Chapter 4
Changing Land-, Sea-, and Airscapes: Sources of Nutrient Pollution Affecting Habitat Suitability for Harmful Algae

Patricia M. Glibert, Arthur H.W. Beusen, John A. Harrison, Hans H. Dürr, Alexander F. Bouwman, and Goulven G. Laruelle

4.1 Introduction

When the first workshop related to the development of a programme related to the Global Ecology and Oceanography of Harmful Algal Blooms (GEOHAB) was held in 1998, there was consensus on two important issues (GEOHAB 1998). It was recognized first that there had been "dramatic increases in the impacts of HABs" and second that there were "no clear-cut demonstrations of the influence of specific activities on the frequency, intensity, and distribution of HABs." In the nearly two decades since the publication of that document, due in sizeable part to the efforts of the GEOHAB Programme and its associated activities, there is a new understanding,

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P.M. Glibert (✉)
University of Maryland Center for Environmental Science, Horn Point Laboratory, Cambridge, MD, USA
e-mail: glibert@umces.edu

A.H.W. Beusen • A.F. Bouwman
Faculty of Geosciences, Department of Earth Sciences—Geochemistry, Utrecht University, Utrecht, The Netherlands

PBL Netherlands Environmental Assessment Agency, The Hague, The Netherlands

J.A. Harrison
Washington State University, Vancouver, WA, USA

H.H. Dürr
Ecohydrology Research Group, Department of Earth and Environmental Sciences, University of Waterloo, Waterloo, ON, Canada

G.G. Laruelle
Department Geoscience, Environment and Society (DGES), Université Libre de Bruxelles, Bruxelles, Belgium

UMR 7619 Metis, Sorbonne Universités, UPMC, Univ Paris 06, CNRS, EPHE, IPSL, Paris, France

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and considerably more direct data, on these issues. It is now unequivocally recognized that the global expansion of HABs is continuing, with increasing abundance, frequency, and geographic extent of HABs (with new species being documented in new areas). It is also well accepted that human activities, especially nutrient pollution and climate changes, are an important, if not the most important, contributor(s) to this expansion (e.g., Heisler et al. 2008; Anderson 2014; Wells et al. 2015; Glibert and Burford 2017 and references therein). Food production and wastewater discharge accompanying the expanding global population contribute the most to nutrient pollution, but these intersect with other anthropogenic effects that have influenced freshwater flow and climate. In this chapter, we take a broad, global view of some of the massive changes that have occurred in nutrient loading as well as other physical factors that affect nutrient retention and therefore habitat suitability for HABs. Our goal here is not to describe which species may benefit under which condition, but to paint the overarching picture of changes that have occurred and that are conducive to increases in habitat suitability for HABs. We also highlight several approaches to modelling nutrient export that have been brought to bear in understanding the magnitude of these trends. While this analysis does not discount the importance of global transport of species leading to new introductions of species to new areas, and the effects of climate change and associated changes in physical structure and related environmental factors that can affect HABs [e.g., Moore et al. 2008; Doney 2010; Fu et al. 2012; see also Chap. 5, Wells and Karlson (2018)], it specifically focuses on the causes and consequences of changing nutrient loads. The rationale is that cells require nutrients for growth, and the magnitude of global nutrient changes over the recent past decades bears emphasizing [Heisler et al. 2008; Glibert et al. 2005; see also Chap. 12, Glibert et al. (2018)].

This chapter begins with an overview of changes in nutrient delivery to both the coastal and open oceans, by examining changes in land-based nutrient pollution and changes in freshwater flow, changes in sea-based nutrient pollution through aquaculture development, and changes in atmospheric nutrient deposition. Both observations and models are used in these global estimates. Second, coastal typology and the retentive nature of the different coastal types are reviewed (e.g., Pitcher et al. 2010; Dürr et al. 2011; Beusen et al. 2016). Third, applying a coastal eutrophication index comparing information on the changes in nitrogen (N) and phosphorus (P) relative to silicon (Si) (Billen and Garnier 2007; Garnier et al. 2010) and various scenarios of future change, projections of algal community composition globally for 2050 are made. Fourth, other modelling approaches of the additive effects of nutrients and climate change are reviewed showing the potential for increased algal blooms for several regions of the world.

The overarching conclusion of this synthesis is that increases in nutrient pollution are a global phenomenon, as is the alteration of water flow, altering nutrient export to coastal waters. Highest export of nutrients—from land-based sources, aquaculture sources, and atmospheric deposition—occurs in Asia, but exports from Europe and the USA are also high. An increasing risk of accelerating nutrient pollution in many world regions (western North America, eastern South America, and much of Africa and Asia) is also indicated. These changes, in turn, are altering the amount, proportion, and chemical forms of nutrients that can and do support HABs. The trajectory of more HABs is likely to increase, synergistically with other global and environmental changes, including those associated with a warmer world.

This synthesis of trends relies largely on two activities with goals that were aligned with GEOHAB: (1) Working Group 132 of the Scientific Committee on Oceanic Research (SCOR) Land-Based Nutrient Pollution and HABs and (2) a project funded under the auspices of the United Nations Environment Programme (UNEP)-Global Environmental Fund (GEF), Global foundations for reducing nutrient enrichment and oxygen depletion from land-based pollution, in support of the global nutrient cycle. The latter specifically addressed the need for more quantitative nutrient analysis, particularly in developing countries, estimated the magnitude of contributions from various nutrient sources within watersheds, and quantitatively analysed relationships between nutrient sources and coastal impacts.

4.2 Land-Based Nutrient Pollution

The global changes in N, P, and Si export to marine and freshwaters have been well documented (e.g., Beusen et al. 2016; Seitzinger et al. 2005, 2010). The global export of rivers, which has been estimated to be 40–60 Tg N per year (1 Tg = 1 million tons), has increased more than twofold since the 1860s (Boyer et al. 2006; Howarth 2008; Beusen et al. 2016). Such changes are a result of both increased human population and increases in per capita demand for nutrient-intensive foods such as meat, as well as energy and water (Seitzinger et al. 2010). Human population has now exceeded 7.5 billion people. Since 2007, most of the global population now resides in urban rather than rural areas, and it is expected that by 2050, about two thirds of all people will reside in urban areas, many of which are along coastlines (United Nations 2014). Most of the world's megacities (ten million people or more) are in China, and Africa and Asia are becoming urbanized at a more rapid pace than other parts of the world. While city growth may be favourable for some services, water and sanitation services have not grown at the same rate. Exploding urban populations have massive impacts on water supplies, and in many parts of the world, sewage remains untreated or minimally treated.

