

RESEARCH REVIEW

Human activities and climate variability drive fast-paced change across the world's estuarine–coastal ecosystems

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Abstract

Time series of environmental measurements are essential for detecting, measuring and understanding changes in the Earth system and its biological communities. Observational series have accumulated over the past 2–5 decades from measurements across the world's estuaries, bays, lagoons, inland seas and shelf waters influenced by runoff. We synthesize information contained in these time series to develop a global view of changes occurring in marine systems influenced by connectivity to land. Our review is organized around four themes: (i) human activities as drivers of change; (ii) variability of the climate system as a driver of change; (iii) successes, disappointments and challenges of managing change at the sea-land interface; and (iv) discoveries made from observations over time. Multidecadal time series reveal that many of the world's estuarine–coastal ecosystems are in a continuing state of change, and the pace of change is faster than we could have imagined a decade ago. Some have been transformed into novel ecosystems with habitats, biogeochemistry and biological communities outside the natural range of variability. Change takes many forms including linear and nonlinear trends, abrupt state changes and oscillations. The challenge of managing change is daunting in the coastal zone where diverse human pressures are concentrated and intersect with different responses to climate variability over land and over ocean basins. The pace of change in estuarine–coastal ecosystems will likely accelerate as the human population and economies continue to grow and as global climate change accelerates. Wise stewardship of the resources upon which we depend is critically dependent upon a continuing flow of information from observations to measure, understand and anticipate future changes along the world's coastlines.

Keywords: climate variability, ecosystems, environmental change, estuarine–coastal, global change, human disturbance

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Introduction

It has been nearly two decades since Vitousek *et al.* (1997) published their seminal paper declaring that we live in a human-dominated planet. Evidence has accumulated since that the human population has transformed landscapes, chemistry of the atmosphere and oceans, biological communities from the top to bottom of food webs, biogeochemical cycling, patterns of pre-

cipitation and runoff, ice extent across the Arctic, and the functions essential for human welfare provided by planet Earth (Röckstrom *et al.*, 2009). Our knowledge of these transformations has accumulated from observations made repeatedly over time that detect, measure and sometimes explain the underlying role of human activities and put these in the context of a variable climate system.

Repeated measurements sustained over decades capture changes provoked by specific human actions such as damming a large river, or fluctuating climate patterns such as the North Atlantic Oscillation. Repeated

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observations before and after such events can be exploited as experiments to learn about the patterns, rates and processes of change in the Earth system as it continues to respond to the expansion of human activities and changing climate. Measurements sustained over time contain information and that information has become essential for understanding environmental change, for establishing policies to conserve resources and sustain processes vital for humanity, for measuring the effectiveness of those policies through adaptive management and for anticipating changes that will unfold under different scenarios of climate change and continued global population and economic growth.

The human population is concentrated along coastlines where we expect signals of anthropogenic changes to be particularly strong. Many different kinds of observational programs have been sustained in marine coastal waters, most over the last several decades. Our purpose here is to synthesize the information contained in time series of measurements made in coastal marine ecosystems influenced by connectivity to land – estuaries, bays, lagoons, inland seas and nearshore shelf waters receiving river runoff. These are highly complex transitional ecosystems between land and sea, so the signals of human disturbance can be confounded by variability of the climate system through its separate influences over watersheds and ocean basins (Feyrer *et al.*, 2015). However, clear signals can emerge when time series are extended long enough and include measurements of key drivers of change both on land and in the coastal ocean.

Our perspective of coastal change is just developing because multidecadal, globally distributed coastal time series were not available a generation ago. However, the observational records now available show that changes at the sea–land interface are even more complex, pervasive and occurring faster than we could have anticipated when Vitousek *et al.* (1997) concluded that ‘we are changing Earth more rapidly than we are understanding it’.

The human dimension

Time series of measurements sustained over the past few decades provide compelling evidence that the human footprint is broad and deep along continental margins where our populations and activities are concentrated. We highlight as examples three anthropogenic drivers of change – nutrient enrichment, water management and fishing.

Explosive growth of the human population and its activities on land have altered global patterns of element cycling, and recent assessments are that increased flows of nitrogen (N) and phosphorus (P) from land to

sea have exceeded planetary boundaries – thresholds beyond which we can anticipate unacceptable consequences for humanity (Carpenter & Bennett, 2011; De Vries *et al.*, 2013). Nutrient enrichment is a powerful driver of change where rivers and sea meet, exemplified by profound changes over a 30-year period in the coastal Black Sea that receives runoff from industrial and agricultural areas of nine countries. By the early 1990s, the Danube River was delivering 60 000 tons of P and 340 000 tons of inorganic N to the Black Sea annually (Mee, 1992). The syndrome of responses to what Nixon (1998) described as ‘enriching the sea to death’ included: occurrences of dense monospecific phytoplankton blooms, decreased water clarity and loss of macrophytes, proliferation of jellyfish, widespread occurrences of hypoxia and loss of a large proportion of the macrobenthos community. Eutrophication at this scale has economic and social consequences, including demise of fisheries that once supported 2 million people and a tourism industry suffering from clouded waters and beaches fouled by decaying organic matter (Mee, 1992). Manifestations of nutrient enrichment are evident along the world coastlines and include 10-fold chlorophyll *a* increase in regions of Chesapeake Bay since the 1950s (Harding & Perry, 1997), doubling of primary production in Danish coastal waters from the 1950s to the late 1970s (Ærtebjerg Nielsen & Gargas, 1984), fourfold expansion of hypoxic zones in the Baltic Sea from 1950 to 1970 (Carstensen *et al.*, 2014) and recurrent massive green tides in coastal waters of the southern Yellow Sea since 2007 (Li *et al.*, 2014).

Water management that captures, consumes and diverts runoff has also profoundly modified coastal ecosystems by altering inputs of fresh water, sediments and nutrients. More than half the world’s large river systems are dammed (Nilsson *et al.*, 2005) and cumulative discharge of midlatitude rivers declined 60% during the 1951–2000 era of intense dam construction (Milliman *et al.*, 2008). Virtually all the fresh water carried by the Colorado River, United States, has been appropriated since construction of the Glen Canyon Dam in 1963. The loss of river inflow shrunk wetlands in the river delta, reduced abundance of bivalve mollusks over 90% and created habitat changes that endanger numerous species in the Sea of Cortez (Baron *et al.*, 2002). Water diversions across the Florida Everglades, United States, reduced freshwater inflow to Florida Bay by about 60%, leading to salinity increase that restructured seagrass communities as the salt-tolerant *Thalassia testudinum* replaced *Halodule wrightii*. This shift has economic and ecological significance because fish densities are highest in seagrass habitats dominated by *Halodule* (Herbert *et al.*, 2011). Sediment retention behind the Three Gorges Dam reduced sediment dis-

charge of the Changjiang (Yangtze) River by 70%, starving its estuary of sediments and causing erosion of the coastal delta and submersion of salt marshes (Yang *et al.*, 2011). Similar changes followed damming of the Ebro River (Vericat & Batalla, 2006), Nile River (Hamza, 2009), Mississippi River (Blum & Roberts, 2009) and Chikugo River (Hayami *et al.*, 2014). Egypt's coastal fishery collapsed after closure of the Aswan Dam in 1965 because it disconnected Mediterranean coastal waters from their riverine nutrient source (Nixon, 2003), and similar losses of nutrient supply, primary production and fish biomass occurred in the Bohai Sea after damming reduced discharge of the Huanghe (Yellow) River by 73% (Fan & Huang, 2008).

