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Perspectives on the eutrophication of the Yahara lakes

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Abstract

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Eutrophication of the four Yahara lakes—Mendota, Monona, Waubesa, and Kegonsa—near Madison, Wisconsin, has been dramatic since the mid-1800s. For Lake Mendota, the erosion of sediments from higher water levels established by the damming of the lake's outlet, plus the agricultural expansion of its watershed, resulted in blue-green algal growths. These impacts, however, were dwarfed by water quality problems stemming from Madison's wastewater inputs that directly entered Lake Monona from the late 1800s through 1936, and then Lake Waubesa until 1958. Blue-green algal blooms were so bad in the lower Yahara lakes that the Madison Public Health Department conducted major copper sulfate treatments during 1925–1954. During the wastewater input years, inorganic nitrogen (N) and especially dissolved reactive phosphorus (DRP) concentrations in the surface waters were very high (particularly in Waubesa and Kegonsa), indicating neither nutrient was limiting algal growth. No P legacy from the wastewater inputs was found in Waubesa and Kegonsa's sediments; minimal P-binding potential due to low iron (Fe) availability is the hypothesized reason. Mendota's algal blooms were not a problem until the mid-1940s when wastewater inputs from upstream communities increased as well as the agricultural use of N and P fertilizers. This increase in eutrophication symptoms coincided with an increase in indices of DRP and inorganic N concentrations in the lake. After wastewater diversion in 1971, blue-green algal blooms persisted in Lake Mendota, and the onus of the problem shifted to agricultural and urban nonpoint source pollution. While much progress has been made in recent years to control these pollution sources to Mendota, manure runoff during late winter continues as a management problem. As evidence, P loadings during January to March constituted 48% of total loadings measured for 1990-2006 in the Yahara River subwatershed. Much of this runoff P was dissolved and not associated with high sediment loads, whereas during other months, more of the runoff P was bound to sediments that could settle out in lower stream reaches prior to entering the lake. However, low P-binding potential of recently deposited sediments in Mendota along with signs of water quality improvements following periods of drought indicate the lake could respond rapidly to nutrient input reductions. Finally, DRP and inorganic N concentrations since 1980 have indicated that algal growth in the Yahara lakes during July–August may have been limited by not only P, but N (especially in the lower Yahara lakes). Aggressive programs to reduce inputs from both nutrients will be important to prevent scum-forming blue-green algal blooms and filamentous algal growths that could become problematic once zebra mussels become established in the Yahara lakes.

Key words: blue-green algae, eutrophication, Kegonsa, Mendota, Monona, nitrogen, nonpoint source pollution, phosphorus, wastewater, Waubesa, Yahara lakes

The Yahara River chain of lakes—Mendota, Monona, Waubesa, and Kegonsa—are located near Madison in south central Wisconsin, USA (Fig. 1). Lake Mendota is one of the world's most studied lakes, stemming from the pioneering limnological research by E.A. Birge begun in 1894 and then with C. Juday at the University of Wisconsin (UW), a tradition of limnological research that continues to this day (Kitchell 1992, Magnuson 2002, Carpenter *et al.* 2006, 2007). Much is known about Lake Mendota because of this research, but fewer studies have been conducted on the other three Yahara lakes (hereafter referred to as the lower Yahara lakes), even though the lakes have been intensely managed to combat eutrophication problems from dense blooms of blue-green algae as well as excessive growths of aquatic macrophytes since the early 1900s (Lathrop *et al.* 1992). However, lake management efforts that began in earnest in

1925 by the City of Madison in response to the massive fertilization from poorly treated sewage discharges to the lower Yahara lakes precipitated an important 25-year monitoring program of all four lakes. Lake sampling on a regular basis was initiated again by the UW and the Wisconsin Department of Natural Resources (WDNR) in the 1970s and is ongoing through collaboration of WDNR and the UW's North-Temperate Lakes Long-Term Ecological Research (NTL-LTER) program. This combination of research, monitoring and management provides an interesting long-term perspective on lake eutrophication.

Not all aspects of the Yahara lakes eutrophication are discussed here. I present, in a historical context, long-term nutrient data and other pertinent information compiled from numerous sources (many as raw data records or unpublished reports). Caveats about the reliability of the long-term data are also presented, along with interpretations to help unravel the eutrophication history of the Yahara lakes. Many of these interpretations have been presented elsewhere, but some surprises are noted. Hopefully the presentation and interpretation of the long-term data will be of interest and value to those studying and managing other eutrophic lakes, as well as provide insights for future research and management activities on the Yahara lakes as they face new challenges from climate change, altered land use, and invasive species.

Lake and watershed characteristics

The watershed of the four Yahara lakes encompasses 996 km^2 of gently rolling to hilly, glaciated terrain (Lathrop 1992a). Lake Mendota's watershed is largely agricultural with urban land uses around the western, southern, and eastern shorelines; urban development is also expanding rapidly throughout the watershed. The direct drainage area to Lake Monona (downstream of Mendota) is mostly urban; the direct drainage areas of lakes Waubesa and Kegonsa are predominantly agricultural, although much of the lake shorelines have dwellings.

Lake Mendota is the largest and deepest lake with an extensive pelagic zone; Waubesa and Kegonsa are significantly shallower, while Monona is intermediate in depth (Table 1). Water temperatures of the surface and bottom waters for July 1995–2006 indicate the strength of thermal stratification in the four lakes. The shallower lakes have a much higher propensity for internal recycling of nutrients from the bottom sediments, indicated by the smaller temperature differences between surface and bottom waters when compared to the deeper lakes. The ranges in annual temperature values indicate the variability in climate affecting the lakes. The lower Yahara lakes also have significantly more rapid flushing rates due to the smaller lake volumes and greater river discharges downstream when compared to Mendota.

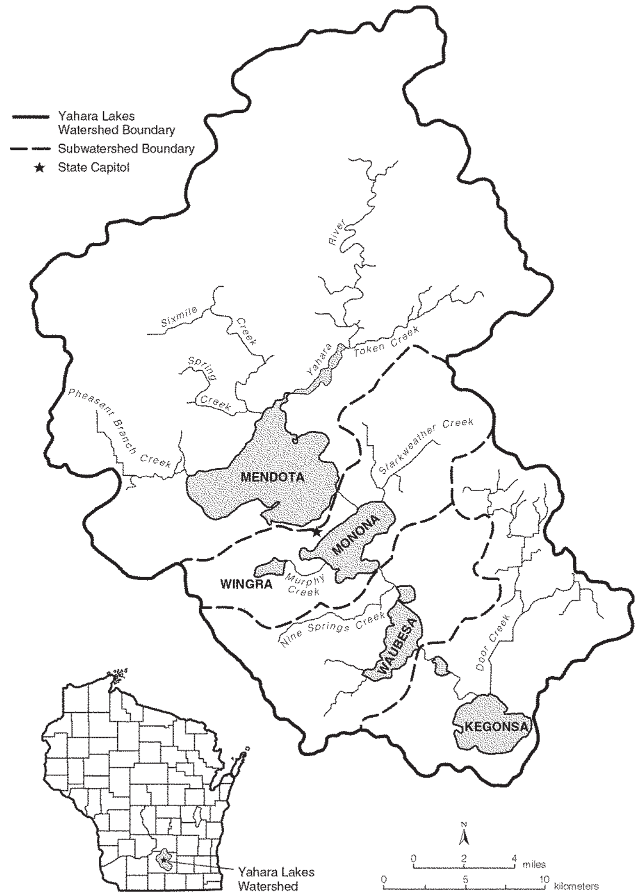

Figure 1.-The Yahara River chain of lakes (Mendota, Monona, Waubesa, Kegonsa) and their watersheds and major tributary streams near Madison, Wisconsin.

Lake monitoring data sources

While the Birge and Juday research conducted during the early 1900s (Frey 1963) forms the earliest scientific database of algal densities (based primarily on unpublished Secchi disk readings) in Lake Mendota (Lathrop 1992b, Lathrop *et al.* 1996), their research focus (especially during the summer) had shifted to northern Wisconsin by 1925 when the UW Trout Lake field station was established. Serendipitously, it was the water quality problems of the lower Yahara lakes (initially Lake Monona and then later lakes Waubesa and Kegonsa) caused by the over fertilization from Madison's poorly treated sewage that motivated the Madison Public Health Dept. (MPHD) to initiate a regular monitoring program on all four Yahara lakes.

This monitoring program had a principle goal of evaluating the responses to whole-lake copper sulfate treatments that were fully initiated by the MPHD in 1925 on Monona and later on Waubesa and Kegonsa to control the noxious summer growths of blue-green algae stimulated by the waste-

Table 1.-Physical characteristics of the Yahara lakes.

	Mendota	Monona	Waubesa	Kegonsa
Surface area (ha)	3,985	1,326	843	1,299
Depth (m)				
Maximum	25.3	22.6	11.6	9.8
Mean	12.7	8.3	4.7	5.1
Flushing rate (yr^{-1})*	0.15	0.91	3.2	2.2
Water temperature (°C)**				
Surface July maximum	23-26	24-28	24-28	24-28
Bottom July minimum	10-12	11-14	15-21	17-25
Watershed				
Direct drainage area (km^2)	562	105	113	141
Drainage area at outlet (km^2)	602	720	842	996

* Flushing rates for Monona, Waubesa and Kegonsa from Lathrop (1979); flushing rate for Mendota from Lathrop *et al.* 1998. Higher discharges in the Yahara River system in recent years would increase listed flushing rates especially for lower Yahara lakes.

**Range of annual max./min. temperatures for 1995–2006 (Source: NTL-LTER database).

water nutrients. Interestingly, Mendota was perceived as the "control" because its water quality problems were minor compared to the lower Yahara lakes, at least until the mid-1940s. The monitoring data were not widely known because the information was considered incriminating to the role of Madison's wastewater in causing the severe lake eutrophication problems in the lower Yahara lakes during the contentious period when legal and budgetary battles were preventing the diversion of the wastewater out of the lakes.[1]

Concern about the sources of nutrients causing the algal problems in the lower Yahara lakes came to a head in the early 1940s with a two-year water quality study that conclusively documented the role of Madison's wastewater (Sawyer *et al.* 1945). By the mid-1940s, however, Lake Mendota's algal blooms for the first time caused efforts to be focused on that lake's sources of nutrients, which were from agricultural and urban runoff as well as wastewater effluents from small communities in the lake's watershed (Hasler 1947, Bartsch and Lawton 1949, Lathrop 1992b). These studies, conducted initially by government scientists with water analyses performed at the State Laboratory of Hygiene, were continued as UW engineering student projects for a few additional years.

The early water quality monitoring of the lower Yahara lakes in general ended when the MPHD's program stopped after 1949, except for a brief period when the wastewater effluent diversion around the lakes occurred in 1958 (Clesceri 1961). Monitoring on a limited basis began again by the WDNR on Monona in 1967 and Waubesa and Kegonsa in 1973. Water quality investigations on Mendota were restarted in the mid-1960s by the UW Water Chemistry Program; research conducted in the early 1970s (Torrey 1972, Sonzongni 1974, Stauffer 1974, Vigon 1976) provide the most complete and reliable database of lake nutrients to that time.

Beginning in 1976, the WDNR began a regular limnological sampling program on all four lakes, a program that lasted through 1994. A seamless transition occurred in 1995 when the UW's NTL-LTER project (with WDNR help) began conducting ecosystem research on Mendota and Monona (Carpenter *et al.* 2006, 2007), while maintaining the basic water quality sampling on Waubesa and Kegonsa.

