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Review

Regime shifts in the marine environment: The scientific basis and political context

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ABSTRACT

Regime shifts in the marine environment have recently received much attention. To date, however, few large-scale meta-analyses have been carried out due to insufficient data coverage and integration between sustained observational datasets because of diverse methodologies used in data collection, recording and archival. Here we review the available data on regime shifts globally, followed by a review of current and planned policies with relevance to regime shifts.

We then focus on the North and Baltic Seas, providing examples of existing efforts for data integration in the MarBEF Network of Excellence. Existing gaps in data coverage are identified, and the added value from meta-analyses of multiple datasets demonstrated using examples from the MarBEF integrated data project LargeNet. We discuss whether these efforts are addressing current policy needs and close with recommendations for future integrated data networks to increase our ability to understand, identify and predict recent and future regime shifts.

1. Introduction

Several definitions of regime shifts have been proposed in the literature. The term was originally restricted to the description of atmospheric processes on multidecadal timescales and their consequences for the physical environment (Rahmstorf, 1999; Rodionov, 2005a,b; Schumacher, 1999). But the term is now applied more broadly. The regime shift concept was first extended to investigations of freshwater (Scheffer et al., 2001a) and terrestrial systems (May, 1977; Scheffer et al., 2001b) and is now used in a similar manner for marine and estuarine systems (Petersen et al., 2008; Polovina, 2005) to describe rapid shifts between two alternate stable environmental states. More recently the definition was broadened even further and now also includes shifts in biological systems as a response to physical drivers (Collie et al., 2004; de Young et al., 2004, 2008; Solow and Beet, 2005). Therefore, changes in species abundance, community composition and trophic organization occurring at regional or greater spatial scales either in response to an external physical or anthropogenic driver or due to internal processes such as wasp-waist control or changes in energy transfer could all constitute a regime shift (Bakun, 2004; Baumann and LeBlond, 1996; Benson and Trites, 2002; Caddy and Garibaldi, 2000; Collie et al., 2004; Hutchings, 2000; Mantua, 2004; McKinnell et al., 2001; Wiltshire et al., 2008) if the restructuring of the ecosystem state and associated triggering of a series of concomitant physical and biological processes ensures that a new quasi-equilibrium state is established (Cury and Shannon, 2004; de Young et al., 2004). The time span during which the regime shift occurs is considered to be short in relation with the duration of the 'before and after' regimes and relative to the overall observation period (de Young et al., 2008). The two regimes may not necessarily be under the control of the same factor either, as different processes can act simultaneously or subsequently (Cury and Shannon, 2004; Möllmann et al., 2009).

Three distinct types of regime shift have been categorized on the basis of the relationship between the response variable (usually biotic) and the control variable (usually external, abiotic or anthropogenic) (Collie et al., 2004): (1) 'smooth', represented by a quasi-linear relationship between the response and control variable, (2) 'abrupt' or 'catastrophic', where a non-linear relationship between the response and control variable means that incremental changes in the forcing factor are amplified to abrupt changes in, for example, population abundance (Scheffer and Van Nes, 2004), or (3) 'discontinuous', whereby the trajectory of the response variable is different when the control variable increases to when it decreases. In the latter case the system is considered to have undergone hysteresis and alternative stable states exist, known as

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multiple basins of attraction (Collie et al., 2004; Hsieh et al., 2005; Petraitis and Dudgeon, 2004; Scheffer et al., 2001a). The ecosystem cannot 'recover' to its former state merely by a small reversal in parameter values and thus alternative stable states occur over the range of parameter values between the two threshold points.

The presence of an environmental threshold, however, does not guarantee that a system has alternate states. Gradual changes in physical parameters may result in steep shifts in equilibrium densities within an ecosystem but without multiple stable points at the same parameter values (Petraitis and Dudgeon, 2004). It has been argued by some authors, e.g. Manly and Chotkowski (2006) that the stable or semi-stable states are relatively rare in biological data and that most biological systems are characterised by underlying trends (red noise) that need to be taken into account in any analysis. Where such trends occur, a regime shift would not necessarily result in an abrupt change from one stable state to another but a reversal in the trend (Manly and Chotkowski, 2006). An alternation between presence/absence or dominance of two species from the same trophic level does not necessarily constitute a regime shift. For example, the majority of observations of switches between two ecosystem states in marine systems, such as between sardine-dominated systems and anchovy dominated systems are now recognized as oscillatory responses to events such as El Nino Southern Oscillation (Beamish et al., 2000) or Pacific Decadal Oscillation (Alheit and Bakun, 2010) while pilchard and herring occurrence are due to sunspot and solar activity cycles (Southward and Boalch, 1988) rather than regime shifts (Beamish et al., 2004a). Despite difficulties with the definition and detection of regime shifts in marine systems, the number of scientific publications referring to or describing regime shifts has grown rapidly over the last few decades. Major regime shifts have been identified globally in the 1920s, 1940s, 1970s and the 1990s centered around the years 1925, 1947, 1977, 1989 and possibly 1998. Responses at the ecosystem level have been observed within the North and Central Pacific, Bering Sea, North Atlantic, North Sea, Baltic and the Benguela Current System (Alheit and Niquen, 2004; Alheit and Bakun, 2010; Alheit et al., 2005; Benson and Trites, 2002; Österblom et al., 2007; Overland et al., 2010; Reid et al., 2001a; Sherman and Duda, 2001). One useful approach to the study of ecosystem responses has been the concept of large marine ecosystems (LME). Using this approach analyses are carried out beyond national boundaries in areas defined on the basis of physical and ecological rather than economic characteristics, particularly bathymetry, hydrography, productivity and trophic relationships (Sherman and Hempel, 2009). For coastal North Sea waters, this approach has contributed to the report of a significant regime shift (Reid and Beaugrand, 2002).

The best studied regime shifts are those that have been driven by climatic changes, which can occur on basin scales. However, regime shifts can also occur on much smaller scales, e.g. overfishing (de Young et al., 2008; Sugimoto et al., 2001) or the introduction of alien species (Williamson, 1996). In productive systems such as estuaries, regime shifts have also been described. They can here be the result of coastal hypoxia events (Liu and Scavia, 2010) eutrophication, i.e. excessive nutrient inputs (Hutchinson, 1973; Petersen et al., 2008; Troell et al., 2005). In estuaries the consequences of regime shifts might be expected to be particularly pronounced as they are often important nursery grounds for fish, they are surrounded by heavily populated coasts and estuaries already have high levels of natural stress. However, it has been argued by Elliott and Quintino (2007) that due to these high levels of stress estuarine organisms are also particularly well adapted to environmental change, buffering potential anthropogenic stressors.

Frequently, multiple causes interact as in the case of the comb jelly *Mnemiopsis leidyi*. Its invasion into the Black Sea was underpinned by a combination of eutrophication and overfishing (Bilio and Niermann, 2004; Oguz et al., 2008; Shiganova, 1998) and led to a total re-organization of the food web.

Community composition, species dominance, timing of reproduction and peak abundance of marine species can all be affected by a regime shift. As individual species occupy different ecological niches and therefore respond differently to environmental conditions, ecosystem function is affected (Hutchinson, 1957; Wiens et al., 2009) e.g. via alterations of predator–prey relationships (so called match–mismatch scenarios). These commonly occur as the result of changes in phenology where rising temperatures cause the reproduction of the prey to shift towards earlier in the year while the reproduction of the predator remains unaffected (Durant et al., 2007; Edwards and Richardson, 2004; Hipfner, 2008). Edwards and Richardson (2004) demonstrated such shifts in a range of dinoflagellates, branchiopods and echinoderm larvae, whereby the seasonal spring and autumn peaks have moved forward significantly, i.e., they occur earlier in the year for some species. Interestingly, the time of the peak bloom actually shifted later in the year for other planktonic species, with no consistent trend amongst taxonomic groups. Such match–mismatch phenomena have also been shown in benthic assemblages (Schmalenbach and Franke, 2010). Strasser and Gunther (2001) found a delay, after severe winters, in the biomass build-up of the common shore crab *Carcinus maenas*, which facilitated a greater abundance of bivalve species as their phenology remained unchanged.

