Effects of dams on riverine biogeochemical cycling and ecology

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Effects of dams on riverine biogeochemical cycling and ecology

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ABSTRACT
Currently, dam construction is a main and growing global anthropogenic disturbance on rivers. Dams have major effects on the physics, chemistry, and biology of the original river, including altering water circulation and retention time, sedimentation, nutrient biogeochemical cycling (especially greenhouse gas emissions), and the amount and composition of the organisms present. Among those, the effect of dams on the riverine material cycle and ecology is especially concerning because of its close relationship with current global environmental problems such as climate change and ecological deterioration. This review thus mainly focuses on nutrient cycling and ecological changes in a regulated river. In the future, research on reservoir–river systems should focus on (1) processes and mechanisms of nutrient biogeochemical cycles, (2) interaction between these processes and ecological change such as phytoplankton succession, and (3) developing mathematical functions and models to describe and forecast these processes and their future interactions.

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Introduction

Rivers are the major links connecting the land to the ocean; they deliver fresh water, carbon, energy, and nutrients to estuaries and coastal seas (e.g., Humborg et al. 1997, Jiao et al. 2007). Rivers also link the land with the atmosphere, exchanging heat, influencing the regional climate, and exchanging gases, affecting global biogeochemical cycles and the global climate (Lauerwald et al. 2015). In past decades, with the increasing demands of a growing human population, the natural river has been strongly disturbed by dam construction to generate hydropower, increase water supply and security, control floods, improve navigation, and provide opportunities for recreation (Bednarek 2001). Thus, the natural connectance among land, rivers, and oceans has declined, and material cycling has been affected with important consequences for the biology of the altered ecosystems.

Until the 1970s, however, the environmental impacts of dams were not widely taken into account. In 1972, the Scientific Committee on Problems of the Environment issued a report of man-made lakes as modified ecosystems (SCOPE 1972), which showed earlier concerns about the physical, chemical, and biological impacts of dams on the downstream rivers. In 2000, the World Commission on Dams presented another report, Dams and Development: a Framework for Decision-making (Report of the World Commission on Dams 2000). In 2010, the International Hydropower Association proposed the Hydropower Sustainability Assessment Protocol. Gradually, the effect of dams on rivers and their connected ecosystems has become widely studied (IHA 2010). The Millennium Ecosystem Assessment noted the dramatic increase in dam construction and consequent water storage, to the extent that flows in 60% of the World's large rivers are moderately or strongly affected, and also noted some of the negative consequences to ecosystems (Millennium Ecosystem Assessment 2005).

Generally, dam construction has 3 major consequences: (1) altered river hydrological cycle, exacerbated by artificial regulation such as anti-seasonal storage, which means reservoirs maintain low water levels during rainy seasons for the control of flood peaks but high levels during low water periods for water storage; (2) altered biogeochemical cycles in the impounded river; and (3) altered ecological conditions in the discontinuous river–reservoir system. These processes interact with each other, and their influences can have local, regional, and global effects. Understanding these processes is the scientific basis for understanding the environmental impacts of dam construction and providing sustainable management strategies for the impounded river. This brief review mainly focuses on these processes and, in addition, discusses current "hot" topics about the impounded river.

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Historical and current states of river damming

Modern dam construction began in 1900 and boomed from about 1950 with the use of concrete and innovation in excavation (Fig. 1). Currently, ~70% of the world's rivers are intercepted by dams (Kummu and Varis 2007), and in China, >80 000 reservoirs were constructed by the end of 2008, among which were >5000 dams higher than 30 m (http://www.chincold.org.cn). Dams are built to store water for various purposes. Accompanied with the rapid increase of dam construction (from 1948 to 2010), the global active storage capacity of reservoirs grew from about 200 to >5000 km^3, >70% of the total global reservoir capacity (7000–8000 km^3; Vörösmarty 1997, Zhou et al. 2016). The number of reservoirs will increase in the future with the restart of the hydropower loan project by the World Bank (World Bank 2009) and the motivation to increase renewable energy sources (Hermoso 2017).

Globally, the extent of hydropower development is not balanced. In Europe, North America, and Central America, >70% of the technically feasible hydropower has been utilized, while this value is <4% in Africa (Wang and Dong 2003, Home 2005). The developed countries have a higher level of hydropower utilization than the developing countries. For example, in China, only ~24% of the hydropower resource has been exploited, much less than the average value of 60% in developed countries. China also has large regional differences; eastern China has exploited 79.6% of its hydropower resources, but southwestern China, which has the richest hydropower resources, has only exploited 8.5% (Liu et al. 2009). Recently, with the numerous proposals to increase dam construction, the Amazon basin has become a hot area of hydropower development. A Dam Environmental Vulnerability Index was consequently introduced based on (1) the vulnerability of the basin to run-off and erosion that could transport nutrients and pollutants to the river, (2) modification to the hydrological regime

