MINIREVIEW

USE OF PHYTOPLANKTON IN LARGE RIVER MANAGEMENT

John D. Wehr
Louis Calder Center-Biological Station, Fordham University, P.O. Box K, Armonk, New York 10504

and

Jean-Pierre Descy
Facultés Universitaires Notre-Dame de la Paix, Unit of Freshwater Ecology, 61 Rue de Bruxelles, B-5000 Namur, Belgium

Key index words: freshwater; large rivers; phytoplankton; river impoundments

Abbreviations: POC, particulate organic carbon; RCC, river continuum concept; SRP, soluble reactive phosphate

Historically, rivers have served as sources of drinking water, fisheries resources, transportation routes, irrigation supplies, and waste removal systems. Human civilization has had many major effects on rivers, dating back more than 5000 years when Egyptians built dams on the Nile to supply water for crops and human consumption. Today, management of large rivers requires a balance between human needs and ecological integrity, although until quite recently, ecological principles have played a minor role in river management (Edwards 1995).

Planktonic algae are an important part of these issues because they play a central role in the functioning of large rivers. Algal communities are major producers of organic carbon in larger rivers, are a food source for planktonic consumers, and may represent the primary oxygen source in many low-gradient rivers (Thorp and Delong 1994, Köhler 1995, Reynolds and Descy 1996). Phytoplankton are responsive to excessive supplies of inorganic nutrients and may pose problems in long stretches of rivers with cultural eutrophication, but may also enhance water quality for humans in rivers affected by agricultural or industrial uses. Algal communities of river systems consist not only of suspended algae, but also a diverse benthic assemblage of macrophytic forms, smaller epilithic species, epiphytes, and sediment-dwelling forms (Reynolds 1996). This review focuses on the use of planktonic algae in river management because of their central importance in larger rivers and because of the growing need for ecosystem-level studies on river plankton.

WHAT ARE LARGE RIVERS?

Ecologists know much more about small tributaries than larger rivers, in large part because they are simply easier to study. Undoubtedly, this is why some authors define “large river” as a fluvial system large enough to intimidate researchers (Hynes 1989) or too large to study without a boat and specialized equipment (Stalnaker et al. 1989). Streams and rivers are classified by geomorphologists according to fluvial and watershed properties, which recognize a continuum of physical features (channel width, average depth, discharge, catchment area) as one progresses downstream (Morisawa 1968). When viewed from this perspective, the distinction between large and smaller rivers may be considered arbitrary (Stalnaker et al. 1989, Townsend 1996). Ecological theory developed for river ecosystems, such as the RCC (Vannote et al. 1980) and Nutrient Spiraling model (Newbold et al. 1981), employ longitudinally integrated processes as factors affecting resource supply and ecosystem function. The relevance and usefulness of the RCC for large rivers has been questioned (Sedell et al. 1989, Thorp and Delong 1994, Townsend 1996), but the definition of “large river” that has emerged from these studies is that of rivers greater than sixth order (based on Strahler 1957). Other workers examining the importance of phytoplankton production to carbon supply and system function have suggested that simply “low-gradient” or “lowland” rivers may be more useful concepts because their metabolism is frequently dominated by algal or other in situ autotrophic organisms, despite the fact that many are light-limited due to depth and turbidity (Baker and Baker 1979, Descy and Gosselain 1994, Köhler 1995, Reynolds and Descy 1996). This latter perspective has merit, because downstream reaches of smaller rivers possess many features common to large rivers (floodplain influences, high turbidity, lateral carbon inputs, lower gradient). Other authors maintain that large rivers should be viewed as systems distinct from upland sections, with ecological features and management problems that deserve particular attention (Sedell et al. 1989, Admiraal et al. 1994). Indeed, large rivers are widely recognized and even defined as manipulated or regulated ecosystems (Ward and Stanford 1983, Décamp 1996). Therefore, our review focuses on management concerns...
of lowland rivers greater than sixth order and the role that phytoplankton may play in these systems.