Global population growth requires growth in agriculture and animal production. In order to support this growth in agriculture, the global manufacture of N-based fertilizers has increased from <10 million metric tons N year^{-1} in 1950 to >150 million metric tons year^{-1} in 2013, with 85% of all chemical fertilizers having been produced since 1985 (Fig. 4.1a, d; Howarth 2008; Glibert et al. 2014b and references therein). In contrast to the enormous expansion in the global use of chemical N fertilizers, the use of P-based fertilizers has shown a much smaller increase, at a rate only about a third that of N (Fig. 4.1b, e; Sutton et al. 2013; Glibert et al. 2014b; Beusen et al. 2016; Bouwman et al. 2017), leading to a global

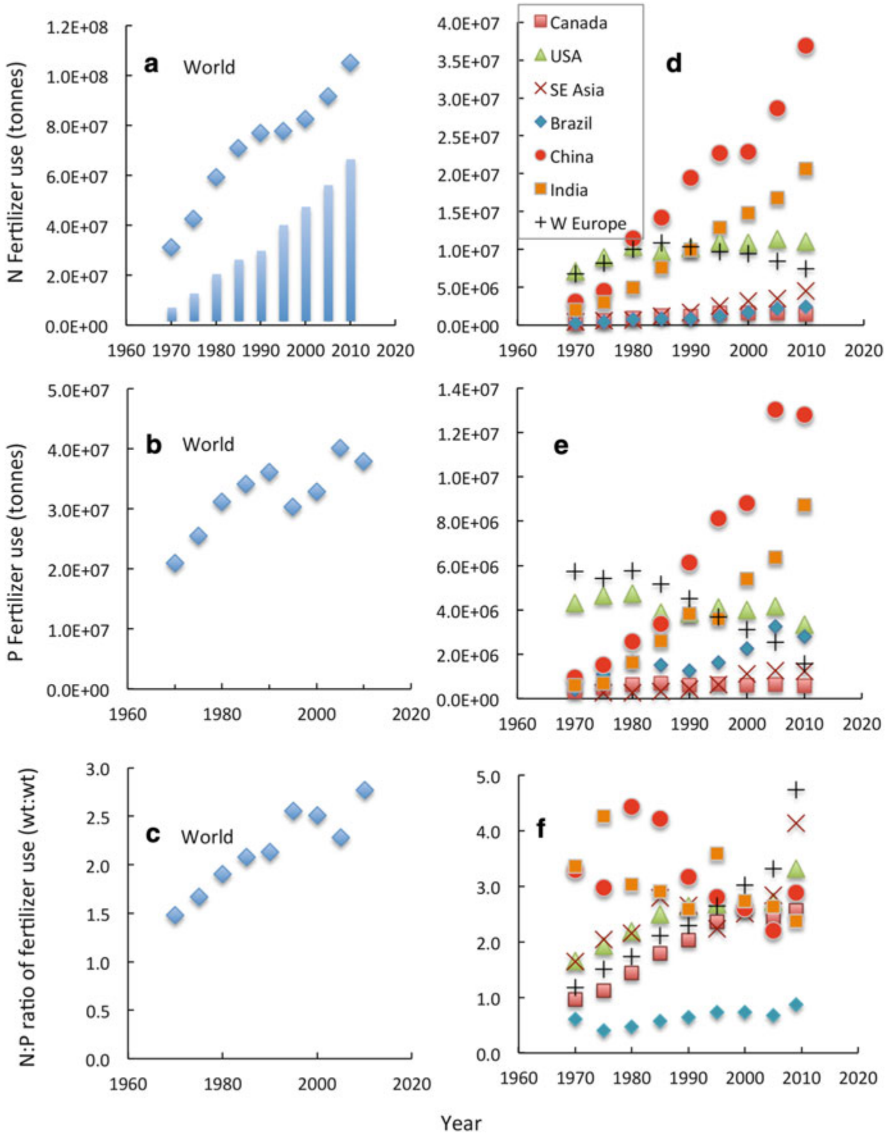


Fig. 4.1 N and P (as P_2O_5) fertilizer use and change in N:P ratio of fertilizer use by weight for the world (panels **a**, **b**, and **c**, respectively) and for selected countries or regions (panels **b**, **d**, and **e**, respectively). Superimposed on the world N use graph (**a**) is the fraction of N use as urea (*bars*). Total N and P data are from FAO (2012a, b) and data are the average of the 3 preceding years for each 5-year period; urea data are from Constant and Sheldrick (1992) through 1990 and estimated at 3.8% growth per year thereafter, comparable to urea data reported from IFA (2014). Figure reproduced from Glibert et al. (2014b) under creative commons licence

increase in N:P (Fig. 4.1c, f). Unlike N, which is fixed from the atmosphere via natural and anthropogenic processes (i.e., the Haber-Bosch process), P must be mined from the earth. Of these two major agricultural nutrients, only 10–30% actually reaches human consumers (Galloway et al. 2002; Houlton et al. 2013), and more than half is lost to the environment in direct runoff and atmospheric volatilization/eventual deposition (Galloway et al. 2014). In the case of N, a single molecule of fixed N can have multiple impacts on water, air, and soil and ecosystems (Galloway and Cowling 2002). Thus, accompanying this agricultural intensification has been an acceleration of the global N cycle, a trend projected to intensify further in the coming decades (e.g., Smil 2001; Glibert et al. 2006, 2014b; Beusen et al. 2015, 2016; Bouwman et al. 2009, 2017).

Nearly 60% of all N fertilizer now used worldwide is in the form of urea [$CO(NH_2)_2$; Fig. 4.1a; Constant and Sheldrick 1992; Glibert et al. 2006, 2014b; IFA 2014]. Global urea use as a fertilizer and feed additive has increased more than 100-fold in the past four decades (Glibert et al. 2006). It is projected that from 2012 to 2017, more than 50 new urea manufacturing plants will be constructed worldwide, half of them in China (Heffer and Prud'homme 2013), contributing to a further doubling of global urea use by 2050 (Glibert et al. 2006, 2014b). Urea can be a significant contributor to both total N and to the fraction used by phytoplankton in estuarine and coastal waters (Kaufman et al. 1983; Harrison et al. 1985; Glibert et al. 1991, 2006; Kudela and Cochlan 2000; Switzer 2008), and the frequency of reports that urea may be used preferentially by many harmful species has increased in recent years (Glibert et al. 2006 and references therein). Urea also rapidly hydrolyzes to NH_4^+ in water, another important N source for phytoplankton including HABs.