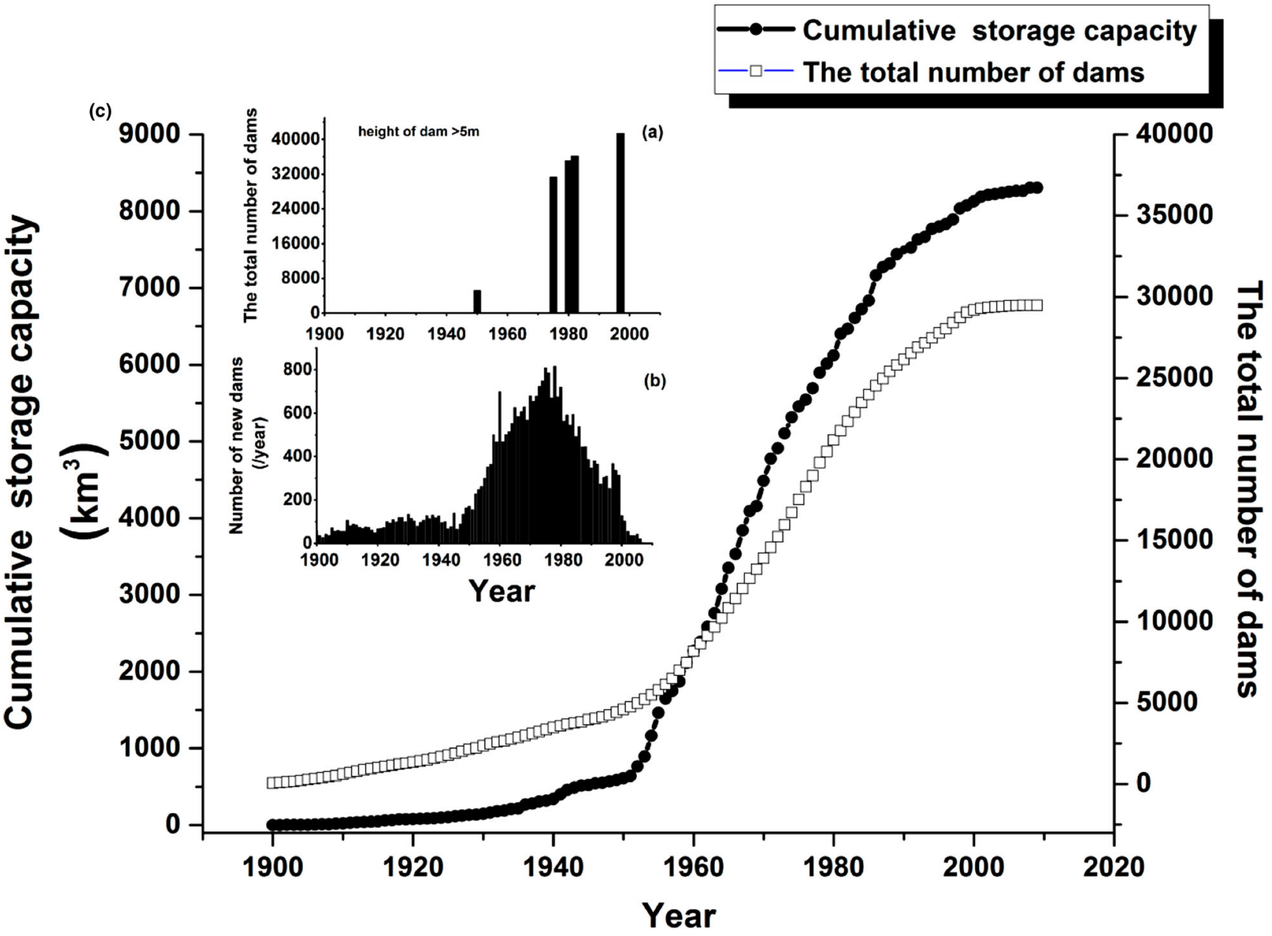

Figure 1. Historical variation in the number and cumulative storage capacity of reservoirs (modified from Chao et al. 2008, Jia et al. 2010). (a) Number of dams in the world with a dam height >5 m (Liu et al. 2009, Jia et al. 2010); (b) number of new dams constructed in the world each year; and (c) total number of dams and the cumulative storage capacity of reservoirs registered in the International Commission On Large Dams (ICOLD; Chao et al. 2008).

and transport of sediment, and (3) quantification of the extent of the river system affected (Latrubesse et al. 2017). With the development of the global economy, especially in developing countries, more dam construction is expected, further intensifying human disturbance on the rivers (Zarfl et al. 2015, Hermoso 2017).

