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Heterogeneity of impacts of high CO_2 on the North Western European Shelf

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Abstract

The increase in atmospheric CO_2 is a dual threat to the marine environment: from one side it drives climate change leading to changes in water temperature, circulation patterns and stratification intensity; on the other side it causes a decrease in pH (Ocean Acidification or OA) due to the increase in dissolved CO_2. Assessing the combined impact of climate change and OA on marine ecosystems is a challenging task: the response of the ecosystem to a single driver is highly variable and still uncertain, as well as the interaction between these that could be either synergistic or antagonistic. In this work we use the coupled oceanographic-ecosystem model POLCOMS-ERSEM driven by climate forcing to study the interaction between climate change and OA. We focus in particular on primary production and nitrogen speciation. The model has been run in three different configurations in order to separate the impacts of ocean acidification from those due to climate change. The model shows significant interaction among the drivers and high variability in the spatial response of the ecosystem. Impacts of climate change and of OA on primary production have similar magnitude, compensating in some area and exacerbating in others. On the contrary, the direct impact of OA on nitrification is much lower than the one imposed by climate change.

1 Introduction

Climate change is exerting multiple stresses on marine ecosystems. The increase of temperature modifies the marine habitat generally encouraging migration of species towards higher latitudes (Beaugrand et al., 2002) with the possible extinction of some cold-adapted species (Hoegh-Guldberg and Bruno, 2010). Temperature further affects organism metabolism (e.g. reducing their thermal tolerance, Pörtner and Farrell, 2008) leading to multiple impacts on the ecosystem like shifts in community composition, e.g. towards smaller species (Perry et al., 2005; Morán et al., 2010). The warming of oceanic water will have also significant indirect effects, e.g. increasing of thermal stratification could lead to reduction of surface nutrient and primary production (Sarmiento et al., 2004; Holt et al., 2012a) and to increasing anoxia in the coastal ocean (Diaz and Rosenberg, 2008).

At the same time, the ocean has absorbed about half of the anthropogenic emissions of CO_2 (Sabine et al., 2004), altering the carbonate equilibrium causing a reduction of pH and other related parameters, known as Ocean Acidification (OA) (Caldeira and Wickett, 2003). Several Oceanic Global Circulation Models (OGCMs) have projected an average reduction of pH between 0.2–0.4 pH units by the end of the century (Orr, 2011) with the Polar Regions being most sensitive. Although the negative trend of pH is a consistent feature projected by all OGCMs, there is much uncertainty regarding the intensity of the OA signal and in particular of its spatial variability. Global models generally have a coarse spatial resolution and therefore they do not include an adequate representation of shelf and coastal sea processes, hence the uncertainty of these predictions on marginal seas is much higher. In the shelf sea environments, the OA signal can be enhanced or mitigated by many processes. Benthic processes, such as community respiration or biologically mediated redox reactions, are a significant source of Dissolved Inorganic Carbon (DIC) and Total Alkalinity (TA) and have a significant impact on the carbon budget and the pH (Thomas et al., 2009; Wakelin et al., 2012). The pelagic production – respiration balance can also impart significant seasonal and diurnal variability, as can the relatively large seasonal temperature signal. Finally, riverine inputs of DIC, TA and nutrients can be as significant or greater in determining the carbonate chemistry of coastal areas as the increasing atmospheric $p\mathrm{CO}_2$ (Borges and Gypens, 2010; Artioli et al., 2012). The variability of carbonate chemistry in shelf seas and their response to OA is consequently significant, hence specialised shelf sea models are required in order to give reliable estimates of the future state of the carbonate system in these environments.

Over the last five years, the number of studies focusing on the consequences of OA on marine chemistry and in particular on the biota has exponentially increased, but results have shown a high degree of variability and sometimes inconsistency as

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well. The main complications are the high degree of species-specific responses and the influence of environmental heterogeneity, resource availability and life stage. For example, a resource replete individual will be far more resilient than a stressed one (Melzner et al., 2011); larvae of bivalves or echinoderms are most sensitive to high CO_2 for the relatively brief period during which they settle and begin to develop their adult carbonate shells (Dupont et al., 2010; Gobler and Talmage, 2013).