Overfishing is the oldest pervasive human disturbance to coastal ecosystems, beginning centuries ago and leading to extinctions of large vertebrates (Jackson *et al.*, 2001). This historical perspective provides a management context for understanding effects of overfishing in modern times that depleted stocks of cod (*Gadus morhua*) in the Baltic Sea (Casini *et al.*, 2008), striped bass (*Morone saxatilis*) in US Atlantic coastal waters (Richards & Rago, 1999), anchovy kilka (*Clupeonella engrauliformis*) in the Caspian Sea (Daskalov & Mamedov, 2007) and led to closure of commercial fishing in San Francisco Bay – once a premier fishing center (Skinner, 1962). Overfishing depletes populations of targeted species, but it also restructures biological communities at all trophic levels as apex predators are removed. Loss of cod from the Baltic sea in the early 1980s reduced predation mortality on spratt (*Sprattus sprattus*), allowing its population to grow while biomass of its zooplankton prey fell and phytoplankton biomass grew through a fishing-induced trophic cascade (Casini *et al.*, 2008). Overfishing of large predators, herbivores and shellfish induced other trophic cascades that contributed to losses of kelp forests in the Gulf of Maine and seagrasses in Moreton Bay, Australia, and mortality of corals in the Great Barrier Reef (Jackson *et al.*, 2001). By reducing biological complexity, the overharvest of large animals has increased the sensitivity of coastal ecosystems to other disturbances such as nutrient enrichment, and it preconditioned the 'collapse' of coastal marine ecosystems measured in recent decades (Jackson *et al.*, 2001). In addition, commercial trawl fishing reduces diversity of benthic communities by removing large epifauna and disrupting the topographic complexity of benthic habitats (Thrush *et al.*, 1998).

Human disturbances have interactive effects that we are slowly beginning to understand from sustained observations. We give two examples, the first from sampling in Ringkøbing Fjord by the Danish National Aquatic Monitoring and Assessment Program that began in 1989. This shallow lagoon is highly enriched

in nutrients from land runoff, and its water exchange with the North Sea is manipulated by operation of a control gate. Water exchange was altered in summer 1995 to enhance inflow of saline North Sea water and dilute nutrient concentrations. The monitoring program detected a shift in the plankton community a year later when zooplankton biomass declined abruptly, populations of copepods *Eurytemora affinis* and *Acartia tonsa* collapsed, and phytoplankton biomass (mean chlorophyll *a* concentration) decreased from 52 to 8.7 mg m⁻³. The loss of plankton followed a sharp population increase of the marine clam *Mya arenaria* in 1996 (Fig. 1a), and it was attributed to filter feeding by *Mya*

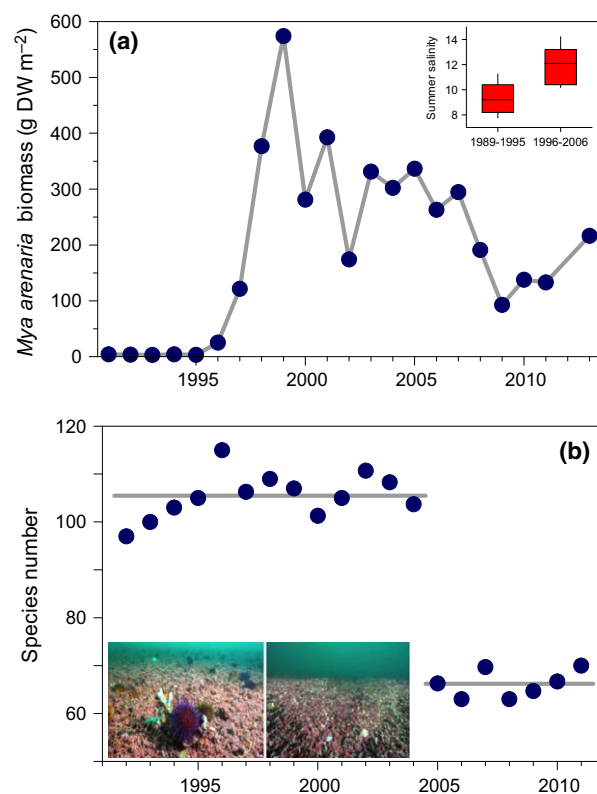


Fig. 1 Time series of: (a) annual mean biomass (g dry weight m⁻²) of *Mya arenaria* in the Ringkøbing Fjord from samples collected 1991 through 2013. The abrupt increase in clam abundance occurred after water exchange with the North Sea was altered in 1995 and salinity in the fjord increased. The inset compares distributions of summer salinity before (median = 9.2) and after (median = 12.1) water exchange was altered. (b) species richness of benthic invertebrates sampled from maerl habitats in the Bay of Brest from 1992–2011. Each point is the mean number of species identified in 3 replicate 0.1-m² sediment samples collected in autumn with a Smith-McIntyre grab. The series is divided into two eras showing the effects of habitat disturbance by clam fishing, including loss of macrofauna (shown in the inset photograph) that began in late 2004. Data are available at <http://www-ium.univ-brest.fr/observatoire/observation-cotiere>.

that outcompeted zooplankton for the phytoplankton food resource (Petersen *et al.*, 2008). These community shifts followed the change in control gate operation that doubled the rate of water exchange with the North Sea, increased salinity in the lagoon (Fig. 1a, inset) and facilitated widespread *Mya* recruitment. As the *Mya* population grew, it displaced deposit-feeding benthic macrofauna, and as phytoplankton biomass decreased, the coverage of rooted vascular plants expanded (Petersen *et al.*, 2008).

The second example comes from decades of sampling in the Bay of Brest, France, by the Institut Universitaire Européen de la Mer. This observational program measures changes in plant and animal communities of maerl beds – calcareous habitats supporting high biological diversity and having unique conservation value (Grall & Hall-Spencer, 2003; Hall-Spencer *et al.*, 2003). Sampling over the period 1992–2011 detected an abrupt drop in macrofaunal diversity after 2004, when mean species number fell from 105 to 66 (Fig. 1b). This decline in species richness followed a change in fishing practices to target the deep-burrowing clam *Venus verrucosa*. Deep dredging removes soft sediments and destroys the three-dimensional structure of maerl beds (see photo inset, Fig. 1b), leading to habitat homogenization and loss of macrofauna such as urchins, tunicates, sponges and large bivalves. The change in fishing practice was provoked by closure of the scallop (*Pecten maximus*) fishery due to high levels of domoic acid (DA) – the agent of amnesiac shellfish poisoning (ASP) that is produced by the diatom *Pseudonitzschia* and accumulates in scallop tissues (Belin *et al.*, 2013). *Pseudonitzschia* blooms were first detected in coastal waters of Brittany during the 1990s, and a first major ASP event occurred in the Bay of Brest during 2004. Proliferation of toxic blooms is a presumed outcome of nutrient enrichment from intensification of agriculture in the coastal landscape (Romero *et al.*, 2013).

Time series from Ringkøbing Fjord illustrate the power of human activities, in this case, a management action to ameliorate eutrophication effects, that can trigger sudden shifts in marine ecosystems with broad and unanticipated consequences. Time series from the Bay of Brest illustrate the complex pathways through which human actions change coastal ecosystems. Anthropogenic nutrient enrichment sustains toxic algal blooms that provoke closure of scallop fishing to protect human health. Fishermen adapt by harvesting alternative resources using methods that severely damage maerl beds. This cascade of actions confounds the challenge of conserving fragile and increasingly rare maerl habitats that require millennia to develop, and it threatens sustainability of commercially important species such as *Pecten maximus* that require live maerl habitat for recruitment.

Climate variability

Situated at the sea–land interface, estuarine–coastal ecosystems are uniquely influenced by the different responses to climate variability expressed over land, in the local atmosphere and in the adjacent coastal ocean. We give examples of each: salinity variability tied to precipitation patterns over land, a phytoplankton change during a period of increasing air temperature and an ecological state change following a shift in climate forcing across an ocean basin.

