# Environmental Impacts—Freshwater Biogeochemistry

17

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**Abstract**

Climate change effects on freshwater biogeochemistry and riverine loads of biogenic elements to the Baltic Sea are not straight forward and are difficult to distinguish from other human drivers such as atmospheric deposition, forest and wetland management, eutrophication and hydrological alterations. Eutrophication is by far the most well-known factor affecting the biogeochemistry of the receiving waters in the various sub-basins of the Baltic Sea. However, the present literature review reveals that climate change is a compounding factor for all major drivers of freshwater biogeochemistry discussed here, although evidence is still often based on short-term and/or small-scale studies.

## 17.1 Introduction

The aim of this chapter is to summarise current knowledge on freshwater biogeochemistry within the Baltic Sea with a focus on riverine nutrient and carbon fluxes in the Baltic Sea catchment. Wherever possible, the chapter outlines current knowledge on the effect of climate change and its interplay with the relevant human drivers of change in freshwater biogeochemistry. Quantifying the effects of individual drivers is a challenging task. The chapter covers the background sources and loads of biogenic elements as well as the additional human sources and loads, transformation of dissolved, particulate and gaseous constituents along the aquatic continuum, and possible coeffects of human and climate drivers on sources and transformation. The current and possible future export patterns of biogenic elements to the Baltic Sea are also addressed.

## 17.2 Overview of Freshwater Biogeochemistry and the Baltic Sea Catchment

### 17.2.1 The Baltic Sea Catchment

The Baltic Sea drainage basin comprises a northern boreal part that drains into the Gulf of Bothnia (Bothnian Bay and

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Bothnian Sea) and a south-eastern part that drains into the southern basins of the Baltic Sea (Baltic Proper, Gulf of Finland, Gulf of Riga, Danish Sounds, Kattegat), see also Chap. 5.

The northern watersheds, draining into the Gulf of Bothnia, are generally sparsely populated (Table 17.1) and less eutrophic than the cultivated watersheds of the south-east. The dominant land cover in the north is boreal forest and wetlands. Bedrock is dominated by acid volcanic and plutonic acid rocks (mainly granites); soil types are dominated by till (Durr et al. 2005). The mean slope of the western Swedish boreal watersheds draining mainly Sweden is much steeper, and the specific run-off (i.e. the water volume per unit area and time) is roughly double that of the eastern Finnish watersheds (Table 17.1).

The watersheds of the southern and eastern parts of the catchment are dominated by agriculture (Table 17.1; Fig. 17.1). Sedimentary rocks dominate the south-eastern part of the Baltic Sea catchment, whereas non- to semi-consolidated sedimentary rocks dominate the watersheds of the Oder and Vistula and consolidated sedimentary rocks dominate the watershed of the Nemunas and Daugava (Durr et al. 2005). Most rivers are lowland rivers with a mean slope of less than 1°. River nutrient loads, especially from the Neva, Oder, Vistula, Daugava and Nemunas rivers, contribute most to riverine mass fluxes to the central and southern basins of the Baltic Sea (Stålnacke et al. 1999a). The greatest nutrient mass fluxes come from the rivers Oder and Vistula, draining Poland and with its 38 million inhabitants, about half the population of the entire Baltic Sea catchment (Hannerz and Destouni 2006). Specific discharge is much less compared to the boreal watersheds.

### 17.2.2 Changes Shaping the Baltic Sea

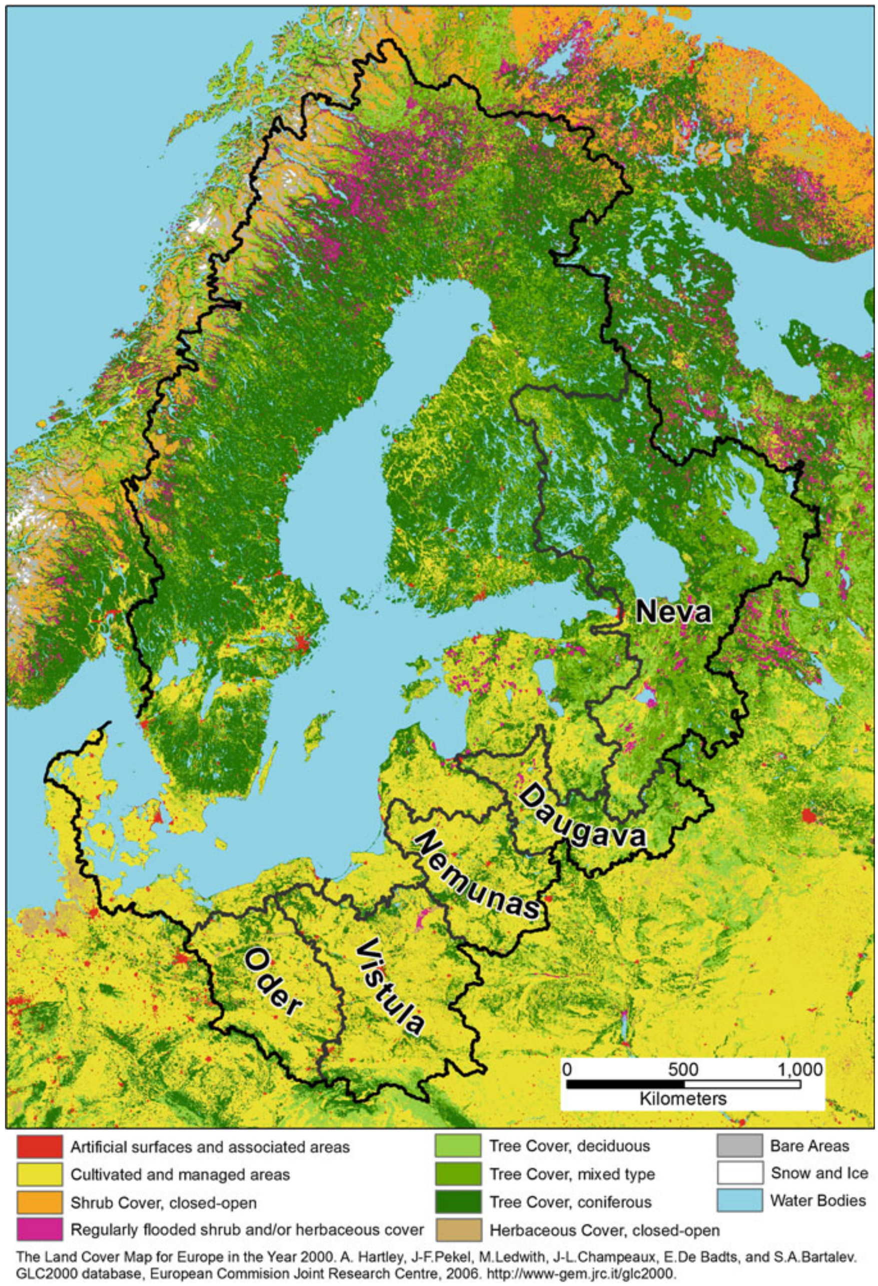
The Baltic Sea is an estuarine system with water residence times of around 30 years and is highly susceptible to changes in riverine loads of biogenic elements (carbon, C; nitrogen, N; phosphorus, P; silicon, Si) (Wulff et al. 1990; Humborg et al. 2007; Meier 2007; Conley et al. 2009; Eilola et al. 2009). In the central parts of the Baltic Sea, a mean salinity of 7 corresponds to about 80 % freshwater and 20 % marine water from the Atlantic. Two major drivers—human lifestyles and global warming—which strongly influence agricultural practices and eutrophication and hydrological patterns for example, could significantly alter the transport of biogenic elements to the Baltic Sea over the near term (Arheimer et al. 2005; Hägg et al. 2010). These changes could be potentially more significant than the variations in

**Table 17.1** Major catchment characteristics of the main Baltic Sea sub-basins: population statistics (2005), land cover (2000) and hydrological properties (average for 1980–2006)

| | Bothnian Bay | Bothnian Sea | Baltic Proper | Danish Straits | Gulf of Finland | Gulf of Riga | Kattegat |
|---|---|---|---|---|---|---|---|
| Area (ha) | 27,029,891 | 23,023,056 | 57,336,809 | 2,735,671 | 41,897,960 | 13,617,922 | 9,008,076 |
| Total population | 1,356,575 | 2,639,277 | 54,100,727 | 4,528,719 | 10,525,186 | 3,794,896 | 3,321,259 |
| Urban population | 468,890 | 771,649 | 18,932,166 | 621,633 | 3,199,319 | 1,252,689 | 609,971 |
| Rural population | 887,685 | 1,867,628 | 35,168,561 | 3,907,086 | 7,325,865 | 2,542,207 | 2,711,288 |
| Run-off ($km^3$ $year^{-1}$) | 105.471 | 96.733 | 110.105 | 6.522 | 114.406 | 33.405 | 33.866 |
| Precipitation (mm $year^{-1}$) | 528.8 | 586.8 | 557.2 | 635.9 | 644.7 | 647.0 | 707.3 |
| Run-off ratio | 0.7379 | 0.7160 | 0.3446 | 0.3749 | 0.4235 | 0.3791 | 0.5315 |
| Temperature (°C) | 0.23 | 2.80 | 7.67 | 8.84 | 4.29 | 6.17 | 6.21 |
| Deciduous forest (%) | 4.24 | 3.34 | 4.87 | 4.81 | 10.07 | 15.59 | 2.67 |
| Coniferous forest (%) | 30.97 | 47.56 | 21.52 | 2.74 | 28.54 | 14.53 | 44.0 |
| Mixed forest (%) | 17.89 | 7.91 | 7.75 | 2.9 | 28.43 | 19.36 | 3.9 |
| Shrub and herbaceous (%) | 21.89 | 19.20 | 5.59 | 1.02 | 8.63 | 16.6 | 5.83 |
| Inland wetlands (%) | 11.59 | 6.02 | 0.52 | 0.55 | 2.53 | 1.93 | 2.46 |
| Maritime wetlands (%) | 0.01 | 0.02 | 0 | 0.21 | 0.01 | 0 | 0.09 |
| Cultivated areas (%) | 3.5 | 6.76 | 54.11 | 77.14 | 6.9 | 29.63 | 26.38 |
| Bare areas (%) | 3.37 | 0.9 | 0.12 | 0.03 | 0.01 | 0.02 | 0.18 |
| Inland water (%) | 5.68 | 6.86 | 2.46 | 1.45 | 13.96 | 1.37 | 11.58 |
| Snow and ice (%) | 0.15 | 0 | 0 | 0 | 0 | 0 | 0 |
| Artificial surfaces (%) | 0.6 | 1.04 | 3.01 | 8.76 | 0.78 | 0.94 | 2.88 |

The run-off ratio gives the share of the annual precipitation discharged by rivers to the Baltic Sea. From the Baltic Environmental Database (www.balticnest.org/bed)

**Fig. 17.1** Landcover in the Baltic Sea drainage basin




riverine fluxes observed over the past 35 years (HELCOM 2004), since changes in lifestyle translate directly into anthropogenic nutrient emissions and riverine fluxes (Howarth et al. 1996; Hägg et al. 2010) and the projected changes in temperature and precipitation are expected to result in fundamental changes within the Baltic Sea catchment (Graham 1999; Graham and Bergstrom 2001; Weyhenmeyer and Karlsson 2009). To date, observations on the discharge regime of major Finnish boreal rivers reveal no changes in mean annual flow during the period 1912–2004 (Korhonen and Kuusisto 2010), but the seasonal distribution of streamflow has changed (see also Chap. 5). Winter and spring mean monthly discharge increased at most observation sites, and the spring peak has become earlier at a third of sites (Korhonen and Kuusisto 2010). However, additional drivers such as atmospheric deposition (Monteith et al. 2007; Weyhenmeyer 2008), management of forestry and wetlands as well as damming and other types of hydrological alteration (Dynesius and Nilsson 1994; Nilsson et al. 2005; Humborg et al. 2006, 2008a) are compounding factors affecting freshwater biogeochemistry, especially in the boreal watersheds.

### 17.2.3 Drivers of Change in Sources, Transformation and Export of Biogenic Elements to the Baltic Sea

Freshwater biogeochemistry in relatively unperturbed aquatic systems within the Baltic Sea catchment and the background load of the biogenic elements C, N, P and Si is the result of its weathering regime, which characterises total ionic strength, acidity (pH) and alkalinity as well as vegetation cover and vegetation type. Generally, weathering reactions charge rainwater with basic cations and anions including dissolved inorganic carbon (DIC), orthophosphate and silicic acid when infiltrating natural soils (Drever 1997). Nitrogen enters the system through biological N fixation, and organic C stems from recently produced biomass (mainly litter and root exudates) (Froberg et al. 2003; Karltun et al. 2005; van Hees et al. 2005; Giesler et al. 2007; Jonsson et al. 2007) and older stored soil organic C (Vonk et al. 2008; Vonk and Gustafsson 2009). However, the bedrock dominated by acid volcanic and plutonic acid rocks as well as the occurrence of coniferous forests and wetlands storing huge amounts of organic C leads to freshwaters in boreal watersheds characterised by low ionic strength, low alkalinity and high concentrations of humic and fulvic acids that form the major pool of dissolved organic carbon (DOC), nitrogen (DON) and phosphorus (DOP). Background loads and concentrations in the cultivated watersheds are difficult to estimate because humans have influenced these landscapes over many centuries. However, the occurrence of sedimentary bedrock in cultivated watersheds and the higher temperatures that increase weathering reactions lead to a higher ionic strength and higher alkalinity.

