**Global Change Biology (8000 word limit, main body text)**

**Research Article**

**Changes in fish community structure in the Southern Ocean can be related to climate change and fisheries management**

**Fish community change in the Southern Ocean**

Joel Williams1 | Nicole Hill1 | Scott Foster2 | Skip Woolley3 | Philippe Ziegler4 | Kerrie Swadling1 | Kaitlin Naughton5 | Otso Ovaskainen6 | Craig Johnson1

1Institute for Marine and Antarctic Science, University of Tasmania, Hobart, Tasmania, Australia

2Data61, CSIRO, Hobart, Tasmania, Australia

3School of BioSciences, University of Melbourne, Parkville, Victoria, Australia

4Australian Antarctic Division, Kingston, Tasmania, Australia

5British Antarctic Survey, Cambridge, United Kingdom

6University of Helsinki, Helsinki, Finland

**Correspondence**

Joel Williams, Institute for Marine and Antarctic Science, University of Tasmania, 20 Castray Esplanade, Battery Point, Tasmania, Australia.

Email: joel.williams@utas.edu.au

**Funding information**

Australian Antarctic Science Program (Project No. 4501).

**Abstract**

Heard Island and McDonald Island are located within the northern half of the Kerguelen Plateau in the Southern Ocean. The Kerguelen Plateau is a biodiversity hotspot with many endemic fish species and the region has economic importance as it supports large fisheries. This region is also a climate change hotspot; water temperatures and oceans currents are changing and shifting. Most existing information on fish species and their distribution in this region is derived from annual random trawl surveys, fishery observations, and indirectly from research on fish predators. In this study we use trawl survey data and contemporary joint species distribution models to understand how the fish assemblages have changed through time and space. Using a hierarchical model of species community approach, we demonstrate that several species, including mackerel icefish, have had substantial changes in distribution and abundance through time and these changes could be related to environmental factors such as SST anomaly and southern annular mode. We found evidence of several species’ distributions responding to temperature variability and are potentially exposed to the ongoing impacts of climate change. This added information can be used by managers and policy makers to ensure sustainable fisheries and the protection of biodiversity into the future.

KEYWORDS

Joint species distribution, fish assemblage, marine park, trawl survey, climate change, fisheries management, ocean

**1 | INTRODUCTION**

The world’s oceans and their ecosystems are changing due to accumulating pressures such as climate change, extractive activities (fishing, mining etc), pollution (litter, land runoff etc) (Coll et al., 2008; Cooley et al., 2022; Hoegh-Guldberg & Bruno, 2010; Jennings & Kaiser, 1998; Pauly et al., 2005). The over exploitation of a single species or degradation of a habitat can have dire consequence at the ecosystem level (Coll et al., 2008; Jackson et al., 2001; Pauly et al., 1998; Worm et al., 2006). The methods for managing and protecting the world’s oceans has also changed and adaptation of marine protected areas, sustainable fishing policies, improved and reduced land-based pollution and runoff. Many of these policy decisions are targeted to specific regions or fishery. For example, changes to fisheries management has led to Australian orange roughy stocks to recover and the fishery becoming sustainable (Doonan et al., 2015; Kloser et al., 2015), and marine protected areas in New Zealand and Australia have demonstrated increases in fish abundance and biodiversity (Allard et al., 2022; Edgar et al., 2014, 2017; Knott et al., 2021).

Climate change adds another layer of complexity and uncertainty. Global ocean water temperatures are warming or cooling, pH is altering, CO2 sequestration is changing, ocean currents are shifting direction and intensity, and the timing and quantity of sea ice melt is changing and this is having ecosystem impacts (Boyd et al., 2016; Constable et al., 2014; Johnson et al., 2011; Wassmann et al., 2011). The changing oceans are resulting in changes to species distributions and leading to biological invasions (Johnson et al., 2011; Schickele et al., 2021). The measure of climate velocity represents the speed and direction of range shift in a species distribution (Arafeh-Dalmau et al., 2021). Understanding how these pressures and climate influence a species or whole communities is an important process to aid adoptive management, monitoring programs, and identify areas that require protection or greater management to avoid further loss (Arafeh-Dalmau et al., 2021; Emblemsvåg et al., 2022). Much of the contemporary marine research has a component that is aimed at understanding how these pressures and climate change is influencing the abiotic and biotic components of the world’s ecosystems (Wassmann et al., 2011).

The key to understanding these environmental processes and ecological change is to have suitable data to inform and provide information. For decades institutes and programs such as the National Oceanic and Atmospheric Administration (NOAA), the Australian Integrate Marine Observing System (IMOS), and Southern Ocean Observing System (SOOS) have been monitoring and collecting oceanographic data over large spatial and temporal scale. Long term (>10 years) biological monitoring or ecological datasets (i.e. species occurrence and abundance data) not as common but they are integral detecting long-term ecological changes. In the era of big data, increased computing capacity and innovative approaches to ecological modelling, researcher’s abilities to ask more complex questions and to model at larger spatial and temporal scales had become increasingly possible (Franklin et al., 2017; Tikhonov, Duan, et al., 2020). This includes working with whole species assemblage data across large spatial scales using joint species distribution models (Norberg et al., 2019; Ovaskainen et al., 2017; Tikhonov, Duan, et al., 2020). These modelling approaches allow ecologist to explore correlations across environmental gradients and produce full-coverage ecological maps for all species, species richness. Ecologist are also interested in using the same model for making future predictions under different scenarios (Evans, 2012).

