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Five critical questions of scale for the coastal zone

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Introduction

The social and ecological systems that comprise human society and its living space are becoming increasingly globalized. From the standpoint of understanding ecosystem behavior, it is getting harder to model a system within their apparently well-defined boundaries because these are ceasing to be the relevant ones. A flutter in the stock market in Tokyo or Hong Kong can affect salmon producers in Norway or farmers in Togo. Concurrent with the globalization of trade, we are witnessing the globalization of the distribution of opportunistic species and the disempowerment of people attempting to manage their own affairs on a local scale. Climate change, as well as other human-accelerated environmental change, can really exacerbate this sense of disenfranchisement. The structure and functioning of coastal ecosystems has evolved over millennia, subject to environmental forces and constraints imposed mainly by local boundaries of catchment and shore, not human culture. However, phenomena that transcend these local scales have emerged as human populations have increased, and especially with the rise of modern global culture. Much of the science of coastal waters is based on abstract considerations of ecosystem behavior in the absence of human "perturbations", such as long-distance transport of materials, introduction of new species, and human accelerated climate and land use change. How can we keep up with the change in scales (temporal and well as spatial) associated with such human activities? What tools can we employ to help people and policymakers to remain in charge in a meaningful manner?

Some critical questions of scale for the coastal zone:

1) *How big are coastal ecosystems and why should we care?*
2) *Temporal scales of change in coastal waters and watersheds: Can we detect shifting baselines due to economic development and other drivers?*
3) *Are footprints more important than boundaries?*
4) *What makes a decision big? The tyranny of small decisions in coastal regions.*
5) *Scales of complexity in coastal waters: the simple, the complicated or the complex?*

1) How big are coastal ecosystems and why should we care?

The short answer to the first half of the question is that coastal ecosystems, and the various entities which comprise and interact with them, span a range of scales, depending upon the definition of "scale" (see Appendix 1, supplementary material) and the issues to be analyzed. For example, an analysis of nutrient inputs and eutrophication might be conducted along the boundaries of the catchment-coast continuum, or include an analysis of agricultural systems and businesses at global, continental (e.g. Europe or at least EU wide) or national policy scales. The question of scale is a fundamental challenge for coastal research necessitating a variety of research approaches and requiring extrapolation from mesocosms in many cases (Boynton et al. 2001). While it has become a truism that "scale" is an important dimension in coastal science and management, it is critical to distinguish its manifestation in different categories of processes (Fig. 1). Physical scale effects are often propagated via intermediate-scale processes, (e.g. energy transferred via turbulent eddies from large areas and time scales to the smallest scales of turbulent mixing). Temporal scales tend to correlate linearly with spatial scales, while ecological effects can "jump" intermediate scales or simultaneously be manifest on different scales (e.g., long-lived species and their structures which tend to result in "memory" effects in ecosystems), so it is more difficult to consider a single time scale corresponding to the spatial scale of an ecosystem. Governance entities, including government agencies, local communities, and the full range of “stakeholder” groups in the coastal zone, exhibit influence at overlapping scales, and are subject to hierarchies and gaps in coverage. Drivers exhibit overlapping asymmetrical effects (large scale drivers affect small scale systems, but not conversely). These effects make selection of a single scale of analysis problematic, and the result is that we must consider a superposition of scales in most cases involving governance of coastal waters (Solidoro et al., 2010a).

<Figure 1 near here>

The answer to the second half of the question is fairly obvious: “Because the response of coastal ecosystems to various pressures (changes in material inputs, freshwater discharge, temperature, etc) depends on how big they are.” Studies of coastal ecosystems frequently attempt to relate physical or biological processes within the systems to characteristic scales, (volume, area, residence time). During the first decade of the Land Ocean Interactions in the Coastal Zone program (LOICZ), coastal nutrient budgets were developed for coastal systems spanning six orders of magnitude in surface area (Smith et al., 2005; Swaney et al., in press). Based on some statistical analyses of LOICZ coastal nutrient budgets, it appears that the net ecosystem metabolism of coastal systems varies inversely with system size, i.e., large systems with long residence times have relatively low metabolic rates, possibly because they are colder, deeper, darker, and may have lower nutrient inputs/volume on average than smaller systems (Fig. 2). This *allometric* relationship (1, supplementary materials) appears to be true whether the systems are heterotrophic (ecosystem respiration exceeding production) or autotrophic (production exceeding ecosystem respiration). Robust statistical analyses using the Olmstead-Tukey test (Olmstead and Tukey, 1947; Sokal and Rohlf, 1995) indicate that such inverse relationships are significant for net-autotrophic systems using either residence time or area as a scale measure, and for net heterotrophic systems using residence time. Other studies have noted that small land margin systems (e.g. coastal lagoons) have relatively high primary and secondary productivity and fish catches compared to larger coastal systems (Houde and Rutherford, 1993). If the productivity of a coastal system varies strongly with its size, then it is likely that management of the system should also vary with size, i.e., one management or decision-making approach should not apply to coastal systems across all scales (Kannen et al 2008). So it seems that for coastal systems, one size does not fit all.

<figure 2 near here>

The size question is also relevant to the globally-aggregated productivity of coastal regions. It is well known that coastlines vary in length depending upon the resolution with which they are measured. This fractal property may also apply to the distribution of sizes of various coastal features: for example, Pareto (power law) distributions are well-known to describe island areas, lakes, and other natural features. If they apply to coastal seas, embayments, estuaries and lagoons, it suggests that small systems are numerically dominant, and possibly critical to overall coastal productivity. The nonlinear nature of the response of these systems (Folke et al., 2004), and their associated ecosystem services (e.g., Barbier et al., 2008) as well as their spatial and temporal variability (Luisetti et al., 2011) implies that there should be a strong scale-dependence in many of the benefits humans obtain from coastal ecosystems as well (Koch et al., 2009). The combination of an uneven distribution of the size of coastal ecosystems, and the nonlinear variation of ecological processes and ecosystem services to ecosystem scale implies that simple area-based extrapolations of estimates of the global contribution of coastal regions probably underestimate their value to human society (Duarte et al., 2008)..

While the above can be considered a variation of the well-established idea that localized “hotspots” or “patches” provide most of the activity in most ecosystems, it suggests that hotspots are dominant features of the coastal zone. Another view is that the cumulative value of a collection of many small, highly-variable autotrophic and heterotrophic systems is simply the (relatively low) productivity of the mean value of the collection. Averaging out the extremes of productivity of the components of a large system is just a property of the central tendency of the statistics, and so the apparent inverse relation between size of a system (measured in various ways) and magnitude of processes (metabolism, percent nutrient export, etc) is the result. Are these fundamental issues relevant to the governance of coastal systems, or is it simply another manifestation of environmental heterogeneity of which we must be mindful in management?

2) Temporal scales of change in coastal waters and watersheds: Can we detect shifting baselines and regime shifts due to economic development and other drivers?

History is long; human memory is short. For most of their history, coastal ecosystems have evolved and developed under the joint influence of their local (though sometimes large) coastal watersheds and oceanic influences. Human accelerated environmental change, particularly landscape transformations, typically beginning with settlement of colonial regions, but also associated with the rise of new technologies, has dramatically increased the influence of terrestrial drivers. These changes include increased nutrient loads and pollutants from a variety of sources; altered hydrological and sediment transport regimes caused by impoundments, other hydraulic engineering structures, and increased impervious surface; and introduced invasive species from microbes to mammals. Many of these effects have been shown clearly in the Chesapeake by Brush (2008) using paleoecological methods and historical records (Fig. 3), and similar impacts are widely recognized in other watersheds (Meybeck, 2003; Turner and Rabalais, 2003; Swaney et al., 2006; Billen et al., 2007; Humborg et al., 2007; Nixon et al., 2008; Solidoro et al., 2010b). Many changes are due to increases in infrastructure – impervious surface, land drainage, damming, hardened coastline, dredging to facilitate shipping, and consequent modification of watershed and estuarine hydrological, biogeochemical and biological processes. For example, the joint effects of dams in reducing sediment transport and projected sea level rise in increasing inundation poses a major threat to the world’s major deltaic systems (Ericson et al., 2006; Woodroffe et al., 2006; Syvitsky et al., 2009). Several studies (e.g., Restrepo et al., 2006; Richmond et al., 2007) have demonstrated the negative impact of runoff and sediment loading to seagrasses and coral systems, even beyond what is typically perceived to be the immediate receiving waters of coastal watersheds, and the need for incorporating watershed management in dealing with these impacts.