2. Regime shifts in the global oceans during the twentieth century 1940s–1960s

In 1947 the Canadian Pacific sardine fishery suddenly collapsed, followed 4 years later by the US fishery (McCall, 1979). These events were thought to be triggered by overfishing and unfavourable environmental conditions (Ware and Thompson, 1991) including a major shift in the winter Pacific Decadal Oscillation (Overland et al., 2010). Survival of Pacific halibut was also negatively affected, with declines in productivity from 1959 to 1972 associated with this event (McCaughran, 1997). High sardine biomass was recorded in the waters around Japan and British Columbia during the 1930s and 1940s, but a dramatic decline followed during the 1950s and 1960s which was attributed to the 1947 regime shift (McFarlane et al., 2000; Schwartzlose et al., 1999). Low numbers of Pacific ocean perch were also recorded post 1947 (Schnute and Richardson, 2001). In the northern Benguela current off the west coast of Africa, a regime shift from a sardine-dominated system to one dominated by jellyfish, pelagic goby and horse mackerel occurred between the 1950s and early 1960s (de Young et al., 2004). Fishing also was to have played a major role in this shift via top-down control (Cury and Shannon, 2004).

2.1. 1970s

More information is available for changes in abiotic and biotic variables surrounding the regime shift that occurred in 1977–1978 reflected in the increased availability of climatic and biotic data. The atmospheric and oceanic drivers responsible were intensification of the wintertime Aleutian Low Pressure Index (ALPI) (Beamish and Boullion, 1993; Trenberth and Hurrell, 1995) year-round cooling of the central North Pacific ocean and a year-round warming of the central north east Pacific and Bering Sea (Ebbesmeyer et al., 1991; Graham, 1994; Manak and Mysak, 1987; Miller et al., 1994; Trenberth, 1990). Oceanographic data for the Sea of Japan also show an abnormally warm regime between 1976 and 1980.

The physical characteristics and biological responses differed between the five main regions within the North Pacific (Alheit and Bakun, 2010). A responsive decline in top predators such as Steller sea lions, fur seals and harbour seals occurred in the Gulf of Alaska and Bering Sea from the mid-1970s (Rosen and Trites, 2000; Trites, 1992). Further south, the opposite trend was recorded in breeding populations of Stellar sea lions between south east Alaska and Oregon (Bigg, 1988). A change from foraging to piscivorous species within the nekton community of the Gulf of Alaska occurred after 1977 (Anderson et al., 1997; Trites et al., 1999).

Many fish populations also showed clear trends: Pacific salmon catches illustrate a decline from average values in the 1950s to low levels by 1977 (Beamish and Noakes, 2002; Beamish and Boullion, 1993; Beamish et al., 2004a). Increases in abundance were also recorded for instance for western Alaska chinook, chum, pink salmon, walleye pollock, central Alaska chum salmon, West Coast mackerel and pink salmon (Beamish et al., 2004a,b). British Columbia coho, pink and sockeye salmon catch increased to historically high levels (Beamish and Boullion, 1993). Catches of demersal species including crab, pink shrimp and Pacific cod were high during the late 1970s and 1980s. Changes in lower trophic levels include an increase in biomass of copepods in the Alaskan Gyre between 1976 and 1977 (McFarlane and Beamish, 1992) and Central Pacific zooplankton (Beamish et al., 2004a).

Decadal variation in North Korean salmon production has been documented from 1920 until the end of the warm phase in 1980, with a large increase in production in 1976 related to fluctuations in oceanographic factors (Roh, 1986). In the Okhotsk Sea and Bering Sea, abundant cohorts of walleye pollock occurred after 1976, reaching a historically high level (Overland et al., 2010; Zhang et al., 2004). Sardine and filefish abundances increased from late 1976 until 1980, when recruitment failure led to a stock collapse (Zhang and Gong, 2001).

2.2. 1980s-present

Between 1989 and 1997, a large drop in winter mean sea level pressure was experienced over much of the Arctic region. In contrast, increased sea level pressure occurred over the North Pacific, North Atlantic and Northern Europe. In the Japan Sea, winter water temperature at 50 m depth showed negative anomalies during 1966–1986, and positive anomalies thereafter until 2004, indicating a shift from a cold to a warm regime around 1986/1987 (Zhang et al., 2007). Many biological components from plankton to predatory fishes responded to this abrupt change from cooling to warming in the Japan Sea in the late 1980s (Zhang et al., 2007). Diatom abundance showed a step change from positive to negative anomalies in 1991. Zooplankton biomass was low around the time of the 1989 regime shift compared to the 1970s and 1990s. A time lag of several years was evident in the demersal fish assemblage. Cold water species declined in abundance and range coverage, whereas warm-water species increased. Japanese anchovy and common squid, and several higher trophic fishes such as yellowtail also significantly increased in abundance in the 1990s. The Tsushima Warm Current ecosystem shift was observed across trophic levels from fish to plankton, and was linked to the Asian monsoon and Arctic Oscillation (Tian et al., 2008). A reversal of the increasing trend in abundance post 1977 also occurred in western Alaska chinook, cum, pink salmon and many more North Pacific fish species (Beamish et al., 2004a). British Columbia's fisheries landings and productivity statistics show synchronous changes since 1989, which are consistent with a change in the composite index of atmospheric forcing, the ALPI (McFarlane et al., 2000). Central Pacific and Alaskan Gyre zooplankton biomass also increased after 1989 (Brodeur and Ware, 1992).

The origins of the regime shift of 1987 remain unclear, but it is likely that the European Shelf Edge Current played an important role (Holliday and Reid, 2001). Strong northward advection of warm water along the European shelf edge coincided with 2 pulses of oceanic inflow into the North Sea in the winters of 1989 and 1998 (Reid et al., 2001b). From 1988 onwards, the NAO index increased to the highest positive level observed this century. Other factors including higher sea surface temperature, earlier and more intense stabilization of the water column, reduced grazing pressure by a change in zooplankton community structure (Greve, 1994) and changing circulation patterns in the North Sea may also have been contributory factors.

Changes in zooplankton community structure occurred in the Southern Benguela in the 1950s when sardine was dominant, and the 1990s when anchovy fluctuated (Richardson and Verheye, 1998). It is unlikely that observed changes in pelagic fisheries played a significant role in driving the changes in the anchovy and sardine in the system (Cury and Shannon, 2004). Unlike in the northern Benguela, the observed change in ecosystem function does not seem to have been extensive enough to label it a shift in regime. Instead the changes are more likely to be fluctuations within the bounds of natural variability (Cury and Shannon, 2004). Fishing has also been an important factor in the regime shift that occurred in the Namibian ecosystem (Cury and Shannon, 2004) by fishing down the foodweb causing a decline in the mean trophic level (Pauly, 1998). Jellyfish have now reached record abundances to the extent that research surveys are being affected and fishing practices disrupted (Sparks et al., 2001).

3. Regime shift detection methods

Regime shifts can only be detected in data sets, which cover a substantial period of time, so that data can be compared from before, during and after a potential regime shift.

