

Review Article

Disrupting biogeochemical cycles – Consequences of damming

Gabriela Friedl^{1,*} and Alfred Wüest²

¹ Swiss Federal Institute of Technology (ETH), FZL, CH-6047 Kastanienbaum, Switzerland

² Swiss Federal Institute of Environmental Science and Technology (EAWAG), FZL, CH-6047 Kastanienbaum, Switzerland

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Abstract. Sustainable management of natural water resources should include environmentally sound dam construction and operation with respect to both upstream and downstream management. Because of slowly evolving alterations in riverine ecosystems following the construction of a dam – due to the sometimes large distances between dams and affected areas, and the interference with other anthropogenic activities – some of the effects of damming may be overlooked. Constructing reservoirs modifies the biogeochemical cycles, such as interrupting the flow of organic carbon, changing the nutrient balance,

and altering oxygen and thermal conditions. The consequences of altered processes may not be immediately apparent and may become obvious only after a long period of time or only in combination with other anthropogenic alterations. It is difficult to give precise predictions of the impacts of a particular dam due to the complexity and individuality of aquatic ecosystems. However, this remains the challenge while planning and constructing new dams. Protecting and restoring river basins has been called for by the World Commission of Dams (WCD).

Key words. Dams; reservoirs; nutrients; nutrient ratios; oxygen depletion; downstream effects.

Introduction

Damming of rivers and pre-existing lakes has a global impact on natural water resources. Almost 800'000 artificial lakes and reservoirs (covering an area almost 500'000 km²; $\approx 3\%$ of the land surface) have been built (Gleick, 1999; McCully, 1996; Gleick, 1993) in order to provide services such as drinking water supplies, hydro-power production, irrigation and flood control. On a global scale, river damming increases the average residence time of river waters by a factor of 3, from 16 days to 47 days (Covich, 1993). The estimated storage volume in reservoirs worldwide is 8'400 km³, compared to a water

volume of 1'200 km³ in natural rivers. This translates in a $\approx 700\%$ increase in the “standing” stock of free flowing river water (Vörösmarty et al., 1997). In certain regions, such as Southeast Asia, man-made lakes and reservoirs comprise by far the greatest area of inland waters (Fernando, 1984).

Impoundments change the characteristics of a water body from “rivers” to “lakes”, affecting not only the hydrology but also physical, chemical, and biological characteristics. These changes include increases in residence time, temperature, stratification, and reduction in turbulence, most often a decrease in particles and turbidity and sometimes an increase of autochthonous primary production. In other cases, shoreline erosion or hydro-power-related discharge may lead to higher turbidity. Additionally, residual flow, hydro-peaking and seasonal changes in the hydrological regime will affect downstream rivers, lakes and reservoirs, their dynamics and

* Corresponding author phone: +41 41 349 2185;
fax: +41 41 823 5210; e-mail: gabriela.friedl@eawag.ch
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water quality. Installing reservoirs will disrupt the natural biogeochemical cycles of carbon, nutrients and metals and possibly affect the whole catchment including downstream ecosystems such as wetlands, estuaries, deltas and adjacent sea areas.

The World Commission on Dams (WCD) has recently summarized the major environmental impacts of large dams (WCD, 2000; Bergkamp et al., 2000). These impacts are mostly negative and are often complex and cumulative. While some changes caused by impoundments are obvious and not debatable, other changes are more subtle and might turn into an adverse effect only after a long period of time or in combination with other anthropogenic activities. Thus these effects are difficult to predict. While the WCD (2000) report provides an overview of all possible effects, the present article concentrates on the disruption of biogeochemical cycles. Since such changes may occur slowly and subtly some potentially detrimental effects may have been overlooked for different reasons (such as missing systematic data, neglecting down-stream effects etc).

Changing rivers into lakes

Reservoirs constitute an intermediate type of water body between rivers and natural lakes having both river-like and lake-like characteristics. The main features differing from a lake are, e.g., (i) the position of the outlet of a reservoir being often located at the deepest part of the water body, (ii) the timing and the variability of the discharge, (iii) the existence of longitudinal gradients in physical, chemical and biological features, (iv) the unnaturally-long shore lines due to dendritic shapes, and (v) often a broad drawdown zone due to high fluctuations in water level. The possibility to withdraw water selectively from different depth has implications on the mixing of the man-made water body. The altered and unnatural timing of discharge affects both the reservoir and the downstream part of the river.

When changing a stretch of a river into a reservoir the slowdown of the flow subsequently evokes particle settling, turbidity decreases and light transmissivity increases, enhancing in-situ primary production. Thus, from the headwater of the reservoir to the dam the river changes from an allochthonous dominated system to a more lacustrine system, where autochthonous production of organic matter dominates. The change from an allochthonous to an autochthonous regime alters the carbon (C), phosphorus (P), nitrogen (N) and silicon (Si) biogeochemical cycles. As each nutrient cycle responds differently, their ratios are also subject to changes (see below). The increase in residence time affects salt concentrations due to evaporation, temperature and oxygen concentration. Both, increased residence time and au-

tochthonous production of biomass will alter redox conditions and subsequently effect nutrient cycling and potential hazardous metal release from the sediments.