The Patos Lagoon receives freshwater runoff from a 200 000 km² watershed in southern Brazil (Odebrecht *et al.*, 2010). Monthly sampling from 1993 to 2012 (Fig. 2a) as part of the Brazilian Long-term Ecological Program shows that salinity fluctuates between 0 and 35, so the estuarine region of the lagoon can function as a freshwater, a marine or a brackish system (Abreu *et al.*, 2010). Annual salinity is strongly correlated with annual precipitation (Odebrecht *et al.*, 2010) that varies with changes in regional atmospheric circulation tied to the El Niño–southern Oscillation system (Grimm *et al.*, 2000). Salinity was low during strong El Niño years of high precipitation (1998, 2001–2003) and low during dry, strong La Niña years (1999, 2005, 2006). The ENSO–salinity linkage is illustrated by the distributions of annual measurements (Fig. 2a inset) showing that median salinity was 20.5 when the Oceanic Niño Index (ONI) was less than –1 (strong La Niña), but 0.5 when the ONI was greater than +1 (strong El Niño). This coupling between ENSO and regional precipitation has ecosystem-scale ramifications: high inflow during El Niño years leads to fast flushing, low phytoplankton biomass, loss of rooted plants (*Ruppia maritima*), reduced abundance and diversity of benthic invertebrates, reduced abundances of dominant fish species, and poor catches of fish and shrimp with economic and social consequences. Dry La Niña years have the opposite patterns (Odebrecht *et al.*, 2010). Repeat observations from the Patos Lagoon illustrate that annual variability of estuaries and their biota track large-scale atmospheric processes through their influence on precipitation and runoff.

Nutrient enrichment of the Baltic Sea has amplified production of phytoplankton biomass, and metabolism of that biomass removes oxygen from bottom waters to create a chronic state of hypoxia (Carstensen *et al.*, 2014). Hypoxia in the Baltic is linked to a second manifestation of nutrient enrichment – massive summer blooms of N-fixing cyanobacteria (*Nodularia spumigena*, *Aphanizomenon* sp.) that aggregate near the surface (Kahru & Elmgren, 2014). Hypoxia stimulates release of

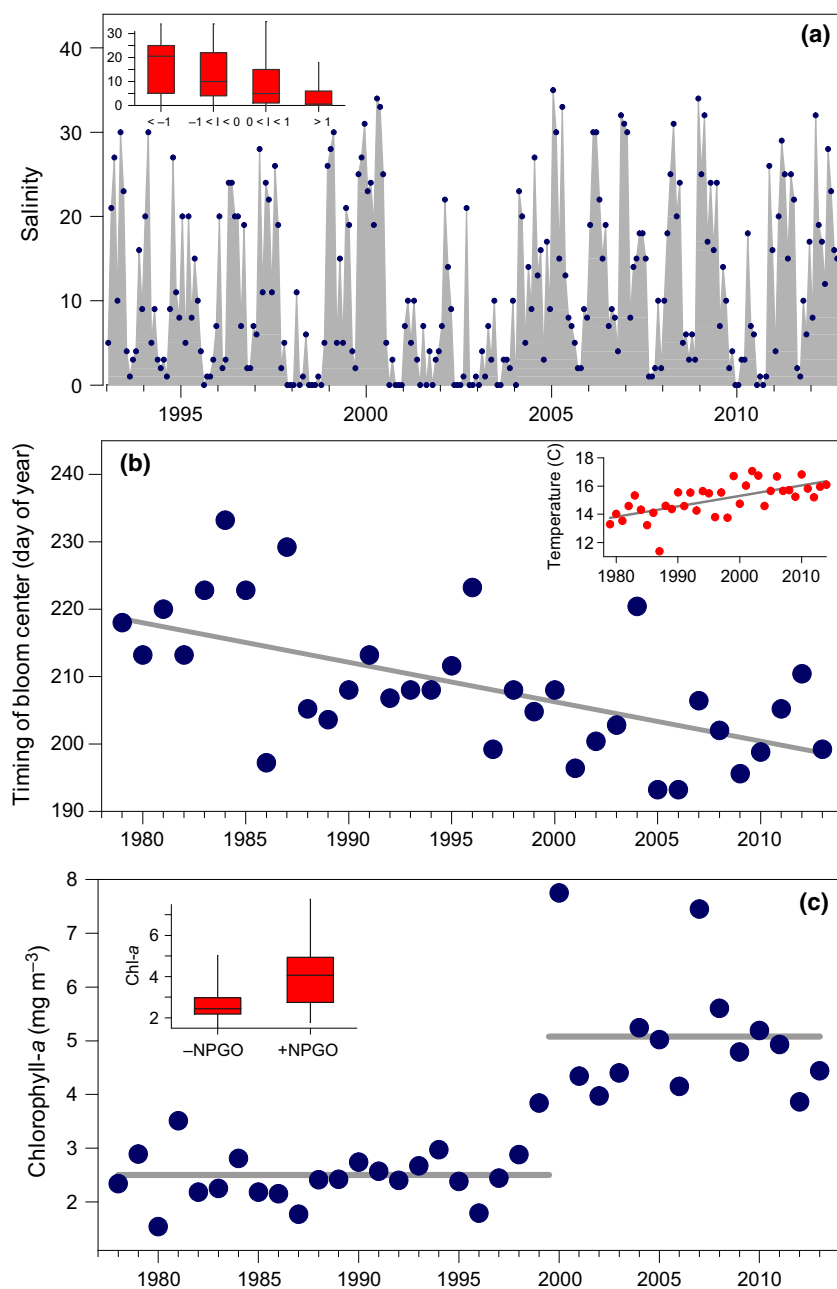


Fig. 2 Time series of: (a) monthly surface salinity in Patos Lagoon Estuary from January 1993 through December 2012. The inset compares salinity distributions for four ranges of the Oceanic Niño Index I: less than -1 (strong La Niña); between -1 and 0 ; between 0 and 1 ; and >1 (strong El Niño). The Oceanic Niño Index was accessed at: http://www.cpc.ncep.noaa.gov/products/analysis_monitoring/ensostuff/ONI_change.shtml. (b) steady decrease in the center of timing of cyanobacteria blooms across the Baltic Sea from 1979–2013 (redrawn from Figure 11A of Kahru & Elmgren, 2014). Day 220 is 8 August; day 200 is 19 July. The gray line is the linear regression of the index for the whole Baltic over time. The inset shows a significant trend of increasing June–August surface temperature (obtained from the Baltic Environmental Database at Stockholm University <http://nest.su.se/models/bed.htm>). (c) mean June–December chlorophyll *a* concentration in South San Francisco Bay from 1978 through 2013 (data accessed at: <http://sfbay.wr.usgs.gov/access/wqdata/>). The horizontal gray bars separate the series into two eras: 1978–1998 (median chlorophyll *a* = 2.4 mg m^{-3}) and 1999–2013 (median chlorophyll *a* = 5.0 mg m^{-3}). The inset compares distributions of chlorophyll *a* in the negative and positive states of the North Pacific Gyre Oscillation (NPGO), accessed at <http://www.o3d.org/npgo/npgo.php>.

sediment-bound P, creating an ideal environment for N-fixing cyanobacteria to sustain a 'vicious circle' of eutrophication (Vahtera *et al.*, 2007). Consequences include large-scale habitat loss (many deep regions of the Baltic are devoid of macrofauna) and blooms of toxin-producing algae that are risks to humans and fisheries. A compilation of satellite-derived data (Fig. 2b) provides a 35-year time series that shows large annual fluctuations in the areal distribution of surface cyanobacteria blooms during summer, with higher coverage after 1997 (Kahru & Elmgren, 2014). From this series, Kahru & Elmgren (2014) calculated an index of bloom timing as the day each year when cumulative accumulation reached half the yearly total. This index decreased 6 days per decade, so the timing of blooms has advanced 20 days, from 8 August to 19 July (Fig. 2b). The most expansive summer blooms occur in periods of warm and sunny weather, so earlier onset might be related to warming of the Baltic (Fig. 2b inset) and earlier start of the stratified season, consistent with projections that cyanobacteria will become increasingly selected in nutrient-rich waters as global warming proceeds (Paerl & Huisman, 2008). A satellite-derived time series from the Baltic Sea illustrates a change in the timing of phytoplankton bloom development during an era of increasing air temperature.

