REPORT

ACIDIFICATION OF SWEDISH SEAS IN A CHANGING ENVIRONMENT: CAUSES, CONSEQUENCES, AND RESPONSES

– AN INTERDISCIPLINARY REVIEW OF CURRENT KNOWLEDGE, KNOWLEDGE GAPS, AND IMPLEMENTATION NEEDS

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SUMMARY

Increasing temperatures, changing weather patterns, shrinking ice-sheets, and shifting species distributions all bear witness to global change. These changes have been most keenly observed in terrestrial systems, however the world's oceans are also changing, albeit less obviously: ocean temperatures are rising; marine species ranges are shifting poleward; and ocean pH is falling (a phenomenon known as ocean acidification[1]). The latest research indicates strongly that the root cause of these changes lays in human policies and behaviours – notably market failures, governance failures, and spurious incentives – that have driven our dependence on fossil fuels, and elevated CO_2 concentrations in the atmosphere. The consequences for natural systems of continuing "business as usual" will be increased rates of ocean warming and acidification that will severely challenge many marine species, and cause substantial shifts in marine ecosystems and the services they provide. How we, as a society, respond will not only depend on the economic, societal and political consequences of these changes, but also on the strategies we identify to mitigate, and adapt to, them.

Here we detail the current state of knowledge of the causes, consequences, and potential responses to ocean acidification in Swedish coastal seas under global change, with a focus on ocean acidification. We note the particular problems – and potential interim solutions – presented by coastal seas, and identify key knowledge gaps, and implementation needs. Among our key findings we note that:

Causes: Current knowledge regarding the extent and impacts of market failures, and of governance failures, in relation to ocean acidification is extremely limited. *Improved knowledge of these issues is essential in order to identify, justify, and prioritize actions to address ocean acidification both nationally and internationally. Lack of relevant incentives and financial support for such research needs to be addressed.*

Consequences: Our current understanding in several key areas is critically lacking: i) how biogeochemical processes combine to impact the development of ocean acidification in Swedish coastal waters; ii) the combined effects of ocean acidification and other key stressors on Swedish coastal ecosystems; iii) the resilience of Swedish coastal ecosystems to near-future change, and the impacts this will have on socially and economically important ecosystem services. Collectively, these knowledge gaps preclude our ability to project likely development, impacts, and societal costs of ocean acidification in Swedish coastal ecosystems. *There is an urgent need to expand research efforts to develop models, conduct long-term experiments, and undertake environmental economic analyses to inform and hone projections of the impacts of ocean acidification in Swedish coastal seas and Swedish society.*

[1] The term "ocean acidification" is used to describe the shift in the acid-alkaline balance of seawater toward more acidic (less alkaline) values. It is caused by increasing levels of CO_2 in seawater (see below).

Responses: Understanding of public attitudes, of relevant mitigation structures, of strategies and policies that can foster adaptive responses to ocean acidification, and of the factors driving public support for those strategies and policies in Sweden, is almost completely lacking. *There is an urgent need to build a schematic understanding of how different information tools, policy instruments, directives, treaties, and laws link to the problem of ocean acidification* per se*, and to develop tools, instruments, and strategies that will maximise the effectiveness of responses.*

This science and implementation needs document is a contribution to Future Earth. It provides a network through which Swedish researchers can collaborate in the regional and global networks of Future Earth. The network will connect with the global research projects of Future Earth in ocean acidification (e.g. SOLAS, IMBER, PAGES, PECS) and provide a nationally coordinated effort within the Future Earth Ocean Sustainability Knowledge Action Network.

1. INTRODUCTION

Globally increasing emissions of anthropogenic carbon dioxide (CO_2), are causing warming of the oceans, melting of sea ice, glaciers and ice sheets, as well as ocean acidification[2]. At a local scale, these processes are being felt as rising sea levels, increased precipitation, reduced salinity, and increased flooding, coastal erosion, and flow of organic and inorganic matter into coastal waters. All of these add to the direct and indirect effects of ocean acidification in different ways. Anthropogenic ocean acidification occurs when atmospheric CO_2 concentration, expressed as partial pressure of CO_2 (or pCO_2), is greater than that in the oceans. The resulting flux of CO_2 from the atmosphere into the oceans causes a decrease in ocean pH. The extent to which this happens depends on level of pCO_2, the existing pH, and (to a small extent) the temperature. Additional, natural, pH fluctuations occur through biological processes, especially where biological activity is high such as when runoff and coastal erosion cause increased input of terrestrial organic matter and subsequent decay. The individual consequences of these varied stressors on marine life are relatively well

Figure 1. Map of the Baltic Sea System (Skagerrak, Kattegatt, Baltic Proper, Bothnian Sea, Bothnian Bay). We use the term "Baltic" or "Baltic system" to refer to the entire region from the Skagerrak to the Bothnian Bay. (Modified from Rönnbäck et al., 2007)

[2] Since the onset of industrialisation, global atmospheric CO_2 levels have increased by ~45% (from ~270 to today's 400 µatm), causing seawater pH to decrease by about 0.1 units. pH is expected to decrease by another 0.3 units by 2100. Because of the logarithmic nature of the pH scale, these changes equate to increases of ~30% and ~100% (respectively) in the hydrogen ion content of seawater (Box 1).

understood. However, we know little about the effects of combinations of these stressors on single species, and far less about their effects on entire coastal ecosystems, communities, society, and economics.

Ocean acidification is identified in the United Nations' Sustainable Development Goal 14.3 "*Minimize and address the impacts of ocean acidification, including through enhanced scientific cooperation at all levels"* [3], and has been classified as one of the nine planetary boundaries of importance for regulating the stability of the Earth system (Rockström et al., 2009). It is a global issue because it is caused by rising amounts of CO_2 in the atmosphere. However, although the atmosphere is almost perfectly mixed, for a number of reasons the degree and effects of ocean acidification are geographically heterogeneous (Steffen et al., 2015).

Here we address the topic of ocean acidification from a Swedish perspective:

- The Swedish coastline of more than 3 000 km encompasses one of the world's largest permanent salinity gradients (Fig. 1) from the high salinity Skagerrak shores of Bohuslän (salinity 30) [4] to the shallow, almost freshwater archipelagos of the northern Bothnian Bay (salinity 3). This gradient not only contains multiple different ecosystems, but also creates differences in seawater chemistry that alter the process, and effects of, ocean acidification. Hence ocean acidification will impact different parts of the Swedish coast very differently.
- Although Sweden is a large country, half of its population lives within 10 km of the coast (SCB, 2014). This intimate connection with the sea is not only vital for human well-being, but also for the country's economy. Marine and maritime sectors include over 5000 companies and are worth more than 95 billion SEK annually (Vinnova, 2013). Activities within these sectors range from international export/import agencies and port authorities to local fishermen and tourism. They collectively support over 200,000 jobs, creating a vital source of income for large ports and cities, and especially for small coastal communities, many of which depend on income from the sea. Although shipping itself is unlikely to be directly impacted by ocean acidification, it can impact ocean acidification locally through release of acidic water (see below). Economic impacts of acidification on other marine and maritime activities in Sweden could be substantial.
- Like many nations, Sweden has limited capacity to influence the global emissions of CO_2 that cause ocean acidification. Nonetheless, Swedish policy, law, and regulations influence the intensity of other stressors (e.g. eutrophication, pollution, fishing) that combine with ocean acidification to impact Swedish coastal seas. Consequently, it is of value to discuss policy options for Sweden, not least because these may be used as a model for other countries in a similar situation.

[3] Ocean acidification is not mentioned in the Swedish National Environmental Objectives, but is relevant to Objective 3 "Only natural acidification" and Objective 10 "A balanced marine environment".

[4] strictly salinity has no units and therefore we refer to salinity throughout this document simply as a number

There is a general deep lack of understanding of the causes, impacts and policy options in Sweden in relation to ocean acidification. This lack of understanding is critical because sustainable development of key aspects of Sweden's marine and maritime sectors depends on the ability to project – and adapt to – the likely effects of these changes on the marine environment and, thereby, on important ecosystem services provided by Swedish coastal seas, (e.g. fisheries, coastal protection, nutrient recycling, recreation, and tourism). Today, Sweden's coastal environment is facing challenges that threaten its sustainability, and therefore also threaten coastal businesses and communities. At the same time, policy measures and incentive structures to address these challenges are largely lacking – or at best not fit for purpose. In addition, with the notable exception of commercial fishing, we have very poor understanding of the economic value of the multiple ecosystem services that Swedish coastal waters provide. Lastly, the effects of ocean acidification in other parts of the world are likely to impact Sweden indirectly. For example, the impacts of ocean acidification on international markets for goods and services from marine systems in the rest of the world may influence market prices in Sweden. This will have economic and social impacts for consumers and producers throughout Sweden, not only close to the coasts.

Despite its potential negative implications for nature and society, the acidification of our coastal oceans is seldom highlighted in media and in public debate in Sweden. Nor has it attracted significant attention within national environmental politics. This absence of concern is also very much reflected in the environmental social science literature, as substantiated by Armstrong et al. (2012) who claim that ocean acidification has generated very few economic or social science studies in any country, despite the fact that the few studies that *have* been conducted anticipate negative impacts on fisheries (Cooley and Doney 2009; Narita, et al. 2016), coral habitats (Brander et al. 2009), and a general reduction in marine ecosystem services (Turley et al. 2010).

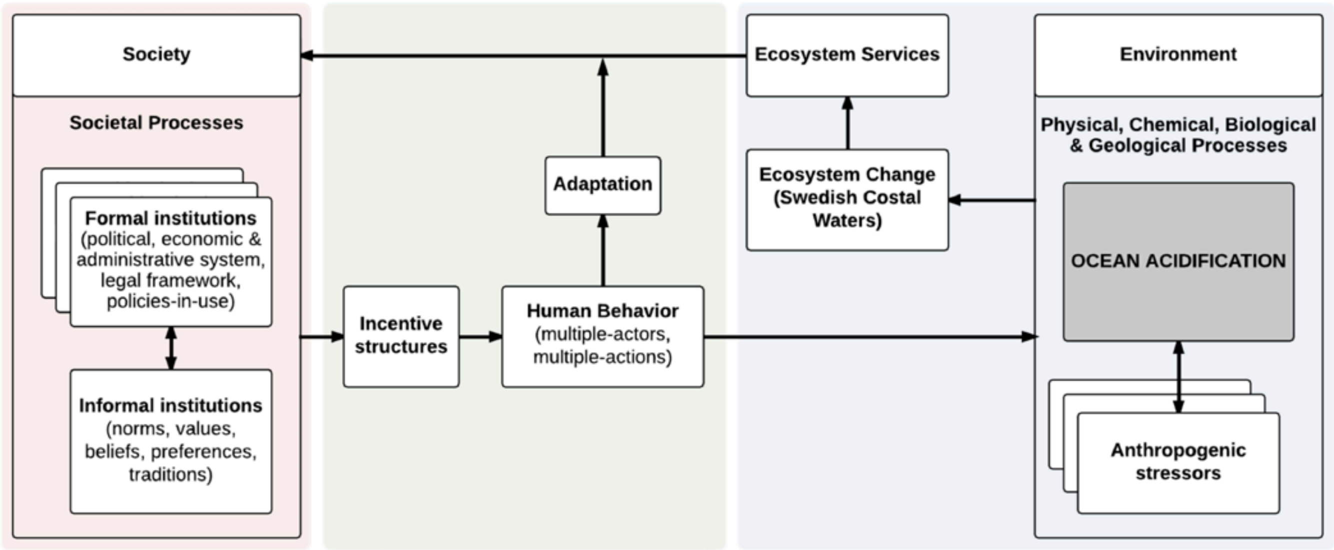


Figure 2. Schematic representation of how ocean acidification interacts with the relationships, processes, and concepts that link society and the natural environment. Ocean acidification originates in the very left of the figure where existing formal and informal institutions create incentive structures that frame and limit the actions of individuals. Those actions are also influenced by individuals' preferences and tastes. When individuals take some of these actions, CO_2 is released to the atmosphere, (for example by driving cars, heating homes, air-travel, etc.), causing ocean acidification (right hand side of the figure). This, in turn, causes ecosystem change in Swedish coastal systems (in combination with other anthropogenic stressors, such as eutrophication, pollution, fishing pressure), and hence influences the services that these ecosystems produce (right hand side). These changes in ecosystem services feedback to society where they affect the individual actors. Potential societal responses to ocean acidification could: i) alter the institutional frameworks that limit actions that trigger acidification (mitigation); ii) smooth the feedback from ocean acidification to society (adaptation); and/or iii) produce a total transformation of the way society interacts with the environment (transformation).

This document outlines the current state of knowledge related to ocean acidification in a Swedish context. In particular the paper provides a holistic interdisciplinary overview of:

1) the primary causes of anthropogenic ocean acidification, with roots in malfunctioning social interactions;
2) the mechanisms underlying ocean acidification, its interactions with other coastal stressors, and its impacts on marine species and ecosystems; and
3) potential societal and political responses to ocean acidification.

Throughout, we use the abbreviation "OA" to refer to ocean acidification, the term "Baltic" or "Baltic system" to refer to all Swedish coastal waters from the Skagerrak to the Bothnian Bay, and the relevant regional terms for the different components of the Baltic system (Fig. 1).

2. CAUSES OF OCEAN ACIDIFICATION

Typically, environmental problems in general and OA in particular can be attributed to two kinds of institutional failures: market failure and governance failure. Here, we first explore the market failure associated with OA, and then deal with governance failures. We then go on to explain how these failures provide spurious incentives that, together with preferences and tastes, steer human actions toward producing CO_2 and hence causing ocean acidification.

2.1 Market failure

Markets for goods and services are key institutions that drive people's actions. In many cases, market activities influence OA because they lead to the release of CO_2 to the atmosphere either through burning fossil fuels or through changes in land-use. Local markets steer, for example, public transport within a city, or domestic heating infrastructure, and thus influence the extent to which these activities generate CO_2 through burning of fossil fuels. By providing places of exchange for goods such as meat, fish, dairy products, or wood products, national and European Union (EU) markets influence the demand for these goods and hence the way they are produced (i.e. use of energy, land-use), which also impacts the generation of CO_2. At the international level, global markets for fossil fuels and air transportation generate substantial direct CO_2 emissions. Many goods and services traded in markets also generate CO_2 as a by-product and thus contribute indirectly to increasing levels of OA.