One of the main aspects of reservoirs is the slowdown of the flow velocity compared to the original river and the related settling of particles. If the particle's sinking velocity is larger than the advective upwelling, caused by the river inflow, the particle will settle out, while the water flows through the reservoir. The sinking velocity v_p (m s^{-1}) depends on the particle's size: According to

Stokes' relation: $v_p = \frac{\Delta\rho}{\rho} \frac{g}{18\nu} D_p^2$, where $\Delta\rho$ (kg m^{-3}) is

the density difference between particles and water, ρ is the density of water ($\approx 1000 \text{ kg m}^{-3}$), $g = 9.81 \text{ m s}^{-2}$ is the gravitational acceleration, ν ($\approx 1 - 1.5 \cdot 10^{-6} \text{ m}^2 \text{ s}^{-1}$) is the viscosity of water and D_p (m) is the particles diameter. The upwelling is caused by the river water, intruding into a reservoir with discharge Q ($\text{m}^3 \text{ s}^{-1}$) at depth z . As a result, the water layers above the intrusion level lift up with the velocity $w(z) = Q(z)/A(z)$ (where $A(z)$ is the lake area at depth z). If $v_p > w(z)$, the particle will settle. For a typical example, such as the Upper Arrow Reservoir (British Columbia, Canada, see map in Fig. 1), the discharge of the Columbia River (average $Q \approx 1190 \text{ m}^3 \text{ s}^{-1}$) and the reservoir surface area ($A \approx 330 \text{ km}^2$) leads to a vertical velocity of $w \approx 0.3 \text{ m per day}$. It follows from the equation

above that particles larger than $D_p > \left(\frac{\rho}{\Delta\rho} \frac{18\nu}{g} \frac{Q}{A} \right)^{1/2}$

will sink to the bottom of the reservoir. For inorganic particles, where $\Delta\rho$ ($\approx 1700 \text{ kg m}^{-3}$) is large, the limiting particle diameter is ≈ 1.8 microns (μm), whereas for organic particles, with $\Delta\rho$ ($\approx 20 \text{ kg m}^{-3}$) being small, the limit would be as large as ≈ 22 microns (μm). Among the inorganic particles, mainly the colloidal fraction remains suspended in the water. In contrast, even the largest organic particles are prevented from settling and can subsequently get washed out, especially if the reservoir outlet is at the base of the dam (Matzinger and Wüest, 2001).

The variation in discharge causes equivalent variation of the washout particle size. In reservoirs with primary production, discharge-related changes of the residence time affect the phytoplankton community because slow-growing algae get washed out, whereas fast-growing algae may withstand flood pulses with their high rate of reproduction (Straskraba et al., 1993). The biotic communities in lakes and reservoirs depends therefore strongly on the timing of the discharge (Komarkova et al., 1996).

The effect of oxygen depletion on changes in carbon cycling

A further potential consequence of the transition from a river to a lake is the depletion of oxygen. The upstream part of the reservoir receives the easily degradable

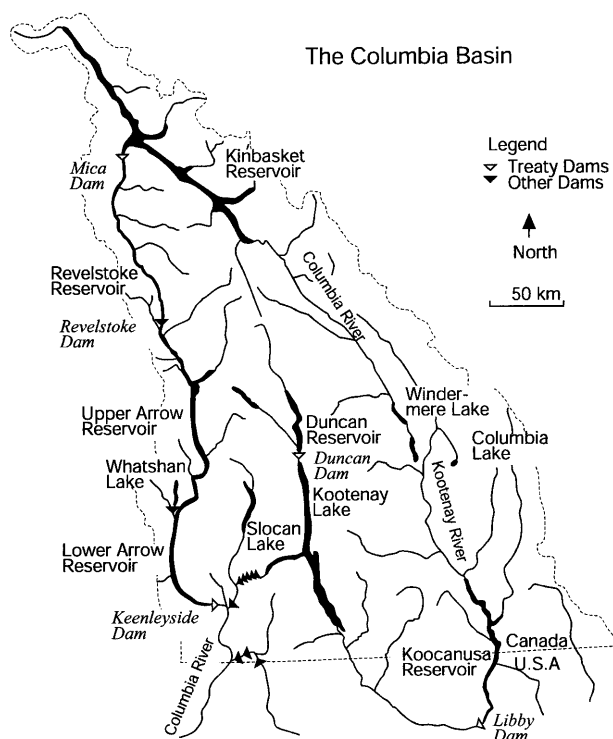


Figure 1. Watershed of the Canadian Columbia River. The triangles indicate dams (pointers directed in the flow direction). “Treaty dams” were built based on the 1963 Columbia River Treaty between USA and Canada. Kootenay Lake and Kinbasket Lake, as well as Upper and Lower Arrow Reservoir were previously lakes, which have been enlarged by damming. Revelstoke Reservoir, Kooconusa Reservoir and Duncan Reservoir are dammed rivers (adapted from Columbia Basin Trust home page: www.cbt.org/html/map.html).