South San Francisco Bay is a shallow marine lagoon situated in an urban landscape and connected to the California Current System (CCS). Human pressures include nutrient enrichment, primarily from treated sewage (Cloern & Jassby, 2012). A 1978–2013 time series of phytoplankton biomass collected by the US Geological Survey revealed an abrupt increase in summer–autumn (June–December) chlorophyll *a* after 1998 (Fig. 2c). This biomass increase followed a climate shift across the North Pacific, indexed as a transition of the North Pacific Gyre Oscillation (NPGO) to its positive state. Phytoplankton biomass in the Bay was twice as high during NPGO+ years compared to NPGO- years (Fig. 2c, inset). Positive NPGO is associated with earlier and stronger coastal upwelling and elevated primary and secondary production in the central CCS (Chenillat *et al.*, 2012). The linkage between NPGO and Bay phytoplankton might operate through two processes: (i) a stronger ocean supply of phytoplankton biomass to the Bay after a post-1998 increase of chlorophyll *a* in California coastal waters (Kahru *et al.*, 2012) and (ii) immigration of anomalously large numbers of demersal predators (juvenile flatfish, crabs and shrimp) into the Bay after 1999 that triggered a trophic cascade by removing bivalves and releasing their grazing pressure on phytoplankton (Cloern *et al.*, 2010). Thus, the shift to a high-chlorophyll era resulted from an amplified source and damped sink for phytoplankton biomass

resulting from a climate-driven increase in the productivity of the adjacent coastal ocean. Time series of phytoplankton, bivalves, fish and crustaceans in San Francisco Bay show that multidecadal oscillations of the ocean–atmosphere system produce changes that propagate into estuaries and bays to reorganize their biological communities.

Managing change

Changes detected in observational programs motivate policies to recover and sustain coastal ecosystems damaged by human disturbances. Effectiveness of those policies is gauged from measurements made after they are implemented (Riemann *et al.*, 2015). There are many successes, and we highlight two examples as recovery from disturbance by sewage disposal and nutrient enrichment.

Sewage disposal

Deep Bay is a shallow bay in Hong Kong waters surrounded by Shenzhen, one of the fastest growing cities and economies in China that has been transformed from a small fishing village in the 1970s to a mega-city of 14 million people. This explosive population growth created serious water-quality problems by the 1980s as growing inputs of organic matter and nutrients caused hypoxia, ammonium toxicity and bacterial contamination (Yin *et al.*, 2010). Measurements by the Hong Kong Environmental Protection Department showed that dissolved oxygen concentrations declined significantly in Deep Bay from 1988 to 2004 (Fig. 3a) as the regional population grew (Fig. 3a inset). The annual mean concentration of dissolved oxygen fell to a minimum 3.6 mg l⁻¹ in 2003, but it has increased steadily since. The trends of oxygen decline, and then increase, were responses to growing inputs of organic wastes that paralleled population growth in the surrounding landscape, followed by policies formulated in 1999 to reduce the number of livestock farms, impose effluent treatment standards on other farms and extend the Hong Kong Sewerage Master Plan to hundreds of villages (Xu *et al.*, 2010; Yin *et al.*, 2010). The Deep Bay example illustrates changes seen along all the world's coasts: organic enrichment depletes oxygen from coastal waters, but the problem can be solved with sewage treatment and actions to reduce inputs of animal waste.

Nutrient enrichment

Tampa Bay, United States, was described as grossly polluted in the 1950s (Johansson & Lewis, 1992). Environmental deterioration was measured in the late 1970s

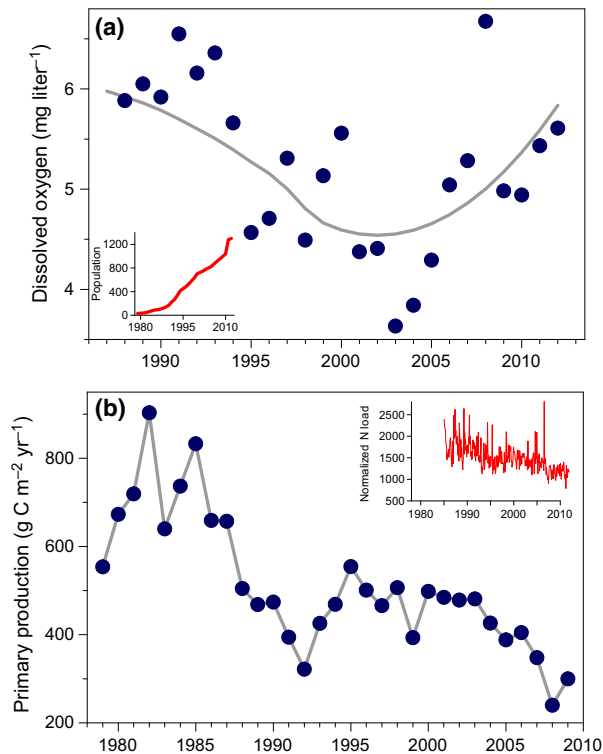


Fig. 3 Time series of: (a) annual mean dissolved oxygen in surface waters of Deep Bay, Hong Kong, based on monthly measurements by the Hong Kong Environmental Protection Department (www.epd.gov.hk) at site DM2 in the inner bay from January 1988 to December 2012. The gray line is a loess smooth with span 0.6. The inset shows population growth of Shenzhen ($\times 10^4$) from 1979 to 2012. (b) annual phytoplankton primary production in Tampa Bay (Hillsborough Bay) from 1979 to 2008. The inset is a monthly time series of flow-adjusted nitrogen input ($\text{kg N } 10^{-6} \text{ m}^{-3}$ freshwater inflow) from January 1985 to December 2013 as an indicator of success of actions to reduce anthropogenic nutrient enrichment.

and 1980s as high phytoplankton biomass, dense shore-line accumulation of macroalgae, reduced water clarity, hypoxia/anoxia, loss of seagrasses and benthic invertebrates, all attributed to nutrient enrichment from wastewater and runoff. The deteriorating conditions were reported by early studies (FWPCA, 1969; Santos & Simon, 1980; Johansson & Lewis, 1992) and subsequent reports (Lewis *et al.*, 1998; Greening & Janicki, 2006; Greening *et al.*, 2014) described the progression of worsening conditions. Measurements by the City of Tampa Bay Study Group showed that phytoplankton primary production exceeded 500 g C m^{-2} each year from 1979 through 1988 – an index that the estuary was hypertrophic (Nixon, 1995). However, as the time series extended, it showed a steady decline of primary production that has been reduced by half (Fig. 3b). This striking reduction of primary production in a once

hypertrophic estuary is a measure of success of policies to reduce nutrient inputs to Tampa Bay that began in the late 1970s and evolved from citizen input and pressure. Advanced sewage treatment was implemented in 1979, and it reduced N loading by half (Greening *et al.*, 2014). Over 300 other projects were implemented to continue reductions of N loading, and their success is measured as a steady decline of flow-adjusted N input to the estuary (Fig. 3b inset). Other signs of recovery include threefold reduction of chlorophyll *a*, disappearance of cyanobacteria blooms, increasing water transparency, expansion of seagrass habitat and biomass by 65% since the 1980s and increased dissolved oxygen in bottom waters (Johansson & Lewis, 1992; Greening & Janicki, 2006; Johansson, 2010; Greening *et al.*, 2014). Time series from Tampa Bay provide an example of an estuary once severely degraded by nutrient enrichment but now on a recovery trajectory after implementation of nutrient-reduction policies motivated by public engagement and voluntary nutrient management actions.

