Regime Shifts in Coastal Marine Ecosystems: Theory, Methods and Management Perspectives

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Abstract
Regime shifts are large-scale and often abrupt ecosystem reorganizations, caused mostly by the impact of multiple stressors. These shifts are characterized by changing ecosystem state, with the new state being persistent and thus difficult to reverse. Regimes shifts have repercussions on the full socio-ecological system with important implications for management. Coastal marine ecosystems are especially prone to such dynamics since they are impacted strongly by anthropogenic and natural drivers. Here, we describe and discuss regime shifts in marine coastal ecosystems. We firstly review the theoretical framework of regime shifts. Then, we show examples of regime shifts based on three management pillars: prevention, restoration, or adaptation. Furthermore, we highlight emergent issues and challenges for management implementation, and we provide a structured multi-step approach for detecting regime shifts. With global climate change, incorporating regime shifts in management should be a priority in order to maintain marine ecosystems and their services.

Glossary
Feedback mechanisms ecological mechanisms stabilizing a regime by amplifying or dampening the response to a force. Positive feedbacks (reinforcing) move the system to an alternate state, while negative feedbacks maintain the status of the system.
Hysteresis the phenomenon for which the return path from B to A is different from the path that led the system from the regime A to B.
Resilience the ability of a system to absorb disturbances and still maintain the same structure and functions. It can be divided in ecological and engineering resilience.
Regime dynamic system configuration being characterized by certain structures and functions. Also known as attractor, stable state, basin of attraction.
Regime shift dramatic and abrupt change in the structure and function of a system causing a shift between two alternative stable states. It follows non-linear discontinuous dynamics.
Tipping points driver threshold separating two dynamic ecosystem states regimes. It is also called as critical threshold or bifurcation point.

Key Points

- Regime shifts are common in marine coastal ecosystems.
- Regime shifts are often abrupt, can range from specific local ecosystem changes to basin-wide ecosystem changes and are difficult to reverse.

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- Regime shifts are often caused by multiple drivers, and may become more frequent under global changes, and can threaten the provision of ecosystem services.
- Theoretical methodological approaches to detect regime shifts are available, however, their applications in real ecosystems and thus management is still limited.
- Approaches for dealing with regime shifts can be summarized in three pillars: prevention, restoration and adaptation.
- Management needs to increase efforts for the integration of regime shift dynamics in order to ensure a sustainable use of marine resources and maintain vital ecosystems.

Introduction

It has long been evident that global oceans are negatively affected by multiple human impacts (Halpern *et al.*, 2008). Coastal and estuarine systems are particularly exposed to severe anthropogenic impacts, due to their importance for human users and proximity to human activities. Coastal ecosystems such as some coral reefs, seagrass beds or mangroves provide key ecosystem services, ranging from recreational use and food provisioning to coastline protection and carbon sequestration (Barbier *et al.*, 2011). Around 44% of the human population live in an area of 150 km to the coast, where they have access to these ecosystem services. (United Nations Atlas of the Ocean, 2002) However, humanity has left a long-lasting footprint on these coastal regions. Marine regions in densely populated areas show worse ecosystem state than regions in uninhabited areas (Halpern *et al.*, 2015). The anthropogenic threats with the largest global impacts include fishing, pollution, habitat destruction, introduction of alien species and climate change (Jackson *et al.*, 2001; Millenium Ecosystem, 2005; IPBES, 2019). These stressors heavily impact the ecosystems or the natural populations and can be divided into low intensity, continuous impacts or abrupt and strong impacts. The way these stressors can impact ecosystems can be linear, non-linear or can lead to a complete reorganization of the system impossible or very difficult to reverse (Scheffer *et al.*, 2001; Rocha *et al.*, 2015a; Conversi *et al.*, 2015). The latter dynamic is what is in the literature referred to as *regime shift* and is the main topic of this chapter.

Regime shifts are increasingly documented in marine ecosystems all over the world, and in most cases attributed to multiple, often interacting human pressures. Among these human pressures, coastal systems are negatively impacted by anthropogenic climate change (IPCC, 2021). The impacts of climate change range from affecting sea temperatures, pH, salinity and sea level (Wong *et al.*, 2014; He and Silliman, 2019). Climate change is impacting the entire planet through different mechanisms and its impact is projected to increase through time if strong mitigation measures are not applied. Climate change might also have an impact on the frequency and intensity of extreme events, such as storms, heat waves, floods and droughts (IPCC, 2021). All these events can lead to major impacts on costal ecosystems (He and Silliman, 2019).

While climate change is a more recent, but a growing threat, fishing activities have been modifying marine and coastal ecosystems for centuries (Jackson *et al.*, 2001). Fishing pressure has been and is still particularly high in the coastal areas (Stewart *et al.*, 2010). Coastal habitats are often key breeding and nursery grounds for harvested fish species, moreover they are easily accessible and thus can be harvested extensively (Jackson *et al.* 2001). Unregulated fishing activities led to overfishing of the resources, resulting in a threat for food provisioning and for the socio-ecological systems (Srinivasan *et al.*, 2010). In some cases, fishing, together with other pressures has induced regime shifts in fish populations and entire ecosystems (Auber *et al.*, 2015). Nowadays, management measures in place are often able to mitigate fishing pressure and favor a sustainable exploitation of marine resources (Hilborn *et al.*, 2020).

Pollution is another potentially important driver of regime shifts. Pollution encompass a wide range of impacts. Pollution can range from land-based plastic pollution (Jambeck *et al.*, 2015), to light and noise pollution of artificial structures (Heery *et al.*, 2017) to chemical pollution through for example the increased input of nitrogen and phosphorus into coastal areas from agricultural run-off. The excess of nitrogen and phosphorus can lead to increased growth of phytoplankton biomass that can have several negative effects for ecosystems. These impacts can include reduced light penetrations, which in turn influences productivity. It can also lead to higher grazing activities, causing high oxygen consumption and, together with biomass decomposition, can favor oxygen depletion (Rabalais *et al.*, 2009). Apart from these prominent drivers, others, such as habitat loss or species invasions, can affect the marine environment.

Anthropogenic drivers often impact the ecosystems in a cumulative way. Impacts can show additive cumulative effects, inducing linear ecosystem changes, or synergistic (positive or negative) effects (Côté *et al.*, 2016) which can induce non-linear ecosystem changes with sudden and possibly irreversible reorganizations of the system (i.e., regime shift) (e.g., (Kirby *et al.*, 2009; Kotta *et al.*, 2018; Ling *et al.*, 2018). These changes, but particularly regime shifts, can have detrimental effects on human health and well-being due to the loss of certain ecosystem services (Cooley *et al.*, 2022). All these varying interactions and unpredictable impacts can make the conservation and management efforts challenging (Brown *et al.*, 2013).

Since regime shifts are, by definition, sudden and difficult to reverse, specific management measures might have to be adopted. The main approaches that can be used to manage regime shifts can be synthetized into three pillars: 1) prevention and early detection, 2) reversal, and 3) adaptation. In the following sections, we will firstly explain the theory behind the science of regime shifts. Next, we will use examples to highlight the possible management responses, structured along these three pillars (prevention and early detection, reversal, adaptation). Further, we highlight the problems or emerging issues of regime shifts and we present a structured, step-wise methodological framework to quantitatively detect them, which may help to uniform the identification and respond to the technical limitations encountered in the literature. Finally, we indicate potential implications for the management of systems prone to regime shifts.

Theory

Since Roman times nature has been described as something that works under fixed, absolute rules. Over the last few centuries, mathematicians and physicists have tried to describe nature and how it works, using the rules and models typical of the mathematical language. However, they realized that natural phenomena often present some erratic behavior that does not fit to the rules and the linear models they were proposing (May, 1972; Holling, 1973; May and Oster, 1976). In the early days of the discoveries, these unpredictable behaviors were considered exceptions from the rules and simple, linear mathematical models were still mainly used to describe the world. However, in the 20th century, it became clear that these non-linear phenomena, visible e.g., in the climatic system, the ecological systems, and oceanography, constituted the rule rather than the exception, and required different descriptive approaches (Lorenz, 1963; Mandelbrot, 1967; May, 1977). The scientist Poincaré, studying shapes and topology, was the first to investigate these types of dynamics, and coined the "bifurcation theory", or the study of changes in a dynamical system (Poincaré, 1890; Crawford, 1991). These changes occur when a smooth and small change in one or more parameters lead to an abrupt and sudden change of the system (i.e., bifurcation). This theory was at first strongly qualitative and was expanded by two other theories in the 1960s: the chaos theory from the work of Lorentz and the catastrophe theory from the models of Thom (Lorenz, 1963; Packard *et al*., 1980; Sugihara, 1994). Chaos theory started to highlight how several natural processes presented chaotic dynamics, meaning that depending on the initial conditions that characterize a system, the system can move randomly between different equilibria. Thus, two identical systems with minimally different initial conditions can develop into very different systems (May and Oster, 1976; May, 1977). Catastrophe theory, instead, attempted through the use of mathematical models, to describe catastrophic bifurcations of systems influenced by a number of external parameters (Thom, 1975; Scheffer, 2009; Petraitis and Dudgeon, 2016). The synthesis of these two theories led in the 1970s to the formulation of the concept of *resilience* defined as the ability of a system to absorb disturbances while maintaining its structure and function, thus remaining in the same stable state (Holling, 1973; Beisner *et al*., 2003; Walker *et al*., 2004).

Resilience can be divided into two components. The first is called *engineering resilience* and describes the capacity of a system to remain close to the steady *stable state* and how fast the system returns to quasi-equilibrium after a disturbance (Carpenter, 2001; van Nes and Scheffer, 2007). The other one is called *ecological resilience* and is the magnitude of disturbance that can be absorbed without changing state, thus without changing functions and structures (Beisner *et al*., 2003; Walker *et al*., 2004). Indeed, disturbances can erode resilience and induce a system to jump suddenly from one equilibrium to another, and these dynamics are described by both chaos and catastrophe theory (May, 1977; Scheffer *et al*., 2001).

Catastrophe theory was formulated by the mathematician Thom and was one of the first attempts to render the bifurcation theory quantitative and applicable to data (Thom, 1975; Jones, 1977; Crawford, 1991). At first, this theory was criticized for being excessively narrow, unable to describe a variety of systems, and based on a purely deterministic framework (Kolata, 1977; Cobb and Watson, 1980; Loehle, 1989; Barkley Rosser Jr, 2007). However, successive development of a stochastic framework favored the utilization of the theory across disciplines such as economics, physics, behavioral sciences and, only marginally, ecology (van der Maas *et al*., 2003; Chen *et al*., 2007; Cunningham and Kwakkel, 2014; Petraitis and Dudgeon, 2015; Diks and Wang, 2016). Catastrophe theory studies systems undergoing abrupt transitions between multiple stable states or equilibria and describes them through differential equations. It describes seven canonical forms of transitions depending on the number of state variables and explanatory variables or parameters (van der Maas and Molenaar, 1996; van der Maas *et al*., 2003; Petraitis and Dudgeon, 2016). Of these seven forms, the "fold" and the "cusp" transitions are the most commonly used. The fold transition describes the abrupt shift of a system depending on one driver, and is the typical example that has been used in the regime shift theory (Scheffer *et al*., 2001; Scheffer, 2009) . The cusp transition, instead, describes the dynamics of a system depending on two interacting drivers from linear continuous dynamics to non-linear discontinuous dynamics characterized by three equilibria (Grasman *et al*., 2009; Petraitis and Dudgeon, 2015; Diks and Wang, 2016; Sguotti *et al*., 2019). To understand if systems behave following a catastrophic behavior, seven indicators (termed flags) of catastrophe have been identified: multimodality, inaccessibility, sudden jump, hysteresis, divergence, variance, and critical slowing down (van der Maas *et al*., 2003; Petraitis and Dudgeon, 2016). These flags have been later incorporated into the regime shift theory and have been used to describe systems undergoing regime shifts. After a few applications to ecological systems, catastrophe theory was "forgotten" and was instead replaced by the regime shift theory (Scheffer *et al*., 2001; Hughes *et al*., 2013a; Conversi *et al*., 2015).