organic matter of riverine origin, which is deposited on the sediment and mineralized. In the deeper part of the reservoir also *in-situ* produced particles (e.g. algae) settling to the sediment consume oxygen. Thermal stratification reduces the exchange between surface and deep water, eventually leading to anoxic conditions (Cooke et al., 1993). Low oxygen concentrations create problems within the reservoir (see below) as well as within the discharged downstream waters. In some countries, discharged water from reservoirs has to meet certain water quality standards. For example many US Federal Energy Regulatory Commission licenses now include minimum dissolved oxygen levels typically of 5 to 6 g m⁻³ in the released water (Mobley, 1997). If reservoirs are used for drinking water supply, the quality standards are even higher (Cook and Carlson, 1989). As a consequence, drinking water suppliers and hydropower generators must frequently add large quantities of (often pure) oxygen to the stored water prior to or during release to mitigate this environmental impact. Different such techniques in use are described by Mobley (1997) or Beutel and Horne (1999).

Within the reservoir, the depletion of oxygen triggers reduction of nitrate, manganese (hydr)oxides, iron (hydr)oxides and sulfate. Reduced products such as Mn(II), Fe(II), NH₄⁺ and H₂S can accumulate in the deep waters. In addition of being toxic to fishes and higher organism, these reduced compounds keep the sediment-water interface anoxic and thus preventing the build-up of iron hydroxide layers (Cook and Carlson, 1989). With the loss of iron to the overlying water, the sediment’s retention capacity for phosphorus decreases. As a result, the productivity of the system is enhanced. Under anoxic conditions, microbial methanogenesis and denitrification lead to the production and potential emission of the greenhouse gases methane (CH₄) and nitrous oxide (N₂O).

Reservoirs as source of methane

Anoxic conditions are often encountered in young reservoirs (Petts, 1984). Newly flooded soil and vegetation can act for a limited period of time as a source of nutrients, as the inundated vegetation begins to decompose and deplete oxygen. Under anoxic conditions the widely abundant biomass will be transformed mainly into carbon dioxide (CO₂) and methane (CH₄). During an initial period after flooding labile organic matter can decompose rapidly and therefore emissions can be huge. A decrease of emissions is expected only after several years, with a time span of 20 years for reservoirs in tropical forests (Galy-Lacaux et al., 1999; Fearnside, 1997) or even more for reservoirs in northern regions (Rudd et al., 1993). In several dammed reservoirs in Brazil the flooded forest continued to decompose over decades, emitting CO₂ and CH₄.

Little basis exists for calculating and predicting such emissions from reservoirs (Rosa et al., 1996; Fearnside, 1995; Rosa et al., 1995), and most of the available data are from boreal (Northern Canada and former USSR) or tropical (Brazil) climates (Lucotte et al., 1997; Rosa et al., 1997; Rosenberg et al., 1995). To fully assess the green house gas emissions caused by building a reservoir, it is essential to determine the emissions from various ecosystems in the watershed before the construction of a dam and after, in order to establish the net emissions. However, a meaningful assessment of the net emission is not an easy task. Firstly, uncertainties in measuring the gross emission arise from spatial differences within the reservoirs, seasonal variations and the formation and escape of CH₄ bubbles, which are difficult to quantify. Secondly, a number of processes have to be taken into consideration to determine net emissions reliably. Atmospheric CO₂ entering the system “reservoir” might be measured as gross emission and has to be deducted. Emission that would have occurred in a non-reservoir

situation, such as from preexisting lakes, soils, wet- and peatlands, and long-term storage in the sediment of the reservoir must also be quantified and subtracted from the gross emission. However, the creation of a reservoir might turn preexisting lakes anoxic and thus trigger the conversion of organic carbon into methane instead of CO₂. As methane has a much higher potential for global warming than CO₂ (56 on a 20 year time horizon, 21 for 100 year; IPCC, 1996) this will have a negative effect on the greenhouse gas budget. On the other hand, wetlands and peatlands might emit less CH₄ after flooding with oxygen-containing water, because methane may get converted to CO₂ within the water body. This conversion is more likely to occur in deep waters, where methane undergoes efficient microbial oxidation to CO₂, whereas methane from shallow waters may form bubble escaping to the atmosphere. And lastly, the loss of the long-term storage capacity of carbon in forest soils has to be added to the gross emission. The studies published so far on methane emissions from large reservoirs illustrate the potential problem. However, careful process studies are required in order to resolve some of the remaining controversies on typical ranges of green house gas emissions per kWh of produced hydropower.