Relatively natural unperturbed conditions can be found in the well-studied River Kalixälven (Ingri et al. 1997; Humborg et al. 2004), in the Simojoki river basin (Lepisto et al. 2008) and in unmanaged headwater catchments (Mattsson et al. 2003; Finer et al. 2004; Kortelainen et al. 2006a). In most other watersheds around the Baltic Sea, freshwater biogeochemistry is affected by following human drivers: atmospheric deposition, forestry and wetland management, eutrophication, and damming and other types of hydrological alteration.

#### 17.2.3.1 Human Drivers

*Atmospheric deposition*. Atmospheric deposition of acids, metals and nutrients peaked in the 1970s and 1980s with the notorious acidification effects observed in lakes and streams of Sweden and Finland (Weyhenmeyer 2008). The effects of atmospheric deposition and acidification are generally more significant in the northern boreal part of the Baltic Sea catchment than in the southern cultivated areas, because of the lower buffer capacity of the surface waters and soils. Moreover, atmospheric deposition is also more significant compared to other drivers in relatively unperturbed watersheds with low population densities. For more detailed discussion, see Chap. 15.

*Forestry and wetland management*. Forests and wetlands are the dominant landscape forms in the northern boreal part of the Baltic Sea catchment and cause the high concentrations of dissolved organic matter (DOM) comprising humic and fulvic acids that colour the surface waters brownish (Laudon et al. 2011). Forest and wetland management (i.e. clear-cutting, ditching, and peat mining) have influenced the hydrology and biogeochemistry of streams, lakes and rivers for centuries (Löfgren et al. 2009; Nieminen et al. 2010). For more detailed discussion, see Chaps. 21 and 25.

*Eutrophication*. Eutrophication is by far the most investigated and well-understood driver of change in freshwater biogeochemistry. Numerous studies illustrate the effects of agricultural practices and urban and industrial point sources on nutrient concentrations in lakes and rivers (Larsson et al. 1985; Rheinheimer 1998; Stålnacke et al. 1999b; Raike et al. 2003; Arheimer et al. 2004; Lysiak-Pastuszak et al. 2004; Ekholm et al. 2007; Humborg et al. 2007; Lindgren et al. 2007; Kronvang et al. 2009; Iital et al. 2010b) and subsequent effects on aquatic ecosystems, such as increased turbidity, anoxia and loss of biodiversity (Blenckner et al. 2006; for more detailed discussion, see Chap. 18). Long-lasting effects due to sediment nutrient release cause less efficient nutrient sequestration and retention of biogenic elements.

*Hydrological alterations*. Damming is more frequent in the boreal rivers owing to its higher effectiveness in terms of power generation (Humborg et al. 2000, 2008b). Major reservoirs located in their headwaters can hold between 30 and 70 % of the annual water discharge (Dynesius and Nilsson 1994; Nilsson et al. 2005). In contrast, damming is much less frequent in the lowland rivers of the south-eastern parts of the Baltic Sea catchment and it is mostly small dams and reservoirs with short water residence times that have been built there (Humborg et al. 2006). Nevertheless, there is a considerable body of literature reporting 'oligotrophication' of river systems as an effect of damming, which is the process of nutrient depletion caused by reduced contact of surface and groundwater with vegetated soils. Similar patterns have been recorded for lakes in the watershed—the higher the lake percentage of the catchment area, the lower the total organic carbon (TOC), N and DOP concentrations/fluxes (Mattsson et al. 2005; Lepisto et al. 2006).

#### 17.2.3.2 Climate Drivers

Another driver affecting river biogeochemistry is climate, principally temperature and precipitation patterns. Although it is still difficult to quantify the extent to which human activities have affected climate in the Baltic Sea catchment, increased temperature affects hydrological properties such as evapotranspiration (positive effect) and river discharge (negative effect), so this is a vital variable for net export of dissolved and particulate matter to the Baltic Sea.

*Temperature*. Biogeochemical fluxes are strongly influenced by temperature. For example, stream temperature affects the uptake of nutrients and thereby their concentration in streams (Rasmussen et al. 2011). Furthermore, temperature regulates microbial activity in soils and waters, which transforms and degrades biogeochemical components. Changes in the timing of ice break-up and snow melt may alter the timing of the spring flood, which has been suggested to cause changes in DOC over time (Hongve et al. 2004; Weyhenmeyer 2008). Areas south of 61°N show high interannual variability and are especially sensitive to warming (Weyhenmeyer et al. 2011). It can be hypothesised that in such climate-sensitive areas daily freeze-thaw of surface soils activates different soil layers and could even increase soil erosion.

In the northern Baltic Sea catchment, permafrost thaw in peatlands is accelerating (Christensen et al. 2004; see also Chap. 6). Kokfelt et al. (2010) found that continued permafrost thaw and related vegetation changes towards minerotrophy (i.e. water supply through groundwater/streams/springs as opposed to precipitation) may increase C and nutrient storage in mire deposits and reduce nutrient fluxes in run-off. They concluded that rapid permafrost degradation may lead to widespread mire erosion and to relatively short periods of significantly increased nutrient loading to streams and lakes (Kokfelt et al. 2010). However, due to the relatively small areas containing permafrost, permafrost degradation is unlikely to cause much change in river discharge (see also Chap. 6).

*Precipitation*. Interannual variability in precipitation can have a large effect on biogeochemical fluxes. Drier-and-wetter-than-normal years have a large effect on the export of nutrients and C within Baltic Sea catchments. At the large scale, total nitrogen (TN) and total phosphorus (TP) river loads to the Baltic Sea correlate well with precipitation and discharge patterns and vary by up to 30–40 % between years (Humborg et al. 2007). Higher (wet years) and lower (dry years) export of DOC has been observed in Finland (Jager et al. 2009). Such patterns are further influenced by changes in snow pack dynamics within the catchment. Pärn and Mander (2012) concluded that the main factor driving an increase in TOC export in Estonia between 1992 and 2007 was the deepening of droughts, that is rising trends in hydrological drought days driven by climate change and magnified by man-made drainage.

#### 17.2.3.3 Net Export of Biogenic Elements

The net export of biogenic elements as particulate, dissolved or gaseous constituents to the Baltic Sea is the result of release from natural and human sources and the transformation of matter along the aquatic continuum formed by streams, lakes and rivers (Fig. 17.2). These transformations include biological processes (formation and degradation of biomass in the widest sense) and physicochemical processes (particle formation, sedimentation, burial and gas exchange) leading to an overall retention of biogenic elements (Behrendt and Opitz 1999; Kortelainen et al. 2004, 2006b; Venohr et al. 2005), and only a proportion of the nutrients originally released from the natural and human sources will finally reach the Baltic Sea. The human drivers (Sect. 17.2.3.1) and climate drivers (Sect. 17.2.3.2) will influence the sources and transformations in different ways. Atmospheric deposition, forestry and eutrophication will clearly influence point and diffuse sources of biogenic elements, whereas these human drivers will have both positive and negative feedbacks on transformation processes and retention. Damming and other hydrological alterations including ditching as part of forestry and wetland management will directly affect water residence times in watersheds, which is a major variable determining the total period over which transformation processes can occur during passage from land to the Baltic Sea and is positively linked to the retention of biogenic elements (Valett et al. 1996; Behrendt and Opitz 1999; Søndergaard 2007). Temperature will affect weathering processes, N fixation rates, terrestrial primary production and the overall kinetics of transformation processes, while an increase in precipitation will increase weathering up to a certain threshold (Kump et al. 2000) and increase nutrient loads to the Baltic Sea, that is decrease the time available for transformation processes and retention (Hong et al. 2012).

## 17.3 Atmospheric Deposition and Freshwater Biogeochemistry

### 17.3.1 Atmospheric Deposition and Waterborne Fluxes

Atmospheric deposition (see also Chap. 15) implies the input of acids, metals, nutrients, particulates and pollutants from the atmosphere to terrestrial and aquatic ecosystems. The far-reaching impact of atmospheric deposition on freshwaters was first recognised in the 1960s. In the Nordic countries, Svante Odén was the first to warn about the consequences of atmospheric deposition, publishing an article in the largest circulation Swedish national newspaper Dagens Nyheter on 24 October 1967 entitled *The acidity of precipitation*. Odén related fish kills to sulphate deposition caused by the burning of fossil fuels. Since then, increasing numbers of freshwater quality problems in the countries

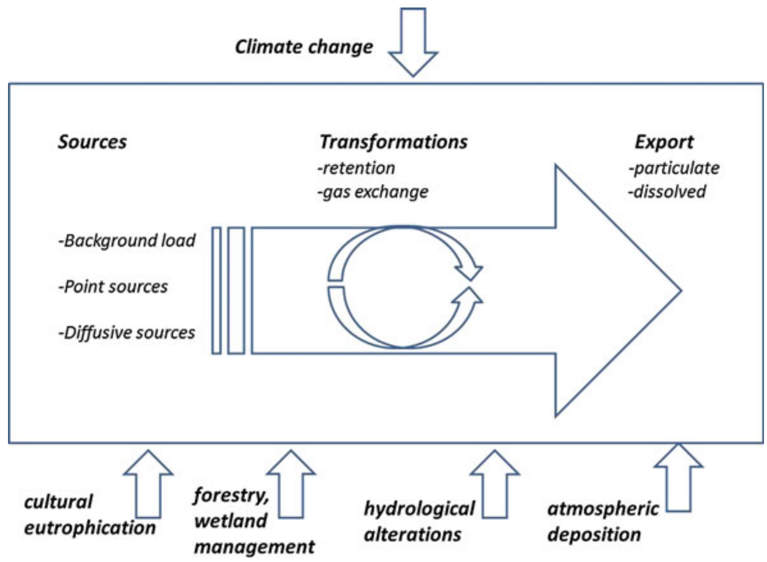



**Fig. 17.2** Conceptual scheme showing how human and climate drivers influence the sources, transformations and export patterns of biogenic elements to the Baltic Sea

surrounding the Baltic Sea have been attributed to atmospheric deposition, from increased nutrient concentrations (particularly N) to increased concentrations of metals and persistent organic pollutant (Hessen et al. 1997; Agrell et al. 1999; Holt 2000; Bindler et al. 2009). Even high levels of radioactive substances accumulated in biota and sediments have their origin in atmospheric deposition. A particularly well-known example is the deposition of radioactive substances after the accident at the Chernobyl nuclear power plant on 26 April 1986, which are still being detected in lake sediments (Lusa et al. 2009). One of the most dramatic changes in freshwaters as a consequence of atmospheric deposition is probably the shift from natural N limitation to P limitation in lakes (Bergström et al. 2005; Bergström and Jansson 2006) as well as C limitation (Weyhenmeyer and Jeppesen 2010). Such shifts have caused substantial change in the biogeochemical cycles of freshwaters, and probably also the Baltic Sea.

The reasons why it took so long to identify the effects of atmospheric deposition on freshwaters are twofold. Traditionally, freshwaters (especially lakes) were studied as individual systems, with little connection to large-scale driving forces (Livingstone and Hari 2008), and the impact of atmospheric deposition on freshwaters is often masked by other drivers and therefore often overlooked.

As background levels began to need quantifying, increasing interest was shown in atmospheric deposition. Background levels for substances originating in atmospheric deposition are generally very low and still relatively uncertain, especially for those substances that are transformed as they move through the environment. Most information on background levels is available from sediment cores. Analysing sediment cores from freshwaters and taking lead (Pb) as an example indicates a significant rise in atmospheric Pb fallout from about AD 1000, followed by rapid increase during the Industrial Revolution. Concentrations peaked in the 1970s and then declined (Branvall et al. 2001). Background levels can also be quantified by methods other than sediment analyses. For example, Hägg et al. (2010) used a statistical approach to estimate a background flux of about 100 kg N $km^{-2}$ $year^{-1}$ from boreal catchments.

Atmospheric deposition to freshwaters is best studied in remote regions. Freshwaters in these areas may be seen as mirrors of the atmosphere and so may be used as reference systems that are highly sensitive to changes in atmospheric deposition and climate. A substantial proportion of the freshwaters draining into the Baltic Sea originates in remote regions. Although atmospheric deposition shows a strong north–south gradient with the deposition of all substances increasing southwards (Weyhenmeyer 2008), the relative importance of atmospheric deposition on freshwater biogeochemical cycling increases towards the remote northern region, where nutrient and pollutant inputs from other sources are small. Atmospheric deposition generally reflects industrial development. At present, atmospheric deposition of metals and sulphate is declining not just over the Baltic Sea area but also over other regions (Ruoho-Airola and Salminen 2003; Harmens et al. 2010; Slemr et al. 2011). The decline in sulphate deposition has resulted in a rapid recovery of freshwaters from acidification (Evans et al. 2001). This recovery has reduced the need for intense liming activities with consequent impacts on the bioaccumulation of mercury (Hg) (Shastria and Diwekar 2008). Nitrogen deposition has also decreased over the northern regions, which has lowered nitrate concentrations in freshwaters (Weyhenmeyer et al. 2007; Kothawala et al. 2011). Some of the freshwaters surrounding the Baltic Sea currently experience nitrate-depleted conditions during summer resulting in periodic N limitation (Weyhenmeyer et al. 2007). If N deposition continues to decrease, a substantial proportion of freshwaters in the Baltic Sea drainage area may shift back towards N-limited systems with consequent impacts on biogeochemical cycling (Weyhenmeyer and Jeppesen 2010).