Human diets are also changing, with increasing proportions of meats as well as refined sugars. Animal agriculture is expanding to meet the dietary demands of an increasing population, and increasingly animal production is carried out in large concentrated animal feeding operations (CAFOs). To understand the scale of this nutrient source, as an example, in the Cape Fear River Basin of North Carolina, USA, it is estimated that there are 5 million hogs, 16 million turkeys, and 300 million chickens produced annually, yielding 82,700 t of N and 26,000 t of P in animal waste (Mallin et al. 2015 and references therein). In China, tens of thousands of CAFOs are estimated to produce 40 times more N pollution than other types of industries, as much of the manure produced in CAFOs is directly discharged into rivers (Ellis 2008). It has been further estimated that if unchecked, these dietary trends will, by 2050, be a major contributor to greenhouse gas emissions and global land clearing (Tilman and Clark 2014).

Spatially explicit global maps of dissolved river N and P export and dominant sources have been derived using a number of different models, including the Global Nutrient Export from WaterSheds (Global NEWS) models (Seitzinger et al. 2005; Dumont et al. 2005; Harrison et al. 2005a, b), and next-generation models, including the Integrated Model to Assess the Global Environment-Global Nutrient Model (IMAGE-GNM; Beusen et al. 2015, 2016). These models have documented not only large differences between regions and over time globally, but differences in nutrient export by element as well (Fig. 4.2a, b). The nutrient source terms that are

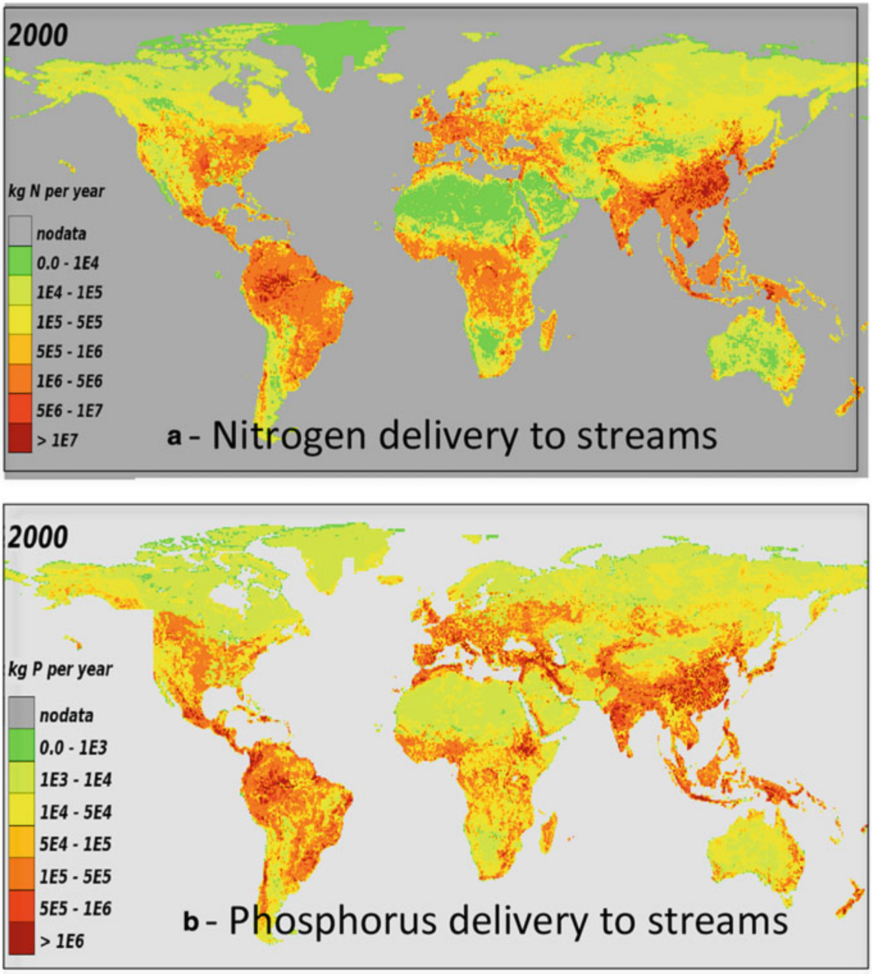


Fig. 4.2 Estimates of N (panel **a**) and P (panel **b**) delivery to streams for the year 2000 based on the Integrated Model to Assess the Global Environment-Global Nutrient Model as described in text. Figures reproduced from Supplemental Material of Beusen et al. (2016) under Creative Commons licence

considered in the Global NEWS models include natural sources such as N_2 fixation and P weathering and anthropogenic sources (nonpoint inputs from fertilizer by crop type, N_2 fixation by crops, atmospheric N deposition, and manure by animal species; point sources from sewage, as estimated by human population and treatment level; Seitzinger et al. 2005, 2010). The models also account for hydrological and physical factors including water runoff, precipitation intensity, land use and slope, as well as in-water removal processes such as dams and reservoirs and consumptive water use, and ultimately estimate nutrient export at the river mouths.

These models have shown that throughout much of Asia, Western Europe, and Eastern North America, fertilizer and manure together constitute the largest source of dissolved inorganic N (DIN) export (Dumont et al. 2005); agricultural sources of DIN (which include fertilizer, animal manures, and agricultural N_2 fixation) collectively contribute about half of the total DIN exported globally (Dumont et al. 2005). On the other hand, for dissolved inorganic P (DIP), human sewage is the largest anthropogenic source throughout much of the world, and inorganic P fertilizers and manures are less significant (Harrison et al. 2005b, 2010). The IMAGE-GNM, which is a coupled hydrology–biogeochemistry model (Beusen et al. 2015, 2016), advances the estimate of flow and retention or export of N and P by not only including the factors above but also including N and P wastewater from urban areas and from aquaculture (see Sect. 4.3) and accounts for additional in-river N transformations (Beusen et al. 2015). Both the NEWS and the IMAGE-GNM models reflect the same trends in nutrient stoichiometry as those in the fertilizer data (Fig. 4.1c), namely, that N:P ratios in water draining to the world's oceans are rapidly increasing (Beusen et al. 2016).

Previously, the Global NEWS models were used to explore the relationships between the HAB species, *Prorocentrum minimum*, and nutrient export [Glibert et al. 2008; see also Chap. 12, Glibert et al. (2018)]. Areas with high DIN and DIP yields, including Eastern Asia, Western Europe, and Eastern North America, coincide with areas in which *P. minimum* blooms have been documented and/or are increasing. This spatially explicit database also allowed identification of relationships between *P. minimum* occurrence and dominant form of N, as sewage, fertilizer, manure, or fixation of atmospheric sources and the extent to which the export of dissolved organic forms of N (DON) was attributable to anthropogenic sources.