Today, many analyses rely on one or a combination of a few long-term data sets (using environmental and biological variables) rather than carrying out analyses over a wide geographic scale. One example of a more comprehensive study is that of Hare and Mantua (2000) in which the authors used Principle Components Analysis on a combined database of 100 individual data sets to detect regime shifts in the Pacific (but see (Rudnick and Davis, 2003) for a criticism of the method). Weijerman et al. (2005) carried out a similar study on over 70 data sets in the North Sea also using PCA and combining it with 'chronological clustering', to confirm the main regime shift periods (1977–1978, 1988) reported previously. However, that study relied to some extent on data from commercially harvested species (in the form of catch data, which is also difficult to standardize and may be affected by many confounding factors including fishing mortality). The IPCC guidelines for assessment reports demand that time series analysing biological changes must be at least 20 years long, extend several years prior to and after the proposed regime shift, and end in 1990 or later (Richardson and Poloczanska, 2008). Weijerman et al. (2005) have set a number of even more stringent criteria for choosing suitable data sets:

1. They should cover at least 30 but ideally more than 40 years.
2. Large spatial coverage.
3. Little or no missing data values in the data set.
4. No obvious local anthropogenic driver for sudden changes in a data series.

Generally, since regime shifts can affect the whole ecosystem, observations should include physical and biological parameters and cover more than one trophic level in order to identify the control mechanisms within the ecosystem (Collie et al., 2004; Tian et al., 2008). Systematic observations should extend over a period of several years before and after the shift to distinguish the inter decadal signal from higher frequency variations.

The methods employed to analyse regime shifts are manifold and involve various degrees of complexity. Many of them originated in the field of climate research and oceanography but are now also applied to biodiversity data sets. Two sets of methods have to be distinguished: Descriptive methods and true statistical significance tests.

3.1. Descriptive (exploratory) techniques

Principal Component Analysis (PCA) is often used as a preliminary analysis to find the dominant abiotic and biotic factors driving a system. PCA is essentially the reduction of a large number of correlated time series and/or variables into a small number of uncorrelated ones (the principal components) containing as much as possible of the original total variance (Mantua, 2004; Schlüter et al., 2008), see (von Storch and Zwiers, 1999) for a review of the method). As an exploratory technique it is usually followed by further tests, e.g. canonical correlation analysis to investigate both patterns in the data and underlying environmental gradients or chronological clustering (Andersen et al., 2009; Bell and Legendre, 1987; Legendre et al., 1985).

Some authors, e.g. Schlüter et al. (2008) have also used PCA to investigate step changes, i.e. regime shifts in time series. Further examples of the application of PCA to environmental and biodiversity data are investigations into inter-annual changes in environmental variables (like sea surface temperature, surface salinity, inflow and wind speed) (Edwards et al., 2007) or in zoo- and phytoplankton counts/abundances (Beaugrand et al., 2000; Edwards et al., 2007; Reid et al., 2001b). Although PCA is widely used it is also of course limited to relationships that are linear. Where non-linear relationships are suspected other methods, such as non-linear multivariate regression models have to be used (Lee and Verleyen, 2007).

The calculation of cumulative sums is a second regularly applied method in recent regime shift studies (Beaugrand et al., 2000; Gieskes et al., 2007; Ibanez et al., 1993). This method involves the subtraction of a reference value, usually the mean of the time series from the time series data points. The residuals are then successively added and plotted in a cumulative sums plot, which will have a positive slope if successive values (e.g. annual means) are above the overall mean and a negative slope if the values lie below the overall mean. This provides a graphical representation of potential regime shifts. This method is now widely applied to biological data (Beamish and Noakes, 2002; Leterme et al., 2005).

3.2. Tests for statistical significance

As with all other techniques a preliminary investigation of the data is necessary to check whether data are normally distributed or whether there are underlying trends, e.g. between species abundance and time (Manly and Chotkowski, 2006). These are often used to relate changes in a time series to possible underlying abiotic factors. Here different regression and correlation methods are in common use. To detect an actual step change in a data series regression analyses with a forward selection step have been employed, e.g. by Wiltshire et al. (2008). Potential change points can also be detected using sequential *t*-tests. In this type of analysis pairwise comparisons are used to check whether the next value in a time series is significantly different from the previous regime. Where this is the case, further observational data are used to confirm or reject the regime shift. This method is problematic in that it is strongly influenced by the chosen length of the regimes (Möllmann et al., 2009).

Gaining more popularity are iterative techniques such as Markov chains Monte Carlo and different bootstrapping techniques (Rodionov, 2005b). These use different algorithms for repeated resampling of a given set of data to test the statistical significance of changes in these data. These methods are used for the detection of step changes as well as for changes in the trends of a dataset. Unfortunately none of the above methods can be recommended universally for the investigation of regime shifts. Whether a method is suitable or not will for instance depend on the type of data (e.g. presence-absence data vs. count data), the data's underlying distribution and whether there are underlying trends in the data, e.g. correlations with temperature or salinity and these factors need to be taken into account when choosing an analysis tool (Manly and Chotkowski, 2006). A recent study involving datasets from multiple trophic levels for both benthic and pelagic systems in UK coastal seas also highlights problems with the identification of 'false positives' when using regime shift detection software, whereby trends in the data can be misinterpreted as step-shifts in ecosystem dynamics (Spencer et al., in press).

4. Management of regime shifts

The regime shifts listed so far have been partly caused by anthropogenic drivers. Human impacts on key physical forcing factors such as climate, fishing pressure, pollution and eutrophication may lower the resilience of ecosystems to the extent that discontinuous regime shifts can also arise from natural disturbance (Collie et al., 2004; Elliott et al., 2007; Folke et al., 2004). In addition, alien species are more likely to become successfully introduced if the system has been disturbed, as natural resistance is degraded by forcing of the system to an alternative state (Williamson, 1996). Building and maintaining resilience of desired or current ecosystem states is therefore likely to be the most pragmatic and effective way to manage ecosystems in the face of increasing environmental change (Scheffer et al., 2001a) but see (Troell et al., 2005).

However, the potential longevity and uniqueness of alternative ecosystem states, causative factors and consequences of regime shifts and linkages between ecosystem components make it difficult to assess the resilience of the system and therefore also the detection, let alone management of regime shifts. Our understanding of large-scale drivers is further limited by our observational capabilities (de Young et al., 2008). In order to predict future regime shifts and develop management responses to rapid alterations in biodiversity and the structure and functioning of ecosystems, we first need to develop robust methodologies for detecting previous regime shifts. Developing research indicates that the delay-time for detection of a regime shift can be rapidly reduced once a functional model has been developed (Spencer et al., in press) under the condition that continuous, long-term time-series data are available from several trophic levels, cover sufficient years before and during the regime shift and continue after the event. If regime shifts (e.g. changes in key species and the food web) have occurred, they can hardly be reversed. Therefore, the prevention of regime shifts is the most effective action. The causes for regime shifts, if identified, can be managed. They are summarized in Fig. 1. Several political initiatives are aiming at prevention or mitigation of these adverse ecosystem changes.