The regime shift theory focuses on the simplest form of catastrophe, the fold (Scheffer *et al*., 2001; Beisner *et al*., 2003; Bestelmeyer *et al*., 2011). Regime shifts are described "as abrupt and dramatic changes in the structure and function of a system that can switch between two alternative stable states" (Scheffer *et al*., 2001; Rocha *et al*., 2015b; Sguotti and Cormon, 2018). However, a system can react to the effect of a driver in three ways (see **Fig. 1**): (1) As the driver increases, the system changes linearly. This means that for each level of the driver, just one level of the state variable exists and thus by reverting the driver, the system will go back to its previous state (**Fig. 1**(a)). (2) As the driver increases the system changes in a non-linear way showing a sudden jump. In this case, for every level of the driver only one level of the system exists and thus when the driver is reverted the

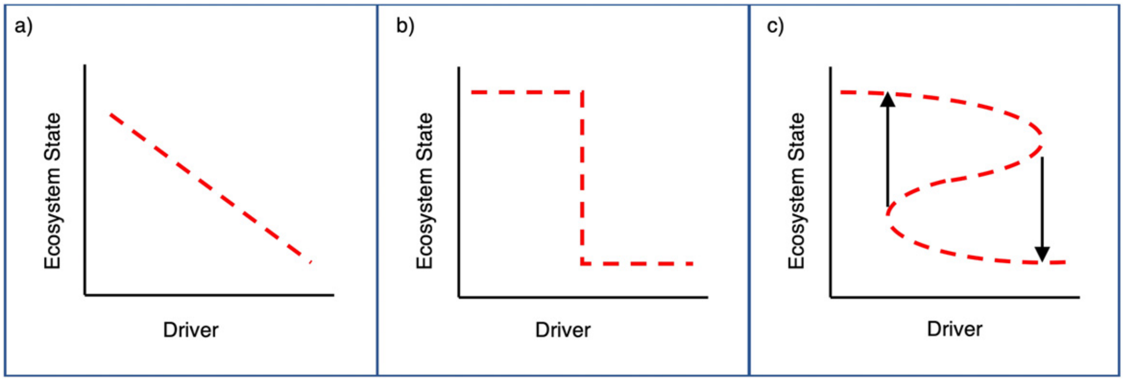

Fig. 1 The drivers state relationship. a) a linear and continuous relationship and b) a non-linear but still continuous relationship between the driver (x-axis) and the state variable (y-axis). In these two cases, the relationship is continuous because for each level of the driver just one level of the state variable exists. c) a non-linear discontinuous relationship between the driver and the state variable. In this case, for every level of the driver multiple states of the state variable exist. This latter case is also known as alternative stable states and represents our definition of regime shift.

system goes back to its previous state. The system presents a non-linear but still continuous dynamic (**Fig. 1**(b)). (3) The last case is the one that presents alternative stable states. As the driver increases the system will first maintain the same structure and function (i.e., resilience), to then suddenly change and move into a completely new state (Scheffer *et al.*, 2001; Beisner *et al.*, 2003; Andersen *et al.*, 2009; Bestelmeyer *et al.*, 2011; Selkoe *et al.*, 2015). Once the driver is reverted, the system will not be able to go back to the previous state from the same path. This behavior is called *hysteresis*, hence in this case for certain levels of the driver two levels of the state variable exist (Scheffer *et al.*, 2001; Andersen *et al.*, 2009; Contamin and Ellison, 2009; Bestelmeyer *et al.*, 2011; Litzow and Hunsicker, 2016; Sguotti *et al.*, 2019). Thus, the system presents a non-linear discontinuous dynamic (**Fig. 1**(c)). Different definitions call regime shifts all these three dynamics, or just the last one (non-linear and discontinuous). Here we define as regime shift the last case, in which a recovery towards a previous state is very difficult.

Regime shifts and their properties are generally explained using the ball-in-valley diagram developed by the work of Poincaré (Poincaré, 1890; Crawford, 1991; Sguotti and Cormon, 2018) (**Fig. 2**). the system (represented by a ball) is characterized by certain properties that maintain it in a certain state or regime (**Fig. 2**(a)). We can take as an example system a coral reef. Percentage of coral cover, the number of herbivorous fishes, or sea urchins density could be some of the characteristics that define the system state (Mumby *et al.*, 2007; Elmhirst *et al.*, 2009; Schmitt *et al.*, 2019). The system state is the valley in which the system lies in (**Fig. 2**(a)). The dimension, width and depth of the valley correspond to the resilience of the system (**Fig. 2**(a)) (Beisner *et al.*, 2003; Vasilakopoulos and Marshall, 2015). If stressors such as e.g. fishing, climate change or a disease start to act on the system, the system might initially be able to absorb these stressors maintaining the same structure. Imagine the stressors as forces that result in the ball being moved up the valley's sides. If the valley is deep and large, the force will not be able to move the ball away. This is exactly the definition of resilience (Beisner *et al.*, 2003; Walker *et al.*, 2004; Vasilakopoulos and Marshall, 2015; Vasilakopoulos *et al.*, 2017).

However, beyond a certain level of stress, the system will be pushed across a *tipping point* and it will move to another state (valley, **Fig. 2**(b)) (Möllmann *et al.*, 2015; Samhouri *et al.*, 2017; Dakos *et al.*, 2019; Heinze *et al.*, 2021). This process can happen in two ways. A very strong, and/or sudden new stressor arrives, capable to push the ball away from the valley and beyond the tipping point, for instance a hurricane. An alternative is that the cumulative stressors acting on the system (fishing, disease, eutrophication), by eroding the resilience, can modify the shape of the valley where the ball is lying, eventually pushing the ball across the tipping point (Vasilakopoulos and Marshall, 2015; Sguotti and Cormon, 2018). This happened, for example, in coral reefs in the Caribbean where at first the high fishing pressure on the herbivorous fishes was masked by the presence of sea urchins that helped the coral reef maintaining the same structure. However, a virus outburst in the sea urchin (*Diadema antillarum*) population, led to a collapse of the sea urchins and together with eutrophication favored the growth of algae which outgrew coral, leading the system to a tipping point (Mumby *et al.*, 2007; Bozec and Mumby, 2015; Roff *et al.*, 2015). In general, these two processes act together: a strong push may lead to shifting in a system already subject to gradual erosion. Once the system crosses a tipping point it will be in a new valley thus a new regime/state, in this case for instance an algae bed (**Fig. 2**(c)). This new state is now resilient (it has a certain depth and width) and is maintained by new *feedback mechanisms* that help its stabilization (**Fig. 2**(c) and (d)) (Norstrom *et al.*, 2009; McGlathery *et al.*, 2013; Steneck *et al.*, 2013; van Nes *et al.*, 2015). Once the tipping point is crossed and the system enters a new state, it is very difficult to bring the system back to its previous state, either because the valley is very deep and it is difficult to push the system back (e.g., when the feedback mechanisms are particularly strong) (**Fig. 2**(d)). This property of regime shifts is called hysteresis and is the key feature of discontinuous dynamics that renders understanding the dynamics and causes of regime shifts critical for management (Beisner *et al.*, 2003; Andersen *et al.*, 2009; Fauchald, 2010; Bestelmeyer *et al.*, 2011; Litzow and Hunsicker, 2016). In some cases, depending on the resilience of the new state, it will be possible to revert the regime shift (**Fig. 2**(c)).

The fold diagram (**Fig. 3**) is another way to visualize regime shifts (Scheffer *et al.*, 2001; Andersen *et al.*, 2009). At the increase in strength of external driver, the ecosystem reacts in a smooth way until, by eroding its resilience, the system reaches a tipping point (F2). At that point, the system will fall into a new state (forward shift) (**Fig. 3**). In the area of the transition, the dotted line, the

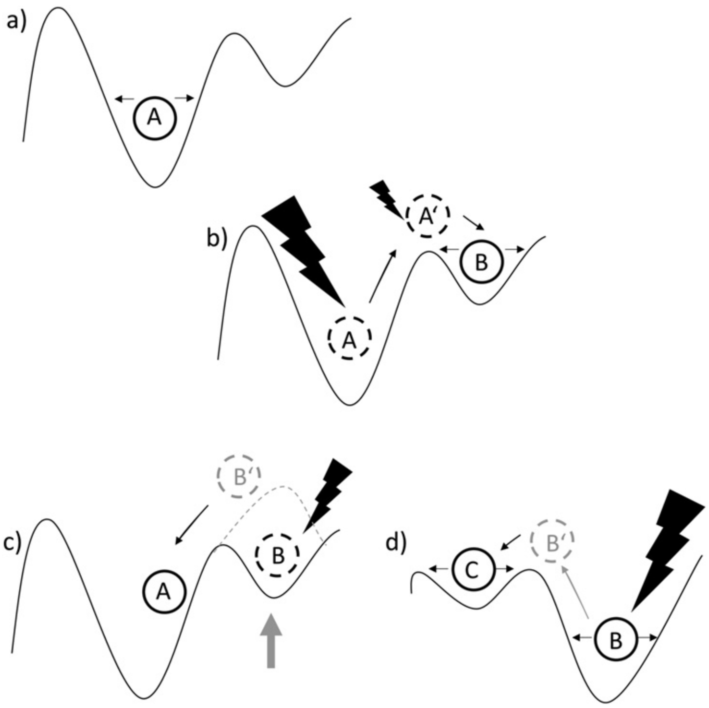

Fig. 2 Example of ball-in-valley diagram. a) The system is represented by the ball "A". It is relatively stable, driven by natural fluctuations and feedback mechanisms (small arrows), but it's functioning and resilience is intact (the ball is deep inside the valley). b) A major disturbance (bolt) pushes the system "A" towards the edge, it is in an unstable state "A" (dotted line), where already a small disturbance can push the system into a new state "B". Here the system is again stabilized by feedback mechanisms. The resilience of the new state "B" (the depth and size of the valley) is smaller than in the previous state. c) Due to the smaller resilience, already a smaller disturbance (bolt), or an erosion of the resilience, by modifying the valley (grey arrow, grey dotted line) is enough to return the system "B" to its previous stable state "A". The system's shift was reversible. d) However, the new state "B" might also be a very resilient state, and thus only strong disturbances can push the system again to a new state. However, due to changes in the ecosystem, the previous state "A" might not exist anymore and a change in "B" might result in a completely new state "C".

system can have three equilibria, two stable and one unstable and theoretically will be attracted by the state in which it is in (Scheffer *et al.*, 2001; Andersen *et al.*, 2009; Bestelmeyer *et al.*, 2011; Sguotti *et al.*, 2019). When the driver reverses (e.g., reduction of fishing pressure, or of sea urchin density), the system will not be able to go back through the same path, and will be locked in the second state. It will be able to jump back to the former state (backward shift) through another tipping point (F1), which is very delayed compared to the first (i.e., at very different conditions) (**Fig. 3**). This is the typical fold visualization used in ecology, in which the dynamic of a system depending on just one driver is shown. A system that undergoes a regime shift presents some characteristics (some of which correspond to the flags of the catastrophe theory indicated previously). It shows *bimodality* in the sense that it will mainly be presented in one or the other state and thus its distribution will be bimodal, and *inaccessibility* since it is theoretically impossible or difficult to find values of the system in the middle state or the area of the transition since the equilibrium is unstable (Bestelmeyer *et al.*, 2011). Moreover, a system that approaches a tipping point in general becomes less stable (since its resilience is eroded) and starts to have an *increased variance* (Scheffer *et al.*, 2009; Lindegren *et al.*, 2012; Dakos *et al.*, 2015). At the same time, every time that a perturbation occurs to a system close to a tipping point, the system will take more time to recover, a phenomenon called *critical slowing down* (Dakos *et al.*, 2008, 2015; Scheffer *et al.*, 2015; Clements *et al.*, 2019). These two last characteristics are particularly important in the current management attempts to anticipate regime shifts and will be described in later sections.

While the regime shift concept is theoretically quite well defined and easy to understand, practically it is really complex to model real regime shifts of ecosystems or populations (Andersen *et al.*, 2009; Bestelmeyer *et al.*, 2011; Vasilakopoulos and Marshall, 2015). Indeed, some limitations exist in the available models and the complexity of the phenomenon to model which have rendered the detection of true regime shifts and the application of this theory to management difficult (DeYoung *et al.*, 2008; Möllmann *et al.*, 2015; Sguotti and Cormon, 2018). Moreover, the utilization of statistical methods that were not able to detect hysteresis but just detect the presence of abrupt changes, have led to detection of many "regime shifts". Because of the inability to test hysteresis, a mistrust towards this branch of science has developed (Dudney and Suding, 2020). Under global changes, more and more tipping points and regime shifts are predicted to be occurring in many different systems and therefore it is important to study them and understand how this concept can be integrated into management (Lenton *et al.*, 2008; Lenton, 2011; Hughes *et al.*, 2013; Eslami-Andergoli *et al.*, 2015; Hossain *et al.*, 2017).