### 17.3.2 Transformations of Nutrients Along the Aquatic Continuum

Background levels assume that transformations in the landscape have reached steady state, that is their natural release is in balance with natural removal processes. Increasing numbers of studies, however, show that transformations can either decrease or increase in their efficiency with changing environmental conditions. For example, Weyhenmeyer and Jeppesen (2010) reported that the efficiency of nitrate removal from freshwaters varies with changes in N deposition. Since substantial fractions of substances are retained or lost in boreal catchments by a wide range of transformation processes, changes in transformation processes within the landscape will have major impacts on river export. Hägg et al. (2010) estimated that about 75 % of the anthropogenic N deposited from the atmosphere was retained within the boreal landscape before entering the sea. Retention capacity varies depending on background levels and the biogeochemical process in question. There are indications that southern parts of the boreal region surrounding the Baltic Sea are more N-saturated than northern parts (Weyhenmeyer and Jeppesen 2010), resulting in a more effective N retention capacity in northern landscapes than southern. This pattern is probably also valid for other substances and will be affected by changes in trends in atmospheric deposition.

### 17.3.3 Climate Impacts on Atmospheric Deposition and the Effect on Waterborne Fluxes

To distinguish the effects of changes in atmospheric deposition from the effects of climate change is a major challenge, mainly because changes in climate strongly covary with changes in atmospheric deposition. There are two reasons for this strong covariation. Over the long term, changes in climate follow changes in carbon dioxide ($CO_2$) emissions that themselves follow changes in the emissions of other substances (Lamarque et al. 2010). While over the short term, deposition patterns for substances from the atmosphere are determined by atmospheric circulation patterns in a similar way to meteorological variables (such as winds and precipitation); thus, short-term variations in atmospheric deposition and climate follow each other (Dayan and Lamb 2005; Weyhenmeyer 2008; Shi et al. 2011). While physical freshwater variables such as temperature, run-off and ice cover can easily be attributed to climate change, it is more difficult to assess climate change effects on chemical and biological freshwater variables (Adrian et al. 2009). Carbon is probably one of the most debated freshwater variables where agreement has not yet been reached on whether it is changes in atmospheric deposition or in climate that are mainly responsible for the recent wide-scale increases. A variety of studies attribute the increase in C concentrations and/or loads mainly to changes in climate variables such as run-off (Andersson et al. 1991; Schindler et al. 1997; Freeman et al. 2001; Worrall et al. 2004; Erlandsson et al. 2008), temperature, growing season length and run-off season duration (Weyhenmeyer and Karlsson 2009), solar radiation (Hudson et al. 2003), the timing of ice break-up and snowmelt (Hongve et al. 2004; Weyhenmeyer 2008), and soil moisture (Worrall et al. 2006). It has been proposed that concentrations and fluxes of DOC are more strongly related to climate and landscape topography than to internal properties of aquatic ecosystems (Mulholland 2003). Long-term trends in the timing of ice freeze-up (getting later) and ice break-up (getting earlier) towards shorter periods of ice cover have been reported for lakes and streams around the Northern Hemisphere (Palecki and Barry 1986; Magnuson et al. 2000; Blenckner et al. 2004, 2007; Prowse and Brown 2010) and can profoundly affect biogeochemical components. Interannual variability in freeze-up and break-up dates is also important (Weyhenmeyer et al. 2011). Other studies attribute the increase in DOC mainly to changes in atmospheric deposition chemistry (Freeman et al. 2004; Bragazza et al. 2006; Evans et al. 2006; Monteith et al. 2007). As Roulet and Moore (2006) concluded, it is probably a combination of different drivers that are responsible for increasing C levels in freshwaters (Fig. 17.3). When a variable is affected by several drivers, patterns become complex and the possibility of additive, synergistic and antagonistic effects occurs. In this context, pH is a good example. In freshwaters, pH has a strong negative relationship with sulphate deposition (Evans et al. 2001) and a strong positive relationship with temperature (Houle et al. 2010). Since temperature is currently rising and sulphate deposition currently falling, the accelerated recovery of pH in freshwaters must be attributed to both.

### 17.3.4 Current and Future Export to the Baltic Sea

Over previous decades and especially in the 1970s and 1980s, atmospheric deposition seems likely to have had a stronger effect on freshwater biogeochemical conditions in the Baltic Sea drainage area than climate. However, this pattern seems to weaken as atmospheric deposition declines. A shift back to climate regulation of freshwater

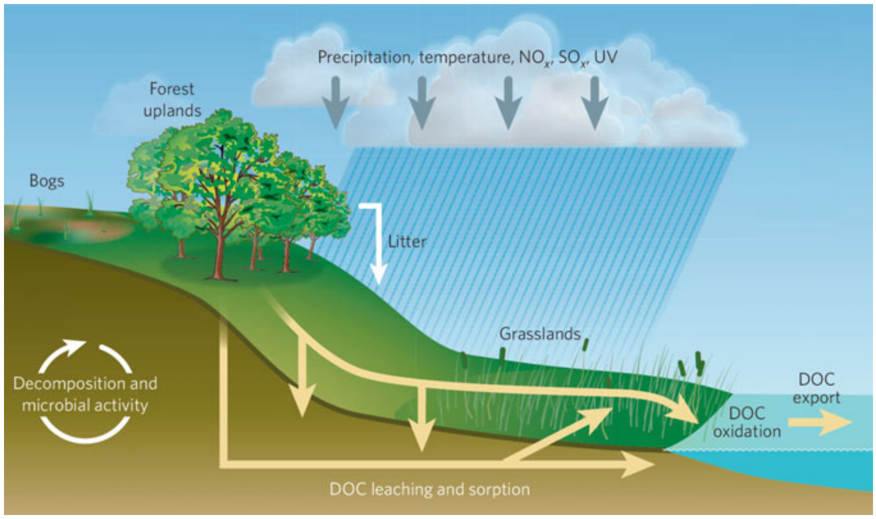



**Fig. 17.3** The decomposition and subsequent leaching of organic matter in bogs, forests and wetlands are the principal sources of dissolved organic carbon (*DOC*) in the terrestrial landscape. Production is mediated by several physical and biogeochemical factors, such as the atmospheric deposition of nitrates and sulphates, moisture and temperature. The rate of export of terrestrial DOC is determined by the rate of production combined with the rate of sorption by mineral soils, and the availability of pathways for water through the environment (Roulet and Moore 2006)

biogeochemistry has already been observed in the United States (Mitchell and Likens 2011). This suggests that the biogeochemical conditions of freshwaters and the Baltic Sea could change rapidly.

## 17.4 Forestry, Wetland Management and Freshwater Biogeochemistry

### 17.4.1 Forest, Wetlands and Waterborne Fluxes

Forests and wetlands dominate the landscape in northern Fennoscandia accounting for more than 85 % of the total land area in northern Sweden, of which wetlands account for 13 % (SLU 2010; see also Chap. 21). In Finnish watersheds, the proportion of upland forests is 29–64 % (average 49 %) and the proportion of peatlands 3–60 % (average 22 %). The percentage of peatlands is highest in the band between 63° and 66°N, whereas the proportion of forests increases towards the south. Over half of the Finnish peatlands, which originally covered a third of the land area has been ditched, mostly for forestry (Aarne 1994). In Estonia, forestry is the dominant land use occurring on 21,974 km$^2$ (50.3 %) of the total territory. Wetlands are extensive landscapes covering 25–30 % of the country's territory, including a substantial fraction of agricultural and forest land. About 70 % of Estonian peatlands are drained or influenced by drainage to an extent that presumably no longer allows peat to accumulate (Ilomets and Kallas 1997).

As the major northern Swedish rivers pass through the landscape from their headwaters in the Scandinavian mountains, they get progressively enriched in TOC and silica (Si); the latter associated with weathering release (Humborg et al. 2004). This enrichment is directly coupled to forest and wetland cover at the landscape level (Humborg et al. 2004; Smedberg et al. 2006). The importance of boreal forests was also emphasised in a study of DOM export along a European climate gradient; the study found that the export of DOC was highest from the Finnish boreal forest sites and clearly associated with forest and wetland catchment coverage (Mattsson et al. 2009). Forests affect the biogeochemical cycles of elements in several ways; trees and in particular coniferous trees are efficient filters for dry and wet deposition increasing the airborne loads of elements to the soil (Robertson et al. 2000). Trees promote soil formation, weathering and C accumulation in soils and also transfer a large amount of current photosynthates to the soil, the latter being an important source of DOC via rhizodeposition (Giesler et al. 2007). Wetlands have a specific role in the boreal landscape since they are hotspots for waterborne organic C (Fig. 17.3). For instance, wetland coverage in the boreal landscape has been found to be positively related to organic C export in streams (Laudon et al. 2011). In Finland, high peatland proportion has been shown to increase average annual leaching of TOC both in managed (Kortelainen et al. 1997, 2006a; Kortelainen and Saukkonen 1998; Mattsson et al. 2003) and unmanaged catchments (Mattsson et al. 2003; Kortelainen et al. 2006a).

The export of DOC and a number of elements is highly seasonal and differs between forest and wetland-dominated sites (Laudon et al. 2011). Long-term monitoring of a boreal mixed forest/wetland drainage basin has shown that wetland-dominated stream catchments are characterised by low DOC concentrations during peak flow events and high DOC concentrations during baseflow conditions (Giesler et al. 2007; Laudon et al. 2011). In forested catchments, concentrations and element fluxes of DOC and elements associated with DOC increase during spring snowmelt (Dyson et al. 2011), whereas other elements are mostly diluted. Typically, elements linked to weathering such as Si, calcium (Ca), magnesium (Mg), potassium (K) and sodium (Na) are highest at baseflow conditions (Smedberg et al. 2006). However, in two forested peat-dominated (drained and undrained) catchments in eastern Finland, DOC concentrations were positively correlated with cation concentrations in both catchments indicating a common peat/groundwater flow path (Dinsmore et al. 2011). Different landscape sources also affect the 'quality' of exported DOC—during baseflow conditions, there is a greater export of older more recalcitrant DOC compared to peak flow conditions where the flux is dominated by younger DOC, that is from surface soil horizons of forest soils (Laudon et al. 2011).

In larger catchments with diverse land cover types, water mixing masks clear patterns; there is also a tendency for decreasing baseflow DOC concentrations with increasing catchment size, which is attributed to the increased influence of deep groundwater or shifts in soil texture from unsorted tills to more sorted fine materials in the lower parts of large catchments (Laudon et al. 2011). A study of water chemistry within the River Kalix catchment showed that spring flood events were dominated by TOC input from upper forest soil horizons and peatlands, while storm peak flow events were dominated by TOC flushed from peatlands. Most of the seasonality and patterns observed in headwater streams are significantly reduced with increasing distance downstream, similar to DOC export and concentrations (Wolock et al. 1997; Humborg et al. 2004; Mattsson et al. 2005; Temnerud and Bishop 2005). This may be caused by a number of factors such as mixing of different sources and degradation of DOM and/or sedimentation of elements along the flow-path. Furthermore, large catchments with longer water retention times are more buffered against variations than headwater systems. Overall, it is clear from many studies that forests and wetlands play a key role in the terrestrial export of a number of elements (Fig. 17.3).

### 17.4.2 Influence of Management Practices

#### 17.4.2.1 Clear-Cutting

The effects of forestry practices on freshwater quality have been investigated in a number of studies in Sweden and Finland (Ahtiainen 1992; Ahtiainen and Huttunen 1999; Finer et al. 2004; Laudon et al. 2009; Löfgren et al. 2009; Nieminen et al. 2010). One of the most important effects of clear-cutting is the reduction in evapotranspiration, resulting in increased run-off, elevated groundwater levels and a change towards shallower flow pathways. Another effect results from the decreased competition from trees after a clear-cut, which increases nutrient availability and nitrification rates. Increased insolation to the soil caused by the absence of a forest canopy will also promote degradation of soil organic matter and mineralisation rates. The impact on the export of elements is, however, linked to the intensity, extent and duration of forest practices, but also reflects climate, topography and soil properties.

In northern Sweden, logging resulted in increased run-off and increased concentrations of Na, K, chloride, TN, TP and suspended material from the two study catchments, whereas nitrate leaching increased only from the catchment without a forest buffer (Löfgren et al. 2009). A high frequency of water sampling during a clear-cut catchment experiment in northern Sweden one year after harvesting showed increased streamwater DOC concentrations during the growing season. This study supports the hypothesis that a raised groundwater level following harvesting caused the increased DOC concentration during hydrological episodes and low-flow conditions (Laudon et al. 2009). In Finland, clear-cutting and subsequent scarification increased total and inorganic P and N, total iron (Fe) concentrations and suspended solids (Ahtiainen 1992). Nieminen (2004) showed that clear-cutting significantly increased the export of DOC and N from drained productive peatlands, while only small increases in P export were found. Buffer zones have shown to be efficient in retaining inorganic nutrients (Silvan et al. 2005; Vaananen et al. 2008; Vikman et al. 2010). Koskinen et al. (2011) showed that the calculated mean annual leaching of P, N and TOC from post-restoration treatment areas was high in comparison with average leaching from undisturbed catchments (Mattsson et al. 2003; Kortelainen et al. 2006a) and average leaching from managed forested catchments (Kortelainen et al. 1997).