Except for data obtained for Lake Mendota by UW students in the early 1970s, most data collected prior to 1980 were not reliable enough at low levels for total phosphorus (P) and sometimes other nutrients to be useful for detecting trends in the Yahara lakes' surface waters not polluted with wastewater (Lathrop 1992b). Based on testing of the MPHD's laboratory methods used during their 1925–1949 monitoring program, I concluded that total P (first analyzed in 1936) was not a reliable indicator of summer trophic conditions in Lake Mendota's relatively unpolluted surface waters during those years (Lathrop 1992b). Dissolved reactive (inorganic) phosphorus (DRP), however, was a useful indicator of P trends in all lakes because concentrations ≤ 0.009 mg/L were reliably tested with many reported results each year being 0.002–0.005 mg/L for unpolluted surface waters.

While much higher DRP concentrations routinely occurred on all lakes during the fall, winter and spring months, very low

[1] This statement is based on my interpretations of a letter I read in the late 1970s while inspecting old water quality records at the former headquarters of the Madison Metropolitan Sewerage District (MMSD). The letter was written around 1949 by the Director of the Madison Public Health Department (MPHD) and sent with a compilation of their lake monitoring data (1925–1947) to MMSD. Recent efforts to locate and copy the letter for archival purposes were unsuccessful.

DRP concentrations in the summer surface waters indicate if the lake was P-limited during the period when blue-green algal blooms occurred. Elevated DRP concentrations clearly indicate algal growth was not limited by P, whereas very low (ideally undetectable) DRP may indicate P limitation as algae can quickly incorporate any dissolved P as soon as it becomes available.

Nitrogen may also have been limiting algal growth (either by itself or co-limitation with P) during the summer months when concentrations were low. Early analytical methods for the various nitrogen forms were not investigated because "standard methods" were available (Lathrop 1992b); however, MPHD detection limits for inorganic N, which was the summation of three separate analytical procedures [ammonium (NH_4-N), nitrate (NO_3-N), and nitrite (NO_2-N) nitrogen] were not as low as modern laboratory capabilities. Because the MPHD testing was geared toward evaluating the wastewater effluent effects on the lower Yahara lakes, years with "low" concentrations of inorganic N found in Mendota can only indicate possible N limitation for algal growth, whereas much higher concentrations found in the other lakes can more reliably indicate N was not limiting. These interpretations recognize that DRP and inorganic nitrogen (N) concentrations can only give an indication of nutrient growth limitation and as such are not absolute proof, as Dodds (2003) has cautioned. To more accurately determine which nutrient is growth-limiting to algae, a combination of algal particulate composition ratios and three metabolic indicators may be needed (Guildford and Hecky 2002).

The most reliable period of record for both total P, DRP and inorganic N on all four Yahara lakes began after 1980 when all water chemistry analyses were performed at the Wisconsin State Laboratory of Hygiene (SLOH), a federally certified laboratory where modern analytical methods, instrumentation, and quality control procedures have been employed that include strict sample-holding times and preservation methods. Analytical capabilities and detection limits also have improved at the SLOH over the years. While detection limits for DRP were <0.004 mg/L since 1980, detection limits dropped in 1989 to <0.002. The most dramatic change in detection limits occurred in 1982, however, when the lab began reporting total P in thousandths of mg per liter compared to hundredths of mg per liter in prior years; detection limits for total P likewise declined from <0.02 to <0.009, and later to <0.007 mg/L. For inorganic N analyses at the SLOH since 1980, detection limits for both NH_4-N and NO_3-N/NO_2-N (combined analysis) dropped from <0.02 mg/L to about <0.007 mg/L in 1991. Thus, I believe the most reliable P and N limitation values in the Yahara lakes are since 1980, with caveats that lower detection limits, especially on inorganic N, make interpretations even more reliable since 1991. (In this study, undetectable inorganic N and DRP concentrations were plotted at one-half their respective detection limits.)

External P-loading analyses have been an important focus of research on the Yahara lakes for many years, especially for Lake Mendota (Sonzogni 1974, Lathrop 1979, Lathrop *et al.* 1998). Phosphorus loadings determined in the late 1940s were based on an extensive continuous stream discharge monitoring network established by the UW Civil Engineering Department. Unfortunately, water sampling conducted until the early 1950s was on an infrequent routine schedule where runoff events were not generally sampled, so those early tributary P loadings were likely underestimated (Lathrop 1979). In the 1960s and early 1970s, average P loadings were estimated from watershed export coefficients (Sonzogni 1974, including unpublished estimates of Prof. G. Fred Lee, UW Water Chemistry Program), but the watershed's glaciated terrain with small internally drained areas complicated those estimates (Lathrop 1979).

In 1976, a watershed monitoring program was initiated with the U.S. Geological Survey (USGS) establishing continuous-flow gaging stations on the major tributaries and storm sewers entering Lake Mendota. Runoff event sampling was also emphasized, which enabled more reliable estimates of annual P loadings on the monitored inflows (Lathrop *et al.* 1998). Fortunately, long-term monitoring has been maintained on two streams and one storm sewer so that the annual variability in P loading to Lake Mendota could be estimated and its effects on lake water quality modeled (Lathrop *et al.* 1998). But even those P-loading estimates for the whole lake are problematic because the flow gaging stations were established well upstream where lake backwater would not affect the water level-discharge relationship derived for each monitoring station. Phosphorus loadings for the unmonitored areas nearer the lake were estimated by applying each year's unit area monitored loadings to the unmonitored areas (Lathrop *et al.* 1998).

More recent USGS-monitored P loadings that were determined for the actual inflow of the Yahara River to Lake Mendota downstream of the wide river estuary (Fig. 1) indicate the estuary traps large amounts of sediment-bound P during high flow events, while releasing P during periods of dry weather flow. This causes the log-normal distribution of annual P loadings estimated from the upstream monitoring station data (Lathrop *et al.* 1998) to be compressed; P loadings to the lake during high runoff years are actually less than earlier estimates, while P loadings during low runoff years are somewhat higher. A more complete analysis of the role of these lower stream reaches on actual P loadings to the lake is currently being conducted.

For this study I used annual P loadings from USGS monitored stations on Pheasant Branch (1975–2006) and Yahara River (1990–2006) as indices of P loadings to Lake Mendota. Since 1990, daily P loadings were determined by the USGS based on an extensive P-sampling program. Pheasant Branch

P loadings for 1975–1989 with less extensive runoff event sampling were computed in Lathrop (1998). All laboratory P analyses since 1980 were performed at SLOH; analyses were performed at another WDNR lab in earlier years. Given the high P concentrations in tributary runoff, I consider the earlier data reliable.

To complement the long-term lake water quality record, data are presented from sediment cores obtained during February 1989 at the deepest location of each of the Yahara lakes. The sediment cores were taken using a "cold finger" corer, where sediments quickly freeze to the outside of the corer tube filled with dry ice; the hollow "tubes" of frozen sediment were later sliced into 2-cm intervals. Samples were oven dried at 105 °C and then later analyzed at the UW-Extension Soil and Plant Analysis Laboratory for a wide range of basic elements.

Yahara lakes in the 1800s

First-time visitor accounts of the Yahara lakes during the early 1800s portrayed lakes Mendota and Monona as having remarkable water clarity (see quotes in Mollenhoff 2005, Carpenter *et al.* 2006). These romanticized descriptions written in diaries and travel logs, sometimes reported in eastern newspapers (David Mollenhoff, local historian and author, pers. comm.), give the impression that the lakes were oligotrophic prior to the arrival of European settlers in the mid-1800s. However, the recurrent fall prairie burnings practiced for many centuries by Native Americans living in the region (Mollenhoff 2003) may have enhanced the transport of inorganic P to the lakes via runoff in the late winter period. Thus, the fertility levels of the Yahara lakes were likely higher than if the burnings had not been done, allowing the watershed vegetation to succeed to forest.

Whatever the trophic-state descriptions of the lakes were immediately prior to European settlement, lake conditions soon changed after Wisconsin became a state in 1848, with Madison the capital. The following year, Lake Mendota's water level was raised approximately 1.5 m due to the damming of its outlet (Lathrop *et al.* 1992). With the rise in water level, erosion from the newly inundated shoreline undoubtedly would have been a major source of sediment and nutrients to the lake. Early survey records and maps indicate that the Yahara River inflow region to Mendota was greatly altered. Prior to the water level rise, the Yahara River meandered as a defined river channel through a large wetland system, and nearby Sixmile Creek flowed into the Yahara River prior to entering Lake Mendota (from 1835 township survey maps and from map in Cheney and True [1893]). The Yahara River wetland is now an open water estuary, and Sixmile Creek enters Mendota directly (Fig. 1).

The mid-1800s was also a time when agriculture expanded rapidly in the region. By 1870, the land area used for crops was under full production, although in early decades small grains (wheat and oats) and hay were the dominant crops, which had less potential for soil erosion than corn, a crop that became increasingly important throughout the 1900s (Lathrop 1992b). The combined effect of a developing farm community and the lake-level rise caused Lake Mendota to have blue-green algal blooms that were first noted in 1882 (Trelease 1889) and later reported in the 1890s (Birge 1898). However, concerns about that lake's water quality apparently didn't become an important issue until after World War II (discussed below in Eutrophication of Lake Mendota – 1940s through the 1970s).

The most pronounced deterioration in water quality in the Yahara lakes first occurred in Lake Monona, which was receiving most of the city's untreated sewage by 1890 (Mollenhoff 2003, 2005). Civic leaders knew as early as 1880 that discharging sewage to the lake was wrong (Mollenhoff 2005), but it would be almost 80 years before the lower Yahara lakes were no longer a receptacle for Madison's wastewater.

Lower Yahara lakes and their management during the 1920s to 1960s

Accounts of early attempts to treat Madison's sewage that was being discharged into Lake Monona along with other waste dumpings (*e.g.*, sugar beet factory waste) indicate the severity of the problem (Flannery 1949, Sonzogni 1974, Lathrop *et al.* 1992, Mollenhoff 2003). The nascent wastewater treatment plants built during the early 1900s soon became overloaded because of Madison's burgeoning population. And, even under the best operating conditions, the treatment plants were not designed to remove nutrients from the effluent.[2] Thus, the lakes became overloaded with P and N, fueling algal blooms.

The nutrient status of the four Yahara lakes beginning in the mid-1920s is well documented in the lake monitoring data obtained by MPHD. Lake Monona DRP concentrations during the summer months (July–August) of 1925–1935, prior to the

[2] In 1914 the Burke plant on the east side of Madison was built and consisted of primary settling tanks and trickling filters (Sonzogni 1974) with its effluent discharging to Lake Monona. In 1928, the first portion of the Nine Springs treatment plant was built and consisted of Imhoff tanks followed by trickling filters and final clarifiers (Sonzogni 1974). At this time the Nine Springs plant received about half of Madison's sewage wastewater, with the plant's effluent discharging into Nine Springs Creek immediately upstream of Lake Waubesa (Flannery 1949). In 1936 all wastewater from Madison and its adjacent communities was sent to an expanded Nine Springs plant (now part of the Madison Metropolitan Sewerage District) when an activated sludge system was added (Sonzogni 1974).

wastewater diversion to Waubesa, averaged about 0.1 mg/L (Fig. 2), indicating the algae were not P limited. Summer DRP concentrations were lower in downstream lakes Waubesa and Kegonsa during those years, whereas DRP was very low in Lake Mendota, upstream from the effluent discharge. DRP concentrations during fall turnover also showed these same fertility difference between lakes (Fig. 3).