The Southern Ocean represents a 10% of the world’s oceans and it plays a pivotal role in oceanic primary production, exports nutrients and oxygen the world’s ocean and support valuable biodiversity (Auger et al., 2021; Constable et al., 2014; Le Quéré et al., 2007; Van de Putte et al., 2021). However, the physical attributes the Southern Ocean, such as water temperature, CO2 sequestration, and ocean currents are changing (Constable et al., 2014; Van de Putte et al., 2021). The Conservation of Antarctic Marine Living Resources (CCAMLR), a leading management body for this region, have acknowledged the need to better incorporate the effects of climate change into decisions on resource, biodiversity, and ecosystem management in the Southern Ocean (Cresswell et al., 2021). The biodiversity of the Southern Ocean is unique and characterised by a high level of endemism in fish species (Constable et al., 2014). This is particularly driven by the sub-order Notothenioids, where 86% of species are endemic to this region (Eastman & McCune, 2000). This group of fishes is also the most abundant on the shelf regions of Antarctica and the Southern Ocean Island such as Heard and McDonald Islands. Both historical and predicted changes in Southern Ocean fish assemblages are mostly unknown. It is expected that the southward shifting ocean frontal systems is likely to have the largest influence on species distributions. However, interpretations of changes in species communities, distribution and abundances need to be careful in incorporating the confounding effects of fishing and fisheries management.

In the study we aimed to establish if the benthic fish community of the HIMI region of the Kerguelen Plateau has changed through time. We then relate any changes in prevalence and abundance of benthic fishes to environmental change, marine reserve zoning or changes to management or fishery practices. To achieve this we use a joint species distribution modelling approach. We were able establish how much of the variation in each species prevalence and abundance is due to environmental filtering and random process, and how these vary across spatial and temporal scales. This joint species modelling approach provides ecosystem level information on the impacts of climate change and fishing and will provide information to facilitate managers to the requirements of CCAMLR. The results from this study have the potential to be implied to other regions of the Southern Ocean and help better understand the implications of climate change, fisheries management, conservation management at a much larger scale.

The Kerguelen Plateau, located halfway between South Africa and Australia in the Indian Ocean sector of the Southern Ocean is a large dominant geographic feature. It is a productivity hotspot, supporting a diversity of marine life as well as supporting a lucrative demersal fishery (Duhamel & Welsford, 2011; Hill et al., 2017). The location and geography of the Kerguelen Plateau means it is highly exposed to the effects of climate change through warming waters and changing ocean currents and polar fronts. Knowledge and information on how climate change is influencing the ecosystem is important for the management and conservation of biodiversity and for managers to meet the obligations of the CCAMLR convention. The management of this region is split with France managing the northern half of the Plateau and Australian managing the central portion of the Plateau. Australia has an exclusive economic zone that encompasses Heard and McDonald Islands. The uniqueness and ecological importance of this region were globally recognised when Heard Island and McDonald Island were World Heritage Listed in 1997 and the formation of a no-take marine reserve in 2002. Species of key economic importance have been well studied for stock assessment and fisheries management. There is a significant knowledge gap of how the fish assemblage as a whole is structured across the Plateau and if and how it has changed through time (Hill et al., 2017). Within the Australian EEZ, Patagonian toothfish and mackerel icefish are the two targeted species of the Kerguelen Plateau. The fishery started as trawl fishery in the 1990s but a change to long-line fishing started in 2003 to maximise catches of Patagonian toothfish while minimising bycatch. There is still some trawling effort to target mackerel icefish. Historically this region was also exposed to significant illegal fishing. This has led to increase is surveillance of the area since the 1990s. Understanding how changes in management and fishery decisions and increased compliance have led to changes at a species assemblage level is important for ecosystem-based management to meets the principles of CCAMLR.

Currently, there is a distinct knowledge gap of how the Kerguelen Plateau benthic fish community is distributed and how it has changed through time. The Australian Government has run a long-term monitoring program that has been designed to be repeated using the same methods on an annual basis. In the study we aimed to establish if the benthic fish community of the HIMI region of the Kerguelen Plateau has changed through time. We then relate any changes in prevalence and abundance of benthic fishes to environmental change, marine reserve zoning or changes to management or fishery practices. To achieve this we use a joint species distribution modelling approach. We were able establish how much of the variation in each species prevalence and abundance is due to environmental filtering and random process, and how these vary across spatial and temporal scales. This joint species modelling approach provides ecosystem level information on the impacts of climate change and fishing and will provide information to facilitate managers to the requirements of CCAMLR. The results from this study have the potential to be implied to other regions of the Southern Ocean and help better understand the implications of climate change, fisheries management, conservation management at a much larger scale.

**2 | METHODS**

**2.1 | Study site**

This study was completed within the Australian EEZ of the Kerguelen Plateau (Figure 1). As the name suggests, it is a large is a large plateau the rises from depths >3000 m, has steep banks, with numerous seamounts, and breaks the surface with three large islands and numerous rocky outcrops. The data used in this study originates from around Heard Island and McDonald Island (HIMI) that are located on the northern half the Kerguelen Plateau (Figure 1). This region is managed and protected by the World Heritage Listed Heard Island and McDonald Island Marine Reserve (Figure 1). This included two areas that are listed as IUCN category 1A marine reserves. HIMI Marine Reserve was established in 2002 and the reserve boundaries were extended in 2014 to encompass 71,000 km2 of ocean. The HIMI region is also of economic interest as it supports valuable fisheries that are managed by Australia under the CCAMLR ecosystem-based management approach (Constable, 2011). Commercial fisheries primarily target Patagonion toothfish, *D. eleginoides*, by long-line fisheries and mackeral icefish, *C. gunnari*, by trawl fisheries. With other species such as *Notothenia rossii* and *Lepidonotothen squamifrons* being historically targeted by trawling (Duhamel & Williams, 2011).



Figure 1. A bathymetry map of the location of each research trawls from the random stratified trawl survey located on the Kerguelen Plateau. The grey-scaled dots represent the year the sample was collected and is scaled 2003 as white through to 20016 as black. The Heard Island and McDonald Island Marine Reserve boundaries are marked by the red polygons. Inset: Location of the study map in relation to the world.

**2.2 | Benthic fish data**

Benthic fish data used in the study comes from a long-term monitoring program established by the Australian Antarctic Division as fisheries independent dataset for monitoring changes in Patagonia toothfish, mackerel icefish and other bycatch managed species. This long-term monitoring program is referred to as the random stratified trawl survey (RSTS). The RSTS provides count data for benthic fish in the HIMI region of the Kerguelen Plateau. The RSTS has been conducted each year since 1997, when the fishery begun. The survey design consists of the HIMI plateau being divided into 10 strata that are defined by regions of similar morphology and commercial fishing effort. Sampling sites are then randomly located within each of these strata. The survey design and methodology has remained mostly the same since 1997, with a few minor adjustments. The consistency in methodology enables comparisons of fish abundance and biomass through time.