While the increase in nutrient loads caused by changes in agriculture practices and land use which occurred during last century triggered eutrophication phenomena over most European and American coastal areas, a reversal in eutrophication trends (so called oligotrophication) is being seen in recent decades in many areas (Stockner et al. 2000, Nixon et al., 2009; Mozetic et al, 2010), probably because of the enforcement of new environmental protection legislation in these regions. This suggests on the one hand that eutrophication trends are reversible, but on the other opens the question of unintended consequences for secondary production, including fisheries (Nixon et al., 2009)

<figure 3 near here>

Direct human modification of coastal lands and waters also continues to increase. Lotze et al. (2006) showed patterns of species and habitat loss, species invasions, and water quality degradation in estuaries and coastal seas around the world. Fifty-five percent of the mangrove areas on the coast of Thailand were lost between 1961 and 1996 through the development of shrimp farming, urban development, mining and other factors, though in recent years there has been some recovery (Giesen et al., 2006). The increasing prevalence of fin- and shellfish aquaculture in coastal waters has significantly modified their nutrient regimes in some cases, either through the direct introduction of nutrients in food or the removal of nutrients with harvest (e.g., Yamamuro et al., 2006). Increased port development and shipping activity associated with the growth of trade has affected coastal water quality in and near ports, including the dredging of channels for increasingly large cargo vessels (Wolanski, 2006). Such disturbance may cause increases in turbidity, alter the flows and flowpaths, and adversely affect fish nursery areas.

Oceanic and climatic influences are also changing and will continue to do so. The coastal zone in the sense of LOICZ (-200 m depth to +200 m) has been shaped by global (eustatic) sea level variations and associated cycles of transgressions and regressions over geological time scales. Reconstructions of (eustatic) sealevel change over the last few hundred thousand years suggest that the present sea level is near the maximum reached over the last 450,000 years. Isostatic (regional) sea level varies in present-day coastal zones, but there is now unequivocal evidence for a small, but crucial, eustatic sea level rise caused mainly by thermal expansion of sea water (Bindoff et al. 2007). Projections of sea level change through 2100 AD predict an increase of between 20 cm and 50 cm in global mean sea level, not including expected regional effects of increased storm surge height or frequency (Fig. 4. Bindoff et al 2007). While this increase may not seem to be dramatic, it could have consequences for coastal protection, and severe ones for extant tidal flats. Whether situated in estuaries, deltas, or on tidal coasts, these play important roles as specific coastal habitats, biodiversity hot spots, and for mitigation of nutrient fluxes from land to sea in the coastal zone. It is particularly dangerous to sinking deltas (Syvitsky et al., 2009). In the geological past, the coastal zone was effectively a "ramp" that accommodated rising/falling sea level by landward/seaward displacement of individual sedimentary and coastal regimes. That degree of spatial freedom is reduced when the coast is enforced by dikes and other structures ("habitat squeeze"). In the narrowing strip between an advancing sea and a fixed coastline, energy of the water increases and results in preferential selection of coarse sediments; fine particles and organic matter are less effectively retained, and are advected to adjacent deeper water. Among other effects, this can decrease areas of tidal flats, and modify river deltas (Fagherazzi et al. 2006).

<fig 4 near here>

The addition of increased storm activity and flooding superimposed on sea level rise projections is problematic. For example, combined storm surge and sea level rise is projected to pose a risk of flooding to the Mekong Delta, despite extensive damming upriver in China (Hoa et al., 2007).   Increasingly, large-scale linkages are being recognized between atmospheric phenomena (such as the Southern Oscillation, ENSO, or the North Atlantic Oscillation, NAO) and the variation of intensity of regional weather patterns (so called "teleconnections"; Wallace and Gutzler, 1981).

Floods and droughts are not the only outcomes of teleconnections. Winter temperature, which is correlated with the NAO in northern Europe (Fig. 5) has a primary role in the development of zooplankton blooms in the subsequent spring and summer (Alheit et al, 2005). Other direct impacts of climate on coastal waters include effects of temperature (air temperature, sea surface temperature (SST), extremes, and gradients), precipitation (e.g. its seasonality) and hydrological regime (flushing rates and circulation) on changes in productivity (Cossarini et al. 2008) and the seasonal timing of its onset and decline (phenology), as well as the degree of benthic-pelagic coupling (Nixon et al., 2008).

A problem shared by many coastal ecosystems subject to changes in terrestrial, oceanic or climatic influences is the lag between cause and effects: we lack early warning indicators of sudden ecological responses to change. Many coastal ecosystems are exposed to gradual changes (climate change, nutrient loading, fishing pressure etc) and are conventionally assumed to respond smoothly to these changes. However, this is not always the case. Examples of abrupt ecological responses to environmental change include:

<figure 5 near here>

- Human-supported species invasions, especially through ballast water transfer (Carlton, 1999; 2001).
- Invasions driven by climate change favoring one species over another, as when a new species is able to displace existing species (Schluter et al., 2008;.Cheung et al., 2009)
- Natural cycles which involve drastic changes, such as long-term cycles in fish stocks (Silvert and Smith 1981, Silvert 1983, Silvert and Crawford 1988; Alheit and Hagen, 1997).
- "Regime shifts" driven by climate change, pollution or overfishing (deYoung et al., 2008).

Increasing numbers of studies show that smooth changes can be interrupted by abrupt transitions to a new state (e.g., Folke et al., 2004; Duarte et al., 2008), so called regime shifts, One perception of regime shifts is that they are transitions from one quasi-stationary state of a system persisting for several years and characterised by low-frequency variability to another stable state with a transition period of a few years. Gradual and possibly disparate changes have little effect until they surpass a threshold, at which the nature and extent of feedbacks in the system change, leading to an abrupt transition. The availability of data of sufficient duration to see these effects has severely limited the identification of regime shifts in the past. However, both theoretical and empirical observations suggest that increased variance of one of several of the variables characterising ecosystems serves as an early warning indicator for an impending threshold (Brock and Carpenter, 2006). And although there are suggestions that regime shifts can be triggered by immediate human action, most regime shifts reported so far appear to be linked to underlying climatic changes.

Research into the non-linear dynamics of ecosystems has implications for management frameworks (Duit and Galaz, 2008), which still tend to manage ecosystems as steady-state or smoothly changing systems and assume that such changes are reversible. Such policies lead to unsustainable futures. Both natural and anthropogenic regime shifts are important to delineate: indicators are needed in order to design governance structures that permit adaptive management and an ecosystem approach.

Temporal scales related to lag times between cause and effect (in DPSIR terminology between pressures, state changes and impacts) can play a critical role in dynamics and abrupt transitions in natural processes, but the same can be said for the institutional response, i.e., the time delay between problem recognition, policy development and political commitment/support to implement a new policy. This can be illustrated by the Multiple Streams framework developed in political sciences (see Zahariadis 2007 on the framework or Meijerink 2005 for an example).

Here, the assumption is that a “problem stream” (the shifting baseline) develops over time. Some scientists might initially recognize it, and subsequently recognition could grow (building a so-called “epistemic community”, see Haas 1989 on epistemic communities or Meijerink 2005 for an example). In response to the problem stream and partly in parallel to it, a policy stream develops, e.g. civil servants or NGOs developing proposals for a new policy, but unable to implement it for lack of political support, funding and resistance to change and its implications, a situation termed “ecosocial anomie” (Limburg and Waldman, 2009) develops. The situation lasts until either the new ideas become mainstream or – typically due to a crisis/catastrophe – a Window of Opportunity (WO) opens. Examples can be found in the development of Dutch coastal defense strategies (Meijerink and Dicke, 2008, Meijerink 2005). In the Netherlands, the need for strengthening coastal defense structures had been recognized, including development of implementation plans, for two decades before a 1953 flood triggered the opening of a WO for implementation.

Such processes, in which a problem stream, a policy stream and a political stream must converge before policy changes can take place, are fairly common. Other examples include the fight against eutrophication of the North Sea, the ban of open access clam fishing and the initialization of a clam aquaculture system in the lagoon of Venice, and the establishment of National Parks in the German Wadden Sea.

3) Are footprints more important than boundaries?

There is an increasing awareness in the ICZM community of the importance of catchment/coast coupling: the recognition that in many coastal areas, coastal management cannot be separated from watershed management. (In the context of estuaries and related coastal systems, Wolanski (2007) refers to the processes that bear on these questions as “estuarine ecohydrology.”). The catchment boundary is a well-defined, natural boundary for understanding hydrological and nutrient linkages between catchment and coast, and even to island or reef systems proximal to the coast (Richmond et al., 2007).

However, infrastructural changes within coastal watersheds can blur or otherwise modify watershed boundaries. Demands for water for agriculture or drinking water can effectively capture parts of neighboring catchments via impoundments and pipelines (as in the New York City reservoir systems) or more dramatically by effectively rerouting major river systems (e.g., in China). Increases in impoundment volumes of dams typically results in increased evaporation, effectively reducing the net precipitation to the watershed and subsequent river flow. Increased storage and “aging” of riverine water affects sediment and nutrient loads as well (Vörösmarty et al, 1997); dams are typically associated with decreased sediment load downstream of the dam due to sediment retention, and the reduced flow downstream also has a lower capacity for sediment transport, which can result in both increased siltation and starved deltas. Impoundments also are often responsible for nutrient trapping, notably silicon, and can severely alter downstream nutrient ratios and associated plankton species distributions (Humborg et al., 2006).