Conditions changed rapidly in 1936, however, when all of Madison's wastewater entered Lake Waubesa. Soon thereafter, summer DRP concentrations declined in Monona while increasing dramatically in shallower Waubesa and Kegonsa. Concentrations continued to climb to very high levels in those two lakes, lockstep with Madison's growing population, until MMSD's wastewater effluent was diverted downstream of the Yahara lakes in 1958 (Fig. 2 and 3). Levels of DRP dropped soon after the diversion due to the relatively rapid flushing rates for Waubesa and Kegonsa. The high DRP concentrations in Waubesa and Kegonsa caused P to behave like a conservative substance (*e.g.*, chloride) with little stimulatory effect on algal growth.

Inorganic N concentrations during July–August and after fall turnover generally were much higher in the lower Yahara lakes than in Mendota during the years when the lower lakes were receiving treated wastewater (Fig. 4). Thus, algae in the lower Yahara lakes were likely not N-limited during the years when sewage effluents were entering the lakes. While summer inorganic N concentrations <0.05 mg/L were sometimes reported for Mendota in the MPHD dataset, concentrations in other years were higher, but this may have been a detection limit problem. Inorganic N concentrations in the lower Yahara lakes during the fall turnover period were generally high and variable throughout the entire long-term record and not particularly informative as an indicator of N trends.

Algal bloom control by copper sulfate

Because of the massive blue-green algal blooms that developed soon after the poorly treated wastewater began entering Lake Monona, the MPHD began experimenting with copper sulfate as an algaecide in 1915 (Mollenhoff 2003, 2005). However, the systematic use of copper sulfate to control algal blooms did not start until 1925 when the MPHD began a program to spray the lake's near shore waters throughout the summer months (Fig. 5 and 6). The chemical control program was expanded to Waubesa and Kegonsa in 1936 as water quality problems became severe in those lakes coincident with all of Madison's treated wastewater being discharged to Waubesa. The amount of copper sulfate applied to Monona dropped after the mid-1940s, probably as a result of improvements in that lake's water quality. Due to perceived environmental concerns, the whole-lake spraying program ended after 1954, even though Madison's wastewater effluent was not diverted from the lower Yahara

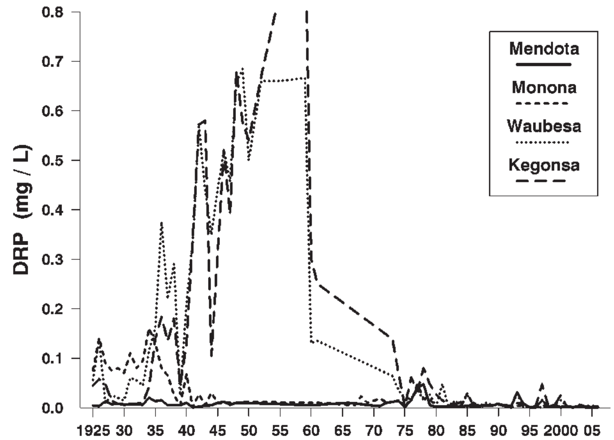

Figure 2.-Median dissolved reactive (inorganic) phosphorus (DRP) concentrations in the surface waters of the Yahara lakes during July-August, 1925-2006.

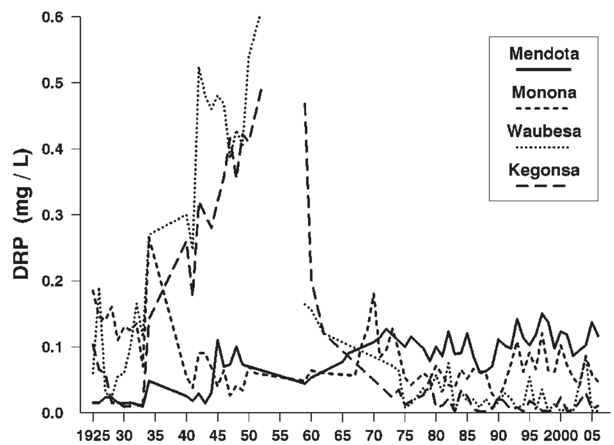

Figure 3.-Median dissolved reactive (inorganic) phosphorus (DRP) concentrations in the surface waters of the Yahara lakes during fall turnover, 1925-2006. (Fall turnover was defined as 1 Nov – 14 Dec for Mendota and 16 Oct – 14 Dec for the lower Yahara lakes.)

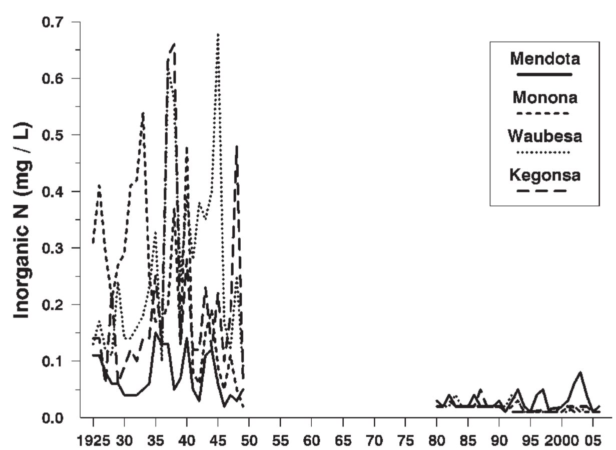

Figure 4.-Median inorganic N concentrations in the surface waters of the Yahara lakes during July-August, 1925-1949 and 1980-2006.

Figure 5.-Barge used in early years of copper sulfate treatments on Lake Monona. A motor boat towed the barge along the lake shoreline during spraying. (Source: Bernard Saley, originally published in Lathrop *et al.* 1992)

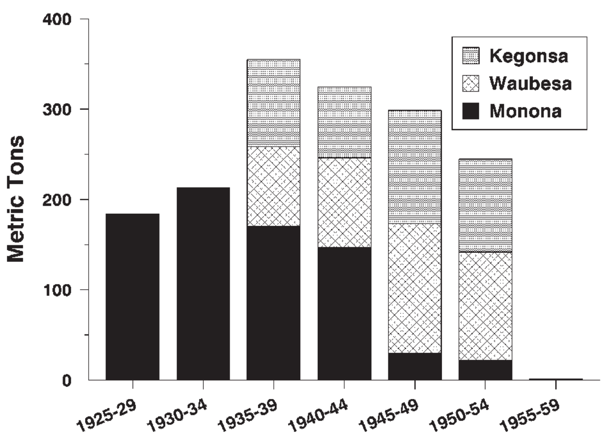

Figure 6.-Metric tons of copper sulfate applied by the Madison Public Health Dept. to lakes Monona, Waubesa and Kegonsa to control summer blue-green algal blooms, 1925-1954.

lakes until 1958. Profiles of sediment copper concentrations confirm the use of large amounts of copper sulfate in the lower Yahara lakes (Fig. 7).

The copper sulfate spraying program to control the dense blue-green algal blooms in the lower Yahara lakes was initially seen as an innovative and beneficial technique, given that no immediate solution was available for preventing the treated wastewater with all its nutrients from entering the lake. One early report indicated that Lake Monona (the only lake being chemically treated at the time) had the best water clarity of all the Yahara lakes, thus allowing greater light penetration for submersed aquatic macrophytes to expand their maximum depth of growth from about 3 m to >5 m (Domogolla 1935). However, the chemical treatments became costly and controversial in later years. After a two-year study in the early 1940s conclusively documented that the overwhelming major source of nutrients causing all the water quality problems in the lower Yahara lakes was from Madison's treated wastewater (Sawyer *et al.* 1945), one might wonder if the "chemical control solution" didn't prolong the political and legal battles to divert the wastewater out of the Yahara lakes, a remedial action that took 13 years from the final report publication date. Unfortunately, Waubesa and Kegonsa were not sampled during the years between the ending of the chemical treatments (1954) and the diversion of the wastewater effluent (1958) to determine what effect the treatments were having on algal densities and water clarity.

Carp and their management

An account of the eutrophication of the Yahara lakes is incomplete without a discussion of trends in carp abundance in response to the wastewater inputs, given the well known role carp have in causing turbid lake environments. Carp were stocked in the Yahara lakes in the late 1800s and soon became abundant (Lathrop *et al.* 1992). While commercial fishing for carp commenced sometime in the early 1900s, carp became so dense in the lower Yahara lakes that the Wisconsin Conservation Department (WCD) in 1934 began a major carp removal program on all four lakes, a program that lasted through 1969 (Lathrop *et al.* 1992). The impact of treated wastewater on carp productivity was immediately apparent when a massive hatch of carp occurred in Lake Waubesa in 1936 (Frey 1940), the year all of Madison's wastewater began entering the lake. In general, annual amounts of carp removed by the WCD were highest when the lakes received wastewater effluents (Fig. 8). Carp removals declined (especially in Monona) after the wastewater no longer entered the lakes, although removal tonnages were also affected by variable fishing effort, especially after 1967 when the WCD merged with a state environmental agency to become the WDNR. This major carp removal program, conducted by state fisheries managers from 1934–1969, was akin to the carp farming that had been practiced in central Europe for centuries (Neess 1949).

Macrophytes and their management

Aquatic macrophytes in the Yahara lakes were historically abundant and diverse except for Kegonsa (Lathrop *et al.* 1992, Nichols and Lathrop 1994). Initially the nutrients from the treated wastewater may have enhanced macrophyte growth, although dense blue-green algal blooms and overabundant carp populations eventually restricted plant densities in the lower Yahara lakes. For example, aquatic macrophytes densely covered large areas of the Lake Waubesa prior to 1936, the year when MMSD's full wastewater effluent first entered the lake and the massive carp hatch occurred. Soon after 1936, macrophytes were sparse (Frey 1940). By the mid-