5. The political context

5.1. Major policy initiatives

The European coastal zone is linked to global trade and transport, is a major source of raw materials and food and contains a

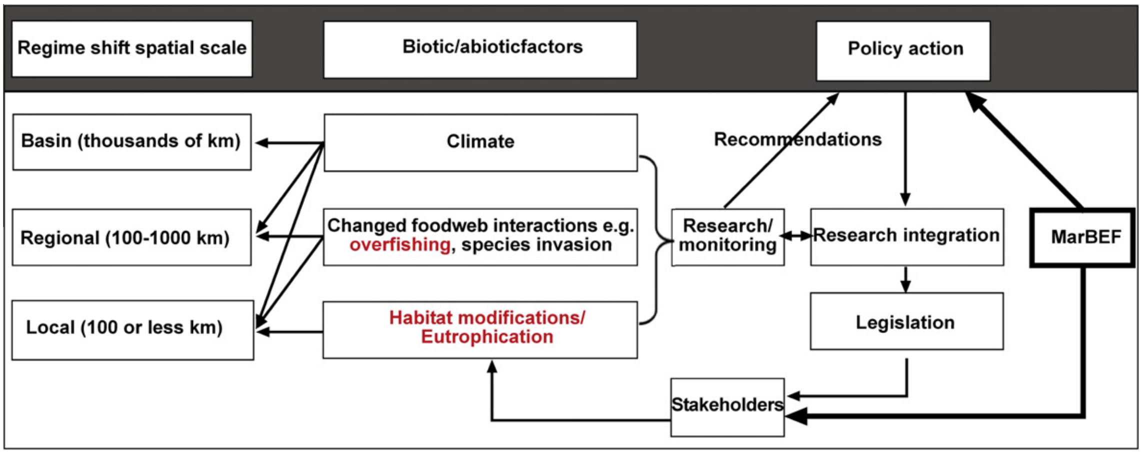

Fig. 1. Summary graphic of the biotic, abiotic and political components involved in the management of potential regime shifts.

diverse array of valuable natural habitats but for a long time, the management of these resources has not been seen in an ecosystem context. The Millennium Ecosystem Assessment Report (http://www.millenniumassessment.org/en/index.aspx) was one of the first to identify that growing populations, increasing human occupation and activity in coastal areas are exerting elevated pressures on coastal ecosystems as a whole potentially reducing their resilience (Agardy et al., 2005). This broader approach to marine resource management has now also been adopted by the European Commission. The Maritime Policy Green Paper, adopted in 2006 stated that while facilitating continuing competitiveness in the global market remained the goal of European policy, this had to be achieved in a sustainable manner (http://europa.eu/scadplus/leg/en/lbv/166029.htm). Such sustainable management needs to take into account a number of anthropogenic drivers such as eutrophication, and in this context the paper explicitly mentions the risks to coastal marine biodiversity (see Table 1 for a summary of the major initiatives).

This broader view of regime shifts is also reflected in the large marine ecosystems (LMEs) concept introduced in Section 1. On a global scale, researchers have described 50 LMEs.

The LME framework has been developed and tested around the world to implement an ecosystem-based approach. It integrates five modules made up of case studies on: (1) ecosystem productivity, (2) fish and fisheries, (3) pollution and ecosystem health, (4) socioeconomic conditions and (5) governance. The modules provide indicators of the changing states of LMEs with regard to ecological condition, socioeconomic consequences and governance rules, i.e. within the LME areas the 5-point strategy includes both the detection of change and options for remedial action (Duda and Sherman, 2002).

With increasing evidence for the ecological consequences of climate change, the European commission has also begun to address the possible impact of climate change on Europe's Marine Resources in its White paper 2009. The White paper again stresses the risks to biodiversity and coastal ecosystems as one key reason why adaptation measures are needed since large-scale biodiversity changes carry the risk of affecting the ecosystem's resilience to further environmental changes (Commission, 2009). It describes the currently fragmented approach to climate change adaptation measures across the European Union and seeks to reduce the latter's vulnerability to climate change impacts by mainstreaming adaptation measures into EU policies and ensuring coherency across spatial boundaries and levels of governance. This common strategy will also form the basis for the Framework of the Integrated Maritime Strategy and the reform of the Common Fisheries Policy.

The Marine Strategy Framework Directive (MSFD), launched in 2008 aims to bring all of the existing legislation on biodiversity under a single European umbrella to reduce the current decline in biodiversity and achieve Good Environmental Status in the Community's marine environment at the level of marine regions and sub-regions by 2020 (European Parliament and Council, 2008; Mee et al., 2008). Crucially the MSFD also requires the participation of local stakeholders in this process (Fletcher, 2007; McCaughran, 1997; Mee et al., 2008).

The Biodiversity Communication 'halting the loss of biodiversity by 2010' released in May 2006 (http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=CELEX:52006DC0216:EN:NOT) was specifically formulated to address negative impacts of socio-economic activities on marine ecosystems. Relevant commitments are also contained in the decisions taken under the Convention on Biological Diversity (CBD) and the United Nations Framework Convention on Climate Change (UNFCCC).

5.2. Regimes shifts in the political context

The implementation of the above directives has direct implications for the management of marine ecosystems including attempts to develop management strategies in relation o regime shifts. Climate change is acknowledged to significantly affect biodiversity with 20–30% of species threatened by a global temperature increase of 2 degrees or more (Thomas et al., 2004). At the same time this increase will also cause shifts in the distribution of species, i.e. immigration into new areas (Commission, 2009). Ecological communities and food web interactions are therefore going to change considerably. This is likely to also promote ecological regime shifts with negative consequences for ecosystem services such as fisheries, particularly in marine ecosystems which are already affected by other human activities (Casini et al., 2008; Kirby et al., 2009; MacKenzie et al., 2007). Therefore the MSFD requires the establishment of reference or baseline conditions in marine ecosystems, i.e. define, where possible, the pristine state of a system, against which future changes can be compared to determine whether Good Environmental Status (GES) is being maintained. The definition of GES descriptors, especially with reference to 'pristine' conditions is still an area of major debate; often targets have been set against an arbitrary benchmark snapshot of the state of an ecosystem, with no consideration of how climate change will alter the dynamics of coastal marine systems away from the reference condition irrespective of adaptation and mitigation actions implemented (Devlin et al., 2007).

Table 1
Directives and conventions published in the last 35 years.

Directive/regulation	Inception	Policy driver	Deadlines
HELCOM Declaration	1974	Towards an ecosystem approach to the management of human activities	
Birds Directive	1979	The maintenance of the favorable conservation status of all wild bird species across their distributional range, identification and classification of Special Protection Areas for rare/vulnerable species, with particular attention to the protection of wetlands of international importance	
Environmental Impacts Assessment Directive	1985	Assess whether large-scale developments likely to cause environmental damage	
Habitats Directive	1992	Means by which the Community meets its obligations as a signatory of the Convention on the Conservation of European Wildlife and Natural Habitats. Establishment of Special Areas of Conservation	Third report in 2012
OSPAR Convention	1992	Towards an ecosystem approach to the management of human activities	Quality Status Report 2010
United Nations Framework Convention on Climate Change	1994	Protect the climate system for the benefit of present and future generations of humankind	
Pan-European Biological and Landscape Diversity Strategy	1996	Support implementation of the Convention on Biological Diversity and integrate ecological considerations into all relevant socio-economic sectors	
Convention on Biological Diversity	1997	Halting biodiversity loss, ensuring the conservation and sustainable use of marine biodiversity, and on the creation of a global network of marine protected areas	2012
European Community Biodiversity Strategy	1998	to anticipate, prevent and attack the causes of significant reduction or loss of biodiversity at the source	
Natura 2000	2000	Establishment of a marine network of conservation areas	
European Strategy for Integrated Coastal Zone Management	2000	Enhance the integration of multiple uses and sustainability in European coastal zones	
Gothenburg target	2001	Halting the loss of biodiversity	2010
Biodiversity Action Plans	2001	Halt the loss of biodiversity	2010
Strategic Environmental Assessment Directive	2001	Assessment of the effects of certain plans and programmes on the environment	
Johannesburg Commitment	2002	Establish a network of Marine Protected Areas	
Malahide Message	2004	Promote cross-border ecological networks	
Thematic Strategy on the Protection and Conservation of the Marine Environment	2005	Promote sustainable use of the seas and conserve marine ecosystems	
Thematic Strategy on the Protection and Conservation of the Marine Environment (European Marine Strategy)	2005	All European seawaters are in good environmental status	Deadline 2021
EC Biodiversity Communication	2006	Halt the loss of biodiversity	2010
Maritime Policy Green Paper	2006	Launch a debate about a future Maritime Policy for the EU that treats the oceans and seas in a holistic way and dovetails with the Lisbon Strategy	
Adapting to climate change in Europe – options for EU action Green Paper	2007	Adaptation strategies	
Marine Strategy directive	2008	Establish a framework for community action in the field of marine environmental policy	2020