As a result a consensus and comprehensive understanding of the effects of OA on the key biological processes has yet to emerge, although a number of synthesis papers are appearing. The international scientific community has generated some general conclusions (Riebesell et al., 2010), mainly drawn on the basis of meta-analysis of different experiments and observations (Doney et al., 2009; Hendriks et al., 2009; Liu et al., 2010; Kroeker et al., 2010, 2013) as follows:

- OA has a measurable impact on many processes spanning all levels of marine ecosystem, from chemistry to biology, from phytoplankton to fishes.
- These impacts show a high interspecific (and sometime intraspecific) variability.
- The magnitude of the impact on a single process or function varies significantly depending on the study. For the projected pH change at the end of the century the impact on single function can be relatively small (Hendriks et al., 2009), but the interaction between multiple affected processes or different levels of the ecosystem or multiple stressors (e.g. Pörtner, 2008) can magnify these effects.

Among all the observed impacts of OA on ecosystem processes, here we focused on two that are now strongly supported by observations (Gattuso et al., 2011).

Firstly Riebesell and Tortell (2011) concluded that the majority of experiments investigating the link between photosynthetic rate and CO_2 in the water showed an increase of Primary Production (PP) with CO_2, though the null hypothesis has not been totally discarded by rigorous meta-analysis (Doney et al., 2009; Hendriks et al., 2009).

Secondly the nitrogen cycle is particularly sensitive to OA as many of the transformations involved are pH dependent (Gehlen et al., 2011). For example with the increase of H^+ ions a shift towards ammonium has been observed in the ammonium–ammonia equilibrium (Wyatt et al., 2010), consequently the biological oxidation of ammonia to nitrite (nitrification) rate has been observed to decrease with pH in several locations (Beman et al., 2011; Huesemann et al., 2002).

In this paper we use a coupled hydrodynamic ecosystem model to estimate the combined impact of climate change (temperature, mixing and nutrient supply) and OA (changes in DIC, photosynthetic rate and nitrification) on the shelf ecosystem.

2 Material and methods

2.1 Model description

This study is focussed on the North Western European Shelf, bounded between latitude 40° N to 65° N and longitude 20° W to 13° E.

The physical model is the Proudman Oceanographic Laboratory Coastal Ocean Modelling System (POLCOMS), a finite difference model able to resolve the fundamental hydrodynamic processes important in shelf seas. The model grid has resolution of 1/9° latitude and 1/12° longitude (~ 12 km) with 42 s-levels in the vertical (Wakelin et al., 2009).

The ecosystem model coupled to POLCOMS is the European Regional Sea Ecosystem Model (ERSEM, Blackford et al., 2004), an ecosystem model based on Plankton Functional Types (PFTs) describing the lower trophic levels. It is a model of medium-high complexity, specifically designed for shelf seas, with independent cycling of the different nutrients (C, N, P, Si) and variable stoichiometry for phytoplankton, bacteria and small zooplankton. The model is able to simulate OA thanks to its carbonate system module (Blackford and Gilbert, 2007), recently developed to take into account the biological feedback and riverine influences on total alkalinity (Artioli et al., 2012). The coupled model has already proved to give good to acceptable simulations of many variables and ecosystem properties (Allen et al., 2007; Allen and Somerfield, 2009; Shutler et al., 2011). More details on the model and the validation of this particular set-up are provided by Artioli et al. (2012) and Holt et al. (2012a).

2.2 Parameterisation of OA impacts

2.2.1 Primary production

In the Pelagic Ecosystem CO_2 Enrichment Study III (PeECE III) mesocosm experiments (Bellerby et al., 2008) a clear relationship between relative carbon and nutrient uptake rates emerged and this relationship was tightly depending on the treatment values. This was used to derive a carbon enhancement factor (Cenh) to describe the observed overconsumption of carbon. This factor has been parameterised depending on atmospheric $p\mathrm{CO}_2$ (Eq. 1), as this is the closer proxy to the treatment values in the mesocosm experiment.

