Regime Shifts – A Global Challenge for the Sustainable Use of Our Marine Resources

Camilla Sguotti and Xochitl Cormon

Abstract

Over the last decades many marine systems have undergone drastic changes often resulting in new ecologically structured and sometimes economically less valuable states. In particular, the additive effects of anthropogenic stressors (e.g., fishing, climate change) seem to play a fundamental role in causing unexpected and sudden shifts between system states, generally termed regime shifts. Recently, many examples of regime shifts have been documented worldwide and their mechanisms and consequences have been vigorously discussed. Understanding causes and mechanisms of regime shifts is of great importance for the sustainable use of natural resources and their management, especially in marine ecosystems. Hence, we conducted a session entitled "Ecosystem dynamics in a changing world, regime shifts and resilience in marine communities" during the 8th YOUMARES conference (Kiel, 13–15th September 2017) to present regime shifts concepts and examples to a broad range of marine scientists (e.g., biologists and/or ecologists, physicists, climatologists, sociologists) and highlight their importance for the marine ecosystems worldwide.

In this chapter, we first provide examples of regime shifts which have occurred over the last decades in our oceans and discuss their potential implications for the sustainable use of marine resources; then we review regime shift theory and associated concepts. Finally, we review recent advances and future challenges to integrate regime shift theory into holistic marine ecosystem-based management approaches.

C. Sguotti (✉) · X. Cormon (✉)
Institute for Marine Ecosystem and Fishery Science, Centre for Earth System Research and Sustainability (CEN), University of Hamburg, Hamburg, Germany
e-mail: camilla.sguotti@uni-hamburg.de; xochitl.cormon@uni-hamburg.de

Introduction

Today, living marine resources represent a primary source of proteins for more than 2.6 billion people and support the livelihoods of about 11% of the world's population (UNESCO 2012; FAO 2014). Oceans worldwide concentrate dense and diversified human activities, e.g., fishing, tourism, shipping, offshore energy production, while experiencing a range of environmental pressures, e.g., increase of water temperature, acidification (Halpern et al. 2008; Boyd et al. 2014). Together anthropogenic and environmental pressures may threaten the integrity of marine systems and their sustainable use, altering their different components in many ways. These ecosystem changes may have great impacts for the social-ecological systems they are a part of, particularly when associated with changes in ecological keystone, cultural and/or commercial species (Garibaldi and Turner 2004; Casini et al. 2008a; Möllmann et al. 2008; Llope et al. 2011; Blenckner et al. 2015b).

The World Summit on Sustainable Development in Johannesburg (2002) provided a legally binding framework to implement the Ecosystem Approach to Fisheries Management (EAFM). This holistic approach aims (i) to conserve the structure, diversity and functioning of marine ecosystems and (ii) to provide the economic benefits of a sustainable exploitation of marine ecosystems. Scientific activities supporting approaches such as the EAFM are hence highly encouraged (FAO 2003). However, the insufficient knowledge on the diversity and entanglement of interactions between the ecological system components (deYoung et al. 2008), as well as their vulnerability to increasing anthropogenic and environmental pressures, may hinder successful management.

Even if systems may react to stressors in a non-linear way shifting suddenly to a different state and losing important ecosystem services, management is indeed still more based on continuous dynamics (Scheffer et al. 2001; Sugihara et al. 2012; Glaser et al. 2014; Travis et al. 2014; Levin and

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Möllmann 2015). Some ecosystems may be able to absorb stronger disturbances than others depending on their characteristics, but in general, marine ecosystems are known to be particularly vulnerable to drastic and unexpected shifts, referred in ecology as regime shifts (deYoung et al. 2008). Such non-linear dynamics may have positive or negative outcomes for the sustainable use of natural resources and their management, therefore they should be taken into account and dealt with great precaution when taking environmental policy decisions (Holling 1973; Carpenter 2001; Scheffer 2009; Rocha et al. 2014a).

In this chapter, we first present some examples of marine ecosystems which have exhibited non-linear dynamics in response to external changes. These examples allow us to highlight different mechanisms potentially involved in regime shifts from an empirical point of view, as well as their potential implications for the sustainable use of marine resources. Secondly, we review the regime shift theory and associated concepts to finally consider recent advances and future challenges of integrating regime shift theory into holistic marine ecosystem-based management approaches.

Marine Ecosystems Regime Shifts All Over the World

Although the regime shift concept is still vigorously discussed, an increasing number of studies provide evidence for the potential of abrupt changes and surprises in marine ecosystems worldwide (Steneck et al. 2002; Beaugrand 2004; Mumby et al. 2007; Möllmann et al. 2008, 2009; Mumby 2009; Bestelmeyer et al. 2011; Frank et al. 2011, 2016; Llope et al. 2011; Beaugrand et al. 2015; Gårdmark et al. 2015; Ling et al. 2015; Vasilakopoulos and Marshall 2015; Auber et al. 2015). These studies, based on empirical observations, highlight mechanisms of regime shifts, firstly formulated by theoretical studies (Holling 1973; May 1977; Scheffer et al. 2001).