#### 17.4.2.2 Site Preparation

Site preparation may add to the effect of clear-cutting; in an experiment by Löfgren et al. (2009), site preparation caused an additional increase in DOC losses of 79 %. A Finnish study (Piirainen et al. 2007, 2009) showed similar results and suggests that site preparation after forest harvesting can increase C, N, P and cation leaching from soils more than the clear-cutting itself. Although this illustrates the effect over the short term, short-term and long-term effects of forestry can be significantly different. Clear-cutting often results in increased concentrations and leaching some years after treatment, but over the long term and after re-establishments of forests this is likely to result in a lower water table and decreasing leaching.

#### 17.4.2.3 Ditching

Ditching has been a common forest practice especially on peat soils and has been found to result in a short-term increase in TOC concentrations (Heikurainen et al. 1978; Moore 1987). Over the long term, ditching lowers the groundwater level and can result in decreased TOC leaching. However, Sallantaus (1994) found no differences in the leaching of TOC between natural fen, natural bog, drained fen and drained bogs several years after treatment. Rantakari et al. (2010) studied long-term effects of ditching in small headwater catchments and showed decreased total inorganic carbon (TIC) and TOC concentrations, but no significant effects on lateral C export due to increased run-off patterns. Reditching, that is managing ditch networks, has been shown to result in decreasing DOC and DON concentrations, and increasing inorganic N, suspended solids and base cation concentrations, whereas no significant changes were found in TN and TP concentrations (Joensuu et al. 2001, 2002). In an experiment carried out in nine pairs of treated and control (no maintenance) catchments located in southern and central Finland, a significant increase in the export of suspended solids for the four-year study period following the ditch network maintenance and aluminium (Al) export increased for one year. The export of N, P and Fe was not significantly changed, and DOC and manganese (Mn) export decreased after the ditch maintenance operation (Nieminen et al. 2010).

#### 17.4.2.4 Multi-Stressors

There are relatively few long-term studies on the effect of different forest practices. However, Kortelainen and Saukkonen (1998) studied average long-term leaching of C, N, P and Fe from 20 headwater catchments representing Finnish forestry land, including the most important forest practices (ditching, clear-cutting, scarification and fertilisation). Ditching was the largest-scale forestry practice in the study catchments. In many catchments, ditching was already ongoing in the early 1960s, although these catchments were not monitored until the 1970s. Considering the differences in catchment size, location, forest type and peatland type as well as different forest practices, the regional differences in the average long-term leaching of total organic nitrogen (TON) and TOC were not large. Interannual variation in TOC in single headwater streams was shown to be greater than spatial variation in average annual TOC fluxes between the catchments (Kortelainen et al. 1997). Regional variation was reduced because concentrations were generally lower in northernmost catchments, while run-off from the study catchments increased to the north. Moreover, the concentrations were higher in the catchments with a high peatland proportion, while run-off from these catchments was slightly lower compared to the catchments with a low peatland proportion. The average annual leaching of TOC, TN, TP and total Fe was greater in southern catchments than northern catchments. Furthermore, high peatland proportion increased average annual TOC, TN and total Fe export both from managed and unmanaged catchments (Kortelainen and Saukkonen 1998; Kortelainen et al. 1997, 2006a; Mattsson et al. 2003).

### 17.4.3 Transformations of Carbon Along the Aquatic Continuum

Receiving lakes and streams are described as 'active pipes' in the export from land to ocean (Cole et al. 2007), and their role as regulators of C cycling has been depicted in several recent studies (Cole et al. 2007; Tranvik et al. 2009; Humborg et al. 2010; Lyon et al. 2010; Einola et al. 2011). Estimates by Tranvik et al. (2009) demonstrated that the global annual emission of $CO_2$ from inland waters is of the same magnitude as $CO_2$ burial by the oceans and boreal lakes seem to be especially important in this context. This has also been emphasised in a number of studies showing that lakes are supersaturated in $CO_2$ (Jansson et al. 2000; Sobek et al. 2005; Kortelainen et al. 2006b). A recent budget calculates the total $CO_2$ efflux from all Swedish lakes and streams to be 2.58 Tg C year$^{-1}$ (Humborg et al. 2010; Fig. 17.4). In-lake respiration, mainly derived from

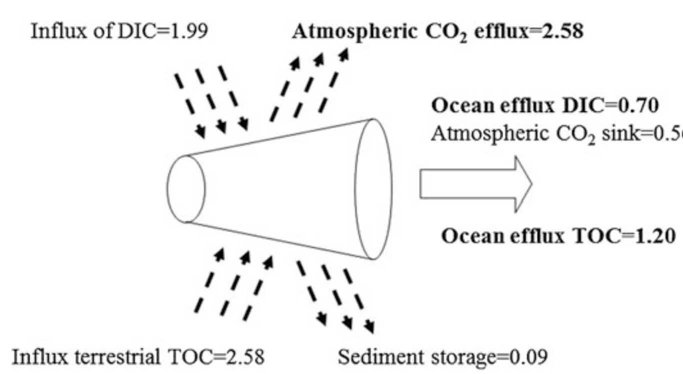



**Fig. 17.4** Schematic view of the major inorganic and organic carbon pathways along the aquatic continuum. Fluxes represent Sweden as a whole and are expressed in Tg C year$^{-1}$ (Humborg et al. 2010)

terrestrial C, has been suggested as an important contributor to the $CO_2$ emitted from lakes (del Giorgio et al. 1999), but inflow of DIC and $CO_2$-rich groundwater can also contribute to lake $CO_2$ (Striegl et al. 2001). The latter was stressed in a study by Humborg et al. (2010) illustrating that the partial pressure of $CO_2$ ($pCO_2$) in a large number of Swedish lakes and streams was strongly related to factors indicative of groundwater influence, that is typically weathering products such as Si, Mg and DIC. The significance of the weathering component in C sequestration has been widely overlooked and may be vital for interpreting future C budgets of major boreal and Arctic watersheds. Weathering reactions consume atmospheric $CO_2$ and form DIC that is locked over geological timescales in the aqueous phase and finally in ocean sediments. The weathering sink of atmospheric $CO_2$ constitutes a negative feedback on atmospheric $CO_2$ due to climate change at northern latitudes (Smedberg et al. 2006; Lyon et al. 2010). Thus, the overall feedback of boreal groundwaters, streams and lakes to atmospheric $CO_2$ is heavily debated. Quantitative estimates are still lacking of the relative importance of groundwater versus in-lake or stream processes contributing to $CO_2$ emissions. Nevertheless, it is clear that future changes in the terrestrial DOC export and changes in the influx of groundwater to the receiving lakes and streams can potentially affect $CO_2$ emissions in aquatic systems. Such changes could be additive in a scenario where both the groundwater influx and DOC export increase.

Dinsmore et al. (2011) found the snowpack to represent a potentially important, and often overlooked, transient C store in boreal snow-covered catchments. Meltwater from the snowpack represented an important source of streamwater $CO_2$ in two forested peatland (drained and undrained) catchments in eastern Finland, contributing up to 49 % of total downstream $CO_2$ export during the snowmelt period in April/May.

### 17.4.4 Climate Impacts on Waterborne Losses from Forests and Wetlands

There are a number of possible effects of changing climatic conditions that may impact waterborne losses from catchments in northern Scandinavia. The two overarching factors are likely to be the amount and seasonality of precipitation and temperature. In the northern Baltic Sea catchment, the spring flood associated with peak snowmelt mainly occurs in mid-May for the Taiga zone and in June for the Fennoscandian mountains (Ingri et al. 2005), but the timing and magnitude of this main hydrologic event may shift under a changing climate. Korhonen and Kuusisto (2010) showed that although no overall changes have been observed in mean annual stream discharge for a large number of Finnish streams, hydrological regimes during winter and spring have changed significantly (see also Chap. 5). This is mainly attributed to winters and springs becoming milder and in consequence late-winter and early-spring discharges increasing. Studies combining several hydrological model simulations (to the end of this century) in the Swedish Regional Climate Modelling Programme show that while results varied depending on the climate change scenario and model boundary conditions, some projections were consistent between runs, for example an overall increased autumn and winter run-off and increased annual run-off volume in northern Sweden (Andreasson et al. 2004). In a similar study focusing on river run-off for the entire Baltic Sea drainage basin, Graham (2004) found a general trend of reduced river flow from the south-eastern Baltic Sea catchment together with increased river flow from the north.

The magnitude and timing of the spring flood is important for the annual fluxes of dissolved and particulate matter in boreal river systems (Woo et al. 2008). Higher spring temperatures and a higher proportion of winter precipitation falling as rain could result in earlier snowmelt and thinner snowpacks. This would lead to spring floods occurring earlier in the year and having a lower magnitude (Andreasson et al. 2004; Woo et al. 2008). However, in high-latitude catchments, sustained sub-zero winter temperatures coupled with increased precipitation may lead to maintained or even increased spring floods (Dankers and Middelkoop 2008). Rantakari et al. (2010) compared TOC fluxes in headwater streams between two climatically different years and found decreased TOC export during the spring ice melt period and increasing export during the rest of the year including snow-cover and snow-free periods. Wet years have been shown to favour the export of TOC from forest-dominated areas (Kohler et al. 2009) and imply that future wetter conditions may increase the TOC export as well as many elements associated with TOC such as Al, Fe, trace elements and potentially harmful elements such as Hg. Empirical models of streamwater fluxes of DOC including both soil temperature and water fluxes have been able to predict seasonal variation in streamwater DOC concentrations reasonably accurately showing that both parameters are important (Kohler et al. 2009). Simulations of a climatic scenario with an average temperature increase of about 2.5 °C and increase in precipitation of 25 % for a boreal headwater stream suggest an increase in the annual TOC export of approximately 15 % (Kohler et al. 2009). The model simulations also indicate that the autumn months are particularly sensitive and that wetter and warmer conditions could cause a TOC increase of up to 5 mg $l^{-1}$ (Kohler et al. 2009).

Temperature effects related to snow cover may also have profound effects on waterborne DOC export from forests (Agren et al. 2010). Higher temperature in organic soils has been shown to increase DOC export, not because of temperature control on production rates but because temperature affects C consumption and microbial activity (Moore and Dalva 2001; Pietikainen et al. 2005). Stedmon et al. (2006) found that seasonal DOM export patterns to a temperate Danish estuary reflected temperature fluctuations in more natural subdrainage areas, whereas precipitation controlled export patterns in sub-drainage areas dominated by agriculture. While increased soil temperature is likely to lead to enhanced export of DOC, it will not necessarily lead to increased fluvial export of nutrients from mire ecosystems (depending on the efficiency of internal nutrient cycling in mires). In a study of Alaskan boreal peatlands, water table depth and soil temperature were found to be significant factors influencing DOC and DON concentrations in streamwater, but it was also shown that bog peatlands retain N (D'Amore et al. 2010).

A snow manipulation experiment in northern Sweden showed that cold winters with a deeper seasonal freeze-thaw layer increased streamwater DOC concentrations during spring snowmelt (Haei et al. 2010). The experiment is interesting in that it shows the importance of winter climate conditions for streamwater DOC export. There is less information available on potential effects of climate change on the export of elements not directly associated with DOC. Concentrations of elements related to weathering normally show an inverse relationship to DOC in forested catchments (Smedberg et al. 2006). The highest concentrations are generally found during baseflow conditions with concentrations diluted during flow events. Potentially, increased weathering or changes in hydrological flow paths with a greater contribution of groundwater could lead to an increased export of elements that are dominant during baseflow conditions. Such changes have been reported from permafrost-affected areas where permafrost thaw is predicted to shift hydrological pathways from being surface water dominated to groundwater dominated (Frey and McClelland 2009). More groundwater formation indicates more weathering and increased weathering consumption of $CO_2$, because more $CO_2$ is transported to deeper groundwater flow depths due to permafrost thaw. This may increase the transport of DIC and the weathering sink of atmospheric $CO_2$, and thus constitute a negative feedback on atmospheric $CO_2$ due to climate change at northern latitudes (Smedberg et al. 2006; Lyon et al. 2009). Actually, permafrost thaw rates of 0.7–1.3 cm $year^{-1}$ have been reported for watersheds in northern Sweden (Lyon et al. 2010). For these tundra catchments, an even contribution of DOC and DIC to the net mass flux of C from the terrestrial environment to and through the surface water system has been suggested. Under future potential scenarios, there could be a corresponding increase in the flux of both DOC and DIC from this landscape due to increased advective travel times associated with deeper flow pathways (Lyon et al. 2009, 2010). Such a potential increase and this particular balance between DOC and DIC are crucial in understanding the relevant feedbacks between future climatic change effects and the hydrological and biogeochemical system. In the Baltic Sea basin, the spatial distribution of permafrost is essentially limited to high-alpine landscapes and sub-Arctic peatlands in Sweden and Finland (Christiansen et al. 2010). There is no quantitative estimate of how this projected permafrost thaw could affect fluvial transport to the northern Baltic Sea basin, but considering the limited spatial extent of permafrost the shifts in export of major ions, nutrients and organic matter are likely to be limited. Climate-related changes in the distribution of mires may also affect landscape fluxes of elements. Fennoscandian mire types can be widely divided into four types: raised bogs, aapa mires, blanket bogs and palsa mires (Pajunen 2005; Parviainen and Luoto 2007). Parviainen and Luoto (2007) investigated the climatic envelopes of these mire types and found that the distributional limits of aapa mires, palsa mires and raised bogs were primarily associated with mean annual air temperature, while blanket bogs were also largely dependent on high levels of precipitation. Only palsa mires were found to have a very narrow climate envelope indicating short-term sensitivity to climate change. Over the short term, climate change is thus unlikely to affect the spatial distribution of wetlands.