Impacts of river damming on the hydrological cycle and physical characteristics

Seasonal thermal stratification

Reservoir stratification conforms to the classic pattern of lake stratification, especially in hydroelectric reservoirs that are usually deep and thus usually develop seasonal thermal stratification. The densimetric Froude number (F) has been suggested to estimate the stratification tendencies in a reservoir (Ledec and Quintero 2003). Stratification is expected when $F < 1$, the severity of which increases with a smaller F; when $F > 1$, stratification is unlikely. For hydroelectric reservoirs, the extent of thermal stratification is influenced by the pattern and extent of water storage and discharge (Fig. 2). One consequence of thermal stratification is that if water is released from the bottom of the reservoir during the period of stratification, water downstream of the dam will greatly differ from that at the reservoir surface, with potential effects on the downstream river for tens of kilometres (Petts1984). This problem could possibly be eliminated by artificially destroying thermal stratification (Lackey 1972, Elçi 2008) or by releasing water from the surface or subsurface.

A reservoir can also affect the downstream river temperature, with consequences for biogeochemical cycling, river ecology, and particularly fish populations. In general, water released from the bottom of a reservoir will be cooler than it would be without the reservoir while water released from the surface of the reservoir will be warmer. In addition, however, reservoirs also dampen the temperature cycle at seasonal, daily, and sub-daily timescales and, in a study of Canadian reservoirs, increased the mean water temperature in September (Maheu et al. 2016).

Storage pattern

Artificial regulation of reservoir water, such as storing or releasing water, changes the flood pulse of the original river, affecting the water balance of the basin and the hydrological condition of the river bank (Liu et al. 2009). After interception by a dam, a significant reduction in the maximum flow occurs in the downstream river (Fig. 3a). In addition, large- and medium-sized reservoirs often have anti-seasonal storage to reduce the reservoir water level in the flood season to cope with flood peaks (e.g., Fig. 3b). This phenomenon is different from a lake, where water level changes correspond to the runoff input minus evaporation and outflow. When cascade reservoirs are constructed, competitive water storage among the reservoirs will occur,

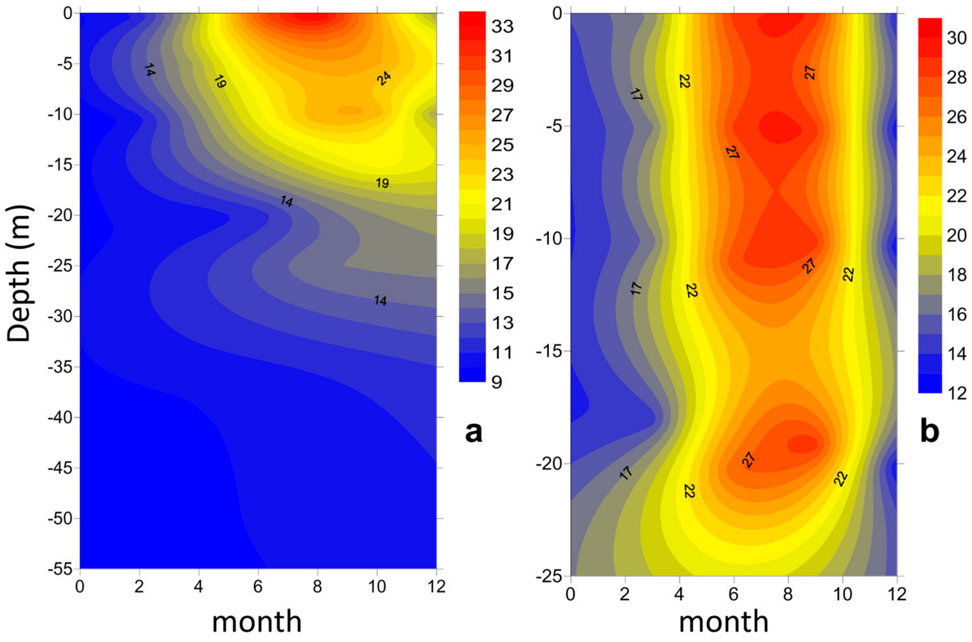

Figure 2. Thermal stratification (water temperature, °C) in selected reservoirs (unpubl. data). (a) Xinanjiang Reservoir with a water retention time >1 year; (b) Wanan Reservoir with a water retention time <2 weeks.