The RSTS uses a trawl net with a 500 mm mish codend liner is towed at a speed of 3 knots (1.5 m/s) for a duration of 30 minutes. Scientific observers are used to record catches using the same observer methods as the fishery. Priority is given to record data on toothfish, icefish and rays and skates but count and biomass data is available for all other species.

While every attempt is made to sample the same area, i.e. trawl tow duration, there is some variability in the distance of the trawls (min = 1.3 km, max = 6.4 km, mean = 2.9 km, SD = 0.3 km). Given the differences in towed distances, for the purposes of modelling, the count estimates are divided by the towed distance to provide CPUE (number of fish per kilometre).

Species that occurred in 10 or fewer trawl samples were removed as these species were considered very rare and difficult to model. Species that were considered pelagic species were also removed as the trawl surveys are designed to sample benthic fishes and there are likely to be biases in how pelagic species are collected in the trawl net.

**2.3 | Spatial and environmental variables**

We selected 15 spatial and environmental variables that could help explain any change in benthic fish prevalence, abundance, and community structure (Table 1). This included data derived from multibeam echo sounder, satellite, climate indices, and oceanographic models.

All environmental variables were tested for correlations using Pearsons correlation to ensure the coefficients between variable was <0.7 to avoid model overfitting.

Table 1. Descriptions for each of the covariates used in the HMSC model, including the range and justification for inclusion.

|  |  |  |  |
| --- | --- | --- | --- |
| **Variable name** | **Range** | **Description** | **Source** |
| Depth | 150-1340 m | Depth (m) at location of the research trawl. | RSTS metadata |
| Slope | 0.00 – 0.75 | Slope calculated from the bathymetry data (100m resolution) using the terrain() function in the R raster package | GeoScience Australia |
| Southern annular mode (SAM) | -2.61 – 3.02 |  |  |
| Indian Ocean dipole (IOD) | -0.49 – 0.56 |  |  |
| Mean monthly sea surface temp. | 1.0 – 4.5 | The monthly mean optimal interpolation SST. Provided by NOAA at 1/4° resolution. The SST data incorporates different platforms, satallites, ships, buoys and argo floats and calculated into regular grids. | NOAA |
| Mean monthly sea surface temp. anomaly | -0.57 – 1.00 | The monthly mean optimal interpolation SST anomoly. Provided by NOAA at 1/4° resolution. The SST anomaly is the difference from the 30 year mean and the data incorporates different platforms, satallites, ships, buoys and argo floats and calculated into regular grids. | NOAA |
| Mean monthly sea surface height | -0.78 - -054 | The monthly mean optimal interpolation sea surface height (SSH). Provided by NOAA at 1/4° resolution. | NOAA |
| Mean monthly Chlorophyl A | 0.02 – 1.82 |  |  |
| Seafloor water temp. | 0.79 – 2.07 | FESOM | FESOM |
| Seafloor current speed | 0 – 0.06 | FESOM | FESOM |
| Marine reserve status | Fished / No-take | The zoning status for the HIMI Marine Reserve established in 2002. | Australian Antarctic Division. |
| Year | 2003 – 2006 | The years we had RSTS with reliable taxonomic resolution and coverage of the above environmental variables | RSTS metadata. |

**2.4 | Statical analyses**

We analysed the RSTS data using Hierarchical Modelling of Species Communities (HMSC) (Ovaskainen et al., 2017; Ovaskainen & Abrego, 2020). HMSC is a joint species distribution model (Warton et al., 2015) that includes a hierarchical layer asking how a species response to environmental covariates depend on species-species and species traits relationship (Abrego et al., 2017). HMSC also utilizes spatially structure latent variables proposed by Ovaskainen et al. (2017) and later expanded to big spatial data by Tikhonov et al. (2020) (i.e. 10,000s points). This approach allows us to investigate how each individual species responds to the environmental covariates.

The RSTS data comprises of the abundance of 38 fish, shark and ray species from between 111 and 195 trawl sites sampled per year from 2003 to 2016. We excluded those species that had less than 10 occurrences in these data, and thus our analyses consist of . [sampling units and response variable]. Each individual trawl was treated as a sampling unit, thus there being sampling units. As the response variable (the matrix of HMSC; see Ovaskainen et al. 2017 ELE), we used the CPUE data for each of the 35 species. Due to the zero-inflated nature of the data, we applied a hurdle model, i.e. one model for presence-absence and another one for abundance conditional on presence (henceforth abundance COP model). We applied probit regression in the presence-absence model, and linear regression for transformed count data in abundance COP model. The count data were transformed by declaring zeros as missing data, log-transforming, and then scaling the data to zero mean and unit variance within each species.

We included as fixed effects (the matrix **X** of HMSC; see Ovaskainen et al. 2017 ELE; where is the number of covariates) 15 variables that we believe will explain the presence and abundance of the species assemblage through time (Table 1). An advantage of HMSC is that is utilises a sample level random factor to account for spatial correlation. Due to the large number of samples in our study we used a nearest neighbour Gaussian process (Tikhonov, Duan, et al., 2020). Due to the study design and the uneven sampling effort across strata, we also included strata as a random factor. To control for temporal correlation we used the numbers of days since the start of the study (i.e. 1 January 2003 = 1).

We fitted the HMSC model using the R-package Hmsc v3.0-12 (Tikhonov, Opedal, et al., 2020) assuming the default prior distributions (see Chapter 8 of Ovaskainen and Abrego 2020). We sampled the posterior distribution with four Markov Chain Monto Carlo (MCMC) chains. The chains were thinned 100 to yield 500 posterior samples per chain and so generating 2000 posterior samples in total. We examined MCMC convergence by examining the potential scale reduction factors (Gelman & Rubin, 1992) of the model parameters.