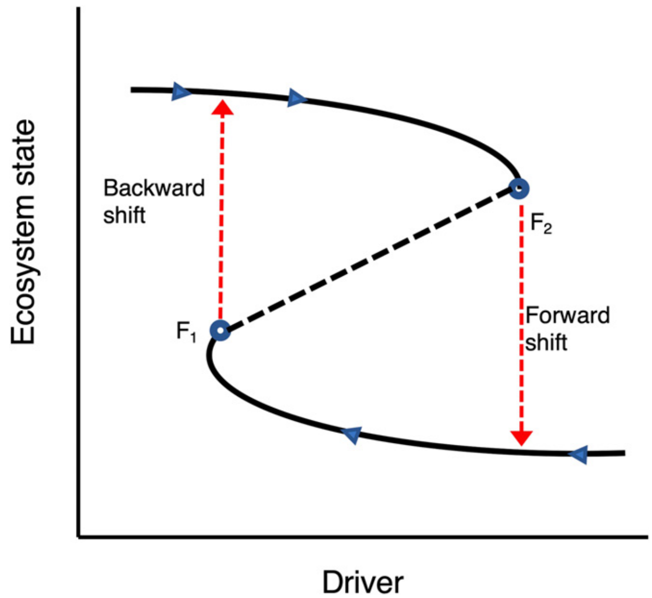

Fig. 3 The fold diagram. The fold diagram is one of the typical ways to represent a regime shift depending on one driver. On the x-axis the driver affecting the ecosystem state (y-axis). The curve line represent the relationship (non-linear, discontinuous) between the driver and the state variable. F1 and F2 are the two tipping points, where after a certain increase in the driver the state variable abruptly change to a new state (the direction of the change is indicated by the red dotted arrows). The dotted black like represents the area below the fold, or the area were three equilibria are possible two stable (the two states) one unstable (the dotted line).

Examples in Marine Coastal Ecosystems

Prevention and Early Detection

Regime shifts are often seen as negative phenomena: this subjective perception is based on the fact that the current system state is seen as favorable for the social-ecological system, being the one the social-ecological system is accustomed to, and also because the new system trajectories might be unknown. Thus, often, the best option for management is knowing what can cause a regime shift in a system, trying to prevent or mitigate the impacts from the stressors and have early detection methods to anticipate regime shifts (Biggs *et al.*, 2009; Levin and Möllmann, 2015). In some circumstances, when the system presents undesired characteristics, a regime shift could be a desirable outcome, in order to achieve a "better" state of the system and thus different management approaches will be used. This could be the case for a system already shifted and for which a reversal is desirable (this will be discussed in the next section), or for a system for which the "pristine" state does not provide as good an outcome as a modified system. Again, the positive or negative perception of a system state is often based on the perceived utility of the sea users.

The opportunity to act in order to prevent a regime shift depends often on the knowledge of the system (or comparable systems) and to the connectivity between system components (DeYoung *et al.*, 2008; Crépin *et al.*, 2012; Graham *et al.*, 2015; Wouters *et al.*, 2015). Recently, there have been advances in approaches that are able to anticipate a regime shift by showing statistical changes in the system time series, i.e. early warning signals. This is possible because systems that are close to a tipping point present the above mentioned phenomenon of the critical slowing down (Scheffer *et al.*, 2009). The critical slowing down happens when the system has a very small resilience and thus these methods can be seen as a estimation of the resilience of the system (Scheffer *et al.*, 2009; Dakos *et al.*, 2015). In general, statistical properties of the time series such as variance, autocorrelation and standard deviation change when the system has low resilience and thus is approaching a regime shift. While these methods are promising, it is not always easy to apply them in real ecosystems, due to high resolution needed in the time series data and some discrepancies in the interpretations (see more in Methods section to detect a regime shift) (Lindegren *et al.*, 2012; Wouters *et al.*, 2015). Attempts to do so have been used to retrospectively detect regime shifts (i.e. once they already occurred), for example in the late 1980s in a time series of phytoplankton biomass in the North Sea (Wouters *et al.*, 2015). The early warning signals indicated a transition to a new state several years in advance the discovered regime shift. Nevertheless, this approach should still be considered with caution as its practical applications remain difficult to determine (Dakos *et al.*, 2015).

Therefore, another way to anticipate regime shifts is to study past regime shifts either in the same system or in similar systems, and identify and monitor the ecosystems indicators that can indicate an approaching tipping point. Of course, to make use of these events, the mechanisms leading to a shift need to be quantifiable and understood. Even if those indicators are known, in order to be able to detect an impeding regime shift in real systems a high level of monitoring is needed. Practical limitations such as the effective spatial extent and scale of e.g. large ocean basins (DeYoung *et al.*, 2008), as well as disproportionate economic and human resources required, limit these large-scale monitoring schemes (Graham *et al.*, 2015). However, there are examples, of achievable approaches to anticipate a regime shift through indicators that are easier to monitor and provide valuable information about the state of and the dynamics in an ecosystem. These indicators can then give time to act and actively provide preventing management measures, such as closing areas for fisheries or reducing fishing pressure as a mitigation measure (e.g. Biggs *et al.*, 2009). These indicators are successfully applied in a range of coastal ecosystems that are very well studied, clearly defined and spatially limited.

In regions such as southern Florida (USA) the zonation of saline tolerant mangroves (halophytes) and hardwood hammock trees that do not tolerate salinity (glycophytes) is under threat of climate change induced sea level rise or storm surges (Sternberg *et al.*, 2007; Ross *et al.*, 2009; Teh *et al.*, 2019). The elevated hardwood hammocks are able to limit soil salinity by a decreased rate of transpiration, unlike mangroves, that increase soil salinity by transpiration (Sternberg *et al.*, 2007). In order to keep freshwater available in aquifers, especially with larger groundwater needs of a growing coastal population, the balance between these two plant communities is important. (Teh *et al.*, 2019; Kh'ng *et al.*, 2021). Thus, a regime shift in this zonation results in the mangroves moving further inland, outcompeting the hardwood hammock, and can threaten the availability of potable water. In this context, the stable oxygen ^{18}O can be used as an indicator for a regime shift from glycophytes to halophytes (Teh *et al.*, 2019). The monitoring of this indicator can help to initiate mitigation measures that could prevent a regime shift. ^{18}O content is found to significantly differ between saline and freshwater plants (different δ^{18}O) and serves as a good indicator of a shift to halophytic vegetation. Its monitoring allows, through modelling techniques, to simulate sea level rise and storm surges and to simulate the competition between glycophytes and halophytes, ultimately, allowing to predict decreasing resilience of freshwater plants around 20 + years before a shift is happening, leaving time for management and conservation actions (Teh *et al.*, 2019).

Also, in coral reefs, there are ways to estimate an oncoming regime shift. Coral reefs provide important ecosystem services, for example being a refuge for many marine species, protecting the coasts against the impacts of storms and tsunamis and being valuable habitats for commercially important fish species ((Woodhead *et al.*, 2019) and references therein). If the corals experience heat stress leading to a bleaching event, the formerly diverse reefs might shift into a stable and persistent macroalgae dominated state (Graham *et al.*, 2015, but see for other trajectories also Norström *et al.* 2009). Processes like fish herbivory, features of the coral habitat or management regimes determine the structure and functioning of coral reef systems. These processes can be quantifiable by for example habitat type, juvenile coral density, depth of the reef, herbivorous fish biomass, marine reserve status or nutrient regime (Graham *et al.*, 2015). Graham *et al.* (2015) used these indicators to evaluate the trajectory of coral reefs after a major climate-induced disturbance. The 17-year data set, starting in 1994, came from reefs in the Indo-Pacific (Seychelles), where the authors identified, among those reefs that had been impacted, reefs which recovered and others which had exhibited a regime shift after a bleaching event in 1998. In order to find indicators for the fate of a reef after future disturbances (e.g., climate-induced bleaching events), the authors found that for example, the density of juvenile corals, could prevent a regime shift as a higher density indicates successful settlement and survival of juveniles that reinforces the resilience of the reef. Also, when a reef was structurally more complex, it was more likely to recover. Deeper reefs were able to re-establish a healthy reef, since shallow reefs are prone to experience more stress. Additionally, already low levels of herbivores biomass, thus grazing pressure on macroalgae can prevent a regime shift as well as low nutrient inputs. The authors noted that marine reserves did not necessarily prevent a regime shift from happening, however, they can have an important role in preserving herbivorous fish biomass. Two indicators, the water depth and the structural complexity of the reef, can both be measured easily, are stable through time and can be collected over large areas, thus serving well as indicators of an impending regime shift. These two measures alone provide already a valuable tool in the management and conservation effort of coral reef systems worldwide and can help mitigate climate-induced shifts.

Recently, it has been shown that profound ecosystem changes can alter the system's properties and with those its soundscape (Rossi *et al.*, 2017). Thus, the soundscape can be used as a monitoring tool, for instance in kelp forests in order to assess the state of the system and the proximity to a regime shift (Gottesman *et al.*, 2020). Overharvesting of key predators can be a driver of regime shifts in kelp forests since it can induce the increase of sea urchins which, feeding on the kelp, can transform the system into barren grounds. Gottesman *et al.* (2020) analyzed a variety of frequencies in order to collect sounds from mammals, fish and invertebrates in different kelp forests. The data ranged from areas inside a marine protected area and outside in California, USA. Using correlations, where knowledge of a relationship between sound frequency and biological variable existed, the study could indicate that an urchin barren had a modified soundscape compared to a healthy kelp forest ecosystem (Gottesman *et al.*, 2020). Accordingly, changing soundscapes, by e.g. a decrease in key predators of sea urchins, and an increase in sounds, indicating a shift to urchin barrens, can initiate management measures, e.g. decreasing fishing pressure, to prevent a shift from happening.

All the approaches described above show that there are many researches, which aim at anticipating a regime shift based on ecological indicators, being key characteristics of the system, or statistical properties of the system time series (as used in early warning indicators). The information gathered through monitoring may then be used to act timely with appropriate management actions. Climate related impacts are almost impossible to act on in a short time frame, thus, management actions, such as, limiting or even prohibiting fishing pressure on key species or reduction in nutrient input can be useful in order to rebuild and maintain resilience of an ecosystem and to prevent the regime shift (Biggs *et al.*, 2009; Graham *et al.*, 2015; Levin and Möllmann, 2015). The ecological indicators used are often context-dependent and it might be difficult to apply the same set of indicators over different systems around the world. However, comparisons across systems can be done in order to facilitate the use of certain indicators in different areas. Moreover, it might be more straightforward to identify suitable indicators for smaller and well-defined systems (e.g. coral reefs, kelp forests, benthic systems), than for larger systems such as coastal open water ecosystems. Even though all these tools are not perfect yet, management should work to improve and find these ecosystem relevant indicators and explicitly include them in their management practices in order to be able to anticipate and thus prevent possible regime shifts.

Reversal

Hysteresis, the delayed response of a system to the reduction of the stressor is the fundamental property of regime shifts that render them very difficult, if not impossible to reverse. However, in certain cases it is possible to reverse a regime shift and go back to the previous, preferred state. Reversal of shifted systems can occur under multiple conditions, and may be driven by natural factors or by human interference, whether intentional or accidental. In the case of managed systems, it is fundamental to understand the conditions and factors that can facilitate restoration to a previous, more desirable state, once a shift has occurred. Similarly, it is important to understand when the drivers of system state are out of control of management actions, and thus might be difficult to reverse the system. Moreover, it is important to keep in mind that the recovery towards a previous state depends also on the temporal scale which we are investigating and thus temporal bias can also exist and give a wrong picture of the regime shift. Fluctuations of environmental conditions may for example lead to oscillations between alternative states in a system: depending on the time scale of observation, these oscillations can be interpreted as regime shifts with reversal, or as mere fluctuations in a continuum of system variability.

The reversibility of regime shifts has been studied in detail in the case of the sea urchin-macrophyte systems (**Fig. 4**). These temperate or high latitude coastal systems typically show kelp (or other macrophytes) forests, hosting high biodiversity, being abruptly wiped out by a proliferation of grazing sea urchins. When predation control is lacking, sea urchins maintain barren areas and actively prevent the recovery of the kelp forests. While the mechanisms behind these regime shifts are quite well understood, (Filbee-Dexter and Scheibling, 2014; Ling *et al.*, 2015), the mechanisms behind the reversal of the barren to macrophytes are less known. In general, sea urchins seem to be controlled by top-down dynamics since their control mechanisms on kelp recruitment is very strong (Christie *et al.*, 2019). Filbee-Dexter and Scheibling (2014) describe two types of feedback mechanisms that promote stability of the barren state: processes that reduce kelp recruitment on barrens and processes that allow sea urchins to maintain high densities on barrens. Only when these two feedback mechanisms are disrupted, the situation can reverse.