This factor has been applied to the specific gross primary production and consequently to the activity respiration that in the model is tightly correlated to the first one (Blackford et al., 2004).

$$C_{\mathrm{enh}} = 0.0005 \cdot \left(p\mathrm{CO}_{2,\mathrm{a}} - p\mathrm{CO}_{2,\mathrm{a}}\big|_{2005}\right) \quad (1)$$

$$g\mathrm{PP}_{\mathrm{enh}} = g\mathrm{PP} \cdot (1 + C_{\mathrm{enh}}) \quad (2)$$

$$\mathrm{act.resp.}_{\mathrm{enh}} = \mathrm{asct.resp.} \cdot (1 + C_{\mathrm{enh}}) \quad (3)$$

A similar relationship has been used previously to assess the response to carbon overconsumption on global primary production (Riebesell et al., 2007) and ensuing modifications to global oxygen distributions (Oschlies et al., 2008).

2.2.2 Nitrification

In ERSEM, the correlation between nitrification rate and pH is parameterised using the linear equation developed by Blackford and Gilbert (2007) based on observations of

Huesemann et al. (2002):

$$\mathrm{nit} = \mathrm{nit}_{\mathrm{ref}} \cdot \max(0; 0.6111\mathrm{pH} - 3.8889) \quad (4)$$

where nit is the nitrification rate, $\mathrm{nit}_{\mathrm{ref}}$ is the reference nitrification rate (set as $2.0\,\mathrm{mmolN\,m^{-3}\,d^{-1}}$), and pH is the actual pH value. This corresponds to halving the nitrification rate at approximately 7.2 pH units, and to a complete cessation below 6.4 units.

2.2.3 Model experiments

Four sensitivity simulations have been run in this work, each using atmospheric forcing and oceanic boundary conditions derived from the IPSL-CM4 climate model (Marti et al., 2006). We used a single member of the ensemble that is representative of the ensemble average response (L. Bopp, personal communication, 2010).

Scenario 1 (named $\mathrm{PD_{nit}}$) simulates present day conditions (1981–2000), using the standard ERSEM parameterisations with no primary production enhancement from $\mathrm{CO_2}$ and nitrification dependant on pH. The nitrification parameterisation is included as standard in the model in order to account for the for the impact of normal pH variability (Hofmann et al., 2011; Artioli et al., 2012) on the nitrogen cycle.

Scenario 2 (named $\mathrm{A1B_{nit}}$) is a far future simulation (2080–2099) that uses outputs of the same ensemble member run under the IPCC AR4 A1B scenario. Otherwise this is similar to PD with no primary production enhancement from $\mathrm{CO_2}$ and nitrification dependant on pH.

Scenario 3 (named $\mathrm{A1B_{nit,PP}}$) is identical to 2 with the enhancement of primary product at high $\mathrm{CO_2}$ activated.

Scenario 4 (named $\mathrm{A1B_{noOA}}$) is identical to 2 with neither nitrification nor primary production dependent on pH.

The diagnostic variables used are the seasonal mean over the simulation period of pH, aragonite saturation, net PP (calculated as particulate primary production), zooplankton biomass and the ammonium to total dissolved inorganic nitrogen ratio ($NH_4 : DIN$).

Net PP and zooplankton biomass are depth integrated on the whole water column, while all the other variables refer to surface values, where not differently specified. Seasonal means have been calculated as means of three consecutive months starting from January (winter = JFM, spring = AMJ, summer = JAS, autumn = OND).

3 Results

The results of all simulation have been analysed as follows in order to highlight the response of the ecosystem to different stressor and processes:

- Scenario $A1B_{nit}$ vs. scenario PD: they differ only in the forcing (present day compared with future) hence comparing their outputs allows to assess the potential impact of climate change, alongside the impacts of the nitrification sensitivity to pH.
- Scenario $A1B_{nit,PP}$ vs. scenario $A1B_{nit}$: these differ only in the inclusion of the PP response to high CO_2, enabling the impact of the potential enhancement of PP to be quantified.
- Scenario $A1B_{noOA}$ vs. scenario $A1B_{nit}$: these differ only in the inclusion of the nitrification response and allow an assessment of the sensitivity of the system to pH controlled nitrification.