We examined the explanatory and predictive powers of the probit model through species specific AUC (Pearce & Ferrier, 2000) and Tjur’s R2 (Tjur, 2009) values. The explanatory and predictive powers of the abundance COP model were measured by R2. To compute explanatory power, we made model predictions based on models fitted to all data. To compute predictive power, we performed 2-fold cross validation, in which the sampling units were assigned randomly to two folds, and prediction for each fold were based on model fitted to data on the remaining fold.

To quantify the drivers of community structure, we partitioned the explained variation among the fixed and random effects included in the model. To address our main study question, i.e. if and how species communities have changed over the study period, we examined species responses to the explanatory variables of the study period, counting what proportion of species showed a positive or negative response with at least 95 % posterior probability. We also examined the species – species interactions, referred to as omega values within the HMSC framework. HMSC estimates the species correlations that

**3 | RESULTS**

Catch per unit effort data (CPUE) from 2,186 trawls from 2003 to 2016 were used is this study. These data included CPUE data for 35 species, after accounting for the removal of very rare species (observed in < 10 trawls). The 35 species included 29 ray-finned fish and 6 Chondrichthyes. The species assemblage was dominated by three species, *Champsocephalus gunnari*, *Dissostichus eleginoides*, *Channichthys rhinoceratus*, that represented >70% of the total CPUE across all years. *Dissostichus eleginoides* was the most ubiquitous species being present in 91% of samples, follow by *Channichthys rhinoceratus* that was present in 56% of samples.

The MCMC convergence of the HMSC presence absence model was satisfactory: The potential scale reduction factors for the -parameters (that measure the responses of the species to environmental covariates; Ovaskainen et al. 2017 ELE) were on average 1.03 (maximum 1.55) for the presence-absence model and for the abundance COP a mean value of 1.00 (maximum 1.01).

The presence-absence model showed a variable fit to the data, the Tjur R2 values ranged from 0.06 to 0.78 among species, with the mean being 0.33 (SD 0.22) (Figure 2a), The AUC values ranged from 0.75 to 0.99 among species, with the mean being 0.94 (SD 0.06). On average, the fixed components of the presence absence model explained 77% of the explained variance for each species (Figure 2b). Specifically, bathymetry related variables explained on average 33% of the variance across species, satellite derived data 15%, climate related variables 14%, year 9%, FESOM modelled variables 4%, and marine reserve zoning 0.5% (Figure 2b). The abundance COP model showed reasonable fit to the data, the R2 values ranges from 0.17 to 0.98 among species, the mean being 0.48 (SD 0.23, Figure 2c). On average the fixed components of the abundance COP model explained 79% of the explained variance for each species (Figure 2d). Specifically, bathymetry related variables explained on average 26% of the variance across species, satellite derived data 24%, climate related variables 15%, satellite derived data 24%, FESOM modelled variables 8%, year 6%, and marine reserve zoning 1% (Figure 2d).

Accounting for only responses that were positive or negative with at least 95% posterior probability, in the presence-absence model 80% of species showed a positive increase in prevalence across the study from 2003 to 2016 (Figure 2a). No significant declines in species prevalence were detected during this study. The abundance COP model had 25 % of species showing a significant increase in abundance (Figure 2b). No species were predicted to decline in abundance. Depth and slope were also important at explaining the prevalence of species with for 71% and 20% of species respectively showing significant effect sizes (Figure 3a). A positive response, higher prevalence in deeper water, was observed for 49% of species. While a negative response, higher prevalence in shallow waters was observed for 23% of species. Most depth response also had a significant second degree highlighting that many species have well defined depth ranges. For example, *C. gunnari* would only be observed in depth <500m and conversely *Macrourus* sp. Would only be seen in depths >500m (Figure S1). Depth was also most important at describing abundance with 34 % of species having a negative or positive response (Figure 3b). Slope was only significant for 11% of species (Figure 2b).

At the climate level, IOD and SAM had variable influence on the occurrence of benthic fish species (Figure 3). Almost half of the species included in the presence absence model had a significant positive or negative response to IOD or SAM (Figure 3a). For abundance the effect wasn’t as strong with six species significantly (three positive, 3 negative) responding to IOD and six species positively responding to SAM (Figure 3b). One third of the species assemblages had positive relationship with SST for presence absence (Figure 3a). However, there were no significant responses in the abundance COP model (Figure 3b). SST anomaly had a mixed response for the presence absence model with three species having a significant positive relationship and eight species having a negative relationship (Figure 3a). SSH, Chlyrophyl A, and the FESOM model variables have a notably smaller significant responses for the presence absence model (Figure 3b). However, these variables were more significant for the abundance COP model. Their responses were highly variable and species specific.

The presence of six benthic fish species were high inside the no-take marine reserves, including a fishery managed species *Lepidonotothen mizops* and *L. squamifrons* (Figure 3a). status (no-take zone v open zone). The abundance of *Gobionotohen acuta* and *Zanchlorhynchus spinifer* where significantly more abundant inside the no-take marine reserve (Figure 3b).

There were several significant species-species interactions (Figure 3). These are either positive or negative correlations between species that cannot be explained by the predictor variables. At the individual trawl level

Species richness was predicted to increase through time with a mean species richness increasing from three species per trawl in 2003 to seven species per trawl in 2016 (Figure 5). Species richness also decreased with depth from a mean of 8 species per a trawl in 200 m depth to 5 species per trawl in 1,000 m depth (Figure 5). There was a positive correlation with species richness and SST with species richness more than doubling over 3 degrees of water temperature (Figure 5).

Chart, timeline

Description automatically generated

Figure 2. A) The Tjur R2 values for each species from the presence absence HMSC. B) The variation partitioning of the fixed and random effect within the presence absence HMSC. C) The R2 values for each species from the abundance COP HMSC. D) The variation partitioning of the fixed and random effect within the abundance COP HMSC. The species names are listed in alphabetical order.

Chart, scatter chart

Description automatically generated

**Figure 3.**  The responses of the species to environmental covariates. Panel A showed the results for the presence-absence model and panel B for the abundance COP model. In both panels, responses that are positive with at least 95 % posterior probability are shown in red, response that are negative with at least 95 % posterior probability are shown in blue, and responses that did not gain strong statistical support are shown in white. The species are ordered according to their phylogeny as illustrated by the phylogenetic tree show in the panels.

