, total phosphorus: 20–30 μg L^{-1}). Recovery continued. By 1975, transparency increased to 4 m and total phosphorus and chlorophyll *a* concentrations decreased to 16 μg L^{-1} and 4 μg L^{-1}, respectively, similar to conditions in 1933. Thus, deterioration in the water quality of Lake Washington was reversible.

Biological regulation of phytoplankton biomass greatly improved water quality in Lake Washington. The appearance of the aggressive phytoplankton grazer, the microcrustacean *Daphnia* spp., coincided with an increase in water transparency to as much as 12.9 m by 1976.

Daphnia appeared after the decline of inedible cyanobacteria (*Oscillatoria* spp.), and thrived due to a sustained 10% decrease in its voracious predator *Neomysis mercedis* in 1965. The decrease in *Neomysis* occurred at about the same time as the introduction of longfin smelt, a predator known to specialize in eating *Neomysis* (Edmondson, 1994). Helped by a rapid flushing rate (2.3 yr) and stable dissolved phosphorus inputs since the end of sewage diversion (Edmondson, 1994), Lake Washington remains mesotrophic.

9.08.3.2 Lake Erie

Lake Erie offers one of the best-publicized examples of the serious impacts of humans on water quality as well as a model of ecosystem rehabilitation, thanks to international cooperation. Of the five North American Great Lakes, Lake Erie was the most seriously impacted by cultural nutrient enrichment. Lake Erie is thought to be naturally more eutrophic than the other Great Lakes because of its shallower depth and the greater erodibility of nutrient-rich soils in its immediate drainage basin (i.e., excluding upstream Great Lakes). Lake Erie can be divided into three basins (Figure 2), distinguished by mean depth, that have distinct morphometric and trophic characteristics. The western basin (3,080 km^2) is the shallowest (mean depth 7.6 m), and most nutrient- and plankton-rich of the Lake Erie basins. The water column mixes completely throughout most of the year with brief and unpredictable periods of thermal stratification between May and September. The western basin is eutrophic, because it is shallower (see Section 9.08.2: Natural Eutrophication), and receives the largest external nutrient loads. The central basin, largest of the three in area (16,425 km^2) and volume has a mean depth of 18.6 m. Here, thermal stratification is established yearly between June and September, causing bottom waters to become hypoxic by late summer. The eastern basin (area 6,159 km^2) is by far the deepest of the three (mean depth 26 m). Thermal stratification is strongest in this basin and persists from June to late October or early November. The eastern basin is considered oligotrophic, and therefore the thick hypolimnion (bottom layer of cold water) experiences only small reductions in dissolved oxygen concentrations (Bartish, 1987). Thus, Lake Erie displays a longitudinal gradient in trophic status; from eutrophic waters in the western basin to progressively diluted and oligo-trophic waters in the outflowing eastern basin.

Because most of the immediate drainage basin is composed of highly erodible sedimentary soils, it is believed that the eutrophication of Lake Erie

began with land clearing associated with European settlement. Phosphorus loading increased slowly after forest removal in the drainage basin in the late nineteenth century, then increased exponentially over three decades up to the early 1970s (Snell, 1987). At that time, the western basin was highly eutrophic, the eastern basin was mesotrophic, and the central basin was meso- to eutrophic (M. Munawar and I. F. Munawar, 1976). When Lake Erie was in its worst state, total phosphorus load was estimated at 1.2×10^4 t yr^{-1}, four times higher than precolonization values (Vallentyne and Thomas, 1978). Parallel to the increasing load, phosphorus concentrations increased fivefold in the central basin, to ~25 µg L^{-1}. By the early

1960s, cyanobacterial blooms appeared in Lake Erie and phytoplankton abundance tripled from measurements taken 40 years earlier (Figure 5; Davis, 1964). The western basin and parts of the central basin developed dense mats of the cyano-bacterium *Aphanizomenon flos-aquae* (Beeton, 1965). Furthermore, dissolved oxygen depletion rates in bottom waters increased (Figure 5) and extensive hypoxia developed: dissolved oxygen concentrations were low (3 mg L^{-1} (ppm) or less) in ~70% of the bottom waters of the central basin during late summer (Beeton, 1963). Major changes also occurred in the bottom fauna of the western basin; the benthic community dominated by the burrowing mayfly (*Hexagenia* spp.) was replaced

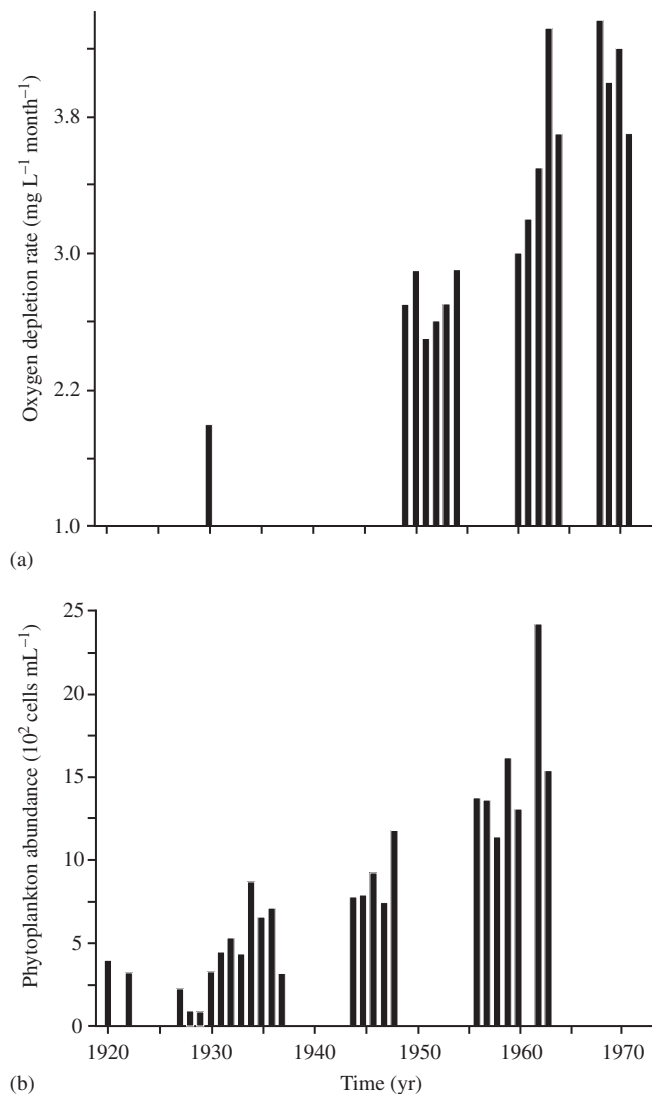**Figure 5** (a) Mean depletion rates for dissolved oxygen during summer in the bottom-water (hypolimnion) of central Lake Erie (reproduced by permission of National Research Council Canada from *J. Fish. Res. Board Can.*, **1976**, *33*, 355–370). (b) Mean phytoplankton abundance in Lake Erie from 1920 to 1963 (reproduced by permission of American Society of Limnology and Oceanography from *Limnol. Oceanogr.*, **1964**, *9*, 275–283).

almost entirely by one dominated by midge larvae (chironomids) and oligochaetes tolerant of low-oxygen conditions. *Hexagenia* spp. disappeared shortly after two periods of anoxia in the early 1950s (Beeton, 1961). Furthermore, the fish fauna changed from natural cold-water fish populations (whitefish, sauger, cisco, blue-pike, and lake trout) to domination by warm-water species (yellow perch, smelt, fresh-water drum, white bass, carp, catfish, and walleye) characteristic of eutrophic lakes.

Due to the severity of eutrophication and extensive hypoxic conditions, programs were developed in the early 1970s to reduce phosphorus loads into the Great Lakes as part of the Great Lakes Water Quality agreement between Canada and the United States. The agreement required that phosphorus concentration of sewage effluent from all large municipal waste treatment plants be limited to 1 mg L^{-1}. A total of 7.5 billion dollars was spent on municipal waste facility enhancement. Total phosphorus loads from direct municipal discharges were reduced by more than 80% by the mid-1980s. Much of this reduction was a result of phosphorus restrictions in detergents; up to 50% of phosphorus in sewage originated from laundry detergents. The Lake Erie ecosystem responded quickly to these management efforts. Total phosphorus concentrations in the western basin had decreased by 50% only a decade after the Great Lakes Water Quality Agreement (Rosa, 1987). Over two decades after the Agreement, the rate of decline in total phosphorus concentrations was ~0.44 μg L^{-1} yr^{-1} (Rosa, 1987; Bertram, 1993). During the same time period, total phytoplankton and cyanobacterial (i.e., *Aphanizomenon flos-aquae*) biomass decreased by about 65% and 89%, respectively (Makarewicz and Bertram, 1991), and mesotrophic phytoplankton species common before eutrophication (*A. formosa*) reappeared (Makarewicz, 1993). During that time, there was also a slight decrease in dissolved oxygen depletion rates in the deeper water (Bertram, 1993). Six fish species tolerant of hypoxic conditions declined in abundance and three species intolerant of degraded water quality recovered (Ludsin *et al.*, 2001).