Changes in forest practices related to improved climatic growth conditions such as a longer growing season might also affect the biogeochemistry of forest stands. This could include changes in tree species, shorter times between plantation and harvest, or more intense forest management. However, all these changes will only take effect over the long term since forest growth is still relatively slow at these latitudes. The effects of climate drivers on terrestrial ecosystems were reviewed in the first assessment of climate change in the Baltic Sea basin, and it was concluded that *It is apparent that trees are growing taller and lusher compared with a few decades ago; net primary productivity has increased, the ecosystems are net carbon sinks. The likely cause is an extended growing season associated with higher average temperatures* (Smith et al. 2008). An overall increase in forest productivity may, however, have a more direct effect since it may increase litter deposition and change the flow of current photosynthates to the soil (Olsson et al. 2005). Whether this also has implications for waterborne export to aquatic ecosystems is still unclear.

### 17.4.5 Current and Future Export to the Baltic Sea

Most studies investigating the effects of geochemical fluxes in relation to landscape properties such as forest or wetlands or forest practices have mainly focused on smaller catchments or headwater streams. The implications at a larger scale may be less pronounced. For instance, Futter et al. (2010) estimated the contribution of short-term increases in nitrate leaching following stem-only harvesting at a larger scale and suggested that this effect accounted for about 3 % of the overall Swedish N load to the Baltic Sea, despite the fact that short-term increases in nitrate leaching can be very pronounced in headwater streams after harvesting (Rosen et al. 1996). Model simulations of N and P fluxes from Swedish forest land to the marine environment suggest that over 93 % of the leaching losses can be attributed to background loads and the small remainder to forest practices (Brandt and Rapp 2008). Similarly, forestry in Finnish river basins was estimated to contribute 9 % of the total N export to the Baltic Sea on average (Lepisto et al. 2006). Kenttämies (2006) estimated the loading of P and N from forestry to be 8 and 5 %, respectively, of the total anthropogenic load in Finland. A major reason for the relatively minor contribution of forest practice to large-scale losses is that the extent of forested land area that is annually affected by forest practices is relatively low; in Finland, about 2.5 % of the entire country (Kortelainen and Saukkonen 1998). Lepistö et al. (1995) showed that within both Swedish and Finnish headwater catchments large-scale forest management practices were needed before any clear effect on spatial variability of N leaching could be detected. Although forestry is often the largest-scale human impact in headwater streams, it is thus reasonable to assume that any effect on element fluxes to the Baltic Sea related to forest management can only relate to large-scale change in forest practices.

Climate is the overall factor that might impact waterborne fluxes of elements. Although several studies from forested headwater streams indicate that, for instance, change in winter conditions affect DOC losses, it still remains to be shown that climate change will also have large-scale impacts in large catchments flowing to the Baltic Sea. Over the coming century, climate change is unlikely to affect the spatial distribution of wetlands in the Baltic Sea basin. In general, however, increased ambient and soil temperatures (including permafrost thaw) and increased precipitation could lead to increased DOC export from wetlands, especially during baseflow conditions. This increase is likely to be more pronounced in the boreal/sub-Arctic regions of the Baltic Sea catchment. In fact, recent scenario studies indicate that DOC production from terrestrial vegetation, modelled by the LPJ-GUESS ecosystem model, could increase by 30–43 % (Omstedt et al. 2012). Rising temperatures causing an increase in net ecosystem production, increasing both the available substrate and the rate of decomposition of plant biomass derived organic matter, provided the most important explanation for the increase in DOC export from wetlands and forests. In the scenario calculations of riverine fluxes, DOC fluxes to the Baltic Sea generally increased, especially in the northern catchments, in the range of 20–50 %, with the greatest increase in the Gulf of Finland. The increasing fluxes resulted mainly from the increasing run-off, since modelled concentration changes in river water were about 10 % (Omstedt et al. 2012).

## 17.5 Eutrophication and Freshwater Biogeochemistry

### 17.5.1 Agriculture, Urban Areas and Waterborne Fluxes

Agriculture dominates the southern and eastern areas of the Baltic Sea drainage basin (Fig. 17.1; Table 17.1) and covers from 7 % of total land area in Finland and Sweden to 60 % in Denmark, Table 17.2. Agricultural production is intensive in large parts of Denmark, Germany, Poland, and southern Sweden and Finland (Table 17.2), whereas more extensive agricultural production is seen in the Baltic States. The agricultural structure differs markedly between countries: in Poland, 70 % of the area is owned by small farms of less than 10 hectares, whereas in Denmark most farms are larger than 100 hectares (FAO 2003; Benoist and Marquer 2006a, b, c, d, e, f, g; see also Chap. 21).

Urban areas and industry which are also concentrated in the southern and eastern parts of the Baltic Sea catchment (Table 17.1) are major point sources for nutrients. Although several countries have already implemented effective wastewater treatment plants (Table 17.3), poor wastewater treatment is still an issue in rural areas and effective sewage treatment would still have a significant potential in the transitional countries (Humborg et al. 2007; Iital et al. 2010b).

The most recent compilation of nutrient loads to the Baltic Sea reports loadings via rivers and coastal point sources of respectively 638,000 t TN and 28,370 t TP (HELCOM 2011, see Table 17.4). The rivers Vistula (Poland), Nemunas (Belarus, Lithuania and Russia), Oder (Poland and Germany) and Neva (Russia) all drain the southern cultivated part of the Baltic Sea catchment and account for the majority of the nutrient inputs to the Baltic Sea (HELCOM 2004). Many rivers of northern Sweden and Finland are still largely unperturbed by human activities (Humborg et al. 2003). The difference between the northern boreal and southern cultivated systems is illustrated by comparing the Kemijoki and Oder rivers. The Kemijoki drains the northern part of Finland and discharges into the Gulf of Bothnia with average concentrations of 0.4 mg TN $l^{-1}$ and 0.02 mg TP $l^{-1}$ for the period 1994–2008. The Oder, draining a large part of Poland, has TN and TP concentrations 10-fold higher (HELCOM 2011). Source apportionment for the riverine nutrient loading of the Baltic Sea in 2000 indicates that TN natural background losses

**Table 17.2** Agricultural land subdivided by crop type, average consumption of fertiliser and manure, and livestock density in countries within the Baltic Sea drainage basin in 2005 (Benoist and Marquer 2006a, b, c, d, e, f, g) including Belarus and Russia (Sweitzer et al. 1996)

| | Denmark | Sweden | Finland | Poland | Lithuania | Latvia | Russia[a] | Belarus[a] | Estonia |
|---|---|---|---|---|---|---|---|---|---|
| Agricultural area, 1000 ha | 2588 | 3096 | 2262 | 13,132 | 2338 | 1302 | 4803 | 4081 | 764 |
| Share of total land cover (%) | 60 | 7 | 7 | 42 | 36 | 20 | 15 | 45 | 17 |
| *Cover of agricultural area (%)* | | | | | | | | | |
| Arable land | 69 | 41 | 60 | 72 | 45 | 42 | 31 | 58 | 72 |
| Forage plants | 17 | 33 | 28 | 6 | 26 | 23 | | 21 | 6 |
| Permanent grass and meadows | 7 | 15 | 1 | 19 | 26 | 28 | 69[b] | 21 | 19 |
| Fallow | 7 | 10 | 11 | 1 | 3 | 6 | | 0 | 1 |
| *Consumption in 2005,* kg N $ha^{-1}$ *agricultural area* | | | | | | | | | |
| Mineral fertiliser | 74.0 | 51.0 | 74.2 | 68.2 | Nd | 25.9 | Nd | 55[c] | 26.3 |
| Manure[d] | 72.2 | 25.4[e] | 42.9 | 38.3 | 27.2 | 17.9 | Nd | 42[c] | 25.7 |
| *Livestock, head* $ha^{-1}$ *agricultural area* | | | | | | | | | |
| Dairy cows | 0.14 | 0.13 | 0.14 | 0.21 | 0.18 | 0.14 | Nd | 0.22 | 0.15 |
| Pigs | 3.68 | 0.58 | 0.64 | 1.42 | 0.48 | 0.33 | Nd | 0.46 | 0.46 |

For Denmark, the area that drains into the North Sea is also included
[a] Agricultural area for that part of Belarus which drains into the Baltic Sea (Sweitzer et al. 1996)
[b] Cover of forage plants, permanent grass and meadows
[c] Consumption of fertiliser and manure is average numbers from National Statistic of Belarus (2010)
[d] $NH_3$ volatilisation from storage and stable excluded
[e] Input to the soil–ammonia ($NH_3$) volatilisation excluded

**Table 17.3** Levels of sewage treatment by country in 2004 (Humborg et al. 2007)

| Country | Primary (%) | Secondary (%) | Tertiary (%) |
|---|---|---|---|
| Belarus | 0 | 50 | 0 |
| Czech Republic | 0 | 61 | 0 |
| Denmark | 2 | 5 | 81 |
| Estonia | 2 | 34 | 34 |
| Finland | 0 | 0 | 80 |
| Germany | 0 | 9 | 85 |
| Lithuania | 33 | 6 | 18 |
| Latvia | 2 | 35 | 33 |
| Poland | 2 | 23 | 34 |
| Russia | 0 | 50 | 0 |
| Sweden | 0 | 6 | 86 |

Primary treatment (lowest effect) is mainly through sedimentation, while secondary and tertiary treatments (highest effect) are mainly through biological filtration and chemical precipitation

**Table 17.4** Total nutrient load by country via rivers and coastal point sources for 2006 according to data reported to HELCOM (HELCOM 2011)

| Country | Total nitrogen load (t $year^{-1}$) | % | Total phosphorus load (t $year^{-1}$) | % |
|---|---|---|---|---|
| Germany | 16,900 | 3 | 490 | 2 |
| Denmark | 53,000 | 8 | 1520 | 5 |
| Estonia | 20,400 | 3 | 790 | 3 |
| Finland | 79,000 | 12 | 3490 | 12 |
| Lithuania | 28,000 | 4 | 1240 | 4 |
| Latvia | 59,500 | 9 | 2800 | 10 |
| Russia | 107,600 | 17 | 4070 | 14 |
| Poland | 152,600 | 24 | 10,240 | 36 |
| Sweden | 121,000 | 19 | 3730 | 13 |
| Total | 638,000 | 100 | 28,370 | 100 |

account for 28 %, diffuse losses for 64 %, and point source discharges for 8 %. For TP, the contributions are 26, 55 and 19 %, respectively (HELCOM 2004). Agriculture accounts for the majority of the diffuse losses: 70–90 % for TN and 60–80 % for TP (HELCOM 2011).

#### 17.5.1.1 Long-Term Trends in Land Use and Nutrient Loads

The N and P content in rivers increased steadily through the twentieth century with the highest concentrations measured during the 1980s and 1990s. Even 100 years ago, the impact from human activities on nutrient losses was substantial. Natural fertilisers were used in agriculture, and the construction of water supply and sewage systems increased the output of waste to inland and coastal waters (Savchuk et al. 2008). In addition, the widespread draining of lakes and wetlands which led to a loss of nutrient retention capacity mainly occurred at the end of the nineteenth century and in the first decades of the twentieth century (Hoffmann et al. 2000; Schernewski and Neumann 2005). Using a modelling approach, Schernewski and Neumann (2005) calculated that the overall nutrient loads to the Baltic Sea have increased by

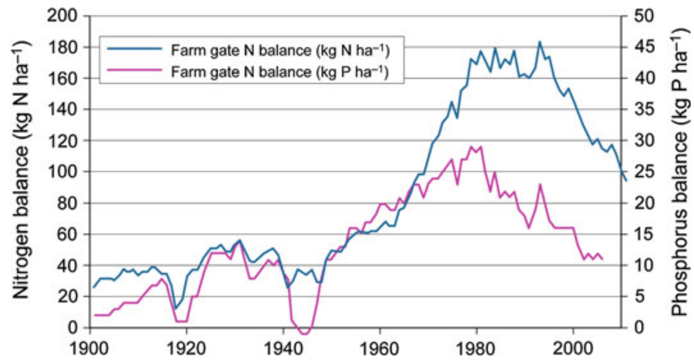



**Fig. 17.5** The farm gate nitrogen (N) and phosphorus (P) balance for Denmark over the period 1900–2010 (Kyllingsbæek and Hansen 2007; Kyllingsbæk 2008, redrawn)

a factor of 2.4 (TN) and 3.1 (TP) over the past 100 years. Savchuk et al. (2008) reconstructed external nutrient inputs from various literature and data sources and estimated a similar increase over the past century: factors of 2 (TN) and 3 (TP). Gadegast et al. (2012) and Behrendt et al. (2008) using the Moneris model found TN loadings from the Oder system increased by a factor of 4.6 between 1880 and 1980.