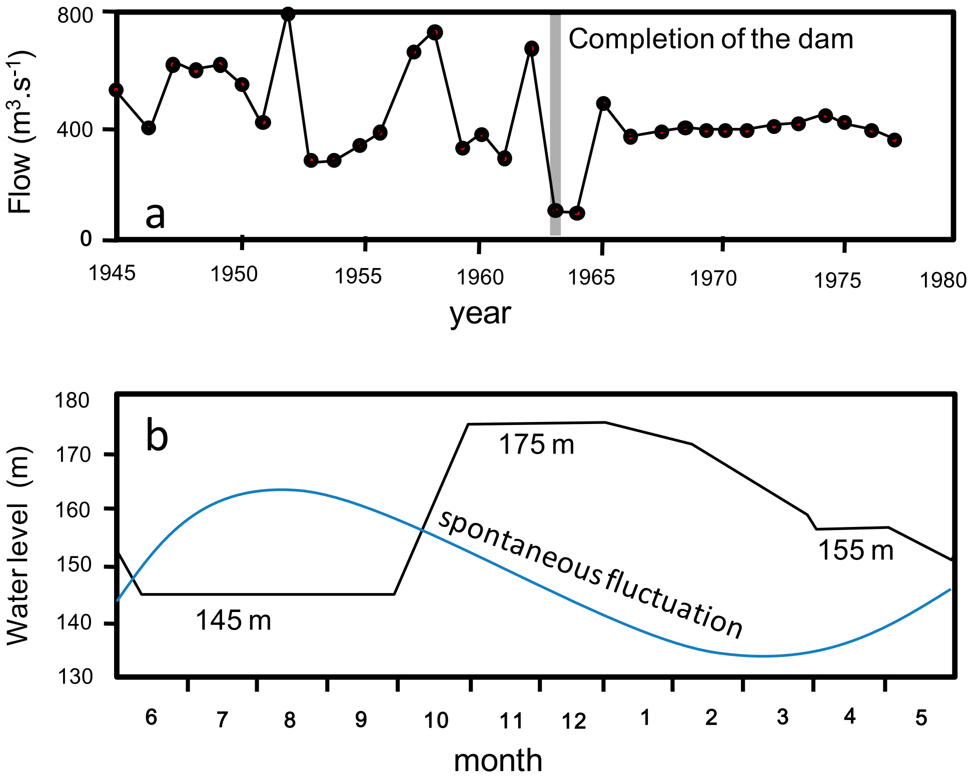

Figure 3. Changes in river flow and water level after river damming. (a) Effects of an upstream reservoir (Lake Powell) on the maximum annual flow of downstream (modified from Chapman 1996); (b) water level change after the Three Gorge Reservoir closure.

reducing water availability in the downstream reaches of the river.

Hydrological retention time

Dam construction obviously changes the retention time of the corresponding river. The retention time within one reservoir can vary between 1 day and several years, greatly prolonging that of the natural river. For continental runoff in free-running river channels with no river damming, the average residence time varies from <16 to 26 d; however, the discharge-weighted global average value is almost 60 d when taking river impounding into account (Vörösmarty and Sahagian 2000). In some strongly regulated river basins, the value can be higher. For example, the water retention time of the Yellow River (upstream of Lijin station; i.e., the whole basin taking into account the 2816 reservoirs), increased from 1year to 4 years after dam construction, ranking the Yellow River in the top 3 in terms of residence time and flow regulation among large river systems in the world (Ran and Lu 2012). The increase of water retention time has a profound effect on the reservoir thermal stratification, riverine elemental cycle, and phytoplankton ecology. For example, with increasing hydraulic retention time, thermal stratification of a waterbody is more likely to occur, and the retention rate of phosphorus in the reservoir is also increased (Duras and Hejzlar 2001). The retention time also has a direct influence on the spatial heterogeneity of the reservoir phytoplankton (Soares et al. 2012). Recent studies show a significant negative exponential relationship between the carbon dioxide (CO_2) release flux and the retention time of reservoirs (Wang et al. 2015).

Effect of dams on riverine material cycling

Sediment transport

Sedimentation within reservoirs is a complex process. The sediment load delivered to the reservoir is controlled by the sediment yield in a basin, and reservoir sedimentation is mainly influenced by the hydraulics of the river, the geometry of the reservoir, and ratio at the entrance to reservoir (width to depth ratio). In addition to sedimentation of material produced in the catchment, stimulation of phytoplankton biomass within the reservoir (discussed later) also produces particulate material that is stored. The distribution of sediment types in a reservoir is shown in Table 1. The reduction in downstream sediment load by reservoir construction may be >75%, as seen in the case studies of Sao Francisco River in Brazil, the Chao Phraya River in Thailand, and the Yellow River in China (Walling 2006). Kummu and Varis

Table 1. A typical distribution of deposited sediment in a reservoir (USACE 1987).

Particle size	Inlet (%)	Mid-reservoir (%)	Outlet (%)
Sand	5	<1	0
Silt	76	61	51
clay	19	38	49

(2007) reported that the operation of dams on the Mekong main channel had approximately halved the sedimentation from 150–170 × 10^9 to 81 × 10^9 kg annually. The magnitude of global suspended sediment flux to the ocean is still unclear but estimated to be in the range of 9.3 Gt yr^{-1} (Judson 1968) to >58 Gt yr^{-1} (Holeman 1968), with recent studies converging around 15–20 Gt yr^{-1} (e.g., Milliman and Meade 1983, Meybeck 1988, Ludwig et al. 1996, Vörösmarty et al. 2003, Walling 2006).