3.1 Impact of high CO_2 on the ecosystem properties

First we look at the impact of high CO_2 on the shelf sea carbonate chemistry (Fig. 1). The model projects a mean decrease of surface pH of about 0.27 units over the whole domain by 2080–2099, but with significant spatial and temporal variability (Fig. 1a). Winter shows a clear difference between the shelf and the oceanic waters, with the former being more “acidified” by about 0.1 units. In spring the decrease in pH reaches its minimum almost in the entire domain, and the spatial gradient is lower. Acidification reaches its maximum in summer and autumn, with a pH decrease exceeding 0.4 units (equivalent to an approximate 2.5 times increase in hydrogen ion concentration). The shelf-ocean gradient is evident also looking at future surface aragonite saturation state (Fig. 1b): here the difference between the two basins is even more critical as the shelf sea surface waters are close to undersaturation during winter and autumn, with localised undersaturation in the German Bight. At the same time coastal waters in the southern North Sea are fully saturated during spring and summer. The vertical gradients on the shelf are also very important. During warm seasons, the water column in the central North Sea stratifies creating two contrasting environments: while the surface is oversaturated, bottom waters reach significant undersaturation ($\Omega \geq 0.7$) over a large area (Fig. 2). pH has a similar vertical gradient, with surface pH up to 0.3 units higher than in the bottom waters due mainly to differences in the production – respiration balance.

Figure 3a shows the difference in net PP driven by changes in temperature and physically mediated nutrient availability: the largest differences occur during the productive seasons when a general decrease in net PP is observed (up to $150\,\mathrm{mg\,C\,m^{-2}\,d^{-1}}$ in the northern oceanic regions in summer and $300\,\mathrm{mg\,C\,m^{-2}\,d^{-1}}$ in the southern oceanic region in spring). On the contrary, the model forecasts an increase of net PP in the Irish Sea, the Celtic Sea and the Icelandic waters, with the latter being the more productive area with an increase in spring up to $150\,\mathrm{mg\,C\,m^{-2}\,d^{-1}}$. In winter and autumn the differences are smaller with a tendency of positive values on the shelf and in the Bay of Biscay. Table 1 summarizes how these changes differ among the PFTs: diatoms and flagellates are barely impacted on an annual basis, but their phenology changes with an earlier bloom in March due to the increase in temperature. On the contrary picophytoplankton and dinoflagellates are the groups responsible for the drop in net PP during spring and summer.

Changes in total zooplankton biomass generally agree in sign with those of net PP (Fig. 3b) with one notable exception: in spring, in the Western Approaches (the oceanic areas west of UK and Ireland) a strong increase in zooplankton biomass (more than 500 $\mathrm{mg\,C\,m^{-2}}$) is observed despite the simultaneous decrease in PP. This decoupling can be explained looking at shift in zooplankton community composition towards the mesozooplankton (Table 1). This group is indeed omnivorous and therefore it can feed also on the grazers to sustain its growth. Furthermore, its dynamic are slower than the other groups and therefore a lag between its dynamic and that of primary producers is expected.

The difference in $NH_4:DIN$ (Fig. 3c) shows also a distinct seasonal cycle loosely following the changes in net PP: in winter the model projects a small increase in ammonium only in the southernmost area of the domain (passing from few percent to 10–15 % of DIN), and the opposite in spring when the rest of the basin shows a small increase. In summer the model projects a decrease in the ratio in the entire domain (with an average drop of 8 % in absolute terms, e.g. from 30 % of ammonium to 22 % of ammonium) while in autumn the differences are lower in absolute value and the positive and negative sign are equally present with the former more focused on the southern part of the domain. This reflects higher temperatures increasing recycling rates.

3.2 Impacts of the enhanced PP

The major consequences of this assumption is a general increase in net PP in most of the domain, particularly in spring, with an average increase around 100 $\mathrm{mg\,C\,m^{-2}\,d^{-1}}$, and in summer limited to the coastal area of the shelf, with peak of more than 200 $\mathrm{mg\,C\,m^{-2}\,d^{-1}}$. This latter pattern is generally associated with riverine discharges (Fig. 4a). In contrast, most of the open ocean experiences a reduction of net PP of about 50 $\mathrm{mg\,C\,m^{-2}\,d^{-1}}$ during summer. Again, not all phytoplankton groups respond in the same way, with Picophytoplankton having a remarkably higher increase in spring compared to the other PFTs (Table 1). Changes in Zooplankton biomass are tightly coupled to PP changes (Fig. 4b).