The Atlantic Cod Trophic Cascade

Surprises in natural systems are relatively common and can happen even in well-studied systems, due to different drivers. One driver of non-linear dynamics is the overfishing of top-predators. Top-predator overfishing may cause the depletion and collapse of their population resulting in unexpected ecosystem structure reorganizations through trophic cascades (Myers and Worm 2005; Fauchald 2010; Llope et al. 2011; Möllmann and Diekmann 2012; Steneck and Wahle 2013). Atlantic cod (*Gadus morhua*) is an important top-predator fish species, which can regulate marine ecosystems through top-down control, and has supported entire human communities through fisheries for centuries (Haedrich and Hamilton 2000; Myers and Worm 2005). After the industrial revolution and the increase of fishing power and capacity around the 1980s–1890s, many cod stocks collapsed bringing high economic losses (Myers et al. 1997; Frank et al. 2016). Multiple analyses conducted in different basins such as in the Baltic Sea or in the Eastern Scotian Shelf, showed that the collapse of cod stocks was caused by a combination of increased fishing pressure and unfavorable climatic conditions (Frank et al. 2005, 2016; Casini et al. 2008b; Möllmann et al. 2008, 2009). The high economic loss and social issues induced, led governments to adopt a range of management measures, such as drastic quota reductions and, in some cases, even fishing moratoria. Nevertheless, despite all the management measures and plans adopted, cod stocks failed to recover (Hutchings 2000; Frank et al. 2011; Hutchings and Rangeley 2011).

One of the reasons advanced to explain these management failures is the undergoing non-linear dynamics known as trophic cascades (Casini et al. 2008a; Star et al. 2011). Indeed, the collapse of this top-predator resulted in a shift from a cod-dominated to a forage fishes-dominated system (Frank et al. 2005; Gårdmark et al. 2015). Before overfishing, adult cod biomass level was high and cod controlled forage fish populations through predation. This hindered the forage fish from negatively impacting younger cod (through predation and/or competition), thus enhancing its biologically sustainable biomass. However, when cod biomass became depleted, the consequently increased forage fish abundance caused a further decline of cod population by increasing their negative direct (predation) or indirect (competition) impacts on younger cod. This feedback loop is then very difficult to reverse (Walters and Kitchell 2001; Möllmann et al. 2009; Nyström et al. 2012). Based on this example, it is clear how such systems can show two distinct configurations depending on their level of top-predator biomass. Of course, changes in mid-trophic levels will also reflect in lower ones, for instance high abundance of forage fishes will likely reduce plankton abundance. Under this new configuration with low cod biomass, a reduction in fishing pressure would likely lead to a very delayed or even none cod recovery, since new mechanisms would keep its population in the new depleted state. To summarize, both Baltic Sea and Scotian Shelf regime shifts were caused by a combination of overfishing and climate variation, and characterized by a trophic cascade (top-down mechanism) due to the depletion of Atlantic cod stocks (Frank et al. 2005; Casini et al. 2008b; Llope et al. 2011; Möllmann and Diekmann 2012). This led to immediate high social and economic losses for cod fishery, followed by a fisheries reorganization in order to adapt to the new ecosystem configuration. Finally, this regime shift led to a considerable increase of fisheries profits due to an outburst of lobster and crustaceans productivity.

The North Sea Regime Shift

The North Sea regime shift involved different mechanisms that induced changes which started at the bottom of the trophic chain and propagated up to higher trophic levels (Reid et al. 2001; Beaugrand 2004; deYoung et al. 2008; Conversi et al. 2010; Lynam et al. 2017). The North Sea regime shift occurred during the 1980s and was mainly induced by a combination of increased sea surface temperatures and changes in hydro-climatic forces (Beaugrand 2004). Due to the increase of sea surface temperature and changes in the water inflows, phytoplankton biomass increased. As a consequence, the zooplankton assemblage, originally dominated by cold waters species, e.g., *Calanus finmarchicus*, shifted to an assemblage dominated by warmer water species, e.g., *Calanus helgolandicus* and gelatinous zooplankton such as jellyfish (Reid et al. 2001; Beaugrand 2004; Möllmann and Diekmann 2012). These changes in the zooplankton community, combined with hydro-climatic changes, propagated to higher trophic levels. Changes in temperature and/or salinity led to an increase of flatfish biomass (Möllmann and Diekmann 2012) while the decline of *C. finmarchicus,* which is the preferred prey of gadoids and especially of cod larvae, led to cod recruitment failures (Beaugrand et al. 2003; Beaugrand 2004) enhancing the negative sea warming effects. These changes in recruitment had a lagged impact on the adult gadoids biomass that, already stressed by overfishing, started to decline inexorably at the end of the 1980s (Hislop 1996). The changes in fish biomass and composition, together with warmer temperatures, favored the emergence of previously scarcely present species such as horse mackerel (*Trachurus trachurus*) and mackerel (*Scomber scombrus*), especially in the northern North Sea (Reid et al. 2001; Beaugrand et al. 2003; Beaugrand 2004).

