Large Rivers and Eutrophication

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Abstract

This paper discusses the effects of nutrients on the physical, chemical and biological components of large rivers in the context of both human costs and ecosystem response. In addition to the direct consequences of adding nutrients, such as increased primary productivity and resultant effects on water chemistry, additional interactions also characterize large river nutrient responses: for example, algal community structure is altered leading to proliferation of nuisance taxa, taste and odor problems, increased water treatment costs, increased toxins (cyanobacterial blooms), and loss of habitat. In addition, nuisance macrophytes (rooted aquatic plants) also increase and affect water chemistry and habitat. As a result of these direct and indirect responses, large rivers suffer impacts to aquatic life and recreational uses. Rivers also function to transport nutrients to downstream ecosystems, and some of the impacts of nutrients on large rivers are transported to downstream lake and coastal receiving waters.

1. Introduction

Water quality in rivers is vital to humans and to maintenance of biotic and ecological integrity. Rivers are part of a continuum of flowing water and no clear delineations exist to differentiate between small streams and large rivers. For example, from an agency sampling point of view, large rivers are not wadable, whereas an alternative definition considers large rivers to be navigable. Some quantifiable characteristics often used to delineate large rivers from smaller streams are drainage basin size and discharge. Some of the stream/river size cutoffs that have been used include basins >100 km^2 for small rivers (Meybeck and Helmer 1989), 100-10,000 km^2 for medium, and > 10,000 km^2 for large rivers (Caraco et al. 2003). Discharge in large rivers typically exceeds 100 cfs (2.83 $m^3 s^{-1}$) for at least part of the year.

Eutrophication refers to increases in plant and algal production. In some waterbody types, this can be a natural process, but in the vast majority of instances, eutrophication is brought about by human inputs of nutrients and is termed "cultural eutrophication" (Dodds 2002). Eutrophication occurs widely in large rivers of the United States, perhaps because large rivers integrate such large land areas. This paper reviews what is known about large rivers, their natural ecology, and the effects of nutrient enrichment on these systems. First, we will provide basic descriptions of river morphology, hydrology, baseline nutrients, and food webs. Then we will consider physical and chemical alterations related to food webs. Finally, we will cover effects of eutrophication on river ecosystem structure and function.

River morphology is dependent on basin geology, climate, and vegetation (Leopold 1994). In areas dominated by depositional sand or gravel, rivers tend to be low gradient and become shallow and broad, or braided. These rivers often have broad, poorly constrained floodplains and numerous riparian wetland features (e.g., the lower Mississippi). Steeper gradients are common when rivers flow through rocky, alluvial areas. These rivers become deeper, have incised canyons and constrained flood plains, and often have areas of intense rapids or whitewater (e.g., the Colorado River).

Generally, as discharge increases, rivers get wider more quickly than they get deeper, increasing the width to depth ratio. This pattern is displayed by the Mississippi River, which is approximately 100 times wider than it is deep in the lower segment (Leopold 1994). Width:depth ratio is more typically 1 to 10 for smaller streams. Characteristic widening causes the influence of riparian vegetation to become less important as the river gets larger. Most rivers in a natural state have riparian wetland habitat, such as oxbows and side channels that are sometimes connected to the main river channel through overland flow or groundwater exchange. Therefore, riparian influences are not completely absent, but are more seasonal based on hydrology. These side channels can be important to the nutrient retention of rivers during floods.

River hydrology depends on climate, vegetation, and slope (Dodds 1997). Therefore, water residence time can be quite variable among rivers. Generally, rivers contain a

larger volume of water and have more stable flow than smaller streams. The water column is usually well mixed vertically, but less so laterally, and a distinctly faster moving thalweg (central region with higher water velocity) and slower moving side channels are common.

Ranges of most naturally occurring elements in rivers span from 1 to 2 orders of magnitude (Dodds 2002) and vary with basin geology and climate (Meybeck and Helmer 1989). Rivers are greatly affected by humans (e.g., wastewater effluent, water withdrawal, nonpoint pollution). Rivers tend to receive more inputs from natural and anthropogenic point and nonpoint sources than smaller streams because they drain larger watershed areas. Also, because many major cities are located on large rivers, their watersheds are home to larger human populations. Increased anthropogenic inputs increase dissolved ion concentrations and suspended material in rivers, such as sediments and nutrients. Therefore, rivers are more likely to be turbid than smaller streams in the same basin (e.g., Meybeck et al. 1999, Whiles and Dodds 2002), and nutrient transport is typically greater than retention (net uptake of nutrients, Alexander et al. 2000, Bernot and Dodds 2005).