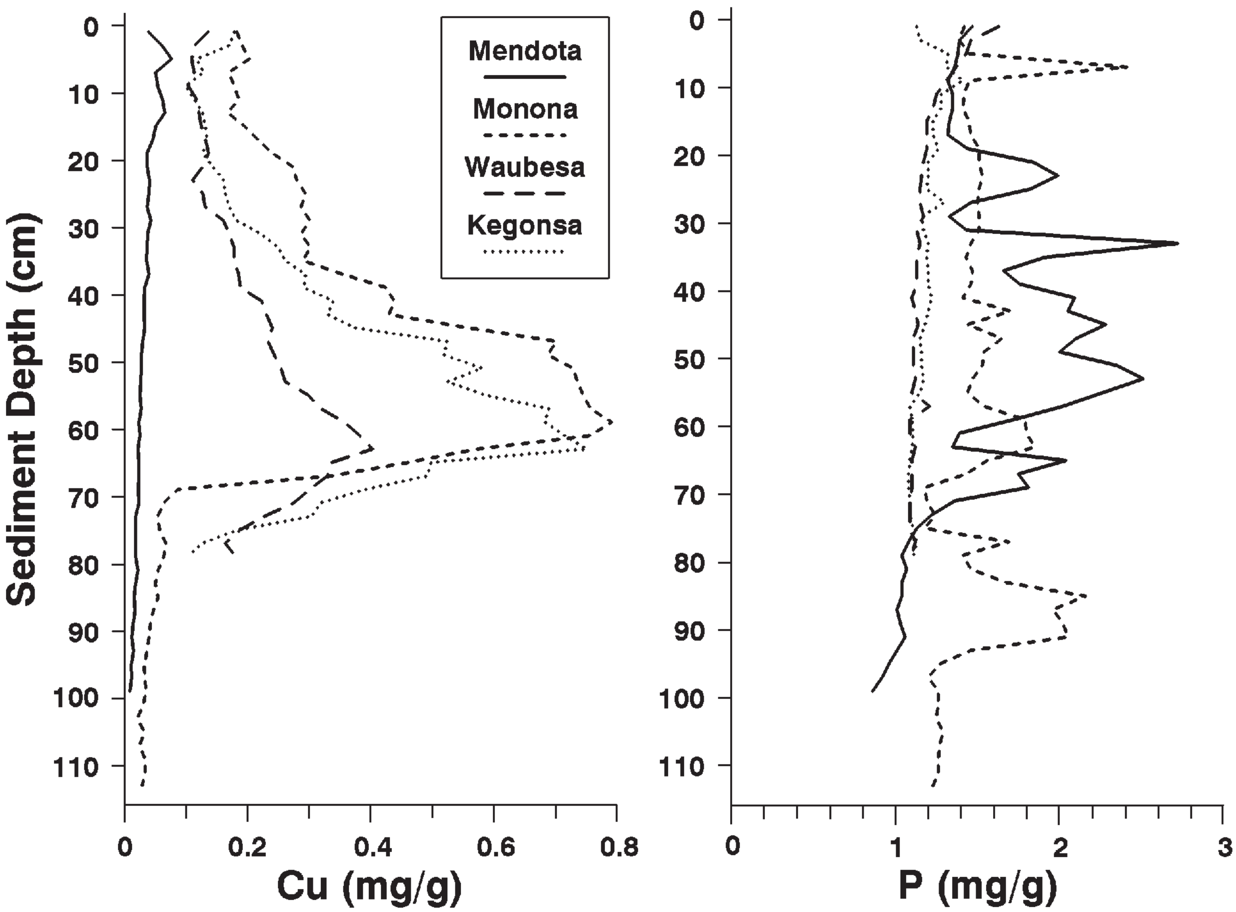

Figure 7.-Total copper and phosphorus concentrations in sediment cores taken from the deepest regions of the four Yahara lakes during February 1989.

1920s, Lake Monona was still infested with rooted "weeds," causing the MPHD to remove the plants using a weed cutting machine, steel cables, and chemicals in shallow waters; macrophytes in deeper waters were not treated because of their recognized importance to fish populations (Domogalla 1935). While the mechanical removal methods were employed in small areas, the MPHD began using arsenic compounds in 1926 (Domogalla 1926) as the primary control method of overabundant macrophytes. Amounts of arsenic used in the Yahara lakes have never been fully documented, but use was apparently extensive through the 1940s (especially in Monona) and then declined until its use stopped after 1964 (Lathrop and Johnson 1979).

Mechanical weed cutters were used more extensively as a macrophyte control method in the 1950s and 1960s. After cutting, the floating plants were removed by workers on barges. This control practice likely promoted the rapid spread of Eurasian water milfoil (*Myriophyllum spicatum*; Lathrop *et al.* 1992), which had invaded the Yahara lakes during the

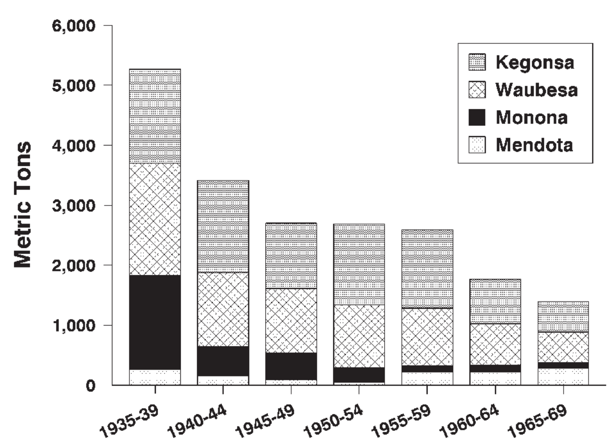

Figure 8.-Metric tons of rough fish (mostly carp) removed by the Wisconsin Conservation Dept. (WCD) from the four Yahara lakes, 1935-1969. Note: Carp removal began in 1934 in Monona. Carp removals declined after 1967 when the WCD became the Wisconsin Dept. of Natural Resources (WDNR) and ended after 1969. (Source: Lathrop *et al.* 1992)

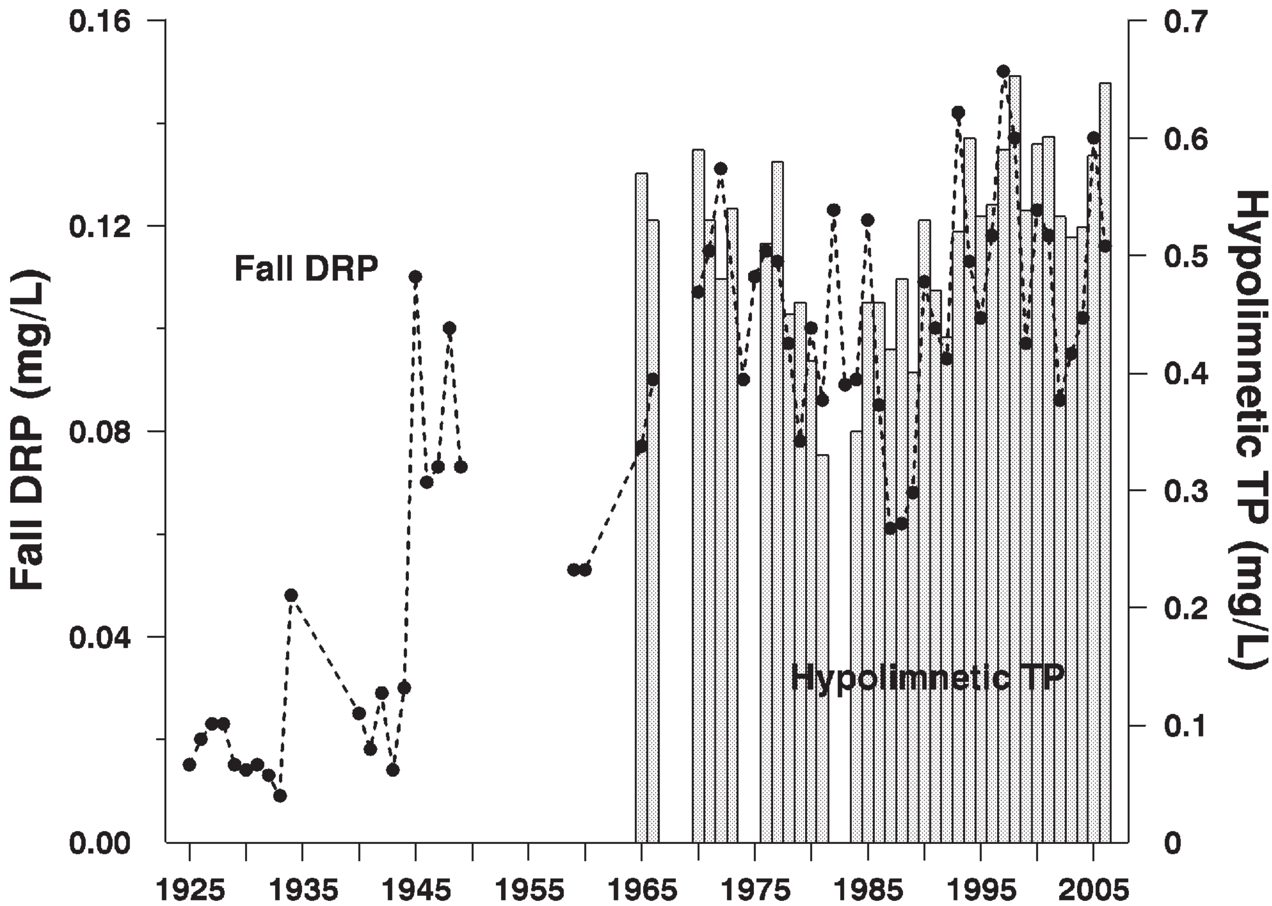

Figure 9.-Median surface water DRP concentrations (solid circles) during fall turnover (1 Nov-14 Dec) and hypolimnetic total P concentrations (bars) at 20 m depth on 1 September (interpolated from sampling dates) for selected years during 1925-2006 in Lake Mendota.

early 1960s (Nichols and Lathrop 1994), because the plant grows rapidly from vegetative fragments. By the late 1960s, mechanical weed harvesters replaced the weed cutters, but milfoil was already flourishing throughout the Yahara lakes. Because Eurasian milfoil also grows with dense foliage near the surface of turbid waters while shading out lower-growing native plants, the exotic plant continues to be a huge water management problem in the eutrophic Yahara lakes.

Eutrophication of Lake Mendota – the 1940s through the 1970s

Summer blue-green algal blooms in Lake Mendota were reported by the late 1800s after the lake's level was raised and its watershed converted to agricultural. In the 1920s, two upstream communities built wastewater treatment facilities that discharged to two streams flowing to Mendota. However, the impact of the wastewater effluents on the lake was probably not important prior to the 1940s because DRP concentrations in the tributary streams during the early 1940s were only slightly higher than the background concentrations of a nearby unpolluted stream (Lathrop 1992b). In addition, MPHD was not using large amounts of copper sulfate in Mendota, as evidenced by low copper concentrations in the bottom sediments (Fig. 7). With attention focused on the water quality problems in the lower Yahara lakes, the prevailing view was that Mendota was the "clean lake" in the Yahara chain.

Water quality in Mendota changed in the mid-1940s after World War II. Concentrations of DRP increased substantially in the two streams receiving the wastewater effluents from upstream communities (Lathrop 1992b). The 1940s was also the period when farmers began using increasing amounts of commercial N and P fertilizers (Lathrop 1992b). Furthermore, corn production increased, leading to increased soil erosion due to prolonged exposure of essentially bare soil. Finally, Madison's population was also expanding, with much of the urban development occurring in the Mendota watershed.