**MANAGEMENT ISSUES**

Many studies document the effects of human disturbances on communities in large and small rivers, but few of these findings have been adequately communicated to water management agencies (Petts et al. 1995). This condition exists despite the fact that most large rivers of the world are now or will soon be substantially altered by human activities (Sparks 1995). A few of the important disturbances include channelization, navigation or hydroelectric dams, removal of adjacent wetlands, toxic pollutants, and cultural eutrophication (Johnson et al. 1995, Petts et al. 1995). Management of these large-river disturbances is complicated by the fundamental feature common to all flowing waters: activities or disturbances at one location affect processes and organisms downstream (Fig. 1). Due to the longitudinal nature of rivers, management problems that arise are frequently more difficult to manage than in lakes. For example, excessive nutrients may enter the system from several points along the main channel, and the forms of nutrients also depend on various watershed processes (e.g. hydrogeological processes as nutrients transfer through soils and groundwater), which are not well understood. Downstream transport of dissolved and particulate substances often requires that management schemes consider large river stretches (>10^2–10^3 km) or entire basins, as well as adjacent drainage areas.

Human disturbances affect not only riverine organisms and processes, but also the links between a river and its watershed, by disrupting lateral linkages with floodplains and groundwater (Townsend 1996). Perhaps this is one reason why computer models that have attempted to predict phytoplankton production or species composition in large rivers have met with only modest success (Billen et al. 1994, Ruse and Love 1997). River ecosystems are actually networks of interactions among physical, chemical, and biological processes, which reach a higher degree of complexity downstream. There are also longitudinal differences in the time scales of chemical and biological processes. Such complexity renders it difficult to design policies and assess the results of management actions. Recently, studies have taken into account riverine, watershed, and land use variables, as well as differences in spatial scale in devising management models for river water and fish habitat quality (Hunsaker and Levine 1995, Roth et al. 1995, Smitz et al. 1997).

Use of phytoplankton in management has had a fairly long history. Concern most often centers on adverse effects, such as taste and odor problems, filter clogging at water treatment plants, and effects on fisheries (Palmer 1962). Fewer studies have documented the beneficial uses of algae in management practice. How algae are applied depends on the problem, as well as on river size and flow regime. For example, in the highly regulated Buffalo River (South Africa), high nutrient loads were reduced by sedimenting algal populations along impounded stretches, resulting in the ability to release relatively low-nutrient water below the dams (O’Keefe et al. 1990). Studies using algal community structure or accumulation of toxic metals have been employed in large-scale analyses of water quality in many streams and rivers (del Giorgio et al. 1991, Whitton et al. 1991, Lowe and Pan 1996). An important complication to their use in management is that the composition, dynamics, and production of algal communities in large rivers is less well understood than in either lakes or small streams.

**RIVER PHYTOPLANKTON ECOLOGY: A SHORT REVIEW**

Various aspects of potamoplankton, the community of suspended algae in flowing water, have been reviewed by Reynolds and Descy (1996). These authors highlight several key points specific to the composition and ecology of planktonic algal assemblages in large rivers. Ironically, many of the most important features emphasized were understood long ago (Kofoid 1908, Krieger 1927, Reinhard 1931, Butcher 1932, Welch 1952). Among questions left unresolved, the origin of phytoplankton assemblages in running waters is most often raised. Even the earliest studies perceived that algal inocula from various sources contributed to the main channel, but potamoplankton is also “native” to rivers, as demonstrated more precisely by Reynolds (1995) and Reynolds and Glaister (1993). The actual sources of inocula (backwaters connected to the main channel or dead zones within the channel) are still uncertain and likely vary among river systems. It is tempting to speculate that most planktonic algae in rivers are meroplanktonic (planktonic with a benthic life stage) and recolonize littoral areas or the river bed for survival and perennation. The benthos clearly serves as the origin of suspended algae in smaller streams (Swanson and Bachmann 1976).
In such cases, the chlorophyll a flux is positively related to discharge. However, in large rivers, phytoplankton biomass varies inversely with discharge (Schmidt 1994); this distinction is an important characteristic that separates small from large rivers.