In rivers, primary production in the main channel can be benthic (i.e., periphyton or macrophytes) or planktonic (suspended algae, also referred to as sestonic). Key factors determining whether the dominant primary producer type is benthic or planktonic are hydraulic residence time, turbidity, and channel morphology. Plankton and macrophytes become more prominent as conditions become more lake-like with slower flushing rates (Van Nieuwenhuyse and Jones1996). Some river food webs can be based mainly on planktonic production (Thorp and Delong 1998). Side channel and oxbow primary production may be driven by macrophytes or emergent vegetation. Some large rivers have lakes periodically connected to them. These mostly act as shallow lakes and the balance of their nutrient input comes from inundation during flooding. If river nutrient load is high when flooding occurs, nutrient loading into lakes can dramatically increase, altering these ecosystems (Hamilton and Lewis 1990).

In silty and sandy rivers with shifting bottoms, most macroinvertebrates are associated with stable patches such as wood, logjams, or rocky areas. In general, large rivers have more diverse fish assemblages than small streams (Matthews 2003). Some large fishes such as sturgeons and paddlefishes are specially adapted to large rivers and not found elsewhere.

Three main theories predicting the major carbon (C) source fueling river food webs are: the river continuum concept (RCC) (Vannote et al. 1980), the flood pulse concept (FPC) (Junk et al. 1989), and the riverine productivity model (RPM) (Thorp and Delong 1994). The major C source according to the RCC is from upstream transport, but planktonic production is acknowledged. The FPC suggests that food webs depend on terrestrial C from the floodplain, and the RPM stresses the importance of local autochthonous C produced in the channel.

Very little is currently known about cycling, processing, and retention of nutrients in rivers other than what can be inferred from other systems. To date, the majority of research has been modeling studies (Alexander et al. 2000, Seitzinger et al. 2002, Darracq and Destouni 2005, Wollheim et al. 2006), and much knowledge of nutrient cycling in lotic systems has been gleaned from smaller streams. Most models predict that nutrient retention per unit length of stream declines with increasing stream size. The model of Alexander et al. (2000), which incorporated data from 374 U.S. stream stations across a large gradient, predicts less in-stream retention and greater transport of nitrogen (N) with increasing stream depth. Additionally, river nutrient concentrations are linked to watershed population, human activity, and energy consumption (Meybeck 1982). The model created by Caraco et al. (2003), using data from 249 North Temperate watersheds, predicted a nitrate export of 360, 630, and 443 kg N km^{-2} watershed area yr^{-1} for small, medium, and large rivers respectively. Population estimates for these watersheds were 38, 117, and 53 individuals per km^2.

Natural N and phosphorus (P) concentrations in rivers vary with basin geology, vegetation, and climate (Meybeck 1996). Natural NO_3 and PO_4 concentrations in rivers range from 0.05 to 0.2 and from 0.002 to 0.025 mg L^{-1}, respectively (Meybeck and Helmer 1989). Rivers tend to have less temporal and spatial variability in nutrient concentrations than smaller streams (Smith et al. 2006). Worldwide it is documented that nutrient concentrations are increasing in rivers by at least 50% (Dodds 2002). Total nitrogen (TN) and total phosphorus (TP) concentrations have been decreasing slightly in the United States over the past 30 years, following a reduction in agricultural intensity (Alexander and Smith 2006). However, changes in river trophic state in response to nutrient reductions are unclear.

Nutrient criteria for streams may be needed to avoid direct toxicity, taste, and odor problems, alterations in biotic integrity, and interference with recreation (Dodds and Welch 2002). Similar issues occur for rivers. For example, many municipalities rely on rivers for drinking water. Nitrate concentrations in drinking water above 10 mg L^{-1} can negatively affect human health by causing methemaglobanemia (blue baby syndrome when it occurs in infants). Excess nutrients can also indirectly harm humans through increased water treatment costs. Algal and cyanobacterial blooms can cause taste and odor problems, and increase toxin levels in river water (e.g., the Murray-Darling River system in Australia, Maier et al. 2001, Davis and Koop 2006). Algal blooms can also interfere with recreational uses such as boating, swimming, fishing, and tourism due to increased phytoplankton, periphyton, and/or macrophyte biomass. Finally, eutrophication of rivers can alter biotic community composition and decrease biotic integrity (Miltner and Rankin 1998).