This regime shift, induced by bottom-up processes, was more qualitative than quantitative in the sense that changes in assemblage and not in total biomass of trophic levels occurred (Beaugrand 2004). The dynamics of these changes highlighted different response time patterns depending on the organisms affected. Indeed, the phytoplankton and zooplankton communities, with their fast life cycles, responded to climatic changes faster than the fish community. Spatial patterns were also different: the coastal areas were less sensitive to change in hydrodynamic conditions, and the regime shift was stronger in the northern North Sea (Reid et al. 2001; Beaugrand 2004; Möllmann and Diekmann 2012). This regime shift completely changed the structure of the North Sea fish community and led to the decline of various commercial species like cod, while the abundance of other species like flatfishes and mackerel increased, consequently having impacts on fisheries (Reid et al. 2001).

Coral Reefs and Kelp Forests Transitions

Other examples of marine regime shifts are coral and kelps transitions (Rocha et al. 2014b). For instance, the Caribbean coral reefs were flourishing ecosystems providing many ecosystem services, sustaining large fish populations and associated human communities. The integrity of the reefs depended on the presence of sea urchins and grazing fishes, which, by eating the algae, maintained the coral reef structure. When the populations of grazing fish started to decrease due to overfishing, nothing seemed to change in the system. Indeed, sea urchins were still able to regulate algae population through predation, preserving the reef structure (Nyström 2006; Standish et al. 2014). However, the ability of the reef to absorb disturbances was already eroded by overfishing, when two concomitant and dramatic events occurred, leading to the total destruction of the reef (Mumby et al. 2007). Sea-urchin populations quickly collapsed due to an illness outbreak, while more nutrients, discarded from the islands, were added to the system, causing rapid eutrophication. In a short time, coral reefs were substituted by algae beds which were not regulated by any top-down (sea urchin predation) or bottom-up (limitation of nutrients) processes. This algae-dominated system is now difficult to reverse due to the feedback mechanisms maintaining the system in its new status (i.e., the number of new algae growing every year can impede the reintroduction of corals, Mumby et al. 2007; Mumby 2009; Kates et al. 2012).

Similarly, kelp forests are highly diverse ecosystems which can maintain flourishing fish populations and offer many services for humans such as fisheries and cultural values (Steneck et al. 2013; Ling et al. 2015). Kelp forests are mainly maintained by fish predation on sea urchins, which controls sea urchin populations. In Australia, overharvesting of predatory fish, coupled with diseases weakening the kelp, led to a boom of the sea urchin population and a shift from high biodiversity kelp forest to poorer urchin's barren (Ling et al. 2015). This state was then difficult to reverse due to various feedback mechanisms such as the increase of juvenile urchin abundance and facilitation of juvenile survival, but also because of the lack of efficient measures to recover the stocks of the sea urchin's predators (Ling et al. 2015). In these two examples, the regime shifts were caused by multiple stressors which altered the regulation (top-down and/or bottom-up) of previously highly productive ecosystems and led to huge economic, social and ecological losses. Similarly to the Atlantic cod example, management measures failed to reverse these unexpected regime shifts due to feedback loop mechanisms (Steneck et al. 2002; Ling et al. 2015).

From Examples to Theory

From these four examples, several conclusions can be drawn. Stressors potentially inducing regime shifts may affect a system gradually, e.g., decline of top-predator due to fishing (Baltic Sea and Scotian Shelf regime shifts), or abrupt and exceptionally, e.g., disease outbreak (Caribbean coral reef destruction). The examples of the Atlantic cod stock collapse and the North Sea regime shift showed that climate change may play and important role in such mechanisms (Beaugrand 2004; Conversi et al. 2015; Yletyinen et al. 2016). In addition, these examples showed the cumulative effects of different stressors and how they may act together in synergistic ways. The mechanisms and processes involved in regime shifts may be induced by top-down and/or bottom-up regulation (Holling 1973; Beisner et al. 2003; Conversi et al. 2015; Pershing et al. 2015). Finally, these examples highlight the importance and necessity to understand regime shifts mechanisms for a sustainable use of marine resources in order to provide ecosystem services and benefits for human communities (Doak et al. 2008). Also, they uncovered some fundamental properties of regime shifts, e.g., the abruptness of changes and their lack or low reversibility (Scheffer et al. 2001, 2015; Dakos et al. 2012). However, due to the complexity and entanglement of the mechanisms involved, defining regime shifts based on empirical evidences is challenging. A review of the concepts associated with regime shifts, which are mostly theoretical (Levin and Möllmann 2015), is essential to understand the non-linear mechanisms potentially involved in complex systems dynamics, particularly in a time of pronounced environmental changes.