Despite the dramatic recovery of Lake Erie through point-source reductions, the lake remains enriched in phosphorus. Over half of the external load of total phosphorus to Lake Erie is from tributaries draining cultivated land. Therefore, attainment of the desired total phosphorus load will depend on further nonpoint-source controls (Dolan, 1993). Lake Erie must be monitored regularly because of its dynamic nature due to heavy use as an international shipping route. Following its introduction to the Great Lakes about 20 years ago and prolific expansion, the zebra mussel (*Dreissena* spp.) has been partly responsible for tremendous improvements in water clarity in recent years (Makarewicz *et al.*, 1999). The Lake Erie restoration story continues to unfold.

9.08.3.3 Lake Tahoe

With its enchanting emerald green waters, Lake Tahoe lies on the border of the American states of Nevada and California, in the Sierra—Nevada mountain range (elevation 1,898 m; Figure 3). Because of its stunning mountainous setting and its naturally clear waters (Secchi disk depth 41 m in 1972), this large (500 km^2) and deep (maximum depth 505 m) lake has long been revered as one of the most beautiful water bodies in the world, thanks in part to its limnological guardian, C. R. Goldman. Naturally ultra-oligotrophic, Lake Tahoe receives low nutrient loads from the drainage basin. On an annual basis, phosphorus inputs to the lake are predominantly from terrestrial sources, similar to Lake Washington. In contrast, atmospheric sources of nitrogen, mostly nitrate, are more significant than drainage basin inputs because of Lake Tahoe's small drainage ratio (1.6; Figure 3). Due to the relative paucity of nitrogen, however, phytoplankton biomass in Lake Tahoe has been controlled, until recently, by the limited availability of ammonium and nitrate (Goldman *et al.*, 1993).

Because of human influences, nitrogen inputs to Lake Tahoe have steadily increased and have fueled a noticeable growth in phytoplankton biomass. Limnological changes started to occur by the middle of the nineteenth century, when the basin was logged to provide timber for mining operations (Jassby *et al.*, 2001). Lake Tahoe has also seen a tremendous population growth and drainage basin development since the 1950s (as of early 2000s: a resident population of 52,000; one million visitors yearly). Shoreline development proceeded with little consideration for runoff and erosion control. To complicate matters, Lake Tahoe's drainage basin has steep slopes that produce high-energy runoff. Therefore, nutrients leached from destabilized and eroding soil on the lakeshore have become a major source of nitrogen to the lake. However, atmospheric deposition provides most of the nitrogen load to Lake Tahoe and this has been increasing steadily due to local combustion of fossil fuels for transportation and heat, and upwind human development (cities of Los Angeles and San Francisco; Jassby *et al.*, 1994; Goldman, 1993). Phytoplankton are quite responsive to atmospheric loads of nitrogen, because they are deposited mostly in bio-available form (i.e., nitrate and ammonium). Conversely, total nitrogen from drainage basins contains a greater fraction of organic nitrogen, much of which is resistant to rapid bacterial degradation to

bio-available forms (Kalff, 2002). As a result of increased deposition of atmospheric nitrogen, the rate of phytoplankton production (primary productivity) has more than quadrupled and Secchi disk depth has decreased ~30% since the late 1960s (Figure 6). With the increase in nitrogen relative to phosphorus loads, phytoplankton growth in Lake Tahoe has gone from limitation largely by nitrogen to strong phosphorus-limitation (Jassby *et al.*, 1995). Despite cultural eutrophication, Lake Tahoe remains ultra-oligotrophic (Secchi disk depth 25 m; Goldman *et al.*, 1993). However, because water and nutrient flushing times are long (650 yr), prevention of additional fertilization of Lake Tahoe is crucial. Reduction of within-basin nutrient sources (e.g., Best Management Practices (BMPs); see Section 9.08.4.2) has been the focus of water quality management. However, the fate of Lake Tahoe appears dim at this moment because atmospheric nutrient sources are variable; they depend on large-scale wind patterns and are difficult to control.

9.08.3.4 Lago Maggiore

Lago Maggiore is located in the foothills of the Alps, just north of the most industrialized part of Italy (Figure 3). With the CNR—Institute of Ecosystem Study (ISE) on its shore, Lago Maggiore has the privilege of being the most thoroughly investigated lake in Italy, both in terms of the number of key biogeochemical parameters measured and the duration of their monitoring. Lago Maggiore is deep (maximum depth 370 m) and only the top 100–150 m of its water column circulates once each year. However, the lake mixes completely over a varying number of years, the mixing being connected with very cold autumns and winters, and with windy periods during March and April, when thermal stratification is weaker (Ambrosetti and Barbanti, 1999). Before major eutrophication began, the lake was oligotrophic (spring total phosphorus concentrations below $10 \, \mu g \, L^{-1}$, Secchi disk depth 10.7 m), and the growth of phytoplankton was limited by the availability of phosphorus (Mosello and Ruggiu, 1985).

Like many lakes around the world, Lago Maggiore was rapidly eutrophied during the 1960s causing a shift in trophic state from oligotrophic to mesotrophic a decade later. During this period, bio-available (soluble reactive) phosphorus concentrations tripled and nitrate concentrations doubled in surface waters and stabilized in subsequent years (Figure 7; Ambrosetti *et al.*, 1992; Ruggiu and Mosello, 1984). As agriculture
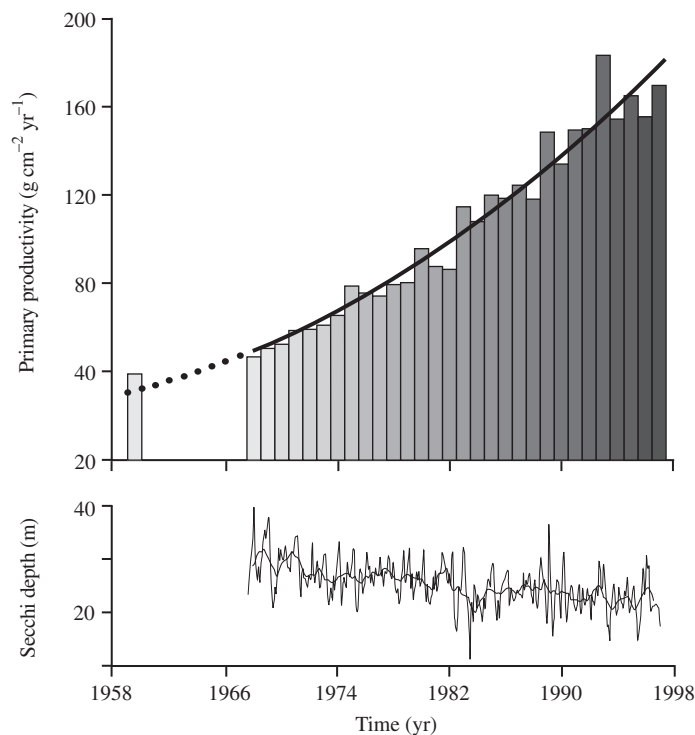