In theory, markets with perfect competition could deliver – and allocate in time and space – the amount of ocean acidification "optimally" for the long-term global well-being of human society. The level of OA would be "optimal" in the sense that any other level or allocation of OA could only improve the situation for one person if someone else was made worse off (Arrow, 1951; Arrow and Debreu, 1954). There are, however, many reasons why markets are almost never characterised by perfect competition and thus do not lead to the efficient provision and allocation of resources in general, and for ocean acidification in particular. For example: market transactions often incur extra costs; information about the relevant characteristics of the good exchanged is not perfect (i.e. negative externalities); some actors have market power and can influence the price; entry and exit from the market is not easy; or property rights to the good are not well-defined, in particular with regard to future generations. These general properties hold true for all goods and services including those that generate CO_2 and ocean acidification. Hence economic theory would suggest that levels of OA greater than that which is optimal for society are essentially due to multiple kinds of market failure[5]. Four kinds of market failure seem particularly pervasive in the case of ocean acidification:

[5] Market failures (Bator, 1958) are often linked to time-inconsistent preferences (Hoch and Loewenstein, 1991), information asymmetries (Stiglitz 1998), non-competitive markets (Tirole, 1988), principal–agent problems (Hart and Holmstrom, 1087), externalities (Laffont, 2008), or public goods (Baumol and Oates, 1988).

a) *Negative externality* (Fig. 3a). OA has a clear nature of negative externality: multiple economic activities generate increased levels of CO_2 in the atmosphere, which increase OA. Property rights to the seas are often ill-defined (especially outside the exclusive economic zone) and emissions of CO_2 have impacts far away from their sources. Hence activities that emit CO_2 do not account for the true cost that these emissions generate. Even when they do account for the true costs of carbon on global warming – such as the substantial set of carbon taxes in Sweden – these must also be globally coordinated, address all sources of carbon, and/or only account for both warming and acidification effects of carbon, (Turley and Gattuso, 2012; Rodrigues, 2016).

b) *Limited information on costs* (Fig. 3b). Information about the impacts of OA, and the processes that influence it (in particular the links between climate change and OA), is limited, and consequently OA-relevant decision-making must deal with substantial uncertainty (Polasky et al., 2011). In particular Armstrong et al. (2012) identify five limitations of knowledge that complicate the economic evaluation of the impacts of OA regarding: i) the effects of ocean acidification in the natural environment; ii) how ocean acidification affects/will affect ecosystem services from the sea; and iii) economic values of those ecosystem services. In addition, there are: iv) methodological limitations with regard to economic valuation of such services; and v) only scant knowledge of human preferences for services from the ocean. Brander et al. (2014) refer to knowledge gaps instead with respect to: i) understanding the relation between changes in the marine environment and socio-economic impacts; ii) the ecosystem services that have been assessed; iii) the distribution of impacts; and iv) the vulnerability of different populations.

c) *Global public good* (Fig. 3c). The capacity of the oceans to buffer the effects of ocean acidification (see Section 3 below) can be characterized as a global public

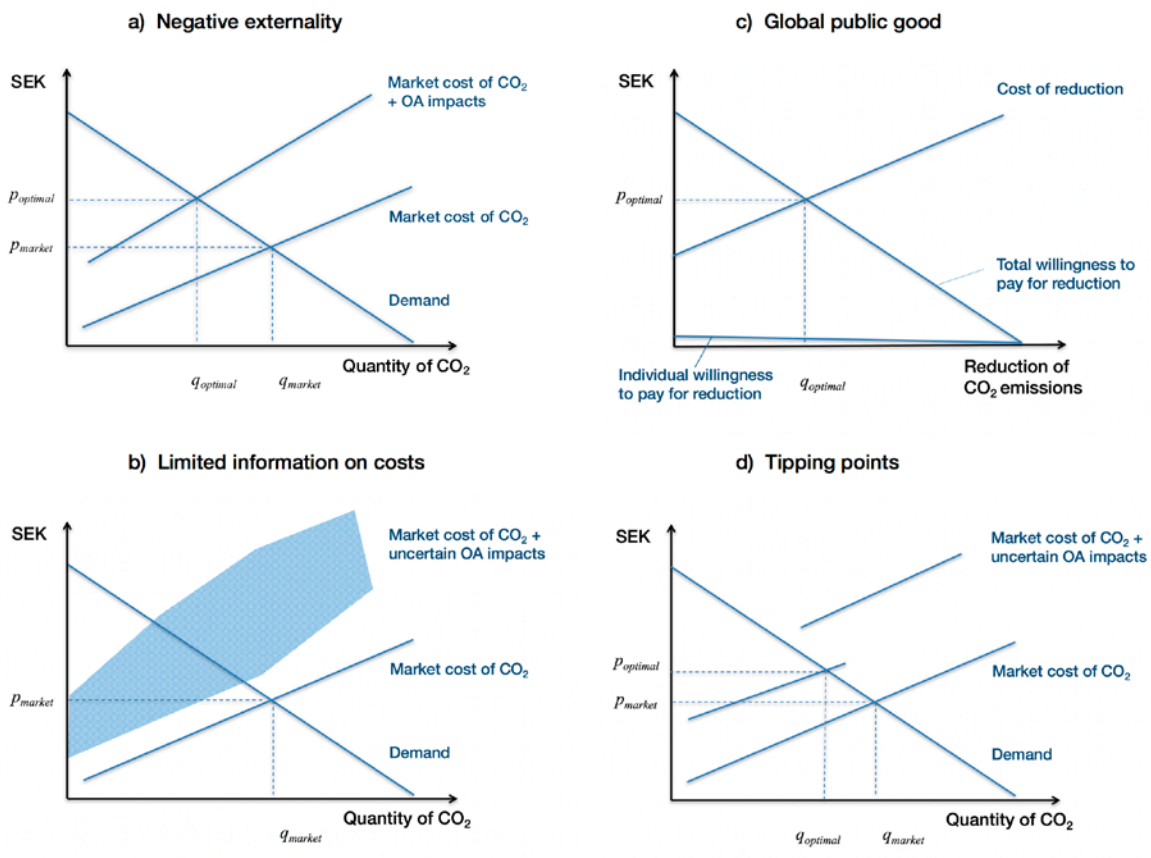


Figure 3. Four market failures causing excessive OA: a) the market does not account for all the cost associated with emitting CO_2; b) there is substantial lack of information about the impacts of OA; c) reducing CO_2 emissions is a public good so individuals will not be willing to pay so much even though collectively they would benefit, hence the market will not reduce emissions sufficiently; d) "tipping points" may occur, such that multiple equilibria are possible for large levels of demand and the system may reach a suboptimal equilibrium.

p_{market} and q_{market} denote respectively the price and quantity that will occur spontaneously on the market while $p_{optimal}$ and $q_{optimal}$ denote the respectively the price and quantity that would be optimal.

good, in the sense that any efforts by an individual to buffer the effects of OA automatically benefit other individuals equally, even at a global level. (In practice, OA is a global "public bad" as others cannot be effectively excluded from being impacted, and impact on one individual does not reduce impacts on others). Decentralized decision-making in markets will generally lead to under-provision of public goods (or over-provision of "public bads"), since the parties generating the public good (bad) do not account for the positive (negative) effects imposed on others. This dimension of the problem is particularly complex because causes and impacts span the entire planet. Hence regulating the provision of OA as a public bad cannot be achieved unless countries co-operate to implement the agreed upon policy (Fig. 3c). A particularly difficult aspect of such regulation is that it would require lump-sum financial transfers between individuals, which is a difficult option within a nation state, and even more doubtful in an international context (Sandmo, 2003).

d) *Tipping points* (Fig. 3d). Ecosystem responses to OA may be rapid, and at the same time influence evolutionary processes (Sunday et al., 2011). Hence, OA will involve slow and fast processes, which may trigger abrupt change in marine ecosystems beyond some particular threshold level of ocean acidification – so-called "regime shifts" (Hughes et al., 2013a,b; Rocha et al., 2015). These characteristics may therefore generate time-inconsistent preferences: preferences about future choices may differ at different points in time because history matters. This also implies that the effects of current levels of OA are likely to impact current and future generations differently.

In addition to these traditional types of market failure, another type of institutional problem, i.e., governance failure, can be particularly pervasive for OA. Hence, substantial justification for the problems caused by OA in Sweden, and elsewhere, can be linked to failures of formal institutions (market and government) to address the issue properly. Addressing these failures is likely to be a substantial part of successful societal responses to ocean acidification.

2.2 Governance failure

Because OA intersects with a number of other drivers and outcomes, including climate change, marine biodiversity, and food security, the ecological and social repercussions of OA are embedded in a highly diffuse and complex institutional setting. As explored by Galaz et al. (2011), a rich flora of actors and international institutions act in this problem domain, and poor political, administrative, and other institutional arrangements, means progress is slow. A key question here is the extent to which – and how – political regimes and institutions facilitate provision of public goods. Democratic institutions tend to lead to more public good provision than non-democratic alternatives (Acemoglu and Robinson, 2006; Sen, 1999; Bueno de Mesquita, 2003; Gandhi and Przeworski, 2006; Lake and Baum, 2001; McGuire and Olson, 1996), and therefore Swedish institutions could provide good examples with which to improve prospects to cope with OA internationally.

Nonetheless, there are a number of reasons why democratic institutions fail to provide public goods and avoid public bads. The strong focus of politicians on re-election doesn't always benefit the public interest (Besley and Coate, 1998), as rational, vote-maximizing politicians are unlikely to risk unpopularity by introducing policies that might be received poorly by the electorate (Page and Shapiro, 1983; Stimson et al., 1995; Burstein, 2003). Thus, the limits of public support also constitute the very tangible limits of the democratic policymaking process (Skocpol, 1997; Williams and Edy, 1999; Wallner, 2009). Similarly, elected leaders often work with short time horizons (Haggard, 1991; Sterner et al., 2006; Keefer, 2007). In contrast, the avoidance of public bads – including mitigation of, and adaptation to, OA – is a truly long-term undertaking. Thus, in the context of OA, it can be argued that poor and dysfunctional government policy intervention related to (e.g.) energy issues and climate change may even worsen OA because it provides incentives that are poorly designed to address OA specifically, and thereby promotes activities that generate more acidification (Anthoff and Hahn, 2010; Helm, 2010).

While there is currently no overarching international, nor national, regime explicitly addressing ocean acidification, a few attempts have been documented in which state actors respond and prepare for its repercussions. Rosen and Olsson (2012) and Fidelman et al. (2012) for example, analyse how states around the Coral Triangle (located in the tropical marine waters of Indonesia, Malaysia, Papua New Guinea, Philippines, and others) collaborate to improve food security in the region and protect marine ecosystems at risk. In addition, several international initiatives (led by NGOs and UN agencies) exist to create awareness around the problem, synthesize and disseminate scientific information, and try to influence high level negotiation arenas such as the UNFCCC, albeit with limited success (see Galaz et al., 2011). Currently, however, efforts to address OA can be described as a "non-regime", since the policies surrounding OA are "characterized by the absence of multilateral agreements for policy coordination among states" (Dimitrov et al., 2007: 231).

2.3 Incentives and Actions

Many types of actor are currently contributing to OA, from local to global level, and in sectors as diverse as shipping to energy production. These actors affect OA either directly *via* CO_2 emissions (e.g. transport, energy production, heating, agriculture) or indirectly *via* interacting stressors (e.g. eutrophication, waste-water discharge, pollution; see Section 3 for more detail). The main sectors of activity that generate CO_2 (Fig. 4) include: land use change (~3.5 Gt CO_2 per year); deforestation (increases albedo, but releases CO_2, and reduces CO_2 capture creating changes in the "land sink" – 11.5 Gt CO_2 per year) and; burning fossil fuel and other industries (~ 34 Gt CO_2 per year; all data means for 2006-2015; Fig. 4).

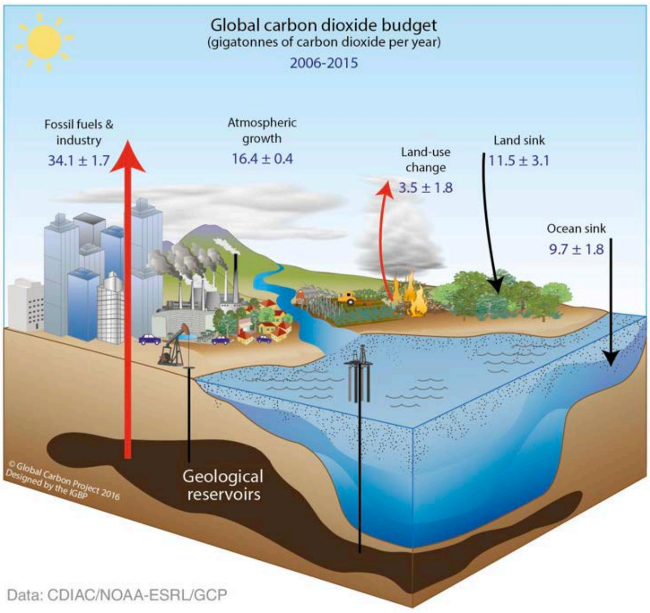


Figure 4. Global CO_2 budget for 2006–2015. Approximately 28% of the CO_2 released from fossil-fuels is absorbed by the oceans, leading to ocean acidification. Units are Gt CO_2 / year.

Existing institutions and governance systems comprise a multitude of policy instruments created to address problems other than OA, but which nonetheless generate incentives that constrain human actions that influence OA (Fig. 2). In part, this is because the diverse norms and preferences that characterise human actors restrict the actions they take. Hence, while formal and informal institutions constitute social restrictions to human actions, preferences and tastes constitute individual restrictions. Both kinds of restrictions provide incentives that steer human action in a direction commonly detrimental to OA (Fig. 2).

The great multitude of different actors generating CO_2 mean that ocean acidification can be characterized as non-point source pollution. However, unlike most non-point source pollutants, which are only measurable after they have entered the environment (making polluting sources costly or impossible to identify; Kampas and White, 2004), CO_2 production can be identified and measured readily at source. Nonetheless, these multitudes of small sources are more burdensome to detect, which provides less incentive to take action against them. Moreover, individual actors can typically only address a tiny part of the problem, which also reduces the incentives for each individual to tackle the problem.