A regime shift can force the status of the ecosystem far from the reference conditions as assessed for national and European conservation directives, and it is unlikely that the arbitrarily selected 'pristine' ecosystem states can be achieved again. Indeed, from a biodiversity perspective it seems neither possible nor necessary to regain the previous 'pristine' species composition of the ecosystem as long as functionality of the ecosystem is assured (Mee et al., 2008; Ruiz and Aide, 2005). For example, Wasmund et al. (2008a) found a shift in phytoplankton species composition in Kiel Bight during the last century, but maintenance of class diversity. If, however, the diatom spring bloom were to disappear or have an altered timing for a prolonged period, as found in 2007 in Mecklenburg Bight (Wasmund et al., 2008b), this would have consequences for zooplankton, zoobenthos and higher trophic levels and therefore exert a potentially negative impact on the ecosystem as well as pose a threat to ecosystem services. The diverse range of problems and pressures on the natural resources of the various marine regions or sub-regions constituting the marine environment in Europe require different and specific solutions. Therefore the requirements of existing directives to determine reference systems for different ecosystems cannot be achieved through the assessment of species richness, both historical and recent. What is required instead is a combined approach of *in situ* monitoring of physical and biological parameters, phenological observations and field experiments are essential to establish the impacts on biodiversity and ecosystem level responses to both regime shifts and the more gradual background of global climate change.

One of the biggest impediments to progress for an efficient ecosystem management is that at present, the period of integration of scientific information into national and European legislation is often lengthy, due to both the time delay necessary to obtain the wide temporal data coverage required to detect change, and the progression of political understanding with regards to the applicability of such data to assist in the delivery policy targets and outputs (Ducrotoy and Elliott, 1997). This could be shortened considerably by providing better mechanisms for both data integration and data dissemination. This problem was also acknowledged in the abovementioned White paper which demands the establishment of a clearinghouse mechanism for biological data, recognizing the need for standardized, widely available and quality controlled data for the analysis of large scale and long-term environmental changes, including regime shifts. The essential role of information on long-term change in marine ecosystems was also highlighted earlier during the initial development of conservation policies in the UK in the 1960s (Hiscock, 1996) and HELCOM, the Helsinki Commission to protect the Baltic Sea marine environment (http://www.helcom.fi/Convention/en_GB/text/).

These directives require information on biodiversity, ecological status and stability of marine environments to provide a knowledge base from which to develop fit-for-purpose management and adaptation action plans. Biodiversity should be taken into account at all stages of the preparation of marine strategies, but especially during the planning and execution of measures to achieve GES in the Community's marine environment at the level of marine regions and sub-regions. This need was also clearly voiced in the Valencia declaration, developed by world leaders in Biodiversity research at the world conference on Biodiversity in November 2008 (http://www.marbef.org/worldconference/docs/The_Valencia_Declaration_20081115.pdf).

The problem is that while regime shifts are currently, with few exceptions, examined on a very large scale, management strategies, which are based on European directives and laws, are being implemented on a more regional and short-term scale. However, to investigate regime shifts, and their consequences for marine biodiversity and ecosystem function, observational data from a large number of locations around Europe need to be compared over a long period of time through the formation of cross-border ecological networks (Jones-Walters, 2007). Only integrated research and monitoring programmes across national and regional scales will make it possible to separate different drivers and to devise effective management strategies for marine biodiversity resources (Beamish et al., 2004a) and ecosystem services such as fisheries (Rothschild and Shannon, 2004). Unfortunately the lack of suitable long-term data provides a considerable bottle neck to carrying out such assessments. Particularly the geographical distribution of time series is a problem when attempting European wide comparisons as some areas are much more intensively monitored than others. This makes it even more important that existing data sets are integrated; gaps identified and filled, and ultimately be made widely available to the international scientific community. There are already some international precedents such as the Census of Marine Life (CoML) Ocean Realm Field Project NaGISA (Japanese for the nearshore zone) which places a focus on coastal ecology. An early focus of NaGISA was the collection of biodiversity data from sites around Asia, but it is now a project of worldwide relevance with many partners in Europe. Within the EU Network of Excellence MarBEF (Marine Biodiversity and Ecosystem Functioning), NaGISA aims at the archival of taxonomic and joint analysis of ecological data based on data collected with standardized methodologies. The MARS network is a foundation created by and open to Europe's marine research stations. MARS serves as a virtual hub for research project dissemination, as a forum for knowledge exchange between European scientists and communicates with international organisations and the managers of European research, including the Commission of the European Community in Brussels and the Marine Board of the European Science Foundation in Strasbourg. The UK Marine Environmental Change Network (MECN) was established by the Marine Biological Association of the UK and the UK government department Defra in 2002 to provide a consistent platform from which to promote sustained physical and biological observations within UK regional seas. It provides a national focal point for scientists and policymakers involved with long-term marine time-series work. MECN is also the national contact point for collaborative links with wider European initiatives such as MarBEF.

6. The LargeNet European database

In recent years, European funding programmes have moved towards the creation of large-scale networks that can acquire the necessary data to support and develop European policies. The network of Excellence MarBEF (Marine Biodiversity and Ecosystem Functioning), funded within the 6.Framework Programme and consisting of 94 European marine institutes, had, for instance, a clear remit to integrate and disseminate knowledge and expertise on marine biodiversity, with links to researchers, industry, stakeholders and the general public. Particularly, with respect to data integration, MarBEF activities have had a considerable impact. Within MarBEF joined analyses of multiple datasets were carried out, probably for the first time on a European scale. But, even more importantly, the data analyses pointed out the current limitations in carrying out large-scale analyses.

The principal aim of LargeNet, a project within MarBEF, was the compilation of multiple datasets individually collected by scientists across Europe in a single, standardized, electronic database (http://www.marbef.org/projects/largenet/index.php). The database is intended to provide a central resource to increase the scope and geographic/temporal scale of analyses already carried out by partner institutes and to facilitate additional meta-analyses to answer regional as well as basin-scale questions regarding changes in biodiversity and their potential drivers. This was to be achieved by assembling a range of datasets with appropriate accompanying metadata and integrating them into a single, quality assured, common format. This was to allow:

1. An assessment of the comparability of the various datasets in terms of methodologies.
2. More intensive analyses of individual data sets.
3. The identification of gaps in data coverage.
4. Meta-analyses involving multiple data sets.