Figure 6 Mean annual primary productivity and mean monthly Secchi disk depth over time in Lake Tahoe (reproduced by permission of Backhuys Publishers from *The Great Lakes of the World (GLOW): Food-web, Health and Integrity*, **2001**, pp. 431–454).
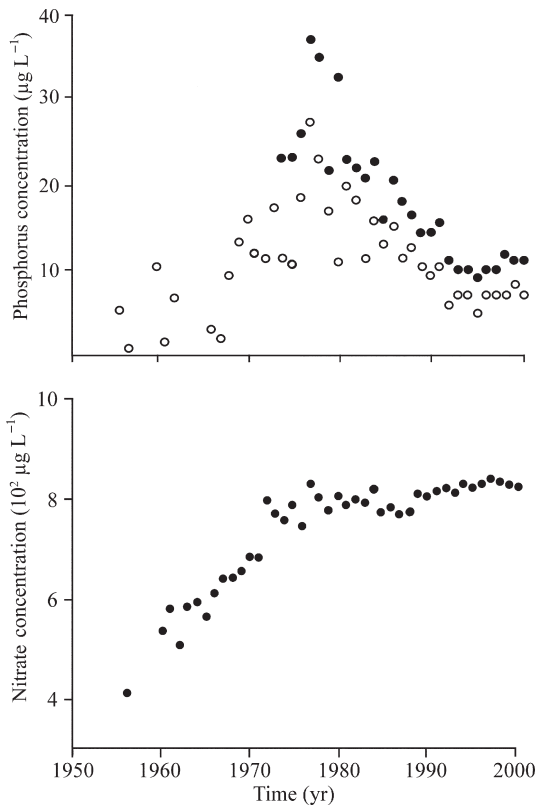
Figure 7 Bio-available (○) and total (●) phosphorus and nitrate concentrations in Lago Maggiore between 1956 and 1991. Mean volume weighted concentrations measured at overturn (reproduced by permission of Istituto Italiano di Idrobiologia from *Mem. Ist. Ital. Idrobiol.*, **1992**, *50*, 171–200; R. Mosello, CNR—Institute of Ecosystem Study, Pallanza, Italy, unpublished data from 1990 to 2000).

in Lago Maggiore's drainage basin is limited, urban sewage from a heavy population base (6.34×10^5 current inhabitants in the drainage basin) provides most of the phosphorus inputs to the lake. Lake nitrogen concentrations seem to be closely related to atmospheric inputs; just under 80% of the nitrogen reaching Lago Maggiore comes from the atmosphere through deposition (Mosello *et al.*, 2001), mostly from the highly industrialized Po Plain situated upwind. Furthermore, three of the biggest lakes draining into Lago Maggiore are highly eutrophic and are important nutrient sources (Mosello and Ruggiu, 1985). Total phytoplankton biomass in Lago Maggiore increased 2.5 times during the two decades after the 1950s (de Bernardi *et al.*, 1988). Over this time, cyanobacteria bloomed for the first time and became the dominant phytoplankton in Lago Maggiore (Manca *et al.*, 1992).

Lago Maggiore has been on the path of recovery since the early 1980s, mostly through the reduction of phosphorus loads. Construction of five treatment plants with a tertiary stage halved the phosphorus load from the shoreline. There has been a further reduction of phosphorus inputs from tributaries along which treatment plants were constructed. Restoration efforts of eutrophic upstream lakes, especially Lago di Lugano, also helped the recovery of Lago Maggiore. Phosphorus loads were also reduced by gradual reductions of the phosphorus content in detergents, from 8% before 1982 to 1% a decade later. (Provini *et al.*, 1992). As a result of these point-source reductions, phosphorus loads from the drainage basin decreased by half from the late 1970s to the late 1980s. Between 1979 and 2000, mean annual total phosphorus concentrations decreased fivefold to ~6.5 $\mu g \, L^{-1}$, while total- and nitrate-nitrogen concentrations have not changed much (Figure 7), probably because of substantial inputs from the atmosphere (Manca *et al.*, 1992; Istituto Italiano di Idrobiologia—CNR, 2001). In response to reduced phosphorus concentrations and because phytoplankton growth was limited by phosphorus, water quality improved and phytoplankton biodiversity increased (Manca *et al.*, 1992). Less than 10 years after the first bloom, cyanobacterial blooms were no longer seen and the dominance of this group had decreased.

9.08.3.5 Lake Victoria

Lake Victoria is located in eastern Africa's rift valley, shared by the countries of Kenya, Uganda, and Tanzania (Figure 3). This ancient lake is an important ecological resource because of its rich diversity of endemic fish species. Due to its impressive size (with a surface area of $6.88 \times 10^4 \, km^2$, it is the second largest freshwater lake in the world after Lake Superior), Lake Victoria is also an important source of cheap protein for millions of people that reside on its shore. Furthermore, the lake regulates the activities of human populations that live downstream in water-scarce locations. Outflow occurs solely north into the White Nile, near Jinja, Uganda. The White Nile crosses into Sudan and links with the Blue Nile in the capital of Khartoum to form the main Nile flow through Egypt to the Mediterranean Sea. Lake Victoria supplies 14% of the water passing through the Nile and thus influences Nile baseflow. The 1964 floods in Cairo were caused by high lake levels due to record rainfalls (Yin and Nicholson, 1998). The water budget of Lake Victoria is controlled largely by evaporation and direct precipitation due to its small drainage ratio and because inputs from tributaries (20% of total water entering the lake) approximately equal outflow over the long term. The contribution of the atmosphere to the nutrient budget of semi-arid lakes is greater, in general,

than in temperate systems because of the relatively high evapotranspiration rates in the former.

Nutrients and biota in Lake Victoria are highly dependent on the physical structure of the water column, which is in turn dependent on seasonality. Seasons in the African Great Lakes region are mediated by the annual cycle of Indian monsoon winds. There are two wet seasons: a longer and wetter rainy season occurs between March and May and the short rainy season falls between October and December. The dry or windy season extends from June to September and is characterized by a cooler climate with strong and persistent southerly winds. The thermocline is strongest during the wet seasons and weakens during the drier, windy season. During the windy season, heat loss through evaporation and turbulence caused by the wind enhances mixing in the water column. Thermocline depth increases substantially during the windy season and complete vertical mixing occurs (Ochumba, 1996). During the rainy season of September to April, runoff, wet deposition of nutrients, and thus, phytoplankton biomass are at their yearly maxima. During the windy season, the open offshore areas are mixed to the bottom and the phytoplankton is pushed to low-light depths, limiting their growth (Mugidde, 1993). Therefore, changes in the phytoplankton biomass of Lake Victoria are determined by seasonality.

Nutrients enter Lake Victoria mainly through two diffuse pathways, or nonpoint sources: land runoff and atmospheric deposition, together accounting for ~90% of phosphorus and ~94% of nitrogen input into the lake. About one-third of the nitrogen inputs to Lake Victoria are from atmospheric deposition and over half of the phosphorus comes from particles associated with runoff from agricultural lands (Scheren *et al*., 2000). Nitrogen loading is dominated by biological fixation in Lake Victoria; the latter provides two-thirds of the total nitrogen supply (Mugidde, 2001). Water bodies in semi-arid climates like Lake Victoria are relatively more dependent on nutrient supply from the atmosphere than temperate systems. High evapotranspiration rates result in relatively less water and nutrients released from the drainage basin, given the relative size of the drainage basin. In addition, undisturbed tropical soils are high in iron and aluminum oxide concentrations that tightly bind phosphorus, making the latter unavailable for dissolution in runoff. Through a process called podzolization, soluble organic acids play an important role in removing iron and aluminum by chelation (Schlesinger, 1997). However, organic decomposition at the floor of tropical forests is so complete that nearly no soluble organic acids percolate through the soil profile, inhibiting podzolization. The growth of

phytoplankton biomass in the East African lake appears to be nitrogen limited. Phytoplankton biomass in Lake Victoria increased only when nitrogen additions were made to the lake, either alone or in combination with phosphorus. However, phosphorus additions alone produced no detectable change in phytoplankton biomass (Lehman and Branstrator, 1994). The total nitrogen to total phosphorus ratio (average of 13 : 1 by weight for 1990 and 1992 to 1996) is slightly lower in Lake Victoria water than the ratio needed for optimum phytoplankton growth (around 16 : 1), providing more evidence for nitrogen limitation of the growth of phytoplankton biomass (Guildford and Hecky, 2000).

Thirty million inhabitants live in Lake Victoria's relatively small drainage basin $(1.94 \times 10^5 \text{ km}^2)$, making it one of the most densely populated areas of Africa (~240 individuals km^{-2}). Rapid population growth (3% yr^{-1}) is adding to already intense environmental pressures. Approximately half of the drainage basin is cultivated and in some areas, cattle densities are as high as human densities. Most (71%) of the energy consumed in sub-Saharan Africa is in the form of fuel wood due to the high poverty level of the populace and because wood gathering imposes no personal financial burden (Davidson, 1992). Therefore, most of the drainage basin has been cleared for rangeland, cultivation, and wood for household burning.