While global models and global trends provide a macroscopic perspective, at the regional to local scale additional sources of nutrients will have to be considered in future modelling efforts. For example, for the Mediterranean Sea, it has been shown that submarine groundwater discharge likely delivers as many nutrients overall as riverine inputs (Rodellas et al. 2015). On a global scale, the N for submarine groundwater discharge is 1–1.4 Tg N (Beusen et al. 2013), but the local effect is much stronger. This is only true for N. Not for P. Even more important for near-shore areas in this region might be direct discharges of specific forms of N and P from wastewater treatment plants. While the global models estimate wastewater input, the specific level of treatment can lead to widely varying nutrient inputs even if servicing the same number of people. As described in the following section, local aquaculture is another source that is increasingly recognized to contribute regionally.

4.3 Changing Seascapes

In concert with changing diets, and in parallel with the increasing production of animals in CAFOs on land, aquaculture has been developing intensively, especially in Asia. This trend is also a function of declining natural fish stocks due to overfishing. Based on data from the Food and Agriculture Organization (FAO) on the United Nations, the annual growth rate in aquaculture was about 9% over the 1970–2000 period in both fresh and marine waters and decreased to ca. 6–7% between 2000 and 2010, with over 70% of this production in developing countries, mostly in Asia (FAO 2016); aquaculture production is expected to increase further in the coming decades (IFPRI 2003; Cork et al. 2005). Global mariculture (marine aquaculture) production is now ca. 40% of total aquaculture production, and in the most recent years, ca. 75% of mariculture is shellfish production; the remainder is finfish, including high-value marine and brackish water species (e.g., salmon, bream) in intensive farming systems in cages and net pens (FAO 2016). In 2010, 60% of global crustacean and 98% molluscan production was in marine or brackish water culture.

Aquacultural finfish production takes place in a variety of systems in freshwater and marine environments with different species and production intensities. Using data for the period 1950–2010 (derived from the database FISHSTAT, FAO 2012a, b, 2016), providing annual production, per species, country, sea, and type of environment, simple nutrient budget models were developed to describe the major flows of nutrients in shellfish and fish aquaculture systems (Bouwman et al. 2011, 2013b). The model approach describes nutrients in feed inputs, feed conversion, nutrients in fish, and the outflow in the form of faeces (solid forms based on apparent digestibility) and dissolved nutrients, as well as retention and recycling in pond and integrated aquaculture systems. Global cultured production of finfish and crustacea contributed an estimated 1.7 million tons of N and 0.46 million tons of P to receiving waters during 2008 (Verdegen 2013). Within the relatively short period from 2000 to 2006, nutrient release from shellfish cultures increased by 2.5- to 3-fold, and much larger increases are predicted in nutrient contributions from shellfish cultures by 2050 (Bouwman et al. 2011). While globally these values are small compared to riverine input, on a regional basis, they can be substantial (Bouwman et al. 2013a). As further described below, where the receiving environment is especially retentive, these nutrients can be retained, recycled, and transformed.

The implications of the expansion in aquaculture with regard to HABs are several (Bouwman et al. 2013a). Toxic and fish-killing algae are commonly associated with finfish and molluscan production (Honkanen and Helminen 2000; Wang et al. 2008), while high-biomass bloom-forming algae are more commonly associated with pond production (Alonso-Rodríguez and Páez-Osuna 2003; Wang et al. 2008). Of particular concern in both shellfish and finfish mariculture is the change in the form of nutrients as a result of excretion and associated microbial remineralization. Between 7 and 12% of the dissolved N waste of finfish consists of urea, the remainder being NH_4^+ (Kaushik and Cowey 1991). As described above, there is a mounting evidence that reduced N forms (in contrast to oxidized N forms), including urea, differentially stimulate the growth of some types of HABs (Anderson et al. 2002; Berg et al. 2003; Glibert et al. 2005, 2016), or may fuel the production of algae on which mixotrophic HABs may feed (Heisler et al. 2008;

Flynn et al. 2013). Relationships between increasing prevalence of HABs and aquaculture operations are increasingly reported with associated economic impacts (GEOHAB 2010). In China, for example, about US$2.5 million in finfish revenues were lost in 2005 due to one HAB event (Li et al. 2009). Single HAB fish-kill events in Korea have been estimated to have cost from US$1 to 100 million in lost fish, while in Japan such events have been estimated to have resulted in losses of fish worth more than US$300 million (GEOHAB 2010).

4.4 Coastal Typology and Anthropogenic Changes in Water Flow: Nutrient Retention Effects

The magnitude and effect of land-based nutrient pollution depend not only on how much nutrient is exported from land sources, but how those nutrients are retained in coastal or receiving waters and how modifications of river flow may alter nutrient export. The retention of N and P is shown to vary greatly with coastal type and latitude. Dürr et al. (2011) defined four major types of near-shore coasts, including small deltas, tidal systems, lagoons, and fjords based on hydrological, lithological, and morphological criteria. These systems represent the estuarine filter, where riverine waters mix with those of the coastal ocean, and the complex interplay of physical and biogeochemical processes takes place altering the riverine nutrient delivery to the continental shelves (Regnier et al. 2013). Fjords are particularly effective in retaining nutrients with characteristic residence times as long as decades, while small deltas have a low filtering capacity. Estuaries in temperate regions of the Northern Hemisphere are particularly important as a sink for nutrients because of the prominence of relatively long residence time tidal systems. In tropical and subtropical regions, retention is much less efficient, and a large proportion of the nutrient inputs to the ocean bypass the coastal filter by discharging directly onto the shelves in large rivers. Large rivers largely bypass the near-shore filter (McKee et al., 2004), while karstic and arheic coasts act as inactive filters. Of the nutrient inputs that enter the estuarine filter, an estimated 22% of the riverine N and 24% of the riverine P are retained (Laruelle 2009; Fig. 4.3a). These values are consistent with the limited number of system scale nutrient budgets derived from either observation (Nixon et al. 1996) or models (Arndt et al. 2009; Volta et al. 2014). These retentions are complementary to those calculated for continental shelves (Sharples et al. 2017) which predict that 75% global DIN and 80% of global DIP that enter continental shelf waters reach the open ocean. These figures suggest that the removal of nutrients taking place in estuaries and continental shelves are equally effective.