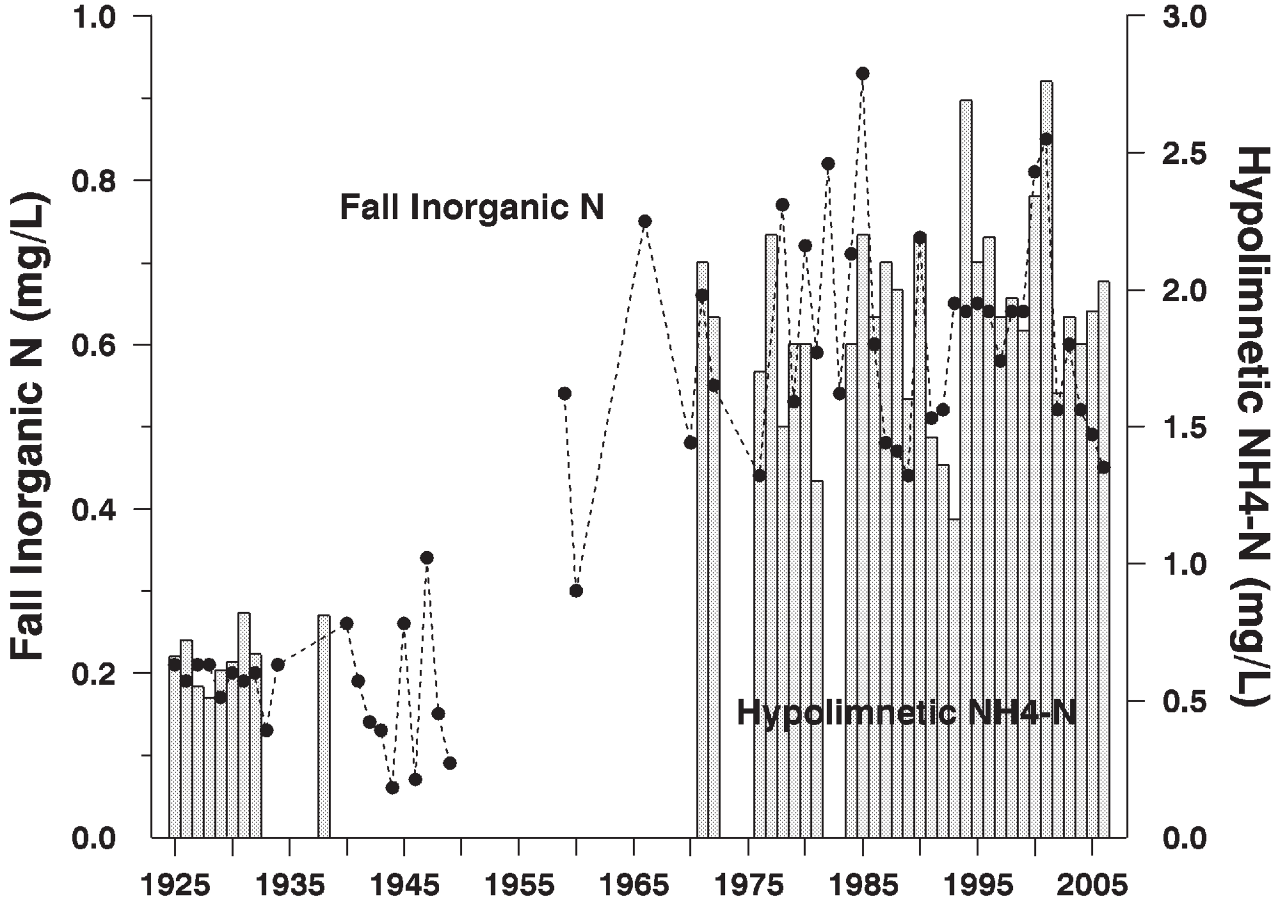

Figure 10.-Median inorganic N surface water concentrations (solid circles) during fall turnover (1 Nov-14 Dec) and hypolimnetic NH_4-N concentrations (bars) at 20 m depth on 1 September (interpolated from sampling dates) for selected years during 1925-2006 in Lake Mendota.

Increased inputs of nutrients from the sewage effluents and agricultural and urban runoff initiated concerns about the eutrophication of Lake Mendota (Hasler 1947, Bartsch and Lawton 1949, Lathrop 1992b, Carpenter and Lathrop 1999, Carpenter *et al.* 2006).

The increase in nutrient inputs and resultant algal bloom problems in Lake Mendota after World War II is corroborated by an increase in both P and N concentrations in the lake. While summertime DRP concentrations in the surface waters remained very low (Fig. 2), fall DRP concentrations increased dramatically in the mid-1940s (Fig. 9). Similar DRP increases were also observed during the winter and spring months (Lathrop 1992b). An increase in inorganic N in the surface waters during fall turnover also occurred sometime between the late 1940s and mid-1960s (Fig. 10). NH_4-N concentrations in the hypolimnion were also higher after 1970 than during the 1920s and 1930s (Fig. 10). The differences in N between the two eras probably reflect the agricultural use of N fertilizers that began in the 1940s, with the annual amounts increasing substantially after the mid-1960s and peaking in the 1980s (Lathrop 1992b).

In an earlier study I concluded that a DRP increase in the hypolimnion of Lake Mendota in the mid-1940s indicated a change might have occurred in the amount of P available for internal loading, which was consistent with the fall DRP jump in the surface waters during the same time period (Lathrop 1992b). The relationship between hypolimnetic and fall DRP concentrations followed the work of Sonzogni (1974), who showed the large DRP mass build-up in Mendota's hypolimnion during stratification was transferred and diluted throughout the whole lake at fall turnover without a decline in DRP mass. But if dissolved ferrous iron [Fe(II)] also builds up in the hypolimnion, then insoluble ferric iron [Fe(III)] oxide compounds are formed as the deeper waters become oxygenated during destratification. The end result is that these Fe compounds will scavenge P as they settle from the water column, causing significantly less DRP in the lake surface waters after fall turnover (Holdren and Armstrong

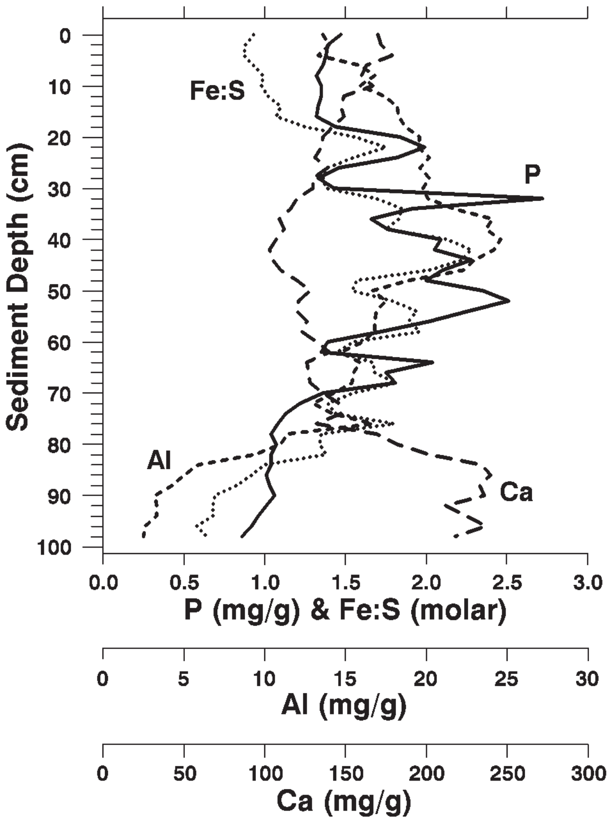

Figure 11.-Profiles of aluminum (Al), calcium (Ca), and phosphorus (P) concentrations, and molar ratios of Fe:S in a sediment core collected at the deepest region of Lake Mendota during February 1989. (Core sectioned into 2-cm increments; analyses performed at the UW-Extension Soil and Plant Analysis Lab.)

1986). Data from Lake Mendota during the 1970s and 1980s indicated little build-up of dissolved Fe(II) in the anoxic hypolimnion or interstitial sediment waters, possibly because the formation of insoluble FeS compounds limited the amount of free Fe(II) (Holdren and Armstrong 1980, 1986, Stauffer 1987a). However, this may not have been the case prior to the mid-1940s.

I now hypothesize that the lower hypolimnetic DRP concentrations reported earlier for the 1920s and 1930s (Lathrop 1992b) may have resulted from aeration of water samples from the anoxic hypolimnion during handling, allowing Fe(III) oxide precipitates to remove P from solution before the sample was filtered and analyzed the next day, the likely procedure of the MPHD given that most if not all of the lakes were often sampled the same day. While I have seen this major loss of P happen in similarly handled hypolimnetic samples from another lake with high Fe, it has not been a significant problem with Mendota samples analyzed at the SLOH since 1980 where unpreserved DRP samples were chilled but generally not filtered and analyzed until the next day. Hypolimnetic DRP concentrations during 1980–2006 represented about 90% or more of total P concentrations. For the UW's early 1970s work, where hypolimnetic DRP samples were filtered immediately upon returning to the shoreline lab and analyzed within a few hours (Sonzogni 1974), DRP averaged about 95% of total P.

Closer inspection of the MPHD's hypolimnetic TP data obtained in some years during 1937–1947, however, indicates that TP concentrations were much higher than DRP in several samples (sampling was too inconsistent to be useful for extending a hypolimnetic TP index to earlier years). Thus, some hypolimnetic Fe(II) may have been available for scavenging P as the lake destratified until conditions changed in the mid-1940s. In recent decades, sulfate (SO_4) concentrations have increased significantly compared to historical data (Lathrop 1992a), which could cause an increase in FeS being formed, reducing the amount of free Fe available for scavenging P from lake waters in the fall or other times of the year.

Another indication of these changes is apparent in Mendota's sediment chemistry record based on the 1989 sediment core (Fig. 11; sediment aging done by Hurley *et al.* [1992] on a core collected in 1987 determined 20 cm of sediment depth to represent about the mid-1940s and 80 cm to represent the late 1700s, assuming a constant mass sedimentation rate). In the 1989 core, sediments deeper than 80 cm had high calcium concentrations (about 60% $CaCO_3$ by weight). The large increase in aluminum concentrations in sediments above the marl layer is indicative of erosional sediment input, which had declined substantially in recently deposited sediments. Phosphorus concentrations also were highest during this erosional period, with variability that closely matched the iron to sulfur molar ratio (Fe:S) profile. Most of the Fe:S ratio variability was due to Fe (data not shown), with sulfur increasing generally in more recent years. Molar ratios near 1.0 in the more surficial sediment layers may indicate that most of the Fe was tied up as FeS; higher ratios deeper in the sediment profile may indicate part of the Fe was available for P binding. Thus, the higher P concentrations in Mendota's deeper sediment layers may indicate more refractory P associated with erosional materials. Research is currently being conducted on Mendota's deep-water sediments to determine the historical changes in the lake's P recycling potential and the chemical forms of Fe, S, and P in the sediments that may affect those potentials. Interestingly, P concentrations in the lower Yahara lakes (especially Waubesa and Kegonsa) showed almost no P increase during the years when lake water P concentrations were exceedingly high from the sewage inputs (Fig. 7), which indicated little P binding potential in those sediments as Fe:S molar ratios were generally <1.0 (data not shown).

As treated wastewater inputs and agricultural and urban nonpoint pollution continued to increase, water quality problems in Lake Mendota became severe in the 1960s. Additional wastewater inputs from another upstream community began entering the lake, which coincided with increasing amounts of N and P fertilizers used by farmers in the watershed (Lathrop 1992b). Coupled with these increases in nutrient loadings, Eurasian milfoil was proliferating throughout all the Yahara lakes, including Mendota. The public outcry for relief from the eutrophication problems in Lake Mendota during the mid-1960s resulted in the formation of the Lake Mendota Problems Committee comprised of university and government scientists and managers. Perhaps its greatest accomplishment was realized in 1971 when the wastewater effluents of the upstream communities were finally connected to the MMSD treatment plant system, ending that source of nutrients to the lake (Sonzogni and Lee 1974). However, as evidenced by subsequent P levels (Fig. 9), water quality in Lake Mendota did not improve with the diversion of the wastewater effluents, and the onus of the problem shifted to agricultural and urban nonpoint source pollution, the major source of water pollution in the U.S. today (Carpenter *et al.* 1998).