Our knowledge of the response of Lake Victoria to local environmental degradation is based on data from scattered sources: water quality measurements from 1960 to 1961 (Talling and Talling, 1965) and the early 1990s (Mugidde, 1993), and sedimentary studies of past aquatic conditions (Hecky, 1993). Due to a lack of long-term field measurements, knowledge of nutrient dynamics in tropical lakes is severely limited. However, Lake Victoria is one of a handful of tropical lakes with seasonal water quality data. Nitrogen and phosphorus deposition to Lake Victoria's sediments increased at different times: nitrogen deposition increased early in the twentieth century until the 1960s when phosphorus deposition began a rapid increase. Bio-available phosphorus and the nitrogen concentrations of the lake water were consistently higher during the rainy season in 1990–1991, compared to 1960–1961 (Figure 8). Recent increases in lake nutrient concentrations may have been caused, in part, by greater atmospheric loading as a result of wood burning and increased soil erosion. Soil disturbances from deforestation and agriculture are likely to have reduced the soil retention capacity of the drainage basin. In response to increased nutrient inputs to Lake Victoria, phytoplankton biomass (estimated as chlorophyll *a*) was over 10-fold higher in 1990–1991, than in 1960–1961.
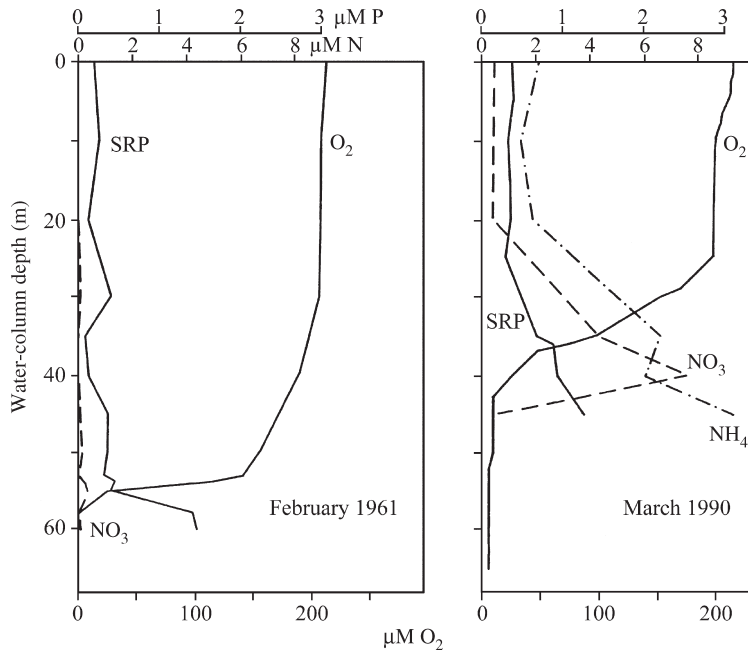
Figure 8 Depth profile of O_2, bio-available phosphorus (SRP), nitrate (NO_3), and ammonium (NH_4, 1990 only) concentrations in February 1961 and March 1990 during Lake Victoria stratification (reproduced by permission of E. Schweizerbart Science Publishers from *Verh. Int. Ver. Limnol.*, **1993**, *25*, 39–48).

Furthermore, the phytoplankton community structure has changed from a community dominated by diatoms, to one dominated by eutrophic species of cyanobacteria that produce substances toxic to humans (*Microcystis* sp., *Anabaena* sp.). Due to an increase in floating particles (e.g., phytoplankton) in the water column, water transparency and the size of the zone in which photosynthesis may occur (photic zone) have decreased. Over a period of 60 years, Secchi disk depth had dropped ~70% by 1990–1991 (Mugidde, 1993) and the area of the lake floor within the photic zone has been halved (Hecky, 1993). Decomposition of organic matter in the deep water has increased dissolved oxygen consumption. As a result, dissolved oxygen concentrations at lake depths of 40–60 m were consistently lower during the rainy season in 1990–1991 than 30 years earlier (Figure 8). Lake hypoxia is now more frequent, more persistent, and affects a greater fraction of Lake Victoria's bottom. Anoxia below a depth of 45 m now affects up to 50% of the lake's bottom area for prolonged periods of time (Hecky, 1993). Tropical water bodies are especially susceptible to dissolved oxygen depletion because of the inverse relationship between temperature and solubility of oxygen in natural waters. As temperature increases, the saturation concentration of oxygen in water decreases and the metabolic rate of decomposers (i.e., oxygen-users) increases (Lewis, 1987). Massive fish kills have been associated with cyanobacterial blooms and the

mixing of anoxic waters from deep layers (Ochumba, 1987). Due to its large volume and relatively small outflow, the water of Lake Victoria has a long flushing time (140 yr; Bootsma and Hecky, 1993). Therefore, the recovery of the lake will be slow because pollutants are retained for a long time.

9.08.3.6 Chesapeake Bay

Chesapeake Bay (Figure 2) is the largest and, historically, the most productive estuary in the continental US, and it has been studied extensively for decades. Its drainage basin (165,760 km^2) spans six states and is highly populated (15 million inhabitants). However, nutrient contributions from urban effluent are thought to be minor compared to the nonpoint source of heavy fertilization for agriculture. Agriculture accounts for about half of the total nitrogen and phosphorus loads to the Bay (Magnien *et al.*, 1995). Because of its shallow depth (mean depth ~9 m), much of the estuary's bottom historically received enough light to support submerged plant biomass. Yet, Chesapeake Bay is deep enough to stratify strongly in the summer.

The growth of phytoplankton is typically nitrogen-limited in the ocean due to enhanced sediment release of phosphorus relative to nitrogen (Caraco *et al.*, 1990), slow rates of nitrogen fixation (Howarth, 1988), and high rates of

denitrification (bacterial conversion of nitrate to nitrogen gas; Seitzinger, 1988). In the Chesapeake Bay estuary, the growth of phytoplankton is limited by nitrogen or phosphorus, depending on the season. Late winter/early spring is the period of greatest flow in Chesapeake Bay and its tributaries. Runoff from the drainage basin, rich in nitrate from fertilizer application, creates high nitrogen concentrations in Chesapeake Bay and its sub-estuaries. Therefore, the most severe phytoplankton blooms occur in the late winter/early spring season. Owing to the wealth of nitrogen, the ratio of dissolved inorganic nitrogen to phosphorus concentrations in late winter can be as high as 90 : 1 by weight and phytoplankton become phosphorus limited (D'Elia *et al.*, 1986). During the summer, when drainage basin runoff is at its lowest, nitrate concentrations in Chesapeake Bay are low and sometimes undetectable (Malone *et al.*, 1988). However, phosphorus concentrations increase due to internal release from bottom sediments through bacterial decomposition and chemical diffusion to top layers. As a result, during the summer the nitrogen to phosphorus ratio of Chesapeake Bay water can be as low as 5 : 1 by weight, and nitrogen limits the growth of phytoplankton (D'Elia *et al.*, 1986).

Eutrophication of Chesapeake Bay has been occurring since Europeans began aggressive deforestation of the drainage basin. By the end of the nineteenth century, 80% of the land was cleared, mostly for agriculture. Due to reduced transpiration potential, water export from the drainage basin increased during this time, resulting in a gradual increase in nitrogen concentrations in Chesapeake Bay. Total annual loading of nitrogen and phosphorus, mostly from terrestrial sources, have increased 6- and 17-fold, respectively, since pre-European times (Boynton *et al.*, 1995). Consequently, the eutrophication of Chesapeake waters has been increasing; the centric to pennate diatom ratio, a sedimentary indicator of eutrophication, increased from 1.3 before European settlement to 5.2 in recent sediments. Along with a doubling in human population and a tripling in inorganic fertilizer use, phytoplankton biomass (estimated as chlorophyll *a*) increased up to fivefold during the two decades prior to the 1970s. Phytoplankton biomass has remained similar or slightly higher since then and phytoplankton blooms are now more frequent and of longer duration (Harding, 1994). The neurotoxin-producing dinoflagellate *Pfiesteria piscicida* was first reported in the Bay in the early 1990s (Lewitus *et al.*, 1995) and was responsible for fish kills and impaired memory capacity in fishermen and field workers that sampled the estuary (Grattan *et al.*, 1998). Also, diatom biodiversity decreased from 100 to 68 species during the three decades before the early

1990s, a further indication of phytoplankton stress in the Bay (Cooper and Brush, 1991). Chesapeake Bay has a long history of eutrophication.