The great change in fluxes of river suspended sediment into the sea in recent decades reflect the strong anthropogenic activities in the basin. An artificial increase in soil erosion has occurred, and also a decrease in sediment flux caused by soil and water conservation and dam interception during the past decades (Walling 2006, Wang et al. 2016a). Among them, reservoir retention is considered to be the main reason for the reduction of suspended sediment flux in global rivers. Syvitski et al. (2005) estimated the accumulation of sediment in global reservoirs over the past 50 years as 100 billion metric tons (Syvitski et al. 2005). The retention of suspended sediments in the reservoirs reduces the reservoir capacity, which negatively affects the ecological environment of the downstream rivers. For example, the lack of sediment in the downstream river leads to habitat reduction, which can increase erosion in the downstream riverbed and estuary.

Major biogeochemical processes in reservoir

Dam construction changes the ecosystem from a "river type" heterotrophic system dominated by benthic biota to a "lake type" autotrophic system based on plankton (Saito et al. 2001). The development of this process is mainly controlled by the retention time and nutrient level of a reservoir. As an autotrophic system develops, the photosynthetic carbon sequestration capacity of the reservoir is strengthened, and dissolved inorganic nutrients are converted to particular material that can be retained. A study on a large reservoir confirmed that the biogeochemical cycle of carbon was enhanced after the river was impounded (Han et al. 2018). Because of the nonconservative geochemical behaviour of carbon (C), nitrogen (N), silicon (Si), and phosphorus (P) in a river, the complex biogeochemical processes in a reservoir significantly change their fluvial fluxes and forms (Fig. 4; Jossette et al. 1999, Ittekkot et al. 2000, Kelly 2001, Hungspreugs et al. 2002, Koszelnik and Tomaszek 2008). Terrestrial organic matter carried by river is usually recalcitrant and not rapidly degraded. Most of

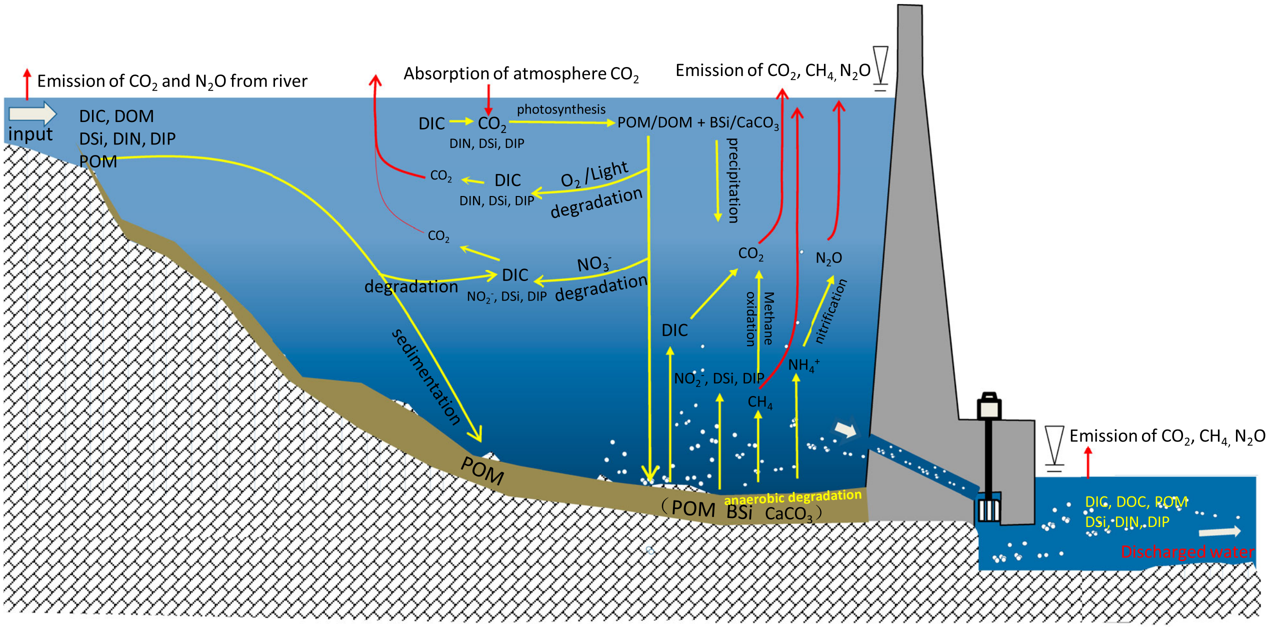

Figure 4. Major biogeochemical processes in a typical reservoir. Red lines refer to the GHGs emission. Yellow lines are the main geochemical processes in reservoirs (e.g., degradation, nitrification, sedimentation, methane oxidation). DIC = dissolved inorganic carbon; DOM = dissolved organic matter; DSi = dissolved silicon; DIN = dissolved inorganic nitrogen; DIP = dissolved inorganic phosphorus; BSi = biogenic silicon; POM = particulate organic matter.

this organic matter is buried in the reservoir sediments, and only a small part is decomposed within the reservoir. By contrast, organic matter newly formed in a reservoir by photosynthesis can decompose rapidly during settlement, returning inorganic C, N, P, and other resources to the waterbody (Fig. 4). Material that decomposes within a few centimetres of the surface of the sediment can lead to denitrification, methanogenesis, and the generation of greenhouse gases (GHGs) such as CO_2, methane (CH_4), and nitrous oxide (N_2O; Fig. 4). According to the research results of Wang et al. (2012), 70% of the newly deposited organic carbon (OC) is decomposed within 12 years, and the remaining part will be retained in sediment in the form of refractory OC (Wang et al. 2012). In addition, unlike natural lakes, artificial reservoirs usually use the bottom water release mode for hydrologic regulation and hydropower generation; therefore, the reducing substances and GHGs in the bottom water of the reservoir are released downstream.