There is evidence that human impacts can contribute to reinforcing the resilience of the barren systems by for example removal of predators such as fish (e.g. in the Mediterranean; (Farina *et al.*, 2020)) or marine mammals (e.g. in Alaska; (Estes *et al.*, 2011; Ellingsen *et al.*, 2015; Burt *et al.*, 2020)), through fishing or hunting. This can lead to uncontrolled proliferation of sea urchins, reducing the possibility of a system reversal. Also, other factors such as pollution, habitat change, or changes in the productivity of the system (Boada *et al.*, 2017) can increase the resilience of the barren and hinder the recovery to the kelp state. Such patterns are observed in sea urchin-macrophyte systems on a global scale, presenting themselves as an intrinsic characteristic of this system (Ling *et al.*, 2015).

Nonetheless, reversals are not uncommon. This is the case, when the mechanisms maintaining resilience in the barren are disrupted, and the recovery threshold is met. Then, the kelp recovery can take place rapidly (Filbee-Dexter and Scheibling, 2014).

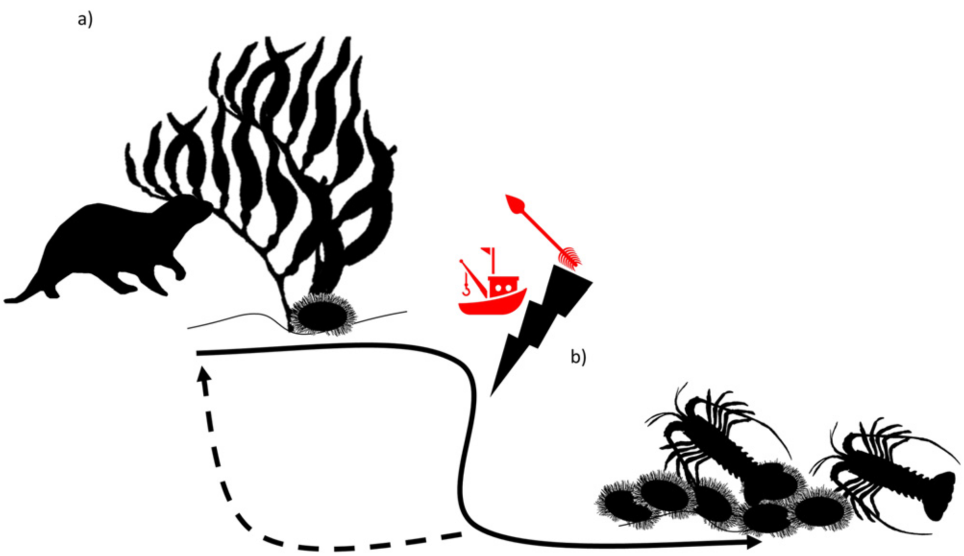

Fig. 4 The kelp forest – sea urchin regime shift. (a) A healthy kelp-forest system can be shifted by drivers such as hunting (red arrow) or fishing pressure (red boat). The predation control on sea urchins is removed. b) The sea urchins proliferate and the system shifted to barren grounds, with new species compositions emerging. The state might be reversible (dashed arrow).

As the main mechanism of resilience of barrens is the density of sea urchins, the key to recovery is the reduction in sea urchin abundance. Natural factors can drive the disruption of the sea urchin barren, through either gradual decline of sea urchins or, abrupt mass mortality events. El Niño events (Vásquez *et al.*, 2006), and climatic extremes (Fagerli *et al.*, 2013; Rinde *et al.*, 2014) have been related to sea urchin decline, possibly through recruitment failure. In Norway, for example, the decline of sea urchin populations in some areas has been associated with sea water temperature increases above a critical threshold, with both gradual increase in temperature or stochastic events exceeding the threshold for short periods of time. This was indicated as potential mechanisms leading to macrophyte recovery. Mass-mortality events of sea urchins can also be due to disease outbreaks (Scheibling *et al.*, 1999; Christie *et al.*, 2019), and toxicity resulting from harmful algae blooms (Clemente *et al.*, 2014; Feehan and Scheibling, 2014; Jurgens *et al.*, 2015). These natural factors often co-occur, with unclear synergies. For example disease outbreaks occurrence seems to be related to warming events (Christie *et al.*, 2019) or to storms. Anomalous storm events may be connected to infection of a parasite pathogen, due to increased sediment movement and large-scale vertical mixing, possibly leading to infections (Scheibling and Lauzon-Guay, 2010; Hernández *et al.*, 2020). Such sea urchin die-offs are suggested to have driven kelp bed recovery in Norway (Fagerli *et al.*, 2013; Christie *et al.*, 2019).

Human activities can promote the regime shift reversal from barren to macrophyte-dominated systems in direct and indirect ways, intentionally or not. Direct mechanisms include extracting sea urchins from the system for human consumption, i.e., harvesting (non-intentional), or culling for restoring the system (intentional). Intentional conservation measures can help the reversal. Here, the clearest case is that of sea otter (*Enhydra lutris)*, considered a keystone species in the North Pacific coastal systems because of its role in controlling sea urchins which can lead to cascading effects on kelp abundance (Paine, 1969). Restrictions on sea otter hunting effectively helped populations to recover in areas such as western Alaska and California. Sea otter populations increase in turn caused a reduction of sea urchins, ultimately leading the recovery of kelp forests and regime shift reversal in many regions (Estes and Palmisano, 1974; Breen *et al.*, 1982). Other types of intervention target directly the sea urchin. (Cebrian *et al.*, 2021) describe approaches for restoration of macroalgal forests in the Mediterranean, including sea urchin removal or exclusion and transplant with restocking of macroalgae.

Indirect mechanisms act through food web effects, and include return of formerly depleted predators, arrival of new predators (Christie *et al.*, 2019) or increase of mesopredators that were already common in the system, but increased in abundance due to a release from top predators (Norderhaug *et al.*, 2021). The increase of sea urchin predators can take place either with introduction or protection of predators (intentional) or by predatory release of mesopredators, often triggered by fishing and leading to a trophic cascade (non-intentional). Fishing, and fisheries management actions, may unintentionally result in alterations of the systems through indirect protection of the urchins or of their predators. For example, in coastal Norwegian systems, the occurrence and abundance of crabs and other mesopredators that consume sea urchins can be affected directly and indirectly by fishing pressures through extraction or through food web effects. Where larger predatory fish species (such as the Atlantic cod (*Gadus morhua*)) have been overfished, the abundance of mesopredators (e.g., decapods) may increase leading to sea urchin depletion and kelp recovery (Norderhaug *et al.*, 2021). Similar cascading effects from mesopredator release (Prugh *et al.*, 2009) leading to kelp recovery are known in other areas such as Maine (USA) (Steneck *et al.*, 2004; Ling *et al.*, 2015).

In general, human impact and natural forcing can also act together to favor a reversal of the regime shift. Thus, food-web dynamics must be considered in combination with environmental factors as they may interact in unpredictable ways. Moreover, cases when the sole predator increase was sufficient to reduce the sea urchin to such low levels, are rare. More commonly, a combination between reduced fishing on mesopredators and increasing capture of sea urchin is required. In Tasmania, marine reserves increased mesopredators and helped to maintain the kelp reducing the risk of grazing by sea urchins (Ling and Johnson, 2014) however, this was not sufficient to lead to a state reversal from barren to kelp forests. This shows the strong resilience of the barren mechanisms (Filbee-Dexter and Scheibling, 2014), and calls for structured management actions toward prevention of regime shifts, rather than recovery with managed reversal from barren to forested states. It is important to note that removal of the sea urchin drivers does not automatically lead to a macrophyte recovery. Mechanisms driving kelp or macrophyte recovery with, for example, deliberate culling or removal (Boada *et al.*, 2017; Cebrian *et al.*, 2021), may prove unsuccessful when the complex dynamics of macrophyte recovery are not fully understood. In Mediterranean coastal systems, for example, the *Cystoseira* (macrophytes) forests do not rapidly recover even after reduction of sea urchin pressure, with less desirable alternative stable states of low-canopy forests or turf algae taking over after the barren are broken. Moreover, even when macrophytes beds do recover, the overall community might shift to a different state with novel elements in the community (e.g., the arrival of Orcas in Aleutine Islands after the recovery of the sea otter population; or the changes in the fish community composition in California recovered kelp beds; see (Ling *et al.*, 2015)). These again prove that reversing a regime shift is really difficult.

Detection and interpretation of regime shifts can also be masked by time scale and confounded by a skewed perception of the system. For example, the famous anchovy-sardine oscillations typical of many upwelling systems such as Peru, California, or the Benguela currents, display recoveries driven and caused by the reversal of climate conditions and their repercussion on the ecosystem (DeYoung *et al.*, 2004). These dynamics are called regime shifts even if they do not really fit into the definition of hysteresis and irreversibility. In the Peruvian upwelling system the change between anchovy and sardine phases, corresponding to alternate regimes, has been thought for decades to be determined by interdecadal oscillations, with particularly strong El Niño Southern Oscillations (ENSO) events, connected with the warm phases of the oscillations in combination with fishing pressure (Chavez *et al.*, 2003, 2008; Bertrand *et al.*, 2020). Traditionally, anchovy abundance has been connected to cold phase periods, with sardines preferring warm periods instead. For example, the anchovy collapse of 1972–1973 was reverted, after a phase of sardine domination, by a change in the general climatic condition in the mid 1990s (Bertrand *et al.*, 2004, 2020; Schreiber *et al.*, 2011; Oliveros-Ramos *et al.,* 2021), In this period, the system entered a cold phase, with a decline of the sardines and an increase in anchovy (**Fig. 5**). However, such cycle, and our understanding about their relationships with the anchovy and sardine dynamics, have changed in the last couple of decades. Since around year 2000, the system displays unusual warm temperature - despite being in a cold phase cycle -, and no strong El Niño events, rather showing new, unusual events such as marine heatwaves. This change in the dynamics of the system is suggested to be related to climate change, and may be modifying the response of the fish to environment in unexpected ways: for example, the lack of recovery of sardines despite the apparently suitable conditions with warm temperature remains unexplained. The change observed in that year may thus represent a regime shift: from regular events and fluctuations between the resources due to the interdecadal oscillations, to a larger influence of global warming on the dynamics of the system.

In this sense, the representation of the system and ultimately, the perception of recovery might be altered by our scale of observation. Here, the interpretation of the regime moved from the earlier consideration of the individual oscillation phases as alternative regimes, to the acknowledgement of the large abundance and fluctuations of Peruvian anchovy experienced during the 20th century as an anomaly, as confirmed by observation of the system at geological time using paleontological record (Salvatteci *et al.,* 2018, 2019). Under this new interpretation, the oscillations observed in the past decades between sardine and anchovy may well represent alternative phases within one stable state. Indeed, an alternative stable state with far less desirable conditions existed in geological past and may be looming in the future under climatic changes (Salvatteci *et al.,* 2022).

In summary, the factors leading to reversal of a regime shift depend on the system's natural features and environmental drivers, on the human pressure and actions, and on the interpretation of the observer. It is increasingly clear that the interaction between natural fluctuations and anthropogenic change may lead to abrupt and durable regime shifts. In the Peruvian ecosystem, anthropogenic climate change seems to be related with disruption of the traditionally observed fluctuation patterns, with anomalous and novel phenomena emerging (Wolff, 2018; Bertrand *et al.,* 2020). These could lead to dire consequences for the ecosystem, including predicted durable changes and the disruption of recovery mechanisms of the anchovy (Bertrand *et al.,* 2018; Salvatteci *et al.,* 2022) (Bertrand *et al.,* 2004). Regime shift reversals are possible, when natural conditions help to reach thresholds that can be crossed. Targeted human intervention through mechanisms that facilitate recovery from the regime shift and adaptation practices can be critical (Bertrand *et al.,* 2018; Oliveros-Ramos *et al.,* 2021; Romagnoni *et al.,* 2022), but sometimes not sufficient. Not least, a broader horizon perspective can help to disentangle legitimate and feasible strive for restoring systems, from long-term patterns of change that may be out of the scope of human intervention.