Eutrophication in Chesapeake Bay has reduced the habitats for fish, rooted aquatic plants, and bottom dwelling organisms. During blooms, phytoplankton not consumed by predators die and settle to the bottom where they is decomposed by bacteria. In response to increased phytoplankton production and sediment load from the drainage basin, sedimentation rates in Chesapeake Bay have increased from 1.5- to sevenfold since the late eighteenth century (Cooper and Brush, 1991). Anoxia associated with bacterial decomposition develops in the bottom layers of Chesapeake Bay, affecting many aquatic organisms (i.e., fish and bottom-dwellers such as oysters). The habitat of these organisms is reduced as they are forced to move to oxic parts of the estuary bottom. Seasonal anoxia has intensified between 1950 and 1980 time period and now occurs annually (Cooper and Brush, 1991; Malone *et al.*, 1988).

The distribution and abundance of submerged aquatic vegetation in Chesapeake Bay has decreased since the 1960s. Severe changes began when Tropical Storm Agnes (1972) caused a dramatic increase in sediment loading from the drainage basin to the estuary. Suspended sediment particles reduced water transparency and caused a drastic decrease in the distribution and abundance of submerged plants. A decade later, the major tributaries of the Bay were barren. Sedimentary analysis of seeds and pollen from aquatic plants of Chesapeake Bay indicates that seeds were deposited to sediments from the eighteenth century to the time of Tropical Storm Agnes; at that time they disappeared abruptly from the record (Orth and Moore, 1983). Such a major decline in aquatic plants is unprecedented in the Chesapeake Bay estuary.

Efforts to clean the Bay began when, in 1982, 1987, and 2000, the states covering most of the drainage basin (Virginia, Maryland, and Pennsylvania), the District of Columbia, the Chesapeake Bay Commission, and the US Environmental Protection Agency, signed agreements to protect and restore Chesapeake Bay's ecosystem. Part of the 1987 agreement was a commitment to achieve a 40% reduction in nitrogen and phosphorus loads to the Bay by 2000, in comparison with 1985. Bio-available phosphorus concentrations had decreased by at least half since the early 1970s due to tertiary phosphorus treatment of sewage effluent (Harding, 1994). The goals of the 2000 Agreement will require even greater reductions in point-source nitrogen loading from wastewater treatment facilities. Currently, 45% of the nitrogen flow from sewage treatment plants is treated and by 2010, 79% is to be treated. In an effort to reduce nonpoint-source nutrient loads,

2.5×10^4 acres of tidal and nontidal wetlands are to be restored by 2010. Also, by 2012, the rate of urban sprawl development of forest and agricultural land in the Chesapeake Bay drainage basin is to be reduced by 30% from the average over 1992–1997. It is too early to determine the level of success achieved by the efforts to clean up Chesapeake Bay.

9.08.4 EUTROPHICATION CONTROL

Eutrophication control methods are time- and watercourse-specific. To predict the success of methods under consideration, the predominant sources of nutrients must be identified and related to changes in water quality. Reduction of nutrient loads from point sources is often the simplest approach when nutrient enhancement stems from known sources, such as sewage. Where drainage basins were disturbed by forest harvesting, forest fires, road building, land development, agriculture, and urbanization, diffuse or nonpoint sources of nutrients may be the most important contributors to aquatic eutrophication. In water bodies, where conditions are such that anoxic water accumulates over the bottom sediments, *in situ* sources can provide most of the nutrients to the euphotic zone, a process termed internal loading. In this case, the management of internal sources may be necessary to complement external nutrient control.

9.08.4.1 Control of Concentrated (Point) Nutrient Sources

Historically, the primary method to control eutrophication has involved reducing or eliminating point sources of nutrients, mostly sewage from urban areas. Diversion of point sources, such as sewage effluent, downstream is a cost-effective method of nutrient load reduction into lakes. The outstanding improvements in water quality in Lake Washington encouraged other communities to consider this management solution. However, diversion is often not a practical management solution because it transfers the problem downstream, rather than eliminating it. Advanced wastewater treatment is a more popular point-source management solution. The addition of a tertiary stage of water treatment can precipitate phosphorus with aluminum sulfate (alum) or calcium hydroxide (slaked lime). However, tertiary treatment entails high initial capital and operational costs. It is therefore crucial to ensure that the aquatic system considered for receipt of tertiary treated water will benefit significantly from phosphorus removal. In Lake Tahoe, the initial strategy to improve water quality was to treat sewage through tertiary phosphorus removal

and then to return the effluent to the lake. However, short-term bioassays indicated that phytoplankton biomass was stimulated by nitrogen, and not phosphorus. Thus, in Lake Tahoe, the nitrogen left in effluents after tertiary treatment would continue to enhance phytoplankton biomass (Goldman, 1993). In Shagawa Lake, Minnesota, phosphorus was the primary factor controlling phytoplankton biomass and 80% of the phosphorus was derived from sewage. An advanced wastewater treatment plant was constructed. This removed 99% of the phosphorus and within two years phytoplankton biomass (estimated as chlorophyll *a*) decreased and Secchi disk depth increased by 50% (Horne and Goldman, 1994). Another method of point-source nutrient removal that has proved effective is the construction of basins, or preimpoundments, that retain nutrient-rich water for a short period of time, allowing nitrogen- and phosphorus-bearing particles to settle out of the water column prior to entering downstream waters.

9.08.4.2 Control of Diffuse (Nonpoint) Nutrient Sources

Diffuse or nonpoint nutrient sources are often most important in drainage basins dominated by agricultural activity. In the USA, agriculture is the most important source of nutrients to lakes and rivers and the third largest source of nutrients after urban runoff and municipal sewage to estuaries (Parry, 1998). Unfortunately, the control of nutrients from nonpoint sources has proved most difficult. Control of runoff from nutrient-rich sources often relies on a suite of methods, together called "BMPs". One or a combination of BMPs may be needed for pollution reduction, these include:

(i) Soil stabilization to minimize the movement of soils and attached nutrients. The addition of chemical soil stabilizers such as high molecular weight anionic polyacrylamide can reduce soil loss ninefold and phosphorus loss five- to sevenfold (Lentz *et al.*, 1998). Grassed outlets can decrease sediment suspension by running water and reduce nutrient concentrations in solution. Revegetation can improve stabilization of soil surfaces through root networks and increased soil content of heavy organics. Conservation tillage reduces the erosive energy of rainfall by leaving crop residues at the land surface. Also, in some cases, buffer strips of vegetation left along the shoreline can absorb excess nutrients and water before they escape from the drainage basin. Livestock exclusion from watercourses prevents bank erosion and direct nutrient inputs from feces.

(ii) Interruption of overland flow. Water treatment techniques such as the creation of artificial

wetlands that collect water and remove nutrients through aquatic plants. Basins can also be constructed to collect runoff water and allow settling of suspended sediment, often rich in phosphorus, before discharge downstream (Brown *et al*., 1981). Other examples include flow regulators such as energy dissipators within culverts, reduced slope angles, and soil roughening.

(iii) Changes in chemical application techniques to minimize excess nutrient availability and reduce export. A decrease in road salt usage or a change to calcium chloride usage can help to preserve roadside ditch vegetation (Goldman and Lubnow, 1992). The use of pellet fertilizers, rather than granular and liquid fertilizers, can reduce the availability of nutrients to weathering by releasing nutrients more slowly. Soil and manure testing should be completed before application to determine the need for fertilization. Ploughing manure into the soil, and the proper timing of chemical applications, such as the application of fertilizer after spring thaw instead of in the fall, can reduce the vulnerability of added nutrients to removal in surface runoff. Additions of slaked lime or alum to manure can greatly reduce nitrate volatilization and phosphorus solubility (Moore and Miller, 1994).

(iv) Reduction of nutrients at their source. Genetic measures can be employed to increase phosphorus absorption capacity of livestock. Much of the phosphorus in corn grain is present in the form of phytic acid, a phosphorus source that is not digestible in monogastric animals and thus is often excreted in waste. Corn with reduced levels of phytic acid can be isolated with chemical mutant induction. Using this method, a 65% reduction in phytic acid has been recorded with no effect on total grain phosphorus content (Ertl *et al*., 1998). Also, intensive grazing by livestock can reduce off-farm nutrient inputs because less feed is needed and imported. Nutrient inputs in feed and fertilizer exceed production outputs in crop and animal produce leaving the farm or drainage basin.