In contrast to the catchment boundaries of coastal systems, the last decade has seen the ubiquitous “carbon footprint” emerging as a touchstone of the environmental movement to alert people to their impact on atmospheric CO_2 levels and earth’s climate (Wiedmann and

Minx, 2008). The ecological footprint (Wackernagel and Rees, 1996) has been used more broadly to emphasize the dependence of individuals on ecosystem services globally. The “foodshed” (Wedden, 1929) or “foodprint” (Billen et al., 2009) takes into account the area required to produce agricultural goods required to meet the food requirements of a population, and is thus related directly to the nutrient demands of cities and other concentrations of human population (Fig. 6). A benefit of the footprint concept is its emphasis on bringing awareness of the relationships between local, individual material consumption and distant, diffusely distributed production. A peculiarity of the ecological footprint concept is that it doesn’t translate directly into a specific geographic location, but rather represents a hypothetical area of land required to sustain an ecosystem, produce food, or absorb CO_2. In contrast, the “foodprint” attempts to associate a “real” region with the food supply of an urban population (Billen et al., 2009). Historically, the “foodprint” of most cities was largely based on local or regional food production. For some cities (e.g. Paris; Billen, 2009) this may still be true; for others (e.g. New York), food supply is inseparable from the global transportation system.

<figure 6 near here>

From the standpoint of nutrient loading, watershed nutrient accounting methods have been used to infer the net cross-boundary nutrient inputs to watersheds from the most significant sources (net food/feed transport, fertilizer, and in the case of nitrogen, atmospheric deposition and biological N-fixation) that have been related to nutrient export from watersheds to coastal waters (Howarth et al., 1996; Boyer et al., 2002; Russell et al., 2008; Fig. 7). These methods, cousins of the various footprint methodologies in that they relate local impacts (both within-watershed and coastal nutrient loads) to transboundary fluxes, reveal the increasing importance of these fluxes in this age of globalization (aptly termed “the Homogecene” in the context of biotic homogenization and reduced biodiversity; Rosenzweig, 2001; Rooney, 2007). Nutrient inputs to coastal watersheds are seen to be increasingly related to changes in food preferences and other market trends (more meat/less grain, corn for processed food and biofuels, etc), long-distance transport of nutrients in food and feed across catchment boundaries, and associated increases in atmospheric N deposition related to vehicular emission sources.

Some material flows to coastal waters (water, sediment, silica) seem to be largely constrained by catchment boundaries and human activities therein. Others (nitrogen, phosphorus) may be more tied to trade and other transboundary effects. In addition, the structure and responses of coastal ecosystems themselves may be changing in response to species flows and resulting changes in biodiversity associated with the onset of the Homogecene (Lotze et al., 2006; Levrel, 2007); one example is the transformative effect of the zebra mussel in North American waters and their biogeochemistry. How can we adapt our thinking and management strategies for coastal waters to these realities?

<figure 7 near here>

4) What makes a decision big? The tyranny of small decisions in coastal regions

“Scale” applies to other dimensions beyond time and space. Human decision making is also characterized by a range of scales, from deciding what to have for lunch, to deciding whether or not to go to war. However, most human decisions are small, made by individuals, and are often assumed to have few obvious consequences beyond the decision at hand. Kahn (1966) examined the consequences to the market of cumulative effects of apparently independent individual decisions and concluded that they can yield undesirable consequences. William Odum (1982) discussed the phenomenon in the context of the environment, noting the example that no one decided to eliminate 50% of coastal marshlands of Connecticut and Massachusetts between 1950 and 1970, though this is what happened. Ehrlich and Kennedy (2005) noted that the ethics of decision making is unclear, that there is a need for a focus on how people make decisions affecting the environment, and called for a "Millenium assessment of human behavior." A recent, provocative piece in the New York Times (Gertner, 2009; Marx, et al., 2007), suggesting that the human brain may be "wired" to give environmental issues like climate change relatively low priority, only increases the urgency of such work. Regulations and governance presumably should protect environments from gross negligence and irresponsible decisions, but is it possible to safeguard vulnerable regions from these incremental effects of small decisions?

Coastal regions seem disproportionately vulnerable to the cumulative effects of unintended "bad" choices because of their relatively high population densities (45% of the world's population lives in the 10% of land area defined by LOICZ as the "coastal zone"; Crossland et al., 2005). Probably more to the point, the rate of migration to coastal regions is higher than to other regions, the result of millions of individual decisions with potentially disastrous consequences. In addition, choices that were relatively innocuous even a few years ago are today more risky due to combined effects of increased vulnerability (i.e., a shifted baseline) and an amplifier effect on consequences of choices due to the nature of modern trade. Historically, most of the small decisions of daily life had little impact beyond local households. Prior to modern food distribution systems, both were procured from local lands or waters. As nutrient accounting can show, the distribution of food and feed can play a dominant role in the level of riverine nutrient fluxes to coastal waters. Modern transport methods assure the arrival of foods across continental scales in a matter of hours or days, thereby guaranteeing a market for fisheries in (previously) remote regions. Fish or fowl for dinner? The fish may have been swimming in coastal waters on the other side of the world a few days before (Berkes et al., 2006). The fowl may be raised in a factory farm, fed on fishmeal from thousands of kilometers away, but the farm discharges waste to local coastal waters.

The world's coastal ecosystems see the direct effects of human activity for the simple reason that more humans live near coastlines, or along major rivers, than in continental interiors far from waters, and are thus especially sensitive to them. This is not an accident of history, but the collective result of individual decisions over time, based on proximity to jobs and other economic opportunities, quality of life, and many other factors all ultimately related to the proximity to coastal waters and their services to humans. Whether the waters support transport and trade, fishing/aquaculture, recreation, serve as a disposal system for sewage and industrial waste, or all of the above, they provide value (i.e., ecosystem services) to the earth's coastal lands, and thus concentrate human populations (UNEP, 2006).

However, the decision to live in coastal areas can ultimately have a dark side, as recent experience with natural disasters in Banda Aceh, New Orleans, Haiti, and Japan has shown. Highly populated coastal regions are especially vulnerable to extreme events of geological or meteorological origin. When the catastrophic, magnitude 8.7 Lisbon earthquake of 1755 rang church bells all over Europe, they tolled for the tens of thousands of lives lost in the quake, the accompanying tsunami and fires, which leveled most of Portugal. The January, 2010 magnitude 7.0 earthquake that struck Port au Prince, Haiti, killed hundreds of thousands and caused devastating damage to infrastructure and personal property in the billions. Both of these coastal cities were local centers of population and commerce. While Port au Prince saw no post-quake tsunami, the relative increase in devastation caused by the less powerful Haitian quake (compared to the Portuguese quake) can be seen as a direct result of the relative increase in population density and development in this coastal city. The more recent magnitude 9.0 earthquake and tsunami in northern Japan, and associated nuclear plant catastrophe, shows that even the world's most technologically sophisticated countries are vulnerable. Tsunamis and hurricanes can wreak catastrophic destruction in the modern day, densely populated coastal zone simply by virtue of the number of people deciding to live there; risk to life and property in coastal regions increases in proportion to population density and development even in regions of quality infrastructural engineering and design.

Increasingly, coastal zone governance turns to engaging "stakeholders" at the local scale as well as scientific expertise (e.g. Tompkins et al., 2008) in the hope of equitable solutions and grass roots support. Stakeholder participation has helped to mitigate or solve many conflicts in coastal areas, particularly at local scales, but Billé (2006), in his discussion of the "illusions" of coastal zone management, notes that one illusion is the notion that consensus in human groups is prevalent at small scales. Stojanovic and Barker (2008) discuss several problems of coastal governance through "coastal partnerships" of institutional stakeholders. If consensus is an illusion even in small groups which are deciding questions of governance for small systems, what chance is there for larger systems?

In fact, governance of small coastal systems may be especially difficult because of the numbers of institutions involved. Murawski (2007) refers to this as the "paradox of scale": the smaller the geographic scale used to define an ecosystem, the more entities are involved in decision making (Fig. 8).