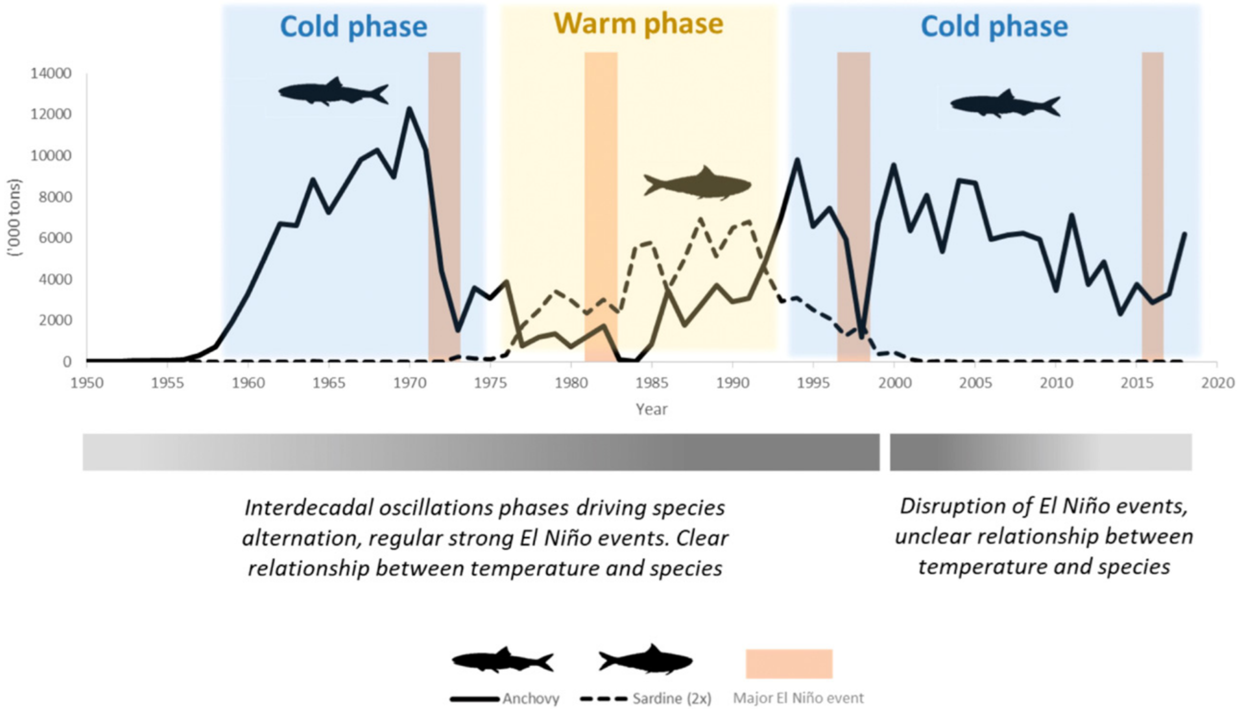

Fig. 5 Catch patterns of anchovy and sardine (magnified x2 for graphical representation) in the Peruvian system. The major El Niño events and the phase of interdecadal oscillations are shown. The connection between interdecadal oscillations and the anchovy-sardine oscillations traditionally connected each interdecadal phase to a regime shift, alternating sardine regimes to anchovy regimes. After the year 2000, however, the mechanism linking warm-water to sardine dominance has broken up: sardine did not recover, and anchovy maintained high biomass even in warm water conditions. A more long-term perspective allows to observe how the year 2000 could be considered the regime shift, moving from an oscillatory regime, to a “stable anchovy” regime, and emphasizing the relative nature of the regime shift characterization, as well as its scale dependency.

Adaptation

Adaptation is intrinsic in the concept of regime shifts. After a regime shift occurrence, the ecological components of the system undergo changes to adapt to the new regime. Similarly, the users and the management system and framework need to adapt to the new conditions (DeYoung *et al.*, 2008; Thrush *et al.*, 2016). Even though adaptation is generally seen in a positive way, such view is extremely context dependent. Under the intrinsic complexity and articulation of the socio-ecological systems relying on marine coastal ecosystems, adapting to changes in the ecological system is neither straightforward for the users nor for the management frameworks (Kluger *et al.*, 2019; Woods *et al.*, 2022).

Sea otters were highly exploited from the North Pacific East Coast at the end of the 20th century (Jackson *et al.*, 2001; Springer *et al.*, 2003; Burt *et al.*, 2020; Rasher *et al.*, 2020). The major driver of extirpation was hunting, since they were targeted for their fur. Sea otters were also key-stone species and, by eating sea urchins and crustaceans, were able to maintain a healthy and productive kelp forest ecosystem which sustained huge populations of fish (Estes *et al.*, 2011; Ellingsen *et al.*, 2015; Burt *et al.*, 2020). The extirpation of sea otters led to a structural change of the entire ecosystem: sea urchins increased in abundance resulting in overgrazing of the kelp beds (Steneck *et al.*, 2002; Ling *et al.*, 2009; Burt *et al.*, 2020). In the meantime, herbivorous fishes were also heavily exploited by fisheries and their populations started to decline. The disappearance of kelp and the decline of fishes lead to a huge increase of crustaceans (Steneck and Wahle, 2013; Steneck *et al.*, 2013; Ling *et al.*, 2015; Burt *et al.*, 2020) (**Fig. 4**). All these changes in the system lead to the adoption of strong management strategies, which forbid the hunting of sea otters. Nevertheless, no clear recovery was seen over the first 20–50 years (Burt *et al.*, 2020). The users of the sea, in particular the fishers, readily adapted to the disappearance of fish and sea otter and started to target crustaceans. Crustaceans were far more profitable than fish and became a large fishery as well as a subsistence fishery. Some fishers did not even remember the system as it was before the otter collapses, a perfect case of the "shifting baseline syndrome" (Pauly, 1995; Jackson, 2001).

Over the last decades, sea otters started to increase again and to eat sea urchins and crustaceans. This lead to collapses of the crustaceans fisheries and was perceived differently by different users (Steneck *et al.*, 2013; Burt *et al.*, 2020). Some saw the comeback of sea otters negatively, perceiving it as a threat to their activity of crustaceans fisheries. These were mainly fishers from a younger generation who never saw the ecosystem as it was before. Others were pleased about the return of the sea otters and saw new possibilities of earning money with tourism or catching fishes (Burt *et al.*, 2020). This first example shows a case of a regime shift, where the users readily adapted to the first shift in the ecosystem, but were reluctant to adapt again to new (actually the former rather pristine) conditions. In this particular case, management was imposed by the State (the prohibition of hunting for otters) in order to restore an ecological depleted state but was seen critically by the coastal population that preferred the state full of crustaceans. Therefore, the importance of understanding how the system might develop and the judgement of what is considered favorable needs always to be discussed when dealing with regime shifts. However, different users might have different ideas of what is a "good state" and this can lead to conflicts.

While the kelp forest transition to sea urchins is one of the most notorious regime shifts happening in coastal areas, together with coral reefs transitions, it is more difficult to study and analyze the regime shifts occurring in coastal fish populations (DeYoung *et al.*, 2008; Elmhirst *et al.*, 2009; Möllmann and Diekmann, 2012; Filbee-Dexter and Scheibling, 2014; Conversi *et al.*, 2015). One striking example is the abrupt collapse of Atlantic cod stocks along the Canadian coast (Haedrich and Hamilton, 2000; Frank *et al.*, 2005, 2016; Sguotti *et al.*, 2019). Here, cod collapses occurred following discontinuous dynamics and thus showing regime shift behaviors (Frank *et al.*, 2005, 2011; Sguotti *et al.*, 2019). These shifts were due to unsustainable fishing pressure coupled with unfavorable environmental conditions and in some cases occurred with a loss of more than 80% of the fish biomass in just two years (Frank *et al.*, 2016; Sguotti *et al.*, 2019). The collapse of the cod population off Newfoundland (Canadian coast) is particularly famous because it was the first collapse of a monitored fish stock in modern times and led to a fishing moratoria, leaving thousands of people without jobs (Haedrich and Hamilton, 2000). A dramatic change in the ecological system resonated through the whole socio-economic chain, leading to a national crisis, an identity crisis for the fishers who were left without employment and a cultural crisis with many fishing villages disappearing. Even though a fishing ban was in place and cod was not directly fished for more than 20 years the stock did not show any sign of recovery (Bundy and Fanning, 2005; Hutchings and Rangeley, 2011; Mullowney and Rose, 2014; Sguotti *et al.*, 2019). This again shows that, once a tipping point is crossed, it is difficult or even impossible to go back to the previous conditions even if drastic management measures are adopted, since new ecological feedback mechanisms can hold the system in the new state. These failed management measures highlight how regime shifts can create a situation of a loss of trust between institutions and society when management measures do not work.

The collapse of cod, a top predator of the marine ecosystems, led to an increase of its preys, in particular crustaceans and lobsters (Plasson, 1994; Steneck and Wahle, 2013). As a consequence, lobster fisheries started to increase in numbers and became more and more profitable in the Canadian shelf coast (Frank *et al.*, 1994, 2005, 2011; Alfonso *et al.*, 2012; Ellingsen *et al.*, 2015). Once again, fisheries adapted to changes in the ecological compartment. However, among the fishers, those already engaged in lobster fisheries or who managed to change and adapt their boats were winners, but those who used to fish cod and did not adapt were losers of this regime shift. This is another important pillar of regime shifts and adaptation; even if an entire sector manages to adapt, individually some people will benefit and some others will be damaged by the ecological regime shifts (Crépin *et al.*, 2012; Lade *et al.*, 2013). At the moment, the fisheries for lobster is strong and profitable, and cod is still depleted. However, these coastal Canadian shelf ecosystems are typically low resilient and especially under high exploitation and climate change lobster populations could dramatically decline, bringing the ecosystem towards a new structure (Bakun, 2006). This could in turn damage fisheries. Thus, the adaptation towards new resources is an advantage when dealing with regime shifts but has to occur in a controlled way in order to avoid new possible, further deleterious tipping points.

In the final example, adaptation to the Baltic Sea regime shift will be described. This regime shift started in the middle of the 1950s and encompassed different trophic levels of the Baltic Sea coastal ecosystems (Alheit *et al.*, 2005; Möllmann *et al.*, 2009, 2015; Conversi *et al.*, 2015; Gårdmark *et al.*, 2015). Until the mid 1950s the Baltic Sea ecosystem was mainly oligotrophic and dominated by seals (**Fig. 6**(a)) (Osterblom *et al.*, 2007). The increased eutrophication coming from land waste and the hunting of the seal, led to a first regime shift of the ecosystem. The new state of the Baltic Sea was highly eutrophic, dominated by demersal fish species, e.g., cod, and with other different species like sprat (*Sprattus sprattus*) and herring (*Clupea harengus*) being controlled by cod abundances (**Fig. 6**(b)) (Casini *et al.*, 2008a; Gårdmark *et al.*, 2015; Reusch *et al.*, 2018; Tomczak *et al.*, 2021). Around the 1970s the cod stocks increased dramatically, allowing the fishery to become one of the biggest of Europe. However, at the end of the 1980s, the strong overfishing on cod stocks, coupled with changes in environmental conditions such as reduced inflow of oxygen-rich saline water from the North Sea and higher temperatures, lead to a collapse of the cod population and a complete reorganization of the ecosystem (**Fig. 6**(c)) (Köster and Möllmann, 2000; Casini *et al.*, 2008b; Leeuwen *et al.*, 2008; Möllmann *et al.*, 2008, 2021; Lindegren *et al.*, 2010; Pershing *et al.*, 2015). Sprat and herring started to dramatically increase and, although fishing for cod was reduced, the new system was maintained by a new feedback mechanism: the so-called "cultivation effect" (Köster and Möllmann, 2000; Bakun, 2006; Casini *et al.*, 2008a; Heikinheimo, 2011). Sprat and herring, eating the larvae of cod, impede the recovery of cod. Because of that, the abundance of the pelagic fishes continuously increased. However, recently, the system is going through a new shift. The high eutrophication has led to the instauration of large anoxic zones in various areas of the Baltic Sea, leading to a die-off of benthos and demersal organisms (Köster *et al.*, 2017; Ammar *et al.*, 2021). The fishing on herring and sprat drastically increased leading to a collapse also of the herring stocks (**Fig. 6**(d)). At the moment, the fisheries have lost their main resources and struggle to stay alive even if many management measures are in place (Aps and Lassen, 2010; Froese and Quaas, 2011; Möllmann *et al.*, 2021).