The objective of this document is to discuss the effects of river eutrophication in terms of a river's physical, chemical, and biological components, and in the context of both human costs and ecological responses. In addition to the direct consequences of adding nutrients, such as increased primary productivity, additional interactions may alter the ecosystem. Due to methodological and technical limitations of large river research, studies on the effects of eutrophication on these systems have been limited. However, the

importance of rivers to watershed, basin, and continental nutrient cycling, storage, and transport is evident.

2. Biological Responses

2.1 Ecosystem Structure

There are few direct effects of eutrophication on the physical structure of rivers. Since eutrophication in rivers causes chemical and biological changes that are tightly linked, we will discuss them collectively. Structural responses in river biota can be altered both directly and indirectly by the addition of nutrients. Primary producers that can use the additional resource directly should be more rapidly affected by the addition than higher trophic levels. There is little empirical evidence of changes in microbial assemblage structure in rivers due to eutrophication. There is no reason to assume, however, that microbes in rivers should behave differently than microbes in smaller streams (Dodds 2006). Dumestre et al. (2001) demonstrated a change in microbial community structure with eutrophication below a reservoir on the Sinnamary River, South America. At this point, not a lot is known about the consequences of a microbial structural change or how such changes would indirectly affect the riverine food web.

Algal responses to nutrients are the most studied feature of river eutrophication. Algal community structure is directly affected by eutrophication, resulting in increased biomass and altered community composition (Butcher 1947). N, P, or both N and P can limit phytoplankton and benthic algal growth in rivers (Watson 1989, Francoeur 2001, Tank and Dodds 2003). Chetelat et al. (2006) found phytoplankton biomass across 46 rivers in Ontario and western Quebec to be more dependent on TP concentration than residence time or light. TP was also found to be strongly correlated to phytoplankton biomass in a study of 115 North American Temperate streams (Van Nieuwenhuyse and Jones 1996).

Indirect effects of eutrophication in rivers are principally reactions to an overgrowth of algal biomass or change in algal community structure, including the proliferation of nuisance and toxic species. For example, there are positive responses of the nuisance alga *Cladophora* to altered nutrient concentrations in the Clark Fork River, Montana (Dodds et al. 1997), which alters the river ecosystem's structure (Dodds 1992). Eutrophication can also alter benthic cyanobacterial species composition and distribution along a nutrient gradient, with many eutrophic cyanobacteria producing taste and odor problems and even toxins (Perona and Mateo 2006). In more lentic riverine habitats, additional nutrients can stimulate *Chara* or *Nitella*, which can also create taste and odor problems. Other problems include hypoxia in parts of, or throughout, the water column, and a reduction in recreation potential of the river. In the Murry-Darling River basin, Australia (Maier et al. 2001, Mitrovic et al. 2003, Davis and Koop 2006) and the Tualatin River, Oregon, reduced flow coupled with high nutrient loading stimulated cyanobacterial blooms and hypoxic conditions in the bottom of the river channel. And in North Carolina, excess phytoplankton biomass was the source of labile carbon for hypoxic conditions in a piedmont river (Mallin et al. 2006).

River macrophytes respond to eutrophication with increases in biomass (Chambers et al. 1994). Empirical evidence for this response was found in the Saskatchewan River, where a reduction in nutrient concentrations led to reduced macrophyte biomass (Chambers et al. 1999). Water hyacinth also does well in high nutrient conditions; it is used in nutrient removal for sewage treatment (Kumar and Garde 1990). Water hyacinth is a nuisance plant in many tropical and sub-tropical regions (Dodds 2002), and increased loading could exacerbate growth problems with this plant and others.

The direct effects of nutrient enrichment on river macroinvertebrates are not known. Indirectly, secondary production should increase with increased primary production and available food sources. For example, deBruyn et al. (2003) found a five-fold increase in macroinvertebrates downstream of a sewage treatment plant effluent in the St. Lawrence River, Montreal. Very little is known to link macroinvertebrates to the quality of food in rivers. Whiles and Dodds (2002) attempted to link suspended sediments to macroinvertebrate biomass in rivers, but a strong pattern could not be established.