The effect of oxygen depletion on the mercury cycle

In certain reservoirs in northern Canada elevated concentration of methylmercury in fish are not uncommon. It was observed that after flooding of new reservoirs, the concentration in piscivorous fish increased. For example methylmercury concentrations in pikes and walleyes from La Grande Reservoir (Quebec) exceeded at times the Canadian Health and Safety limits for food (Rosenberg et al., 1995; Hecky, 1991).

To a certain extent, mercury is present in all soils. Elevated concentrations are partly due to the geological underground and partly due to atmospheric deposition of mercury prior to damming (Martinet-Cortizas et al., 1999; Rasmussen et al., 1998; Alloway, 1995; Engstrom et al., 1994;). In the latter case, Hg originates from anthropogenic activities such as copper and zinc smelting, burning of fossil fuels and from chemical industries. In soils mercury is present mainly in inorganic form as Hg²⁺ (Steinnes, 1995). This major fraction strongly adsorbs to organic matter and to a lesser extent to mineral surfaces (Alloway, 1995). The mobility and the availability to plants are very low. In aquatic systems inorganic mercury is transformed into methylmercury. The concentration of methylmercury is a result of the balance between production of the compound and degradation by microorganisms (Bodaly et al., 1997; Gilmour et al., 1992; Hecky et al. 1991). Production within and remobilization from the

sediment is especially intense under anoxic conditions, often encountered in young reservoirs (Herrin et al., 1998). It is believed that sulfate-reducing bacteria, active under anoxic conditions, trigger the conversion of inorganic Hg to methylmercury. Thus, the balance between methylation and demethylation is shifted to high methylmercury concentrations within the sediments and in the overlying water (Bodaly et al., 1997; Gilmour et al., 1992). The compound enters the food chain and accumulates in higher organisms and fishes or it is directly adsorbed from water in the gills of fish (Paterson et al., 1998; Hecky et al., 1991). The final methylmercury concentration in the fishes is determined by mercury input, the net balance between methylation and demethylation and the magnification factor in the food chain. Hecky et al. (1991) found that the methylation/demethylation balance rather than the total mercury concentrations in the substrate determines methylmercury concentrations in fish. Methylation seems to be favored in the presence of terrestrial material. The microbial influence on the methylation/demethylation balance is not yet fully understood. Flooding of areas with terrestrial organic matter and occurring sulfate reduction seem to be the trigger for enhanced methylation. The highest methylation rates are therefore to be expected in shallow reservoirs, where anoxic waters cover organic-rich soils.

Downstream effects of changes in oxygen concentrations

If the river water is rich in organic compounds or nutrients, the storage of river water may cause significant decrease in dissolved oxygen concentrations in the deep-water layers of the reservoir. Since the oxygen concentration in the discharged water follows the reservoir concentration at withdrawal level, deep outlets may release low-oxygen water into downstream during stratification periods. Such water can contain, in addition, reduced compounds that may lead to reduced assimilation capacities of the downstream river, which is especially important if domestic and industrial effluents are present. A sufficient supply of dissolved oxygen is essential for the integrity of river ecosystems and vital to the self-purification process within rivers.

The downstream availability of oxygen is a function of the load of reducing compounds and organic substances, and the rates of respiration, photosynthesis and gaseous exchange with the atmosphere. The re-aeration of river water depends on the saturation deficit ($c_s - c$) of oxygen and on the exchange velocity (v_{ex}) across the air-water interface, which in turn depends on the turbulence level of the river. The volumetric re-aeration of the river water can be calculated as: $dC/dt = K \cdot (c_s - c) =$

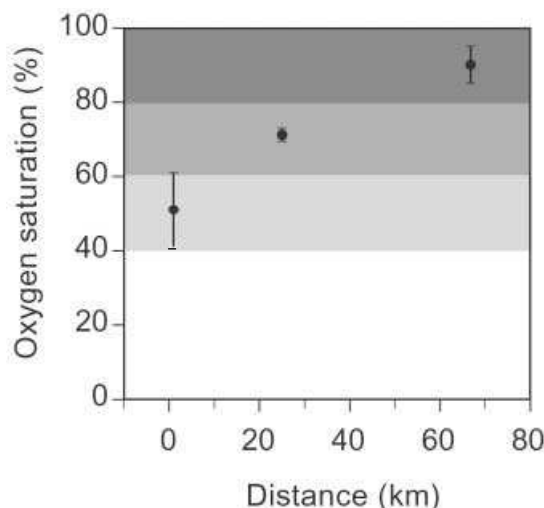


Figure 2. Oxygen saturation in Murray River at three different distances downstream of Lake Hume Dam (Murray River Basin, Australia). The values are averaged over the summer months December and January and over the years 1974–1976 (Data from Walker 1978).