**Composition and selection.** The most successful algal groups in large rivers are diatoms and green algae; the latter tending to dominate in summer conditions, although this pattern is far from consistent. Cryptophytes and cyanobacteria may also at times be abundant, whereas other groups, chrysophytes, dinoflagellates, and Euglenophytes are usually less developed, despite records of blooms of particular taxa in specific conditions (e.g. Ohio River: Wehr and Thorp 1997, St. Lawrence River: Hudon et al. 1996). Plankton surveys provide lists of up to several hundred taxa, but the list of common and truly planktonic taxa is much less (Reynolds and Descy 1996), and (unlike benthic macroalgal species), few if any planktonic taxa are specific to rivers (Round 1981).

The principal selective factors affecting river phytoplankton were clearly summarized by Reynolds (1988, 1995):

- Nutrient limitation is unlikely in rivers, despite the influence of algal growth on nutrient (Si, P) levels. Nutrients in rivers are often in considerable excess of algal requirements, such that models based on nutrient resource ratios in lakes (Tilman et al. 1982) are not applicable in most rivers.
- Phytoplankton abundance and production are controlled by discharge. This is related to residence time, channel depth, and dilution rate and affects water transparency and sedimentation.
- Maintaining stable or persistent phytoplankton communities is impossible in large (deep and turbid) rivers if these systems exist as fully mixed reaches. Therefore, river mixing must be incomplete (Margalef 1960), which is provided by heterogeneity in channel morphology.

The river environment selects for fast-growing species able to cope with wide variations in light conditions, which depend on incident light, water transparency, and depth (not unlike turbid, well-mixed shallow lakes; Reynolds et al. 1994). There are very few studies demonstrating nutrient limitation in larger rivers (Moss and Balls 1989) or dependence of algal growth upon P availability (Reynolds and Descy [1996], but see Basu and Pick [1996] for a contrary view). Such issues are important when dealing with the effects of eutrophication in rivers. Analysis of long-term data shows that increases in algal biomass that have occurred in recent decades in many European and North American rivers may have resulted from increased nutrient inputs coupled with hydrological changes and river regulation.

**Biomass, longitudinal variation, and driving variables.**

A wide range of phytoplankton biomass has been reported for rivers, from <1 μg (Orinoco River, Brazil: Lewis 1988) up to ≈400 μg chlorophyll a·L⁻¹ (several rivers: Jones 1984, Moss et al. 1984). The largest values are comparable to those reported for highly eutrophic ponds. It is in the higher range where potamoplankton may cause harmful effects for water use and even decreases in oxygen concentrations. However, as algal photosynthesis is often the major supplier of oxygen to slowly flowing rivers, phytoplankton loss may also result in water quality problems (Descy 1992).

That potamoplankton biomass exhibits variations along the course of a river has been known for many decades (Welch 1952). Simulation models show this pattern develops in four phases, progressing downstream (no plankton in the headwaters, increase, maximum, decline), and results from interactions among river morphology, hydrology, light availability, and algal growth rate (Descy and Gosselain 1994, Garnier et al. 1995). Growth rates in eutrophic rivers strongly depend on the balance between algal photosynthesis and respiration, which is positive in shallow, transparent river channels and may become negative in deep and turbid reaches. This deficit in phytoplankton growth is particularly true in river estuaries, such as the tidal Hudson River, New York (Cole et al. 1992). Phytoplankton in lower reaches are likely imported from upstream (less turbid) stretches that support positive net algal production. These longitudinal dynamics are driven by downstream transport of plankton into reaches where growth fluctuates in response to physical factors; therefore, measures against eutrophication must consider the entire river system.

The view that potamoplankton development in large rivers takes place mostly in the main channel and is heavily dependent on physical characteristics is applicable not just to regulated or eutrophic rivers. The same processes may govern phytoplankton growth in rivers with extensive floodplains, as shown by Lewis (1988) for the Orinoco River. On an annual basis, plankton gross production was 9.5 g C·m⁻²; the floodplain contributed only 0.25 g C·m⁻². At the upper end of the production range, the River Loire, France (Lair and Sargos 1993, Lair and Reyes-Marchant 1997), provides an example of a river with massive potamoplankton growth, but limited flow regulation, where exchanges (nutrients, organisms) with the littoral zone substantially enhance plankton development.