Little is known about the influence of increased nutrient loading on vertebrate riverine biota such as fish, amphibians, reptiles, and birds. Miltner and Rankin (1998) analyzed data across 1657 sites in Ohio and found a deleterious effect on fish communities when TN and TP levels exceeded natural background levels in lower order streams, but found no effect in larger rivers. Fish communities depend on a natural nutrient balance within the system, but not much is known about the response of fish communities in large rivers to the cascade of effects caused by an imbalance in nutrients. Any effect of enrichment is most likely indirect and probably unpredictable. The exception is that low dissolved oxygen excursions (hypoxia) can occur in stagnant, nutrient-rich rivers, and these conditions have obvious negative effects on fish.

2.2 Ecosystem Function

Eutrophication of rivers can influence both heterotrophs and autotrophs (Dodds 2006). Major ecosystem functional responses to eutrophication should stem from changes in the structural and functional roles of microbial assemblages and primary producers within the system. However, not much is known about the effects of elevated inorganic nutrient loads on heterotrophic communities except that decomposition of litter in small streams is stimulated by higher nutrients (Dodds 2006). Autotrophic responses are better studied.

General responses of primary production in rivers should be less than in smaller streams because of increased turbidity and light limitation. However, there is no reason to believe that the underlying physiological responses of primary producers to increased nutrients, such as increased biomass and altered community structure, seen in smaller streams should not hold true in rivers. For example, when Dodds et al. (1997) plotted the relationships between nutrients and benthic algal biomass for the Clark Fork River, they did not differ substantially from those derived from many studies of small streams. When environmental conditions are right, primary producer responses can cause whole river effects. For example, more primary production and increased heterotrophic

microbial activity creates greater diurnal oxygen swings that could lead to areas of hypoxia within the river, such as those observed in the Murray-Darling (Maier et al. 2001), Clark Fork River (Dodds et al. 1997), Tualatin River (http://www.waterencyclopedia.com/Mi-Oc/Nutrients-in-Lakes-and-Streams.html), and others. Hypoxia typically occurs during periods of very low discharge or in rivers with limited flushing rates.

Changing biomass and composition of primary producers can alter lotic food webs and the availability of resources to higher trophic levels. Phytoplankton production may supply a significant portion of productivity in medium to large rivers that are not highly turbid and do not completely mix because they have zones of limited water replacement (e.g., Thorp et al. 1998, Wehr and Descy 1998). How increased production cascades up the food web is not well characterized for many rivers.

Structural changes in lotic biota may alter ecosystem function, such as nutrient cycling. However, direct measurements of nutrient and metal cycling are also rare for larger rivers. It is not known if or how increased nutrient loads would change cycling rates, such as nitrification and denitrification, though information published for small streams suggests functional effects, such as decreased proportional retention with increased nutrient loading.

Rivers also function to transmit nutrients to downstream habitats. Turner et al. (2003) estimated river discharge into oceans for major algal nutrients. These values are 16.2, 21, 2.6, 3.7 to 5.6, and 194 Tg yr^{-1} for NO_3-N, total nitrogen (TN), dissolved inorganic phosphorus (DIP), total phosphorus (TP), and dissolved silica (DSi), respectively. If nutrient retention becomes saturated and channelization lowers the ability of rivers to retain nutrients (Bernot and Dodds 2005), then the potential for transmitting eutrophication downstream increases. Increased transport of total nutrients to impoundments, lakes, and coastal waters, would likely lead to hypoxia. The nutrient ratios in the discharge of the world's 10 largest rivers approach the Redfield ratio, which may increase coastal phytoplankton production and occurrence of harmful algal blooms due to a shift to N or Si limitation (Justic et al.1993, Justic et al. 1995).

In conclusion, river food webs and water quality are potentially negatively impacted by eutrophication. While effects of increased nutrients on rivers are not as clearly defined and as well studied as they are in small streams or lakes, several aspects of river ecosystem structure and function may be altered by increases in nutrients. These include changes in community composition and increases in primary production and biomass. Changes in ecosystem structure and function can lead to problems with biological integrity, water quality, and the value of large rivers for recreation.

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