9.08.4.3 Control of Internal Nutrient Sources

Reduction of the visible symptoms of eutrophication (i.e., green scum) can be achieved by various methods. An indirect method that has received considerable attention, although it is still in the experimental phase, is biomanipulation or the manipulation of aquatic food chains to reduce phytoplankton biomass. The idea is to reduce predation on aquatic grazers such as large zooplankton, which feed on phytoplankton. This method has proven successful in increasing water transparency and dissolved oxygen concentrations and reducing phytoplankton biomass (Shapiro and

Wright, 1984; Vighi *et al*., 1995). However, stabilization of nonequilibrium populations is difficult and unpredictable over the long term (Kasprzak *et al*., 1993). Another method of aquatic vegetation reduction involves mechanical harvesting of macrophytes or surface blooms. This method provides immediate relief from conditions that impair water recreation, but it requires repeated application, is costly, and often spreads the problem out over a larger area. Chemical control of phytoplankton blooms with algicides, such as copper sulfate, can also keep the biomass of phytoplankton at a minimum. However this method requires continuous applications; the suppression effects are only temporary because previously organic-bound phosphorus is released. Further, the use of herbicides such as copper sulfate raises concerns about their possible toxicity to other biota. Complete mechanical circulation of the water column, achieved by vigorous mechanical aeration of deep waters, can reduce phytoplankton biomass by pushing the phytoplankton to greater depths, where light is insufficient for their growth.

Sediments can be an important source of nutrients. The most popular method to prevent phosphorus release from sediments is inactivation. Phosphorus inactivation involves chemical treatment of the water column with aluminum sulfate/sodium aluminate (alum) or lime to precipitate phosphorus out of the water column. Once in the bottom sediments, the precipitate can continue to adsorb phosphorus and prevent its release to the overlying water (Prepas *et al*., 2001a). Phosphorus inactivation can be highly effective: in a survey of treated American lakes, phosphorus loading from sediments was reduced by about two-thirds and treatments are expected to last 10 years and 15 years for shallow (polymictic) and deep (dimictic) lakes, respectively, before reapplication (Welch and Cooke, 1999). However, only relatively small lakes have potential for phosphorus inactivation because of the high costs of chemicals. Hypolimnetic oxygenation without destratification can eliminate the reducing environment at the sediment–water interface and diminish nutrient release from sediments. Hypolimnetic oxygenation in Amisk Lake, Alberta, from 1988 to 1993 increased dissolved oxygen concentrations in the hypolimnion fivefold to near an average of 5 mg L^{-1} in the treated basin. This change improved the habitat for fish and their food base. Also, surface water chlorophyll *a* and total phosphorus concentrations were reduced such that a lake previously classified as eutrophic was reclassified as mesotrophic (Prepas *et al*., 1997). Withdrawal of nutrient-rich, hypoxic, water from the hypolimnion can reduce the transfer of nutrients from the deep water to surface waters, thereby reducing the potential for increased

phytoplankton biomass. In 10 lakes, hypolimnetic withdrawal decreased epilimnetic total phosphorus concentration by $11\% \, \text{yr}^{-1}$ on average. Operational costs are very low, especially when passive siphoning is employed (Nürnberg, 1987). Finally, mechanical removal of sediments by dredging is a very effective but expensive method to reduce nutrient inputs to systems with high internal loads. For example, in lake Trümmen, a shallow Swedish lake, cyanobacterial biomass decreased after dredging only 0.5 m from the lake bottom (Cronberg *et al.*, 1975). The challenge of dredging, however, is to find a use, like crop fertilization, for waste sediment. Decreases in epilimnetic total phosphorus concentrations from internal loading control can often be detected within two to three years.

9.08.5 THE FUTURE—OPPORTUNITIES FOR EUTROPHICATION MANAGEMENT

The primary focus of any drainage basin management program should be to restrict external nutrient loads. With such action, lakes with small internal nutrient sources can be restored quickly, as in the case of phosphorus and in Lake Washington. However, where internal loading is a significant component of the annual nutrient load, restoration can be lengthy. For example, Shagawa Lake (Minnesota) is projected to take 80 years to achieve a 90% reduction in sediment phosphorus load (Chapra and Canale, 1991). Early action to prevent eutrophication, as was done in Lake Tahoe, can minimize accumulation of phosphorus-rich sediments. Otherwise, the removal of stockpiled phosphorus in bottom sediments can require a long and costly cleanup. There are too few Lake Washingtons and Lago Maggiores, where resources were relatively abundant to assess and to remedy the problem.

Worldwide, the greatest unresolved water quality concerns are in countries with the fewest resources, where demand for water is growing and research and restoration funding sources are scarce. The fate of aquatic resources in the world's poorest countries should be of concern to all countries. Because the toxicity of drinking water often increases with nutrient enrichment, excess nutrients should be treated as toxic substances and banned. There is an urgent need for the universal regulation of nutrients to protect drinking water supplies and aquatic biodiversity.

REFERENCES

Ambrosetti W. and Barbanti L. (1999) Deep water warming in lakes: an indicator of climatic change. *J. Limnol.* **58**, 1–9.

Ambrosetti W., Barbanti L., Mosello R., and Pugnetti A. (1992) Limnological studies on the deep southern Alpine lakes Maggiore, Lugano, Como, Iseo, and Garda. *Mem. Ist. Ital. Idrobiol.* **50**, 117–146.

Bartish T. (1987) A review of exchange processes among the three basins of Lake Erie. *J. Great Lakes Res.* **13**, 607–618.

Bartsch A. F. (1972) Nutrients and eutrophication—prospects and options for the future. In *Nutrients and Eutrophication: The Limiting Nutrient Controversy* (ed. G. E. Likens). The American Society of Limnology and Oceanography, Lawrence, Kansas, pp. 297–300.

Beeton A. M. (1961) Environmental changes in Lake Erie. *Trans. Am. Fish. Soc.* **90**, 153–159.

Beeton A. M. (1963) *Limnological Survey of Lake Erie 1959 and 1960. Great Lakes Fisheries Commission*, Ann Arbor, MI, Report no. 6.

Beeton A. M. (1965) Eutrophication of the St. Lawrence Great Lakes. *Limnol. Oceanogr.* **10**, 240–254.

Bertram P. E. (1993) Total phosphorus and dissolved oxygen trends in the central basin of Lake Erie, 1970–1991. *J. Great Lakes Res.* **19**, 224–236.

Bootsma H. A. and Hecky R. E. (1993) Conservation of the African great lakes: a limnological perspective. *Conserv. Biol.* **7**, 644–656.

Bosch J. M. and Hewlett J. D. (1982) A review of catchment experiments to determine the effect of vegetation changes on water yield and evapo-transpiration. *J. Hydrol.* **55**, 3–23.

Boynton W. R., Garber J. H., Summers R., and Kemp W. M. (1995) Inputs, transformations, and transport of nitrogen and phosphorus in Chesapeake Bay and selected tributaries. *Estuaries* **18**, 285–314.

Brown M. J., Bondurant C. E., and Brockway C. E. (1981) Ponding surface drainage water for sediment and phosphorus removal. *Trans. ASAE* **24**, 1478–1481.

Caraco N., Cole J., and Likens G. E. (1990) A comparison of phosphorus immobilization in sediments of freshwater and coastal marine systems. *Biogeochemistry* **9**, 277–290.

Chapra S. C. and Canale R. P. (1991) Long-term phenomenological model of phosphorus and oxygen for stratified lakes. *Water Res.* **25**, 707–715.

Codd G. A. (1995) Cyanobacterial toxins: occurrence, properties and biological significance. *Water Sci. Technol.* **32**, 149–156.

Cooper S. R. and Brush G. S. (1991) Long-term history of Chesapeake Bay Anoxia. *Science* **254**, 992–996.

Cronan C. S. and Aiken G. R. (1985) Chemistry and transport of soluble humic substances in forested watersheds of the Adirondack Park, New York, USA. *Geochim. Cosmochim. Acta* **49**, 1697–1706.

Cronberg G., Gelin C., and Larsson K. (1975) Lake Trummen restoration project: II. Bacteria, phytoplankton and phytoplankton productivity. *Verh. Int. Ver. Limnol.* **19**, 1088–1096.

Crosby J. M., Bradford M. E., Mitchell P. A., Prepas E. E., McIntyre L. G., Hart Buckland-Nicks L., and Hanson J. M. (1990) In *Atlas of Alberta Lakes* (eds. P. Mitchell and E. Prepas). University of Alberta Press, Edmonton, Alberta.