In addition to direct or indirect actions that affect OA, some actors also contribute at multiple scales to make marine ecosystems more vulnerable to the impacts of OA. This is a broader context of issues that we do not discuss here (but see section 3.3).

2.4 Knowledge Gaps and Implementation Needs[6]

i) Current knowledge about the extent and impact of different market failures in relation to OA and how these interact with each other is extremely limited, especially in a Swedish context. Improved knowledge of these issues is essential in order to identify, justify and prioritize actions to address OA, both nationally and internationally:

Identify the types of market failure relevant to OA and project their consequences, focussing on those that have strong effects on OA.

Quantify the importance of different types of market failure, and interactions among them.

Identify the magnitude of market failures relevant to OA in Sweden, and globally.

ii) Although market failure is a key issue, there are currently few incentives to conduct research into the economic aspects of OA, and financial support for such research is lacking.

Improve financial incentives to support projects targeting the economic aspects of OA.

iii) Current knowledge about the extent and impact of different governance failures on OA is scant. For example, little is currently known about the appropriateness of various existing policies, legal provisions, mechanisms and administrative systems that address either the main cause of OA (i.e. increasing atmospheric CO_2), or the additional stressors that may influence resilience and adaptation. This lack of knowledge prevents informed assessment of the current institutional framework within which OA arises, and

[6] Knowledge Gaps are given in Roman text, Implementation Needs in *italics*

subsequently the design of additional or modified measures to deal with OA. Therefore research should:

Analyse the coherency and/or conflicts among and within the diverse components of the national, EU, and international legal and administrative systems relevant for OA causing activities. Specifically the measures, and at what level, i.e. local, national, regional, international.

Analyse contradictions, overlaps, and gaps in existing institutional arrangements that cause OA, as well as how these challenges can be amended, focusing on: how feasible different measures may be in light of how well they complement existing legal structures at the relevant level; and the degree of readiness to pursue the required changes among concerned actors.

3. CONSEQUENCES OF OCEAN ACIDIFICATION

In addition to drivers such as OA and warming that are changing marine systems on a global scale, Sweden's coastal seas are also subject to local drivers such as eutrophication, runoff, pollution, tourism, fishing, and aquaculture. Local differences in the strength and timing of these drivers, and the natural heterogeneity of coastal seas, create far greater variability and change in coastal ecosystems than we see in the open ocean. Coastal ecosystems of the Baltic have changed markedly during the past 40 years, and recent work has shown this is partly attributable to local and regional warming and freshening (Olsson et al., 2012), but also to eutrophication (Olsson et al., 2015). The importance of OA in these processes is largely unknown, not least because most biologists were unaware of its potential importance until relatively recently (Gattuso and Hansson, 2011). This Section summarises available evidence for the mechanisms and consequences of ocean acidification (OA) and its interactions with other common drivers in Swedish coastal seas.

3.1 The Geological Context

The geological record provides an important environmental archive of past ocean acidification events and how past marine ecosystems have responded to changes in pH and ocean biogeochemistry. Several events have occurred during which ocean pH has decreased markedly (e.g. Hönisch et al., 2012; Table 1), however it is

Table 1 Past drivers of oceanic extinction events. Note that ocean acidification has been implicated in several major extinction events (see Harnik et al., 2012 for more detail).

Time period[b]	Drivers and Threats[c]						
	Acidification[d]	Anoxia[e]	Warming	Cooling	Habitat Loss[f]	Overexploitation	Pollution
Ordovician-Silurian (~444 Ma)		○	○	●	●		
Late Devonian* (Frasnian-Famennian; ~374 Ma)		●	○	●	●		
End Permian* (~251 Ma)	●	●	●	○	●		
Early Triassic (~245 Ma)	○	●	○	○	○		
Triassic-Jurassic* (~202 Ma)	●		●				
Early Jurassic* (Pliensbachian-Toarcian; ~183 Ma)	●	●	●	●			
Aptian-Albian (~112 Ma)	●	●		○	○		
Cenomanian–Turonian (~93.5 Ma)	●	●	○				
Cretaceous–Paleogene (~65.5 Ma)	●		○	●	○		
Paleocene–Eocene Thermal Maximum* (~56 Ma)	●	○	●				
Eocene–Oligocene (~34 Ma)				●	○		
Mid–Miocene Climatic Optimum (~14.7 Ma)			●		○		
Historical (~10 Ka)			○		●	●	○
Modern	●	●	●		●	●	●

[b] bold text indicates extinction events, asterisks indicate global reef crises

[c] closed circles indicate high confidence, open circles indicate lower confidence

[d] causes of acidification include volcanism, bolide impacts, and methane clathrates in the past, and fossil fuel use in the present

[e] causes of anoxia include warming, eutrophication and ocean stratification

[f] causes of habitat loss include sea-level fall in the past, and habitat degradation and coastal development in the present

important to note that today's changes in marine chemistry – especially the rate of change – are unprecedented in the last 300 million years. Consequently it is difficult to project potential impacts, not least because we cannot at present predict whether ecosystems will be able to acclimate and genetically adapt fast enough to retain resilience to OA.

Example I: The last deglaciation (~17 800 -11 600 years before present [BP])

During the last deglaciation, atmospheric CO_2 increased 30% (from 189 to 265 μatm), leading to a ~ 0.15 unit drop in sea surface pH (Hönisch and Hemming 2005). This change corresponded to a decrease of 0.002 units per 100 years, which is much slower than the present rate of ~ 0.1 units/100 years, and projected change of up to 0.3 units before the end of the century (see below). The pH shift during the last deglaciation led to a drop in shell weight of key planktonic organisms (phytoplankton and single-celled calcifying organisms called foraminifera; Barker and Elderfield, 2002; Beaufort et al., 2011), suggesting that these organisms were negatively impacted.

Example II: The Paleocene-Eocene Thermal Maximum ("PETM" ~56 million years BP)

The PETM involved the release of large amounts of carbon (>2000 Gt C) over a relatively short period (the initial event was ~5000 - 10 000 years). This period was characterised by a sharp increase in temperature followed by intense dissolution of calcium carbonate at the sea floor, and a decline in deep ocean pH of 0.3 units over ~5000 years (Penman et al., 2014). Simultaneously, half of the deep sea benthic foraminifera went extinct and many others reduced in size (Speijer et al., 2012; Winguth et al., 2012; Yamaguchi et al., 2012), and ecosystem productivity and dissolved oxygen concentrations both decreased markedly. Effects on higher marine organisms are unclear due to a lack of suitably preserved fossil specimens[7], however extensive anoxia and warming have been documented (e.g. Sluijs et al., 2014). Overall, the PETM lasted for ~200 000 years.

[7] Reliable fossil data for larger organisms, such as molluscs and echinoderms, are lacking due to their relative scarcity in the fossil record (compared to plankton) and the (geologically) short duration of the event.

3.2 Today's Environment

The Chemical Basis of Ocean Acidification

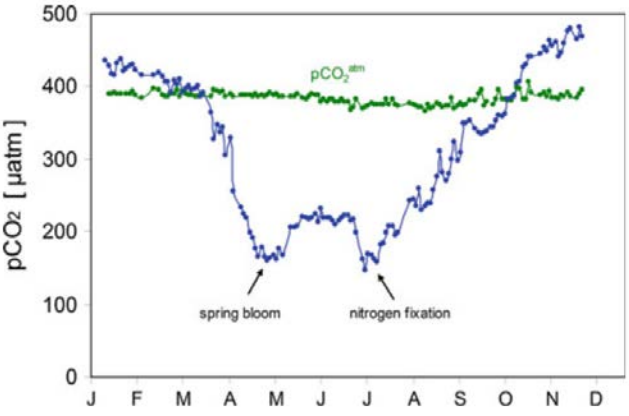


Figure 5. Seasonal changes in phytoplankton productivity absorb dissolved CO_2 during spring and summer (blue line), creating "draw down" relative to that in the overlying atmosphere (green line). Data from central Baltic Sea, taken from Schneider et al. (2015).

The surface ocean continuously exchanges CO_2 with the atmosphere. When atmospheric CO_2 concentration (typically expressed as partial pressure of CO_2, or pCO_2) is greater than that in the oceans, there is a flux of CO_2 from the atmosphere into the oceans (and *vice versa*). This flux is influenced by photosynthesis (more strictly, primary production), which reduces seawater pCO_2 in the summer, and by decomposition and biomineralisation causing the release of CO_2 in the winter. These processes create a seasonal cycle of pCO_2 both in the atmosphere and the ocean. As the atmosphere mixes much faster than the ocean the atmospheric signal is diluted and less pronounced: for instance the seasonal amplitude in atmospheric pCO_2 at the latitude of Sweden is ~ 10 µatm, while that in the coastal ocean can be several hundred µatm (Fig.5).

Box 1. Atmospheric CO_2 and the acidification of seawater

Unlike the other atmospheric gases, when CO_2 dissolves in seawater it reacts chemically with the water:

$$CO_2\,(aq) + H_2O \leftrightarrows H_2CO_3^- \leftrightarrows H^+ + HCO_3^- \leftrightarrows 2H^+ + CO_3^{2-}$$

The pH of the seawater determines which of these chemical species dominate. Higher pH drives the system farther to the right. At seawater pH's typical of Swedish coastal waters (7.5 – 8.5) the carbonate system is dominated by the HCO_3^- terms, and hence dissolving CO_2 in seawater leads to an increase in proton (H^+) concentration, and hence increased acidity. Dissolved inorganic carbon (DIC) buffers the dissolution of CO_2 in seawater by the CO_3^{2-} ion reacting with the CO_2. Because DIC and alkalinity decline from the Swedish west-coast, through the Baltic Proper and into the Bothnian Bay, the effects of increasing pCO_2 are greater in the Baltic than on the west coast, (see main text).

Since the onset of the industrial revolution, seasonally-averaged atmospheric pCO_2 has increased from around 280 µatm to > 400 µatm today, and consequently the average pCO_2 of the surface ocean has also increased. Seasonal fluctuations in pCO_2 are superimposed on this mean increase, leading to elevated extremes of pCO_2. This increasing pCO_2 causes a decrease in ocean pH, but the extent to which this happens depends not only on the pCO_2 itself, but also on other properties of the seawater, notably the existing pH, the buffering capacity (the "alkalinity") and (to a small extent) the temperature (see Box 1). The chemical buffer capacity, i.e. the change in pH for a given change in pCO_2, is determined by the total concentration of all the bases in the seawater, measured as total alkalinity. In most of the world's oceans alkalinity is closely correlated with salinity. Hence, buffer capacity typically decreases with decreasing salinity, with the result that the seasonal pH range is usually larger in low salinity coastal waters than in the more saline open ocean. For example, in the Bothnian

Bay the low salinity and alkalinity cause the seasonal pH range to exceed 1 pH unit (Fig.7c). However, in the Baltic Sea, this alkalinity:salinity relationship is complicated by extremely low salinities, large differences in local geology that influence the alkalinity independently of salinity, and strong seasonality in runoff (Box 2).

Box 2. Salinity and Alkalinity and pH in the Baltic

In the Baltic, salinity is determined by the combination of substantial runoff of freshwater together with limited exchange of seawater with the North Sea. In those parts of the Baltic system most distant from the North Sea, (Bothnian Bay and eastern Gulf of Finland), the salinity is below 3, rises to around 7 in the Baltic Proper, and then rises rapidly from ~8 in the southern Danish straits to ~15 in the Kattegat just a hundred km or so to the north. Salinity continues to rise to the north and west, reaching 30 in the western Skagerrak (Fig. 1). River runoff affects not only salinity, but also alkalinity – and hence local pH – because the alkalinity of the runoff itself varies depending on the geology of the drainage basin. Thus, the northern Bothnian Bay, which is largely surrounded by granite bedrock, has relatively low alkalinity and pH because the alkalinity of the runoff is low. In contrast, the limestone bedrock that characterises the watersheds flowing into the Gulf of Riga has very high alkalinity, and thus pH in the Gulf of Riga is much higher (Fig.6).

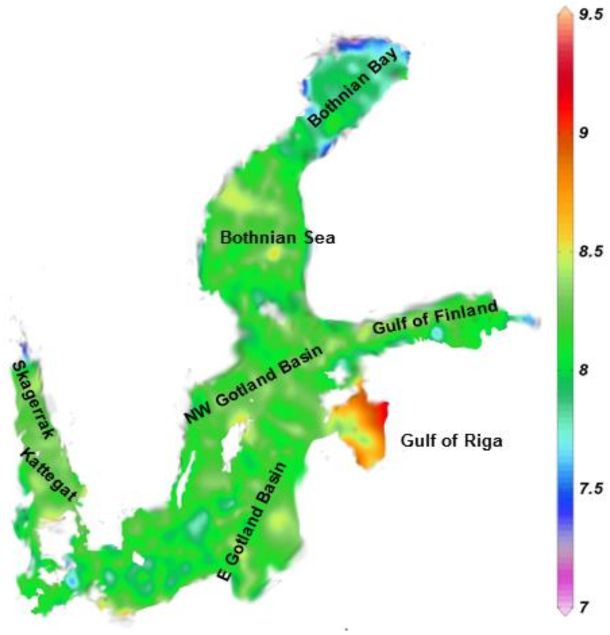
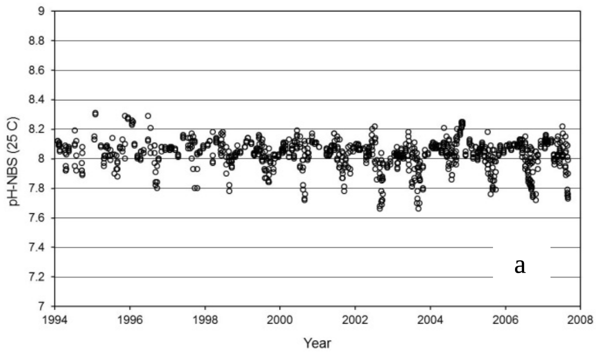
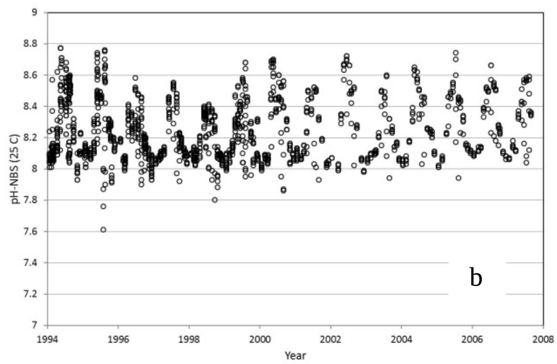


Figure 6. Long-term mean surface water (0-20m) pH in the Baltic Sea.