Despite many studies to the contrary, nutrient limitation should not be excluded as a factor in the longitudinal development of river phytoplankton. Data from rivers in southeastern Ontario and western Quebec reveal a positive relationship between total P and chlorophyll a (Basu and Pick 1996). We suggest, however, that a clear demonstration of nutrient limitation in river systems is generally still lacking. Such a relation is likely true in systems with few nutrient sources, long residence times, and high water clarity (or shallow depth). Few large rivers worldwide combine these conditions, and many if not
Role of loss processes in river phytoplankton dynamics. Phytoplankton development in some rivers may be favored because of lower grazing pressure and reduced sedimentation rates (Cole et al. 1991, 1992). There are some data indicating that zooplankton biomass in lotic environments is lower than in lakes (Pace et al. 1992, Thorp et al. 1994) and that phytoplankton–zooplankton interactions are weak in most river systems (Köhler 1995, Basu and Pick 1996). However, declines in potamoplankton numbers or biomass do occur in otherwise favorable conditions for algal growth (de Ruyter van Steveninck et al. 1992, Gosselain et al. 1994). These declines may result from poor light availability combined with zooplankton grazing and sedimentation losses. However, few field studies have quantified the impact of zooplankton grazing on phytoplankton in large rivers (Thorp et al. 1994, Gosselain et al. 1998). The primary constraint in rivers is residence time, which affects both phytoplankton and their grazers (Pace et al. 1992, Thorp et al. 1994). Rivers usually select for small-bodied zooplankton with an ability to grow rapidly enough to compensate for limited residence times (Viroux 1997), but which may have a low capacity to regulate phytoplankton biomass. Several authors have reported decreases in phytoplankton numbers brought about by benthic filter feeders, especially zebra mussels (Dreissena polymorpha; Effler et al. 1996, Roditi et al. 1996, Strayer et al. 1996) and other invaders, such as Corbicula sp. and Corophium curvispinum (Bachmann et al. 1995). Voracious feeding rates coupled with very large populations have dramatic effects on phytoplankton, water quality, and ecosystem processes in large rivers. These recent and dramatic developments in many large rivers (e.g. Mississippi, Ohio, St. Lawrence, Hudson) call for specific management actions.

Almost no studies have shown that sedimentation in rivers is an important loss process for phytoplankton. To date, sedimentation losses have been mostly calculated, taking into account water velocity, depth, and specific settling velocity of algal species (Carling 1992). However, it is likely that sedimentation is a selective factor that may suppress diatom numbers and perhaps prevent their dominance in shallow river stretches, especially in regions of very low flow (Reynolds 1995).

CASE STUDIES AND OTHER MANAGEMENT ISSUES

Eutrophication in large rivers. Most large European rivers have a long record of eutrophication attributed to human influences (Friedrich and Viehweg 1984, Descy 1987), which raises concern about the sustainability of domestic uses of river water. This is especially true when river water is abstracted and intensively used, such as with the River Meuse (Belgium), a source of drinking water for about six million people (Descy 1992). Although increased algal growth is commonly thought of as resulting principally from increased nutrient loading, it is far from clear that this is true in most rivers. For example, in the case of River Danube (Hungary), there are excellent algological records spanning >40 years (Szemes 1967, Kiss 1994). A 10-fold increase in phytoplankton densities occurred during the 1970s (Fig. 2) without a noticeable increase in the nutrient supply since the end of the 1950s. Rather, changes in suspended matter transport, brought about by reservoirs in the German and Austrian sections, improved water transparency and light availability for algal growth. It is also likely that increased residence time from the exploitation of the upper Danube favored the development of euplanktonic species in the river.