An accurate quantification of removal of nutrients by the estuarine filter generally requires extensive modelling work. The development of reactive transport models (RTMs) has advanced the capture of these complex hydrological properties (Arndt et al. 2009; Regnier et al. 2013; Volta et al. 2014). However, such complex

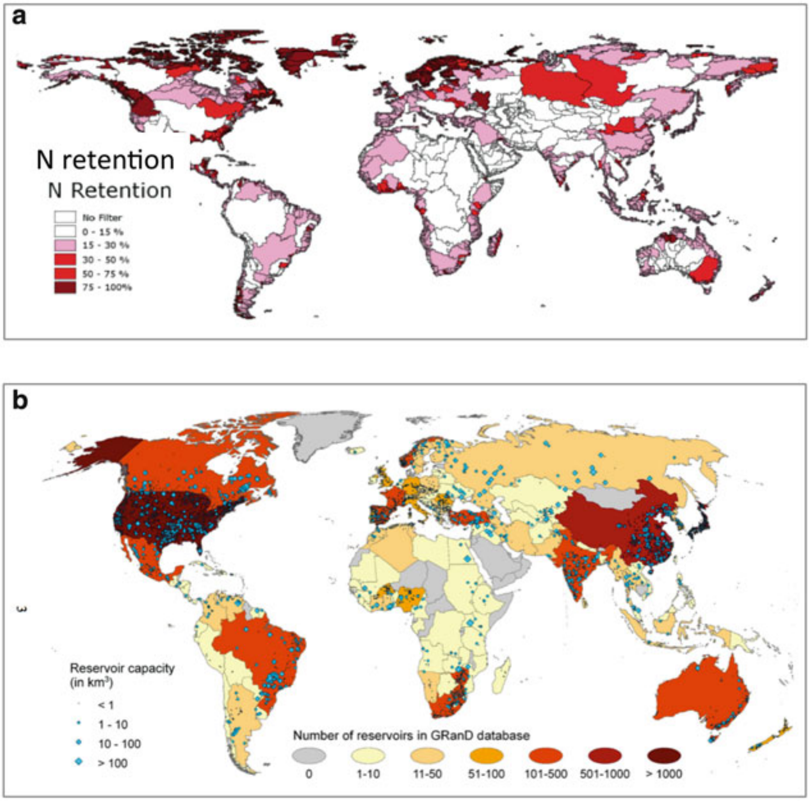


Fig. 4.3 Nitrogen retention in coastal gridcells (panel **a**). Reproduced from Laruelle (2009). Global distribution of large dams and their reservoirs, overlaid with the percentage of sediment trapped within watersheds by dams (panel **b**). Reproduced from http://www.gwsp.org/products/grand-database/global-reservoir-and-dam-grand-database-project.html (open access)

and data-intensive work can generally only be performed one system at a time, and regional applications of RTMs with less data demanding setups (Volta et al. 2014) remain in their infancy (Volta et al. 2016; Laruelle et al. 2017). As a consequence, global estimates of the efficiency of the estuarine filter in terms of nutrient retention still currently rely on up-scaling from relationships derived from a limited number of well-studied systems. Such methodology was applied by Laruelle (2009) combined with the coastal typology established by Dürr et al. (2011) to develop a global scale, spatially explicit model of the coupled N and P cycles in the estuarine zone. The model consists of a set of generic biogeochemical box models for each coastal type and was applied to estimate the estuarine retention of N and P from rivers. The model consists of a "ribbon" of cells distributed along the entire global coastline at a half-degree resolution. The model calculates the nutrient retention in each coastal cell using generic box models relying on simple empirical relationships. This typology-based modelling tool can provide an interface between spatially explicit global models of river discharge of nutrients (Seitzinger et al. 2010) and ocean global circulation models (Bernard et al. 2010, 2011; Heinze et al. 1999). The modelled retention of N and P for individual boxes compares reasonably well to data for local studies.

Coastal retention has important implications for HABs. On the one hand, in retentive regions, nutrients can be retained, recycled, and transformed, keeping the site in a nutrient-enriched condition. This can lead to enhanced phytoplankton accumulation, high-biomass HABs, and hypoxia. Coastal retention is also important in understanding impacts of aquaculture-derived nutrients, whether they remain within confined regions, in which case they are more likely to promote HABs, or whether they are likely to be flushed in a well-circulated system. Retentive zones are also important for HABs, especially those that form cysts, as the cysts accumulate and can lead to blooms whenever favourable conditions occur. Shallow, enclosed coastal lagoons can be subject to recurrent blooms once a population of HAB becomes established.

Coastal typology and features such as banks, canyons, and islands affect not only the retention of nutrients from land-based sources but also affect local circulation patterns that may transport cells or nutrients from offshore to inshore. As Pitcher et al. (2010) summarize, a rise in bottom topography can create conditions conducive to upwelling, while offshore islands may result in eddies that can influence the distribution of phytoplankton including HABs. For example, on the US West Coast, regions such as the Juan de Fuca Eddy, the Farallon Island regions, the Heceta Bank, the Monterey Bay, and the Santa Barbara Channel are all regions where *Pseudo-nitzschia* blooms are frequently observed due to their retentive nature (Hickey and Banas 2003; Pitcher et al. 2010).

In addition to effects of natural coastal typology and coastal features on nutrient retention and export, the export of nutrients to receiving waters is also affected by anthropogenic alteration of river discharge. It has been estimated that by 2030, 93% of all rivers will be affected by dams (Grill et al. 2015). Thousands of dams of a capacity of more than 1 MW have been constructed during a first construction boom in the 1950s and 1960s, and a second boom is now underway, mostly in developing economies (Grill et al. 2015; Zarfl et al. 2015; Fig. 4.3b). In fact, retention of nutrients in lakes increased >60% between 1900 and 2000 due to reservoir construction and other anthropogenic flow alterations (Beusen et al. 2016). Dam and reservoir construction and other in-river consumptive uses, as well as channelization of flow, have greatly modified the timing, magnitude, form, and stoichiometry of nutrient delivery to the coast as P is retained more efficiently than N during processing and transport in soils, groundwater, riparian zones and streams, rivers, lakes, and reservoirs (Beusen et al. 2016). For the major global dams, reactive P (total dissolved P + reactive particulate P) is more efficiently reduced (by 43%) than reactive Si (dissolved Si + reactive particulate Si) (by 21%) and total N (12%) (Maavara 2017; Maavara et al. 2014, 2015), thus increasing the N:P and N:Si ratios of riverine delivery to coastal areas. Changes in Si availability have also occurred due to sediment trapping and elemental transformations following construction of dams (e.g., Billen et al. 1999; Vörösmarty et al. 2003; Beusen et al. 2005, 2009; Syvitski et al. 2005; Harrison et al. 2012). In addition to the reductions in overall river flow that occurs from dam construction, large river systems can become fragmented preventing free movement of organisms, and severe modification of river flow alters temperature regimes and dramatically reduces sediment transport (Vörösmarty et al. 2010; Lehner et al. 2011; Liermann et al. 2012). Collectively, it is increasingly recognized that altered flow modifies not only salinity but also nutrients, in terms of total loads and proportions, and these changes can create conditions where phytoplankton assemblages change in composition. For example, retention of Si upstream following construction of the Three Gorges Dam in China has been considered to be an important factor leading to altered nutrient proportions favouring HABs in East China Sea (e.g., Zhang et al. 2015).