Between 2006 and 2008, LargeNet has compiled, centrally archived and standardized data collected by 19 MarBEF partner institutes. The LargeNet database currently holds 69 data sets covering the biodiversity of pelagic, rocky shore and soft bottom habitats. It comprises 542,000 distributional records for 4525 species at 17,117 unique stations spanning a longitudinal range from 24 degrees West to 59 degrees east and 43.2 (37.9–81.1) degrees of latitude from the Arctic, Baltic, North Sea, English Channel, Irish Sea, North East Atlantic and Mediterranean Sea. Every data point was standardized in terms of taxonomic name (using ERMS as standard), geographic location (WGS84 latitude and longitudes) and scientific units of measurement (SI units) to ensure a common format was adopted across the database. LargeNet is not the only result of the extensive data acquisition within MarBEF. The MANUELA project (specializing in meiofauna) and Macroben (macrozoobenthos) databases are further examples of these activities. Their mechanisms of archival and data integration are identical to those used in LargeNet thus facilitating integration of information from these systems. The most extensive dataset, the CPR survey data, although not integrated into the LargeNet database was also archived as part of MarBEF activities and was also made widely available via EurObis.

7. Integrating between MarBEF data sets: North Sea and Baltic regional data

Here we will review in more detail the situation in the North and Baltic Seas on the basis of taxon groups, taking into account the data collected in the LargeNet project (supported by additional literature data for data sets not included in the database, and contributions from external colleagues) and whether these can be used to address requirements of existing and planned European directives.

7.1. The North Sea

The North Sea is the most intensively studied area in Europe with respect to regime shifts. Several factors make it an important system for a review of regime shift studies, their methodologies and political relevance: 1. The large number of available long-term data sets, with all components of the marine food web sampled (Table 2), 2. The heavy use of ecosystem services and therefore the need for efficient management strategies, 3. The dense human population in the coastal areas surrounding the North Sea giving the issue of protecting the North Sea as a region of high biodiversity and as an exploitable resource even more urgency (Winter and Bartholomae, 2006).

7.1.1. Phyto- and zooplankton

The only formal regime shift analyses were carried out with phytoplankton and to some extent zooplankton, using data from the SAHFOS continuous plankton survey and the Helgoland Roads long-term data sets. These have shown shifts in phytoplankton and zooplankton abundance and composition during the regime shifts in the 1970s and 1980s (Leterme et al., 2005; McQuatters-Gollop et al., 2007; Reid, 1975; Reid et al., 1998; Schlüter et al., 2008). The 1980s regime shift, probably linked to increased Atlantic

Table 2

Summary of major North Sea biotic data series including information on the sampled parameters and statistical methods used for the evaluation of regime shift and long-term changes in general (series shaded in grey have been archived in one of the MarBef databases). (See below-mentioned references for further information.)

Time series	Location	Duration	Taxa covered	Criteria fulfilled?	Shifts detected	Methods used
North Sea						
Norderney	North of the Eastfrisian Island of Norderney	1978-present	Macrozoobenthos		Yes, in response to change in sign of NAO, but changes were gradual, no true regime shift	EOF (variant of PCA) followed by CCA
BIS data set	Dutch Coast	1985–2004	Macrozoobentos	No		
SAHFOS	Many transects throughout the North Sea	Started in 1930s others in 1958	Phytoplankton, zooplankton	Yes	1978, 1983–87	PCA, cumulative sums/anomaly plots
Helgoland Roads	Island of Helgoland, southern, German Bight	1962-present	Phytoplankton	Yes	Shifts confirmed for 1978/1988	PCA, CCA, Bayesian techniques, regression analysis
Waddensea Station Sylt	Island of Sylt, at border between Germany and Denmark	1984-present	Zooplankton	No	Possibly a minor shift in 1997 (increasing zooplankton abundance)	Correlation analysis, no specific regime shift methods used
Wadden Sea Station Sylt	Island of Sylt, at border between Germany and Denmark	1987-present	Zhytoplankton	No	Increased periods of nitrogen limitation since 1991	Loebl et al. (2007)
L4 zooplankton monitoring	Western English Channel	1988-present	Zooplankton	No	?	?
SPEEK (MANUELA)	Kwintebank, sandband on the Belgian Continental Shelf	Sampled in 1978, 1997, 2003 and 2004	Nematodes	No	Yes, but as a consequence of sand extraction	Multivariate methods, no specific regime shift methods used
Dove Marine Lab	Northumberland coast, UK	1972-present	Macrozoobenthos, Zooplankton			
Baltic						
IOW Phytoplankton sampling		1980-present	Phytoplankton abundance, temperature	No	Decrease in diatom abundance in the late 1980s	
N3	Kiel Bay	1986–2004				
IOW macrozoobenthos monitoring	Mecklenburg Bight, Darss Sill, Greifswalder Bodden, Pomeranian Bight	1980-present	Macrozoobenthos abundance, biomass (wet/dry weight)	No	Apparently no large-scale changes observed (Zettler et al., 2006)	Biodiversity indices, multidimensional scaling but no formal regime shift analysis, Loess smoother
Estonian monitoring programme	Several locations	Since 1954	Phytoplankton, zooplankton and ichthyoplankton	No	Monitored species introductions	No formal regime shift analyses carried out
Estonian monitoring programme	Vaeinameri archipelago sea	1993–2003 (data from the 60s also available)	Macrozoobenthos	No	Several species introductions documented, several amphipod and isopod species lost from the system	Orav-Kotta et al. (2004)
Latvian Fisheries Research institute	Gotland Basin, Bornholm Basin, Gdansk Deep	1959–1997	Mesozooplankton	No	Increase in sprat abundance, changes in mesozooplankton composition and	Sequential t-tests, pre-whitening technique (Möllmann et al., 2008)
Finnish Institute of Marine Research	Gotland Basin	1965 (approximately annual sampling intervals, no sampling in 1976)	Macrozoobenthos	No (the series started in 1965, but with low, infrequent sampling)	Changes in the abundance of macrozoobenthic species in relation to the level of oxygenation	Mainly descriptive studies so far

inflow, was discernable throughout the system including fish populations such as the horse mackerel (Reid et al., 2001a,b). Particularly a shift from a diatom to a dinoflagellate dominated phytoplankton community was observed during the 1980s regime shift. Schlüter et al. (2008) and Wiltshire et al. (2008) confirmed the known regime shift periods in the 1970s and 80s in the North Sea by carrying out principal components analysis on data from the Helgoland Roads and other long-term data sets. One of the most important determinants of the shift in 1987 was winter SST. Repeated cold events appear to have facilitated the shift itself whereas continued warming might have contributed to the persistence of the new regime (Schlüter et al., 2008). Despite these changes, the authors concluded that the lower trophic levels, i.e. phytoplankton and zooplankton are actually remarkably resistant to long-term change.

A study by Pitois and Fox (2006) showed long-term changes in biomass and composition of zooplankton near the North Frisian island of Sylt. An analysis of data from the LargeNet database also showed an increase in carnivorous zooplankton, e.g. *Oithona similis* in relation to temperature since 1984 in this area (Martens and Van Beusekom, 2008). However, since the series only started in 1984 no meaningful formal regime shift analyses are yet possible.

7.1.2. Meiobenthos

There is a marked lack of datasets that can be used to assess possible regime shifts using meiobenthic organisms. Meiobenthos is a generally neglected component of the benthic fauna (Kennedy and Jacoby, 1999), although it is generally accepted that the dominant meiobenthic taxon, the nematodes are ideal tools to detect changes in the benthic environment due to their ubiquitous distribution, high abundances, high diversity (with species ranging from tolerant to highly sensitive to disturbances), short generation times and direct benthic development (Heip et al., 1985; Schratzberger et al., 2000). The increase in primary production detected in the North Sea during the 1988 regime shift will affect the benthic environment as well, since about 20% of the ungrazed primary production is deposited on the sediment (Lancelot et al., 2005). Seasonal short term increases of phytodetritus on the seafloor indeed triggered significant changes in nematode community composition and standing stock at different locations in the North Sea (Schratzberger et al., 2008; Steyaert, 2003; Vanaverbeke et al., 2004) and thus it can be expected that regime shifts affect the meiobenthos on the long term. Although no formal comparative regime shift studies on the datasets involved could be carried out, the above examples demonstrate that the lack of incorporation of meiobenthos in regular long-term monitoring programmes presents a gap that should urgently be filled. Current political initiatives require us to follow an ecosystem approach to establish marine reference conditions. This is not possible with the current data coverage.