($v_{ex}/h \cdot (c_s - c)$), where K = re-aeration coefficient, h = mean depth, c_s = saturation concentration of oxygen and c = actual oxygen concentration (Durum 1971). For a smooth flow, the exchange surface of the river is equal to the river area and the re-aeration varies only with the river depth (h) and the oxygen deficit ($c_s - c$). However, rough hydraulic structures can cause the air-water contact area to become many times larger. Subsequently the re-aeration depends also on the slope of the river. Accordingly the re-aeration is fastest in turbulent, steep and shallow rivers (Cirpka et al., 1993). At a given oxygen loading, slow flowing and deep lowland rivers have up to ≈ 10 times lower re-aeration capacities (O’Keeffe et al., 1990; Vörösmarty et al., 1997).

A typical situation of water quality degradation due to the deep release of oxygen-depleted water is found below Lake Hume Reservoir in the Murray River (mean water discharge of $133 \text{ m}^3 \text{ s}^{-1}$), Australia. The long residence time of 270 days results, especially in summer, in oxygen depletion of the bottom water in the 41 m deep reservoir. The released Murray River water contains on average 50% and 70% oxygen saturation after 0 and 25 km downstream of the reservoir, respectively (Fig. 2). 100 km downstream, the oxygen finally reached values typical for reference sites at the Murray River not influenced by Lake Hume. The long distance reflects the rather slow re-aeration of the river due to its large water depth and its gentle slope (Walker et al., 1978).

In opposition to low dissolved oxygen levels, supersaturated concentrations of oxygen and nitrogen can also occur due to reservoirs. Supersaturation below dams has typically two origins: (i) air, which is entrained as the

water plunges down the spillway and which is subsequently dissolved into the water, and (ii) high oxygen concentrations that occur in the surface water of reservoirs with appreciable primary production. If water is drawn from the epilimnion, the two effects are combined and cumulate. The best-known, and best-researched example of dam-related supersaturation is the dam-cascade on the Columbia River, USA (CBR, 2001). High level of total dissolved gases (oxygen, nitrogen or both) can lead to gas bubble disease in fishes (Mesa and Warren, 1997). If the fishes are exposed for several hours to total gases in excess of 115%, then emboli accumulate in gill capillaries causing death by anoxia (Mesa et al., 2000).

Phosphorus and nitrogen cycle

With the sedimentation of organic particles, either allochthonous or autochthonous, nutrients are withdrawn from the river system. Although nutrients are partly recycled from the sediment, increased delivery to the sediment can result in higher burial and net sedimentation rates. It was shown that reservoirs can act as efficient sinks for phosphorous and to a lesser degree of nitrogen (Garnier et al., 1999; Uhlman et al., 1995; Uhlman and Horn, 1992;). The retention capacity, however, can vary from year to year depending on the hydrological conditions and input. In Chaffrey Dam Reservoir (Australia) for example, phosphorous retention varied from 16–98% (Sherman, 2001). For nitrogen, sediments either acted as a net sink with maximum retention of 89% or as a net source (Sherman et al., 2001). Especially for phosphorous, reservoirs can exceedingly reduce the amount transported by the river. This effect might be welcomed in eutrophic and polluted rivers. In oligotrophic regions, however, the nutrient depletion might cause changes in the food web and thus create severe problems for fishes downstream of the impoundment.

In Kootenay Lake (British Columbia, Canada), a natural oligotrophic lake, phosphorous decreased markedly after the construction of two dams on its tributary (Duncan Dam on Duncan River and Libby Dam on Kootenay River, Fig. 1). The dark line in Figure 3 shows the history of phosphorous loading to Kootenay Lake (Pieters et al., 1998). The high loading during the 60s was partly due to the operation of a fertilization plant on the upper Kootenay River. After 1969, pollution controls were progressively implemented and the plant was closed in 1987. As a result of both, the pollution control and the retention of nutrients in the upstream impoundments, phosphorous loading dropped during the 80s. The input decreased to a value below the so-called historic level, accounting for the situation prior to dams and anthropogenic pollution (Ashley et al., 1999).