<figure 8 near here>

In addition to scale issues, spatial disconnections, e.g. between decisions upstream and distant environmental consequences downstream, often complicate governance, much as lags between actions and consequences do in the time dimension. Major policy decisions, such as undertaking the massive redistribution of river waters to serve the needs of irrigation and drinking water as is being done with the Changjiang River in China (Xue et al., 2007), can have fairly direct environmental implications downstream. Reducing river flows in coastal areas changes the salinity regime, moving the salt front upstream and potentially increasing the risk of groundwater intrusion of salinity and contamination of drinking water supplies (Xue et al., 2007). Reduction of river flow means a concomitant reduction of sediment transport from upland soils, potentially starving river deltas of sediment. The episodic nature of sediment transport (Xu et al., 2005) and its redistribution in the river network is an especially problematic aspect of sediment management and a source of uncertainty. Woodroffe et al. (2006) point out that while sediment processes are well understood at small time scales because of reliable short-term fluid dynamics, and long time scales (> 1000 years) due to the availability of stratigraphic analyses, uncertainties are highest at the intermediate time scales most relevant to managing sediment. Indirect effects can be subtler: introduction of invasive species, shifts in nutrient ratios and corresponding ecosystem changes can easily result from changes in flow regime (Li et al., 2007). All of these flow-related catchment phenomena illustrate the obscuring effect of the spatial dimension on relationships between decisions (large or small) made upstream and consequences (often large) in downstream coastal waters.

5) Scales of complexity in coastal waters: the simple, the complicated or the complex?

As pointed out by Dennison (this volume) Glouberman and Zimmerman (2002) have differentiated between simple, complicated and complex classes of problems and management in the context of the Canadian medical establishment. This distinction between categories of problems also applies to the tools available to deal with them. Simple problems can be characterized by simple models (recipes); complicated problems by more complicated models. Truly complex problems or phenomena often transcend the capability of current models, or decision processes.

Ecological modelers have recognized for years that models can exhibit nonlinear and often counterintuitive behavior, and so must be created and used judiciously (e.g., Silvert, 2000; Sterman, 2002). Solving the “Goldilocks problem” of matching the appropriate level of model detail to the question at hand is a Holy Grail of modeling; on the one hand, simple models are desirably parsimonious, even though many can exhibit complex behavior (Swaney et al., 2008). On the other hand, models that are too simple (e.g., not “biogeochemical” in the sense of Tett and Wilson, 2000) run the risk of being unrealistic and, if used as a guide for management, conceivably dangerous. Calls for new integrated modeling tools (e.g., Kroeze, et al., 2008) are appropriate and necessary, but probably not sufficient.

The appropriate level of model complexity depends on the problem to be addressed, and the data available to drive it. While in many cases complex models are needed for use in predicting environmental status of coastal waters (e.g., oxygen levels, trophic status, etc; Kroeze et al., 2008; Solidoro et al. 2005), simpler models may be adequate to serve as functional descriptions of ecosystem services (Barbier, 2008; Russo and Kareiva, 2009) or first-order responses to environmental change (Rahmstorf, 2007). Intermediate complexity tools may be sufficient to serve as screening models, and statistical relationships may be adequate to detect changes in flows (Kunzewicz et al 2009), or to establish relationships between specific human activities and coastal impacts (Howarth et al., 1996). However, examples of coastal waters in which “complicated” models fail to address the scope of the problem are increasingly common; the Baltic Sea is one example of a complex system subject to changes in terrestrial nutrient loads (Humborg et al., 2007), abrupt transitions (i.e. apparent regime shifts) in its biogeochemical cycles and oxygen conditions (Vahtera et al., 2007; Savchuk et al., 2008) and in food web structure (Osterblom et al., 2006). Its governance is further complicated by international boundaries and multiple, hierarchical management bodies with a conventional steady-state perspective of the relationship between loading and ecosystem response (Österblom et al. 2010). New theories of governance that incorporate ideas of complex adaptive systems may be useful in such cases (Duit and Galaz, 2008).

Datasets used for coastal classification are also scale-dependent. Coastal typologies intended to characterize regional to global-scale variation of ecosystems and their drivers in coastal waters have typically relied on global datasets of half-degree or greater resolution, and have discriminated coastal regions based on static properties of coastal or associated terrestrial or oceanic grid cells. This “Aristotelian” approach to coastal typology has limitations (see Appendix 2, supplementary material). It seems increasingly likely that attempting to categorize the controls on eutrophication at local to regional scales will require more highly resolved data in time and space. Duarte et al. (2008) have shown several examples of nonlinear response of eutrophication to nitrogen loading in coastal waters, including hysteresis and apparent shifts in “baseline” response, suggesting that a return to previous trophic status (i.e., “restoration”) may not be possible. However, these phenomena would probably be overlooked if the analyses were performed at a larger scale. Coastal typologies that aim to resolve controls on ecosystem responses have different data requirements than those resolving gross regional patterns of climate or land cover. Clearly, there is no sense in trying to characterize complex responses of coastal waters without sufficiently highly resolved datasets to both drive and validate the models. For example, Yamamuro et al. (2006) have described shifts from submerged aquatic vegetation (SAV) to phytoplankton dominance in Japanese coastal lagoons, in one case resulting in a catastrophic loss of the fishery and in another, an increase of commercial mollusk production. Yamamuro and Icely (this volume) describe conflicts in governance at different scales which result in similar problems. What is the appropriate level of model complexity to characterize such behavior? This represents a modeling as well as a data management challenge.

Finally, the revolution in technology that has brought us GIS, satellite-based remote sensing, and the capability to follow tracers and biomarkers through ecosystems has also streamlined trade and transportation in the business world, making it simpler to move and track materials globally. Now that we have entered the Homogecene, it is clear that the same revolution is responsible for the homogenization of culture, the creation of material flows between distant ecosystems, and the associated degradation of the environment. Modern tracking systems, such as those used in business to follow shipments door-to-door, remain unused, or at least underutilized, in research, possibly due to privacy concerns and other legal issues. Do these represent a new class of tools appropriate for addressing the unique problems of environmental impacts of globalization? Cutting edge technologies, including so called “smart dust” (http://en.wikipedia.org/wiki/Smartdust) and other networks of environmental sensors integrated with analytical software (e.g. Harmon et al., 2009; Galaz et al 2010) have the potential for monitoring spatiotemporal change in coastal ecosystems. Beyond research, can they inform coastal decision making and governance? And, if so, at what scale?

Preliminary answers from LOICZ experience in coastal science

The questions posed above don’t have obvious single answers – if they did, they would not be very fundamental questions. In addition, we are equipped with incomplete understanding based on limited knowledge, which changes as time goes on, corresponding to the ongoing dynamic of change in the coastal zone. This is true of most questions related to the science and management of the coastal zone. However, we can propose a few partial answers:

1) How big are coastal ecosystems and why should we care? While there is no single “size” or scale to coastal ecosystems, we know that they are influenced by activities well-beyond their immediate boundaries. Human activities in their watersheds have degraded coastal ecological communities, increased areas of eutrophication and dead zones, and reduced abundance of fish and shellfish (Bricker et al., 2007). Critical scaling parameters, such as mean residence time and other measures, can provide bases for multidimensional classification of coastal water bodies, inform us as to the sensitivity of a given water body to various loadings, and thus suggest appropriate management action (Bricker et al. 2003; Buddemeier et al 2008; Swaney et al. 2008).

2) *Temporal scales of change in coastal waters and watersheds: Can we detect shifting baselines due to economic development and other drivers?* This is a tough one. Human memory is tied to its own relatively short time scales, ranging from the duration of the work day, to the length of a political cycle, to the time between human generations. Time delays in societal and political response to (recognized) regime shifts add to those in the cause-effect relations in ecosystem change. Together, these delays pose a particular challenge for the dynamics of societal response which demand new forms of management which account for

unpredictability of system behavior as well as ambiguity within the society in framing the responses. Without concerted effort to document change in the environment, gradual degradation escapes our notice, as can the gradual, cumulative impacts which can lead to relatively swift transitions in ecological communities (references). This is why monitoring our coastal waters and their watersheds is of paramount importance. Collecting, curating, and synthesizing monitoring data, and analyzing it with the new insights and novel techniques that science continually provides us, is essential to preserving the quality of coastal environments. Establishing common, accessible datasets for general use (such as those developed by the FAO), and developing accessible tools to analyze data is a start. We must share our results with our peers and students, but also with the coastal governance community, as clearly, straightforwardly and with as little bias as possible. As data-collecting technologies proliferate, we must continue to develop new ways of assimilating, interpreting and visualizing the data, including illustrative models and material budget methods, GIS, coastal typologies, and other methods of synthesizing information, so that it is informative for stakeholders, educators and decision makers. In addition, different modes of communication and dialogue between scientists, policy makers and the public need to form part of the evaluation of shifting baselines in order to include societal valuation and perception of the detected changes.