Other success stories

The recovery of dissolved oxygen in Deep Bay exemplifies successes in many urban estuaries of policies mandating advanced wastewater treatment. Chronic problems of noxious algal blooms, hypoxia and fish kills disappeared from the Potomac Estuary, United States, after sewage inputs of P and BOD (biological oxygen demand) were reduced over 90% (Jaworski, 1990). Sewage treatment also removes toxic contaminants, and monitoring programs documented returns of benthic invertebrates and fish to the heavily polluted Nervión Estuary, Spain (Borja *et al.*, 2006), and Mersey Estuary, United Kingdom (Hawkins *et al.*, 2002), after metal loadings were reduced by advanced wastewater treatment. Three decades of measurements documented significant decreases of phosphorus, ammonium and chlorophyll *a* concentration in the northern Adriatic Sea following improvements in sewage treatment and a ban on phosphates in the Po River basin (Mozetič *et al.*, 2009). Actions that reduced nutrient inputs to Tampa Bay were similar to those taken by the Danish government that reduced N inputs to its coastal waters by 50% and P inputs by 56%. In response, chlorophyll *a* concentrations decreased, water clarity improved and eelgrass meadows expanded, but the anticipated reduced frequency of low-oxygen events was not realized because of confounding effects of changing climate (see below). The Danish example demonstrates that partial recovery from nutrient pollution is attainable at a national scale (Riemann *et al.*, 2015).

Other policies have successfully addressed disturbances such as fishing, flow manipulations and habitat loss. Although many marine fisheries are overexploited, some stocks of estuarine-dependent fish have recovered from overharvest after implementation of protective measures. The once-depleted stock of striped bass in Chesapeake Bay was fully recovered by 1995 after a 1980s management plan was adopted to protect females from harvest before they spawned at least once (Richards & Rago, 1999). Conservation efforts beginning in the mid-20th century have recovered once-depleted populations of several bird species and gray and harbor seals in the Wadden Sea (Lotze *et al.*, 2005). Other policies have returned damaged coastal habitats to a more natural state through flow management and habitat restoration. A 1996 diversion of river water into the Berre Lagoon, France, decreased salinity, caused intense eutrophication with anoxia events and altered biological communities. However, salinity increased, algal biomass decreased and the zooplankton community returned to one characteristic of Mediterranean lagoons after the diversions were restricted by a 2006 ruling of the European Union Court (Delpy *et al.*, 2012). Most of the tidal wetlands around San Francisco Bay were lost to filling or diking by the early 20th century, but a large-scale restoration project began in 2003 and has since reconnected 1200 hectares of commercial salt ponds to the Bay, on target toward a goal of restoring 6100 hectares of wetlands to promote recovery of native plant, fish and waterfowl populations (www.southbayrestoration.org).

Tougher problems remain

A key to these policy successes is that they addressed a well-understood, single-stressor problem where solutions were feasible. Many other policies have failed to meet expectations, and resource management at the sea–land interface is proving to be enormously difficult. Long-term studies provide explanations why. First, our observational records show that transformations of estuarine–coastal ecosystems into states not previously seen are often associated with changes in multiple processes whose effects are difficult to isolate. Collapse of the Caspian Sea's anchovy kilka population was caused by the combined effects of fishing mortality and recruitment failure after the introduction of the non-native predator *Mnemiopsis leidyi* in 1999 (Daskalov & Mamedov, 2007). Overfishing of cod was a primary mechanism of the four-level trophic cascade in the Baltic Sea, but these broad community changes were compounded by the eutrophication-induced spread of hypoxic waters (Carstensen *et al.*, 2014) and changes brought by a shift of the North Atlantic Oscillation Index (NAOI)

to its positive phase (Alheit *et al.*, 2005; Casini *et al.*, 2008). Fishery resources have declined in the South Yellow Sea because of the collective effects of harmful algal blooms, overexploitation and climate change (Li *et al.*, 2014). In all these cases, no single management action will return altered communities to an earlier state.

Outcomes of nutrient-reduction policies have varied across ecosystems and range from the expected downward trajectory of phytoplankton biomass and production measured in Tampa Bay (Fig. 3b) to unexpected resistance to change or even continuing increases of phytoplankton biomass in Chesapeake Bay, some coastal sites in Denmark and the Dutch Wadden Sea (Carstensen *et al.*, 2011). The unanticipated increases of chlorophyll *a* following reduced nutrient inputs reflect another challenge to environmental management as 'shifting baselines' (Duarte *et al.*, 2009) – changes in ecosystem properties (e.g., CO₂, temperature and food-web structure) that regulate the amount of phytoplankton biomass produced per mole of N or P. For example, the chlorophyll *a* increase in San Francisco Bay (Fig. 2c) resulted from a climate-driven trophic cascade that reduced grazing losses and increased the accumulation of algal biomass produced from sewage-derived nutrients (Cloern & Jassby, 2012). Although great progress has been made in reducing N and P inputs to Danish coastal waters, the frequency of low-oxygen events has unexpectedly increased because a changing climate regime (calmer winds, higher air temperature) has intensified stratification and reduced vertical mixing that aerates bottom waters (Riemann *et al.*, 2015).

Many long-term observations contain surprises, and these reflect a third management challenge – our imperfect understanding of how multiple stressors interact to bring about change. Decades of observation in Thau Lagoon, France detected new appearances of picocyanobacteria (*Synechococcus*) and the toxic dinoflagellate *Alexandrium catenella* in 1995. This major ecological change followed two changes that give picoplankton a competitive advantage over diatoms – a 3 °C increase of summer temperature and sewage treatment that reduced phosphate concentrations 10-fold. The unexpected appearance of *Alexandrium* is explained by its mixotrophy – capability of ingesting picocyanobacteria as a food resource. Thus, the combination of warming and oligotrophication created an environment promoting a shift from diatoms to picocyanobacteria and a mixotrophic consumer (Collos *et al.*, 2009). Some surprising changes attract international attention, such as the spectacular green tides of macroalgae (*Ulva prolifera*) that accumulated in N-enriched coastal waters of the southern Yellow Sea and

disrupted sailing competition during the 2008 Olympic games. This problem will be difficult to manage because the sources of nitrogen include accelerating inputs from the Yangtze River and from marine aquaculture sources that doubled in less than a decade (Li *et al.*, 2014). Lastly, the management challenge is confounded by the emergence of previously unidentified ramifications of stressors such as nutrient enrichment. Observational programs show that some coastal waters are becoming acidified faster than the open ocean because of the combined effects of eutrophication (enhanced metabolism and CO₂ production) and atmospheric CO₂ uptake (Cai *et al.*, 2011). This threat to sustainability of calcifying organisms was not recognized a decade ago, but it has become another strong motivation for managing nutrient supplies to estuaries (Wallace *et al.*, 2014).

Discoveries from sustained observation

Our review identified a wide range of physical, biogeochemical and biological changes in the world's estuarine–coastal ecosystems, illustrated with 50 examples in Table 1. Four general lessons emerge from a synthetic view of changes captured by repeated observation over time.

Fast paced change

Time series are valuable because they provide rates of environmental changes, many of which occur faster than we could have anticipated a few decades ago. Examples above showed 200-fold growth of *Mya arenaria* biomass in Ringkøbing Fjord in only 4 years; 60% reduction of primary production in Tampa Bay over 7 years; 50% reduction of dissolved oxygen in Deep Bay over a decade; and an abrupt decline in number of benthic invertebrate species in the Bay of Brest. Large annual variability and fast-paced change seem to be characteristic features of estuarine–coastal ecosystems. As examples, winter temperature in Gyeonggi Bay, Korea, increased 3.5 °C from the 1960s to the 2000s (Jahan & Choi, 2013). The dinoflagellate component of spring bloom biomass in the Gulf of Finland increased from 10% to 70% over a decade (Klais *et al.*, 2011). Riverine inputs of dissolved inorganic P to the Changjiang Estuary increased 574% from 1970 to 2013 (Wang *et al.*, 2014). Oxygen and more than 50 fish species returned to the anoxic and fishless Thames Estuary within a decade after advanced sewage treatment was implemented (Andrews & Rickard, 1980). But Egypt's coastal fisheries collapsed within 5 years after completion of the Aswan Dam (Nixon, 2003), and Turkey's annual

anchovy catch in the Black Sea plummeted from 338 000 to 15 000 tons in just 3 years (Mee, 1992).