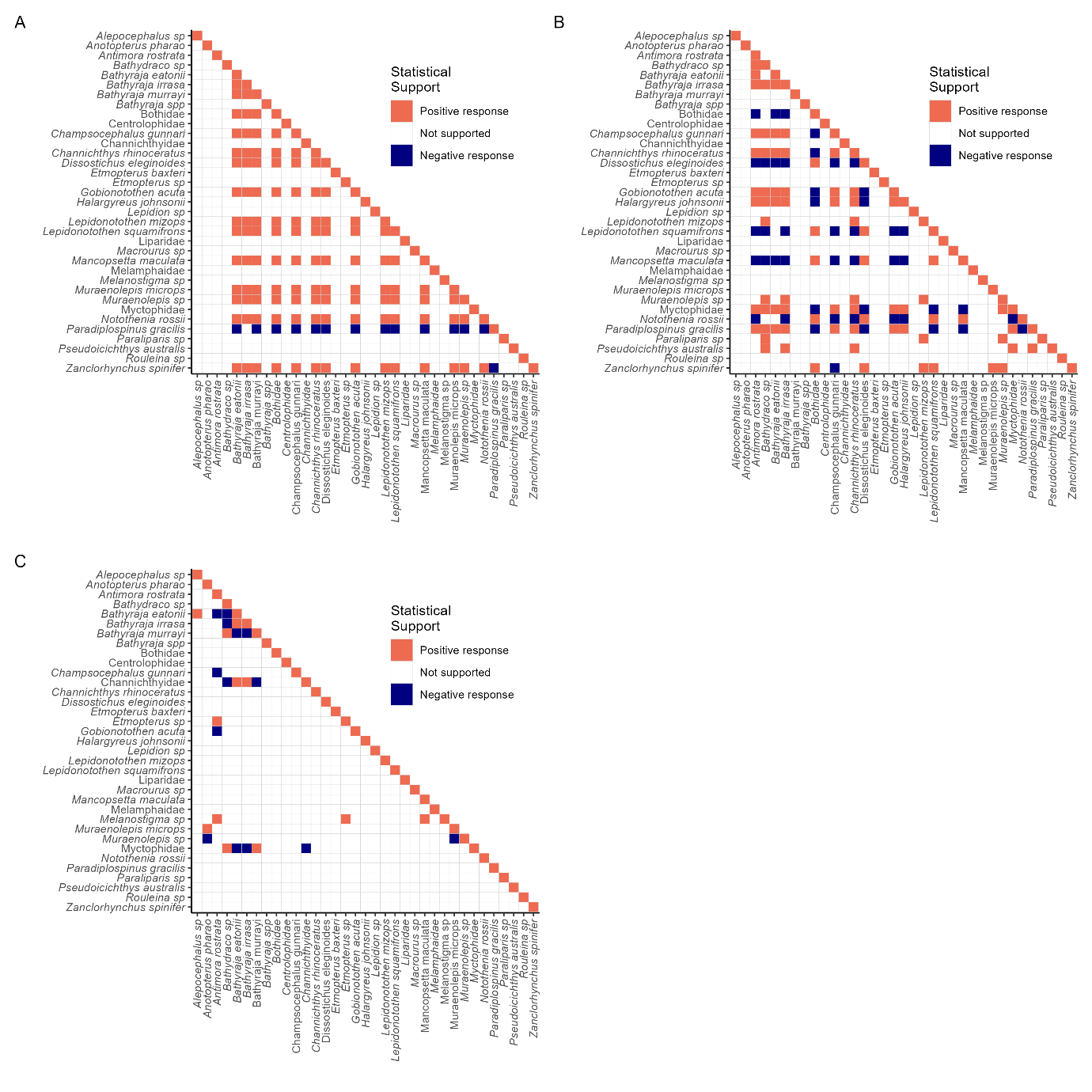
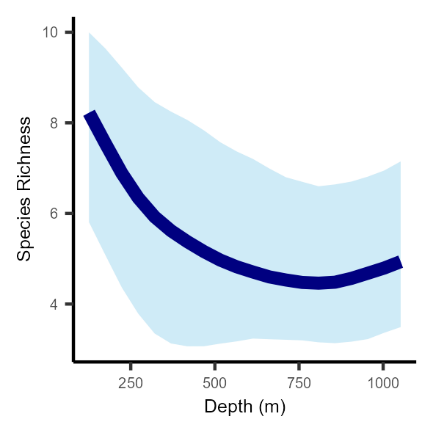
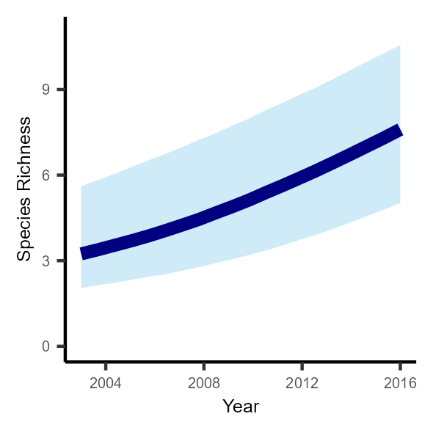


Figure 4. Modelled species – species interaction plots at various spatial and temporal scales; 1) at the individual trawl level, 2) at the strata level, and 3) at the days since start of study level.

Chart, line chart

Description automatically generated

Figure 5. Predictions of species richness

**4 | Discussion**

The Southern Ocean climate is changing and the management, compliance, and fishery led changes in operating have also changed, combined this is influencing the benthic fish community. In this study we relate seascape, climate and oceanography with the prevalence and abundance of benthic fishes. We demonstrated that benthic fish species richness has been increasing since 2003. Furthermore, most species of benthic fish have increased in prevalence and abundance across the study area. With the available data and modelling approach it is difficult to disentangle the specific in what is driving this increase in specie richness, prevalence, and abundance. We hypothesise that it is likely due to combination of the cessation of illegal fishing, a change in fish practices, fisheries management, and the changing environmental conditions that favour productivity in this region.