3) *Are footprints more important than boundaries?* Because of modern technologies, we have made the world a smaller place. Prior to the industrial revolution, the long-distance transport of materials associated with trade and other human activities was a negligible component of coastal material cycling. Today, “footprints” matter as much as the boundaries on a map which define coastal water bodies and their watersheds because they represent the dominant role of human activities in biogeochemical cycling, and we will have to recognize the spatial and temporal scales at which decisions are made that influence footprints. Global decisions (e.g. at the WTO) can create local impacts on footprints at many locations. Therefore, both boundaries and footprints are important for planning and management. More efforts must be made to develop global datasets at finer resolution, e.g. better than country-wide totals or averages, or half-degree grid cells), and developed for natural boundaries (watersheds, water bodies) rather than political or administrative ones. This is especially crucial for understanding watershed interactions with coastal waters. For the practical management of nutrient loads to coastal waters, relatively simple nutrient accounting methods, such as the Net Anthropogenic Nitrogen Input (NANI) methodology (Hong et al. 2011; Boyer et al. 2002) can estimate the magnitudes of net inputs to coastal watersheds and show the relative importance of local versus distant sources. On the governance side, careful analysis of decision making “actors” and the scales at which they operate requires integrated assessments and a range of methodologies and tools from both natural and social sciences. A global example of an integrated assessment is the Millennium Ecosystem Assessment (www.millenniumassessment.org); a regional example is the Coastal Futures project, which examined impacts of offshore wind farms in the German North Sea, including perceptions of the local population, effects on regional economic development and impacts on ecosystem services (Lange et al. 2010). Such assessments rely on GIS-based tools to assess physical characteristics as well as actor-oriented analytical frameworks used in political science (e.g. Scharpf 2000).

4) *What makes a decision big?* It is fairly clear that the joint impact of small, individual decisions can have as much or more impact on coastal waters than “large” executive decisions made by those charged with coastal governance, though the difficulty of assigning individual responsibility to the many decisions that can collectively amplify into environmental catastrophe is the root of what Hardin called the “tragedy of the commons”. Attempts by governing bodies to provide structure and accountability to decision making over the range of

issues affecting coastal regions are fraught with pitfalls and limitations, as discussed above, though there have been some partial successes in grappling with multiple scales of coastal governance. An example is the Resource Management Act (RMA) in New Zealand. The RMA has been deemed, to some extent, a success both nationally and internationally (http://en.wikipedia.org/wiki/Resource_Management_Act_1991). The Act established one single framework in place of separate frameworks divided between different agencies and sectors (e.g., land use, forestry, pollution, water, etc), and was the first national scale statutory planning strategy to adopt sustainability as a core principle, and "sustainable management" as the basis of regulations. The RMA is based on the concept that decisions made locally about a proposed activity are to be made on basis of effects, putting forward the expectation that science-based knowledge be applied to management issues. Unfortunately, the RMA has had failures, including in the area of water quality, which has continued to degrade in New Zealand during the recent period of land development. This appears to be rooted in the inability of regional bodies to prepare and implement standards consistent with the aims of the Act in the face of development. For other environmental issues (e.g., aquaculture) the Act has been more successful. Overall, the Act has provided a unified framework for local bodies to work toward, and some progress has been evident. Such attempts at coordinated decision making over local, regional and global levels (Olsen et al, this volume) represent the best hope for sustainable governance of the coastal zone.

5) Scales of complexity in coastal waters: the simple, the complicated or the complex?

Scale matters in coastal waters, for a variety of reasons, as we have seen. We also need to communicate this to those who manage the waters and mediate between stakeholders, and to help provide them with methods to manage appropriately to the scales of the problem at hand on a case by case basis. We may be aided in management by new sensor technologies and monitoring networks for coastal environments, but we also must learn to live with uncertainty in decision making and develop governance systems that are flexible and quickly adaptable in the light of new insights or observations of unintended changes in systems (e.g. through monitoring). We need to recognize that coastal governance has scale issues of its own, and that constraints imposed by institutions at various scales are part of the problem. An awareness of the issues of scale is a first step towards avoiding simplistic "off the shelf" solutions to complex problems. Transdisciplinary approaches incorporating natural and social sciences within integrated problem assessments may improve understanding of problems (including interaction between scales), that might be help decision makers frame solutions to complex multiscale problems.

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List of figures, with captions

Figure 1: Relationships between time and space scales for a) physical factors, b) biological factors, c) governance and d) drivers of ecosystem change in coastal waters and watersheds.

Figure 2: Apparent net ecosystem production for LOICZ budget sites as a function of system area (log-log plots). a) Net heterotrophic systems (p-r <0; absolute value used). b) Net autotrophic systems (p-r > 0). (Smith et al. 2005).

Figure 3: (a) The Chesapeake watershed at different historical periods: pre-Colonial up to 1650, showing a forested landscape with a large beaver population; early Colonial, when ~40% of the land was deforested for agriculture; intensive agriculture, when up to 80% of the land was under cultivation; and urbanization, when about half of the watershed is in forest and urban and suburban centers characterize some parts of the watershed. (b) The increase in the human population. (c) The historical record of land use. (d) Sales of fertilizer in the Chesapeake watershed. (e) Historical record of nitrogen fluxes into the Potomac River. (f) The historical record of the crab and oyster harvest. (g) Pollen representation of land use in the upper estuary. (h) Paleoecological record of sedimentation rates in the upper estuary. (i) Pollen profile showing a gradual increase in dry taxa after colonization. (j) Nitrogen influx into the Chesapeake Bay. (k) The change in the ratio of planktonic to benthic diatom species in the central mesohaline section of the estuary (Brush, 2008)

Figure 4. Historical and future estimates of sea level change, relative to the present, from the 2007 IPCC report (Bindoff et al. 2007)

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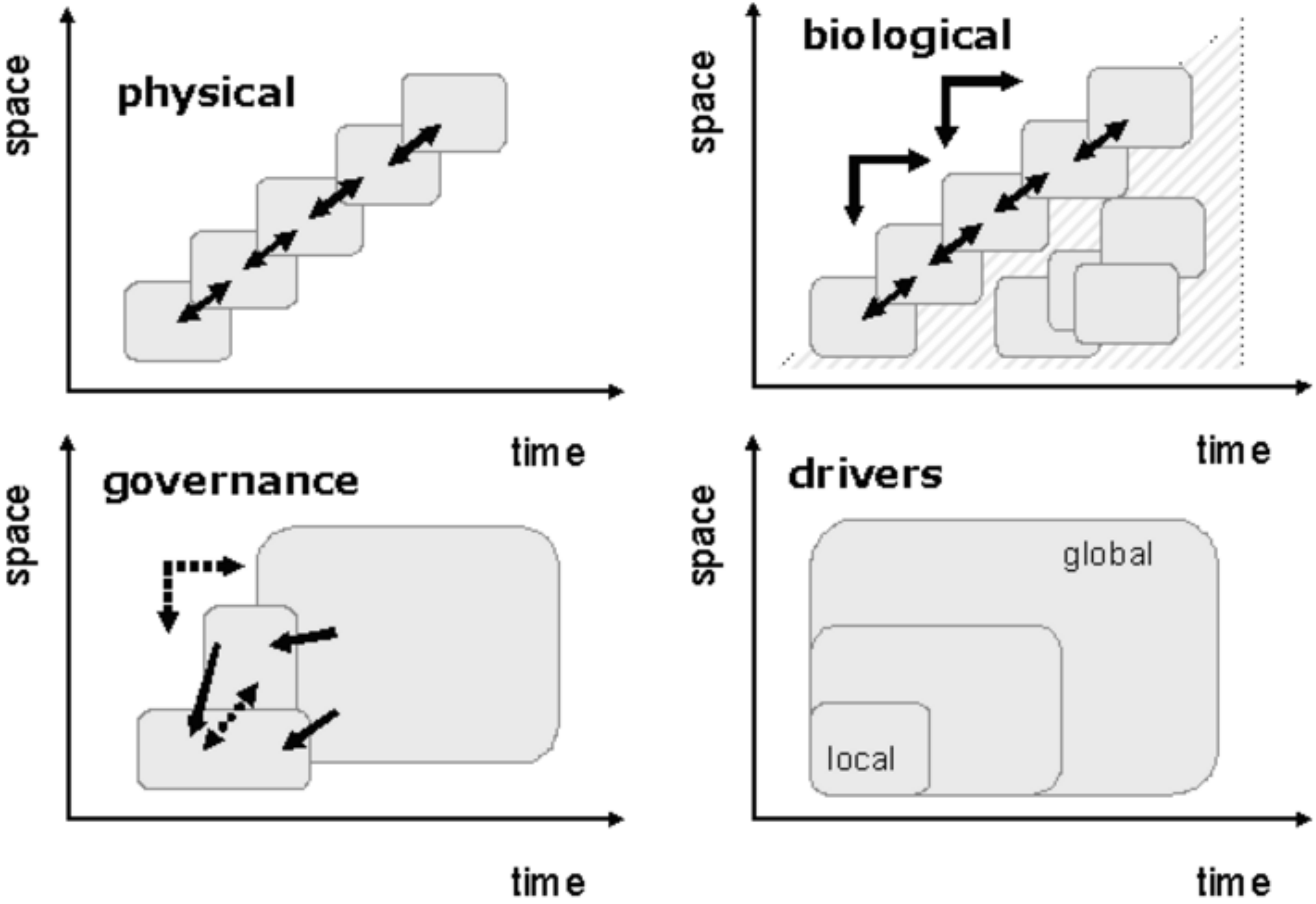
Figure 6: Terrestrial net primary production of agricultural and forested land (Autotrophy) (a), and total heterotrophic consumption by human and domestic animals and by forest communities (Heterotrophy) (b) in the Seine watershed. Distribution of the autotrophy: heterotrophy ratio (c): Yellow areas are in approximate equilibrium; green areas are autotrophic, orange or red areas are heterotrophic (Billen et al 2007). The Paris metropolitan area is highlighted in red, indicating a high net consumption of nutrients, with implications for the downstream waters of the Seine.