These large differences in alkalinity – notably between the Bothnian Bay and Gulf of Riga –cause substantial differences in average pH in these locations (Box 2, Fig. 6). However, alkalinities of the surface waters of the Baltic Proper are broadly similar to those of the Kattegat and Skagerrak. Consequently average pH of these waters are also generally similar (Fig.6) even though their salinities differ strongly.

Notwithstanding these large-scale patterns, small differences in alkalinity combine with differences in the extent of primary production to cause much greater seasonal variability in pH in the Baltic Proper compared to the Kattegatt / Skagerrak (Fig.7). This effect is amplified by the nutrient-rich conditions in the Baltic Proper that cause a stronger draw-down of CO_2 during the productive season, which results in higher pH during the summer (Fig.7b).





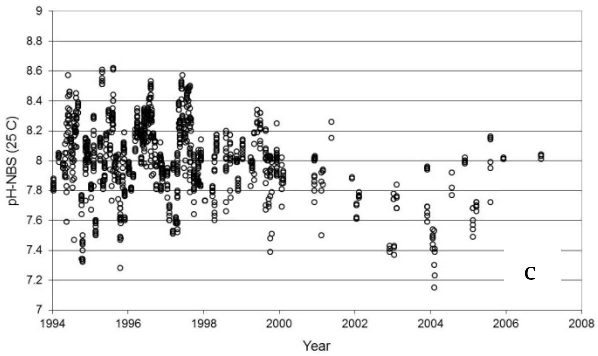


Figure 7. Time series of pH in the top 20 m of the Kattegat (a), Baltic Proper (b) and Bothnian Bay (c). Note that the observational frequency decreased substantially after 2000 in the Bothnian Sea.

Seasonal changes in CO_2 uptake and release also have implications for other important chemical processes in coastal seas. Photosynthesis by micro- and macro-algae, and marine plants such as seagrasses, consumes “acidic” CO_2 and hydrogen ions (H^+) (Box 3). This is the summer pH increase noted above. In winter, the decomposition of organic matter reverses this process, producing hydrogen ions and lowering the pH. This is the winter decrease in pH noted in Figure 5a. Primary production occurs close to the surface in the photic zone, but much of the decomposition occurs after this production has died and settled to the seafloor, and therefore decomposition normally occurs deep in the water column or at the sediment surface. In waters with limited exchange this can sometimes completely deplete the available oxygen, resulting in bottom waters with very low oxygen (“hypoxic”), or no oxygen (“anoxic”). This impacts the seawater chemistry in these waters resulting in increased solubility of some metals (notably manganese and iron), and in some cases the production of elemental sulphur. These processes all have implications for the pH of the seawater (Box 3).

These biogeochemical reactions are more pronounced in coastal seas than in the open ocean because of input from land; nutrient emissions from waste-water treatment and agricultural runoff have caused eutrophication in the Baltic Sea, which has led to increased areas of anoxia as well as marked shifts in plankton communities characterised by periodic blooms of cyanobacteria. The latter are a consequence of chemical reactions between iron(III) and phosphate, which in oxic sediments form precipitations that trap the phosphate, but which is dissolved and released when the sediment becomes anoxic and iron(III) is reduced to iron(II). Thus, in anoxic– typically bottom – waters phosphate is released back to the water column where, in time, it is mixed up to the surface where it stimulates primary productivity, leading to an increase in pH fluctuations (Fig. 5, and see below).

In addition to influencing alkalinity, river runoff also contains dissolved and particulate organic matter. Some of the particulate organic matter decays by microbial activity, adding to local deoxygenation and ocean acidification. This also holds for the dissolved fraction, but this fraction has further impact on pH as these dissolved molecules contain carboxyl groups (i.e. they are organic acids). Although uncertainty is large, climate projections generally suggest the flux of terrestrial organic matter to the coastal seas will increase as a result of increased precipitation, potentially adding to ocean acidification and deoxygenation. Model computations show increased precipitation over northern Sweden, particularly in the winter months (Christiansen et al., 2015).

Seasonal variation and methodological issues preclude reliable detection of trends in measurements of seawater pH in Swedish coastal seas. Recent modelling, however, projects that the combination of this variation with increasing anthropogenic atmospheric CO_2 will lead to greater pH variation and lower pH minima in the surface waters (Fig. 8). Importantly, although average pH in the scenario plotted in Figure 8 does not drop below 7.8 until the last decade of this century, winter minimum pH already begins to fall below this value by the year 2040. Thus, for species that are sensitive to low pH, the effects of acidification may be felt far sooner than predicted from the mean annual pH.

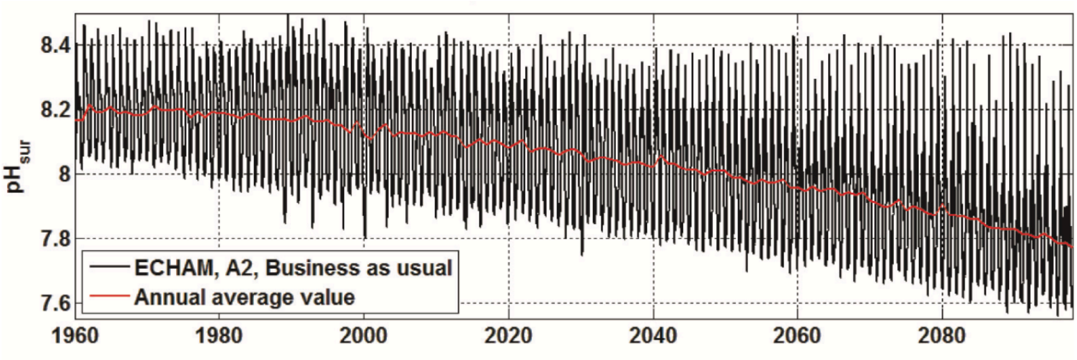


Figure 8. Daily pH values for surface water in the Eastern Gotland Basin projected from the ECHAM global climate model and the SRES A2 “business as usual” scenario. Note that summer maximum pH remains mostly constant until ~2090, whereas winter minimum pH declines almost linearly throughout the modelled period (from Omstedt et al., 2012).

Box 3 Chemical consequences of Ocean Acidification in the Baltic System

In the surface mixed layer of the oceans, the photosynthetic capture of light energy to combine CO_2, macro-nutrients (such as nitrate and phosphate), and micro-nutrients in the form of trace metals (Me^{2+}, such as iron(II)), to create organic matter and oxygen can be formulated as:

$$140\ CO_2 + 16\ H^+ + 16\ NO_3^- + HPO_4^- + Me^{2+} + 123\ H_2O \rightarrow$$

$$\rightarrow (CH_2O)_{91}(CH_2)_{16}(NHCH_2CO)_{16}(CHPO_4Me) + 172\ O_2 \qquad (1)$$

Decomposition of sedimenting organic matter in deeper water runs in the opposite direction, releasing CO_2 and H^+. In the following formulation this CO_2 release is illustrated by balancing it to the bicarbonate ion, HCO_3^-, the dominating form of dissolved inorganic carbon at typical seawater pH (Box 1).

$$(CH_2O)_{91}(CH_2)_{16}(NHCH_2CO)_{16}(CHPO_4Me) + 172\ O_2 + 17\ H_2O \rightarrow$$

$$\rightarrow 140\ HCO_3^- + 156\ H^+ + 16\ NO_3^- + HPO_4^- + Me^{2+} \qquad (2)$$

Thus decomposition produces hydrogen ions, and hence lowers pH. This reaction normally occurs deep in the water column or at the sediment surface. In waters with limited exchange the decomposition process (reaction 2) can sometimes completely deplete the available oxygen, resulting in strongly hypoxic or anoxic bottom water. Under these circumstances other "electron acceptors" are needed to replace oxygen in the decomposition process. The most energetically favourable electron acceptor after oxygen is nitrate, and hence in hypoxic and anoxic areas, decomposition leads to denitrification:

$$(CH_2O)_{91}(CH_2)_{16}(NHCH_2CO)_{16}C(MeHPO_4) + 112\ NO_3^- \rightarrow$$

$$\rightarrow 140\ HCO_3^- + 11\ H^+ + 56\ N_2 + 16\ NH_4^+ + H_2PO_4^- + M4^+ + 45\ H_2O \qquad (3)$$

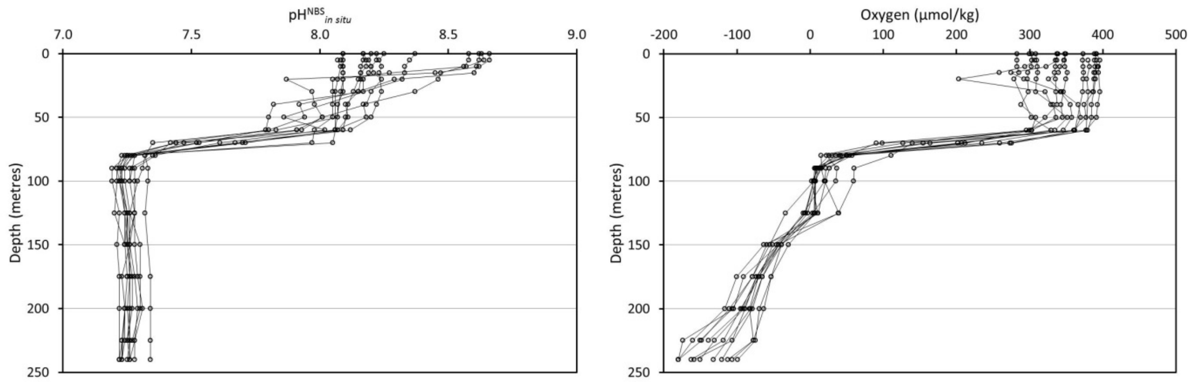
When comparing reactions (2) and (3) it can be seen that denitrification generates far fewer hydrogen ions per bicarbonate ion produced. If decomposition proceeds to deplete all the nitrate then other electron acceptors step in. In seawater these are (in order) manganese(IV), iron(III) and sulphate. When these are used as electron acceptors the following reactions (4 – 6) occur (here organic matter is simplified to "carbohydrates"; $CH_2O_{(org)}$):

$$CH_2O_{(org)} + 2\ Mn(IV)O_2 + 3\ H^+ \rightarrow HCO_3^- + 2\ H_2O + 2\ Mn(II)^{2+} \qquad (4)$$

$$CH_2O_{(org)} + 4\ Fe(III)OOH + 7\ H^+ \rightarrow HCO_3^- + 6\ H_2O + 4\ Fe(II)^{2+} \qquad (5)$$

$$CH_2O_{(org)} + 0.5\ SO_4^{2-} \rightarrow HCO_3^- + 0.5\ H^+ + 0.5\ HS^- \qquad (6)$$

These reactions have very different impacts on pH as both manganese and iron reduction consume H^+, whereas sulphate reduction produces H^+. An important consequence of this is that the pH in the sulphide bottom-water that occurs in the Baltic Proper has close to constant pH, even if the sulphide concentration increases with depth (Fig.9).



3.3 Interacting Anthropogenic Drivers in Swedish Coastal Seas

The multiple drivers that influence our coastal seas rarely operate in isolation. Historically in the Baltic warming has been accompanied by freshening. On shorter timescales saline inflows through the Danish Strait can increase deep-water oxygen concentration, and eutrophication enhances phytoplankton production which increases seasonal pH variation (see above). Interactions among drivers can have synergistic, additive, or antagonistic effects on marine species, depending on the timing and relative magnitude of the drivers. For marine organisms, the situation is further complicated by the fact that drivers have *direct* and *indirect* effects: as their name implies, direct effects operate directly on the organisms concerned, e.g. the effects of warming and freshening on growth rate, whereas *indirect* effects arise when drivers influence a given species (or process) indirectly by changing species interactions, e.g. freshening may decrease the growth rate of a competing species, or acidification may reduce the quality of a food resource. Indirect effects can be complex, and difficult to quantify, and consequently there are very few examples published, although it is clear that indirect effects can be at least as important as direct effects (e.g. direct and indirect effects of warming and acidification on benthic microalgae in seagrass beds of the Kattegatt; Alsterberg et al., 2013).

Shipping

Exhausts from the shipping industry contribute to acidification of surface waters through the release of acidic gases as SO_x and NO_x. While the International Maritime Organisation is working to reduce emissions of these gases to the atmosphere, the corresponding terrestrial emissions peaked some decades ago with the result that shipping is an increasingly important source of these acid depositions to the Baltic (Omstedt et al., 2015). Reducing the marine SO_x emissions to the air can be achieved either by burning low-sulphur fuel or by installing scrubber systems that absorb the SO_x in a counterflow of seawater spray. Open-loop scrubbers without any treatment of the resulting effluent transfer the resulting acid direct to surface waters. Modelling studies have shown that the effects are small on a basin scale (Turner et al., in preparation), but can match the acidification due to CO_2 in heavily trafficked areas close to major harbours (Stips et al., 2016).

Warming

In concert with rising global temperatures, the Baltic has warmed by 0.4 - 1.5°C over the last 150 years (Gustafsson et al., 2012). Recent projections for the end of this century show that sea surface temperatures in the Baltic system may rise by a further 2-3°C on average (Meier 2015), although there is substantial seasonal and regional variation (maximum warming in the northern Bothnian Sea in summer by ≤ 4.4 °C, and minimum warming of ~1.5°C in the Kattegat throughout most of the year; Meier, 2015[8]). Deeper waters are projected to warm more slowly (Meier, 2015).