Changes in pH following from increased biological uptake of CO_2 (Fig. 4c) mimic the pattern of changes in PP. During spring a small increase of 0.016 pH unit on average is simulated compared to the non-PP enhancement scenario in most of the domain, with a maximum in river influenced regions of up to 0.1 units. In summer the coastal areas are still projected to have a higher pH than in the non-PP enhanced scenario, while the rest of the domain is experiencing a small decrease. Changes in saturation state follow the same pattern but they are statistically not separable from the one due to only climate. Finally, the feedback of the enhanced PP on the nitrogen speciation is low (few percent of maximum difference) with no clear pattern except a minor decrease of ammonium in summer.

3.3 Sensitivity of the ecosystem to pH-dependant nitrification

As expected, turning off the dependence of this process on pH will impact the $NH_4 : DIN$ ratio: this tends to decrease in winter and autumn particularly on shelf where the low pH slows down the nitrification in the A1Bnit scenario. However, the sensitivity of the $NH_4 : DIN$ ratio to pH changes is lower than to the enhanced PP and even lower than the one due to the climate change: we used as measure of the sensitivity the range between the 10th and 90th percentile of the difference over the whole domain and the entire 20 yr run. Turning off the pH correlation resulted in a change varying between −0.7 % and 0.5 % (in absolute terms), compared to a range between −2.2 % to 1.3 % due to enhanced PP and −4.1 to 2.7 % due to climate change.

Furthermore this variation on nitrogen speciation does not have significant impact on the rest of the ecosystem, with very small changes (if any) in net PP, zooplankton biomass, pH and aragonite saturation state.

4 Discussion

This works highlights how some potential impacts of ocean acidification in the shelf ecosystem are likely to show significant spatial and temporal heterogeneity and how they are closely interconnected with those of climate change. Although the average decrease of pH is consistent with the mean estimate provided by global modelling approaches, the impact is not homogeneous in time and space. Atmospheric CO_2 concentrations are clearly the driver of OA, but are not the only factor regulating the carbonate system in the shelf seas. Biological processes are particularly important in defining the seasonality of the impact: in the areas where an increase in net PP is projected relatively less acidification is observed (e.g. in spring in the Northern Atlantic or in the southern North Sea in summer). Similarly the undersaturation of aragonite projected in bottom waters in the central North Sea is caused by the accumulation of DIC during spring and summer due to community respiration (both pelagic and benthic) and the concurring stratification that prevents ventilation.

The increase in temperature plays two contrasting roles. From one side it decreases the solubility of CO_2 in water, promoting the outgassing and therefore preventing further acidification. This leads to a larger increase in atmospheric $p\mathrm{CO}_2$. At the same time temperature increases the values of the dissociation constants of the carbonic acid system leading to an increase in the more dissociated forms and a consequent decrease in pH. This effect is partially responsible of the self-ocean gradient in the OA signal: the shelf warms up more than the ocean, seen in both historical observations (Holt et al., 2012b) and these simulations, (Holt et al., 2012a) with differences in temperature varying between 1 °C and 4 °C depending on season and a relative difference in pH of up to 0.05 units.

Another concurring reason for the winter shelf/off-shelf gradient of the decrease in pH is linked to circulation and the freshwater balance: the nordic open ocean surface waters are projected to experience the smallest change in salinity contrarily to shelf sea waters where a significant freshening is projected. Given that TA is mostly parameterized as linear correlation of salinity following Millero (1995) (see Artioli et al., 2012, for further details on the parameterization), this gradient leads to a stronger decrease of TA on the shelf and therefore to lower buffer capacity of the shelf system.

Riverine inputs are another important source of variability. In the coastal areas affected by high riverine discharge, the combined effect of high primary production and high variability in the DIC and TA discharged by rivers lead to the highest seasonal variability in pH. It is also important to note that in all the scenarios only the freshwater discharge has been changed, but all concentrations (nutrients, dissolved inorganic carbon and total alkalinity) have been kept as in the present day. Clearly a change in riverine loadings could significantly modify the near coastal results presented here. The impact of enhanced PP on the carbonate system is relatively small, generally lower than the model uncertainty. Nevertheless in the coastal areas of the southern North Sea pH can increase up to 0.1 units because of the enhanced PP, almost halving the impact of OA in this region.