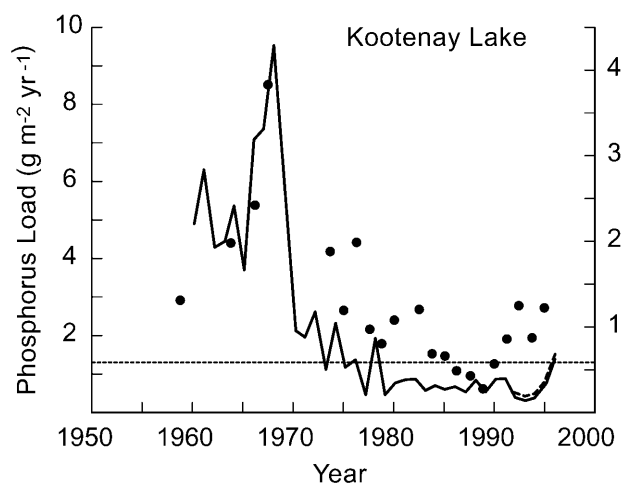


Figure 3. Phosphorus load to (dark line; left scale) and kokanee spawners (dots; right scale) returning from Kootenay Lake (British Columbia, Canada). It is assumed that the drastic decline of the kokanee spawners is due to the decrease of the phosphorus input since 1968 (Ashley et al., 1997). Since the mid 70s, phosphorus load is estimated to be below historical load, due to trapping in the upstream reservoirs (adapted from Pieters et al., 1998; p 9.21/22).

The reduction in phosphorous loading in the 70s and 80s, after the closure of the dams and the successful pollution control was closely followed by declines in phytoplankton and zooplankton. Subsequently the catches of kokanees, a land-locked Sockeye salmon, declined as well (Ashley et al., 1997). The worrying decline of kokanee stocks initiated a fertilization program. Fertilizer is added since 1992 to the lake in the form of liquid ammonium polyphosphate with the goal to stabilize phosphorous concentration at the historical level (Ashley et al., 1997). The artificial fertilization was designed in such a way to compensate for the nutrients, lost in the upstream man-made dams. As a result, the phosphorous concentration in the lake increased since the fertilization program started in 1992 (Fig. 3). Simultaneously the numbers of kokanee spawners (dots in Fig. 3) went up as well (Ashley et al., 1999; Pieters et al., 1998).

Oligotrophication and migrating fish

Probably the largest impact of dam construction is on riverine fishes. Salmonid, trout and sturgeon have been particularly effected by impoundments obstructing their migration, which involves the entire river system. From a geochemical point of view, migrating fish transport nutrients with their bodies from the ocean into the riverine system. A decline of migratory fishes can result in a significant loss of nitrogen (Helfield and Naiman, 2001) and phosphorus supplied by decomposing adult carcasses. In oligotrophic systems, as found in watersheds in remote areas, for instance in NW America or Scandinavia, any

loss of nutrients can be vital to the productivity of the ecosystem and fish stocks. Under natural conditions, anadromous fish add ocean-born nutrients to rivers while they are dying. The natural fertilization of this process may account for more than 30% of the phosphorus input in an oligotrophic lake (Stockner et al., 1996). Therefore the decrease in returning numbers of adult anadromous fishes may add to declining nutrient loads and subsequent oligotrophication of headwaters (Larkin et al., 1997; Bilby et al., 1996).

Nutrient ratios

The efficiency of a reservoir in retaining nutrients varies for different compounds. Lakes and reservoirs are especially efficient in retaining siliceous shells of diatoms produced in the water column. After blooms, diatoms sink faster to the lake bottom than other phytoplankton does. The slow dissolution prevents dissolved silicate from diffusing back into the water column, thus exporting Si into the sediment. Phosphorus on the other hand is recycled more intensely within the epilimnion, since sinking rates for algae other than diatoms are generally slower and at least part of the organically bound phosphorous is released rapidly within the water column. Therefore the P/Si ratio of the water in the outlet might change significantly. The loss of Si in man-made lakes has resulted in declines of dissolved silicate in many river waters after dam closure (Conley et al. 2000; Conley et al., 1993; Wahby et al., 1982, van Bennekom and Salomons, 1981; Mayer et al., 1980). The decline in Si transport to the coastal seas in turn can change the ratio of the nutrient available for the phytoplankton community and hence affects the whole food web of such an ecosystem (Humborg et al., 2000; Turner et al., 1998). However, it is difficult to distinguish between the effects of Si decline, due to retention, and P and N increase, due to eutrophication. Both processes can lead to high N/Si or P/Si ratios with adverse effects on the ecosystem. High N/Si ratios, for example, indicate excess N with regard to diatom needs. A ratio of 1 has been proposed as a threshold with higher values promoting the growth of non siliceous phytoplankton and disadvantaging diatoms (Turner and Rabalais, 1991)

Many catchments were eutrophied and dammed simultaneously over the last 50 years (Humborg et al., 2000; Rahm et al., 1996; Cosiacu et al., 1996; Turner and Rabalais, 1991). One example is the Danube River, which has been turned into a series of lakes with the “Iron Gate” hydroelectric plant (Romania) being the largest impoundment on the river (Petrovic, 1996). At the same time, the Danube watershed is highly populated and industrialized and therefore the river receives considerable loads of nutrients with agriculture as the main source. These high