This warming impacts marine species in several key ways. Chemical dissociation of H_2CO_3 to HCO_3^- and H^+ is temperature dependent such that pH declines with

[8] note that these projections exclude the Skagerrak

increasing temperature. However, warmer waters also carry less gas and hence CO_2 content declines with increasing temperature, with the net result that pH increases slightly as temperature rises. For most species in Swedish coastal waters, warmer temperatures also increase growth rates. For example, warming (in combination with freshening and eutrophication) is projected to increase biomass of summer phytoplankton blooms (Meier et al., 2012). For individual species, the consequences of warming will depend on species thermal tolerance norms (Boyd et al., 2013), and how rapidly these – and species distributions – change (see "Invasive Species" below). Warming is projected to favour some species of cyanobacteria, leading to prolonged blooms of these species (Neuman, 2010), and likely cause shift toward smaller zooplankton (Suikkanen et al., 2013). Acidification has similar effects (see below). These changes are likely to have negative impacts on filter-feeding animals in the benthos because cyanobacteria represent lower-quality food (Karlson et al., 2008), and warming (and acidification) have been shown to reduce the food value of phytoplankton for zooplankton (Dahlgren et al., 2011, Rossoll et al., 2012).

Freshening

Salinity is a key factor determining distributions of species in Swedish coastal waters (Bonsdorff, 2006). Biodiversity typically increases with increasing salinity from the Bothnian Sea and Baltic Proper, through the Danish Strait to the Skagerrak (Fig.10; Bleich et al., 2011).

Climate projections for this century indicate that the Baltic will freshen as a result of increased precipitation (Christiansen et al., 2015), and that this is likely to be greatest in the Belt-Seas around Denmark (by a salinity decrease of ~2), and lowest in the Bothnian Bay (0.5 - 1, Meier 2015). Importantly, because freshening is projected to be greater in deep water than in surface waters (Meier, 2015) stratification in the Baltic Proper is projected to decline, perhaps leading to more overturning and increased supply of oxygen-rich water to deep waters, which may in turn influence the chemical reactions outlined in Box 3.

As noted earlier, increased input of freshwater also increases input of organic carbon. This is a potentially strong driver in the northern Baltic (Bothnian Sea and Bay) where it may cause more energy to be cycled toward the microbial loop, possibly reducing energy transfer to higher trophic levels (Wikner and Andersson, 2012). Similar shifts toward the microbial loop have been observed in response to ocean acidification e.g. Endres et al., 2014, (see below). Freshening is also projected to push distribution limits of marine species out toward the North Sea (Bleich et al., 2011) – particularly key ecosystem-structuring species such as seagrasses and blue mussels (Viitasalo et al., 2015) – and to reduce densities of seagrass (Boström et al., 2004). Freshening is also likely to impact sexual reproduction in macroalgae (low salinity reduces fertilization success, Malm et al., 2001, Serrão et al., 1999). At the same time, ranges of predatory freshwater fish such as pike and perch are likely to expand (Olsson et al., 2006).

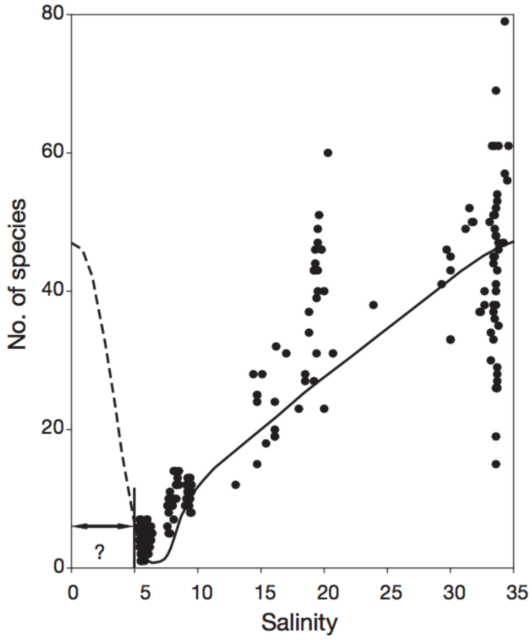


Figure 10. Number of species vs salinity for 72 locations throughout the Baltic system. Dotted line indicates predicted species diversity at low salinity (freshwater species tolerant of low salinity). From Bleich et al. (2011).

<u>Hypoxia and Eutrophication</u>

Coastal hypoxia is widespread in the Baltic and has been increasing since the 1950s (Conley et al. 2011). Anoxic “dead-zones” are also spreading in the deeper basins throughout the Baltic system (Diaz and Rosenberg, 2008). As noted earlier, this is being driven by several processes that stimulate primary production, and subsequently increase decomposition, remineralisation, and hypoxia at the sea-floor (Meier et al., 2015; Schneider et al., 2015). In the northern Baltic, hypoxia correlates with reduced benthic productivity (especially in sheltered areas, Weigel et al., 2015). Where eutrophication does not cause hypoxia, however, the eutrophication-induced increase in the food supply to the benthos can increase the energy available to filter-feeders to buffer against negative effects of acidification (e.g. Thomsen et al., 2013).

In a broader context, hypoxia and eutrophication in combination with overfishing have triggered regime shifts from cod-dominated to sprat- and herring-dominated ecosystems in the Baltic Proper (e.g. Casini et al., 2009), but not in the Danish belt (Lindegren et al., 2010). Eutrophication and overfishing can also have negative impacts on shallow coastal seagrass ecosystems (Moksnes et al., 2008), that mimic those of warming and acidification (Alsterberg et al., 2013, see below), and may act synergistically.

Invasive Species

Marine species distributions are moving poleward as the oceans warm (Molinos et al., 2016). Unlike terrestrial systems, marine species are tracking current climate velocities relatively well (Pinksy et al., 2013, Hiddink et al., 2015) – a response that may be due to the higher degree of connectivity in most marine systems. Within the next 80 years these marine range-shifts are projected to lead to an increase in species turnover (β-diversity – a measure of species differences among habitats), and hence shifts and homogenization of species composition in temperate regions such as Sweden (Molinos et al., 2016). To some extent this has already been observed in the Skagerrak and Kattegat (Hiddink and Coleby 2012), however, connectivity between the Kattegat and the rest of the Baltic system is constricted by the Öresund and Danish Belt (Johannesson and André 2006), and this may in turn constrict the flow of new species.

Importantly, while any shifts in species composition will change the *status quo*, the geologically recent nature of the Baltic (Björck et al., 2008) makes the concept of "invasive" species somewhat problematic: all species have arrived since the last deglaciation (Bonsdorff 2006), and almost half have arrived within the last 40 years, probably via shipping (Galil et al., 2014). Invasive species typically have broader environmental tolerances than native species, although whether this also applies to OA-tolerance is not known.

Notwithstanding their relatively recent origins, Swedish marine ecosystems today deliver valuable ecosystem services, and projected changes in community composition will inevitably change those services. For example, the invasive polychaete *Marenzelleria* spp. burrows deeper than native species and therefore changes the bioturbation and biogeochemistry of soft-sediment substrata. This reduces hypoxia/anoxia, which in turn changes the N and P dynamics of the system (Maximov et al., 2015) making more phosphorus available for primary production. While there is considerable uncertainty about the timing of such events, there is a strong likelihood that further introductions of new species will change ecosystem services.

3.4 The Biological Consequences of Ocean Acidification and other drivers

Separating the effects of ocean acidification from other drivers is inevitably artificial: as we have already seen, pH of coastal waters is heavily influenced by the photosynthesis and respiration of organisms (and hence also by eutrophication and temperature), and is also dependent on alkalinity (≈ salinity). Research to date suggests that at least some Baltic species are well adapted to these fluctuations and thus may be able to manage some degree of ocean acidification. Nonetheless, the combination of declining mean, and increasing variance, in pH (Fig. 8) will further stress organisms – and evidence suggests this may have detrimental consequences. Here we summarize the state of knowledge of the effects of ocean acidification on important species and systems of Swedish coastal waters.

Effects on Primary Productivity

Primary productivity in Swedish coastal waters is adapted to the large seasonal fluctuations in pH (Figs 6, 8). Given this high degree of pH variability, marine plants and algae might be expected to be relatively robust to further anthropogenic increases in CO_2.

A series of comprehensive mesocosm experiments from the Baltic Proper and the Kattegatt / Skagerrak, broadly confirm these expectations, at least for phytoplankton: increasing CO_2 had no impact on primary production of phytoplankton, nor on succession in phytoplankton – except when mineral nutrients were limiting (Paul et al., 2015; Bach et al., 2016; Spilling et al., 2016). When nutrients were not limiting, increasing CO_2 shifted phytoplankton community composition toward much smaller picoplankton (Schulz et al., 2013), which stimulates the microbial loop and reduces energy flow into zooplankton (Bermudez et al., 2016; Endres et al., 2014). As zooplankton are important prey for fish communities in Baltic ecosystems (Österblom et al., 2007; Möllman et al., 2009), and play a dominant role in energy flux from surface waters to the benthos, this interaction between acidification and eutrophication may play an important role in future regime shifts.

The nitrogen-fixing cyanobacteria that dominate summer phytoplankton blooms in the Baltic are not limited by availability of inorganic nitrogen. As noted earlier, hypoxia-induced release of phosphorus – the other key limiting nutrient – from the sediment also stimulates primary production. Nonetheless, responses of cyanobacteria to elevated CO_2 are not overwhelmingly positive: different species display positive, negative, or no responses to acidification (Eichner et al., 2014), such that overall community-level responses are subtle (Paul et al., 2016). For spring-bloom diatom and dinoflagellate species, responses to increased CO_2 are also variable: diatoms from the Skagerrak increased growth rates under elevated CO_2 (Kremp et al., 2012), whereas growth of a toxic dinoflagellate was unaffected, (although toxin production increased with CO_2, Kremp et al., 2012). Again, community-level responses to elevated CO_2 are small and subtle (Sommer et al., 2015). For calcifying phytoplankton such as coccolithophorids, acidification is widely reported to have negative impacts, however responses vary widely among different clones (Langer et al., 2009).

Benthic macroalgae also respond variably to increasing CO_2: growth of filamentous "opportunistic" green- and red-algal species in the Baltic responded positively to acidification, whereas the more robust brown alga *Fucus* was not affected (Pajusalu et al., 2013; Al-Janabi et al., 2015, 2016). Similarly, although seagrasses are generally expected to respond positively to elevated CO_2, available evidence suggests this only occurs with sufficient nutrients and/or in combination with other factors, notably warming (Alexandre et al., 2012; Eklöf et al., 2012).

Effects on Secondary Productivity

Effects of OA on marine animals in Swedish waters are highly variable, with many moderating factors and influences. Among the most tolerant and adaptive taxa identified to date are copepods, which are generally very tolerant to high pCO_2

(Kurihara, 2008). Studies in Baltic copepods show a high degree of phenotypic buffering to OA in many traits (Vehmaa et al., 2016; Almén et al., 2016). This buffering is increased by trans-generational inheritance of plasticity (Kattegatt, Thor and Dupont 2015; Baltic Proper, Vehmaa et al., 2016), although there may be limits to this and higher pCO_2 levels may still have negative effects (Vehmaa et al., 2016). Other work in the region has found significant variation in responses among different copepod populations (Thor and Oliva, 2015), indicating that susceptibility also varies with location. Many of these experiments cover multiple generations, and therefore the capacity for acclimation and adaptation in these taxa over the decadal timescales (with hundreds of generations) over which near-future OA will impact, is likely to be even greater than that summarized here and hence the likely effects of OA will be small. Nonetheless, because copepods are central to the foodweb of the Baltic Proper, any OA-induced shifts in their dynamics may influence other trophic levels (see below).

In contrast to zooplankton, benthic and demersal animals generally respond negatively to OA. This is evidenced by multiple papers from ecologically important Baltic species of bivalve (Janson et al., 2013, Gazeau et al., 2013, van Colen et al., 2012), echinoderm (Dupont et al., 2008, Hernroth et al., 2011), crustacean (Hernroth et al., 2012), and demersal fish (Gräns et al., 2014, Jutfelt et al., 2013). Nonetheless, here too there are exceptions and modifying factors, and some studies have reported neutral (Havenhand and Schlegel 2009) or even positive (Dupont et al., 2010) effects. Known modifying factors include food (energy) availability and environmental history. For example, increased energy availability can reduce negative impacts of OA in barnacles (Pansch et al., 2014), and several studies show extreme OA tolerance in bivalves from parts of the western Baltic that routinely experience strong upwelling / low pH events (Thomsen et al., 2010, 2013). As in the copepods, there are also several studies demonstrating acclimation to OA over relatively few generations in benthic invertebrates (e.g. Dupont et al., 2013) and fish (Schade et al., 2014). More data are required before generalisations can be made on the potential for transgenerational acclimation in these species.

Surprisingly, there are few data from Baltic waters for effects of OA on the bivalves *Mytilus trossulus/edulis* and *Macoma baltica*, which are key ecosystem components (Niiranen et al., 2013; but see Jansson et al., 2013). Similarly, there are very few data on the effects of natural daily fluctuations in pCO_2, which dominate the system (see above). The effects of such fluctuations have been shown to be very different to those of the constant pCO_2 used in most experiments, to the extent that *some* barnacles in fluctuating acidification (mimicking diurnal fluctuation plus OA) grew far better, whereas some grew much worse. Thus, natural daily fluctuations in pH may select phenotypes that benefit under future OA (Eriander et al., 2015). Again, more data are needed before any conclusions can be drawn.

In commercially important fish species such as cod, sprat, herring, and halibut that are key components of Skagerrak/Kattegatt and Baltic Proper ecosystems, OA effects can vary substantially. For example, OA has negative effects on larval growth in halibut (Gräns et al., 2014), herring (Franke and Clemmesen 2011), and cod (Frommel et al., 2013), but neutral effects on sperm motility and swimming performance of adult cod (Frommel et al., 2011, Melzner et al., 2009). Modelling has suggested that climate impacts on cod stocks in the Baltic Proper will be negative, even if fishing pressure is reduced (Meier et al., 2012), however potential effects of OA were not included in that study. Recent work has estimated that direct and indirect effects of OA may reduce recruitment of western Baltic and Atlantic cod stocks by 90% (Stiasny et al., 2016), and consequently it seems likely that overall anthropogenic impacts on cod – and probably most commercially important fish – stocks in the Baltic will be negative. The magnitude of the economic costs of these impacts has not been clearly identified (see below).