The nutrient surplus, the balance between the application and harvest offtake of N or P on agricultural land, is an indicator for nutrient losses from agriculture. Figure 17.5 illustrates the dramatic intensification of Danish agriculture, especially following the Second World War. The graphic also shows the successful regulation of nutrient surplus in Danish agriculture over the past three decades. For a detailed assessment of long-term change in land use see Chap. 25.

In Europe, change in land use over the past few centuries has been characterised by alternating expansion and contraction of agricultural areas. Expansion was a consequence of increasing food demand caused by a growing population. By the end of the seventeenth century, owing to better agricultural methods (new rotation systems and manure use), productivity increased and so a period of contracting cultivated area began. This lasted until the mid-eighteenth century when continued population growth meant the agricultural area again expanded, continuing for the next 200 years. Since the late 1950s, the agricultural area has been relatively constant, although productivity has still increased owing to new agricultural methods and technological improvements (Rabbinge and Vanlatesteijn 1992; Rabbinge and van Diepen 2000).

Land use changes in the southern Baltic Sea catchment over the past 200 years are comparable to the developments in Europe and are strongly influenced by the industrialisation and fundamental socio-economic transformation after the Second World War and after the European socialist period. However, there are no studies on land use changes during the past 100 years for the Baltic Sea basin as a whole. For the southern, cultivated part of the basin, however, existing studies (Behrendt et al. 2008; Gadegast et al. 2012) of the Oder catchment may typify trends: land use was relatively constant before 1950. Agriculture and forests occupied about 66 and 25 % of the land area, respectively. Due to population growth, the urban areas increased slightly to 4 %. After the Second World War, a decreasing trend in agricultural land use began, and by the end of the communist era in Poland, agriculture accounted for 59 % of the land area. Areas with high yield were used more intensively, and the derelict land was converted to forest. Forested areas thus increased from 22 % in 1950 to 28 % in 1990 (Hirschfeld et al. 2009; Fig. 17.6). Following the transition from a state-

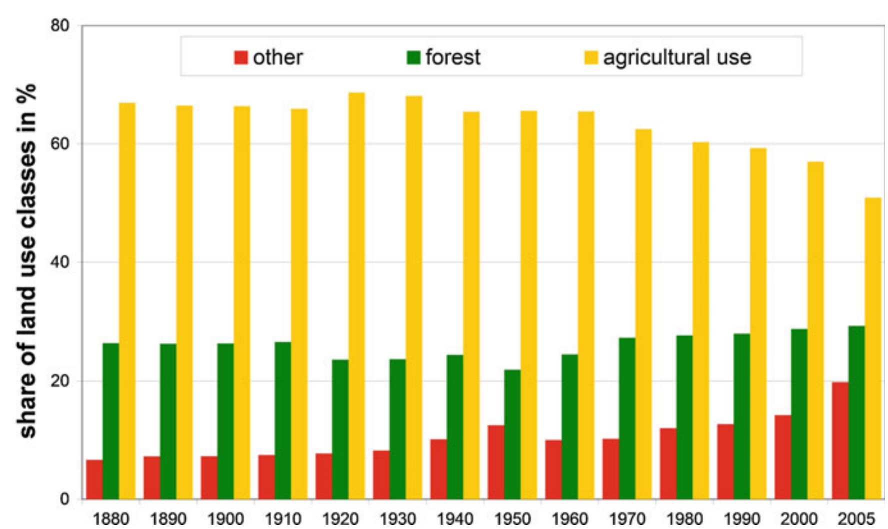



**Fig. 17.6** Land use in the Oder River catchment over 1880–1940 (Gadegast et al. 2012) and for the whole of Poland over 1950–2005 (Hirschfeld et al. 2009)

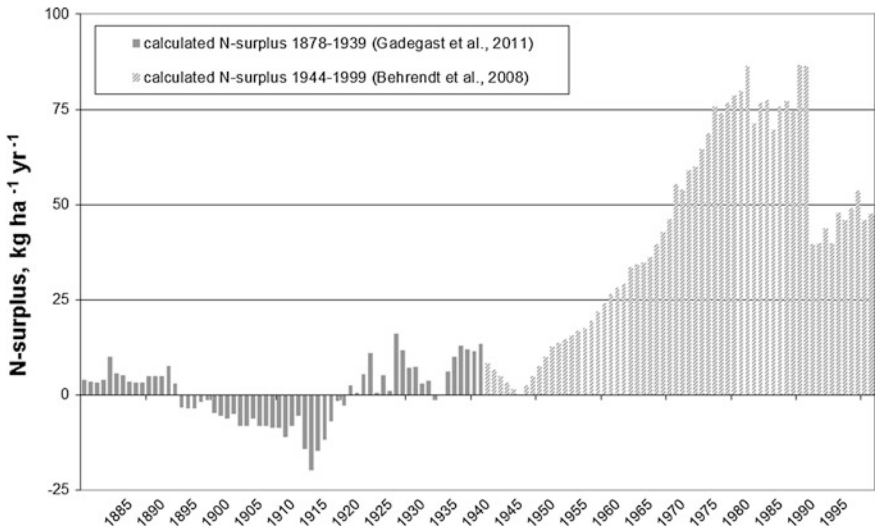



**Fig. 17.7** Nitrogen (N) surplus on agricultural areas in the Oder River catchment, 1878–1939 (Gadegast et al. 2012) and 1944–1999 (Behrendt et al. 2008)

controlled to a free-market economy, agriculture experienced a recession. However, after Poland and the Czech Republic joined the EU in 2004, agricultural land use has again intensified.

The long-term N surplus in the Oder River catchment is shown in Fig. 17.7 for 1878–1939 (Gadegast et al. 2012) and 1944–1999 (Behrendt et al. 2008). Up to 1890, the mean N surplus was around 5 kg N ha$^{-1}$ year$^{-1}$. Between 1890 and 1920, the N balance was negative for three reasons: higher crop yields and therefore a higher N uptake; use of human excrement as an organic fertilizer ceased; and a reduced input of mineral fertiliser. During the First World War especially, blockades prevented the import of products such as 'Chile saltpetre' and 'Guano'. Since 1920, the N surplus has increased. This is due to the invention of the Haber–Bosch process which enables the manufacture of inorganic fertiliser. The average application of mineral fertiliser on agricultural areas in the Oder River system increased from 9 kg ha$^{-1}$ year$^{-1}$ (1921) to 26 kg ha$^{-1}$ year$^{-1}$ (1938/39). After the end of the Second World War, N surpluses increased constantly due to intensification of agricultural production and increased use of mineral fertiliser until the collapse of agriculture in Poland and the Czech Republic in 1989.

Long-term change in DIN loads entering the Baltic Sea via the Oder River is shown in Fig. 17.8. DIN loads increased initially by about 43 % mainly due to growth in the urban population. Waste water treatment became necessary, and nutrient release from point sources such as sewer systems and waste water treatment plants increased. The increasing deposition of nitrogen oxides ($NO_X$) and $NH_Y$ from the atmosphere due to growing industrialisation and the increasing N surplus in agriculture since 1920 influenced DIN loads to the Baltic Sea. The increase in DIN loads continued, almost tripling during 1955 to 1980 with a

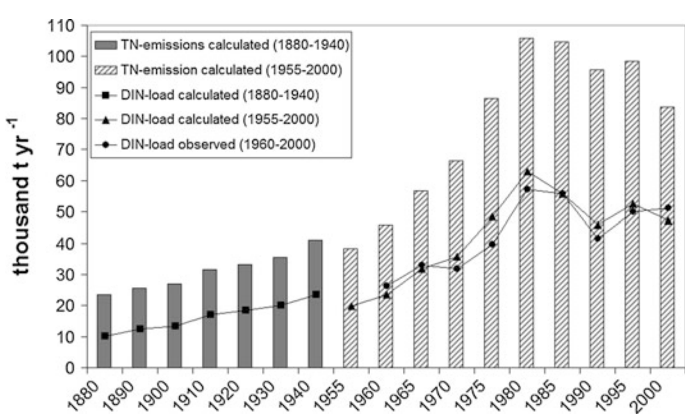



**Fig. 17.8** Long-term change (1880–2000) in total nitrogen (TN) emissions to surface waters and in the calculated and measured loads of dissolved inorganic nitrogen (DIN) in the Oder River at the station Krajnik Dolny. 1880–1940 (Gadegast et al. 2012) and 1955–2000 (Behrendt et al. 2008; Venohr et al. 2010)

maximum of 63,000 t year$^{-1}$. DIN loads decreased around 1990 due to reduced N surpluses following the collapse of agriculture in Poland and the Czech Republic. From 1990 to 2000, DIN loads showed a slight increase as a result of a revitalisation of agriculture (Behrendt et al. 2008; Venohr et al. 2010).

#### 17.5.1.2 Recent Trends in Nutrient Loads

Decreasing trends in riverine nutrient concentrations or loads have been reported. Raike et al. (2003) observed decreasing P concentrations in Finnish rivers and lakes formerly heavily polluted by industrial and municipal discharges. Similar observations were made in Estonia (Iital et al. 2010a, 2010b), Latvia (Stålnacke et al. 2003) and for the Nemunas river (Sileika et al. 2006) where the dissolved inorganic phosphorus (DIP) load decreased 31–86 % between 1986–1991 and 1997–2002. In Denmark, the TN load from point sources has reduced by 74 % since 1985, and the TN loads in 86 streams draining smaller agricultural catchments by 32 % (Kronvang et al. 2008). The decreases in TN concentrations in Estonian rivers relate to substantial reductions in fertiliser use, decreased agricultural land area, decreased point source load and increased self-purification capacity of soil water systems (Iital et al. 2010b). However, increasing TN concentrations are also being observed in some rivers due to higher diffuse loading, for example Finnish rivers (Raike et al. 2003) and the Nemunas river (Sileika et al. 2006). Overall, a trend analysis for the Baltic Sea on total waterborne annual loads (riverine loads plus direct coastal loads) indicates an increase in TN loads from 1994 to 2008, although this was not statistically significant, and a significant decrease (442 t year$^{-1}$) for TP loads (HELCOM 2011).

#### 17.5.1.3 Agriculture and Weathering

Agricultural activities affect chemical weathering and with this probably the release of the nutrient Si from minerals in several ways. First, practices such as tilling and changes in land use alter chemical weathering fluxes positively (Paces 1983; Pierson-Wickmann 2009). An additional driver of increased chemical weathering fluxes is the increasing application of mineral fertilisers, specifically lime or carbonates (Tilman et al. 2001). Evidence for long-term (decades) trends in DIC fluxes have been identified for the Mississippi catchment, but to date have not been observed in the Baltic Sea area. Agricultural acidification, for example, associated with the application of N fertilisers may have increased fluxes of cations and decreased DIC fluxes in the past (Semhi et al. 2000; Perrin et al. 2008; Pierson-Wickmann et al. 2009). However, no studies have been identified showing this for the Baltic Sea area in detail. Mineral fertilisation increases crop production and the C-pool in plants (Ma and Takahashi 1990; Alvarez and Datnoff 2001). A hypothetical increase in terrestrial C-pools of standing stocks may contribute to an increased particulate organic carbon (POC) efflux to the Baltic Sea, specifically if precipitation patterns change towards those enhancing soil erosion. However, no data are known which justify this hypothesis. Moreover, there is no large-scale estimate on how agriculture has affected the fluxes of biogenic elements from sulphidic soils in the boreal watersheds of Sweden and Finland. On the coastal plains of Finland, approximately 3000 km$^2$ of acid sulphate soils have developed as a result of intensive agricultural drainage of waterlogged sulphide-bearing sediments (Åström et al. 2007).

### 17.5.2 Transformations of Nutrients Along the Aquatic Continuum

Retention is the permanent removal or temporary storage of nutrients and other biogenic elements within a system (von Schiller et al. 2008). Depending on the hydrological pathways along which biogenic elements are routed through the catchment, retention processes may significantly alter the concentration of these elements before they reach the marine recipient (Stålnacke et al. 2003).

In the terrestrial part of the hydrological cycle, retention processes include deposition of eroded soil and associated biogenic elements such as in buffer zones (Uusi-Kämppä 2006; Pärn et al. 2011), sequestration of C and nutrients into the organic soil pool (Lal et al. 2011), adsorption of dissolved P onto inorganic soil constituents (Litaor et al. 2003) and denitrification: a microbial dissimilation process in which dissolved nitrate is reduced to gaseous forms of N (Seitzinger 1988). Several studies have demonstrated very high N removal rates and high efficiency (up to 100 % removal) due to denitrification in groundwater-fed wetlands and wetlands subject to overland flow (Haycock and Pinay 1993; Sabater et al. 2003). N retention in groundwater is strongly dependent on hydraulic residence time. For 17 Danish catchments, Andersen et al. (2001) reported groundwater retention of 20–80 % along a gradient of increasing retention time. Minerogenic soils in general have a high affinity for P, and excess P is usually strongly sorbed in soils until a critical degree of saturation is reached (Hooda et al. 2000). Most north-western European countries, however, experience a net input of P to agricultural land (Leinweber et al. 2002), which makes the soil more vulnerable to P losses via erosion and leaching (Sharpley and Rekolainen 1997). As an example, the average P content in Danish agricultural soils increased from 3200 to 4600 kg P ha$^{-1}$ between 1900 and 2000 due to the intensification of agriculture and increased use of fertilisers (Rubæk et al. 2005).