Eutrophication and its management in the Yahara lakes since the 1970s

Programs to reduce the nonpoint sources of pollution were conducted during the 1970s and 1980s, with cost-sharing monies available to install best management practices (BMPs). But the programs were voluntary, many of the BMPs were untested, and participants were too few to result in significant reductions in P loadings to Lake Mendota (Lathrop *et al.* 1998, Carpenter and Lathrop 1999, Carpenter *et al.* 2006). Because controlling nonpoint P inputs seemed like an intractable problem, a biomanipulation project using piscivore stocking to induce a trophic cascade to ultimately reduce algal densities and increase water clarity in Lake Mendota was initiated in 1987 (Kitchell 1992, Lathrop *et al.* 2002). A historical analysis of Secchi disk records for Lake Mendota indicated that when planktivorous fish were abundant in the lake, water clarity was significantly less than when planktivorous fish were not abundant (Lathrop *et al.* 1996). While it has been hard to maintain large densities of piscivorous fish in a popular urban fishing lake like Mendota (Kitchell and Carpenter 1993, Johnson and Carpenter 1994), the biomanipulation project has helped sustain improved water clarity to this day due to high densities of large-bodied *Daphnia pulicaria* during the spring and early summer months (Lathrop *et al.* 2002, Carpenter *et al.* 2006).

After the biomanipulation project commenced on Lake Mendota, efforts to control nonpoint source pollution were accelerated. In 1994, the Lake Mendota Priority Watershed Project began with a three-year inventory phase that estimated P loadings were about 48% from cropland, 21% from barnyards, and 19% from urban construction site erosion (Betz *et al.* 2005). Stream bank erosion and pollution from established urban areas made up the remaining P inputs to the lake. The project plan called for a P-loading reduction goal of 50%. After the project plan was approved and state cost-sharing monies made available, the 11-year implementation phase of the project began with the implementation of many BMPs along with the institution of other corrective actions (*e.g.*, creation and enforcement of a county-wide construction site erosion control ordinance). While the project will not terminate until the end of 2008, progress has been made toward reducing P inputs to the lake (Betz *et al.* 2005, Ventelä and Lathrop 2005), but the 50% P reduction goal will likely not be met.

One challenge is the imbalance in the P budget for the agricultural watershed, where much more P has been imported as fertilizers and feed supplements than exported as crops and animal products (Bennett *et al.* 1999). This imbalance led to an increase in soil P levels recorded from the 1970s to the 1990s, with high soil P concentrations becoming even more of a problem today. Another problem is an increasing number of animal units on farms in the watershed while the land available to spread the manure is decreasing through urbanization. In some farm fields nearest the source of the manure, soil P concentrations are exceedingly high (Cabot *et al.* 2004). While it appears that the reductions in soil erosion have been achieved as part of the priority watershed project (Betz *et al.* 2005), the manure management problem continues to drive the external P loading dynamics for Lake Mendota.

The importance of manure management is illustrated by an analysis of daily P loading data determined for the USGS Yahara River monitoring station (upstream from Lake Mendota). From 1990 to 2006, 48% of the total P loading for the entire 17-year period occurred between January and March (mostly as runoff and not baseflow) when the ground was usually frozen, with four years having very large P loadings in late winter (Fig. 12). During April–June, when intense thunderstorms were common but before substantial crop growth, 28% of the P loading occurred. Suspended sediment loadings for 1990–2006 indicated the seasonal differences in dissolved P versus particulate P loadings. For the same amount of daily P loading in runoff, daily sediment loadings during January–March were about 25% of daily sediment loadings in runoff during the rest of the year (Fig. 13). While only a small portion of the collected water samples were processed for dissolved P analysis, DRP concentrations obtained for runoff events during January–March were generally $\geq 50\%$ of total P, whereas DRP represented a significantly lower proportion of total P in runoff samples for other months

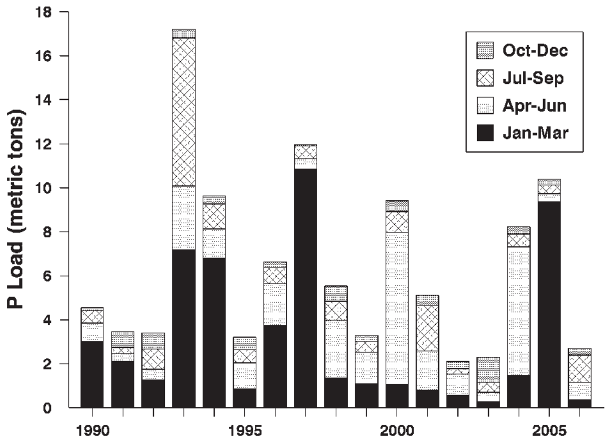

Figure 12.-Seasonal phosphorus loadings for the Yahara River tributary inflow to Lake Mendota, 1990-2006. (Seasonal data are summation of monitored daily P loads determined by the U.S. Geological Survey at the Yahara River at Windsor gaging station.)

Figure 14.-Late winter snowmelt runoff from an agricultural field with manure applied on frozen ground in the Lake Mendota watershed. (Photo: Herb Garn, U.S. Geological Survey, Middleton, Wisconsin)

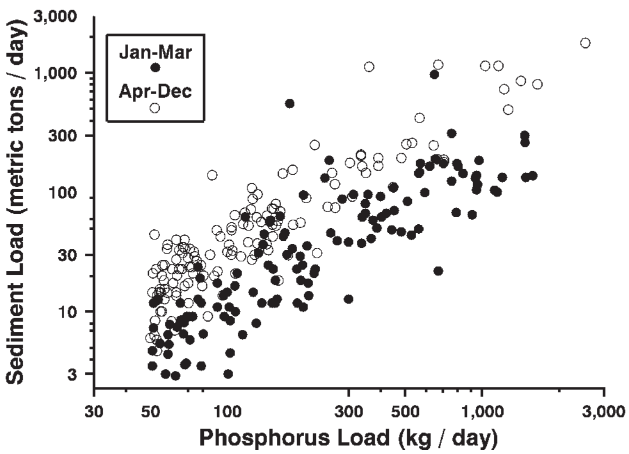

Figure 13.-Daily total P loads versus suspended sediment loads for January-March (solid circles) and April-December (open circles) periods during 1990-2006 at the USGS Yahara River at Windsor gaging station.

(USGS published monitoring data). Consequently, not only is manure causing soil P levels to be high, thereby increasing sediment-bound P loads during the spring and summer months, much of the applied manure P is entering the lake during the late winter period (Fig. 14) in a highly biologically available state with little potential for P attenuation via sediment settling in the lower stream reaches. Similar findings were reported for runoff events in this same subwatershed during 1976-1980 (Lathrop 1986).

Trophic conditions of the Yahara lakes since 1980

Probably the most interesting question today is: What are the recent Yahara lakes' trophic conditions and how have they responded to the changes in P loadings? Now that the wastewater inputs are not influencing the lower Yahara lakes, the major proportion of their P loadings is coming from the outlet of the upstream lake (Lathrop 1990, DCRPC 1992). With Mendota's P status dictated by urban and especially agricultural runoff (Lathrop *et al.* 1998), water quality improvements in that lake cascade down through the lower Yahara lakes.

The availability of high quality data (with low analytical detection limits) for the summer months since 1980, when P concentrations (both dissolved and total) in the surface waters have been the lowest, illustrates how the Yahara lakes have responded in both P concentration and water clarity, measured by Secchi disk readings. Data for June are not included in these analyses (except for occasional sampling done at the end of the month when the early July biweekly routine sampling was scheduled) because June is generally part of the spring clear-water phase when Secchi disk readings are large prior to the normal onset of blue-green algal blooms that historically have occurred from July into September in Lake Mendota (Lathrop and Carpenter 1992, Lathrop *et al.* 1996). The Pheasant Branch tributary P loading data for 1975–2006 are a useful indicator of external P inputs (Fig. 15), along with the shorter term record for the Yahara River inflow (Fig. 12).

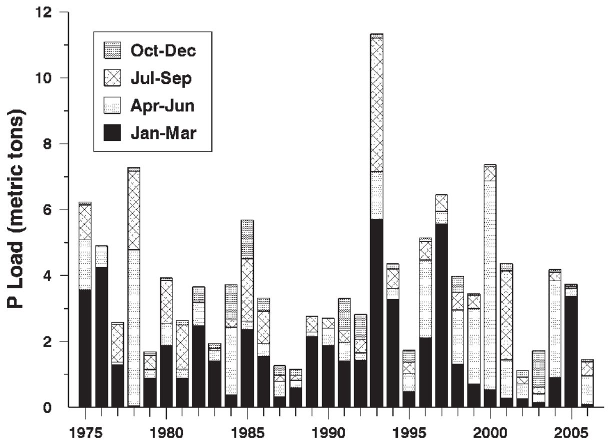

Figure 15.-Seasonal phosphorus loadings for the Pheasant Branch tributary inflow to Lake Mendota, 1975-2006. [Seasonal data are the summation of daily P loads determined for the U.S. Geological Survey's Pheasant Branch at Middleton gaging station. Records were computed by Lathrop (1998) for 1975-1989 and by the USGS for 1990-2006.]

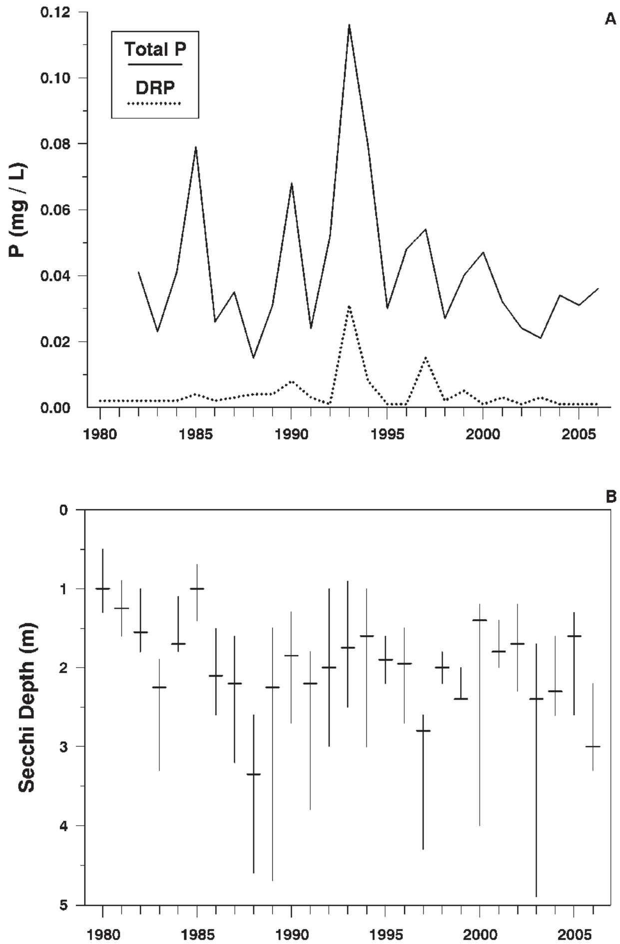

Figure 16.-(a) Median total P and dissolved reactive P (DRP) concentrations in Lake Mendota during July-August of 1982-2006 and 1980-2006, respectively. (b) Secchi disk transparencies in Lake Mendota during July-August, 1980-2006. Median reading for each summer is designated by a short horizontal line; range of readings is designated by thinner vertical line.

In Lake Mendota, surface water total P (TP) concentrations have been highly variable during July–August since 1980 (Fig. 16a). Because most P was in the particulate phase, TP concentrations are highly correlated to algal biomass. Concentrations were the lowest in 1988 as of result of a two-year drought with reduced P loadings (Fig 15); TP concentrations were highest in 1993 following very high spring and summer runoff events. Since 1993, summer TP concentrations in Mendota have been generally declining, although TP has rebounded somewhat from 2003 following another two-year period of low P inputs (Fig. 12 and 15). Interestingly, median DRP concentrations in the lake surface waters during July and August were above analytical detection in a number of years during the 27-year record (Fig. 16a). In particular, DRP was elevated in 1993, 1994, and 1997 when P loadings were high during the late winter runoff period as well as in a few other years (1985 and 1990). In all these years, TP was relatively high in the lake during the summer. Thus, these elevated median DRP concentrations quite likely indicate algal growth was not P-limited for much of those summers.