Figure 7: Net anthropogenic nitrogen input (kg-N/km^2/yr) in a) northeastern US watersheds and b) Hudson River watershed subbasins. The larger scale patterns on driven by latitudinal gradients of population, agriculture, and atmospheric N deposition. Within the Hudson catchment, the intense levels of anthropogenic nitrogen input on the coast correspond to the large food imports of the New York City metropolitan area and local elevated N deposition associated with high automobile densities. (Hong and Swaney 2009)

Figure 8: The range of scale of institutional involvement at for coastal systems of various size scales. The “paradox of scale” is that the smaller the geographic size of the coastal ecosystem, the more entities that “contribute” to its governance (i.e. the largest regions are governed only by international bodies; the smallest by local to international entities) (Murawski 2007).

Figure(s) 1


physical
space
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Figure(s) 2

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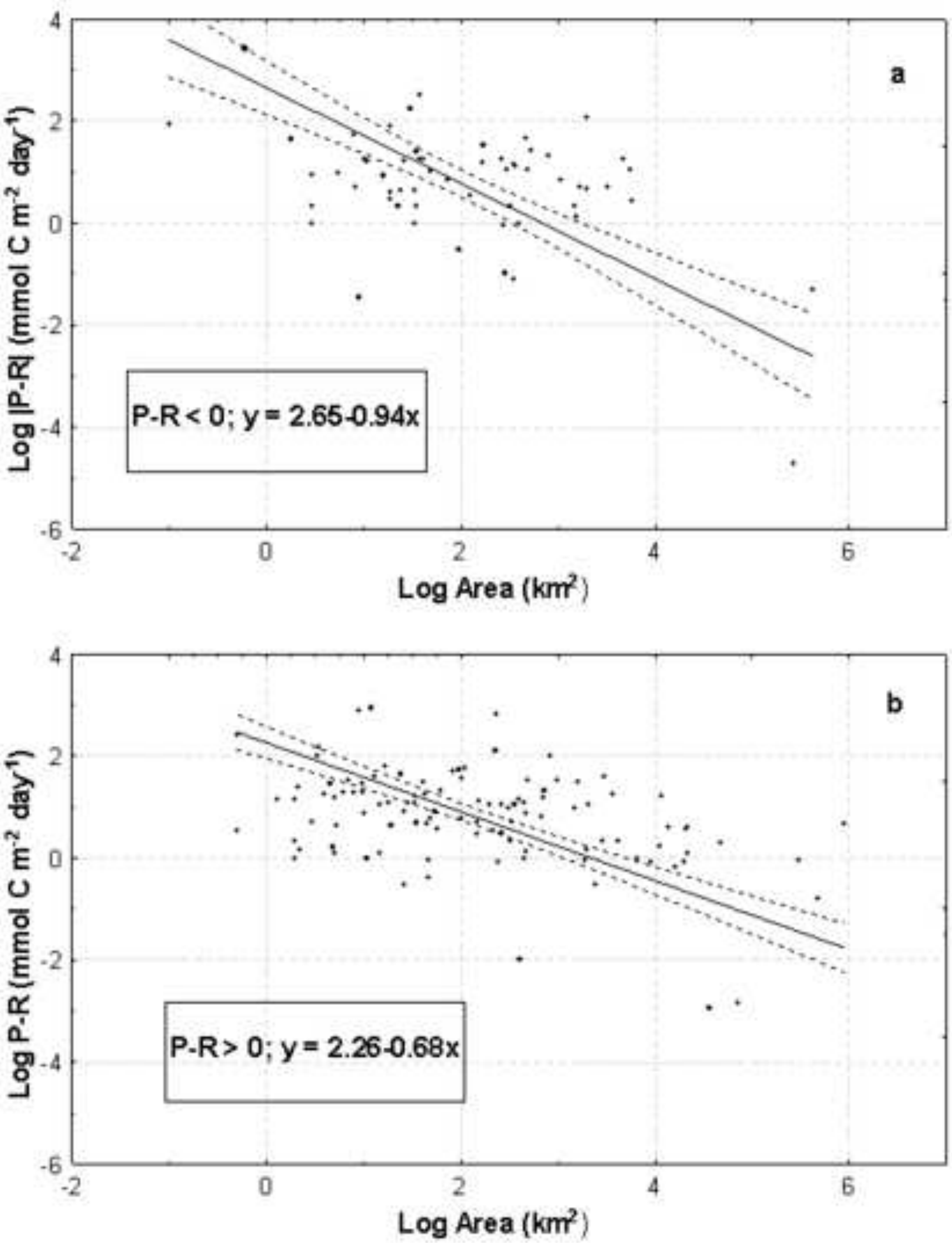
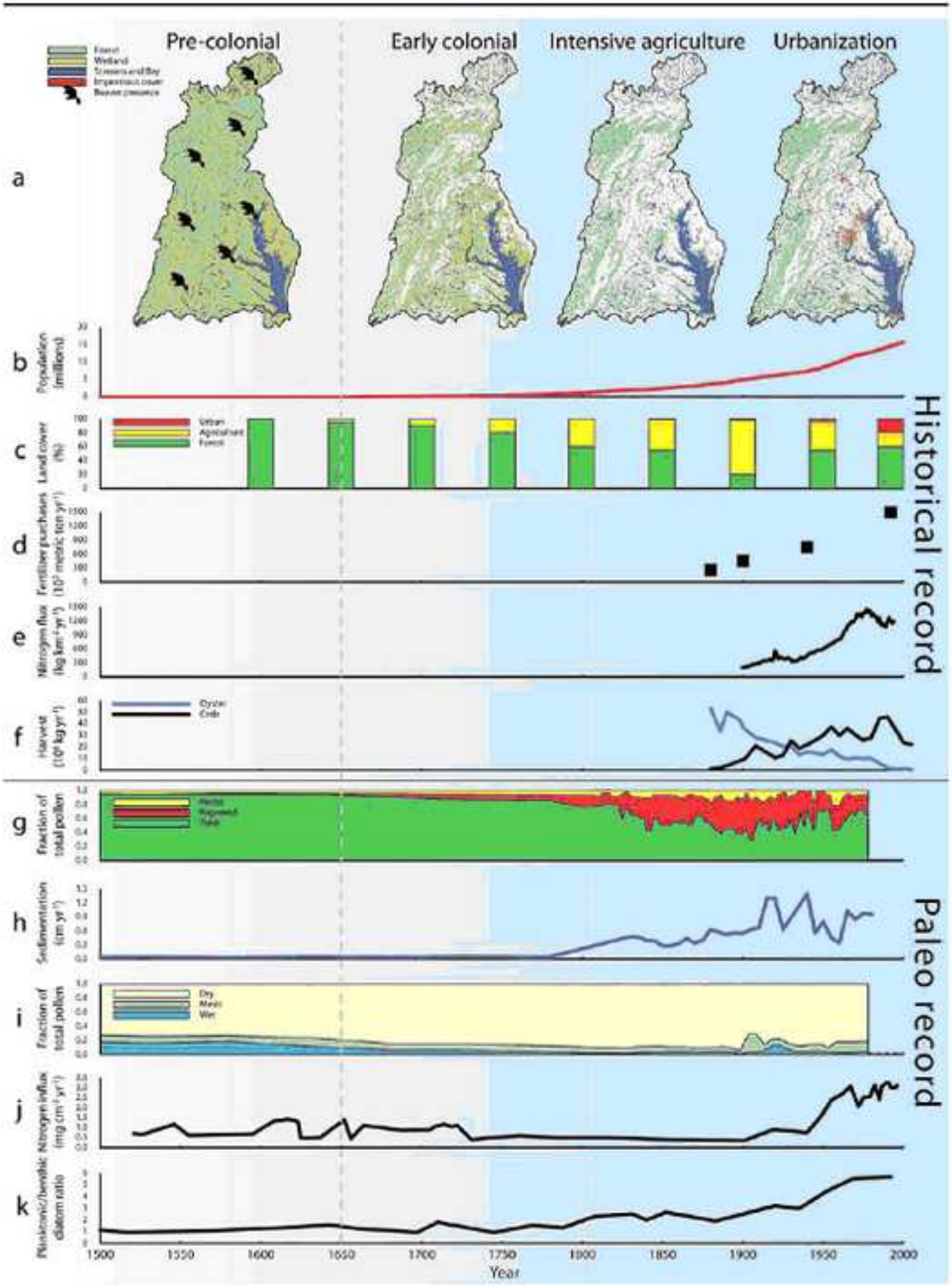
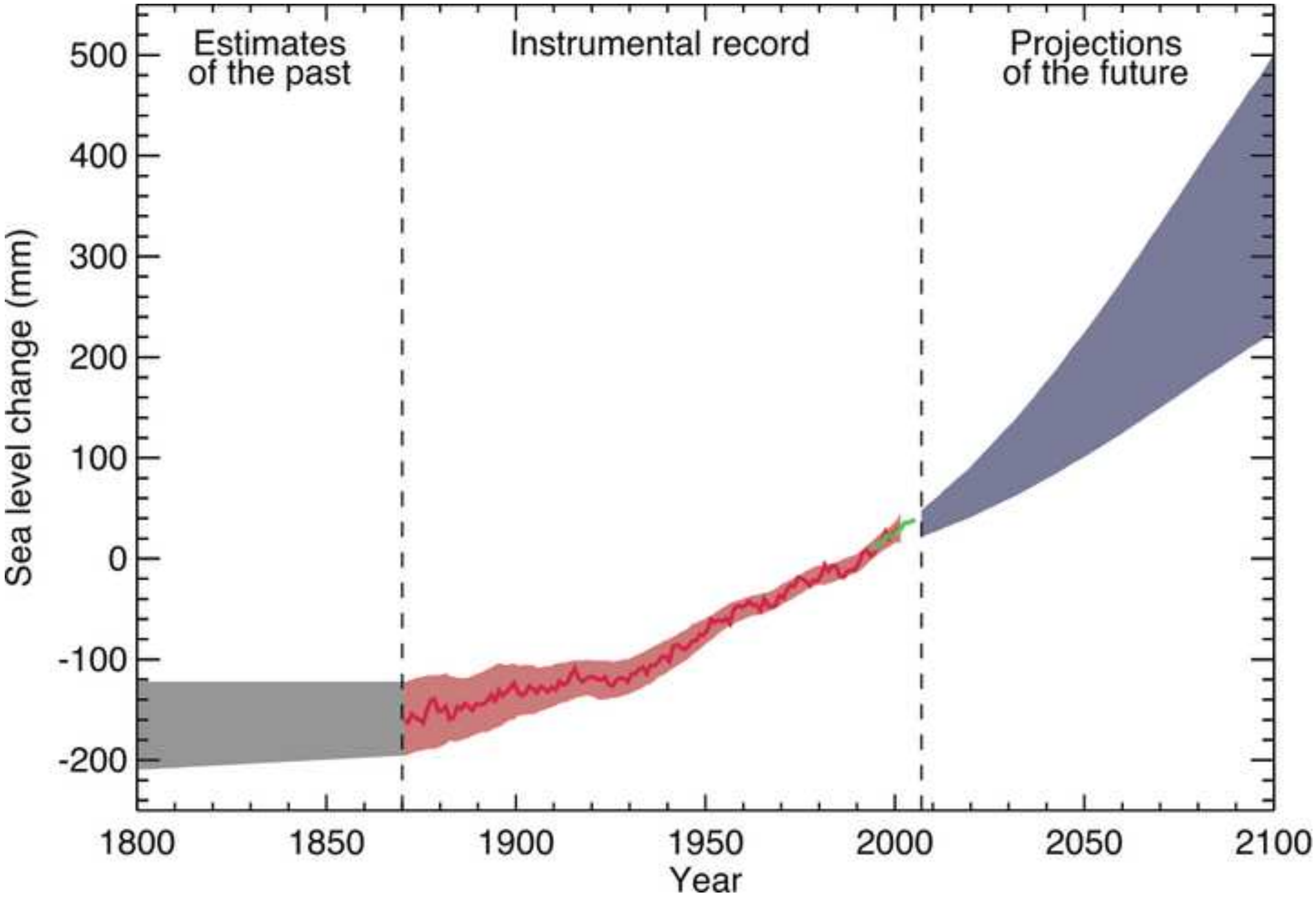