This is an example in which a regime shift was seen as a positive outcome (the passage from an oligotrophic to an eutrophic ecosystem with high abundances of cod) and where human adaptation to this shift had a negative outcome. Indeed, the fishery readily adapted to the new state in the 1970s and started to fish the cod stock. Likewise, this adaptation happened with the increase of herring and the consequent overfishing. While adaptation is vital for populations and countries to survive ecological shifts, management measures need to be introduced to support a stable and healthy ecosystem (Levin and Möllmann, 2015; Selkoe *et al.*, 2015; Sguotti and Cormon, 2018; Ingeman *et al.*, 2019; Woods *et al.*, 2022). When this does not happen, adaptations may be risky and become a new source of possible regime shifts. They may also harm the resilience of the ecological system (Woods *et al.*, 2022). This example is also particularly interesting because the desired state, the one in which cod was abundant, was not the pristine ecosystem state, but the result of intense human drivers. This shows that past, "pristine" states are often forgotten and that it is difficult to see which is the right environmental state the system should go back to (Pauly, 1995; Jackson, 2001; Pinnegar and Engelhard, 2008). This is relevant in a management context in which ecosystem baselines are generally defined as targets to restore ecosystems to a "pristine" state. The presence of regime shifts, as the ones described here, should actually move the management towards different principles.

As we have seen in these examples, once a regime shift occurs, the ecological system will adapt and change into something new, dominated by new structures, new species, and new mechanisms. The new state might be considered worse or better compared to the previous one, but this is rather a subjective or economic definition that not necessary refers to the ecological system state. In general, when ecological regime shifts occur, the users of the sea will be able to adapt, especially if management measures are flexible enough to allow so (Woods *et al.*, 2022). This means, for instance, allowing fishers to changing gears and change target species. At the same time, management should aim to promote and maintain resilience in order to avoid new possible tipping points due to maladaptation (Walker *et al.*, 2004; Rockström *et al.*, 2009; Darling and Côté, 2018; Woods *et al.*, 2022). This adaptation could however require a rather long time or could involve different people compared to the ones that were previously involved. Present management should change and move towards a new resilience paradigm in order to favor flexibility but also maintaining ecosystems structures and functions.

Emergent Issues

In the previous sections of this chapter, we illustrated various examples of regime shifts, in different ecosystems (e.g. kelp forest, coral reefs, coastal open ocean ecosystems), caused by different drivers and with a variety of repercussion on the social and economic systems (Mumby *et al.*, 2007; Möllmann *et al.*, 2009, 2015; Conversi *et al.*, 2015; Ling *et al.*, 2015; Pershing *et al.*, 2015; Burt *et al.*, 2020). These examples, give us the possibility to discuss more in detail some emergent issues of regime shifts, that are still hard to tackle and that render this topic more complex and more difficult to study.

The first emergent issue is the so-called problem of scales, both spatial and temporal (Tuya *et al.*, 2005; DeYoung *et al.*, 2008; Blenckner *et al.*, 2015; Conversi *et al.*, 2015; Möllmann *et al.*, 2015). From a spatial point of view, it is easy to detect and understand a regime shift when it occurs in a defined area with a clear and distinguishable community structure and physical habitat (van Nes *et al.*, 2007; Elmhirst *et al.*, 2009; Blamey and Branch, 2012; Hempson *et al.*, 2018; Eklöf *et al.*, 2020). A transition of a coral reef to an algae bed or of a kelp forest to an urchin barren is thus clearly detectable. Instead, a transition of a shelf sea from a system dominated by demersal fish to a system dominated by forage fish is more difficult to detect and may go unnoticed for a long time (Hughes *et al.*, 2013; Conversi *et al.*, 2015; Möllmann *et al.*, 2015). Moreover, larger coastal ecosystems can extend over a large area and therefore the changes might not be equally distributed in all the areas of the system (Tomczak *et al.*, 2013b;

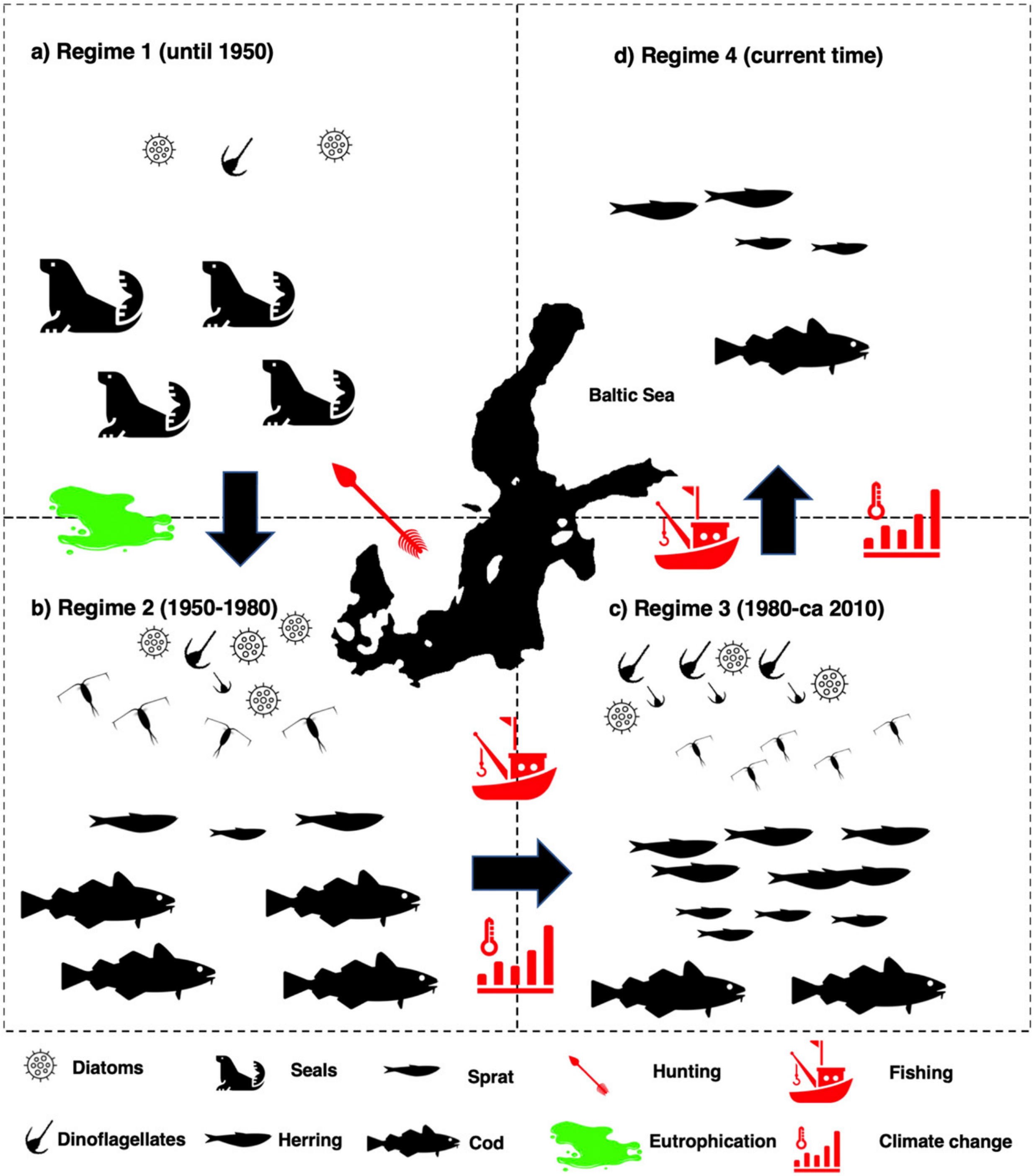

Fig. 6 The Baltic Sea regime shift. a) The first regime of the Baltic Sea with high abundance of seals. The arrows indicate how the state of the Baltic Sea changed and the colored icons indicate the drivers inducing these changes. b) the regime 2 (1950–1980s): an eutrophic state of the system with high abundance of cod, dominating the pelagic species. c) the regime 3 (1980s to 2000): overfishing of cod with an increase in pelagic fish biomass, d) regime 4, the current time: widespread anoxic zones leading to a decrease in overall fish biomass. The icons indicated in all the regime show in a simplify way how the ecosystem changed. The legend of the icons can be found below. In the middle of the plot the Baltic Sea.

Fisher *et al.*, 2015; Möllmann *et al.*, 2015; Rocha *et al.*, 2018; Eklöf *et al.*, 2020). This is also true for different coral reefs that might be located in the same larger area but might experience different levels of the disturbances (Tuya *et al.*, 2005; Hempson *et al.*, 2018; Gizzi *et al.*, 2021). Thus, albeit it can be used as an early warning signal (Kéfi *et al.*, 2007), fragmentation makes regime shifts difficult to detect and especially in large marine ecosystem it is difficult to overcome. Studies can be made that investigate multiple spatial scales to understand whether the regime shift has occurred throughout the system or if it is spatially limited to a particular area of a certain system. The relevant spatial scale to consider would be the one that is useful for management and therefore where results related to regime shifts can be directly translated into management advices (Jiao, 2009; Swaney *et al.*, 2012; Levin and Möllmann, 2015; Selkoe *et al.*, 2015).

Similarly, also the temporal scale of regime shifts might vary across the different organisms. Small organisms such as plankton might react faster to changes than long-lived animals like large fish or even mammals (Beaugrand, 2004; DeYoung *et al.*, 2008). Thus, while the response of the lower trophic levels to a change in one or more drivers can take one year or even less time, several decades could be needed to fully complete the regime shift in all the trophic levels. This makes it even more complicated to study these types of dynamics. The concept of regime shift is generally associated with something that occurs fast and abruptly. However, the documented regime shifts are often occurring slowly and gradually (over decades); one of the reasons why they go unnoticed (Hughes *et al.*, 2013). While these different temporal scales of the shifts and their long occurrence time seem to complicate even more the regime shift science, these temporal mismatches can be actually useful to predict regime shifts. Indeed, small organisms and lower trophic levels could be used as sentinels of the changes and could be helpful to predict the regime shift if this is a bottom-up shift.

Another issue of the regime shifts is that they are often initiated by cumulative drivers. A driver, for instance fishing mortality, can be studied in detail and a threshold could be detected that, if passed, could induce a regime shift (Contamin and Ellison, 2009; Samhouri *et al.*, 2017). However, if other drivers such as temperature or eutrophication interact with the fishing mortality, this threshold can shift and even a sustainable fishing mortality can actually cause a regime shift (Halpern *et al.*, 2008; Rockström *et al.*, 2009; Hughes *et al.*, 2013; Carpenter *et al.*, 2017; Sguotti *et al.*, 2019). Thus, it is really difficult to understand how the drivers will interact, which drivers will be responsible for the shifts and ultimately which mechanisms will induce the transition.

The usually held perception that regime shifts are a negative event may be a misconception, as such anthropocentric judgment depends on the perspective we are looking at them. We saw that for example some fishers were happy about the transition because the new system state was economically more beneficial to them, while others preferred to go back to the previous state because they were culturally more linked to it (Haedrich and Hamilton, 2000; Steneck and Wahle, 2013; Steneck *et al.*, 2013; Burt *et al.*, 2020). Thus, regime shifts (independent of the direction of change) always bring repercussions on the full socio-ecological systems (Lade *et al.*, 2013; Rocha *et al.*, 2015a; Biggs *et al.*, 2018). Adaptation is possible but always comes with some delays and costs. Therefore, we should aim at maintaining systems since a new transition could lead to undesirable states or a reversal of a shift might simply be not possible because the conditions have changed and the system cannot move back. Understanding the effects of possible regime shifts on the full socio-ecological system is important to predict which costs will be associated with the transition apart from ecological losses.

Methods to Detect Regime Shifts

While regime shifts are well documented and relatively well understood; the methodologies available to detect or predict them are still rather limited, rendering the detection of this type of dynamics in empirical data extremely difficult (Andersen *et al.*, 2009). The limited number of methods is due to the complexity of the phenomenon which comprises dynamics including bifurcations, discontinuities and which are often caused by two or more drivers (Beisner *et al.*, 2003; Petraitis and Dudgeon, 2016). Approaches, such as theoretical models or experiments might be more successful in detecting regime shifts, but are less directly applicable to management.