The Regime Shift Theory

Different mathematical frameworks lead to the development of the regime shift theory (Jones 1975, 1977; Thom 1975; Crawford 1991), describing how changes in some controlling factors can lead to huge and abrupt changes in various systems (e.g., biological, physical, behavioral; Jones 1975; Carpenter 2001; Scheffer et al. 2001). Marine regime shifts can be defined as dramatic and abrupt changes in the system structure and function that are persistent in time, where the system can range from a single cell to a population or an ecosystem (Beisner et al. 2003; Scheffer and Carpenter 2003). Due to the high number of terminologies and definitions used in the literature, a glossary was added to this chapter in order to have consistent and clear definitions. All terms highlighted in italics in the following text can be found in the glossary section (Box 1).

Box 1: Glossary

Regime shift:	dramatic and abrupt change in the structure and function of a system causing a shift between two alternate stable states following discontinuous non-linear dynamics and exhibiting three equilibria. There are some debates about the definition and *critical transition* or *phase shift* might be considered synonyms depending on the literature.
Resilience:	capacity of the system to absorb disturbances and reorganize in a way that it retains the same functions, structure, identity and feedback mechanisms, potentially impeding a regime shift.
Regime:	dynamic system configuration maintaining certain structures and functions. It is also known as *stable state, basin of attraction* or *domain of attraction*.
Tipping point:	threshold separating two dynamics regimes. It is also known as *critical threshold* or *bifurcation point.*
Feedback mechanism:	ecological mechanisms stabilizing a regime by amplifying (positive) or damping (negative) the response to a forcing. Positive feedbacks (reinforcing) move the system to an alternate stable state, out of equilibrium. Negative feedbacks (balancing) maintain the status of the system, close to the equilibrium dynamics.
Hysteresis:	phenomenon for which the return path from regime B to regime A, is drastically different from the path that led from regime A to regime B.

The easiest way to understand and visualize *regime shifts* is the example of the ball-in-cup or ball-in-valley diagram developed from the pioneer work of Poincare in the 1800’s in Crawford 1991; Fig. 1). The ball represents the study system, for instance the Caribbean coral reef. The system reef (our ball) has certain parameters such as coral abundance, coverage, and biodiversity. The system state is represented by the valley in which our ball (system) lies (regime 1 in Fig. 1). The dimension of the valley (width and height in our two dimensions’ figure) corresponds to the

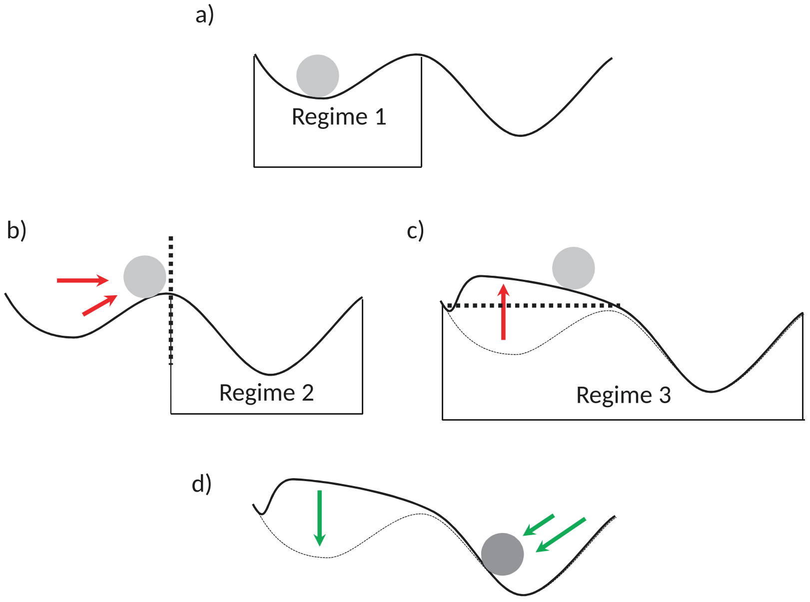

Fig. 1 Regime shift theory represented by ball-in-cup diagrams (Crawford 1991). The ball represents the system and the cups (or valleys) the system states (see text for more information). The thick dotted lines represent the tipping points. The arrows represent disturbances, red for disturbances inducing a shift and green for reversed disturbances having no effects. (**a**) System in its original state. (**b**) Regime shift induced by changes in system state variables. (**c**) Regime shift induced by change in system parameter variables. (**d**) System in its new state showing hysteresis. Referring to our Caribbean example (section "Coral reefs and kelp forests transitions") the light grey ball represents coral reef dominated system while the dark grey ball, the algae dominated system

resilience of the system state. For instance, even when the Caribbean coral reef system was stressed by intensive fishing on grazing fishes, the system maintained its original state and did not shift because its resilience was high (i.e., the sea urchins were able to maintain top-down regulation on algae, Mumby et al. 2007). Indeed, when the valley is large and deep, the ball/system remains in it, maintaining its structure, despite the disturbances. Repetitive disturbances such as overfishing and eutrophication did, however, reduce the system resilience (the valley became narrower and shallower) and when a strong disturbance occurred (here a disease outbreak), the system shifted abruptly to a new state (i.e., algae beds). This new state is now resilient, maintained by new *feedback mechanisms* that help its stabilization, e.g., the higher survival of algae and the non-recovery of grazer fishes (Beisner et al. 2003; Roe 2009; Conversi et al. 2015). Resilience is defined as the capacity of the system to absorb disturbances and reorganize, so as to still retain essentially the same functions, structure, identity and feedback mechanisms (Holling 1973; Beisner et al. 2003; Vasilakopoulos and Marshall 2015; Folke 2016).