7.1.3. Macrozoobenthos

Long-term shifts in coastal macrozoobenthos off Norderney have been correlated with fluctuations in the sign of the NAO index. Particularly an increase in biomass was observed after 1989. These changes were probably caused by successive warm winters (Kröncke et al., 1998). Changes in benthic communities were also shown on the island of Sylt in the Wadden Sea, where records for sea grass beds extending back to the 1930s, Reise et al. (2008) showed that the many areas previously covered by seagrass and high densities of amphipods, have since the late 1980s been replaced by green algal mats. However, as regular re-investigations of the original survey only began in 1988 these drastic changes cannot automatically be classed as a regime shift as no data for the transition into the new phase exists. In a further study in the western North Sea Frid et al. (2008) investigated the changes in a macrozoobenthos time series in the western North Sea (Northumberland Coast). Although evidence for potential regime shifts was found, the patterns did not seem to be consistent across taxa. Several distinct but potentially linked factors including changing winter temperatures and carbon flux to the benthos were found to exert an influence, but none of them was dominant throughout.

Potential regime shifts (although again not statistically confirmed) have been described as the result of species invasions e.g. *Sargassum muticum*. The latter showed that such invasions can also have beneficial effects. In the case of *S. muticum* biodiversity of the system increased after the invasion due to a greater habitat complexity (Buschbaum et al., 2006; Wiltshire et al., 2010). Although important long-term changes were shown in the above examples, the data series underlying them have not yet been subjected to a formal regime shift analysis.

7.2. Baltic Sea

7.2.1. Phyto- and zooplankton

Alheit et al. (2005) showed an increase in phytoplankton biomass after 1987 in both North Sea and the Baltic similar to that found in the CPR data. The steady increase in chlorophyll *a* concentrations in the Baltic Proper, at least until the late 1990s (Suikkanen et al., 2007; Wasmund and Uhlig, 2003; Wasmund and Siegel, 2008), may not be considered a regime shift, but the sudden disappearance of diatoms from the spring blooms and replacement by dinoflagellates in the Baltic Sea since 1989 (Wasmund and Uhlig, 2003; Wasmund et al., 1998) may have drastic consequences for the food chain. Indeed, Alheit et al. (2005) found a regime shift that was detected on three trophic levels (phytoplankton, zooplankton and fish). While the changes at lower trophic levels, particularly increases in dinoflagellate abundance, were observed in Both the North Sea and Baltic, the responses of fish were more complex and caused by a combination of physical and biological factors (Alheit et al., 2005). The reason was a warming of the winter water since 1988, which prevented the convective spring overturn of the water column (Wasmund et al., 1998), and increasing warm-water inflows into the halocline in summer (Alheit et al., 2005). The regime shift at the end of the 1980s was confirmed by Möllmann et al. (2006) for the central Baltic Sea and the Gulf of Riga, using Principal Component Analysis and Chronological Clustering. They also identified a shift during the early 1990s (1993–1994) in these waters. The identified regime shift was discontinuous, having been caused by climatic factors but being maintained by food web interactions (Möllmann et al., 2009). Also in the North Sea, the regime shift of 1988 was related to temperature and weather conditions, similar to North Pacific studies, which indicates a shift in climate-ocean interactions throughout the entire temperature zone of the northern hemisphere (Weijerman et al., 2005).

7.2.2. Macrozoobenthos

Although many long-term records exist, no formal regime shift analyses have been carried out, and many accounts are descriptive in nature. Zettler et al. (2008) provide an overview on macrozoobenthos species composition in the German part of the Baltic, compiling data reaching back to 1839. However, two-thirds of these data were collected after 1980. During this period of time changes in the benthic composition are visible which are partly relatated to eutrophication (Zettler et al., 2008). Orav-Kotta et al. (2004) documented the disappearance of several amphipods, isopods and polychaetes from the Vaeinameri archipelago. At the same time, the polychaete *Marenzelleria viridis*, which originates from the East coast of the USA, invaded the area and other parts of the Baltic (Zettler et al., 2002) but did not become abundant. In the Aaland Archipelago on the other hand this species became dominant

(Perus and Bonsdorff, 2004). Gröger and Rumohr (2006) determined species richness (species per m^2) and found that the species richness at several sites in the Kiel Bight between 1968 and 2000 was strongly influenced by the NAO winter index and salinity, however for the latter factor a four year time lag was observed. The authors speculate that larvae of macrobenthic organisms are transported into the Western Baltic and it takes several years before these species become apparent in the macrozoobenthos records. Similar results have been found for an earlier period (1975–1976) by Laine et al. (1997) for the Gotland Basin, although the authors did not address larval drift as a potential factor but invoked increased oxygen levels as facilitator of higher rates of colonization of a larger number of macrobenthic species such as polychaetes (Laine et al., 2003).

8. Discussion: The status quo and the way forward

While previous European directives have focused on particular sectors or issues, such as heavy metals, nitrates or fisheries, more recent ones have adopted a more holistic, ecosystem approach (Apitz et al., 2006; Mee et al., 2008). Particularly the Marine Strategy Framework Directive requires the establishment of environmental reference conditions against which future changes can be gauged. Such an approach certainly makes sense in the case of regime shifts, as their consequences can propagate through the whole ecosystem (Polovina, 2005; Polovina et al., 1995; Reid et al., 2001a). A good example is fisheries management, since studies across the globe have documented impacts of regime shifts on fisheries (Alheit and Bakun, 2010), either the complete collapse of fisheries or the shifts in commercially harvestable species, the best example being the switches between walleye pollock and squid in the Japan/East Sea which do not only have ecological consequences but they are also harvested by different types of fisheries and at different times of the year necessitating considerable adjustments in the fisheries sector (Zhang et al., 2007). In order to best manage commercial stocks and preserve sensitive ecosystems it is also vital that we improve our understanding of whether climate change will increase or decrease the likelihood that an ecosystem will revert back to its original state after a regime shift has occurred (Brander, 2010).

A comprehensive approach to ecosystem management requires comprehensive data on 1. All ecosystem components in a particular area and 2. The same type of data for different regions in Europe so that analyses can be carried out on different spatial scales. However, the process of data acquisition and subsequent analyses for the LargeNet project within MarBEF, as well as the additional studies described in this global review, revealed an imbalance between data available for physical parameters and fisheries on the one hand and biodiversity data on the other. This is the case even for heavily sampled areas such as the North Sea (Vanden Berghe et al., 2009; Vandepitte et al., 2010). Even the few large-scale studies that have been carried out, e.g. by Weijerman et al. (2005) could not use data where all trophic levels (e.g. phytoplankton, zooplankton and fish) were sampled in the same location.

Comparisons of biodiversity data sets remain problematic for several reasons, including different sampling regimes and frequencies but particularly insufficient temporal overlap between time series and a very unbalanced data coverage for different taxon groups, for instance a complete lack of long-term data for the meiobenthos and a lack of comparable phytoplankton data. This is particular important, as even a regime shift apparent in a single food web component, e.g. plankton, may often, with a time lag, trigger changes in benthic communities (Gröger and Rumohr, 2006). There is therefore a strong case for maximizing the output from existing data series by adopting, and crucially, maintaining standard data aggregation programmes run by large networks such as the Network of Excellence MarBEF. These networks have implemented an independent data management team that is acting on behalf of all partners to collect and ensure data quality assurance, collating it using a standardized format that can be simply searched and bespoke subsets of data extracted for use in meta-analyses.