Nutrient retention

The complicated geochemical process within a reservoir and deep water release from the dam greatly change the nutrient speciation and fluxesnts in a river. These modified nutrients are then transported to the downstream river and, finally, influence the ecosystem of the estuary and marginal sea (e.g., Humborg et al. 1997, Jiao et al. 2007).

The first concern on the consequences of dams on nutrient retention was about Si. Because dissolved Si is an essencial element for the growth of diatoms, the lack of Si in water and the imbalance in the proportion of Si, N, and P will lead to the succession of aquatic primary producers from diatom to non-diatom. Humborg et al. (1997) found that after the completion of the Iron Gate Reservoir in the late 60s of last century, the flux of Si transported by The Danube was greatly reduced. They estimated that about 600 kton of dissolved Si was retained in the Iron Gate Reservoir every year (Humborg et al. 1997). The large amount of Si retention has caused a rapid succession of diatoms to non-diatoms in the coastal waters of the Black Sea and significantly changed the structure and function of the marine ecosystem. Friedl et al. (2004) recalculated the continuous mass balance data and found that the Iron Gate dams only intercepted 4% of the dissolved Si. Based on this estimate, they noted that most Si retention may occur in the newly built reservoirs in the upper reaches of the Danube, not in the Iron Gate (Friedl et al. 2004). Undoubtedly, dam construction results in a significant Si retention and decrease in the flux of dissolved Si (DSi) to the sea (Humborg et al. 1997, 2006, Wang et al. 2010, Maavara et al. 2014), with potential consequences for coastal primary productivity (Gong et al. 2006). At the global scale, the retention of DSi (as SiO_2) in lakes and reservoirs is 163 Gmol yr^{-1} (9.8 Tg yr^{-1}), and the total active Si retained is 372 Gmol yr^{-1} (22.3 Tg yr^{-1}; Maavara et al. 2014).

In freshwater sysytems, P is often a major limiting factor for primary productivity and is easily adsorbed by suspended sediment, which leads to a higer retention rate of P in reservoirs. The long-term accumulation of P in reservoirs potentially risks eutrophication if it is released to the water, as can occur during sediment–surface anoxia (Wang et al. 2016b); however, P retention also reduces the nutrient transport of P downstream. For example, because the 2 dams were built in the upper reaches of the Kootena Lake in Canada, the P input to the lake has declined sharply, directly reducing plankton abundance and the productivity of fisheries on the lake (Friedl and Wüest 2002). Although the P retention rate varies greatly among reservoirs, a reservoir generally is still a P sink. Recently, Maavara et al. (2015, 2017) estimated the global P and OC retention by river damming. Total P (TP) trapped in the global reservoirs was estimated as 22 Gmol yr^{-1} in 1970 and 42 Gmol yr^{-1} in 2000, and retention of reactive P was 9 Gmol yr^{-1} in 1970 and 18 Gmol yr^{-1} in 2000; however, the global TP loading to rivers had changed in the same period, but only from 312 to 349 Gmol yr^{-1}. Consequently, the rapid increase of TP and reactive P retention was mainly caused by the rapid expansion of dam construction between 1970 and 2000 (Maavara et al. 2015), and the volume of reservoirs increased from about 3000 km^3 in 1970 to almost 6000 km^3 in 2000 (Lehner et al. 2011). By 2030, about 17% of the global river TP load is forecast to be sequestered in reservoir sediments, and the main increase will be from Asia and South America, especially in the Yangtze, Mekong, and Amazon drainage basins (Maavara et al. 2015).

The global mineralization of OC in reservoirs exceeds C fixation, and about 75% of OC in reservoir sediments is allochthonous. OC burial in reservoirs is forecast to be about 4.3 Tmol yr^{-1} by 2030, a 4-fold increase relative to 1970. The net mineralization fluxes (OC mineralization [R] minus primary production [P]) decreased from 2.6 Tmol yr^{-1} in 1970 to 1.3 Tmol yr^{-1} by 2000 and is forecast to increase to 2.2 Tmol yr^{-1} by 2030 because of the many new dams planned for the 21st centrury. Further estimates show that in-reservoir burial plus mineralization eliminates 6.9 Tmol yr^{-1} of OC, accounting for ~19% of total OC carried by rivers to the oceans by 2030 (Maavara et al. 2017). Comparatively little is known about N retention in impounded rivers, perhaps because of the more complex N cycle in the reservoir–

river system. A case study in a regulated Mediterranean river indicated that the river course below a dam acted as net sinks of total dissolved N, and this high net uptake by organisms (autotrophs and heterotrophs) below dams could reduce N export to downstream ecosystems (von Schiller et al. 2016).