4.5 Changing Airscapes

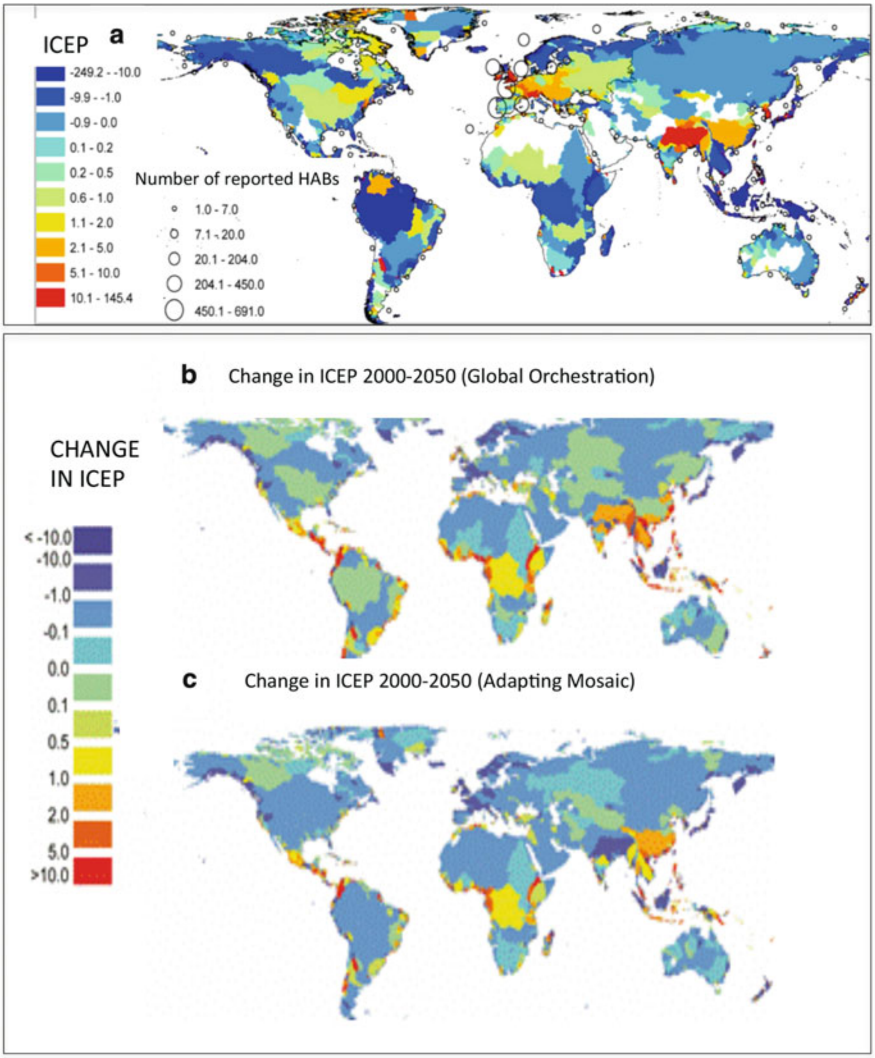
In many regions, atmospheric deposition of N is not only high but increasing (e.g., Howarth 2006; Galloway et al. 2008; Duce et al. 2008). Atmospheric deposition of N reaches >700 mg N m^2 $year^{-1}$ in many regions, particularly the downwind plumes from major cities (e.g., Duce et al. 2008). Globally, emissions, mainly of ammonia (NH_3) from land, together with combustion-related emissions of nitrogen oxides (NO_x), contribute 100 Tg N $year^{-1}$ to the atmosphere (Fowler et al. 2013). This N is derived from increasing NO_x emission from fossil fuel burning and from volatilization of animal manure and other land-based fertilizer applications. Older estimates for both European and US coastal waters suggested that anthropogenic atmospheric N deposition contributes from 10 to 40% of new N loading (Jaworski et al. 1997). Recent modelling has shown that there has been a threefold increase in soluble N deposition over the land and a twofold increase over the ocean in terms of human activities since 1850 (Kanakidou et al. 2016). All forms of chemically reduced N relative to oxidized N are also increasing in atmospheric deposition, driven largely by emissions of NH_3 from agriculture. Eastern China has experienced not only high dry deposition fluxes but also has experienced the greatest increase in dry deposition fluxes over the past decade globally (Jia et al. 2016). Moreover, emissions of NH_3 are further projected to increase and to become the dominant component of N emissions with rising temperatures (Fowler et al. 2013). For HABs, this is significant, as, has been noted above, many HAB taxa may preferentially use, grow more rapidly on, or become more toxic, when the N substrate is in chemically reduced form (Glibert 2017 and references therein).

4.6 Eutrophication Potential and Global HAB Distribution

Building on the understanding that nutrient loads are changing both in total quantity and quality, there have been various efforts to relate these nutrient compositional changes to changes in phytoplankton community assemblage. One approach that has attempted to quantify the change in nutrient ratios globally is the Indicator for Coastal Eutrophication Potential (ICEP) approach (Billen and Garnier 2007). This approach estimates the total production of non-Si algal biomass that can develop in a water body in excess of that which would be required to sustain Si biomass, i.e., non-diatom versus diatom biomass (Billen and Garnier 2007).

This index has been compared to the globally available HAB data, maintained in the Intergovernmental Oceanographic Commission (IOC) HAEDAT database (Fig. 4.4a). For this analysis, each observed HAB was assigned to the corresponding Large Marine Ecosystem (LME) region. The results for the year 2000 show an agreement between positive ICEP values and observed HABs, especially in the European region, but the Asian relationship, where nutrient loading from land, sea, and air sources is especially high, was not strong (Fig. 4.4a). However, it is important to note that HAEDAT is a metadatabase under continued development containing records of harmful algal events from various parts of the world, but by virtue of differences in regional reporting, it does not yet provide a complete global perspective. The available information on individual events varies greatly from event to event and from country to country. Regions such as the ICES area (North Atlantic) since 1985 and from the PICES area (North Pacific) since 2000 are well represented, but HAB events from Asia, South America, and North Africa are generally still underrepresented in this database. For example, the vast expansion of both high-biomass HABs and toxic HABs, including those causing paralytic shellfish poisoning in Asia in the past several decades (e.g., GEOHAB 2010), is not at present well documented in HAEDAT. Undergoing efforts to enlarge HAEDAT will facilitate comparisons with the ICEP index in the future.