Evolutionary Adaptation

For species that occur on both sides of the steep salinity gradient at the Kattegatt / Baltic boundary, genetic diversity is substantially greater in Skagerrak and Kattegatt populations than in populations from the Baltic Proper and Bothnian Sea (Johannesson and André, 2006). This may have arisen for any of several reasons, including bottlenecks during recent colonization events, and presently lower population sizes in the Baltic. In addition, most species of marine origin in the Baltic Sea live close to the limits of their environmental tolerance levels (notably salinity), which tends to: i) reduce population sizes further (Bridle et al., 2010), and; ii) prevent local adaptation because gene-flow occurs from locations with more benign conditions (Kawecki, 2008). Increased rates of asexual reproduction in the Baltic Sea have been hypothesized to counteract this, by retaining favourable genetic combinations across generations in marginal habitats (Silvertown, 2008).

Theory predicts that greater genetic diversity should provide more opportunities for adaptation (Reed and Frankham, 2003), and that this in turn should provide greater resilience to environmental perturbation (de Mazancourt et al., 2013; Anderson et al., 2015) [9]. Theory also predicts that lower population sizes increase the role of stochastic factors such as genetic drift relative to the forces of natural selection, restricting the capability of populations to adapt (Polechová and Barton, 2015). Consequently, we might expect that the capacity of low genetic diversity populations in the Baltic Proper to adapt to OA would be less than in the relatively diverse populations of the Skagerrak and Kattegatt. At present there are no data with which we can test this prediction. However, despite their reduced genetic diversity, it is clear that some populations in the Baltic system are strongly tolerant of extreme levels of OA, (at least when food availability is high, Thomsen et al., 2013), and that Baltic populations have adapted rapidly in the past to stressors such as salinity change and pollutants (Johannesson et al., 2011) – which has led to the formation of at least one new endemic species with increased tolerance levels (Pereyra et al 2009). In the absence of further data it is impossible to reach a conclusion, however the scant available evidence suggests that theoretical expectations are not borne out and that there may be greater capacity for evolutionary adaptation to OA in Baltic populations than theory would suggest.

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[9] Experimental tests of role of intra-specific genetic diversity in resilience to marine climate change are few but show that resilience of Baltic seagrass beds to an extreme warming event was positively related to genetic diversity (Reusch et al., 2005, Ehlers et al., 2008)

Ecosystem-level change

Mesocosm investigations of the ecosystem-level effects of OA on phytoplankton communities have been discussed earlier. For higher trophic levels in planktonic systems, mesocosm studies have shown that warming and OA in combination have varying outcomes: gelatinous zooplankton thrive under these circumstances (Winder et al., in review; and see Richardson and Gibbons, 2008), whereas copepod zooplankton responded negatively to OA, but positively to warming, switching the ecosystem from bottom-up to top-down control under warming (Paul et al., 2016b). In shallow benthic mesocosms (seagrass beds), OA alone had few or no effects but once again, warming (alone and in combination with OA) played a major role in community composition and productivity (Alsterberg et al., 2013; Eklöf et al., 2012, 2015).

Benthic-pelagic coupling plays an important role in Baltic ecosystems, not least for the transfer of planktonic primary production to the benthos (benthic macrofauna are an important food source for fish, notably cod), but also for water-quality (Griffiths et al., in press). Consequently, the observation of negative impacts of OA on some bivalves (see above) could be significant for coastal ecosystem function. Future projections show that zoobenthos biomass in Baltic shallow ecosystems is also likely to decline in response to eutrophication, hypoxia, warming and freshening (Weigel et al., 2015). Such changes could amplify the negative effects of OA, leading to substantively decreased ecosystem function.

In a broader context, freshening in combination with OA and eutrophication-induced hypoxia are likely to have negative impacts on Baltic cod – a major predator – and hence on the trophic cascade that has been implicated in regime-shifts between cod-dominated, and sprat/herring-dominated systems (Casini et al., 2009; Frommel et al., 2011). This shift includes cascading effects on zooplankton, although recent work suggests these changes might not always carry through to affect phytoplankton and primary productivity (Viitasalo et al., 2015). Decrease in cod stocks – and those of other predatory fish – can also be caused by climate-change impacts on the number of egg predators (Bergström et al., 2015) and by overfishing, which has cascading effects on the structure of seagrass ecosystems (Baden et al., 2012), and zoobenthos (Olsson et al., 2013).

Modelling of the open Baltic Sea ecosystem shows that this may change significantly under future changes in climate, fishing and nutrient input (Niiranen et al., 2013). However, Baltic Sea ecosystem/food-web models do not currently account for the effects of OA. In other regions additive, synergistic, and antagonistic effects of OA and other drivers have been observed in different ecosystem management scenario models (e.g. Griffith et al., 2011; Kaplan et al., 2010). Including the effects of OA in ecosystem modelling of the Baltic Sea is an important, and currently lacking, step that calls for: i) empirical knowledge/data on the effects of OA on species physiology, demography, and ecosystem responses; and ii) modelling approaches that describe the key processes and functions affected by OA at the levels of individual organisms, species, and ecosystems (e.g. Koenigstein et al., 2016).

The importance of Biodiversity

The positive relationship between biodiversity and ecosystem resilience is known from both terrestrial and marine systems (e.g. Maestre et al., 2016; Sgro et al., 2011; Folke et al., 2004). Increased diversity within species (genetic diversity) and among species (species richness and evenness) both contribute to functional diversity within the ecosystem, which increases the ability of the system to resist – or recover rapidly from – a disturbance (i.e. "resilience"). Examples from marine ecosystems include genetic diversity enhancing the resilience of seagrass beds (Reusch et al., 2005), genetic and species diversity enhancing the resilience of coral reefs (Hughes et al., 2003), and of species diversity (complementarity) enhancing the resilience of the California Current pelagic ecosystem (Lindegren et al., 2017). The extent to which resilience to OA is enhanced by biodiversity is less well-understood, although the principles should still apply. Importantly, work in Swedish seagrass beds has indeed confirmed that warming and OA in combination reduce biodiversity and resilience (Eklöf et al., 2012).

These clear relationships between biodiversity, ecosystem functioning, ecosystem services, and resilience (Worm et al., 2006), indicate potential pathways for societal adaptation to OA (see section 4 below).

3.5 Consequences for Ecosystem Services

Oceans and coastal ecosystems generate important ecosystem services and economic values that influence human welfare directly through their use, but also indirectly through their complex interactions with other parts of the socio-ecological system in which they are embedded. Following TEEB (The Economics of Ecosystems and Biodiversity; Sukhdev, 2008) the definition of ecosystem services can be divided into: a) supporting (e.g., biogeochemical cycling, primary production, food web dynamics, diversity, resilience); b) provisioning (e.g., fish and fisheries); c) regulating (e.g., climate, mitigation of eutrophication, biological regulation, regulation of hazardous substances); and d) cultural (e.g., recreational fisheries, tourism; Millennium Ecosystem Assessment, 2005; Sukhdev, 2008). Marine and coastal systems are increasingly under pressure from direct human activities such as overfishing, pollution and habitat destruction. However, indirect human activities now play a larger role in shaping marine ecosystem dynamics and their potential for ecosystem service generation worldwide (Norström et al., 2016). Examples of such indirect drivers are urbanization, changed consumption patterns, and human migration, which indirectly impair the health of marine resources and habitats (Österblom et al., 2016). In Swedish waters, eutrophication has been the main cause of ecosystem change resulting in reduced capacity of marine ecosystems to provide goods and services. Causal relationships between individual threats and ecosystem changes are, however, rarely straightforward. Ecosystem disturbances can act over a long time, resulting in very gradual changes in their structures and functions. Alternatively, when disturbances are strong – or have been acting over a long time – systems may experience a rapid shift that changes their structure and functions and also their ecosystem services (Scheffer et al., 2001).

Identifying and forecasting the effects of OA on ecosystem services in Swedish waters is difficult due to the many interactive factors discussed earlier, however, it is obvious that OA can impact a wide range of ecosystem services (a-d above). Among the provisioning ecosystem services alone there are multiple examples, such as the production of fish – a key ecosystem service that can be impacted in several ways: negative effects of OA on Baltic cod and other coastal fish species, (Frommel et al., 2011; Franke and Clemmesen 2011); the widely-reported negative effects of OA on the aquaculture industry (e.g. mussel and oyster growth; Barton et al., 2012; Gazeau et al., 2010, 2011, 2013); and indirect effects on fin-fish farming by ocean acidification in distant waters that support forage fish incorporated into aquaculture feeds. Similar examples can be found for supporting, regulating, and cultural ecosystem services in Swedish coastal waters.

Currently available information is insufficient to permit even an order of magnitude estimate of the cost of these impacts to Swedish society. However, the potential risks for economic losses can be illustrated by looking at the Swedish fishery sector (industrial and recreational). In 2013 more than 1.6 million recreational fishermen landed 16 000 tonnes of fish. The value of this fishery – approximated by the costs and investments that the recreational fishermen are willing to pay – was estimated to be SEK 5.8 billion (SCB, 2014). This sum included travel within Sweden, accommodation, fishing gear and licenses, fuel, investments in boats, etc., but is probably an underestimation of the total willingness to pay for these activities. The industrial fishery sector landed 177 million tonnes in 2013, at a value of SEK 1.3 billion (HAV, 2014).

Swedish aquaculture production is small and mainly dominated by freshwater finfish (11 thousand tonnes produced in 2014; SCB, 2015). Although the overall value of Swedish aquaculture is low (SEK 370 million in 2014; SCB, 2015) there is presently a momentum for the industry to grow. Targeted research funding, new aquaculture centres, and new food policies that emphasize the need for Sweden to become more self-subsistent will push the aquaculture industry forward. Aquaculture for feed production (mussels in the Baltic) is also developing and is threatened by potentially negative impacts of OA.

A very preliminary attempt to quantify and value the impacts of OA on Norway's fishing industry, (both capture fisheries and aquaculture), indicated annual net loss in the order of several million NOK (Armstrong et al., 2012). The same study estimated that costs arising from the negative impacts of OA on the regulating service of carbon storage would be several orders of magnitude higher than those for fisheries and aquaculture. These figures are unlikely to apply to Sweden, however equivalent data for Swedish marine ecosystem services are lacking.

3.6 Knowledge Gaps and Implementation Needs

i) In addition to anthropogenic CO_2 emissions, several geochemical processes, such as input of terrestrial organic matter, microbial decay of organic matter, and changes in river discharge, impact the development of ocean acidification in Swedish coastal waters. The combined effects of all these processes on OA are not trivial to project. Further, several of these geochemical processes also impact the oxygen environment, which adds stress to the system. In order to understand the broader effects of OA on species and ecosystems, there is a need to:

Study historic environmental records, better quantify the relevant geochemical processes impacting OA and other stressors, and develop models to project future levels of these stressors under different forcing.

ii) The resilience of ecosystems to perturbations such as OA is dependent, in part, on the sum of all the stressors on the system. In the Baltic, regional and local anthropogenic drivers such as eutrophication are projected to influence the magnitude of pH fluctuation (Fig.8). Fishing pressure and dissolved toxicants add further stress to Swedish marine ecosystems. The combined direct, and indirect effects of these drivers can be non-additive and therefore difficult to anticipate.

There is an urgent need to identify the extent and magnitude of the direct and indirect effects of multiple stressors (including OA) on key Swedish marine species, ecosystems, and ecosystem services.

iii) Modelling tools for some Baltic ecosystems and food-webs are relatively well-developed and well verified, whereas models for other regions (notably the Kattegatt and Skagerrak) are less well developed. None of these models, however, incorporate the potential effects of OA on biota, and consequently our ability to project the likely outcomes of OA for ecosystems and key ecosystem services is poor.

There is an urgent need to develop holistic ecosystem modelling frameworks that incorporate the effects of multiple anthropogenic drivers, including OA, at large spatial scales.

iv) There are no data currently available on the adaptive capabilities of Baltic organisms to OA. It is therefore impossible to project long-term changes to Baltic ecosystems. This is most critical for species that are key components of ecosystems, such as those that provide habitats for other species. Priorities should include:

Assessing the capacities, and limitations, of key Baltic species to adapt to ocean acidification over multiple generations.

Determining the importance of within-species (genetic) diversity for ecosystem resilience.

Projecting potential cascade effects on the trophic network from adaptive capabilities (or lack thereof) of individual species.

v) We currently know too little to make a valid quantitative assessment of the likely social and economic risks, and vulnerabilities, arising from biological responses to OA. Hilmi et al. (2013) list the requirements for completing such an assessment. For Sweden these are:

Determination of the fraction of GNP that depends on provisioning ecosystem services such as fishing and aquaculture, and cultural services such as coastal tourism, that depend on OA sensitive ecosystems;

Identifying likely shifts in species composition, and hence food-value, of seafood under OA

Projecting changes in human populations dependent on the coastal zone

Determining the vulnerability and sensitivity of these coastal populations to environmental change and assessing their capacity to adapt.

4. RESPONSES TO OCEAN ACIDIFICATION

Although ocean acidification (OA) is a key consequence of increasing CO_2 emissions and can have significant negative effects for both ecosystem services and the human activities that rely on them, literature on societal responses to OA, more specifically, is almost non-existent. With a few prominent exemptions (e.g. Harrould-Kolieb, 2016; Billé et al., 2013; Rau et al., 2012; Kelly and Caldwell, 2012; Harrould-Kolieb and Herr, 2011; Kelly et al., 2011; Cooley and Doney 2009; Pacala and Socolow, 2004) the majority of research on responses relevant to OA focuses explicitly on other factors that drive and/or threaten marine systems, notably the overarching concept of climate change. As indicated in Section 2, this also applies to most, if not all, of the institutions and policies already in place that currently affect levels of OA: thus far political efforts have been prompted by concerns other than reducing ocean acidification. Nonetheless, research on environmental politics and policy (in particular with regards to global climate change) is indeed highly relevant for the problem of OA, either because we can translate lessons from other empirical areas to fit the problem (for example administrative structures, multilateral cooperation, use of policy instruments, etc.), or because addressing one well-researched major problem (CO_2-induced climate change) also directly affects the degree of OA.