Many forms of change

The patterns of change contained in time series are diverse and include linear trends of, for example: earlier seasonal development of cyanobacteria blooms in the Baltic Sea (Fig. 2b), warming of Narragansett Bay surface waters over four decades (Borkman & Smayda, 2009b), decreasing summer pH in lower Chesapeake Bay over three decades (Waldbusser *et al.*, 2010), decreasing silicate concentrations in shelf waters of the NW Black Sea from the 1960s to mid-1980s (Yunev *et al.*, 2007), sea-level rise along the US Pacific coast from 1935 to 2000 (Cayan *et al.*, 2008) and decreasing transparency (Secchi depth) in the Ariake Sea, Japan (Hayami *et al.*, 2014). Other trends are nonlinear, such as the accelerating expansion of Chesapeake Bay's hypoxic volume from 1950 to 2000 (Hagy *et al.*, 2004) and accelerating rates of sea-level rise projected under some scenarios of global warming (Cayan *et al.*, 2008). Other patterns include inflection points between, for example, eras of decreasing and then increasing dissolved oxygen in Deep Bay (Fig. 3a); eight decades of steadily increasing point-source N input to Danish straits followed by two decades of decreasing N input (Conley *et al.*, 2007); and loss of eelgrass (*Zostera marina*) in Elkhorn Slough, United States, from 1965 to 1980 followed by two decades of eelgrass recovery (Hughes *et al.*, 2013).

The era of depressed fishery landings off the Nile River delta after completion of the Aswan Dam was followed by a surprising recovery of the fishery attributed to another human disturbance – increasing inputs of N and P from fertilizer and sewage runoff (Oczkowski *et al.*, 2009). Pattern complexity builds as time series lengthen, and a striking example is the 1918–2009 series of pelagic fish catch in Denmark's Limfjorden showing high-amplitude oscillations with periods of several decades (Riisgård *et al.*, 2012). The few records of this duration provide an important context for interpreting observations from only a few decades because ecological time series are nonstationary – their variance grows as they are extended and capture more processes of change (Cloern & Jassby, 2012).

Perhaps the most surprising discovery from recent decades of observation is that abrupt, ecosystem-scale changes are common. We showed examples as sharp increases of *Mya* in Ringkøbing Fjord (Fig. 1a) and chlorophyll *a* in San Francisco Bay (Fig. 2c) and decreased infauna diversity in the Bay of Brest (Fig. 1b). Many other abrupt changes have been observed such as the restructuring of benthic faunal

Table 1 Fifty examples of change measured in the world's estuarine-coastal ecosystems by repeated observations over time

| Place | Observational period | Change | Mechanism of change | Reference |
|----------------------------|----------------------|---|--|-----------------------------------|
| Baltic Sea | 1950–1970 | 4-Fold expansion of hypoxic zones | Nutrient enrichment | Carstensen <i>et al.</i> (2014) |
| Chesapeake Bay, USA | 1950–2001 | Increasing volume of summer hypoxia from 0 to 3.6×10^9 m ³ | Nutrient enrichment | Hagy <i>et al.</i> (2004) |
| Mississippi Delta, USA | 1950–2006 | >80% Reduction of sediment input | River damming and water diversions | Blum & Roberts (2009) |
| Danish coastal waters | 1950s–1970s | Doubling of primary production | Nutrient enrichment | Ærtebjerg Nielsen & Gargas (1984) |
| Chesapeake Bay, USA | 1950s–1990s | 5- to 10-Fold chl- <i>a</i> increase in seaward regions | Nutrient enrichment | Harding & Perry (1997) |
| Huanghe River delta, China | 1950s–1990s | >60% Reduction in river discharge and sediment load | River regulation | Fan & Huang (2008) |
| Bohai Sea, China | 1950s–1990s | Reduction of fish diversity from 146 to 73 species | Water diversions and nutrient enrichment | Fan & Huang (2008) |
| Narragansett Bay, USA | 1950s–2000s | Reduced chl- <i>a</i> , primary production, benthic metabolism (oxygen uptake, nutrient regeneration) and demersal fish biomass | Warming, increased cloudiness and reduced nutrient inputs | Nixon <i>et al.</i> (2009) |
| San Francisco Bay, USA | 1950s–2010s | Reduced sediment supply, shift from net deposition to net erosion, 40% reduction of freshwater inflow, reorganizations of biological communities, elimination of hypoxia | River damming, water diversions, species (>200) introductions, advanced sewage treatment | Cloern & Jassby (2012) |
| Changjiang Estuary, China | 1955–2008 | 70% Reduction in suspended sediment input | River damming (Changjiang) | Yang <i>et al.</i> (2011) |
| Narragansett Bay, USA | 1959–1997 | Shift from a winter-spring bloom pattern to a summer bloom pattern | Altered climate (temperature, solar radiation, wind) associated with positive NAO | Borkman & Smayda (2009a) |
| Mediterranean Coast, Egypt | 1960–1970 | 7-Fold loss of fisheries landings | River damming (Nile) | Nixon (2003) |
| Black Sea shelf waters | 1960s–1990s | Decreasing silicate concentration at rates from 0.7–1.5 $\mu\text{M yr}^{-1}$ | River damming (Danube) | Yunev <i>et al.</i> (2007) |
| Thames Estuary, UK | 1964–1979 | Increased fish diversity from 2 to 98 species | Advanced wastewater treatment | Andrews & Rickard (1980) |
| Chesapeake Bay, USA | 1966–2002 | Increased chl- <i>a</i> and hypoxia; decreased landings of oyster (<i>Crassostrea virginica</i>) and abundances of copepod <i>Acartia tonsa</i> and sea nettle <i>Chrysaora quinquecirrha</i> | Nutrient enrichment, warming, altered species interactions | Kimmel <i>et al.</i> (2012) |
| Potomac Estuary, USA | 1970–1985 | 60% Reduction of algal growth, increased DO and decreased fecal coliform density | Advanced sewage treatment that reduced P and BOD loadings | Jaworski (1990) |
| Gulf of Trieste, Italy | 1970–2005 | Doubling of copepod biomass and advancement of seasonal biomass peaks | Warming and altered circulation patterns | Conversi <i>et al.</i> (2009) |

Table 1 (continued)

| Place | Observational period | Change | Mechanism of change | Reference |
|------------------------------|----------------------|--|--|--------------------------------|
| Changjiang Estuary, China | 1970–2013 | 338% Increase in riverine DIN input and 574% increase in riverine DIP input | Intensification of agriculture and animal production | Wang <i>et al.</i> (2014) |
| Bay of Brest, France | 1970s–1990s | Shift in the seasonal pattern of diatom production | Altered Si:N ratio and introduced species (<i>Crepidula fornicata</i>) | Chauvaud <i>et al.</i> (2000) |
| Narragansett Bay, USA | 1970s–2000s | Sediments transformed from a net source to net sink of N ₂ gas | Climate-induced decrease in primary production | Fulweiler <i>et al.</i> (2007) |
| Black Sea shelf waters | 1971–1985 | 52% Reduction of freshwater input from the Dnieper River | River damming and water diversions | Mee (1992) |
| Elkhorn Slough, USA | 1971–2011 | Two decades of eelgrass (<i>Zostera marina</i>) decline followed by three decades of recovery | Nutrient enrichment, sea otter recovery | Hughes <i>et al.</i> (2013) |
| Ariake Sea, Japan | 1972–2010 | 35% Reduction in autumn Secchi depth | River channelization and migration of the turbidity maximum | Hayami <i>et al.</i> (2014) |
| Atlantic Coast, USA | 1973–1983 | 90% Reduction of striped bass commercial catch | Over harvest | Richards & Rago (1999) |
| Mediterranean Coast, Egypt | 1975–2001 | >10-Fold increase in fisheries landings | Nutrient enrichment | Oczkowski <i>et al.</i> (2009) |
| North San Francisco Bay, USA | 1977–1990 | Loss of the summer chl- <i>a</i> maximum and 60% reduction in primary production | Introduced species (<i>Potamocorbula amurensis</i>) | Alpine & Cloern (1992) |
| Gironde Estuary, France | 1978–2009 | chl- <i>a</i> decline | Shift to a climate pattern of reduced precipitation, river inflow, and nutrient concentrations | Chaalali <i>et al.</i> (2013) |
| South San Francisco Bay, USA | 1978–2013 | Abrupt increase of summer chl- <i>a</i> | Trophic cascade induced by a North Pacific climate shift | Fig. 2c |
| Central Baltic Sea | 1979–1996 | Increased dinoflagellate abundance and decreased abundance of diatoms and copepod <i>Pseudocalanus</i> sp. | Shift to a positive phase of the winter NAO (warmer, stronger westerly wind) | Alheit <i>et al.</i> (2005) |
| Tampa Bay, USA | 1979–2008 | 50% Reduction of phytoplankton primary production | Policy to reduce N and P loadings | Fig. 3b |
| Baltic Sea | 1979–2013 | 20-Day advancement of the center (date) of cyanobacteria bloom development | Synchronous with a warming trend (Figure 2b inset) | Fig. 2b |
| Baltic Sea | 1980–2006 | 4-Fold decline of cod biomass, 3-fold increase of spratt biomass | Cod harvest and shift to positive NAO index | Casini <i>et al.</i> (2008) |
| Yellow Sea coast, China | 1980s–2010s | First massive macroalgae (<i>Ulva prolifera</i>) green tide in 2007 | Nutrient enrichment | Li <i>et al.</i> (2014) |
| Atlantic Coast, USA | 1981–1995 | >10-fold increase in recreational striped bass CPUE | Harvest regulation | Richards & Rago (1999) |