Crumpton W., Isehart G. M., and Mithell P. L. (1992) Nitrate and organic N analyses with second-derivative spectroscopy. *Limnol. Oceanogr.* **37**, 907–913.

D'Arcy P. and Carignan R. (1997) Influence of catchment topography on water chemistry in southeastern Québec Shield lakes. *Can. J. Fish. Aquat. Sci.* **54**, 2215–2227.

Davidson O. (1992) Energy issues in sub-Saharan Africa: future directions. *Ann. Rev. Energy Environ.* **17**, 359–404.

Davis C. C. (1964) Evidence for the eutrophication of Lake Erie from phytoplankton records. *Limnol. Oceanogr.* **9**, 275–283.

de Bernardi R., Giussani G., Manca M., and Ruggiu D. (1988) Long-term dynamics of plankton communities in Lago Maggiore (N. Italy). *Verh. Int. Ver. Limnol.* **23**, 729–733.

D'Elia C. F., Sanders J. G., and Boynton W. R. (1986) Nutrient enrichment studies in a coastal plain estuary: phytoplankton growth in large-scale, continuous cultures. *Can. J. Fish. Aquat. Sci.* **43**, 397–406.

Dillon P. J. and Kirchner W. B. (1975) The effects of geology and land use on the export of phosphorus from watersheds. *Water Res.* **9**, 135–148.

Dolan D. M. (1993) Point-source loadings of phosphorus to Lake Erie: 1986–1990. *J. Great Lakes Res.* **19**, 212–223.

Dunn J. (1996) Algae kills dialysis patients in Brazil. *Br. Med. J.* **312**, 1183–1184.

Edmondson W. T. (1969) Eutrophication in North America. In *Eutrophication: Causes, Consequences, Correctives*, Publ. 1700. National Academy of Science/National Research Council, Washington, DC, pp. 129–149.

Edmondson W. T. (1975) Recovery of Lake Washington from eutrophication. In *Recovery and Restoration of Damaged Ecosystems* (eds. J. Cairns, Jr., K. L. Dickson, and E. E. Henricks). University Press of Virginia, Charlottesville, VA, pp. 102–109.

Edmondson W. T. (1994) Sixty years of Lake Washington: a curriculum vitae. *Lake Reserv. Manage.* **10**, 75–84.

Ertl D. S., Young K. A., and Raboy V. (1998) Plant genetic approaches to phosphorus management in agricultural production. *J. Environ. Qual.* **27**, 299–304.

Goldman C. R. (1993) Failures, successes and problems in controlling eutrophication. *Mem. Ist. Ital. Idrobiol.* **52**, 79–87.

Goldman C. R. and Lubnow F. S. (1992) Seasonal influence of calcium magnesium acetate on microbioal processes in 10 northern Californian lakes. *Resour. Conserv. Recy.* **7**, 51–67.

Goldman C. R., Jassby A. D., and Hackley S. H. (1993) Decadal, interannual, and seasonal variability in enrichment bioassays at Lake Tahoe, California–Nevada, USA. *Can. J. Fish. Aquat. Sci.* **50**, 1489–1496.

Grattan L. M., Oldach D., Perl T. M., Lowitt M. H., Matuszak D. L., Dickson C., Parrott C., Shoemaker R. C., Kaufman C. L., Wasserman M. P., Hebel J. R., Charache P., and Morris J. G., Jr. (1998) Learning and memory difficulties after environmental exposure to waterways containing toxin-producing *Pfiesteria* or *Pfiesteria*-like dinoflagellates. *Lancet* **352**, 532–539.

Guildford S. J. and Hecky R. E. (2000) Total nitrogen, total phosphorus, and nutrient limitation in lakes and oceans: is there a common relationship? *Limnol. Oceanogr.* **45**, 1213–1223.

Harding L. W., Jr. (1994) Long-term trends in the distribution of phytoplankton in Chesapeake Bay: roles of light, nutrients, and streamflow. *Mar. Ecol. Prog. Ser.* **104**, 267–291.

Hecky R. E. (1993) The eutrophication of Lake Victoria, East Africa. *Verh. Int. Ver. Limnol.* **25**, 39–48.

Hobbie J. E. and Likens G. E. (1973) Output of phosphorus, dissolved organic carbon, and fine particulate carbon from Hubbard Brook watersheds. *Limnol. Oceanogr.* **18**, 734–742.

Horne A. J. and Goldman C. R. (1994) *Limnology*. McGraw-Hill, New York, NY.

Howarth R. W. (1988) Nutrient limitation of net primary production in marine ecosystems. *Ann. Rev. Ecol. Syst.* **19**, 89–110.

Hutchinson G. E. (1973) Eutrophication. *Am. Sci.* **61**, 269–279.

Istituto Italiano di Idrobiologia—CNR (2001) Ricerche sull'evoluzione del Lago Maggiore. Aspetti limnologici. Programma quinquennale 1998–2002. Campagna 2000. (ed. Commissione Internazionale per la protezione delle acque italo-svizzere). Ist. Ital. Idrobiol.—CNR, 77pp.

Jassby A. D., Reuter J. E., Axler R. P., Goldman C. R., and Hackley S. H. (1994) Atmospheric deposition of nitrogen and phosphorus in the annual nutrient load of Lake Tahoe (California–Nevada). *Water Resour. Res.* **30**, 2207–2216.

Jassby A. D., Goldman C. R., and Reuter J. E. (1995) Long-term change in Lake Tahoe (California–Nevada, USA) and its relation to atmospheric deposition of algal nutrients. *Arch. Hydrobiol.* **135**, 1–21.

Jassby A. D., Goldman C. R., Reuter J. E., Richards R. C., and Heyvaert A. C. (2001) Lake Tahoe: diagnosis and rehabilitation of a large mountain lake. In *The Great Lakes of the World (GLOW): Food-web, Health and Integrity* (eds. M. Munawar and R. E. Hecky). Backhuys Publishers, Leiden, The Netherlands, pp. 431–454.

Jenny H. (1980) *The Soil Resource*. Springer, New York.

Kalff J. (2002) *Limnology*. Prentice Hall, New Jersey.

Kasprzak P., Krienitz L., and Koschel R. (1993) Biomanipulation: a limnological in-lake ecotechnology of eutrophication management? *Mem. Ist. Ital. Idrobiol.* **52**, 151–169.

Lehman J. T. and Branstrator D. K. (1994) Nutrient dynamics and turnover rates of phosphate and sulfate in Lake Victoria, East Africa. *Limnol. Oceanogr.* **39**, 227–233.

Lentz R. D., Sojka R. E., and Robins C. W. (1998) Reducing phosphorus losses from surface-irrigated fields: emerging polyacrylamide technology. *J. Environ. Qual.* **27**, 305–312.

Lewis W. M., Jr. (1987) Tropical limnology. *Ann. Rev. Ecol. Syst.* **18**, 159–184.

Lewitus A. J., Hesian R. V., Kana T. M., Burkholder J. M., Glasgow H. B., and May E. (1995) Discovery of the "phantom" dinoflagellate in the Chesapeake Bay. *Estuaries* **18**, 373–378.

Ludsin S. A., Kershner M. W., Blocksom K. A., Knight R. L., and Stein R. A. (2001) Life after death in Lake Erie: Nutrient controls drive fish species richness, rehabilitation. *Ecol. Appl.* **11**, 731–746.

Magnien R., Boward D., and Bieber S. (1995) *The State of the Chesapeake 1995*. USEPA, Annapolis, MD.

Makarewicz J. C. (1993) Phytoplankton biomass and species composition in Lake Erie, 1970 to 1987. *J. Great Lakes Res.* **19**, 258–274.

Makarewicz J. C. and Bertram P. (1991) Evidence for the restoration of the Lake Erie ecosystem. *Bioscience* **41**, 216–223.

Makarewicz J. C., Lewis T. W., and Bertram P. (1999) Phytoplankton composition and biomass in the offshore waters of Lake Erie: pre- and post-*Dreissena* introduction (1983–1993). *J. Great Lakes Res.* **25**, 135–148.

Malone R. C., Crocker L. H., Pike S. E., and Wendler B. W. (1988) Influences of river flow on the dynamics of phytoplankton production in a partially stratified estuary. *Mar. Ecol. Prog. Ser.* **48**, 235–249.