Furthermore, local physical and environmental conditions will, apart from the nutrient loading and element ratios used in the ICEP concept, determine the propensity of a coastal marine ecosystem to develop high-biomass algal blooms or hypoxia. This global ICEP-HAEDAT comparison, as was the case with the global projection of *P. minimum* blooms described above, is considered a first step in our understanding of the relationships between HABs and global nutrient loads and their changes. Moreover, measures such as nutrient yields and ICEP values are annual averages whereas HABs frequently are ephemeral events, and there has been no effort made to incorporate the event time scale. Thus, there may be a temporal mismatch. Notwithstanding all these limitations, on a global basis, these projections illustrate the propensity for high-biomass HABs to occur where and when dissolved N and P yields are high.



b Change in ICEP 2000-2050 (Global Orchestration)

c Change in ICEP 2000-2050 (Adapting Mosaic)

Fig. 4.4 The eutrophication index ICEP estimated globally for the year 2000 (panel **a**) and number of HABs (based on the HAEDAT data). Changes in ICEP globally estimated for the years 2000–2050 with the assumption associated with two different development scenarios, Global Orchestration (panel **b**) and Adapting Mosaic (panel **c**)

4.7 Future Projections: Millennium Ecosystem Assessment Scenarios

Future projections of eutrophication were made using a suite of assumptions described in the Millennium Ecosystem Assessment (MA) (Cork et al. 2005) and the ICEP approach. These assumptions were, in turn, based on storylines developed by the Intergovernmental Panel on Climate Change (IPCC) and translated into changes of the main anthropogenic drivers, i.e., demography, economic development, and agricultural production (Alcamo et al. 2006). Although the MA actually defines four scenarios that differ in terms of environmental management and in degree and scale of connectedness among and across country borders, here we focus on only two of these scenarios for illustration. The "Adapting Mosaic" (AM) scenario was developed assuming proactive and regional environmental management, while the "Global Orchestration" (GO) scenario assumes reactive environmental management and a trend towards globalization (Alcamo et al. 2006).

In the coming decades, coastal zones in many world regions are almost certain to see increases in river export of N and P, even accounting for increased retention by reservoirs. The MA scenarios for 2050 show major increases in N and P river export to coastal ecosystems, particularly in South and Eastern Asia and in many countries in South America and Africa. In order to guarantee food security for populations in developing countries, and to prevent land degradation or restore soil fertility, fertilizer use will have to increase in these parts of the world. As a consequence, nutrient losses by leaching, volatilization, and runoff will inevitably increase. At the same time, urbanization and lagging sewage connection and treatment of wastewater will lead to increasing nutrient discharge to surface water in developing countries. In contrast, important decreasing trends are projected in Europe and stabilization is projected in North America and Australia. Industrialized countries are assumed to reduce nutrient discharge to rivers by developing improved wastewater treatment systems and also by reducing NH_3 volatilization, leaching, and runoff by improved nutrient management.

Meanwhile, Si river export is decreasing globally as a result of eutrophication and retention in the increasing number of reservoirs in the world's river systems. The result of these simultaneous changes of N, P, and Si is an increasing ICEP value in many world regions [western part of North America, eastern part of South America, and many parts of Africa and Asia, indicating an increasing risk that severe problems associated with eutrophication may occur (Fig. 4.4b, c)]. It is worth noting that while changes in Africa were slow between 1970 and 2000, in the coming decades, changes in nutrient stoichiometry may be more significant, probably the result of the expected fast population growth and all associated societal and economic changes, such as increasing food and energy production. With increased dam construction and due to the preferential retention of P over Si and N in reservoirs, rivers that are N-limited are under greater risk of future Si limitation (Maavara 2017).

Historical data suggest that HAB risk increased considerably between 1970 and 2000. Scenario analyses for 2050 indicate that this risk will further spread (South

America, Africa) and increase in areas with current high risk (Eastern Asia). There are also large parts of the world where the HAB risk is expected to decrease as a result of higher efficiency of nutrient use in agriculture and improved wastewater treatment. This is particularly so in the AM scenario, which is a scenario with an orientation towards proactive environmental management and simple, local solutions (Fig. 4.4c).

4.8 Future Projections: Global Ecosystem Modelling Approaches

The trajectory of more HABs is only going to be additive with other global changes, such as those associated with a warmer world [Wells et al. 2015; Sinha et al. 2017; see also Chap. 5, Wells and Karlson (2018)]. Average sea surface temperatures are expected to rise as much as 5 °C over the coming century, leading to a freshening of many oceanic regions due to ice melt and altered precipitation (e.g., Moore et al. 2008; Doney 2010; Fu et al. 2012, and references therein). These changes, in turn, will alter stratification, availability of nutrients and their forms and ratios, pCO_2, and light regimes among other factors (e.g., Boyd and Doney 2003), all of which control the extent to which HABs become established, recurrent features and likely will create many "windows of opportunity" for HABs to thrive.

Two modelling approaches highlighting the effects of nutrients and climate changes as multistressors are given here. First, a suite of model projections of the effect of climate change, together with spatially explicit nutrient loads, was undertaken to estimate the potential change in HAB distribution in several regions of the globe, NW European Shelf–Baltic Sea system, NE Asia, and SE Asia (Glibert et al. 2014a). In this modelling effort projections of the effects of climate and nutrient changes on the potential for expansion of specific harmful algal genera were made by applying a coupled oceanographic-biogeochemical model (Holt et al. 2009), combined with a suite of assumed physiological "rules" for genera-specific bloom development and habitat suitability (Glibert et al. 2014a). Habitat suitability was defined by a ratio of $NH_4^+:NO_3^-$ that exceeded 1, and an inorganic N:P ratio that was stoichiometrically imbalanced, together with genus-specific temperature and salinity criteria. Using the dinoflagellates *Prorocentrum* and *Karenia* spp. as examples, since they are globally common genera that are often associated with eutrophication, the risk of future expansion was examined in several oceanographic regions, including northwestern European Shelf–Baltic Sea system and northeastern Asia. Climate projections for "present-day" (years 1980–1990) and "future" (years 2090–2100) temperature scenarios were based on the IPCC Assessment Report on Climate Change (the "A1B" midline scenario; Solomon et al. 2007).