Table 1 (continued)

| Place | Observational period | Change | Mechanism of change | Reference |
|----------------------------------|----------------------|---|---|---------------------------------|
| Forth Estuary, UK | 1982–1999 | Nearly 10-fold reduction in mercury content of mussels (<i>Mytilus edulis</i>) | Policy to reduce metal inputs from industrial and domestic sources | Dobson (2000) |
| Chesapeake Bay, USA | 1985–2008 | Significant pH decreases in polyhaline waters | Nutrient enrichment | Waldbusser <i>et al.</i> (2010) |
| Thau Lagoon, France | 1987–2006 | New appearances of picocyanobacteria and toxic dinoflagellate <i>Alexandrium catenella</i> | Summer warming and 10-fold reduction in SRP | Collos <i>et al.</i> (2009) |
| Deep Bay, Hong Kong | 1988–2012 | Significant decline, followed by significant increase of DO | Policy to reduce N and P loadings | Fig. 3a |
| Nervión Estuary, Spain | 1989–2003 | 4–6 Fold increase in species richness of benthic macrofauna | Advanced wastewater treatment | Borja <i>et al.</i> (2006) |
| Ringkøbing Fjord, Denmark | 1989–2006 | >10-fold increase in benthic invertebrate biomass, >3-fold decrease of zooplankton biomass, >5-fold increase of chl-a | Altered water exchange | Petersen <i>et al.</i> (2008) |
| Gulf of Trieste, Italy | 1989–2009 | Reduced nutrient concentrations, chl- <i>a</i> and zooplankton biomass | Climatic era of low precipitation and runoff | Mozetič <i>et al.</i> (2012) |
| Danish coastal waters (45 sites) | 1990–2013 | 50% reduction of N inputs and 56% reduction of P inputs | Policy to reduce N and P loadings | Riemann <i>et al.</i> (2015) |
| Danish coastal waters (45 sites) | 1990–2013 | Significant trends of decreasing N, P, chl- <i>a</i> and macrofauna biomass | Policy to reduce N and P loadings | Riemann <i>et al.</i> (2015) |
| Caspian Sea | 1991–2004 | Near complete loss of anchovy kilka (<i>Clupeonella engrauliformis</i>) spawning stock | Introduced species (<i>Mnemiopsis leidyi</i>) | Daskalov & Mamedov (2007) |
| Odense Fjord, Denmark | 1992–2008 | >2-fold increase in sediment oxygen consumption and CO ₂ and NH ₄ release | Nutrient reduction and introduced species (<i>Arenicola marina</i>) | Kristensen <i>et al.</i> (2014) |
| Gyeonggi Bay, Korea | 1992–2010 | Shift from spring and autumn diatom blooms to winter diatom and summer dinoflagellate blooms | Climatic era of increased temperature, precipitation and solar radiation and calmer winds | Jahan & Choi (2013) |
| Bay of Brest, France | 1992–2011 | 37% Decrease of benthic infaunal species richness | Bottom disturbance by dredging harvest of clams | Fig. 1b |
| Patos Lagoon, Brazil | 1993–2012 | Salinity fluctuations between 0 and 35 | Runoff variability tied to ENSO | Fig. 2a |
| Northern Adriatic Sea | 1998–2007 | Decreasing chl- <i>a</i> trend (>5% yr ⁻¹) | Sewage treatment and phosphate detergent ban | Mozetič <i>et al.</i> (2009) |
| Yellow Sea coast, China | 2003–2011 | Doubling of DIN load | Mariculture | Li <i>et al.</i> (2014) |

BOD, biological oxygen demand; CPUE, catch per unit effort; DIN, dissolved inorganic N; DIP, dissolved inorganic P; DO, dissolved oxygen; N, nitrogen; ENSO, El Niño Southern Oscillation; NAO, North Atlantic Oscillation; P, phosphorus; SRP, soluble reactive P.

communities and their biogeochemical functions in Denmark's Odense Fjord (Kristensen *et al.*, 2014), doubling of copepod abundance in Gulf of Trieste

(Conversi *et al.*, 2009) and decreases of chlorophyll *a* in the northern Adriatic Sea (Mozetič *et al.*, 2009) and Gulf of Trieste (Mozetič *et al.*, 2012), fish populations in the

Caspian Sea (Daskalov & Mamedov, 2007) and nitrogen-assimilative capacity of the Skagerrak (Conley *et al.*, 2007).

Phytoplankton as indicators of change

Monitoring programs in estuarine–coastal waters usually include some component of phytoplankton sampling as an indicator of change that can inform strategies of ecosystem-based management such as the European Union's Marine Strategy Framework Directive (Mc Quatters-Gollop *et al.*, 2015). Time series of phytoplankton biomass and community composition have been sustained long enough to begin revealing how phytoplankton communities respond to changes in the climate system. Over three decades of observation in South San Francisco Bay (NE Pacific) detected abrupt changes after the NPGO became positive in 1999: increased primary production, threefold chlorophyll *a* increase during summer–autumn (Fig. 2c), a shift from a seasonal pattern of spring blooms to a pattern of spring and autumn blooms (Cloern *et al.*, 2007), appearances of new species and disappearances of others (Cloern & Jassby, 2012). Phytoplankton communities respond similarly to changes in climate patterns across all ocean basins. A 1992–2010 time series from Gyeonggi Bay, Korea (NW Pacific), documented a state change after 1999 when mean chlorophyll *a* concentration increased and the seasonal pattern of spring and autumn diatom blooms changed to a pattern of winter diatom and summer dinoflagellate blooms (Jahan & Choi, 2013). A 38-year time series from weekly phytoplankton sampling in Narragansett Bay, United States (NW Atlantic), detected an abrupt change in winter–spring abundance of *Skeletonema* spp. separating a 1959–1980 era of high-abundance from a 1980–1997 era of low abundance. Loss of this key diatom species signaled a shift away from an annual cycle of winter–spring blooms in the 1960s to a summer bloom pattern in the 1990s (Borkman & Smayda, 2009a). A 1981–1999 time series from the central Baltic Sea showed an era of declining diatom biomass and increasing dinoflagellate biomass after 1987 (Alheit *et al.*, 2005), and chlorophyll *a* began to increase in the Gironde Estuary, France (NE Atlantic), at the same time (Chaalali *et al.*, 2013). Each of these changes followed a change in a large-scale climate pattern – the North Pacific Gyre Oscillation (San Francisco Bay), the winter Monsoon Index MOI (Gyeonggi Bay), the Gulf Stream northern Wall index (Narragansett Bay) and/or the North Atlantic Oscillation Index (Narragansett Bay, Baltic Sea, Gironde Estuary).