Manca M., Calderoni A., and Mosello R. (1992) Limnological research in Lago Maggiore: studies on hydrochemistry and plankton. *Mem. Ist. Ital. Idrobiol.* **50**, 171–200.

Moore P. A., Jr. and Miller D. M. (1994) Decreasing phosphorus solubility in poultry litter with aluminum, calcium, and iron amendments. *J. Environ. Qual.* **23**, 325–330.

Mosello R. and Ruggiu D. (1985) Nutrient load, trophic conditions and restoration prospects of Lake Maggiore. *Int. Rev. ges. Hydrobiol.* **70**, 63–75.

Mosello R., Barbieri A., Brizzio M. C., Calderoni A., Marchetto A., Passera S., Rogora M., and Tartari G. (2001) Nitrogen budget of Lago Maggiore: the relative importance of atmospheric deposition and catchment sources. *J. Limnol.* **60**, 27–40.

Mugidde R. (1993) Changes in phytoplankton primary productivity and biomass in Lake Victoria (Uganda). *Int. Ver. Theor. Angew. Limnol. Verh.* **25**, 846–849.

Mugidde R. (2001) Nutrient status and planktonic nitrogen fixation in Lake Victoria, Africa. PhD Thesis, University of Waterloo.

Munawar M. and Munawar I. F. (1976) A lakewide study of phytoplankton biomass and its species composition in Lake Erie, April–December 1970. *J. Fish. Res. Board Can.* **33**, 581–600.

Nürnberg G. K. (1987) Hypolimnetic withdrawal as lake restoration technique. *J. Environ. Eng.* **113**, 1006–1017.

Ochumba P. B. O. (1987) Periodic massive fish kills in the Kenyan part of Lake Victoria. *Water Qual. Bull.* **12**, 119–122.

Ochumba P. B. O. (1996) Measurement of water currents, temperature, dissolved oxygen and winds on the Kenyan Lake Victoria. In *The Limnology, Climatology, and Paleoclimatology of the East African Lakes* (eds. T. C. Johnson and E. O. Odata). Gordon and Breach, Australia, pp. 155–167.

OECD (Organization for Economic Cooperation and Development) (1982) *Eutrophication of Waters: Monitoring, Assessment, and Control*. OECD, Paris.

Orth R. J. and Moore K. A. (1983) Chesapeake Bay: an unprecedented decline in submerged aquatic vegetation. *Science* **222**, 51–53.

Parry R. (1998) Agricultural phosphorus and water quality: a US Environmental Protection Agency perspective. *J. Environ. Qual.* **27**, 258–261.

Prepas E. E. and Rigler F. H. (1982) Improvements in quantifying the phosphorus concentration in lake water. *Can. J. Fish. Aquat. Sci.* **39**, 822–829.

Prepas E. E., Field K. M., Murphy T. P., Johnson W. L., Burke J. M., and Tonn W. M. (1997) Introduction to the Amisk Lake Project: oxygenation of a deep, eutrophic lake. *Can. J. Fish. Aquat. Sci.* **54**, 2105–2110.

Prepas E. E., Pinel-Alloul B., Chambers P. A., Murphy T. P., Reedyk S., Sandland G., and Serediak M. (2001a) Lime treatment and its effects on the chemistry and biota of hardwater eutrophic lakes. *Freshwater Biol.* **46**, 1049–1060.

Prepas E. E., Planas D., Gibson J. J., Vitt D. H., Prowse T. D., Dinsmore W. P., Halsey L. A., McEachern P. M., Paquet S., Scrimgeour G. J., Tonn W. M., Paszkowski C. A., and Wolfstein K. (2001b) Landscape variables influencing nutrients and phytoplankton communities in Boreal Plain lakes of northern Alberta: a comparison of wetland- and upland-dominated catchments. *Can. J. Fish. Aquat. Sci.* **58**, 1286–1299.

Provini A., Marchetti R., and Tartari G. (1992) The Italian lakes: trophic status and remedial measures. *Mem. Ist. Ital. Idrobiol.* **50**, 147–169.

Rawson D. S. (1960) A limnological comparison of twelve large lakes in Northern Saskatchewan. *Limnol. Oceanogr.* **5**, 195–211.

Redfield A. C. (1958) The biological control of chemical factors in the environment. *Am. Sci.* **46**, 205–221.

Riley E. T. and Prepas E. E. (1984) Role of internal phosphorus loading in two shallow, productive lakes in Alberta, Canada. *Can. J. Fish. Aquat. Sci.* **41**, 845–855.

Roby K. B. and Azuma D. L. (1995) Changes in a reach of a northern California stream following wildfire. *Environ. Manage.* **19**, 591–600.

Rosa F. (1987) Lake Erie central basin total phosphorus trend analysis from 1968 to 1982. *J. Great Lakes Res.* **13**, 684–696.

Ruggiu D. and Mosello R. (1984) Nutrient levels and phytoplankton characteristics in the deep southern alpine lakes. *Verh. Int. Ver. Limnol.* **22**, 1106–1112.

Ruttner F. (1952) *Fundamentals of Limnology*, 3rd edn. (Translated in 1953 by D. G. Frey and F. E. S. Fry). University of Toronto Press, Toronto, ON.

Ryding S.-O. and Rast W. (1989) *The Control of Eutrophication of Lakes and Reservoirs*. UNESCO, Paris.

Scheffer V. B. and Robinson R. J. (1939) A limnological study of Lake Washington. *Ecol. Monogr.* **9**, 95–143.

Scheren P. A. G. M., Zanting H. A., and Lemmens A. M. C. (2000) Estimation of water pollution sources in Lake Victoria, East Africa: application and elaboration of the rapid assessment methodology. *J. Environ. Manage.* **58**, 235–248.

Schindler D. W. (1977) Evolution of phosphorus limitation in lakes. *Science* **195**, 260–262.

Schlesinger W. H. (1997) *Biogeochemistry: An Analysis of Global Change*. Academic Press, San Diego, CA.

Seitzinger S. P. (1988) Denitrification in freshwater and coastal marine ecosystems: ecological and geochemical significance. *Limnol. Oceanogr.* **33**, 702–724.

Shapiro J. and Wright D. I. (1984) Lake restoration by biomanipulation. *Freshwater Biol.* **14**, 371–384.

Silbergeld E. K., Grattan L., Oldach D., and Morris J. G. (2000) *Pfiesteria*: harmful algal blooms as indicators of human: ecosystem interactions. *Environ. Res.* **82**, 97–105.

Smith V. H. (1986) Light and nutrient effects on the relative biomass of blue-green algae in lake phytoplankton. *Can. J. Fish. Aquat. Sci.* **43**, 148–153.

Snell E. A. (1987) Wetland distribution and conversion in southern Ontario. Working Paper No. 48. Inland Waters/Lands Directorate, Environment Canada.

Søballe D. M. and Kimmel S. T. (1987) Advection, phytoplankton, biomass, and nutrient transformations in a rapidly flushed impoundment. *Arch. Hydrobiol.* **105**, 187–203.

Talling J. F. and Talling I. B. (1965) The chemical composition of African lake waters. *Int. Rev. ges. Hydrobiol.* **50**, 421–463.

Trimbee A. M. and Prepas E. E. (1987) Evaluation of total phosphorus as a predictor of the relative biomass of blue-green algae with emphasis on Alberta lakes. *Can. J. Fish. Aquat. Sci.* **44**, 1337–1342.

Vallentyne J. E. and Thomas N. A. (1978) *Fifth Year Review of Canada–United States Great Lakes Water Quality Agreement*. International Joint Commission, Windsor, ON.

Van den Hoeck C., Mann D. G., and Jahns H. M. (1995) *Algae: An Introduction to Phycology*. Cambridge University Press, Cambridge, UK.

Vighi M., Sandroni D., and Ferri G. (1995) Biomanipulation of trophic chain in two small eutrophic lakes. *Mem. Ist. Ital. Idrobiol.* **53**, 157–175.

Watson S. B., McCauley E., and Downing J. A. (1997) Patterns in phytoplanktontaxonomic composition across temperate lakes of differing nutrient status. *Limnol. Oceanogr.* **42**, 487–495.

Welch E. B. and Cooke G. D. (1999) Effectiveness and longevity of phosphorus inactivation with Alum. *Lake Reserv. Manage.* **15**, 5–27.

Yin X. and Nicholson S. R. (1998) The water balance of Lake Victoria. *Hydrol. Sci. J.* **43**, 789–811.