Summer water clarity during 1980–2006 has similarly varied (Fig. 16b). Many of the lowest Secchi disk readings occurred during the early 1980s when the smaller-bodied *Daphnia galeata mendotae* dominated as compared to years since 1988 when the larger-bodied *D. pulicaria* dominated (Lathrop *et al.* 2002; NTL-LTER project, unpubl. data). The reasons for the exceptional summer water clarity in 1988 have been attributed to lower P inputs and lake concentrations, as well as greater algal grazing from the large-bodied *Daphnia* and less internal nutrient recycling from more stable temperature stratification (Lathrop *et al.* 1999, 2000). While 1990 was a year of very dense *Aphanizomenon flos-aquae* blooms, blue-green algal blooms since then in general have not been as severe, at least in the occurrence of shoreline scums. The summers of 1993–1994 had very high P concentrations in the lake, yet water clarity was average (Fig. 16a and b). In 1997, water clarity was exceptionally high, even though late-winter P loadings were high, resulting in summer DRP concentrations being well above analytical detection, indicating algae were not P limited. The low P loadings of 2002–2003 also produced low P concentrations in the lake surface waters and periods of good summer water clarity in 2003. While median summer Secchi disk readings were relatively low during 2000 and 2005 following elevated spring and late-winter P loadings, respectively, the effect was not long-lasting. In addition to summer 2003, water clarity was also high during summer 2006.

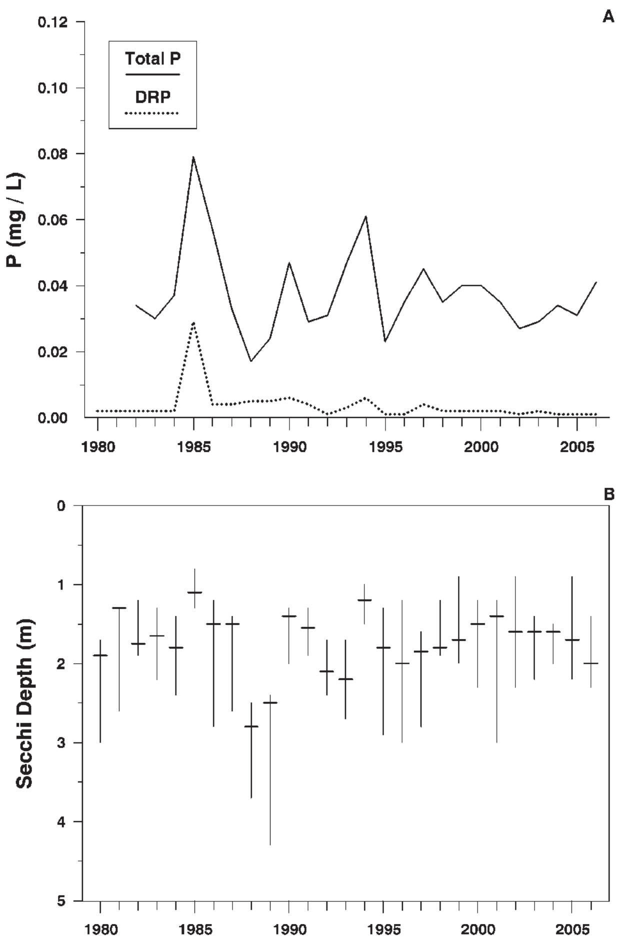

Figure 17.-(a) Median total P and DRP concentrations in Lake Monona during July-August of 1982-2006 and 1980-2006, respectively. (b) Secchi disk transparencies in Lake Monona during July-August, 1980-2006. Median reading for each summer is designated by a short horizontal line; range of readings is designated by thinner vertical line.

Thus, Lake Mendota might be near a threshold where it is not P-limited during the summer, as indicated by elevated DRP concentrations in the surface waters after periods of high P loadings, but the threshold also is reversible if P loadings decline. Internal P recycling from the lake's sediments would cause a lag in this response, but the apparently low P-binding potential of the sediments indicates the response could be rapid. This is corroborated by how soon DRP concentrations in the surface waters became low again after years of high external P loadings (Fig. 16a). Summer water clarity also improved following about two years of low runoff, further indicating the rapid response of the lake to changes in loading. While it has been well documented that internal P loading is much more important in supplying P for algal growth than external loading during the summer months in Lake Mendota (Stauffer and Lee 1973, Stauffer 1987b, Soranno *et al.* 1997), the lake's memory of external P loadings seems short-lived.

Algal densities and water clarity during the summer can also be controlled by zooplankton grazing as well as nutrient supplies (Lathrop *et al.* 1999). This may help explain the high water clarity during summer 1997 when DRP and inorganic N concentrations in the surface waters were both relatively high. Phytoplankton may have been dominated by more edible species that normally are not important when inorganic nutrient concentrations are low.

The P and Secchi disk record since 1980 for Lake Monona is less complicated. The highest surface water total P concentrations occurred during the summers of 1985 and 1994 when water clarity was worst, indicating the correlation between particulate P and blue-green algal biomass (Figs. 17a and b). The effect of the 1987–1988 drought was apparent in low TP concentrations and high water clarity during the summers of 1988–1989. Water clarity during the other years in the 27-year record has been relatively consistent with 1.5–2.0 m median Secchi readings. Only in summer 1985 was DRP substantially above analytical detection, although other summers in the late 1980s through 1997 indicated slightly elevated median DRP concentrations. The slightly lower Secchi disk readings in Monona than in Mendota probably reflect Monona's greater potential for internal recycling, given the lake's shallower mean depth.

Summer TP concentrations were generally much higher and Secchi disk readings much lower in Lake Waubesa than in Monona, reflecting Waubesa's much greater propensity for internal recycling due to its shallowness (Fig. 18a and b). Median Secchi disk readings ranged around 1.0 m for most summers during 1980–2006, with a few summers having somewhat higher Secchi readings. For the first two decades of the summer record, surface water DRP concentrations were well above analytical detection, indicating the lake was not P-limited. However, DRP has been lower in recent years indicating a potential shift to P limitation for algal growth.

The summer P and Secchi disk record for Lake Kegonsa (Fig. 19a and b) reflects that shallow, round-shaped lake's propensity for internal recycling. Except for 1987–1989, years associated with the drought and lower P loadings, TP concentrations were high and summer blue-green algal blooms dense, with Secchi disk readings averaging <1 m. Median summer DRP concentrations have been consistently above analytical detection for most summers, indicating the lake probably was not P-limited, although DRP has been low in some recent summers. In both Waubesa and Kegonsa, undetectable DRP concentrations when P limitation might be occurring have not resulted in improved water clarity.

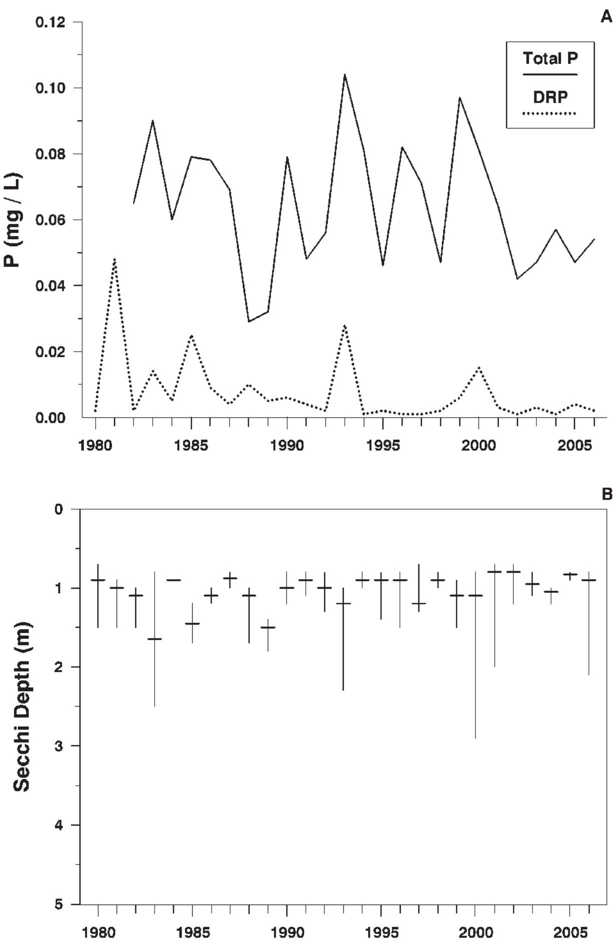

Figure 18.-(a) Median total P and DRP concentrations in Lake Waubesa during July-August of 1982-2006 and 1980-2006, respectively. (b) Secchi disk transparencies in Lake Waubesa during July-August, 1980-2006. Median reading for each summer is designated by a short horizontal line; range of readings is designated by thinner vertical line.

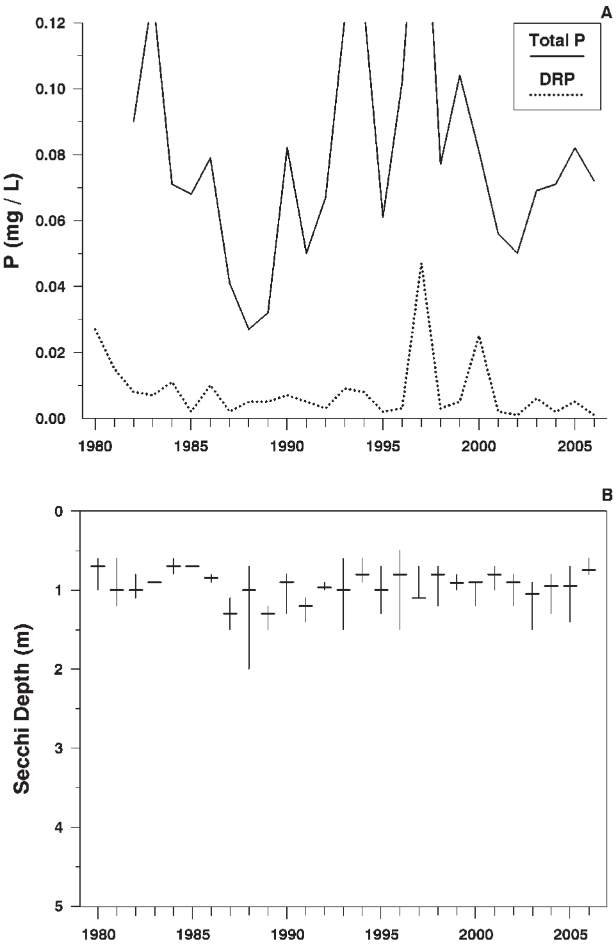

Figure 19.-(a) Median total P and DRP concentrations in Lake Kegonsa during July-August of 1982-2006 and 1980-2006, respectively. (b) Secchi disk transparencies in Lake Kegonsa during July-August, 1980-2006. Median reading for each summer is designated by a short horizontal line; range of readings is designated by thinner vertical line.