Some perturbations may act either on the system state variables (pushing our ball from its valley into a new one, e.g., disease outbreak, Fig. 1b) or on the system parameter variables (modifying the shape of the valley, hence affecting system resilience, e.g., overfishing and eutrophication, Fig. 1c; Beisner et al. 2003). As highlighted by the Caribbean coral reefs example, it is the combination of multiple mechanisms that generally causes a system to shift from a stable state to another (Biggs et al. 2012). This shift of a system between two alternate stable states is the foundation of regime shift theory (Carpenter 2001; Scheffer et al. 2001). The separation point between two *regimes* (or alternate stable states) is the so-called *tipping point* (Selkoe et al. 2015). Once crossed, the system will shift to a new regime with new characterizing parameters. Clearly, once a tipping point is crossed, it is not easy to push the ball back in its original valley, since the new valley is deep and large, thus highly resilient, and/or the original valley might have disappeared. This can hinder a return of the system to the previous state even when disturbances stop (e.g., fishing ban, end of disease outbreak) or are reversed (Figs. 1d and 2; Beisner et al. 2003). This property of regime shifts is called *hysteresis* and can be defined as the phenomenon for which the return path of a system from the altered to the original state can be drastically different from the one which have led to this altered state (Beisner et al. 2003; Bestelmeyer et al. 2011). Hysteresis is a typical feature of discontinuous regime shifts and can be detected when the relationship between the stressors and the system differs depending on the regime (stable state) of the system (Scheffer and Carpenter 2003; Bestelmeyer et al. 2011).

Another way to visualize the regime shift is the fold bifurcation curve (Fig. 2; Scheffer et al. 2001). The system reacts in a smooth way to condition changes until a tipping point

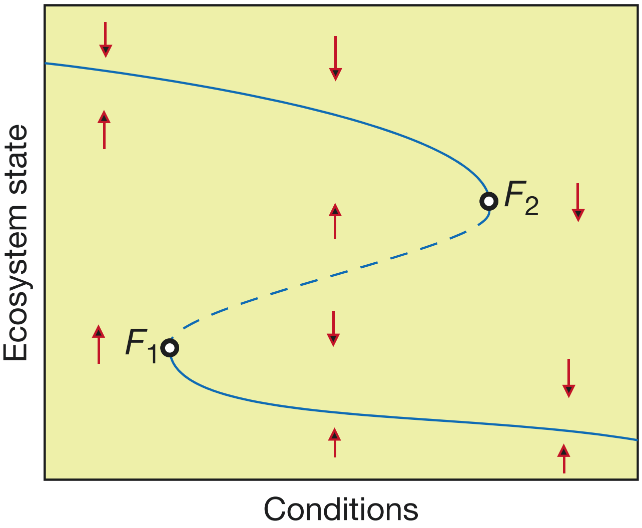

Fig. 2 Fold bifurcation curve (Reproduced from Scheffer et al. 2001). The dashed line represents the unstable equilibria and the border between the two alternate stable states represented by plain lines. F1 and F2 represent the tipping points

(F1 or F2) is reached and the system jumps from one state to another. In the area of discontinuity (Fig. 2, dashed blue line) the system can present three equilibria. As evidenced by this visualization, systems that show such behavior are difficult to reverse to previous state even when condition changes are reversed (hysteresis). Although some debates exist regarding the definition of regime shift we adopted the definition of Scheffer et al. (2001) and Selkoe et al. (2015) of an abrupt change over time with discontinuous dynamics exhibiting hysteresis. This is opposed to phase shifts *sensu* Selkoe et al. (2015), where system state's response to condition change is continuous, e.g., a logistic response, with two states but only one equilibrium.