Empirically, regime shifts are often detected in ecosystems or populations following a multi-steps approach (Scheffer *et al.*, 2001; Bestelmeyer *et al.*, 2011). A first indication of regime shift is the presence of *abrupt changes* in the time series of the analyzed systems (Andersen *et al.*, 2009; Bestelmeyer *et al.*, 2011; Lindegren *et al.*, 2012; Sguotti *et al.*, 2019; Möllmann *et al.*, 2021). The definition of abrupt changes varies depending on the system, but it usually implies a sudden and rather abrupt shift in the mean or variance of a system. A range of methods exist to detect change points in time series and generally look at sudden changes in mean or variance or both (Andersen *et al.*, 2009; Bestelmeyer *et al.*, 2011). Some of the most common methods used in ecology are STARS (Sequential t-test), the R-packages *strucchange* and the bayesian change point analyses *bcp* (Rohrbeck, 2013.; Rodionov and Overland, 2005; Erdman and Emerson, 2007; Bestelmeyer *et al.*, 2011) (**Fig. 7**). All these tools are rather easy to use and give straightforward results. However, they are not able to distinguish simple abrupt changes from true regime shifts (i.e., discontinuous dynamics) since they cannot detect the presence of hysteresis, one of the fundamental properties of regime shifts. One of the limits of the regime shift literature in aquatic ecosystems is that often the only method used to detect regime shifts is the change point analysis, thus creating a mismatch between definitions and results. This has somehow reduced the trust in this concept (Dudney and Suding, 2020).

Another proof of regime shift dynamics is the presence of bimodality in the state variable (**Fig. 7**) (van der Maas *et al.*, 2003; Bestelmeyer *et al.*, 2011; Petraitis and Dudgeon, 2016). The presence of bimodality (a bimodal distribution of the state variable) indicates that two different alternative states are present, supporting the hypothesis of regime shifts (Scheffer *et al.*, 2001; Beisner *et al.*, 2003; Andersen *et al.*, 2009; Selkoe *et al.*, 2015).

Then, finally after detecting abrupt changes and bimodality in an empirical system, the presence of hysteresis must be proved. In general simple visualization tools can be applied to detect hysteresis such as the driver-state plot (Andersen *et al.*, 2009; Bestelmeyer *et al.*, 2011) (**Fig. 7**). The driver-state plot is just a plot of the state variable depending on one driver. If for every level of the driver just one level of the state variable exists, then the system presents linear dynamics. Instead, if for one level of the driver multiple states of the variable exist and the relationship follows a loop-like shape then it is a sign that the system presents hysteresis to that driver and thus that regime-shift like dynamics might exists. In this case, modelling the system with a continuous model would be wrong, since hysteresis is present and thus different approaches need to be used. One possibility is to fit two different models to the time series before and after the shift or to use a model which can incorporate a threshold such as thresholds generalized additive models (tGAM) (Ciannelli *et al.*, 2004; Bestelmeyer *et al.*, 2011; Vasilakopoulos and Marshall, 2015; Vasilakopoulos *et al.*, 2017). These modelling approaches might not be the standard statistical tools but are fundamental to understand how the system behave before and after the shift and how the relationship with an external driver has changed.

Most of the papers dealing with regime shifts detection based on empirical data use some part or the total multistep approach described. Recently, some other methods have been proposed to model regime shifts (**Fig. 7**). One such method is the Integrated Resilience Assessment (Vasilakopoulos and Marshall, 2015; Vasilakopoulos *et al.*, 2017; Ma *et al.*, 2021; Tsimara *et al.*, 2021; Hidalgo *et al.*, 2022). This method aims at detecting regime shifts and calculating resilience of a system depending on one driver. The system needs to be reduced to a time series (e.g. through a statistical reduction technique (see below)) and subsequently a model comparison between GAMs and threshold GAMs is made in order to detect whether regime-shift like dynamics are present. If the tGAMs results as the better model, a resilience index based on the distances of the system from the stable states determined by the GAM is computed. In this way, it is possible to understand, when the tipping point occurred and what the resilience of the system is at the different times. However, all these models cannot really model real bifurcations and explore hysteresis.

The stochastic cusp model, is able to model true tipping points using a mathematical equation of the catastrophe theory (see Theory) (Petraitis and Dudgeon, 2015; Diks and Wang, 2016; Sguotti *et al.*, 2019). Moreover, it models the dynamics of a state variable depending on two interactive drivers (Thom, 1975; van der Maas *et al.*, 2003; Grasman *et al.*, 2009; Petraitis and Dudgeon, 2016). One of the drivers is the so-called splitting factor and is the driver controlling the dimension of the state variable. The

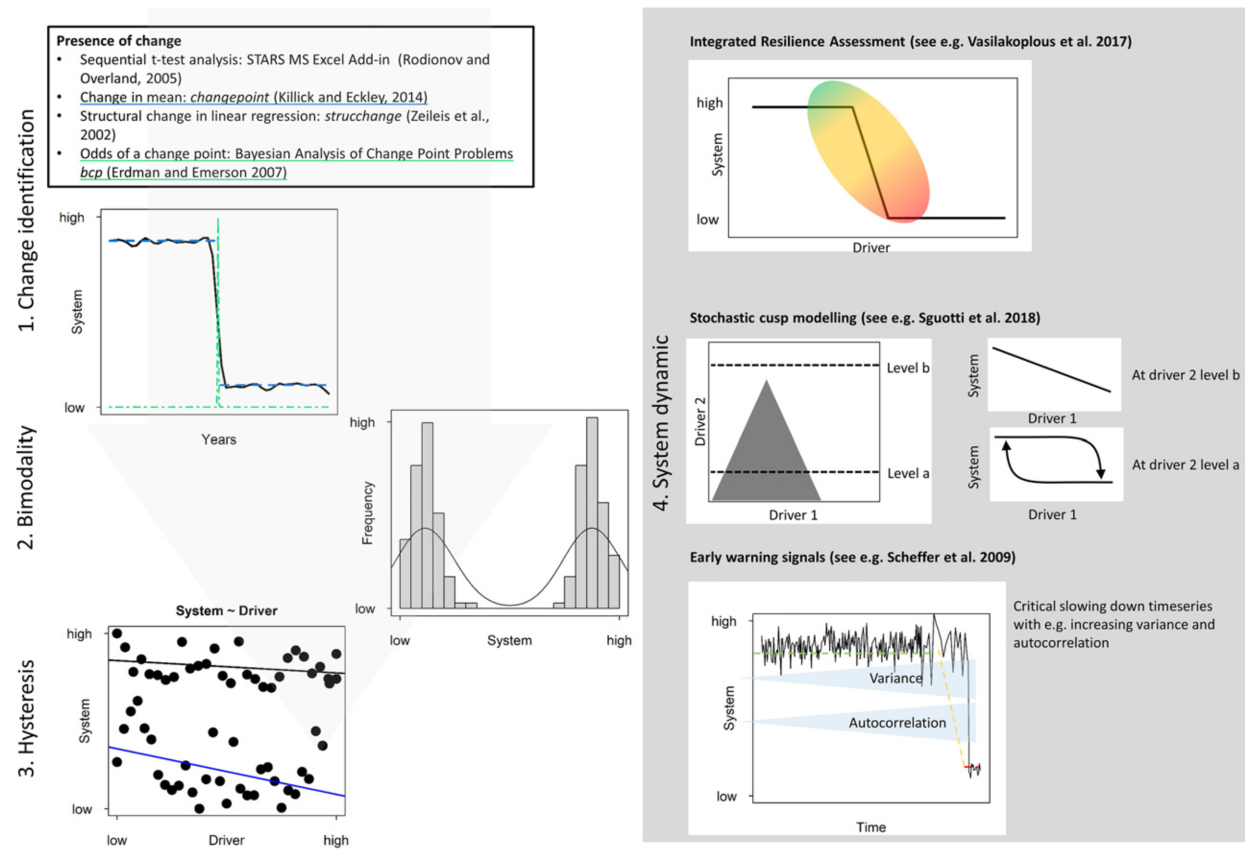

Fig. 7 Overview of methods to detect a regime shift. 1) A change can be identified in a time series using several change point detection methods. 2) Identification of bimodality in the time series 3) Identification of hysteresis: One driver level has several system state levels. 4) Investigating the system dynamics using tools like integrated resilience assessment, stochastic cusp modelling and early warning signals.

second driver is the bifurcation factor which controls whether the system follows a linear or a discontinuous dynamic (regime shift) (Diks and Wang, 2016; Sguotti *et al.*, 2019). Thus, this model is able to model the presence of true bifurcations and of hysteresis moving a step forward compared to the multi-step approach. Moreover, it is able to detect whether the system is in a stable state (outside of the bifurcation area) and thus if it is resilient and present hysteresis depending on the two drivers (Ceja and Navarro, 2012; Petraitis and Dudgeon, 2015; Diks and Wang, 2016; Sguotti *et al.*, 2019). While it is evident that as all the other model, the cusp is a simplification of reality, this model seems rather important to detect discontinuous dynamics but it is still seldomly used in ecology and more applied in economic or behavioral science.

Another group of methods that is able to detect possible tipping points, is the Early Warning Signal (EWS) framework (Scheffer *et al.*, 2009). EWS, examining statistical properties in the time series, such as increase of variance, increase of autocorrelation are able to detect the presence of tipping points flags such as the critical slowing down (Dakos *et al.*, 2008, 2016; Scheffer *et al.*, 2015) (**Fig. 7**). Indeed, a system approaching a tipping point is a system with very low resilience and thus will not be able to recover quickly after perturbations occurred (critical slowing down). Statistically, this can be seen by the increase in autocorrelation and the increase in variance of the time series (Dakos *et al.*, 2012; Litzow *et al.*, 2013; Scheffer *et al.*, 2015; Wouters *et al.*, 2015; Clements *et al.*, 2019; Zhang, 2020). There are a range of indicators that are part of the EWS framework and can be used to detect the proximity to a change point. However, these methods are often used on modelled time series because empirical time series are too noisy. Nevertheless, they could potentially be very helpful in the study of regime shifts and very useful for management (see the management section).

All the methods described above use time series of the state variable and of the drivers to detect regime shifts. Thus, if the dataset is a simple time series, for example through a monitoring time series of kelp cover and sea urchin densities, they can be directly applied. In the case of a multivariate dataset, for instance time series of different species in an ecosystem (e.g. the ecosystem wide changes in the Baltic Sea), a reduction technique such as Principal Component Analysis (PCA), Empirical orthogonal functions (EOF) or Dynamic Factor Analysis (DFA) needs to be applied (Beaugrand, 2004; Vasilakopoulos *et al.*, 2017; Hidalgo *et al.*, 2022; Hunsicker *et al.*, 2022).

Additionally, there are other methods that can be applied directly to food webs. Generally, the work on food webs is more related to understanding the resilience depending on species interactions or functional traits and thus detect the presence of potential tipping points that could cause regime shifts (Allesina and Tang, 2012; Tomczak *et al.*, 2013a; Allesina *et al.*, 2015; Landi *et al.*, 2018). To study food webs is, however, often complicated due to the large quantity of data and information needed. In some cases, data driven approaches can be used to describe changing relationships and understand how the resilience of a network might have changed. Often, studies that look at regime shifts in food webs use theoretical models and test possible mechanisms of regime shifts (Biggs *et al.*, 2009). Even though these methods might be helpful in understanding how change may occur, they are difficult to apply when data are scarce, and difficult to be translated directly to management. However, information about regime shifts mechanisms could help to understand what species to protect in order to avoid the tipping point. Similarly, ecosystem models such as e.g. Ecopath with Ecosim, Osmose or Atlantis include methods for identification of changes in the system through community and ecosystem indicators.

Finally, understanding regime shift dynamics and helping to find indicators can also be conducted through experimental or field work approaches. In the case of experiments, it is relatively straightforward by applying different level of the stressors to a system to see how the system reacts, if it shows an abrupt change and how much time it needs to recover (Graham *et al.*, 2013; McGlathery *et al.*, 2013; Steneck *et al.*, 2013; Melis *et al.*, 2019; Schmitt *et al.*, 2019). In this way, the possible threshold level of the drivers and also the hysteresis of the system can be obtained. These types of experiments have been conducted for instance to study the effect of removing herbivores on kelp, or to study the effects of climate change on coral reefs and urchins barren simulated in laboratory. Experiments that are more complex could test the regime shift on the "entire" trophic chain through mesocosms experiments or experiments in the total assemblage. Even if these systems do not fully represent the natural structure of the ecosystem, they can still provide indications about driver levels, mechanisms and possible recovery. Finally, direct field work observations can also be used to study regime shifts. For instance, the study of different coral reefs exposed to varying magnitude of the drivers, can help to understand whether they have similar or divergent recovery path and if the recovery can be achievable (e.g. Graham *et al.*, 2015).