In the social science literature on OA (e.g. Billé et al., 2013; Cooley and Doney, 2009), suggestions for responses have been roughly categorized as: i) mitigation of the main source of OA, i.e. anthropogenic CO_2 emissions; ii) mitigation of other greenhouse gas (GHG) emissions; iii) mitigation of other local/regional factors that contribute to (or compound the effect of) OA, for example pollution, eutrophication and loss of biodiversity; iv) adaptation of ecosystems and human activities that build resilience to OA and; v) a range of different restoration activities. Here we address these suggestions under the headings of mitigation and adaptation.

4.1 Mitigation

Mitigation as a strategy

By mitigation we mean a strategy to treat the problem at its root by reducing its very causes. As OA is primarily a problem of anthropogenic CO_2 emissions, many (though not all, see Harrould-Kolieb, 2016) of the policies already in place, with the purpose of alleviating climate change, will also positively affect acidification. In the long term, limiting fossil fuel emissions of CO_2 to the atmosphere, as well as reducing the amount of CO_2 already in the atmosphere (Billé et al., 2013), are the only mitigation strategies available to comprehensively address OA (Pacala and Socolow, 2004; Cooley and Doney 2009). However, as is evident from efforts to combat climate change, such strategies are typically not “quick fixes” for a problem (see Sterner et al., 2006), but rather long-term strategies that require a wide range of efforts on the global scale, and are both complex and politically contested. In the short-term, therefore, initial mitigation responses to OA should instead focus on the local or regional level, where already existing management structures and legal systems might serve to also address the root causes of OA (Kelly et al., 2011; Cooley and Doney, 2009).

Changing behaviour with national-level policy tools

As illustrated in Section 2 on causes of OA, human behaviour in the form of anthropogenic carbon dioxide emissions – but also other local activities such as pollution, resource extraction, ecosystem management or rather the lack thereof – lie at the root of the problem. Thus, OA can be readily described as a classic collective-action problem, in which lack of coordinated behaviour or cooperation among actors responsible for OA results in suboptimal outcomes for the collective (Olson, 1965). A typical mitigation-response would therefore be to change these negative behaviours toward more desirable patterns of action. However, since limiting OA implies the avoidance of a public bad (see “Causes” above), it will rarely be provided in sufficient quantity, (if at all), because of the difficulties of aggregating all actors’ willingness to pay for it. Hence, following public choice theory, there is an imminent risk of free-riding. This risk arises regardless of whether or not the collective willingness-to-pay for an additional limitation of CO_2 emissions is large (e.g. Samuelson, 1954; Clarke, 1971; Ostrom, 1990). At the individual level, voluntary behavioural changes should therefore not be anticipated to any significant extent. Some form of intentional, third party coordination or *governmental coercion* (Mansbridge, 2014) is usually necessary to initiate cooperation, especially when the number of involved actors is very large, dispersed over vast geographical areas, and therefore mostly anonymous to each other. Such governmental coercion is typically realised via different types of policies and policy measures. On administrative levels where regional, national or local governments hold authority, governmental involvement usually takes the form of various policy tools aimed at changing the incentive structures governing behavioural choices, either by increasing the attractiveness of preferred behaviour or by exacerbating the negative impact of an undesired behavioural choice.

Governments across the world, Sweden included, have proposed, developed and implemented many pro-environmental policy measures in their attempts to overcome large-scale collective action problems and, thus, to induce positive individual-level behavioural changes (IPCC 2014; Jordan, 2005; Coria and Sterner, 2012). Here we organise these measures as legal, market-based and informative.[10] Rather than focusing on OA *per se*, relevant current literature is concerned with policy measures aimed at lowering CO_2 emissions for the purpose of mitigating climate change. Evaluating the merits of different climate policy designs is a complex task, and over time governments have based their evaluations on a rather differentiated set of criteria (Konidari and Mavrakis, 2007). Effects on the target, i.e., to cut CO_2 emissions, must be considered in combination with the cost of monitoring and enforcing policy compliance, the fit within the current political-institutional

[10] An alternative would have been to, e.g., use Vedung’s (1998) seminal tripartite classification of policy instruments as sticks, carrots, and sermons. The primary distinction between different types of policy instruments is usually the amount of coercion they imply (see also Sterner and Coria 2012).

system, and the possible side-effects of implementation (IPCC, 2014). However, the extent to which a policy instrument succeeds in effectively and efficiently bringing about OA-mitigation depends not only on technical or political-administrative factors, but also on the reactions of the target audience. In particular, the extent to which a policy instrument enjoys broad public support has been shown to affect both its short- and long-term performance significantly (Matti, 2010; Stern, 2008; March and Olsen, 2004; Ostrom, 2005).

1. *Legal policy tools*

A multitude of legal policy tools that aim to address both the long-term (i.e. CO_2 emissions) and short-term (i.e. pollution, resource management, biodiversity) is already in place at the regional, national and sub-national levels. Although not designed to address OA specifically, these existing legal frameworks need not change substantively in order to do so, but they do need implementation (Billé et al., 2013). The main source of OA – CO_2 emissions – is subject to both international agreements and EU-directives, and is therefore an integrated part of Swedish national law that particularly affects the behaviour of business and industry. EU climate change law includes the EU emissions trading scheme (Directive 2003/87/EC), measures for the promotion of energy from renewable sources (Directive 2009/28/EC), and for energy efficiency (Directive 2012/27/EU). Similarly, emissions of other important stressors that compound the effects of OA (for example emissions of SO_x and NO_x, as well as eutrophication caused by release of nitrates and phosphates to water) have already been subject to extensive regulatory efforts, although prompted by concerns other than OA; e.g. the Convention on Long-range Transboundary Air Pollution (CLRTAP), the Ambient Air Quality Directive (2008/50/EC) and the National Emission Ceilings (NEC) Directive (2008/50/EC).

Further, several legal instruments have been developed with the primary, or secondary, aim of protecting the marine environment. Most notable among these are the two framework directives on water (the Water Framework Directive, 2000/60/EC) and the marine environment (the Marine Strategy Framework Directive, 2008/56/EC), the Nitrates Directive (Directive 91/676/ EEC), which aims specifically at reducing water pollution caused or induced by nitrates from agricultural sources, and the Urban Waste Water Directive (Directive 91/271/EEC). However, the fact that all major EU measures in these areas take the form of directives (as opposed to directly applicable regulations) means that it is for the member states to transpose the EU acts into binding measures applicable to individual operators at the domestic level (e.g. Lindegarth et al., 2016; Langlet and Mahmoudi, 2016). This transposition generally results in significant variation between member states as to how the EU measures are applied and, in some cases, with respect to how the EU requirements are construed.

In Sweden, relevant EU-directives are implemented *inter alia* through the Act (2004:1199) on emission trading, the Air Quality Ordinance (SFS 2010:477), and the Environmental Code (SFS 1998:808). The Environmental code and associated legislation also include various rules for the agriculture sector, targeting eutrophication specifically. The national and local levels have, at least formally speaking, significant possibilities to regulate nutrient leakage through measures pertaining to land use. OA has been mentioned briefly in preparatory works to legal acts relating to climate change mitigation and to protection of the marine environment, but has not directly impacted the elaboration of specific measures. In the legal literature with a national or regional focus, OA has received very little attention.

In contrast to OA, legal aspects of the management of fisheries have been more extensively discussed in the legal literature, both as regards international law and that of the EU, but again hardly ever from an OA perspective (Christiernsson et al., 2015; Churchill and Owen, 2009; Lado, 2016). In Sweden, fisheries are mainly regulated by the EU. Since conservation of marine biological resources under the EU's common fisheries policy is the exclusive competence of the Union this lack of OA perspective is predominantly a matter that must be addressed though EU law. There is some limited room for individual member states to regulate fishing in the vicinity of their own coast and conducted by their own fishing fleet (Christiernsson et al., 2015). Whereas these powers are not insignificant in a local perspective, they cannot be relied upon to provide a system-level impact on the stressors relevant to OA. So far, the effects of fishing on OA impacts have not been a significant concern for decisions on fish quotas or for the adoption of other management measures. Marine protected areas, which may serve as a means to increase the resilience of ecosystems subject to multiple stressors, are partly regulated at the EU level (e.g. through minimum requirements of protection as part of the Natura 2000 network, and through restrictions posed by EU fisheries policy), and partly at the national level. Also here, however, decisions are not to any noticeable degree influenced by concerns related to OA.

2. *Market-based policy tools*

In general, economic policy responses aim to alter incentive structures by directly addressing market failures (i.e. negative externalities, information deficits, public goods-provisions) that give rise to unwanted behavioural patterns. Typically, regulating CO_2 emissions, or emissions of other GHG's and pollutants, using economic policy instruments is done through quantity regulation (the amount that should be produced), price regulation (taxes), or mixed regulation (e.g. Cap and Trade; see Hepburn, 2006). Such policies have been extensively studied in the context of climate change reduction, but in principle OA-mitigation could be achieved by the same means. The use of CO_2 taxes is widely regarded as one of the most cost-effective means of limiting emissions and changing behaviour, and has been implemented in Sweden since 1991 with gradually increasing public support (Jagers and Hammar, 2009; Jagers and Matti, 2016). Similarly, taxes on the commercial use of fertilizers and pesticides as well as on (land-based) NO_x and SO_x emissions have been in place since the early 1990s. Although economic regulation of emissions may seem straightforward, research has also pointed towards several traps that need to be negotiated for them to be efficient, including optimal distribution of reductions, cross-country coordination, and support for technological innovations (Somanathan et al., 2014; Hoel and Karp, 2001; Greaker and Hagem, 2008). The price-instrument is also applied to target more specific behaviours, rather than pricing the emissions *per se*, although with a similar outcome. For example, the congestion-charges currently implemented in Stockholm and Gothenburg discourage private car use and hence limit emissions from fossil fuels, and the electricity consumer tax directly targets households' energy-usage.

Other types of economic policy tools that currently are, or could be, directed towards emissions-reductions include pull-instruments that subsidize more favourable alternatives. These can target both the production of alternative energy-sources (e.g. feed-in-tariffs for biofuels or other renewables), consumer-behaviour (e.g. eco-car subsidies), or investments in the management of biodiversity (e.g. preservation of fish-stocks, water-quality improvement; OECD, 2016; Söderberg, 2011). Several such subsidies are already implemented in Sweden.

3. *Informative policy tools*

Informative policy tools serve to highlight the problem at hand, with the dual purpose of initiating voluntary action and increasing support for the implementation of legal and economic policy tools. In the context of a common pool resource where too much uptake can lead to a regime shift in the resource renewal rate there is evidence that providing information about resource dynamics to resource users triggers management that is at least as efficient as a traditional quota system without regulation (Lindahl et al., 2016). Information can also help address the market problems of asymmetric information. Examples of information instruments include eco-labelling or certification schemes for products or technologies, and collection and disclosure of data on GHG emissions by significant polluters (Krarup and Russell, 2005). Such types of policy can also trigger changes in social norms if the information can change the perception of large groups in society about what is accepted behaviour and what is not (Nyborg et al., 2016). This is, however, complex as the reasons for a consumer choosing an eco-labelled product are many and not always well correlated with information/knowledge about environmental impacts (Jonell et al., 2016; Jagers et al., 2016). Nonetheless, as research demonstrates that people are largely unaware of OA and its consequences (Leiserowitz et al., 2010; Capstick et al., 2016; Frisch et al., 2015), and people in general are unsupportive of solutions to unfamiliar problems (e.g. Stern et al., 1999), increasing the information on the causes and consequences of OA is a potentially important tool for legitimizing further mitigation-policies.

A second aspect of information relates to optimal provision of research in order to improve understanding of OA. Identifying what levels of OA might be acceptable would require assessing all the trade-offs between valuable economic activities that generate CO_2 and the harm to society caused by the resulting increase in OA. Consequently, the optimal price of CO_2 to society cannot be calculated, making it difficult to calibrate any policy instrument targeting OA in Sweden. This task is currently intractable given the low level of knowledge in particular about the impacts of OA. More knowledge is needed about the problem and its causes and about ways of targeting the problem (Armstrong et al. 2012; Turley and Gattuso, 2012; Brander et al., 2014). As with

OA-mitigation, however, research is a public good which necessitates governmental intervention to be provided in ample quantities.

The necessity of international cooperation

In addition to the collective action problem presented by OA on a national/regional-level, OA is a global public bad implying the need for cooperation between countries for its mitigation. However, the lack of a supranational authority and limited enforcement mechanisms suggest that legal measures to initiate multilateral cooperation must take the form of voluntary agreements and treaties, in which compliance is largely dependent on the will and ability of individual states. This can be extremely tricky to achieve, in particular agreements of how costs and benefits should be distributed across national borders, and negotiating multiple free-riding problems (cf. Barrett, 1994, 1999, 2003, 2008; Aldy et al., 2003). Although not directed towards OA specifically, several international legal regimes addressing both the long- and short-term sources of OA are already in place on several administrative levels.

Box 4. Eutrophication in the Baltic: HELCOM and BSAP

In Sweden, marine eutrophication is very much a regional concern because excessive nutrient loads in the Baltic Sea are the result of emissions and runoff from all the states in the catchment area – and even from states further away. The regional response has primarily been in the form of the 1992 Convention on the Protection of the Marine Environment of the Baltic Sea Area (Helsinki Convention) with its governing body the Baltic Marine Environment Protection Commission, or Helsinki Commission (‘HELCOM’). The parties to the Helsinki Convention have adopted the ‘Baltic Sea Action Plan’ (BSAP, HELCOM 2007) as a platform for regional implementation of the EU’s Marine Strategy Framework Directive in the region. This has been prompted by the fact that all Parties to the convention (except the Russian Federation) are also members of the EU and, hence, there is a strong need to coordinate the work carried out by those states as parties to the Helsinki Convention and as Members of the EU. BSAP, which specifies the measures required in order to address *inter alia* eutrophication, is not a binding legal act but may affect interpretation of the Helsinki Convention. BSAP deals with issues relevant to freshwater acidification, but OA is not specifically addressed.