This high variability could be beneficial or a detrimental depending on species and processes. For example, the physiological stress due to hypercapnia can increase the metabolic cost for maintaining basic functions and consequently reduce the scope for growth (Stumpp et al., 2011). It is possible that organisms could benefit from the variability of OA signal, when the future pH is close to the present days the organisms may get some physiological respite allowing them to save energy or restore damaged functionality. Experiments involving transplantation of organisms from low pH areas to areas at normal pH around natural CO_2 vents have already demonstrated that this is possible (Rodolfo-Metalpa et al., 2011). In contrast, if the fluctuation in environmental conditions is high then critical thresholds could be breached more frequently, individuals could be permanently impaired even if this threshold is reached only for a short period of time. This could be the case, for instance, when transient undersaturation coincides with critical life history stages such as bivalve settlement.

Although the uncertainty on the impact of OA on PP is still high, and the parameterization used in this work preliminary, some key messages can be drawn from this study.

The potential change in net PP due to OA alone is of the same order of magnitude of the potential changes due to climate change. The heterogeneity indicates that sometimes these have the opposite sign and tend to cancel out (e.g. in the Central North Sea in spring or in the Southern North Sea in summer), whilst sometimes they have the same sign and OA magnifies the climate change impact. This is in agreement with work from Tagliabue et al. (2011) who tested similar hypotheses at a global scale. This impact is transferred up into trophic network to zooplankton and one could surmise potentially to higher trophic levels.

In this initial study we assumed that all phytoplankton functional groups (diatoms, flagellates, picophytoplankton and dinoflagellates) respond similarly to high CO_2 by increasing productivity with the same rate. However the phytoplankton community composition changed, with a higher increase in picophytoplankton compared to the other groups. This shift in community composition could change if we considered that there are some evidence that smaller species are more likely to respond to OA than the larger ones (Egge et al., 2009). Moreover, we also observed in some areas a decrease in net PP due to the following mechanism. Enhanced spring blooms remove a larger proportion of available nutrients from the off-shelf water columns which already contain less nutrients due to increased stratification (Holt et al., 2012a). Consequently a more nutrient depleted environment remains for summer production.

At domain scale this change in community composition tends to balance the shift due to climate change: the model simulates that the groups more responsible for the total decrease in PP due to climate change are the late bloomers, i.e. picophytoplankton and dinoflagellates.

The change in phytoplankton community composition is mirrored by the zooplankton community, with an equal increase of the two larger groups. The magnitude of the change is again of the same order of magnitude of the one induced by climate change, but with the opposite sign.

The effect of climate change (particularly the increased temperature and stratification) on the NH_4 : DIN ratio is more important than the direct and indirect effects of OA. The strongest signal occurs in spring with an increase in northern areas mostly due to decreased nitrate input from the open ocean, and a decrease in the southernmost part of the domain (and in the entire domain in summer) where the higher temperature enhances nitrification. Biological processes can modulate this effect particularly on the shelf where mineralization of organic matter can increase the ammonium concentration.

5 Conclusions

This paper shows how the impacts of OA and climate change are strongly interlinked: even when simple parameterizations are implemented, the combination of the effects can lead to complex results. OA could either exacerbate or compensate climate change impacts depending on season, area and process. Furthermore, the effects of these global drivers can magnify along the trophic network. This suggests that an ecosystem and multi-driver approach to study global changes is necessary, because we demonstrate that single driver-process responses can be negated or exacerbated by feedbacks from another driver-process response or ecosystem feedback.

This complexity enhances the heterogeneity of the system and of the impacts. Even considering a relatively simple chemical process like the carbonate equilibrium in the seawater, despite a homogeneous increase in atmospheric $p\mathrm{CO}_2$, the variability of pH decrease is significantly high due to the interaction with physical and biological processes. It is therefore crucial to highlight this variability when impacts of OA and climate change are assessed and so identify more or less vulnerable regions. This should be also taken into account in future experiments in order to expose the organisms to treatments that are closer to their future environment. Similarly, marine ecosystem models should consider more deeply the variability of the biological part of the ecosystem including improved physiological realism (Allen and Polimene, 2011) in order to be able to better represent the ecosystem response to global change and to better interface with experimental science.