Greenhouse gas emissions

Reservoir GHGs based on C, CO_2, and CH_4 are derived from OC mineralized in the reservoir or direct input of CO_2 produced in the catchment (Maberly et al. 2013), and their emission occurs by diffusion across the air–water interface and, especially for CH_4, ebullition. GHG production and emission fluxes from a reservoir are closely related to reservoir age, latitude, and retention time (Barros et al. 2011, Ometto et al. 2013, Wang et al. 2015, Deemer et al. 2016). The reservoir surface is usually dominated by the diffusive flux of CO_2, even when bottom anoxia leads to high CH_4 production because of conversion of upwardly diffusing CH_4 to CO_2 by methanotrophic bacteria. When water is released from the bottom of the dam, however, CH_4 emissions can be very high. This source possibly contributes 50–90% of total CH_4 emissions from tropical or temperate hydroelectric reservoirs (Abril et al. 2005, Kemenes et al. 2007, Maeck et al. 2013).

Rates of surface diffusion of GHGs among reservoirs vary along broad geographic gradients, and low-latitude tropical reservoirs typically emit GHGs at greater rates per unit area than high-latitude temperate and boreal reservoirs (Barros et al. 2011). Barros et al. (2011) ascribed this latitudinal pattern of emission to water temperature and higher flooded biomass in tropical regions, which favours the production of GHGs. Average emissions of 3500 mg m^{-2} d^{-1} of CO_2 and 300 mg m^{-2} d^{-1} of CH_4 have been found in tropical reservoirs compared to CO_2 values of 387~1400 mg m^{-2} d^{-1} and CH_4 values of 2.8~55 mg m^{-2} d^{-1} from temperature reservoirs (mostly hydroelectric reservoirs; St. Louis et al. 2000, Soumis et al. 2005, Lima et al. 2008, Barros et al. 2011). Chanudet et al. (2011) estimated diffusive fluxes to the atmosphere from 2 Southeast Asian subtropical reservoirs to be −466~1680 mg m^{-2} d^{-1} for CO_2 and 12.8~190 mg m^{-2} d^{-1} for CH_4, comparable to other tropical reservoirs. Few studies have focused on reservoir N_2O emission. A case study in the Wujiang cascade reservoirs showed that the average flux of N_2O emission from the reservoir surface was about 0.45~0.64 μmol m^{-2} h^{-1} (Liu et al. 2011), similar to a natural lake, for example Lake Taihu in China (0.41–0.58 μmol m^{-2} h^{-1}; Wang et al. 2009) and Lake Kevaton in Finland (0.09–0.50 μmol m^{-2} h^{-1}; Huttunen et al. 2003).

Lima et al. (2008) first estimated CH_4-C emission from the global hydroelectric reservoirs as 100 Tg yr^{-1}; however, Barros et al. (2011) suggested this value was 3 Tg yr^{-1}, with CO_2-C emission of 48 Tg yr^{-1} in the hydroelectric reservoirs, while the values from Hertwich (2013) were 76 and 7.3 Tg yr^{-1} for CO_2-C and CH_4-C, respectively. The large range in published estimates could be caused by the different estimates of global reservoir surface (Mendonça et al. 2012, Teodoru et al. 2012, Mosher et al. 2015), and possibly high spatial heterogeneity of CO_2 and CH_4 fluxes along long and narrow reservoirs.

Effect of dams on riverine ecology

River continuum concept

The river continuum concept describes the longitudinal gradient of physical conditions such as geomorphological and hydrological factors in pristine rivers (Vannote et al. 1980, Tornwall et al. 2015). Biological communities are adapted to these gradients and vary predictably along the river from the headwaters to the mouth. The headwater regime is strongly heterotrophic (the ratio of photosynthesis to respiration [P/R] < 1) and has coarse particular matter and invertivores as the main biological species. The mid-regime is autotrophic ($P/R > 1$) and has fine particular matter and piscivorous, invertivorous, and planktivorous species. Finally, the downstream regime gradually returns to heterotrophy due to turbidity (Fisher 1977, Vannote et al. 1980). Dams interrupt the river continuum altering geomorphology, water quality, temperature regime, and flow regime, and result in upstream–downstream shifts in biotic and abiotic patterns and processes. The serial discontinuity concept views impoundments as major disruptions to longitudinal resource gradients along river courses (Ward and Stanford 1983, 1995). The impacts under impoundment have been studied with respect to geomorphology, temperature, flow, invertebrates, fish, and other factors (e.g., Kondolf 1997, Jakob et al. 2003, Poff and Zimmerman 2010, Jones 2011, Winemiller et al. 2016). For example, in regulated rivers the deviation in flow from the natural regime may shorten food chains and reduce aquatic biodiversity, thus altering the structure and function of rivers (Power et al. 1996, Wu et al. 2004). Periphyton biomass recovers quickly from the disturbance cause by a reservoir, usually within 5 km downstream, while benthic invertebrate richness varies considerably, with both increases and reductions observed at near-dam sites and varying in recovery downstream (Ellis and Jones 2013, 2016). Therefore, the aquatic ecology of dammed rivers is unpredictable. Below we highlight the effects of damming a river on the phytoplankton and fish communities.