In the aquatic part of the hydrological cycle, that is river systems including rivers, lakes, riparian areas and floodplains, N retention processes include biotic assimilation, denitrification and sorption (Herrman et al. 2008). Recent studies on agricultural streams with high nitrate concentrations indicate that in-stream nitrate removal may not increase proportionately with nitrate availability due to nitrate saturation of the microbial community responsible for denitrification (Bernot et al. 2006; Herrman et al. 2008). The study by Herrman et al. (2008) suggested a concentration of 2 mg $NO_3$–N $l^{-1}$ as a threshold, above which the streams become increasingly saturated with nitrate and export substantial N. Other research has identified hydraulic retention time (in-stream water residence time) as a key stream characteristic controlling N removal (Valett et al. 1996). In lakes, permanent N retention occurs as denitrification but also by incorporation in sedimenting organic matter that is permanently buried on the lake bottom (Søndergaard 2007). As for streams, hydraulic retention time is found to be a key factor controlling N retention in lakes (Windolf et al. 1996; Søndergaard 2007; Herrman et al. 2008). On average for 69 Danish lakes, 43 % of the N input was permanently retained (Jensen et al. 1990). Based on data reported by contracting countries, HELCOM (2004) estimated a 30 % N retention of the gross load entering river systems in the Baltic Sea drainage basin.

Retention processes for P in rivers, riparian areas and floodplains comprise sorption to suspended solids and bottom sediment, deposition of particulate matter on the river bed, biotic assimilation by algae and macrophytes, and sedimentation on inundated riparian areas and floodplains (Kronvang et al. 1999; Pärn et al. 2011). Storage of P within the river is often considered a temporary sink only, as the P build-up in biomass and sediment during summer is flushed out during high winter flows (De Witt 1999; Schulz et al. 2003). Permanent in-stream retention was explained by Svendsen et al. (1995) as sorption of DIP to Fe and Al oxides and hydroxides. In-stream net retention is probably of minor importance (Vassiljev and Stålnacke 2005), whereas P storage by sedimentation on inundated riparian areas and floodplains can be considerable; rates of up to 127 kg P $ha^{-1}$ $year^{-1}$ are reported (review by Hoffmann et al. 2009). In lakes, P retention occurs via sedimentation of particulate-bound forms or via uptake and incorporation of dissolved P by plants and subsequent sedimentation (Søndergaard 2007). In the drainage areas of entire river systems, P retention in lakes is often considered the most important permanent sink (Svendsen et al. 1995; Vassiljev and Stålnacke 2005). Kronvang et al. (1999) found an average retention of 3 kg P $ha^{-1}$ $year^{-1}$ for 18 shallow Danish lakes. Vassiljev and Stålnacke (2005) using a model approach estimated lake P retention to be around 30–35 % in the 44,000 $km^2$ Lake Peipsi catchment. However, the retention rate in some lakes is currently negative due to high internal P loading from the sediment following a reduction in the external load (Jeppesen et al. 1999). For the Baltic Sea drainage basin, HELCOM (2004) estimated that on average 31 % of the gross P load to river systems is retained.

Retention is not constant over time. Hoffmann et al. (2000) estimated that in Sweden a retention capacity of 30,000 t N has been lost since 1865 due to extensive drainage of wetlands and lakes. A similar loss of nutrient retention capacity is seen in other cultivated areas of the drainage basin (Brookes 1987; Andersen and Svendsen 1997). Over the past 30 years, recreation of the natural nutrient retention capacity of river systems by remeandering of streams allowing temporary inundation of riparian areas and floodplains, and recreation of wetlands and lakes have attracted much attention as means to reduce diffuse nutrient loading (Cooke et al. 1993; Hoffmann et al. 2011).

### 17.5.3 Climate Impacts on Waterborne Losses from Agriculture and Urban Areas

Rising temperatures have prolonged the growing season by 4 weeks or more across the southern part of the Baltic Sea drainage basin since 1982 (Høgda et al. 2007). There have been more droughts in large parts of western and eastern Europe (Trenberth et al. 2007); however, in Denmark, precipitation has increased by 100 mm over the past 50 years (Larsen et al. 2005). Rainfall intensity has increased (Frich et al. 2002), and this has led to severe summer flooding in Europe (Christensen and Christensen 2003). Analysis of data from 17 catchments across Europe (Bouraoui et al. 2009) revealed that climatic variables and in particular total precipitation explained most of the variance found in the nutrient load measured at the catchments outlet. DIN concentration was mainly controlled by the extent of the agricultural area, whereas P concentration was mostly controlled by precipitation intensity (calculated here as the amount of rainfall during a rainy day) and population density. Farmers are currently adapting to climate change, particularly in terms of changing the timing of cultivation and selecting other crop species and cultivars (Olesen et al. 2011). For example, the cultivation of maize, a heat-demanding, warm-season crop, is expanding northwards and increased by a factor of 9 in Denmark from 1990 to 2010 (Fig. 17.9).

The projected climate change—higher temperatures, greater precipitation in some areas, and more extreme events—will affect all hydrologic pathways for biogenic elements and thus loading to the Baltic Sea.

Agro-ecosystems are strongly affected by environmental conditions and thus by climate change. Plants respond to rising atmospheric $CO_2$ concentration by increasing resource use efficiencies for radiation, water and N (Olesen and Bindi 2002), which reduces the risk of N leaching. In experimental studies, Downing et al. (2000) showed a wheat grain yield

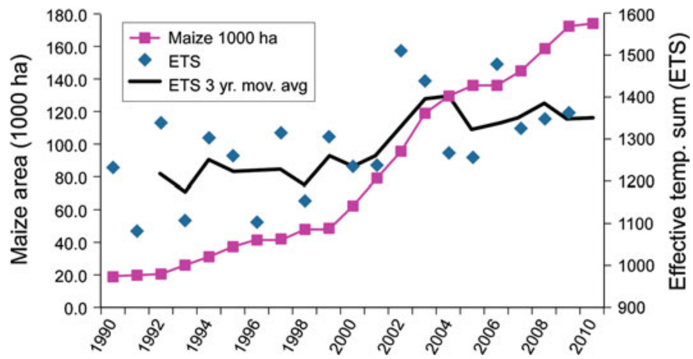



**Fig. 17.9** Area with maize in Denmark and effective temperature sum (*ETS*). ETS is calculated as the sum of daily mean temperatures above 6 °C from 15 April to 30 September. Maize requires an ETS above 1200 °C (redrawn from Olesen 2008)

increase of 28 % for a doubling of current $CO_2$ concentration. However, increased temperature reduces crop duration for many annual crops and hence yields. This could lead to a further expansion of warm-season crops (e.g. maize and sunflower) into areas currently dominated by small-grain cereals and oilseed crops (Olesen and Bindi 2002). Earlier harvest of crops and later planting of winter crops may result in a prolonged period of bare soil in autumn (Jeppesen et al. 2011), which would increase the risk of nutrient loss by leaching and surface loss processes. In addition, soil organic matter turnover would increase under higher temperatures, which would increase the risk of leaching of nutrients and DOC (Patil et al. 2010), particularly in connection with heavy precipitation (Eckersten et al. 2001). Soil erosion and surface run-off are also expected to increase (Michael et al. 2005; Nearing et al. 2005) and thus loss of nutrients and other biogeneic elements to river systems.

The current intensive agricultural production of cereals in most Danish riparian areas is becoming increasingly difficult to sustain due to increased autumn and winter precipitation (Jeppesen et al. 2011). The problems could increase considerably under the projected climate change, and in many riparian areas, intensive agricultural production may cease in the future (Andersen et al. 2006). Abandoning cultivation and artificial drainage of riparian areas would decrease nutrient losses and aid C sequestration (Lal et al. 2011).

### 17.5.4 Current and Future Export to the Baltic Sea

As temperature is one of the parameters controlling denitrification, the rate of denitrification in soils, streams, wetlands and lakes will increase with increasing temperature (Veraart et al. 2011). Using a nutrient accounting approach covering all major watersheds of the Baltic Sea basin, Hong et al. (2012) found that the fraction of net anthropogenic N and P inputs exported as riverine fluxes decreased with increasing temperature (Fig. 17.10), that is the overall watershed nutrient retention which is about 70 % for all anthropogenic N and about 90 % for all anthropogenic P increased with temperature. On the other hand, Hong et al. (2012) found the fraction of net anthropogenic N and P inputs exported as riverine fluxes increased with river discharge, and that is projected to increase in future (Graham and Bergstrom 2000). Similar observations were found by Howarth et al. (2012) using a global dataset of 154 watersheds, of which 36 were in Sweden. Thus, the effect of higher nutrient retention in a warmer climate may be counteracted by a shorter hydraulic retention time following increased percolation through soils and run-off through river systems (Jeppesen et al. 2011). Increased run-off during winter and more extreme rainfall events will affect the interaction between streams and their riparian areas. Andersen et al. (2006) projected a 50 % increase in the number of days with overbank flooding by a lowland river by 2071–2100 under the IPCC A2 SRES scenario, relative to the control period (1961–1990). Because the P deposition rate increases with the magnitude of the inundation event (Kronvang et al. 2007), P retention may increase in a warmer climate. In lakes, a temperature-mediated release of Fe-bound P from the sediment has been observed (Jeppesen et al. 2009). Longer stagnation periods in lakes could increase anoxia and sediment P release. Nutrient depletion in the epilimnion in summer can shift phytoplankton blooms towards autumn when a deepening thermocline allows nutrient inputs from deeper layers (Kangro et al. 2005). As a result, the internal P loading of lakes may increase and the net P retention in lakes may decrease in a warmer climate.

To date, there are few large-scale studies evaluating possible future N loadings from the Baltic Sea catchment area as a whole. Using a multivariate statistical approach, Eriksson-Hägg et al. (2012) estimated that the net effect of changes in climate (i.e. hydrology) and in animal protein consumption may be a possible increase in TN flux ranging from 3 to 72 % (Fig. 17.11) between the major watersheds,

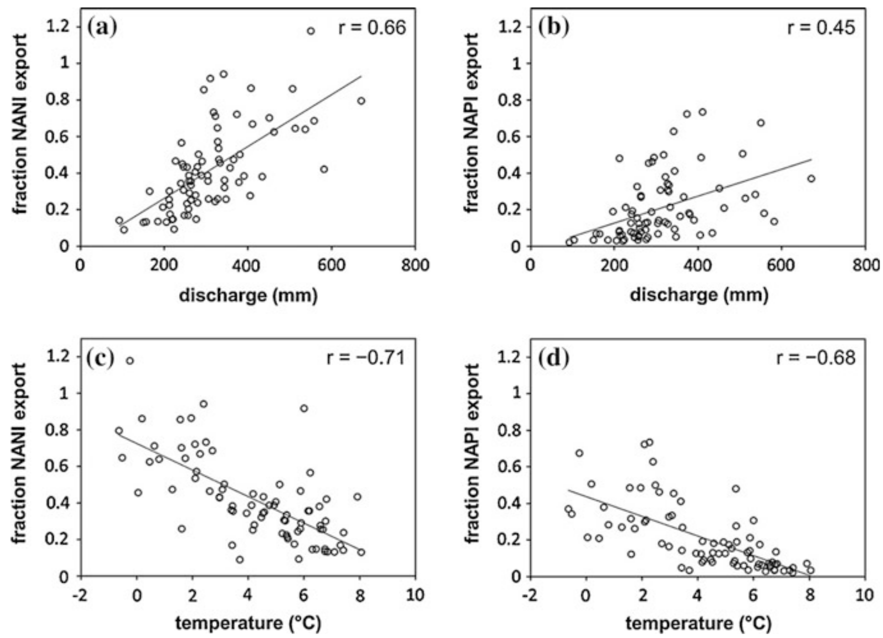



**Fig. 17.10** Fraction of net anthropogenic nitrogen inputs (NANI) and net anthropogenic phosphorus inputs (NAPI) exported as riverine fluxes controlled by discharge and temperature (Hong et al. 2012)

on average 20–30 %. Overall, the authors concluded that demand for animal protein will be fundamental to controlling nutrient loading in the Baltic Sea ecosystem and may be a major block to fulfilling the environmental goals of the Baltic Sea Action Plan.

## 17.6 Hydrological Alterations and Freshwater Biogeochemistry

### 17.6.1 Hydrological Alterations and Waterborne Fluxes

Many rivers in the Baltic Sea catchments have been regulated. In general, the boreal/sub-Arctic rivers entering the Gulf of Bothnia have a higher specific run-off, especially the northern Swedish rivers which have a steeper catchment slope compared to the rivers draining the south-eastern catchments of the Baltic Sea. Damming is therefore more frequent in the boreal rivers owing to its higher effectiveness in terms of power generation; major reservoirs located in the headwaters can hold 30–70 % of their annual water discharge (Dynesius and Nilsson 1994). In contrast, damming is less frequent in the lowland rivers of the south-eastern catchment of the Baltic Sea, and minor dams and reservoirs with short water residence times were mostly built there. Studies in major global rivers have demonstrated that river regulation and damming lead to decreased nutrient transport to coastal seas, although the mechanisms vary (Humborg et al. 2000). The primary mechanism responsible for the decrease in nutrient loads was initially thought to be the trapping and subsequent sediment burial of nutrients in the form of diatoms that were autochthonously produced in the reservoirs due to decreasing water currents and improved light conditions, termed the 'artificial lake effect' (Van Bennekom and Salomons 1981). Most studies focussed on dissolved Si (DSi) fluxes, because this nutrient removal might be overcompensated by anthropogenic N and P inputs downstream of the reservoir and from other sources, and no such compensation occurs for DSi (Fig. 17.12). Thus, river regulation and damming lead mainly to lower N:Si ratios in the receiving coastal water bodies favouring non-siliceous phytoplankton blooms (Humborg et al. 1997; Dai et al. 2011).