A similar phytoplankton increase was observed in the River Meuse, Belgium, by water supply companies since the early 1970s (Descy 1992). Strong increases in mean annual chlorophyll a occurred with no clear trend in P concentrations within the Belgian sector (SRP: 50–100 µg·L$^{-1}$). Despite increases in N concentrations in the Meuse (especially NO$_3^-$),...
as in other European rivers (Van Dijk 1996), levels in the 1970s were already in the mg L\(^{-1}\) range, well exceeding algal demand. Phytoplankton blooms in the Meuse may have built up in the French sector (Léglize and Salleron 1988) as a result of water treatment plants lacking P-stripping and other tertiary treatment capabilities. The Meuse received increased P in river reaches that already had optimal hydrological features for phytoplankton growth. As a consequence, algal biomass represented an average of 58% of the total POC transported in the upper Belgian section of the Meuse; similar values have been reported for other large rivers (Descy and Gosselain 1994).

Similar long-term, high nutrient levels (1960 to present) and flow regulation have also prevailed in the Ohio River (Seilheimer 1963, Nall 1965, Wehr and Thorp 1997). Under these conditions, it is unclear whether hydraulic or chemical factors play the major role in favoring high biomass. The upper Mississippi also supports large populations of algal species regarded as typical of eutrophic waters (Microcystis aeruginosa, Aphanizomenon flos-aquae, Aulacoseira granulata, A. italica), and although nutrient levels were generally high, there was no evidence that standing crop correlated with N or P concentration (Lange and Rada 1993, Huff 1986). Authors also contend that discharge and temperature were the most likely controlling factors of algal biomass through navigation dams and channelization in the Mississippi. Such conditions are common in most large rivers of the world and illustrate that even when long-term nutrient data are available, it is not always straightforward to make conclusions about the primary causes of eutrophication.

Some eutrophic rivers carry such heavy burdens of N and P that identification of the critical limiting nutrient for regulation by a management agency may not be possible. This is because standard nutrient addition bioassays (e.g. Cain and Trainor 1973, Gerhart and Likens 1975) will not produce the necessary growth responses when supplies already exceed demand. An alternative method has been devised for the periodically hypereutrophic Neuse River (North Carolina), in which “dilution bioassays” apply stepwise dilutions of N, P, Fe, and trace metals (Paerl and Bowles 1987). Such tests identify the nutrient(s) most responsible for triggering massive algal growth and help set target nutrient levels that management agencies may recommend for river systems.

Solutions to reduce nuisance blooms in these very eutrophic rivers may lie in hydraulic control (i.e. natural flow conditions) whenever possible but must also involve P reduction. At least a 90% reduction in P is required in many large rivers because most of the algal species that thrive in large rivers such as in the Danube, Meuse, and Ohio (e.g. small species of Stephanodiscus, Cyclotella, and Chlorella-like green algae) have half-saturation growth constants \(\leq 10\ \mu\text{g P L}^{-1}\) (Van Donk and Kilham 1990, Lampert and Sommer 1997). This is \(\leq 1\%\) of current P concentrations in many rivers. Furthermore, taking into account intracellular storage and downstream displacement of water masses, careful treatment strategies must be designed to induce P limitation in large rivers. To develop and test such strategies, reliable simulation models must quantify nutrient inputs, river network features, and biological responses.

Potamoplankton occupies a key place in the oxygen budget of a river at least during the growing season. On one hand, algal photosynthesis is a major source of oxygen production in most large and/or lowland rivers. On the other hand, organic matter production in eutrophic rivers in excess of grazer requirements is principally degraded by heterotrophic bacteria following phytoplankton mortality, and can lead to severe oxygen depletion. These deleterious effects may be seen as a result of river-based power plants, as demonstrated in model simulations of the James River estuary, Virginia (Smith and Jensen 1974). These studies show how increased eutrophication may interfere with industrial uses of river water, affecting algal mortality and physiological changes from thermal shock. Further effects include chlorine treatment and entrainment, the effects of which are often difficult to sort out within the complex situations in the field. Finally, eutrophication of large rivers discharging into the sea have also had serious effects on water quality of coastal regions, through increases in nutrient loading and changes in nutrient ratios (Rabalais et al. 1996, Billen and Garnier 1997, Lorenz et al. 1997).