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Table 1. Seasonal mean across the whole domain of the difference between the A1Bnit scenario and the PD ($A1B_{nit}$-PD) and the enhanced production run ($A1B_{nit,PP}$) and the $A1B_{nit}$ scenarios for primary production split into PFT and for the zooplankton biomass. Both data are depth integrated.

		Diatoms	Flagellates	Picophytopl.	Dinoflag.	Mesozoopl.	Microzoopl.	Het. nanofl.
		$mg\,C\,m^{-2}\,d^{-1}$	$mg\,C\,m^{-2}\,d^{-1}$	$mg\,C\,m^{-2}\,d^{-1}$	$mg\,C\,m^{-2}\,d^{-1}$	$mg\,C\,m^{-2}$	$mg\,C\,m^{-2}$	$mg\,C\,m^{-2}$
$A1B_{nit}$ -PD	Winter	10.68	4.03	9.49	0.83	69.45	5.81	−7.71
	Spring	−11.6	−1.5	−33.97	−21.4	150.97	−107.00	−29.36
	Summer	0.83	−4.73	−18.47	−19.68	−53.16	−244.2	−43.49
	Autumn	2.78	0.85	1.46	−0.07	−8.45	−68.97	−32.25
	Annual mean	0.67	−0.34	−10.37	−10.08	39.70	−103.59	−28.95
$A1B_{nit,PP}$- $A1B_{nit}$	Winter	5.77	3.95	13.01	1.12	28.06	24.48	11.33
	Spring	6.75	7.42	31.36	10.29	58.04	78.80	16.51
	Summer	2.62	−0.39	6.80	2.41	5.25	4.24	−1.92
	Autumn	1.48	1.69	6.36	1.93	25.34	13.41	4.61
	Annual mean	4.16	3.17	14.33	3.94	29.17	30.23	7.63

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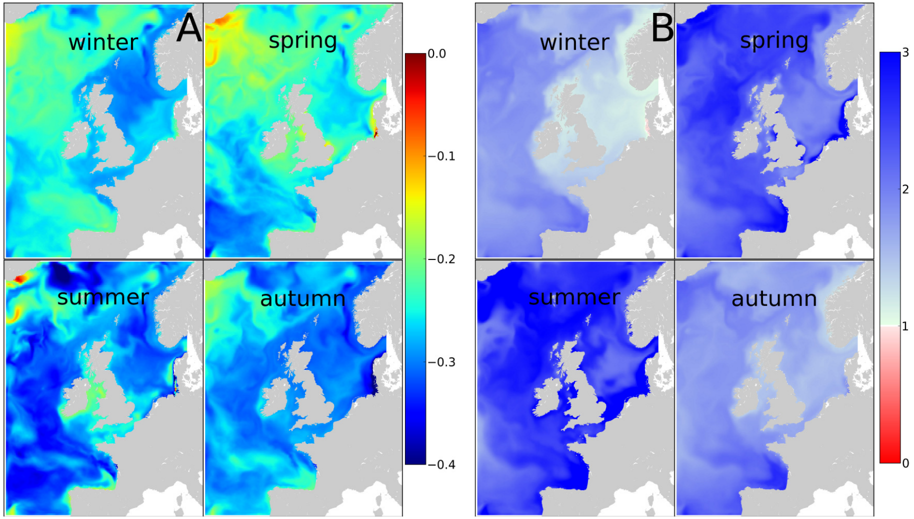

Fig. 1. Impacts of climate change and OA in the carbonate system as projected by the $A1B_{nit}$ scenario: on the left **(A)** absolute difference in surface pH compared to the present day scenario, on the right **(B)** future surface saturation state of aragonite.