A grand challenge now is to understand the mechanisms underlying these changes in phytoplankton

biomass, phenological patterns, primary production and community composition. Mechanistic links to the climate system are obscure (Borkman & Smayda, 2009b; Jahan & Choi, 2013), but clues are emerging as time series are extended and capture changes in ocean–atmosphere circulation patterns as natural experiments. The chlorophyll *a* increase in San Francisco Bay was a response to strengthened coastal upwelling and high production of marine phytoplankton and predators that were carried or migrated into the Bay. Changes in Gyeonggi Bay occurred during a period of calm winds and winter warming after the MOI decreased in 1988–1989. Warming advanced the timing of stratification that triggers blooms and explains the shift of bloom timing from spring to winter (Jahan & Choi, 2013). Warming of the Baltic Sea during an era of positive NAOI similarly altered seasonal mixing patterns that shortened duration of the spring diatom blooms and prolonged stratification that selects for dinoflagellates (Alheit *et al.*, 2005). A 1987 state change in the Gironde Estuary occurred following a transition to a warm–dry era when the estuary became strongly influenced by intrusion of marine water and its plankton communities (Chaalali *et al.*, 2013). At a coarse level, we now understand from time series collected at many sites that the climate–phytoplankton linkage involves many processes including changes in wind speed and direction, solar radiation, freshwater inflow, temperature, seasonal stratification patterns, coastal currents and source waters that exchange with estuaries, and marine organisms that migrate into estuaries.

Change is systemic

Multidecadal observations also show that changes in phytoplankton dynamics are integral components of broader, ecosystem-scale changes across all biological communities and environmental factors that regulate them. For example, systemic changes in Chesapeake Bay occurred as a succession of change points beginning in the late 1960s that included (i) increased water temperature, volume of hypoxic water and chlorophyll *a* concentration and (ii) decreased abundance of the copepod *Acartia tonsa*, production of juvenile oysters (*Crassostrea virginica*) and fisheries landings that were attributed to eutrophication, warming, overfishing and shifting winds (Kimmel *et al.*, 2012). Habitat loss, over-exploitation and pollution restructured biological communities in the Wadden Sea where eelgrass meadows and habitat-building invertebrates have been lost, opportunistic green algae have replaced red algae and 144 species including whales, some bird species, diadromous fish and large groundfish are extinct or

severely depleted (Lotze *et al.*, 2005). The sediment supply, geomorphology, hydrology, habitat matrices and connectivity, water chemistry and biological communities of San Francisco Bay have been transformed over the past century by river damming, water diversions, urban and agricultural pollution, over harvest and introductions of hundreds of plant and animal species (Cloern & Jassby, 2012).

Long-term observations also reveal the tight coupling between changes in biological communities and the biogeochemical functions they provide. For example, observational time series from Narragansett Bay illustrate the ecosystem-scale ramifications of changing phytoplankton dynamics. Systemic changes occurred during a period of warming and increased winter cloudiness when phytoplankton biomass and primary production declined and the winter–spring bloom disappeared (Nixon *et al.*, 2009). Loss of primary production during the cold season reduced the supply of organic carbon to the sediments, resulting in a 75% loss of demersal fish biomass, reduced benthic metabolism (oxygen consumption, nutrient regeneration) and a surprising change in N-cycling as bay sediments switched from functioning as a net producer to a net consumer of N₂ gas (Fulweiler *et al.*, 2007). This example illustrates the unique value of time series for discovering systemic changes as linkages between variability of climate, biological communities and biogeochemical functioning of estuaries.

Time series from the Bay of Brest illustrate two other processes of coupled biological and biogeochemical change. First, nitrogen enrichment of the tributary rivers over a period of decades reduced the Si:N molar ratio in the Bay from 2 to 0.33, leading to Si-limitation of diatom production during spring blooms. Loss of spring diatom production was balanced by a second change, introduction of the mollusk *Crepidula fornicata* whose filter feeding retained biogenic Si (diatom frustules) in sediments that was later mineralized to support new diatom production in summer (Chauvaud *et al.*, 2000). Thus, the combined effects of nitrogen enrichment and introduction of a filter feeder altered the seasonal cycles of silica availability and diatom production. This example also illustrates that phenological shifts can be driven by processes other than changing climate.

Concluding perspective

Observational programs sustained over decades reveal the power of human actions to transform ecosystems at the sea–land interface and the responsiveness of these ecosystems to decadal fluctuations of the climate system.

Many estuarine–coastal ecosystems appear to be in a continuing state of change, so the concept of variability around a quasi-equilibrium state is not useful for guiding research or management. Instead, human actions are creating novel ecosystems with habitats, biogeochemistry and biological communities outside the natural range of variability. Thus, management toward ecosystem reconciliation (Rosenzweig, 2003) has become a more realistic goal than restoration toward some earlier state.

Patterns of change are complex, and the challenge of unraveling the causes of ecosystem change is daunting in the coastal zone where human pressures are concentrated and effects of climate variability over land intersect with different effects over ocean basins. Complexity also arises from the barely-understood synergistic effects of multiple processes such as nutrient enrichment, species introductions, fishing and changing climate. Thus, effective management of coastal ecosystems requires a holistic systems perspective grounded in research to discover how multiple processes operate together to bring about transformative change. Discoveries that come uniquely from sustained observations must play a central role in this research.

Observational programs also reveal how estuaries function as systems where all component parts – habitats, connections to land and ocean, biological communities and biogeochemical processes – are tightly interlinked. Changes in one have effects that ramify throughout the system. This implies that simple measurements such as salinity, temperature, dissolved oxygen and chlorophyll *a* are powerful indicators of ecosystem-scale change.

Although we know with certainty that changes will continue into the future, there is great uncertainty about how those changes will unfold. Ecological time series are nonstationary, so reliable forecasting cannot

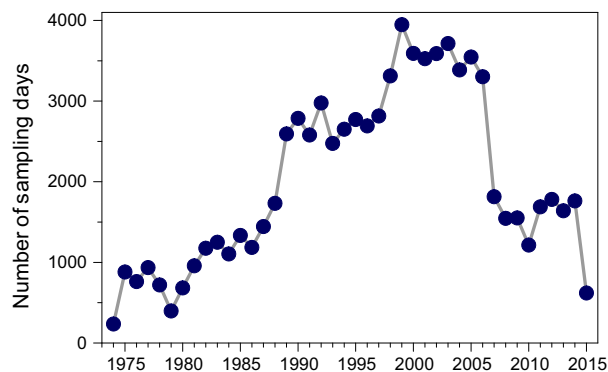


Fig. 4 Time series of the number of sampling days per year in Danish coastal waters, showing decreased effort at water-quality monitoring the past decade (from the Danish National Aquatic Monitoring and Assessment Program).

be based on patterns of historical change. As in water management, ecosystem management will require new tools for dealing with a nonstationary world where continuity of observations is critical (Milly *et al.*, 2008).

Unfortunately, even in the face of fast, systemwide changes that cannot be predicted, observational programs are increasingly difficult to fund and existing programs are at continuing risk of elimination (McQuatters-Gollop *et al.*, 2015). Denmark's National Aquatic Monitoring and Assessment Program provides an iconic example of the value of observational data to measure the effectiveness of environmental policies and explain why they do not always meet expectations. But the sampling effort of this program has been reduced by half over the past decade (Fig. 4), reflecting a global trend. For example, measurement of primary production in Tampa Bay (Fig. 3b) stopped in 2010, ending the longest time series of phytoplankton production in the world's estuaries (Cloern *et al.*, 2014), and zooplankton sampling by the Chesapeake Bay Program was terminated in 2002 (Kimmel *et al.*, 2012). In the scientific community, we must become more effective at communicating to policymakers the critical importance of sustaining and expanding observational programs to accumulate the knowledge required to cope with the inevitable changes that will develop as the planet continues to warm and as human pressures continue to grow along coasts.

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