**Effects of exotic invaders.** Recent literature has focused on phytoplankton declines resulting from biological invasions in large rivers, as well as lakes. Such decreases in phytoplankton biomass have been documented from several European and North American rivers (de Ruyter van Steveninck et al. 1992, Descy 1993, Effler and Siegfried 1994, Gosselain et al. 1994, Garnier et al. 1995, Köhler 1995), although not all have received a definitive explanation. Zooplankton grazing may play a role in some cases (Gosselain et al. 1998), although short-lived zooplankton that develop in rivers are generally unable to control algal biomass over longer periods of time (i.e. several weeks). The most studied filter feeders affecting phytoplankton declines is the zebra mussel Dreissena polymorpha, which invaded North American rivers in the early 1990s (Effler et al. 1996, Roditi et al. 1996, Strayer et al. 1996, Cope et al. 1997, Gist et al. 1997). In the Hudson River, large populations built up between 1992 and 1994 and reduced phytoplankton abundance by a factor of 5 to 10 compared to preinvasion levels (Caraco et al. 1997). Moreover, unlike small planktonic herbivores, zebra mussels apparently feed effectively on most phytoplankton sizes, thereby controlling a large fraction of riverine algal biomass. Similar ef-
effects have been described in the Seneca River, New York (Effler et al. 1996), where large densities of mussels also contribute to low oxygen concentrations. Although perhaps less dramatic, some European rivers have suffered from recent invasions by Dreissena and other exotic filter feeders (Bachmann et al. 1995). What measures should be taken to reduce the spread of these organisms is not clear, and there is a debate about whether they may be limited by availability of substratum or food (Strayer et al. 1996). It is reasonable to assume, however, that pollution control measures to reduce algal proliferation (e.g. reductions in P inputs) may have at least two beneficial effects on these rivers: (1) decreases in the intensity and frequency of nuisance blooms and (2) a smaller food supply for zebra mussels. Some predictions may be possible through simulation models, provided that reliable data on filter feeder densities, population dynamics, and in situ filtering rates are available. But long-term algal monitoring coupled with reductions in nutrient loading is still necessary.

Cyanobacteria in rivers. A large contribution of colonial cyanobacteria to potamoplankton has been most often reported from downstream reaches of warm-temperate and tropical rivers, such as the Rio Salado, Argentina (O’Farrell 1993), Nagdonk River, South Korea (Ha et al. 1998), Neuse River, North Carolina (Paerl and Bowles 1987), and Darling River, Australia (Hotzel and Croome 1994). However, cyanobacteria blooms comprising potentially toxic strains have been reported periodically in cold temperate rivers. These include the Ohio River over at least three decades (Seilheimer 1963, Nall 1965, Petersen and Stevenson 1989, Wehr and Thorp 1997) and the upper Mississippi since the 1930s (e.g. Reinhard 1931, Baker and Baker 1979, Huff 1986). Clearly, cyanobacterial blooms in large rivers are not a recent phenomenon: Microcystis blooms have been reported since 1930 in the Potomac River, Maryland (Krogmann et al. 1986). However, slow growth rates of many of these organisms likely do not allow them to develop large populations in large rivers unless there are relatively high residence times. Accordingly, in the Rhine and Meuse (Germany and Belgium), bloom-forming cyanobacteria build large populations in lower reaches of these rivers, but only for short periods during summer (Ibelings et al. 1998). Interestingly, the 1995 survey in the Rhine indicated that cyanobacteria developed large populations in summer in the Untersee, but quickly disappeared in the river downstream. A significant portion of the bloom (Planktothrix agardhii, plus small Chroococcales) was later observed near Lobith, \( \approx 800 \) km downstream of Untersee.

The conditions under which cyanobacterial blooms develop in the Murray River (Australia) have been studied in detail by Bormans et al. (1997). Although slow flow (current speed \( <0.05 \text{ m s}^{-1} \)) has been put forward as a key factor, temperature stratification is also usually necessary for the development of Anabaena circinalis blooms. It is interesting to note that the model used for simulating the Anacystis nidulans-Anabaena sp. alternation in the Murray River comprised a detailed physical submodel and a simple algal submodel based on algal growth (as a function of light) and a buoyancy coefficient for the two taxa. Sufficient discharge seems the best measure to suppress cyanobacterial blooms, perhaps along with artificial destratification (Webster et al. 1996).