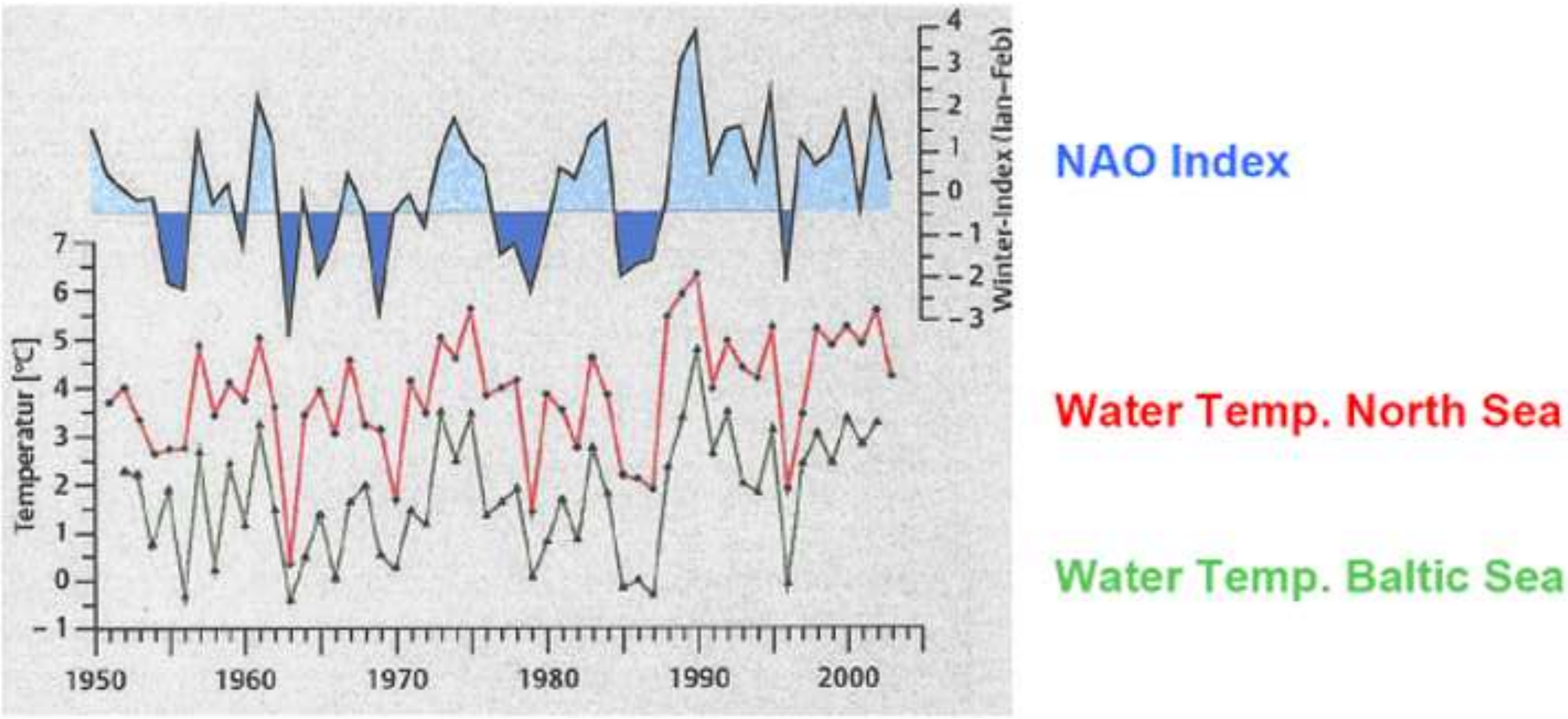
Figure(s) 3

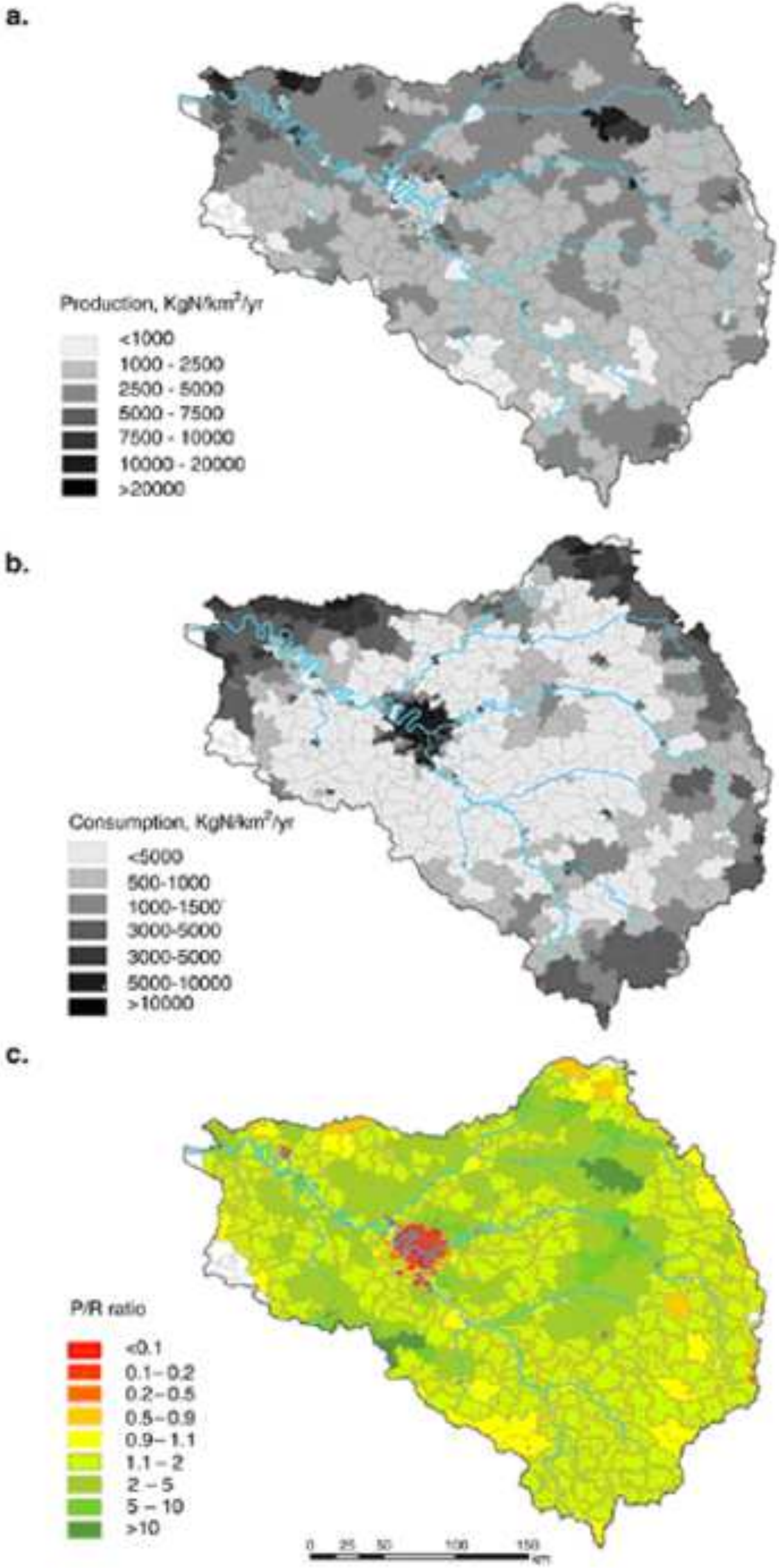

Pre-colonial
Early colonial
Intensive agriculture
Urbanization
Forest
Wetland
Streams and Bay
Impervious cover
Beaver presence
a
b
Population (millions)
c
Land cover (%)
Urban
Agricultural
Forest
d
Fertilizer purchases (10^3 metric ton yr^{-1})
e
Nitrogen flux (kg km^{-2} yr^{-1})
f
Harvest (10^6 kg yr^{-1})
Oyster
Crab
Historical record
g
Fraction of total pollen
Herbs
Ragweed
Tree
h
Sedimentation (cm yr^{-1})
i
Fraction of total pollen
Dry
Mesic
Wet
j
Nitrogen influx (mg cm^{-2} yr^{-1})
k
Planktonic/benthic diatom ratio
Paleo record
1500
1550
1600
1650
1700
1750
1800
1850
1900
1950
2000
Year

Figure(s) 4

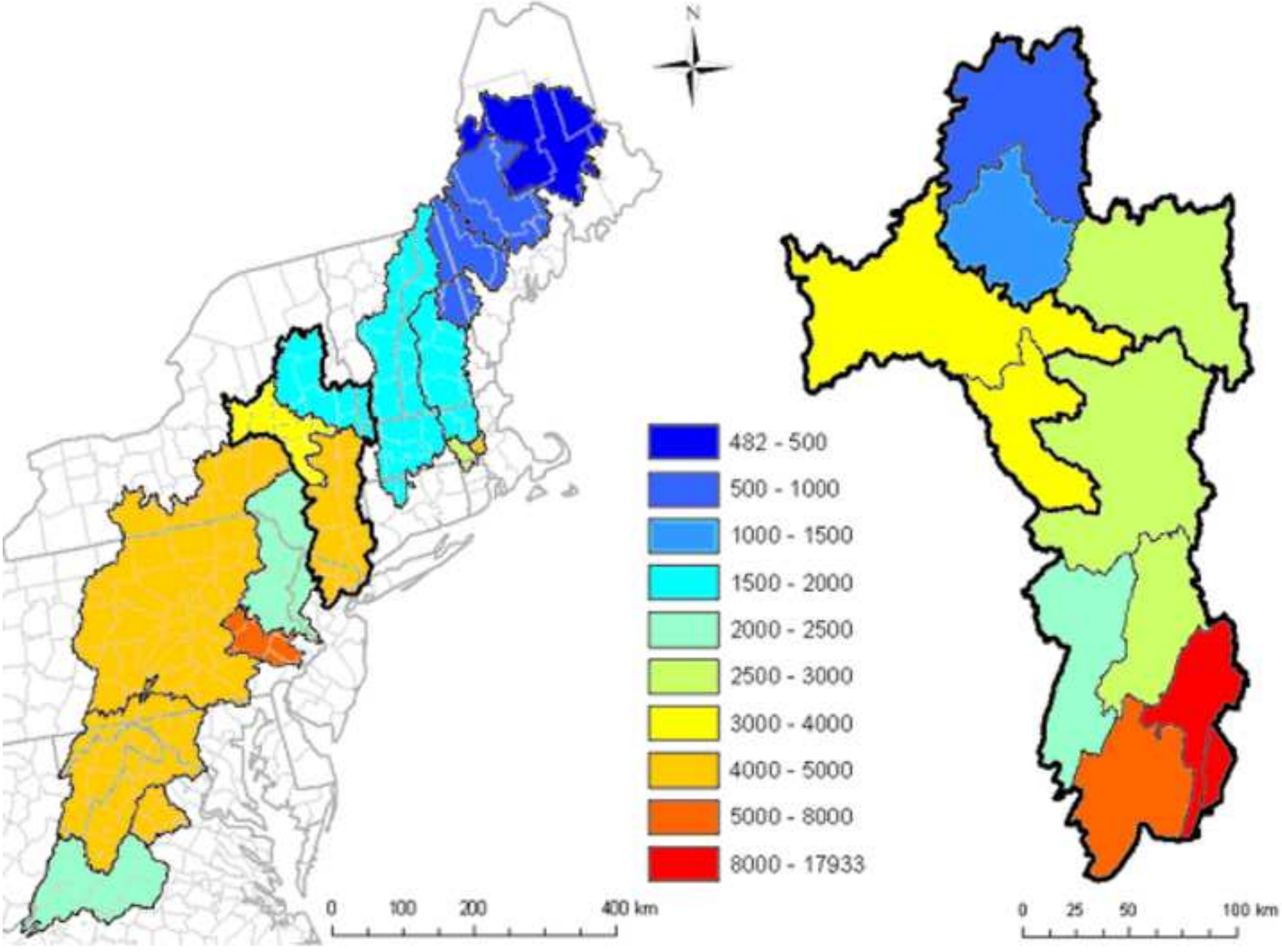

Estimates of the past
Instrumental record
Projections of the future
Sea level change (mm)
500
400
300
200
100
0
-100
-200
1800
1850
1900
1950
2000
2050
2100
Year

Figure(s) 5


NAO Index
Water Temp. North Sea
Water Temp. Baltic Sea
Temperatur [°C]
Winter-Index (Jan–Feb)
7
6
5
4
3
2
1
0
−1
4
3
2
1
0
−1
−2
−3
1950
1960
1970
1980
1990
2000


a.
Production, KgN/km²/yr
<1000
1000 - 2500
2500 - 5000
5000 - 7500
7500 - 10000
10000 - 20000
>20000
b.
Consumption, KgN/km²/yr
<5000
500-1000
1000-1500
3000-5000
3000-5000
5000-10000
>10000
c.
P/R ratio
<0.1
0.1– 0.2
0.2– 0.5
0.5– 0.9
0.9– 1.1
1.1 – 2
2 – 5
5 – 10
>10
0
25
50
100
150
km

Figure(s) 7


N
482 - 500
500 - 1000
1000 - 1500
1500 - 2000
2000 - 2500
2500 - 3000
3000 - 4000
4000 - 5000
5000 - 8000
8000 - 17933
0
100
200
400 km
0
25
50
100 km

Figure(s) 8

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