Phytoplankton

As the main primary producers, river phytoplankton succession after damming is an important issue. Dams can cause major changes in the phytoplankton community in the river, estuary, and adjacent sea (e.g., Humborg et al. 1997, Jiao et al. 2007). In a river–reservoir unit, the dominant Bacillariophyta (diatoms) in rivers change to coexisting Bacillariophyta, Chlorophyta (green algae), and Cyanophyta (blue-green algae) in mesotrophic reservoirs or shift to dominance by Cyanophyta in eutrophic reservoirs (Wang et al. 2013). Phytoplankton succession in the impounded river is not directly caused by the physical obstruction of the dam but can be attributed to changes in hydrological and geochemical conditions after damming. For example, phytoplankton community succession in karst cascading reservoirs was influenced by Si and P stoichiometry (Wang et al. 2014a), whereas in a tributary of the Three Gorges Reservoir, phytoplankton diversity was controlled by hydraulic retention time and nutrient limitation (Xiao et al. 2016). Phytoplankton dynamics in the impounded river is nonlinear, and the mechanisms responsible need further research. Bacteria, heterotrophic nanoflagellates, ciliates, and zooplankton usually show similar longitudinal variation to phytoplankton from river inflow to the dam reservoir, and in addition to retention time and temperature, bacterial community composition can be affected by allochthonous or autochthonous input of OC (Simek et al. 2008, 2011). Generally, the study of the effect of dams on riverine ecology is still at an initial stage, especially from the aspect of the coupling of ecological shifts and nutrient biogeochemical cycle.

Fish

The effect of dams on fish ecology (e.g., spawning, migration, and diversity) has long been a concern (e.g., Bonner and Wilde 2000, Ziv et al. 2012, Winemiller et al. 2016). Regionally, the construction of the Three Gorges Reservoir, for example, resulted in a substantial decline in carp larval abundance of the middle Yangtze River (Wang et al. 2014b). For example, the Three Gorges Dam and Gezhouba Dam on the Yangtzi River threaten the Chinese sturgeon (*Acipenser sinensis*) by interrupting its migratory pathway to upstream spawning grounds (Xie 2003). In a river–reservoir unit, the lotic fish species prefer the fluvial zone, the running-water fish species prefer the transition zone, and lentic fish species prefer the lentic zone. In addition, a shift from lotic to lentic habitat induced by damming could decrease lotic aboriginal fish species and increase exotic fish species (Fan et al. 2015). Globally, dam obstruction decreases fish biodiversity, and catadromous or anadromous taxa such as lampreys (*Lampetra* spp.), eels (*Anguilla* spp.), and shads (*Alosa* spp.) are at particular risk of species loss (Fu et al. 2003, Liermann et al. 2012). In addition, different types of hydropower turbines have different effects on fish; Kaplan horizontal bulb turbines are reportedly less deleterious than vertical axis turbines typically used in accumulation reservoirs (Cella-Ribeiro et al. 2017).

Concluding remark

The construction of dams for various purposes, but particularly hydropower, is booming (Zarfl et al. 2015) and likely to accelerate. Given the complex interactions among the land, rivers, estuaries, and the atmosphere, the consequences of dam building will inevitably have complex knock-on, ecological, and social effects. International initiatives such as the Paris Agreement reached at the 21st Conference of Parties (COP21; also called the 2015 Paris Climate Conference) in December 2015 are encouraging countries to move toward a greater reliance on renewable energy production. Hydropower currently accounts for >80% of renewable energy (Zarfl et al. 2015). Hermoso (2017) noted that while this might prove beneficial for global C emissions, it would likely prove detrimental to local freshwater ecosystems; consequently, a requirement for international guidance and legislation has been implemented to evaluate the benefits of new dam construction compared to the ecological and societal costs. Clearly, the need for robust scientific evidence for these evaluations is urgent and growing. Currently, although numerous case studies exist at specific sites, it is difficult account for regional heterogeneity in conditions to produce advice at a global scale. In the role of reservoirs in global biogeochemical cycling, future research should focus on the following aspects: (1) investigating processes and mechanisms of nutrient biogeochemical cycles; (2) coupling these cycles with ecological conditions, such as phytoplankton succession; (3) developing mathematical functions and models to describe and forecast these processes and their interaction with local, regional, and global factors; and (4) producing strategies and measures to mitigate negative ecological effects of a dam on a river ecosystem.

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