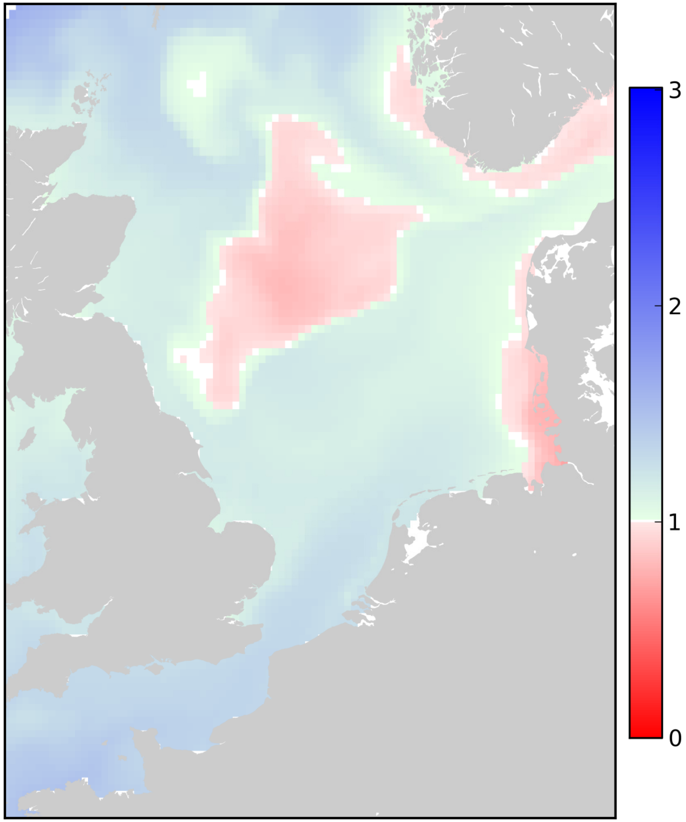

Fig. 2. Mean values of the monthly mean minimum of saturation state of aragonite in the bottom waters in the A1B$_{\mathrm{nit}}$ scenario. Red areas highlight undersaturation.

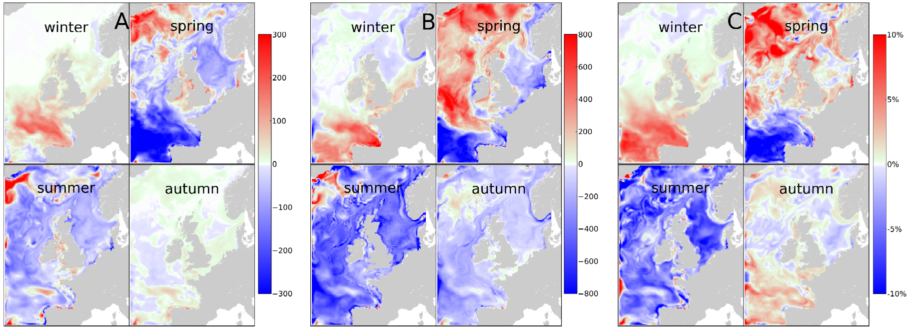

Fig. 3. Impacts of climate change and OA in the eco system as projected by the A1B$_{\text{nit}}$ scenario. **(A)** difference in net primary production (depth integrated, $\text{mg C m}^{-2}\,\text{d}^{-1}$), **(B)** difference in zooplankton biomass (depth integrated, mg C m^{-2}), **(C)** absolute difference in the $NH_4:DIN$ ratio in the surface waters (%).

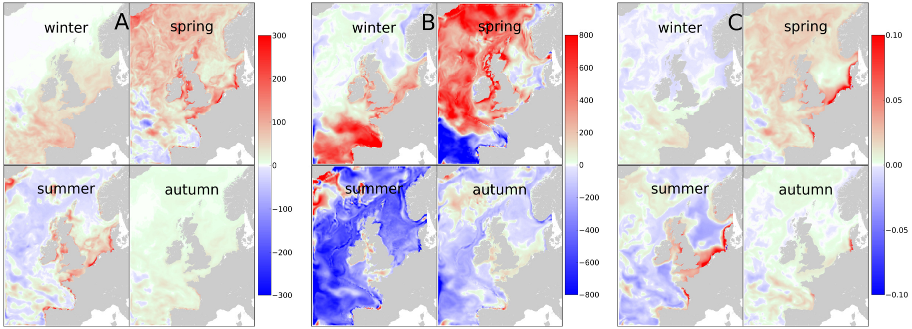

Fig. 4. Impacts of the enhanced primary production due to OA (differences between $A1B_{nit,PP}$ and $A1B_{nit}$ scenarios). **(A)** absolute difference in net primary production (depth integrated, $mg\,C\,m^{-2}\,d^{-1}$), **(B)** difference in zooplankton biomass (depth integrated, $mg\,C\,m^{-2}$), **(C)** difference in pH in the surface waters.