Globally, strategies for CO_2 mitigation have been sought through the international climate regime based on the United Nations Framework Convention on Climate Change (UNFCCC) and subsequent instruments, including the Kyoto Protocol and the recent Paris Agreement. The development of the global climate regime has not been significantly affected by increasing knowledge about OA and there are no provisions that are explicitly aimed at or relate to OA. This has triggered proposals for the elaboration of a specific international agreement focusing on combating OA (Kim, 2012) or for at least highlighting OA as a problem separate from climate change within the present agreements (Harrould-Kolieb, 2016). These global-level tools do not directly affect the legal situation of individuals or companies, but have to be implemented in domestic (or in the case of the EU, regional) legal systems. When implementing international agreements individual states also have significant discretion in choosing instruments and methods that are consistent with their legal traditions and political preferences. This entails significant diversity in the rules and mechanisms subsequently employed in different jurisdictions (e.g. Box 4).

Conditions potentially affecting successful implementation of policy measures

The extent to which policy instruments effectively and efficiently target CO_2 emissions is dependent on how sensitive they are to the reactions of their target audience (Matti, 2010; Stern, 2008). This is crucial to prospects for reducing Swedish contributions to OA by re-orientating various actors' activities and behaviours (Steg et al., 2005; Sagoff, 1988; Berglund and Matti, 2006; Guagnano et al., 1995; Thøgersen, 2005). In what follows, we briefly account for some of the major factors affecting actors' support for pro-environmental policies, including those aimed at governing ocean acidification[11].

First, individual-level factors such as personal preferences and morals drive policy support. For issues closely related to collective interests such as OA, psychological [12] and moral-normative concerns play a significant role in guiding attitude-formation (Berglund and Matti, 2006). By this account, one of the most comprehensive and widely applied individual-level models, the Value-Belief-Norm (VBN) theory of environmentalism (e.g. Stern et al., 1995), is worth mentioning. VBN postulates that the extent to which an individual is motivated to act pro-environmentally, by for example actively supporting the introduction of a policy measure aimed at cutting CO_2 emissions, is ultimately the result of a number of factors activating a personal norm or moral obligation to behave in a specific way. Research demonstrates that these enduring and trans-situational goals play an important role as explanatory factors for individuals' behavioural predispositions (Grafton and Permaloff, 2005; van Deth and Scarbrough, 1995; Feldman and Zaller, 1992; Sniderman et al., 1991).

However, factors tied specifically to policy design also affect support, and several studies describe how people, even while subscribing to strong environmental values and beliefs, do not systematically act accordingly (e.g. Martinsson and Lundqvist, 2011; Jagers, 2009; Krantz-Lindgren, 2001). This indicates that factors other than moral notions affect policy support. One important finding is that policy support varies considerably across specific policy features, both in terms of policy design and type of behaviour being targeted. For example, policy instruments aimed at increasing the attractiveness of pro-environmental behaviour (e.g. subsidies for zero-emission fuels), are generally granted more support than instruments decreasing the attractiveness of environmentally harmful behaviour (e.g. taxing high-emission fuels). Similarly, suggestions for making new and more sustainable alternatives available to the public are received more positively than those targeting a reduction of established practices (Nilsson and Martinsson, 2012; Jagers and Matti, 2010; Steg et al., 2005; Eriksson et al., 2006). This also implies that specific beliefs about the consequences of a policy measure mediate the effect of general

[11] Due to space limitations, we deliberately leave out the "usual suspects" such as socio-economic and demographic conditions.

[12] Primarily economically oriented social scientists typically classify different psychological factors as characteristics of individual preferences. Preferences are individual properties that relate to how people choose to order alternative choices, based on the degree of satisfaction or utility they provide. For example some people may prefer crayfish over cod while other people would be better off with cod than with crayfish. These people's different preferences may impact on the way they relate to different levels of ocean acidification.

environmental beliefs on policy support. Following the multi-attribute evaluation model (Samuelsson and Messick, 1995), the individual's evaluation of a policy measure is hypothesized to be based on at least three dimensions: a) whether the measure is perceived to *effectively* contribute to solving/ improving the environmental problem; b) whether the measure is perceived to be *fair*; and c) the extent to which the measure is perceived to impact actors' *freedom of choice* (Kallbekken et al., 2013; Schuitema et al. 2010; Schade and Schlag, 2003; Joireman et al., 2001; Jakobsson et al., 2000). Thus, for policy makers to successfully develop and eventually implement a policy instrument aimed at targeting and reducing the Swedish contribution to OA, i) it is important that it is accepted and supported by the involved actors and, ii) the probability that it *will be* considered acceptable and be supported is conditioned by the degree to which the actors perceive it to be both effective, fair, and have a limited effect on the actors' freedom.

Second, although only focusing on the case of Sweden in this paper, it is worth mentioning that apart from individual-level motivation and policy design, there are also strong indications that country-specific contextual factors are important in gaining support for public policy measures addressing OA. The mere fact that policy choice varies considerably between countries (Sterner and Coria, 2012) supports this assumption. For example, communal preferences for environmental protection in general are typically attributed to levels of environmental concern triggered by general *economic affluence* (Franzen and Vogl, 2013). Furthermore, recent studies have linked a country's quality of government both to policy choice and to the public's attitudes toward these choices. For example, Harring (2014a) shows that high levels of corruption in a country drive public support for stricter use of legal regulations, rather than market-based instruments. Lastly, cross-country differences in political culture, i.e. the dominant norms and values in a society (e.g. Eckstein, 1996), have proved decisive both for individuals' policy attitudes and for governmental action (e.g. Cherry et al., 2014; Inglehart and Baker, 2000; Schwartz, 2006; Bardi and Sagiv 2003). More than anything, these results indicate that there is no unified OA-mitigation policy panacea in sight. Since the design – not to mention implementation – of such a policy would require sufficient support from various actors, the great variation in economic, cultural, political and institutional conditions among different countries speak strongly in favour of joint targets but country/region-specific policies.

4.2 Adaptation

Adaptation as a strategy

Unlike mitigation, adaptation strategies focus on treating the *symptoms* of OA by adjusting natural or human systems such that damage is reduced, and/or beneficial opportunities taken advantage of (Adger et al., 2007). Thus, adaptation does not target the causes of the problem but rather aims to maintain social well-being in spite of OA. These types of strategy are likely to be easier to apply than mitigation strategies, especially in the short-term, because symptoms of OA are problematic at local and regional levels where people also can address them (Cooley and Doney, 2009). Hence adaptation usually requires less co-ordination effort than global mitigation policies, and those efforts are typically located at the local and regional level where relevant national institutions are usually already in place. Nonetheless, practical examples of OA-adaptation remain scarce, and the barriers to negotiate are in many ways the same as those for mitigation-strategies, including collective action-problems and information-deficits. Three broad types of adaptation strategy can be identified, spanning structural-physical, social, and institutional adaptation. All of these require further government policies and programs to be initiated and funded (IPCC, 2014):

1. *Strategies that aim to reduce the negative impacts of acidification on marine ecosystems*

   As noted in Section 3, OA is one of multiple anthropogenic stressors that impact marine ecosystems and their services. Importantly, however, ecosystem resilience to ocean acidification can be strengthened in the short-term by alleviating pressure from other stressors, e.g. by reducing the rate and magnitude of eutrophication, pollution, and fishing. Because ecosystems with higher diversity are more resilient to other forms of environmental stress, and available data suggest this also applies to stress from ocean acidification, ecological restoration may be a particularly valuable tool in maintaining / increasing diversity and hence increasing resilience to OA (Worm et al., 2006). Hence, there are synergies with efforts to conserve or restore biodiversity (e.g. CBD, 2014).

   Adaptation of human activities such as fisheries and aquaculture will become limited when entire ecosystems and life cycles are disrupted beyond a critical threshold, however, before such thresholds are reached adaptation options remain. For example, the effects of acidification and warming on shallow seagrass ecosystems are similar to the effects of eutrophication and increased fishing pressure: by corollary, reducing eutrophication and fishing pressure will (in the short-term) directly offset the effects of acidification and warming. Relocation of human activities is another adaptation option, and therefore there is a need for well-connected and representative networks of marine protected areas to ensure resilience at seascape scales. Counteracting acidification by alkalinisation may be useful in hotspots such as coastal environments but has very limited potential and feasibility at larger scales (e.g. IDDRI, 2012; Weatherdon et al., 2015).

2. *Strategies that aim to adapt the way society organizes to meet the new reality*

   The impacts of ocean acidification are likely to affect ecosystem services produced in the oceans (Section 3). Turley and Gattuso (2012) list broad categories of impacts on these services including fisheries, aquaculture and food security, coastal protection, tourism, climate regulation, and carbon storage. These changes imply that some ecosystem services may be strengthened while others are reduced. The fact that some provisioning services will be negatively affected (e.g. landings of fish for human consumption) will lead to price increases. A key question is then to what degree those fish may be substituted? For example, fish as source of protein will become more expensive than other sources of protein and thus market forces will automatically steer the economy towards less dependence on fish. Conversely, although examples are currently lacking, OA may positively affect some species, resulting in new, alternative, fisheries.

If such transition processes are slow and costly, progress may be accelerated by: i) compensating losers (e.g. fishermen); ii) providing transitional support; and/or iii) stimulating innovation to accelerate emergence of alternatives and technical replacement solutions.

In some cases, however, substitutes for the good or service damaged by OA may not be available (e.g. loss of fisheries and tourism income due to degradation of high diversity coral reef systems; Worm et al., 2006). Under these circumstances, societal priorities will be forced to change toward different, less-damaged (or undamaged) goods and services. In the extreme case when life-support systems are affected and no substitute is available this could have catastrophic impacts on human well-being. This is true even if the absence of substitutes is temporary. These types of response can be deployed by individual countries and could be a part of the solution for Sweden if mitigation is unsuccessful.

3. *Strategies that aim to provide monetary compensation for people who lose from ocean acidification.*

Changes in the provision of marine ecosystem services arising from OA will likely generate redistributions of resources between user groups. For example, the different responses of two Arctic fishing communities to the disappearance of north-west Atlantic cod stocks led to two very different outcomes – one community lost substantially while the other was able to target other species and increased income (McCain et al., 2015). Compensating such groups that lose-out, and/or helping them adapt to the new situation by stimulating education, investment, etc. are also viable adaptation strategies. However, avoiding the establishment of spurious incentives that effectively reward some sections of society for not managing the change is vital, and therefore such strategies should be transitional (Dixit and Londregan, 1995).

Increasing adaptive capacity

In addition to the categories of adaptation strategy outlined above, broader adaptation potential can be increased through capacity-building activities such as infrastructural improvements, increasing institutional capacity, information, and access to resources (Smit et al., 2001). Although these measures are typically more relevant for developing countries, they remain relevant in a Swedish context, especially at local governmental levels.

4.3 Knowledge Gaps and Implementation Needs

Mitigation

i) A comprehensive overview of mitigation structures in Sweden is currently lacking. In order to understand the scale of the problem and scope of the different mitigation structures and strategies there is a need to:

Undertake research that builds a schematic understanding of how the different directives, treaties and laws link to the problem of OA and to each other.

- *Specifically to undertake research that builds understanding of how current legal and market-based policy tools reinforce (or counteract) each other in the aim of changing human behaviour.*
- *Outline how mitigation-responses, in particular already existing management structures and legal systems on the local and regional level, vary within Sweden, and the extent to which they serve to address the OA-problem.*

ii) Although a range of policy tools that aim to decrease CO_2-emissions by changing human behaviour are currently in place, few are geared directly towards the problem of OA. Thus, we do not know how a change in framing from climate change to OA would affect the acceptability of policy tools.

There is a need to analyse whether increased public knowledge of the causes and consequences of OA, and specifically targeting policy tools to address this problem, will increase public policy support.

iii) The global nature of OA requires cooperation among states, but the prospects for coordinated policy efforts, as well as the extent to which policy diffusion or transfer is possible, are unclear due to contextual variations. To better understand the potential for cross-national efforts as well as for Swedish lesson-learning from abroad we need to:

Assess all policies currently in place that influence OA in Sweden and the extent to which they also succeed in doing so at the global level.

Increase understanding of the importance of political-economic context for policy support, and how this interacts with individual-level mechanisms driving policy support and behavioural change.

<u>*Adaptation*</u>

If we are to continue to derive benefits from key marine ecosystem services, then the rate of adaptation to OA in our human systems must at least keep pace with rates of ecological change. Recent research (Creighton et al., 2016) highlights the need to:

i) *Foster resilience through habitat repair and protection:* Environmental restoration increases local biodiversity and functional diversity, and Marine Protected Areas (MPAs) may reduce fishing pressure and other local disturbances. Collectively these measures increase ecosystem resilience to OA. In Sweden, the concept of managing biodiversity for ecosystem resilience is implemented nationally, and locally in several local communities, although not always with a view to managing impacts of climate change (Wamsler et al., 2016). Moreover, existing management plans for MPAs in the Baltic region fail to account for the need to maintain genetic diversity in order to maximise adaptive potential and thereby enhance resilience (Laikre et al., 2016).

There is a need to develop, evaluate, and implement a strategy for environmental restoration and establishment of MPAs that increases ecosystem and evolutionary resilience to climate change and OA.

ii) *Improve resource allocation strategies:* Substantial impacts of OA on the generation of ecosystem services and hence on human well-being, will likely lead to redistribution of resources between individuals that may be perceived as unfair by the people who lose out.

There is a need to identify which societal groups are likely to win and to lose from OA.

Further there is a need to develop strategies regarding whether or not to redistribute resources (at least during a transition phase) in order to smooth the impacts for those who lose (e.g. compensation systems, support to new activities, etc.) Insodoing it will also be essential to consider the impacts of these measures on incentives for innovation in order to avoid lock-ins into systems that maintain old structures that are not adapted to the new situation.

iii) *Fine tune fisheries management systems:* There is a growing number of studies showing that resilience to global pressures such as OA can be increased by reducing locally-managed anthropogenic pressures (e.g. fishing effort). Thus changing local management practices for non-OA stressors may reduce threats from OA to key ecosystem services.

There is a need to test the extent to which alternate fisheries management strategies at local and regional levels (e.g. temporary closures, MPAs, reduced catch limits) can ameliorate the impacts of OA on Swedish coastal ecosystems.

iv) *Enhance whole of government approaches and policies.*

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