While the risk of potentially harmful cyanobacterial blooms in large temperate rivers is small, some systems with high residence times or connected to eutrophic ponds or lakes may still be prone to bloom development (e.g. River Bure, Moss et al. 1984). Nonetheless, it is important to remain concerned with the possible expansion of cyanobacterial species (Couté et al. 1997, Padišák 1997) and to be able to assess the risk of harmful blooms in fluvial systems.

ARE POTAMOPLANKTON USEFUL INDICATORS OF THE ECOLOGICAL STATUS OF RIVERS?

In lakes, phytoplankton biomass has been used for decades to assess trophic status and to identify artificial eutrophication induced by human activities (Wetzel 1983, Harper 1992). Moreover, detailed studies based on long-term records of phytoplankton numbers and taxonomic composition have shown that, in general, phytoplankton are very sensitive indicators of various environmental changes (Maberly et al. 1994). So far, no such biomass–nutrient indicator system has been successfully applied in large or lowland rivers, mainly because longitudinal dynamics prevent the reliable use of simple trophic scales based on chlorophyll \( a \). Potamoplankton dynamics respond primarily to physical factors and may fluctuate considerably in time and space. Perhaps this is why phytoplankton abundance and biomass data from large rivers frequently do not correlate strongly with nutrient chemistry, such as in the River Severn, U.K. (Ruse and Love 1997), or may even correlate negatively, as in the Ohio River (Wehr and Thorp 1997).

Physical factors, along with nutrients and biotic interactions (grazing), also influence community composition, but we are a long way from being able to sort out which are the most important determinants at the species level. Even at the level of major taxonomic groups, only basic trends can be identified (Reynolds 1994). Thus, our present limited understanding of processes that determine potamoplankton composition currently prevents management agencies from developing predictive tools for assessing ecological quality based on phytoplankton community structure in large rivers. Attempts to use planktonic algae as indicators of river water quality using a community structure approach similar to methods employed for benthic diatoms and ma-
croinvertebrates (Plafkin et al. 1989, Lowe and Pan 1996) have so far been unsuccessful (Williams 1964, 1972, del Giorgio et al. 1991). Studies have shown that changes in phytoplankton composition reflect not only variations in water quality, but also changes in physical variables and biotic interactions. Variations in water chemistry may alter relative proportions of a few dominant taxa but often have little effect on the overall assemblage.

The ability to apply phytoplankton-based assessments of the ecological status of large and lowland rivers lies in a better understanding of the growth rates and requirements of suspended algal species in rivers. As some studies have shown (e.g. Kiss 1994), major changes in a watershed can be detected by biomass changes and by alteration of the growth pattern of certain potamoplankton species. Therefore, a system may be devised that evaluates ecological change by contrasting the present community with reference data from unaltered stretches or time periods. Moreover, as good historical data are rarely available, the reference situation may be simulated through a comprehensive river phytoplankton model, based on known individual river phytoplankton growth submodels. The potamoplankton models currently available, however, are still unable to simulate the potamoplankton composition under the dynamic conditions that prevail in rivers, although the attempts made so far are promising (Dauta 1983, Garnier et al. 1995, Cloat and Le Roux 1997, Ruse and Love 1997). We suggest that the main difficulty in developing a potamoplankton model lies not in incorporating more river–watershed features, examples of which already exist (e.g. Smitz et al. 1997); rather, new models must include information on physiological parameters of the algal taxa present. This has yet to be attempted comprehensively for any large river system to date. While this goal may appear to be a very large task, detailed lake phytoplankton models have been developed using these principles quite successfully (Hilton et al. 1992, Reynolds 1996, Reynolds and Irish 1997). Therefore, river ecologists should build on this knowledge and adapt new algal growth models for river environments.

Thanks go to Dr. James H. Thorp, Clarkson University, New York, and Dr. Wim Admiraal, University of Amsterdam, for sharing their ideas on large rivers and phytoplankton. This paper is contribution 174 from the Louis Calder Center—Biological Station.


Gist, D. H., Miller, M. C. & Brence, W. A. 1997. Annual reproduc-