Resilience, feedback mechanisms, tipping points and hysteresis are important attributes of regime shifts (van der Maas et al. 2003; Bestelmeyer et al. 2011). These properties make regime shifts extremely important in the real world and have profound implications for management (Travis et al. 2014; Selkoe et al. 2015; Angeler et al. 2016). Imagine having as system a fish population. When you start fishing, the population still manage to absorb the perturbation and might decline, but would remain in a state with high biomass, high recruitment, a certain growth rate, etc. At some point the fishing pressure, usually combined with other external stressors, increases so much that the population collapses and its internal mechanisms change. The exploited population is now in a new state at low abundance, possibly with different growth and mortality rates. Now suppose that we are the managers. We could assume that reducing the fishing pressure to pre-collapse levels would make the population quickly rebound. This could work in a context of linear dynamics but if the population has crossed a tipping point and it is now in a new alternate stable state, controlled by new mechanisms that cause hysteresis, recovery of the system may be slow and difficult, or even impossible. From this example, the importance of regime shift appears clear. In order to apply efficient and useful management measures, we should aim to detect regime shifts in advance or, at least, we should consider the possibility that an exploited system can show non-linear behaviors, and apply precautionary management approaches (Holling 1973; Carpenter 2001; Scheffer and Carpenter 2003; deYoung et al. 2008; Dakos et al. 2012; Punt et al. 2012; Levin and Möllmann 2015). Many marine ecosystems have undergone drastic shifts often resulting in new ecologically structured and/or economically less valuable states (Conversi et al. 2015; Möllmann et al. 2015). These regimes shifts have brought catastrophic ecological and social consequences (Rocha et al. 2015), such as economic losses, social issues and losses of ecosystem services (Casini et al. 2008a; Möllmann et al. 2008; Blenckner et al. 2015b). Thus, since several processes at several levels of the ecosystem are often involved, it appears evident from these examples that an ecosystem approach to management of marine ecosystems prone to regime shifts is essential (Long et al. 2015).

Challenges and Implications of Regime Shifts for Management Purpose

To include the concept of regime shift into management perspectives, multiple *a priori* steps have to be made to first identify the mechanisms and the drivers involved (feedback loops, interactions, etc.), and then integrate this information into suitable and adapted policy. The documentation of a broad range of regime shift examples, involving different mechanisms applied to different ecosystems may be very useful to compare the various processes involved, to understand potential implications in a better way (Rocha et al. 2015) and therefore to adapt management to local characteristics (deYoung et al. 2008). In this context, the Regime Shift Database (Rocha et al. 2014b), based on a participatory approach, aims to review regime shifts of social-ecological systems worldwide with a particular focus on regime shifts having a potential large impact on human well-being and ecosystem services. This database, available online (www.regimeshifts.org), is an initiative led by the Stockholm Resilience Centre to increase general knowledge and understanding of regime shifts and associated concepts and to help managers and policy makers to take these concepts into account in their future decisions.

Knowledge of different mechanisms and local characteristics of regime shifts may facilitate their detection. Indeed, the first step and challenge to consider regime shifts in management, is to actually detect them (Carpenter 2001; deYoung et al. 2008; Rocha et al. 2015). For instance, regime shifts in the North Sea and English Channel communities were only detected 10 years after they occurred (Beaugrand 2004; Auber et al. 2015). This late detection may partly be explained by the very large scale at which these shifts occurred and highlights the need of studying different spatial scales when wanting to understand ecosystems processes and dynamics. Similarly, temporal scales of changes might be different depending on the lifespan of the affected organisms and might lead to temporal lags in system responses to stressors (Holling 1973; deYoung et al. 2008) as it was the case in the North Sea. These differences in spatio-temporal patterns need to be addressed and disentangled as they might hinder or delay regime shift detection and exacerbate social and economic consequences (Levin 1992; Scheffer and Carpenter 2003; Kerkhoff and Enquist 2007; Levin and Möllmann 2015). It might also be necessary to disentangle regime shifts (*sensu* Selkoe et al. 2015) from simple logistic dynamics and highlight hysteresis (which requires additional observations in time). For these reasons, regime shift detection requires long and extensive observation datasets of the system which is generally costly in time and money (Carpenter 2001; Scheffer et al. 2009; Levin and Möllmann 2015). Moreover, the required time to obtain time series of suitable length might prove too long, particularly when such shifts strongly impact ecosystems services and human well-being. For these reasons, experimental studies are necessary to enhance the understanding of systems responses to disturbances (Angeler et al. 2016). Particularly, experiments may help to understand multi-causality and dual relationships between stressors and systems which generally participate in hindering detection of regimes shifts (Scheffer and Carpenter 2003; Conversi et al. 2015; Levin and Möllmann 2015).

While regime shifts detection may be delayed, their unexpected and abrupt behavior hinders regime shift prediction, which is necessary to ensure effective management measures. In addition, a post-regime shift detection may result in increased management challenges, particularly due to hysteresis, as described in the previous section for coral reefs (Mumby et al. 2007; Mumby 2009), kelp forests (Steneck et al. 2002) and various fish stock shifts (Myers et al. 1997; Hutchings 2000; Myers and Worm 2005; Hutchings and Rangeley 2011). Challenges in prediction may be partly related to the common use of linear relationships to statistically describe natural processes which need to be overcome in favor of more realistic (thus more complex) models (Holling 1973; Ludwig et al. 1997; Scheffer and Carpenter 2003). Indeed, the non-linear relationships between stressors and system variables need to be understood to be able to correctly predict system responses. Also, a new branch of science has been currently developing regime shift indicators, the so-called early-warning signals, to anticipate regimes shifts. These signals are generally based on the fact that the recovery of a highly disturbed system to an equilibrium is slow, i.e., critically slowing down (Scheffer et al. 2001, 2015; Dakos et al. 2012; Lindegren et al. 2012). Indeed, when systems are close to tipping points, their stability decreases, generally leading to an increased variability, and autocorrelation of the data describing them. These indicators work well with simulation models but still they have some limitations in predicting shifts using empirical data (Dakos et al. 2008, 2017; Scheffer et al. 2009; Dai et al. 2013). They may be constrained by the length of the times series available and/or the limited amount of data, by methodological assumptions and/or sampling errors (deYoung et al. 2008; Lindegren et al. 2012). Moreover, they are not suitable to predict stochastically driven shifts. To overcome these limitations Lade and Gross (2012) developed a new approach to detect early warning signals with reduced time-series. Lindegren et al. (2012) recommended a multiple approach based on knowledge of the system and its local characteristics (key ecological thresholds, relationships with drivers), data availability, sensitivity and bias of the analysis carried out. Such advances need to be followed by the scientific community to develop more approaches overcoming these limitations. Alternative sources of data, e.g., public records and narratives, must be found and used, particularly when ecological data are not available, and systems must be monitored at an appropriate time scale to ensure shift detection as early as possible.