In 2003 the commercial fishery operating in this area started moving away from trawl-based fishery to long-line fishery to increase catches of Patagonian toothfish while decreasing by-catch. This change was driven by the fishery itself to maximise efficiency and minimise bycatch. It is undoubtable that seabed trawling has an impact on benthic habitats and recovery times from trawl impacts can be varied from years to decades (Pitcher et al., 2015, 2022). It is plausible that the reduction trawl fishing has allowed the recovery of benthic habitats to the benefit of benthic fishes. It is also plausible that small changes in environmental conditions, for example higher SST, have increased productivity to the benefit if benthic fishes. It is suggested that warming waters in high latitude areas is resulting in increase in primary productivity due to reduce water column mixing (Doney, 2006). Chlorophyll A from satellite data was included in the model and showed mixed results for prevalence and little explanation for abundance(Doney, 2006). However, we needed monitoring to provide measurements of other data such as zooplankton abundance to better understand the role of climate change on influencing productivity of benthic fishes in the Southern Ocean.

Using a joint species distribution modelling framework, we established a significant effect of year on the prevalence and abundance of many benthic fish species across the Australian EEZ of the Kerguelen Plateau. Furthermore, the model predicts an increase in species richness across the study site during study period. Notably, there was no evidence of any decrease in prevalence or abundance for any of the benthic fish species included in the model. These changes were predicted to occur between the years 2003 and 2016. Coincidently in 2003 the demersal fishery in this region started changing from trawl dominated fishery to long-line dominated fishery (REF). This was brought by the fishery wanted to maximise catches of targeted Patagonian toothfish, while minimising bycatch. There is also evidence of an environmental signal influencing prevalence and abundance. It is difficult to disentangle to causal mechanisms driving these increases. However, evidence suggests that removal of trawling is likely to benefit demersal species through habitat recover and reduction in bycatch (refs for removal of trawling). Without baseline, pre-fishing, data, and knowledge it is difficult to establish if this indicates a return to pre-trawling biogeography of demersal fish on the Kerguelen Plateau. In the instance of the Australian EEZ on the Kerguelen Plateau, it is feasible that the combination of fisheries management and voluntary changes by the fishers has outweighed any impacts of the changing climate. Though it can not be ruled out if some species have benefited or prefer the changing climate, warming waters etc. These results highlight the importance of understanding the influence of resource management coupled with the changing climate and oceans.

The Patagonian toothfish and mackerel icefish fisheries are highly managed fisheries. Quotas for fishing within the Australian EEZ or HIMI region are set and reviewed annually by AFMA and CCAMLR. >>>>>>> Fisheries management coupled with spatial management (i.e marine protected areas or marine reserves). Marine reserves are a viable and effective adaptation strategy to improve the outlook of climate change (Roberts et al., 2017).

Many of the species that make up benthic fish assemblage are poorly understood with data o

Other factors the influence the biogeography . Depth had the greatest influence on prevalence and abundance of most species. This result highlights the fact most species have a specific or preferred habitat defined by depth. Many of the species observed in these data are data poor. Very little is known about the biology, ecology, and distribution of these species. For many species, this is the first exploration of spatial and temporal trends. This makes it difficult to understand their vulnerability to pressures from climate change and fishing.

See (Le Quéré et al., 2007) for understanding about the role of SAM in climate change.

IPCC 2022 report suggests a climate driven ecosystem-based management framework is not need to measure and reduce pressures to adapt to climate change.

This study demonstrates the benefit and value of the annual random stratified trawl survey, a fisheries independent survey. Without long-term data collected using the same comparable methodology it would be possible to detect these changes accurately and confidently. The random stratified component ensures adequate special coverage and avoids sampling biases that increase or decrease effect sizes.

In conclusion, in this study we have demonstrated that improving fish practices that not only improve the fishery but also improve biodiversity over large geographical areas. The use of joint species distribution models has allowed us to demonstrate the benefits over a species assemblage scale, including species that previously haven’t been investigated. This type of information and knowledge is vital for organisations such as CCAMLR

REFERENCES

Abrego, N., Norberg, A., & Ovaskainen, O. (2017). Measuring and predicting the influence of traits on the assembly processes of wood‐inhabiting fungi. *The Journal of Ecology*, *105*(4), 1070–1081.

Allard, H., Ayling, A. M., & Shears, N. T. (2022). Long-term changes in reef fish assemblages after 40 years of no-take marine reserve protection. *Biological Conservation*, *265*, 109405.

Arafeh-Dalmau, N., Brito-Morales, I., Schoeman, D. S., Possingham, H. P., Klein, C. J., & Richardson, A. J. (2021). Incorporating climate velocity into the design of climate‐smart networks of marine protected areas. *Methods in Ecology and Evolution / British Ecological Society*, *12*(10), 1969–1983.

Auger, M., Morrow, R., Kestenare, E., Sallée, J.-B., & Cowley, R. (2021). Southern Ocean in-situ temperature trends over 25 years emerge from interannual variability. *Nature Communications*, *12*(1), 1–9.

Boyd, P. W., Cornwall, C. E., Davison, A., Doney, S. C., Fourquez, M., Hurd, C. L., Lima, I. D., & McMinn, A. (2016). Biological responses to environmental heterogeneity under future ocean conditions. *Global Change Biology*, *22*(8), 2633–2650.

Coll, M., Libralato, S., Tudela, S., Palomera, I., & Pranovi, F. (2008). Ecosystem overfishing in the ocean. *PloS One*, *3*(12), e3881.

Constable, A. J. (2011). Lessons from CCAMLR on the implementation of the ecosystem approach to managing fisheries. *Fish and Fisheries* , *12*(2), 138–151.

Constable, A. J., Melbourne-Thomas, J., Corney, S. P., Arrigo, K. R., Barbraud, C., Barnes, D. K. A., Bindoff, N. L., Boyd, P. W., Brandt, A., Costa, D. P., Davidson, A. T., Ducklow, H. W., Emmerson, L., Fukuchi, M., Gutt, J., Hindell, M. A., Hofmann, E. E., Hosie, G. W., Iida, T., … Ziegler, P. (2014). Climate change and Southern Ocean ecosystems I: how changes in physical habitats directly affect marine biota. *Global Change Biology*, *20*(10), 3004–3025.