Initial studies from the early 1990s comparing unregulated and regulated boreal rivers in the Baltic Sea catchment showed that overall the element concentrations (i.e. base cations and anions, trace metals and nutrients) are much lower in regulated rivers than unregulated rivers (Brydsten et al. 1990). From later studies in boreal and sub-Arctic Swedish watersheds, it was hypothesised that perturbed surface water–groundwater interactions as a consequence of hydrological alterations in the rivers of the northern Baltic Sea watershed lead to changes in weathering conditions. Thus, less weathering may be the major reason for the reduced DSi loads observed in these oligotrophic boreal rivers (Humborg et al. 2000, 2002, 2006). Particle trapping of biogenic Si (BSi)—mainly diatom shells—behind dams is the main reason for the reduced Si loads in the cultivated watersheds of the southern catchment area of the Baltic Sea (Humborg et al. 2006).

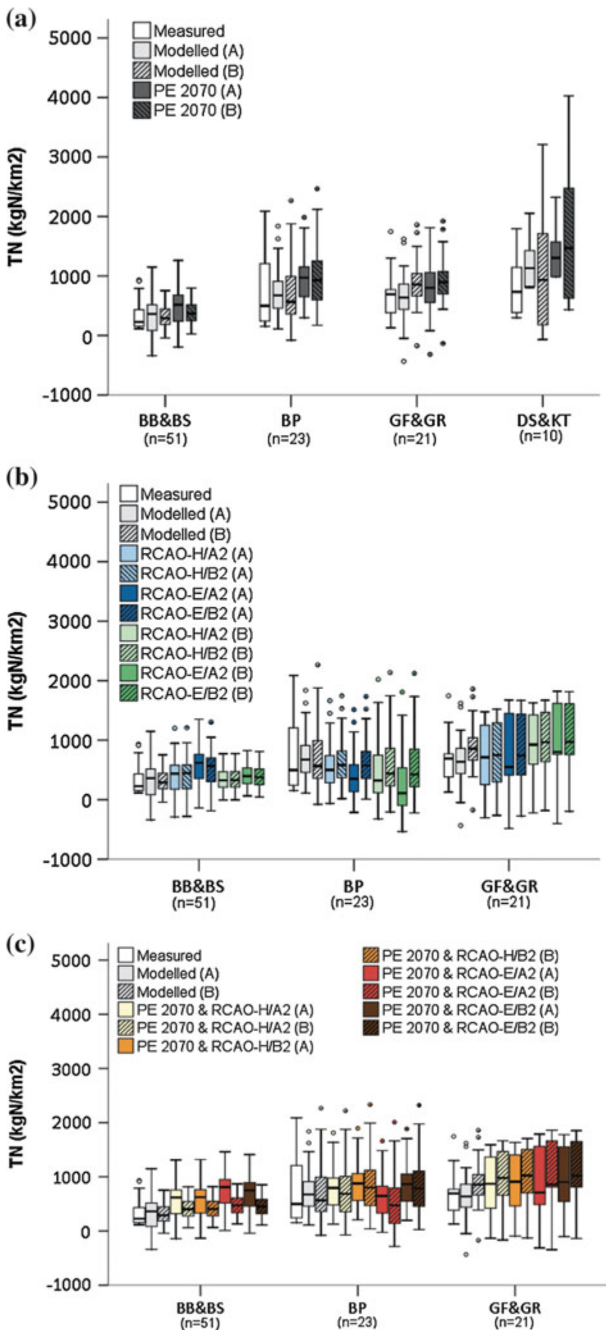



**Fig. 17.11** Box and whisker plots with measured (mean 1992–1996), modelled (1992–1996) and projected future total nitrogen (TN) fluxes (kg N km$^{-2}$ year$^{-1}$) for the four sub-basins: Bothnian Bay and Bothnian Sea (BB & BS), Baltic Proper (BP), Gulf of Finland and Gulf of Riga (GF & GR), and Danish straits and Kattegat (DS & KT) using the all-catchment regression (A) or the basin-specific regression (B), non-area weighted data. **a** Primary emissions scenario PE 2070. **b** Four climate scenarios: the regional climate model RCAO driven by two general circulation models (HadCM, ECHAM5) and two IPCC SRES scenarios (A2 and B2). **c** The net scenarios with both changed primary emissions and changed climate (Hägg et al. 2010)

### 17.6.2 Climate Impacts on Regulated Rivers and the Effect on Water-borne Fluxes

Damming of rivers has led to a moderate increase in lake area; for most Swedish watersheds, the share of the total watershed covered by man-made lakes has increased by only a few percentage (Smedberg et al. 2009). Most regulated rivers show reduced seasonality in water discharge compared to non-regulated rivers due to the controlled use of reservoir water, but total water discharge has not decreased despite the potential increase in evaporation as a result of increased lake area (Carlsson and Sanner 1994). Whether a change in climate will alter evapotranspiration patterns in regulated watersheds is not yet known, but with the projected increase in precipitation in the boreal part of the Baltic Sea catchment which is heavily dammed, overall water discharge to the Baltic Sea may also increase in the regulated rivers. Warmer summers in a future climate could cause a significant increase in evaporation and affect the seasonal discharge patterns of the regulated watersheds.

Hydropower accounts for 19, 47 and 77 % of electricity generation in Finland, Sweden and Latvia, respectively; for the other riparian countries, hydropower is less important, generating less than 5 % of electricity (Lehner et al. 2005). Model studies using different regional climate model (RCM) simulations to examine impacts on hydrology for the Luleälven watershed in Sweden revealed an overall increase

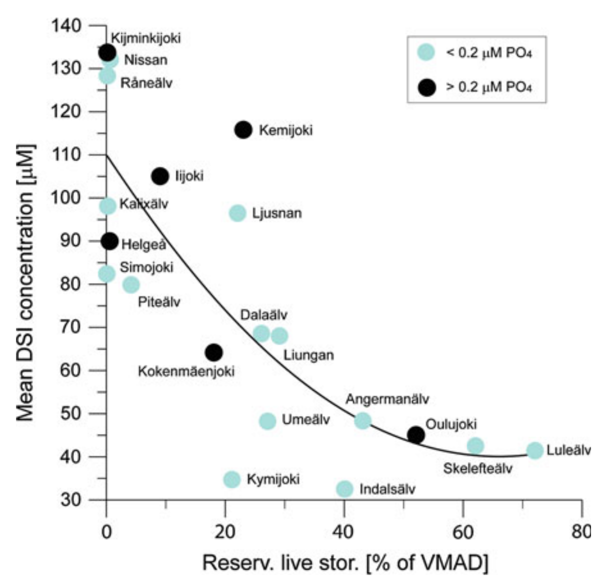



**Fig. 17.12** Dissolved silicon (DSi) concentration versus reservoir life storage in regulated Swedish and Finnish rivers (redrawn from Humborg et al. 2000)

in river flow, earlier spring peak flows and an increase in hydropower potential (Graham et al. 2007). A pan-European study presented a model-based approach for analysing the possible effects of global climate change on Europe's hydropower potential at a country scale (Lehner et al. 2005). The results indicated that even following moderate climate and global change assumptions, major change in discharge regimes could be expected, leading to a potential increase in hydropower potential of 15–30 % for Finland, Sweden and Latvia. It is not yet known whether this potential could be exploited by means of new reservoirs and dams or by increasing existing dams.

### 17.6.3 Current and Future Export from Regulated Rivers

Annual riverine DSi loads to the Baltic Sea may have decreased by about 400,000 t year$^{-1}$ or by a third over the past 100 years. This estimate is based on two independent approaches, one addressing the hydraulic load or water residence time as a key variable for nutrient retention in aquatic systems (Humborg et al. 2008b) and the other comparing DSi yields (t DSi km$^{-2}$ year$^{-1}$) in relatively unperturbed watersheds and regulated watersheds (Conley et al. 2008). In addition to deposition of BSi behind dams, river regulation leads to less weathering as shown in a model study by Sferratore et al. (2008). Lower Si fluxes were observed in River Luleälven compared to the neighbouring River Kalixälven, although specific discharge is much higher in the former despite a similar geological setting. Since damming of the River Luleälven changed the pathways of waters, that is third- and fourth-order streams were affected by regulation, surface water–groundwater interactions in these streams were interrupted, leading to less DSi input from groundwater; diatom blooms reducing DSi concentrations behind dams could be ruled out as diatom growth is relatively low in these ultra-oligotrophic rivers. However, in the southern eutrophic watersheds, the BSi (silica bound in diatom shells) concentration can be up to 100 μM and, thus, significant for the overall riverine Si transport to the Baltic Sea (Humborg et al. 2006). For other dissolved constituents such as nutrients and DOC, no large-scale estimates of changes in river load as an effect of hydrological alterations exist.

## 17.7 Conclusion

Although the effect of climate change cannot yet be quantified on a Baltic Sea basin-wide scale, some key findings can be drawn:

- Long-term trends in the timing of ice freeze-up and ice break-up indicate shorter ice-cover duration in many boreal watersheds. Observations on the discharge regime of major boreal rivers reveal no change in mean annual flow but a change in the seasonal distribution of streamflow with potentially large impacts on the redistribution of organically bound C and nutrients from land to the Baltic Sea. Winter and spring mean monthly discharge increased at many observation sites, and the peak flow in spring has become earlier. Areas with a mean annual temperature around 0 °C (i.e. around 61°N) are most sensitive to further warming.
- Over past decades and especially in the 1970s and 1980s, atmospheric deposition probably had a stronger effect on freshwater biogeochemical conditions in the Baltic Sea drainage area than climate. This pattern seems to change as atmospheric deposition decreases, however. It is likely that control of freshwater biogeochemistry will shift back from dominance by atmospheric deposition to dominance by climate. If this occurs, biogeochemical conditions in freshwaters and the Baltic Sea could change rapidly.
- Numerous studies indicate that forests and wetlands play a key role in the terrestrial export of organic-bound nutrients and C. Ditching and clear-cutting often result in increased concentrations and leaching some years after the treatment has ended, but over the long term and after re-establishing forest, this could result in lower water table levels and decreased leaching. However, current forest management only affects small parts of the overall area covered by boreal forests each year and the effect on fluxes of organic-bound nutrients and C to the Baltic Sea is probably minor. Over the short term, climate change is unlikely to affect the spatial distribution of wetlands, except for palsa mires that cover too small an area in the boreal watersheds to be significant for element fluxes to the Baltic Sea. Results from small-scale field and modelling studies indicate that increased temperature and precipitation could increase DOM transport to the Baltic Sea significantly. However, the large-scale impacts on the Baltic Sea basin are still unknown. Even a northward shift in boreal forest (i.e. Norwegian spruce) with climate change may alter quite significantly the biogeochemistry of the northernmost rivers over the long term. Modelling studies of the effect of changes in vegetation cover and structure for river loads to the Baltic Sea are ongoing but have not yet been published.
- Agricultural practices and urban sources significantly increased N and P concentrations in the rivers draining the cultivated watersheds of the southern Baltic Sea catchment. Nutrient loads from these rivers to the Baltic Sea increased several fold over the past 150 years, peaking in the 1970s and 1980s. A slight decrease is seen in P loads only over recent years probably due to improved sewage treatment from urban areas, especially in Poland. Changes in climate are not uniform across the

cultivated southern catchment area; in the south-western part (i.e. Denmark and western parts of Germany), precipitation has increased since the 1980s and farmers are currently adapting to a warmer and wetter climate by selecting heat-demanding and nutrient-demanding crops like maize. Whether increased fertiliser use, which may even occur in the transitional countries like Poland and the Baltic States, as a result of the European Common Agriculture Policies or changes in lifestyle will lead to an increased nutrient flux to the Baltic Sea is still unknown. This is because water discharge especially in the south-eastern part of the catchment is projected by some modelling studies to decrease, and the retention of nutrients (denitrification of N in soils and groundwater, river bank deposition of P) may increase. Both processes would largely compensate for the projected increase in fertiliser applications that may increase nutrient leaching. However, there are still too few catchmentwide modelling studies on water discharge and observation and modelling of retention patterns are too small scale to allow overall estimates of these processes for the Baltic Sea. Initial studies (that still need further scientific elaboration) indicate that TN fluxes to the Baltic Sea may increase up to 70 % as a result of changes in water discharge and changes in lifestyle (including change in the demand for animal protein).

- Hydrological alterations have lowered the nutrient flux to the Baltic Sea, especially for Si. Overall, the dissolved Si fluxes to the Baltic Sea have decreased by a third over the past 100 years as a result of damming and eutrophication of natural lakes. The projected increase in precipitation in the boreal part of the Baltic Sea catchment could increase hydropower potential by 15–30 %. It is unlikely that the potential formation of new lakes by damming would change the amount of water entering the Baltic Sea through rivers; however, seasonal patterns in water discharge could change significantly as they become smoothed through the operation of the dams (i.e. saving water during spring freshet and using water during winter).



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