Because prediction of regime shifts is so challenging, and because the potential consequences for ecosystem services and human well-being may be abrupt and very difficult (or even impossible) to reverse, precautionary approaches are recommended (Holling 1973; Carpenter 2001; Scheffer and Carpenter 2003; Selkoe et al. 2015). When managing systems prone to regime shifts, risks and uncertainties must be assessed before any management action is taken (Levin and Möllmann 2015; DePiper et al. 2017). Risk assessment requires a clear definition of the system of interest, its potential tipping points, as well as suitable indicators. However, all the challenges already mentioned (multiple-causality, dual relationships to drivers, spatio-temporal different patterns, limitation of data, etc.) may impede the definition of appropriate indicators (Kelly et al. 2015; Selkoe et al. 2015). For instance, Vasilakopoulos and Marshall (2015) showed that the spawning stock biomass (SSB) of Barents Sea cod did not suffice to detect a regime shift of this population, while SSB levels are generally the reference points used in current fishery management plans (single- or multi-species advices), and sometimes the only ones. These results evidence the need to base scientific advice to fishery managers on the monitoring of several ecosystem (community/population) parameters, particularly when suspecting potential impending shifts. Similarly, stressors effects may be unclear when studied individually, while their importance may appear only when combined with other stressors (Rocha et al. 2015; Vasilakopoulos and Marshall 2015). The factors undermining resilience (eutrophication, global warming, species invasion, etc.) should be of prior concern as small variations in stressors might lead to large changes in ecosystem structure and/or functioning when resilience is eroded (Ricker 1963; Ludwig et al. 1997; Scheffer et al. 2001; Beisner et al. 2003; Scheffer and Carpenter 2003). The quantitative assessment of risk and associated resilience is difficult and challenging. Economic cost-benefit analysis might be useful when trying to quantify risks for ecosystem services (Carpenter 2001), however, it might totally underestimate them when too narrow-focused, e.g., focusing on yield in fisheries while neglecting age-structure of the stock (deYoung et al. 2008). Quantitative assessment of resilience may prove very useful but requires an extensive amount of data particularly in complex systems (Vasilakopoulos and Marshall 2015). Therefore, qualitative analysis and/or conceptual models may be preferred (DePiper et al. 2017), particularly when studying data-poor systems or when dealing with complex adaptive systems such as social-ecological ones.

Despite the increasing effort in scientific research, even when risk (or resilience) may be assessed, ecological uncertainties (about system evolution) and livelihood uncertainties (about impacts on human communities) related to regime shifts are high (Pindyck 2000). When managing social-ecological systems (SES) prone to regime shifts, policy makers must face these uncertainties and different management strategies might emerge: reducing or limiting system stressors (mitigation), building up system resilience (adaptation) and/or reversing a shift (restoration, Kates et al. 2012; Angeler et al. 2013). These strategies might have different outcomes, benefits, costs and efficiency depending of goals and focus of management as well as the status of the system (Selkoe et al. 2015; Lade et al. 2015; Fenichel and Horan 2016; Mathias et al. 2017). For example, because of hysteresis, building up resilience might be more effective and less costly than restoration measures (Selkoe et al. 2015). These measures might also require different levels of governance. For instance, the reduction of tuna fishing effort in the Pacific Ocean would require an international consortium for management to be efficient while similar measures applied to a coral reef fishery would be relevant at the local management scale. In addition, when mitigation generally requires international and global management (e.g., gas emissions reduction), building up systems resilience (adaptation) may succeed at local scales, countering global inaction (Rocha et al. 2015). While decreasing variance of a system may seem a good idea, Carpenter et al. (2015) highlighted the adverse effects for system resilience management. Staying within a safe-operating space (Rockström et al. 2009), including uncertainties around tipping points and using history as guideline (Fenichel and Horan 2016; Liski and Salanié 2016) might, however, prove effective and reduce risks of management failures. Adversely, managers might need to erode resilience of a system to tip it towards a preferable regime, i.e., more pristine or more valuable (Derissen et al. 2011). This so-called transformation would require intentional changes in the institutional framework in which the utilization of marine systems (e.g., including switch to a novel management system), as well as a transparent and equitable redistribution of benefits across stakeholders takes place (Selkoe et al. 2015). Uncertainties may as well increase immediate costs, and even if costs of inaction would be high in the future, they might hinder immediate decisions (Pindyck 2000; Selkoe et al. 2015).