This will in future remove the problem of a lack of comparable long-term biodiversity data, which is clearly shown in Fig. 2. The graphs for pelagic and soft bottom data show that the maximum number of time series with a temporal overlap is 6 in each case, with the period of overlap being less than 10 years. (Figs. 2a,b). This time period covers only one of the previously documented regime shifts, which still presents a considerable impediment to the meta-analyses we aspire to.

Currently, to make such data comparable they need to be adjusted to the lowest common denominator, often losing valuable information and resolution in the process. The fact that the comparison of different long-term data with respect to regime shifts is still not possible, demonstrates that, while integrating existing data sets is clearly valuable (Hiscock, 1997), it is not sufficient and should be preceded by a further integration step at the point of data collection with all efforts being made to standardize sampling regimes and just as importantly to support vulnerable data series facing the risk of being shut down. With many European data collection programmes being chronically underfunded, continued financial support for these efforts is therefore also vital.

9. Summary and final recommendations

To summarise, a more holistic approach is required to tackle both adaptation and mitigation with respect to the consequences of regime shifts in the marine environment. The particular scientific challenge of analysing and managing regime shifts in the context of the Water Framework and Marine Strategy directives underlines 1. The importance of maintaining existing monitoring programmes and 2. Establishment of new ones in distinct areas that might be particularly vulnerable to climate change or are of particular ecological/economic significance. These should exploit the experience of current successful monitoring programmes such as HELCOM and OSPAR to avoid duplication (Ducrotoy and Elliott, 1997). Crucially, emerging directives such as the MSFD must consider and not only fund data integration but also the data acquisition process to improve comparability between data sets in future. Consequently we recommend the following actions:

1. The different monitoring programmes have to be conducted jointly by the riparian countries of a particular region, according to agreed sampling programme and analysis techniques to facilitate future large-scale biodiversity assessments (Scholes et al., 2008).
2. The regional monitoring programmes have to be joined under a European umbrella. This includes a common strategy, sampling and analytical methods according to European standardization (methods according to CEN, species lists ERMS), harmonized data treatment and data storage in a common database. The need for this has already been acknowledged implicitly by the European commission through its plans for the development of a European Data Clearing house mechanism. This will fasten the transition from scientific findings into coherent policy.
3. Intercalibrations and other measures for the quality assurance of biodiversity data (e.g. taxonomical courses, establishment of taxonomic expert groups or taxonomic centres) need to be adopted.
4. Regular assessments of the environmental state by the scientists involved, using appropriate statistical tools, also taking

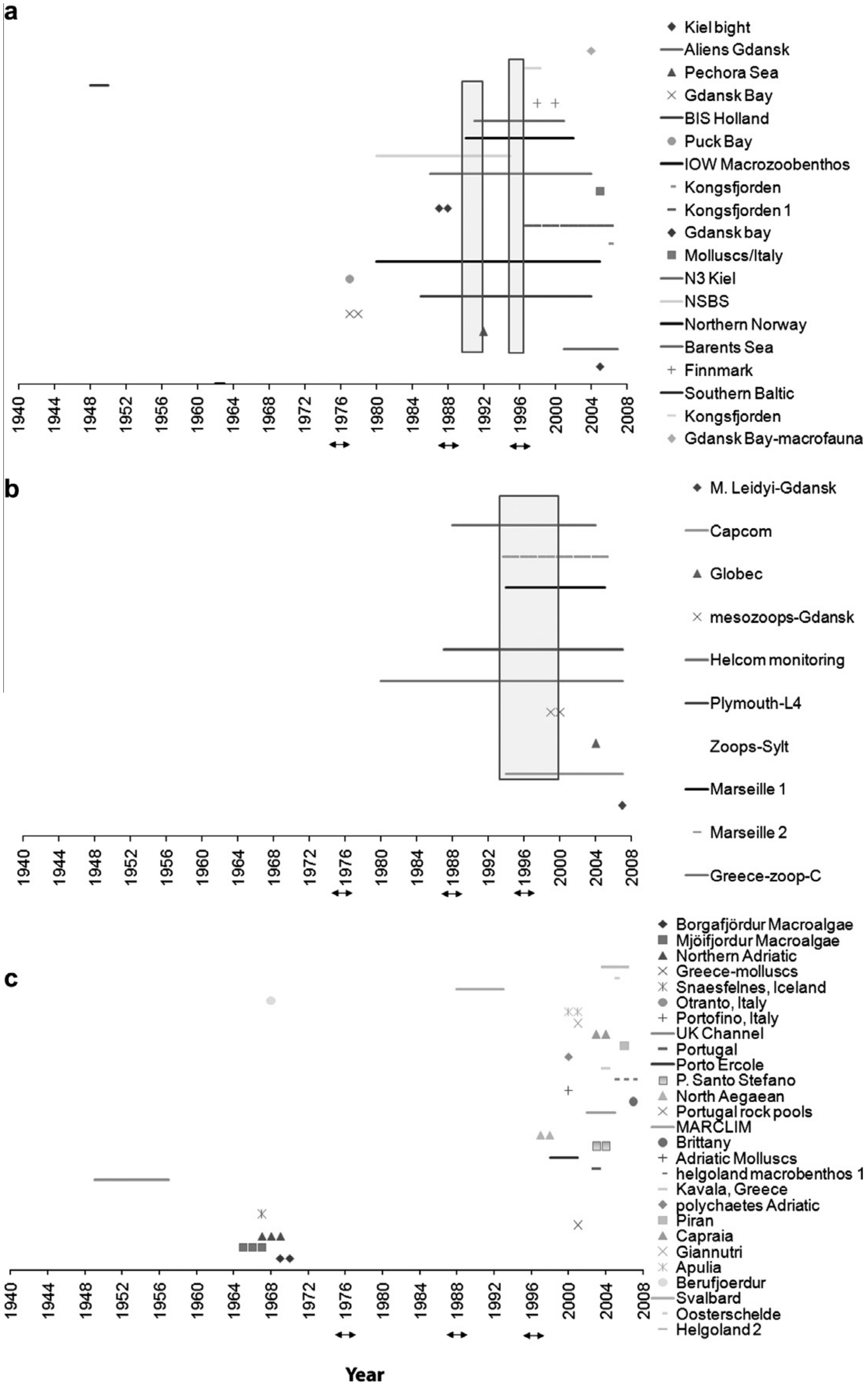

Fig. 2. Graphic showing the temporal overlap in the 69 data sets within the LargeNet database: (a) soft bottom benthos data, (b) pelagic data and (c) rocky shore data. Arrows indicate the periods of known – and investigated-regime shifts. Grey boxes indicate the time periods where the maximum numbers of time series overlap.

into account socio-economic demands are necessary. Outreach of the results to the stakeholders and recommendations for management.
5. The focus on as yet understudied key taxonomic groups for instance the meiobenthos must be increased to include them in regime shift analyses.
6. The integration of data obtained with novel methods, e.g. molecular time series have to be considered, as they can considerably increase the taxonomic resolution of biodiversity data.
7. Better taxonomic integration of data sets and support of taxonomic experts have to become a focus: Coherent collaborative biodiversity research is possible only with detailed taxonomic expertise. Multiple datasets from different sites are only useful if similar and comparable taxonomic expertise is applied to these data sets to avoid bias in the identification process and the resulting data sets. These studies are particularly important in the context of the Marine Strategy Framework directive for the definition of regional reference conditions and to achieve or maintain Good Environmental Status.

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