Nitrogen and its potential as a growth-limiting nutrient to algae

While much emphasis has been placed on P as the most important nutrient controlling eutrophication in lakes worldwide (Vollenweider 1976, Schindler 1977, 2006, Smith 1983, Cooke *et al.* 2005), N can also play a role in many systems (Vollenweider 1968, Goldman and Horne 1983, Guildford and Hecky 2000, Wetzel 2001, Howarth and Marino 2006). The shift to N becoming the more important growth-limiting nutrient can occur in lakes where excessive P loading has caused them to become more productive (Wetzel 2001), especially as enhanced denitrification reduces N availability in fertile aquatic systems (Howarth and Marino 2006). In such circumstances, nitrogen (N_2)-fixation by certain species of blue-green algae can compensate for low inorganic N supplies, with P limitation resulting (Schindler 1977, Smith 1983). However, the nitrogenase enzymes needed for N_2-fixation require Fe, which if in short supply can prevent this compensation mechanism for low inorganic N availability (Goldman and Horne 1983, Howarth *et al.* 1988). In Lake Mendota, N_2-fixation has not been shown to be significant (Brezonik 1968, Torrey 1972), quite possibly due to the lack of available Fe, as discussed earlier.

Thus, N has the potential to be a limiting nutrient for algal growth during summer in the Yahara lakes given three factors: (1) little Fe is available for N_2-fixation, (2) denitrification rates are potentially strong throughout the entire eutrophic lake-river-wetland system, and (3) P loading rates are high. Indications of N limitation have occurred in the surface waters almost every summer in the lower Yahara lakes and many (but not all) summers in Lake Mendota since 1980. Although laboratory detection limits for both NH_4-N and NO_3-N/NO_2-N dropped in 1991, median inorganic N concentrations during July–August have almost always been below detection for both N forms in the lower Yahara lakes since 1980 (Fig. 4). In Mendota, inorganic N was more likely to be above detection in July before dropping below detection in August (data not shown), with some summers exhibiting relatively minor median inorganic N concentration spikes comprised almost exclusively of NH_4-N. The even greater lack of inorganic N during July–August in the lower Yahara lakes is not surprising given their major water source was from the upstream lake where little inorganic N was being exported in the summer months. In contrast, Mendota receives groundwater and stream baseflow that are high in NO_3-N (Lathrop 1979, USGS tributary monitoring data), but denitrification in the Yahara river estuary and other wetland-associated inflows may remove much of that source of inorganic N before it reaches the lake.

Similar to the high DRP concentrations in hypolimnetic waters that provide the source of internal P loading, NH_4-N concentrations were also high in hypolimnetic waters of Lake Mendota (Fig. 10) and as such are a source for internal N loading. However, N:P molar ratios of NH_4-N and DRP in these hypolimnetic waters during late summer (based on concentration data presented in Figs. 9-10) were much less than the Redfield N:P ratio of 16:1 often used to indicate the threshold between N and P limitation for algae (Wetzel 2001, Kalff 2002). Similar concentrations and ratios have also occurred in Lake Monona. Thus, at times of internal mixing events during the summer after earlier available supplies of inorganic N have been depleted, N limitation may be promoted over P limitation in all the Yahara lakes.

These indications of N limitation since 1980 in the Yahara lakes were generally not apparent in the detailed UW sampling of Lake Mendota during the summers of 1970–1972. For the majority of sampling dates during July and August, both NH_4-N and NO_3-N concentrations were well above the reported detection limits that ranged between 0.01 and 0.05 mg/L for both constituents (Torrey 1972, Sonzogni 1974); however, both NH_4-N and NO_3-N concentrations were low (≤ 0.01 mg/L) in Mendota's surface waters during the summer of 1966 (Brezonik 1968). Unfortunately, high-quality inorganic N data with low detection limits were not obtained on the lower Yahara lakes prior to 1980 to help corroborate if those lakes were potentially N-limited.

A major shift from P-limitation to N-limitation in the Yahara lakes would logically require a decrease in inputs (external or internal) of N and/or an increase in inputs of P. A decrease in N inputs to Mendota was not likely given the upward trend of NO_3-N concentrations in stream baseflow and groundwater in agricultural watersheds in recent decades. Also in Mendota, hypolimnetic NH_4-N and fall inorganic N concentrations did increase between the mid-1940s and mid-1960s (Fig. 10), but concentrations have remained relatively constant since then. Other than the large increase in DRP concentrations after the mid-1940s, hypolimnetic TP and fall surface water DRP concentrations have increased only slightly since the mid-1960s, with a period of somewhat lower concentrations during the 1980s (Fig. 9). As previously stated, the lower Yahara lakes (especially Waubesa and Kegonsa) have had detectable DRP concentrations during many summers, whereas inorganic N has been generally undetectable. Whether algal growth in the lakes is N limited under these conditions or whether light (or some other factor) is limiting algal growth requires further study. Because of the poor water clarity, the photic zone is much shallower than the epilimnetic mixing depth.

Another question that needs to be addressed is whether co-limitation of P and N has been occurring when both DRP and inorganic N were undetectable in the Yahara lakes. Bacteria (including cyanobacteria) can store P luxuriously as polyphosphates (Reynolds 1984, Hupfer *et al.* 2007), but intracellular storage mechanisms for inorganic N are less important (Kalff 2002). Thus, undetectable DRP in a eutrophic lake's surface waters during summer does not necessarily mean that P is growth limiting, especially if inorganic N is also undetectable. The algae may simply be storing P while their growth is N-limited. Another explanation is that different species of algae present in the same environment are limited by different nutrients (Fitzgerald 1969) or that other bacteria are competing with blue-green algae for nutrients. (After a 32-year career focusing on P as the nutrient controlling blue-green algal blooms in the eutrophic Yahara lakes, I am troubled by how little I know about the role of N.)

While scientists and managers have stressed P reduction programs for eutrophic freshwater lakes, much less attention has been given to N control mechanisms. In addition to the general belief that P is the most important growth-limiting nutrient in lakes, P also has a biogeochemical sedimentary cycle that is more easily controlled than N (Cooke *et al.* 2005, Cooke 2007). Howarth and Marino (2006) argued that the emphasis on P limitation in lakes prevented scientists for many years from recognizing that N can be the most important limiting nutrient in many estuarine systems. Even so, the authors stressed that N control strategies for coastal waters should go hand in hand with P control. This conforms to Tyrell's (1999) argument that N may be the 'proximate limiting nutrient' controlling local algal growth in relatively short periods of time in the ocean, whereas P may be the 'ultimate limiting nutrient' controlling total system productivity over long timescales.

The eutrophic Yahara lakes could be another system where reducing both P and N inputs from the watershed would have short-term and long-term benefits for controlling summer blue-green algal blooms. Fortunately, many best management practices to curb agricultural and urban nonpoint source pollution reduce inputs of both nutrients even though managers have emphasized P reduction aspects of the practices in the past. However, additional attention on reducing N fertilizer application rates may be warranted given the potential importance of N as a growth-limiting nutrient for algae.

Reducing inputs from both N and P makes even more sense for the future eutrophication management of the Yahara lakes given that zebra mussels may soon become infested as they have recently invaded lakes <50 km away. While the central pelagic waters of the Yahara lakes likely will become clearer as the mussels filter microscopic algae, shallow water conditions could become worse if scum-forming blue-green algal blooms pile up, and especially if filamentous algae growths proliferate from high dissolved nutrient levels in the water. Thus, the long-term limnological record for the deep central locations of the Yahara lakes will show improvements in lake water quality, but the public viewing the lakes from the shoreline will know otherwise.

Summary

Eutrophication of the four Yahara lakes has been dramatic since the mid-1800s when settlers of European ancestry first altered the landscape. For Lake Mendota, sediment erosion from higher water levels established by the damming of the lake's outlet, and from the rapid agricultural expansion of the lake's watershed, resulted in blue-green algal growths in ensuing years. However, the early eutrophication problems in Mendota were dwarfed by water quality problems stemming from wastewater inputs from Madison's inadequately treated sewage that directly entered Lake Monona from the late 1800s through 1936, and then Lake Waubesa until 1958 when Madison's wastewater effluent no longer entered the lakes. Blue-green algal bloom problems were so bad in the lower Yahara lakes that the Madison Public Health Department conducted major copper sulfate treatments throughout the summers of 1925–1954. During the years of wastewater inputs, inorganic N and especially DRP concentrations in the surface waters were very high (particularly in Waubesa and Kegonsa), indicating that N and P were not growth-limiting to algae. Once wastewater no longer entered the lakes, nutrient levels dropped rapidly. However, Waubesa and Kegonsa showed no legacy of the high P concentrations in sediments deposited during the wastewater input era; minimal P-binding potential due to low Fe availability in the lakes is the hypothesized reason.

Mendota's algal blooms were not considered to be a problem until the mid-1940s when wastewater inputs from upstream communities increased as well as the agricultural use of N and P fertilizers. This increase in eutrophication symptoms coincided with an increase in indices of DRP and inorganic N concentrations in the lake. Higher fall DRP concentrations may have been due in part to a change in Fe availability to scavenge DRP from the water column prior to and immediately following turnover, but research is ongoing to confirm this. Eutrophication problems came to a head in the mid-1960s coincident with the invasion and rapid expansion of Eurasian water milfoil, an exotic macrophyte well adapted to nutrient-rich systems with turbid waters. After the wastewater effluents were diverted from Lake Mendota in 1971, blue-green algal blooms persisted, shifting the onus of the problem to agricultural and urban nonpoint source pollution. Efforts to curb these P input sources to the lake have intensified in recent years with the Mendota Priority Watershed Project, a state-funded project with an 11-year implementation phase ending in 2008. While much progress has been made in recent years to control these pollution sources to Mendota, particularly related to controlling sources of soil and sediment erosion, manure management continues as a major problem. Besides rising soil P levels from the over-application of manure to agricultural lands, manure runoff during late winter constitutes a huge P loading source to Mendota. As evidence, P loadings during January to March were 48% of total loadings measured for 1990-2006 in the Yahara River subwatershed. Much of this runoff P was dissolved and not associated with high sediment loads, whereas during other months, more of the runoff P was bound to sediments that could settle out in lower stream reaches prior to entering the lake.

With Lake Mendota being the major source of nutrients to downstream Lake Monona and on down the Yahara River chain, nonpoint source pollution reductions from Mendota's watershed have paramount importance for solving the eutrophication problems in all the Yahara lakes. The apparent lack of Fe-binding potential for storing P in Mendota's recently deposited sediments, coupled with the lake's significantly greater water clarity following two separate two-year drought periods with reduced P loadings, provides an indication that reducing nonpoint source P inputs can produce noticeable reductions in algal blooms within a reasonable period of time.

And finally, high quality N and P data (with low analytical detection limits) collected since 1980 indicate that algal growth in the Yahara lakes during July–August may be limited by not only P, but N (especially in the lower Yahara lakes). Further research is needed to elucidate the role of N versus P as growth-limiting nutrients for algae, but the evidence is strong enough to suggest that both P and N inputs to the Yahara lakes should be controlled. While most of the management effort currently is focused on controlling P inputs, fortunately many best management practices reduce both nutrient inputs. However, additional attention on controlling N fertilizer application rates may be warranted.

Reducing both P and N will be important to prevent scum-forming blue-green algae blooms and especially filamentous algal growths that could become problematic in shallow lake waters once zebra mussels invade and proliferate in the Yahara lakes. This invasion may soon occur given the exotic mussels are now present in lakes <50 km away. Because zebra mussels, through their filtering of microscopic algae, will likely produce clearer water in the open-water regions of the Yahara lakes, the long-term limnological record collected at the deepest locations will indicate improvements have occurred in lake water quality. However, the public viewing the lakes from the shoreline will know otherwise.

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