Cooley, S., Schoeman, D., Bopp, L., Boyd, P., & Donner, S. (2022). Oceans and Coastal Ecosystems and their Services. *IPCC AR6 WGII*. https://epic.awi.de/id/eprint/56137/

Cresswell, I. D., Janke, T., & Johnston, E. L. (2021). *Australia state of the environment 2021: overview, independent report to the Australian Government Minister for the Environment, Commonwealth of Australia, Canberra*. https://www.researchgate.net/profile/Ian-Cresswell/publication/365450543\_soe2021-overview/links/6375db5454eb5f547cdc178b/soe2021-overview.pdf

Doney, S. C. (2006). Oceanography: Plankton in a warmer world [Review of *Oceanography: Plankton in a warmer world*]. *Nature*, *444*(7120), 695–696. nature.com.

Doonan, I. J., Fu, D., & Dunn, M. R. (2015). Harvest control rules for a sustainable orange roughy fishery. *Deep Sea Research Part I: Oceanographic Research Papers*, *98*, 53–61.

Duhamel, G., & Welsford, D. (2011). *The Kerguelen Plateau: marine ecosystem and fisheries*. Société française d’ichtyologie.

Duhamel, G., & Williams, R. (2011). History of whaling, sealing, fishery and aquaculture trials in the area of the Kerguelen Plateau. *The Kerguelen Plateau: Marine Ecosystem and Fisheries*, *2011*, 15–28.

Eastman, J. T., & McCune, A. R. (2000). Fishes on the Antarctic continental shelf: evolution of amarine species flock? *Journal of Fish Biology*, *57*(sa), 84–102.

Edgar, G. J., Stuart-Smith, R. D., Thomson, R. J., & Freeman, D. J. (2017). Consistent multi-level trophic effects of marine reserve protection across northern New Zealand. *PloS One*, *12*(5), e0177216.

Edgar, G. J., Stuart-Smith, R. D., Willis, T. J., Kininmonth, S., Baker, S. C., Banks, S., Barrett, N. S., Becerro, M. A., Bernard, A. T. F., Berkhout, J., Buxton, C. D., Campbell, S. J., Cooper, A. T., Davey, M., Edgar, S. C., Försterra, G., Galván, D. E., Irigoyen, A. J., Kushner, D. J., … Thomson, R. J. (2014). Global conservation outcomes depend on marine protected areas with five key features. *Nature*, *506*(7487), 216–220.

Emblemsvåg, M., Werner, K. M., Núñez-Riboni, I., Frelat, R., Torp Christensen, H., Fock, H. O., & Primicerio, R. (2022). Deep demersal fish communities respond rapidly to warming in a frontal region between Arctic and Atlantic waters. *Global Change Biology*. https://doi.org/10.1111/gcb.16113

Evans, M. R. (2012). Modelling ecological systems in a changing world. *Philosophical Transactions of the Royal Society of London. Series B, Biological Sciences*, *367*(1586), 181–190.

Franklin, J., Serra-Diaz, J. M., Syphard, A. D., & Regan, H. M. (2017). Big data for forecasting the impacts of global change on plant communities. *Global Ecology and Biogeography: A Journal of Macroecology*, *26*(1), 6–17.

Gelman, A., & Rubin, D. B. (1992). Inference from Iterative Simulation Using Multiple Sequences. *Schweizerische Monatsschrift Fur Zahnheilkunde = Revue Mensuelle Suisse d’odonto-Stomatologie / SSO*, *7*(4), 457–472.

Hill, N. A., Foster, S. D., Duhamel, G., Welsford, D., Koubbi, P., & Johnson, C. R. (2017). Model-based mapping of assemblages for ecology and conservation management: A case study of demersal fish on the Kerguelen Plateau. *Diversity & Distributions*, *23*(10), 1216–1230.

Hoegh-Guldberg, O., & Bruno, J. F. (2010). The impact of climate change on the world’s marine ecosystems. *Science*, *328*(5985), 1523–1528.

Jackson, J. B., Kirby, M. X., Berger, W. H., Bjorndal, K. A., Botsford, L. W., Bourque, B. J., Bradbury, R. H., Cooke, R., Erlandson, J., Estes, J. A., Hughes, T. P., Kidwell, S., Lange, C. B., Lenihan, H. S., Pandolfi, J. M., Peterson, C. H., Steneck, R. S., Tegner, M. J., & Warner, R. R. (2001). Historical overfishing and the recent collapse of coastal ecosystems. *Science*, *293*(5530), 629–637.

Jennings, S., & Kaiser, M. J. (1998). The Effects of Fishing on Marine Ecosystems. In J. H. S. Blaxter, A. J. Southward, & P. A. Tyler (Eds.), *Advances in Marine Biology* (Vol. 34, pp. 201–352). Academic Press.

Johnson, C. R., Banks, S. C., Barrett, N. S., Cazassus, F., Dunstan, P. K., Edgar, G. J., Frusher, S. D., Gardner, C., Haddon, M., Helidoniotis, F., Hill, K. L., Holbrook, N. J., Hosie, G. W., Last, P. R., Ling, S. D., Melbourne-Thomas, J., Miller, K., Pecl, G. T., Richardson, A. J., … Taw, N. (2011). Climate change cascades: Shifts in oceanography, species’ ranges and subtidal marine community dynamics in eastern Tasmania. *Journal of Experimental Marine Biology and Ecology*, *400*(1), 17–32.

Kloser, R. J., Sutton, C., Krusic-Golub, K., & Ryan, T. E. (2015). Indicators of recovery for orange roughy (Hoplostethus atlanticus) in eastern Australian waters fished from 1987. *Fisheries Research*, *167*, 225–235.