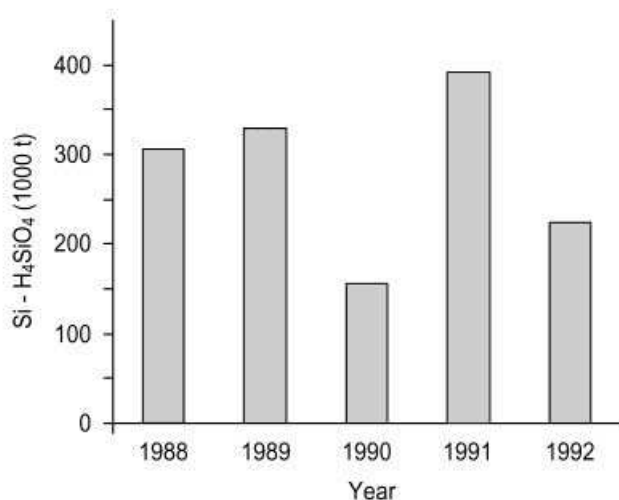


Figure 4. Load of dissolved silicate of the Danube river, estimated from measurements at Sulina station (main branch in the of Danube delta) after Cociasu et al. (1996).

loads of phosphorus and nitrogen lead to extensive diatom blooms in the reservoirs of the Iron Gate. With the sedimentation of these blooms, a significant quantity of dissolved silicate is removed from the reservoir system. Based on biogenic silicate concentrations in the sediment (Reschke, 1999) and the total amount of sediment trapped behind the dam (Panin et al., 1999), a biogenic silicate retention of about 500 kt Si a^{-1} can be estimated. This value agrees with what was estimated by Humborg et al., (1997) as “missing” silicate in the coastal Black Sea since the closure of the Iron Gate in 1975. This is an important part of dissolved silicate transported by the Danube. Between 1988 and 1992 about 300 kt Si a^{-1} reached the Black Sea (Fig. 4; load estimates based on measurements at Sulina Branch, Danube Delta; Cociasu et al., 1996).

While the Iron Gate reservoir seems to be an important sink for dissolved silicate, there are no data available on P or N retention within the impoundment. N/Si and Si/P ratios measured in the water column during a summer cruise indicate that both nutrients are depleted, just as dissolved silicate. The nutrient ratios were 0,5 for N/Si and 200 for Si/P (Reschke 1999). At the river mouth N/Si values of 4–5 were reported (Humborg, 1995; Garnier et al., in press). P and N are supplied below the dam through agricultural activities and wastewater effluent whereas no significant source exists for Si (Brunner, 1997; Friedl et al., in press). As a result of the apparent depletion of dissolved silicate in the riverine delivery, average dissolved silicate concentrations measured in the Black Sea, offshore the Danube Delta, decreased from $55 \mu\text{M}$ before the closure of the Iron Gate to around $20 \mu\text{M}$ today (Fig. 5). The frequency of diatom blooms over the last decades has decreased and dinoflagellates and gelatinous species have become more important (Humborg et al., 1997).

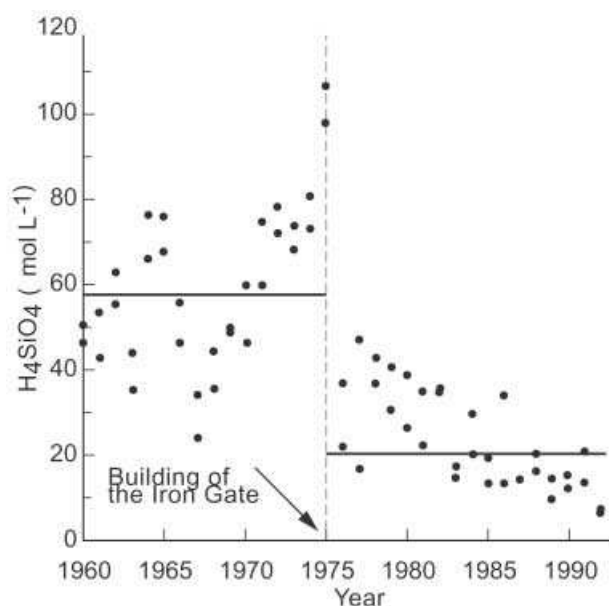


Figure 5. Averages of dissolved silicate concentration in seawater at Constanta-station in winter (Northwestern Black Sea, near the Danube Delta). The dissolved silicate concentration dropped after closure of the Iron Gate, the biggest dam of the Danube River. The bold lines indicate overall medians 1960–1974 and 1975–1992. (Adapted from Cosiacu et al., 1996.)

As a result of the shift in phytoplankton communities, the food chain of the NW Black Sea has changed leading to a drastic decline in fishery since the 70s. Historically the Black Sea has been one of the most fish productive marine regions in the world. Since the 70s number and quality of fish catches declined. Whereas 30 years ago economical valuable mackerels and bonito were caught, today's catches consist only of anchovies (Tolmazin, 1985). Similar effects are described for the River Nile and the coastal sea off Egypt (Halim, 1991) and the Mississippi river basin (Turner et al., 1998).