Model projections indicated variable habitat expansion of *Karenia* spp. and planktonic *Prorocentrum* spp. HABs under the applied assumptions of climate change (Glibert et al. 2014a; Fig. 4.5). Along the northern European coast, there

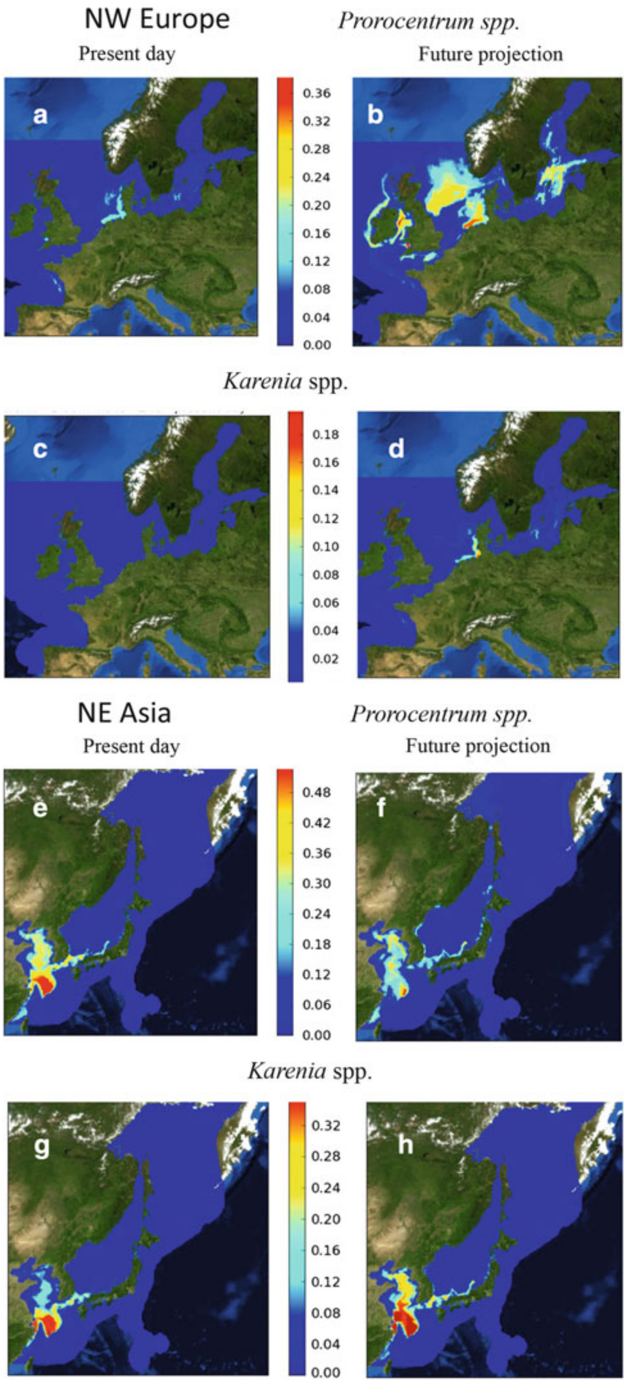


Fig. 4.5 Output of the coupled oceanographic-biogeochemical model described for NE European/ Baltic Sea region. Panels (**a**) and (**b**) depict the spatial distribution of habitat suitability (spatially explicit fraction of time of the year for which all suitable conditions were met) for *Prorocentrum*

was a considerable expansion in the number of months annually conducive to both HAB genera, but this was most notable for planktonic *Prorocentrum*. The expansion of these HABs in the future scenarios was less for the Asian coast, but there appears to be a northern geographic expansion. The projected increases in temperature and nutrient conditions suggest an even greater potential for expansion of these blooms, but the overlap of these conditions could limit the manifestation of these effects. A temporal mismatch in suitability of conditions for growth should not be viewed as evidence for limited potential for expansion. Rather, such a difference points to the possibility of expansion should there be a change in the timing of any one of the parameters. Climate forcing may alter the timing of nutrient loads relative to seasonal warming that may in turn alter the alignment of suitable conditions. Overall, the model projections described here showed the future potential expansion for these two HAB genera in two large oceanic regions, and it is foreboding. Many factors are involved, and different species may show quite different projections. Nevertheless, collectively it is clear that the expansion of HABs, exemplified by *Karenia* spp. and planktonic *Prorocentrum* spp. in some regions, is likely to continue in the coming years.

Supporting evidence for the worsening of eutrophication together with climate changes, and therefore the propensity for the worsening of HABs, comes from a recent model that explored climate change-induced precipitation changes and that showed a large potential for increases in N loading and eutrophication by the end of the century (Sinha et al. 2017). Estimates in this effort were derived from the Climate Model Intercomparison Project Phase 5 (CMIP5) models and the "business-as-usual" scenario. While the empirical model was specific to the USA, the model was applied globally by seeking regions that met specific criteria, and a number of regions with similar conditions were identified. In so doing, the Sinha et al. (2017) study reported that large portions of Asia, especially India and Eastern China, had conditions similar to some regions in the USA, including the Mississippi–Atchafalaya River Basin, and the northeast and Great Lakes regions. As discussed throughout this chapter, these regions are already global nutrient hotspots, as well as regions of frequent and/or increasing HAB occurrences, and thus these model projections serve to underscore the magnitude of the challenge of managing nutrient loads.

Fig. 4.5 (continued) spp. for present (which encompasses period from 1980 to 1990) and future conditions projected using A1B IPCC scenarios for climate change (which encompasses period from 2090 to 2100). Panels (**c**) and (**d**) are the same except for conditions suitable for *Karenia* spp. Panels (**e**)–(**h**) are the same, except for NE Asia. Reproduced and modified from Glibert et al. (2014a) with permission from the publisher

4.9 Conclusions

Recognizing the vast anthropogenic effects that nutrient pollution, harvesting and production of food (including associated fertilizer use), and altered hydrodynamics for water consumption or electricity generation are having on the globe is fundamental to understanding how these changes affect ecological function and biodiversity, including microbial biodiversity and HABs. Nutrient pollution in retentive coastal zones has fundamentally different effects than nutrient input into systems with less retentive properties. Nutrient hotspots are clear around the globe, with severe nutrient loading issues in Asia, Europe, and the USA. Scenario analysis shows that these and emerging regions around the globe will continue to face nutrient-related problems, including expanding HAB issues for decades to come. Additionally, climate change is not only altering environmental conditions, but it is altering the seasonality and timing of co-occurring suitable factors for HAB growth.

There are many opportunities to advance the understanding of HABs and environmental changes, and continued international collaborative programmes, like Global NEWS and GlobalHAB [see also Chap. 22, Berdalet et al. (2018)], will be essential to further our understanding of changes in HABs. Such advances in understanding of both HABs and the environmental conditions to which they are exposed need to be achieved across the full spectrum of scales and across the land-, sea-, and airscapes that are so rapidly changing. Multiple, co-occurring changes such as nutrient pollution, increasing reservoir capacity resulting in the increased retentiveness of rivers, and increasing global temperatures all suggest that proactive management will be required to stabilize or reduce HAB occurrences.

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