Knott, N. A., Williams, J., Harasti, D., Malcolm, H. A., Coleman, M. A., Kelaher, B. P., Rees, M. J., Schultz, A., & Jordan, A. (2021). A coherent, representative, and bioregional marine reserve network shows consistent change in rocky reef fish assemblages. *Ecosphere* , *12*(4). https://doi.org/10.1002/ecs2.3447

Le Quéré, C., Rödenbeck, C., Buitenhuis, E. T., Conway, T. J., Langenfelds, R., Gomez, A., Labuschagne, C., Ramonet, M., Nakazawa, T., Metzl, N., Gillett, N., & Heimann, M. (2007). Saturation of the southern ocean CO2 sink due to recent climate change. *Science*, *316*(5832), 1735–1738.

Norberg, A., Abrego, N., Blanchet, F. G., Adler, F. R., Anderson, B. J., Anttila, J., Araújo, M. B., Dallas, T., Dunson, D., Elith, J., Foster, S. D., Fox, R., Franklin, J., Godsoe, W., Guisan, A., O’Hara, B., Hill, N. A., Holt, R. D., Hui, F. K. C., … Ovaskainen, O. (2019). A comprehensive evaluation of predictive performance of 33 species distribution models at species and community levels. *Ecological Monographs*, *89*(3), e01370.

Ovaskainen, O., & Abrego, N. (2020). *Joint Species Distribution Modelling: With Applications in R*. Cambridge University Press.

Ovaskainen, O., & Soininen, J. (2011). Making more out of sparse data: hierarchical modeling of species communities. *Ecology*, *92*(2), 289–295.

Ovaskainen, O., Tikhonov, G., Norberg, A., Guillaume Blanchet, F., Duan, L., Dunson, D., Roslin, T., & Abrego, N. (2017). How to make more out of community data? A conceptual framework and its implementation as models and software. *Ecology Letters*, *20*(5), 561–576.

Pauly, D., Christensen, V., V., Dalsgaard, J., Froese, R., & Torres, F., Jr. (1998). Fishing down marine food webs. *Science*, *279*(5352), 860–863.

Pauly, D., Watson, R., & Alder, J. (2005). Global trends in world fisheries: impacts on marine ecosystems and food security. *Philosophical Transactions of the Royal Society of London. Series B, Biological Sciences*, *360*(1453), 5–12.

Pearce, J., & Ferrier, S. (2000). Evaluating the predictive performance of habitat models developed using logistic regression. *Ecological Modelling*, *133*(3), 225–245.

Pitcher, C. R., Ellis, N., Venables, W. N., Wassenberg, T. J., Burridge, C. Y., Smith, G. P., Browne, M., Pantus, F., Poiner, I. R., Doherty, P. J., Hooper, J. N. A., & Gribble, N. (2015). Effects of trawling on sessile megabenthos in the Great Barrier Reef and evaluation of the efficacy of management strategies. *ICES Journal of Marine Science: Journal Du Conseil*, *73*(suppl\_1), i115–i126.

Pitcher, C. R., Hiddink, J. G., Jennings, S., Collie, J., Parma, A. M., Amoroso, R., Mazor, T., Sciberras, M., McConnaughey, R. A., Rijnsdorp, A. D., Kaiser, M. J., Suuronen, P., & Hilborn, R. (2022). Trawl impacts on the relative status of biotic communities of seabed sedimentary habitats in 24 regions worldwide. *Proceedings of the National Academy of Sciences of the United States of America*, *119*(2). https://doi.org/10.1073/pnas.2109449119

Roberts, C. M., O’Leary, B. C., McCauley, D. J., Cury, P. M., Duarte, C. M., Lubchenco, J., Pauly, D., Sáenz-Arroyo, A., Sumaila, U. R., Wilson, R. W., Worm, B., & Castilla, J. C. (2017). Marine reserves can mitigate and promote adaptation to climate change. *Proceedings of the National Academy of Sciences*, *114*(24), 6167–6175.

Schickele, A., Guidetti, P., Giakoumi, S., Zenetos, A., Francour, P., & Raybaud, V. (2021). Improving predictions of invasive fish ranges combining functional and ecological traits with environmental suitability under climate change scenarios. *Global Change Biology*. https://doi.org/10.1111/gcb.15896

Tikhonov, G., Duan, L., Abrego, N., Newell, G., White, M., Dunson, D., & Ovaskainen, O. (2020). Computationally efficient joint species distribution modeling of big spatial data. *Ecology*, *101*(2), e02929.

Tikhonov, G., Opedal, Ø. H., Abrego, N., Lehikoinen, A., de Jonge, M. M. J., Oksanen, J., & Ovaskainen, O. (2020). Joint species distribution modelling with the r-package Hmsc. *Methods in Ecology and Evolution / British Ecological Society*, *11*(3), 442–447.

Tjur, T. (2009). Coefficients of Determination in Logistic Regression Models—A New Proposal: The Coefficient of Discrimination. *The American Statistician*, *63*(4), 366–372.

Van de Putte, A. P., Griffiths, H. J., Brooks, C., Bricher, P., Sweetlove, M., Halfter, S., & Raymond, B. (2021). From Data to Marine Ecosystem Assessments of the Southern Ocean: Achievements, Challenges, and Lessons for the Future. *Frontiers in Marine Science*, *8*. https://doi.org/10.3389/fmars.2021.637063

Warton, D. I., Blanchet, F. G., O’Hara, R. B., Ovaskainen, O., Taskinen, S., Walker, S. C., & Hui, F. K. C. (2015). So Many Variables: Joint Modeling in Community Ecology. *Trends in Ecology & Evolution*, *30*(12), 766–779.

Wassmann, P., Duarte, C. M., Agustí, S., & Sejr, M. K. (2011). Footprints of climate change in the Arctic marine ecosystem. *Global Change Biology*, *17*(2), 1235–1249.

Worm, B., Barbier, E. B., Beaumont, N., Duffy, J. E., Folke, C., Halpern, B. S., Jackson, J. B. C., Lotze, H. K., Micheli, F., Palumbi, S. R., Sala, E., Selkoe, K. A., Stachowicz, J. J., & Watson, R. (2006). Impacts of biodiversity loss on ocean ecosystem services. *Science*, *314*(5800), 787–790.