Adaptive co-management might be ideal when cooperation between local and global stakeholders is possible (Plummer et al. 2017). However, it might slow down management processes opposed to the potential flexibility and responsiveness of local stakeholders required for a good management of regime shift effects (deYoung et al. 2008; Horan et al. 2011; Blenckner et al. 2015a; Rocha et al. 2015; Valman et al. 2016). Similarly, polycentric governance holds great potential at the international scale but is vulnerable to negative interactions between institutions and weak coordination (Galaz et al. 2012; Mathias et al. 2017). In both cases, the question of responsibility might be raised in case of management failures (Baumgärtner et al. 2006; Fenichel and Horan 2016). Local and/or global stakeholder cooperation, as well as responsiveness, may be improved by the knowledge of the stressors involved in regime shifts mechanisms, their shared interactions with the different components of the system, and the different scales at which they interact (Rocha et al. 2015). Such knowledge may also help policy makers to set suitable management targets otherwise challenged when uncertainties are high.

Finally, the integration of management and regime shift theory may prove quite complicated. The complex responses to stressors, the multiple, cross-disciplinary interactions between each system components, the high uncertainties and the different stakeholder perspectives and conflicts need to be understood and accounted for when considering regime shifts (and/or resilience) in social-ecological systems (SES) management decisions. This requires holistic and integrative approaches such as integrative ecosystem assessment (IEA, (Levin and Möllmann 2015). In this context, scientists have recently developed frameworks to conceptualize SES and assess their sustainability and uncertainties (Ostrom 2009; Leslie et al. 2015; Levin et al. 2016). Particularly, these frameworks allow the combination of classic scientific information and local stakeholders' ecological, cultural and/or social knowledge of the system. These conceptual models may be used to promote interdisciplinary research, discussions between stakeholders, and allow a holistic management strategy evaluation after their operationalization (Levin and Möllmann 2015; Levin et al. 2016; DePiper et al. 2017).

Conclusions

Regime shifts are abrupt changes that can happen in complex systems worldwide at different temporal and spatial scales, depending on the resilience of the systems (Scheffer et al. 2001; deYoung et al. 2008). It is extremely important to study and understand these mechanisms since many regime shifts have led to catastrophic changes including ecological, social and economic losses worldwide (Mumby 2009; Steneck and Wahle 2013; Blenckner et al. 2015b). Despite the fact that many studies and methods have focused on the detection of regime shifts, there is still a lot to be done to achieve marine ecosystem management integrating resilience and regime shifts (Travis et al. 2014; Selkoe et al. 2015; Angeler et al. 2016). New tools, such as early warning signals or new ways to assess the resilience of different systems, combined with an in-depth study of the mechanisms and stressors affecting natural systems are a good start to incorporate resilience and regime shift into policy-maker decisions (Carpenter and Brock 2006; Scheffer et al. 2009; Dakos et al. 2012, 2017; Ling et al. 2015; Vasilakopoulos and Marshall 2015). Since regime shifts often affect many components of an ecosystem in different ways, ecosystem-based management (EBM) is necessary to include effectively regime shifts into management considerations (Blenckner et al. 2015a; Levin and Möllmann 2015; Long et al. 2015; Rocha et al. 2015). To make this holistic approach effective and to preserve the natural environment and ecosystems in a more integrative way, there is a real need to translate regime shift and resilience concepts from theory to applications (Punt et al. 2012; Travis et al. 2014; Selkoe et al. 2015). Recently, the operationalization of social-ecological systems (SES) conceptual models have shown promising improvements in this sense (Leslie et al. 2015; DePiper et al. 2017). Due to the different spatial and temporal scales at which regime shifts can act, i.e., from extremely local to global, and the high degree of associated uncertainties, innovative and flexible management options need to be developed at different levels of governance. For instance, Rockström et al. (2009) suggested a management at the planetary boundaries. Such management would require, in addition to adaptive management and polycentric governance, a societal shift in order to achieve a fair use of global resources, and a transformed economy (Hughes et al. 2013; Lade et al. 2013; O'Brien et al. 2014). Finally, we can expect that the increasing awareness of the implications of regime shifts and associated concepts for human well-being worldwide will likely lead to more precautionary management approaches, while new tools and technics will be developed to achieve an integrative and efficient management of our natural resources.

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Appendix

This article is related to the YOUMARES 8 conference session no. 5: "Ecosystems Dynamics in a Changing World: Regime Shifts and Resilience in Marine Communities". The original Call for Abstracts the abstracts of the presentations, and the report of the session can be found in the appendix "Conference Sessions and Abstracts", chapter "10 Ecosystems Dynamics in a Changing World: Regime Shifts and Resilience in Marine Communities".

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