All the methods illustrated here are currently being used in marine ecological science to understand and detect regime shifts. Nevertheless, their application to management is still far away and the inclusion of the concepts of regime shift and resilience just started to be discussed recently. The combination of all these methods, from experiments, to field work to finally statistical analyses could help to really understand regime shifts and apply meaningful management measures in order to protect our ecosystems.

Management

The concept of regime shifts has been hardly implemented in management legislation worldwide, due to the unclear contextualization and hard-to-measure dynamics (Woods *et al.*, 2022). In particular, the difficulties reside on where the tipping point lies, how the pressures can interact and what are the mechanisms that lead to the regime shift. For instance, while it is relatively easy to understand if a fish stock is overexploited thanks to quantitative metrics calculated for reference and limit levels of a viable population, it is more difficult to understand where the threshold of no-recovery will be reached and how the drivers will potentially modify this threshold. The consequences of non-linear and discontinuous changes are highly visible to managers. However, in the lack of clear metrics and definitions, the legislator can do very little to explicitly include measures to avoid or deal with regime shifts. One limitation is the fact that thresholds are often only identified after they have been crossed. In this sense, only shifts that have already occurred can be quantified, and thus managed. Moreover, another limitation that prevent the inclusion into management is the fact that multiple cumulative drivers often interact in causing regime shifts. To account for these, the development of Safe Operating Spaces for the systems has been proposed and being advanced to be included into management (Rockström *et al.*, 2009; Carpenter *et al.*, 2017).

For system shifts, where recovery to a previous state seems very hard to reach, like in the shelf systems, a feasible management option is to be open to adaptation and provide flexibility to the users of the sea. However, for systems where recovery is possible, past experience might be valuable for avoiding future shifts. In the Peruvian upwelling system, where in the past regime shifts from high to very low anchovy biomass have been triggered by a combination of interdecadal oscillations, El Niño events and high fishing pressure, substantial, science-based management measures have been taken. Here, the successful prevention of further regime shifts through a strong reduction in the fishing pressure on the anchovy stock during the El Niño events has concretized with the avoided shift under the El Niño in 1997–1998. This is one of the few cases of demonstrated success of regime shift prevention. The prevention of the shift was facilitated by two key factors: a strong stock monitoring and management in place; and the possibility of capitalizing on early warning signals, in the form of El Niño forecasts (Bertrand *et al.*, 2020; Oliveros-Ramos *et al.*, 2021).

It is clear that the many, if not most mechanisms of regime shifts are driven by human actions. According to (Folke *et al.*, 2004), the likelihood of regime shifts in a system increases when resilience is already reduced by e.g., fishing, pollution, climate change, and increase of magnitude, frequency, and duration of disturbance regimes. These combined, often synergistic factors lead to regime shifts. Rocha *et al.* (2015a) point at the need to understand the patterns of variables that co-occur, causing regime shifts. Most frequent causes are food production, climate change and coastal development. The authors highlight how management failures can be expected when the root causes and key drivers are not understood, and management actions rather focuses on well understood, data-rich variables (Rocha *et al.*, 2015a). As such, monitoring of ecosystem state indicators, identification and management of the key drivers and their impact on system resilience, should be the goal for managing regime shifts. Thus, being able to prevent and early detect approaching regime shifts.

The temporal mismatch between management scale and processes driving population dynamics and regime shifts require solutions for dealing with short- and long-term fluctuations differently. (Freón *et al.*, 2005) and DeYoung *et al.* (2008) propose a two-level management strategy, thus combining a conventional single-species management to deal with short-term fluctuations with a long-term ecosystem management strategy. Fostering resilience in management and an adaptive management strategy have been also proposed. Practical examples, however, remain scant. The case of the Baltic Sea represent a valuable example: here, the spatial scale of the regime shift that is known to have occurred is visualized with differential effects across the system, with effects on some species being much larger than others (Yletyinen *et al.*, 2016). This points at the fact that the multiple drivers acting at several scales lead to problems in defining appropriate scales for management and defining responsibilities and actions (Österblom *et al.*, 2010).

From the conservation perspective, management of regime shifts presents more success stories. Here, the scale matters (as described by DeYoung *et al.* (2008)): localized systems might be easier to restore, especially when the effort required is resource-demanding. Kelp or macrophytes restocking seems to be an effective approach to help the system to recover, however the high cost make this a restricted effort. Similarly, efforts of restoration of predators (e.g. the sea otter in Alaska; seals in the Baltic Sea) proved successful for managing the system in a desired direction. However, as described above, the desirability is strongly user-dependent, with many users, both in Alaska and Baltic examples, not seeing positively the recovery of top predators due to perceived negative impact on their activities. Additionally, in areas such as California, the system moved to a different state upon kelp recovery, leading to the development of a different species community (Ling *et al.*, 2015) and reference therein). Moreover not all efforts are successful: for instance the efforts for restoring the kelp forest can prove unsuccessful despite high efforts, leading to the realization that avoiding tipping points is by far the best option (Filbee-Dexter and Scheibling, 2014). Once again, clear understanding of the dynamics at place in the local context are a key for understanding the shifts, to identify specific threshold and to put in action effective strategies to avoid crossing tipping points (Marzloff *et al.*, 2016).

One of the most immediate tools that could be integrated into management are Early Warning Signals (Biggs *et al.*, 2009; Wouters *et al.*, 2015; Zhang, 2020). Early warning signals for regime shifts have been studied extensively (e.g., Carpenter *et al.*, 2011; Chevalier and Grenouillet, 2018); however, their ability of anticipating the shift may depend on the data available. Moreover, even if the shift is anticipated, intrinsic dynamics of the system as well as the reactivity of the management action could frustrate their application (Contamin and Ellison, 2009). More recently, (Alegre Stelzer *et al.*, 2021) reviewed experimentally studied Early Warning Signals for aquatic systems. They found that, while the majority of the EWS proved successful in most cases, none was consistently successful; moreover, they point at a mismatch between experimental results and applied, real-life cases. The authors warn about extrapolation of current results for management, calling for further integration between experimental and applied work toward applicability.

Ultimately, we can conclude that regime shift prevention should be the priority, however the identification of thresholds with early warning signals remains a challenge. Regime shift reversal is a possibility only under certain circumstances, mostly determined by local and global environmental factors on which the manager has little influence - with due exceptions such as local land-based pollution for example. In addition, the high cost, high uncertainty of efforts for regime shift recovery, with the high risk that the new status does not correspond to the expectations, are discouraging on the aspect of ecosystem recovery. This does not undermine efforts for reducing the impact on the system, nor explicit conservation efforts such as the successful sea otter story; but rather warn on the unexpected turnovers. Only in areas with very clear understanding of the system dynamics such effort can be

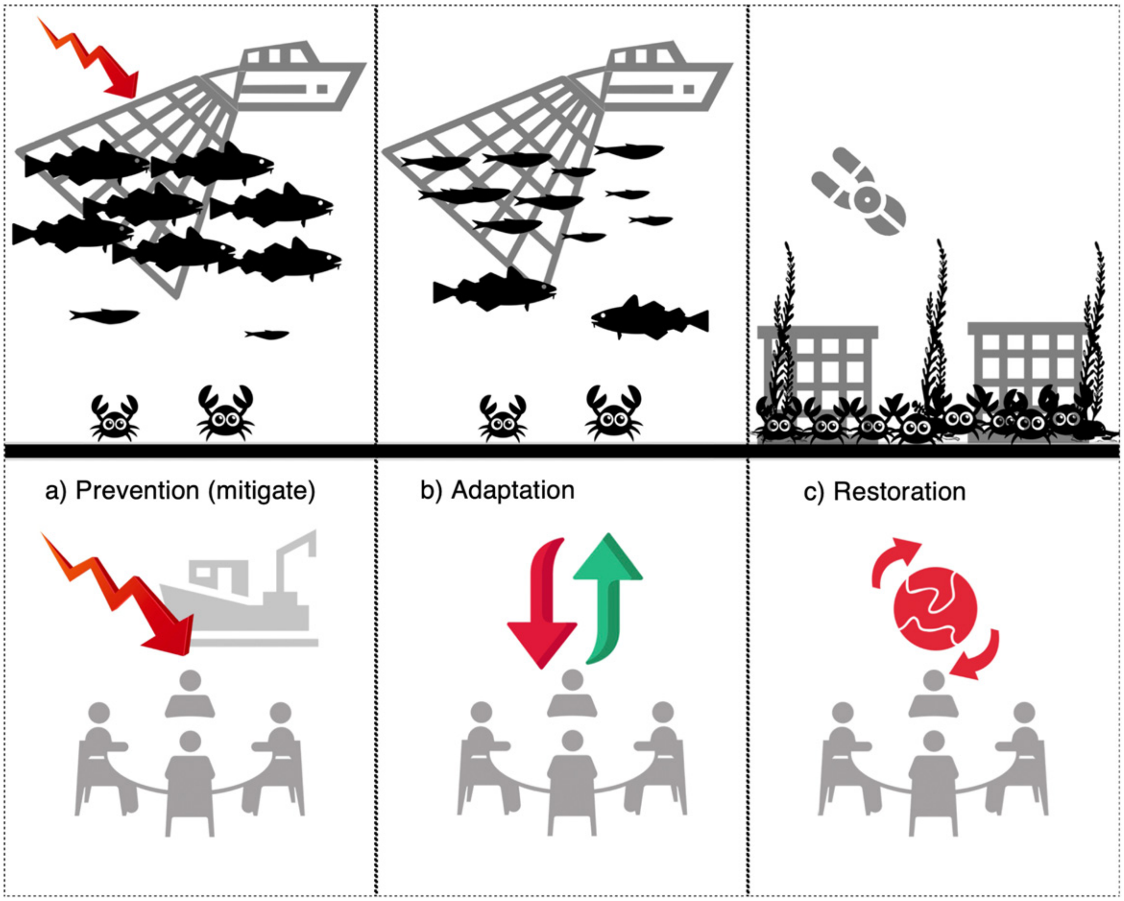

Fig. 8 The three management options discussed in this chapter: a) prevention and early detection, b) adaptation, c) restoration. In the prevention case a driver, in this case, fishing effort on demersal fishes has to be reduced in time in order to avoid the tipping point. b) If prevention does not work, and the tipping point is crossed, the fishing industry should adapt (if management allows it) and for instance change fishing target (here forage fish). c) However, if adaptation is done without following management principles, the system could switch again to a new state (case c, dominated by crustaceans). In this case restoration, by actively removing crustaceans and algae would be the only option to try to recover the system.

safely deployed. Unfortunately, such clear understanding is very uncommon. Thus, in order to be able to include regime shift considerations into management longer monitoring time series need to be collected and studies of methods application to detect past or present tipping points need to be conducted. Only in this way, by increasing knowledge about drivers, mechanisms, modelling approaches and systems, it will be possible to include such dynamics into management.

Conclusions

Regime shifts are common in marine coastal ecosystems all over the world due to the strong anthropogenic impacts these systems are exposed to. These drastic changes, being extremely difficult to revert, can be a huge thread to the ecosystem services provided to the human population and thus to the full socio-ecological system. However, we showed several examples, on how these changes can be either anticipated by monitoring the ecosystem, be it the change of a known indicator, or a statistical change in a time series, i.e. early warning signals (**Fig. 8**(a)). They can also, to some extent, be reverted, and we showed that reversal is very much dependent on the spatial and temporal scale in which we investigate and on the system. Local, smaller scale shifts in benthic systems, like for example in the kelp forests, are often easier returned to a previous state, than an open water system (**Fig. 8**(c)). Finally, we show the other alternative on how management can react to a regime shift, by providing flexible and open approaches to the users of the sea, so that they are able to adapt to the reorganized system (**Fig. 8**(b)). This was for example clearly shown in the cod-lobster example in Eastern Canada, even if, also adaptation can come with some delays and losses.

We clearly show that regime shifts are ubiquitous. However, the usage of methods to detect them before or even after they occurred is often not straightforward and the methods present some limitations. Because of that, management is still not prepared to tackle the problem of regime shifts and implement this concept. Nonetheless, we showed that by using a multi-step approach and a variety of methods as an ensemble, it is possible to detect regime shifts and to estimate the resilience of a system. Estimating the resilience can be essential in identifying early the pressures that need to be reduced or the system components that need rebuilding in order to prevent a regime shift. It is fundamental to continue the research on these methodologies, in order to be able to incorporate them into management plans, since under global climate change, regime shift dynamics will occur more often. Only in this way, by advancing science and providing tools for management we can assure that we will sustainably exploit our resources and maintain a healthy planet.

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