Harmful algal blooms

Changes in nutrient ratios in coastal seas have been postulated to be responsible not only for shifts in algal communities, but also to favor the growth of toxic algae. Although blooms of toxic algae are a naturally occurring phenomenon, the frequency and severity of harmful algal blooms (HABs) world-wide has multiplied in the past decades (Hallegraef, 1993; Smayda, 1990). Although the exact reasons for the increase are unclear, it is known that nutrients can stimulate harmful phytoplankton species in several ways. The relative availability of specific nutrients can trigger the growth of a selection of phytoplankton species and correspondingly certain HAB species might be stimulated selectively (Smayda, 1990). As

human activities have altered these nutrient supply ratios, harmful forms might be favored.

Changes in N/P ratios were reported to trigger red tide events (Zhang, 1994). Diatoms, the vast majority of which are harmless, require dissolved silicate as nutrient, whereas other phytoplankton do not. Therefore diatoms are sensitive to a decline in Si/N and Si/P ratios. Growth of diatoms can be reduced by dissolved silicate limitation, giving rise to other species such as dinoflagellates, which has many more toxic species (Smayda, 1997; Paerl, 1997; Radach et al., 1990). In addition to the changes in the ratio of nutrients, increasing quantities of nitrogen and phosphorous in coastal areas may result in higher abundance of toxic phytoplankton as well (Pearl, 1997). Habitat alteration, changes in water temperature and turbulence or transport by ship's ballast water can also be responsible for the spread of HABs (Smayada, 1997).

Change in the ratio of nutrients entering the coastal system may also trigger shifts in the food web (Turner et al., 1998). Enclosed seas such as the Northwestern Black Sea, the North Sea or the Baltic Sea are especially susceptible to alterations in nutrient inputs. Thus, modifying the nutrient cycles in the catchment by providing more nutrients (from agriculture runoff and wastewater inflow) or retaining nutrients selectively (biogenic silicate in reservoirs behind dams), or the combination of both, can have dramatic consequences on coastal ecosystems.

In addition to the nutrient supply other factors such as physical forcing, physiology and trophodynamics influence the occurrence of harmful algal blooms. Yet these relationships are far from understood and currently the topic of international research programs (GEOHAB, Global Ecology and Oceanography of Harmful Algal Blooms).

Beyond down-river – final remarks

Building and operating dams modifies the aquatic ecosystem. The consequences are well recognized locally where the rivers are changed into lakes and where the hydrological regime of the downstream river is altered below the dam. The effects occur most prominently locally and receive considerable attention on regional levels. However, the alteration due to the establishment of large dams causes effects far beyond the local scale, transgressing international borders and reaching far outside the system boundaries including deltas and estuaries.

Although substantial knowledge exists on the causalities, it is often difficult to proof cause and effect over large distances. For example, estuarine integrity depends on a balance of riverine freshwater and oceanic seawater

influences. Reducing the variability in freshwater input disturbs this balance. Prolonged low-flow conditions allow saltwater to intrude deeper and for longer time periods into the estuary, potentially increasing the local salinity. For example, the estuary of the Volga River in the Caspian Sea is subject to a fourfold increase in salinity as the water discharge dropped by 70% due to extensive damming. Another example is the Don delta (Azov Sea, Russia), where salinity doubled since damming started at the Don River (McCully, 1996). In the tropics, the balance between freshwater discharge and saltwater as well as sediment transport governs the mangrove forests. Deterioration of these mangroves is often related to changes in freshwater discharge and sediment transport and thus to upstream river damming (Snedacker, 1984).

Many of the discussed effects are problematic only in combination with other man-made alterations. Oxygen depletion in the discharged water of a dam might in some cases have little effect on the river ecosystem downstream by itself. However, if it is combined with high loading of organic matter from sewage and agriculture the ecosystem might suffer heavy deterioration. The repercussions of an altered process may only become obvious after a long period of time, e.g. many years. Salmon not reaching the spawning grounds will slowly alter the P cycle of those waters.

Biogenic silicate retention as discussed for the Danube River might only lead to a slight change in the phytoplankton community in the coastal sea. The simultaneous increase of P and N loads in the Danube lead to enhanced degradation of the coastal ecosystem.

To assess such "fine-tuned" effects, long term records of chemical and biological parameters will be necessary to successfully challenge these questions. For the evaluation of potential effects, water quality model help to reconstruct pre-damming conditions and evaluate the importance of different processes. Modeling can deliver useful results, where large amounts of data are available for the analysis (for example for the river Seine; Even et al., 1998). For most river basins, reliable and homogeneous data is too rare for such modeling approaches (Garnier et al., in press).

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