Ecological Indicators 155 (2023) 110984

Contents lists available at ScienceDirect

Ecological Indicators

journal homepage: www.elsevier.com/locate/ecolind

Original Articles

Can we use recovery timescales to define Good Environmental Status?

Robert B. Thorpe [a,*], Michael Heath [b], Christopher P. Lynam [a]

[a] Centre for Environment, Fisheries, and Aquaculture Science (CEFAS), Pakefield Road, Lowestoft, Suffolk NR33 0HT, UK
[b] Department of Mathematics and Statistics, University of Strathclyde, Glasgow, Scotland G1 1XQ, UK

ARTICLE INFO

Keywords:
Ecosystem Management
Good Environmental Status
North Sea
Recovery Timescales
Resource Depletion

ABSTRACT

Ecosystem-based management is mandated by international legislation, including the Marine Strategy Framework Directive (MSFD) in the EU. This introduces a requirement for marine environments to achieve "Good Environmental Status" or GES, implying that the ecosystem is in a healthy and biodiverse state which does not limit the management options of future generations. Indicators of GES typically refer to the current or past state; however, an alternative approach that defines GES in terms of being able to recover to the appropriate reference unperturbed state within 30 years if human activities cease has been suggested. In this study we evaluate this "longest recovery timescales" (LRT) approach using the StrathE2E2 "big picture" model, an end-to-end ecosystem model designed to evaluate both top-down and bottom-up effects at an ecosystem level. We ask whether the approach is enough to prevent severe depletion as well as ensuring recovery at some future time. We also ask whether implementation is practical given uncertainties in defining appropriate baselines for recovery, defining what recovery looks like relative to this baseline, and taking account of natural variability. We find that the main issues with implementation of LRT are a) defining the appropriate baseline for recovery in a changing environment, and b) ensuring that there is stakeholder acceptance of any recommended actions in the event that they differ substantially from current policy. Subject to these two issues, we conclude that the LRT method is a valuable addition to management in support of achieving GES alongside existing methods that focus on current or near-future states.

1. Introduction

The ecosystem-based management (EBM) approach aims to ensure that the natural environment is managed in a sustainable manner which respects the needs all users of ecosystem goods and services (Christensen et al., 1996). EBM has been promoted by international convention (e.g. the UN Convention on Biological Diversity - UNEP 1998, CBD, 2014), by national legislation across Europe and the wider world (Kidd et al., 2011; Rudd et al., 2018; O'Higgins et al., 2020). It is supported by transnational scientific organisations such as the International Council for the Exploration of the Sea (ICES) and OSPAR (The Convention for the Protection of the Marine Environment of the North-East Atlantic) which provide evidence and advice. EBM recognizes the need to take a holistic approach to understanding ecosystem level change, including explicitly accounting for the governance structures involved in interpreting, enacting and enforcing legislation (Borgström et al., 2015).

Motivated by these aims the EU has enacted the Marine Strategy Framework Directive (MSFD: EU, 2008) with the aim of achieving "Good Environmental Status" or GES for the marine waters within the EU by 2020 (Lynam et al., 2016). Similarly in South Africa, the National Biodiversity Strategy and Action Plan aims to achieve 'Good Ecological Condition' which refers to ecosystems that are intact or largely intact with minimal modification from a natural state (Department of Environmental Affairs, 2015). In the United States, implementing the ecosystem-based approach to management includes the development of quantitative indicators and criteria that can be used to assess overall ecosystem status (Leslie and McLeod, 2007). Where ecological data are lacking, such as in South Africa, expert judgment is often used to set targets for marine biodiversity indicators (e.g., Driver et al., 2011; Department of Environmental Affairs, 2015).

In Europe, GES is defined in terms of 11 "descriptors" (Table 1).

In support, scientists have developed modelling tools and indicators which can help determine whether GES is being achieved by considering each descriptor in turn (Shin et al., 2012, Piroddi et al., 2015; Smit et al., 2021). This is not a simple task since one needs to know what meeting the general GES descriptors entails for each of them and how good status

* Corresponding author.
E-mail address: robert.thorpe@cefas.gov.uk (R.B. Thorpe).

https://doi.org/10.1016/j.ecolind.2023.110984
Received 9 February 2023; Received in revised form 18 September 2023; Accepted 20 September 2023
Available online 27 September 2023
1470-160X/Crown Copyright © 2023 Published by Elsevier Ltd. This is an open access article under the CC BY-NC-ND license (http://creativecommons.org/licenses/by-nc-nd/4.0/).

Table 1
Descriptors of "Good Environmental Status" according to the Marine Strategy Framework Directive (MSFD).

Descriptor	Summary Description
D1	Biodiversity is maintained.
D2	Non-indigenous species do not adversely alter the ecosystem.
D3	The population of commercial fish species is healthy.
D4	Elements of food webs ensure long-term abundance and reproduction (ICES, 2014b).
D5	Eutrophication is minimised.
D6	The sea floor integrity ensures functioning of the ecosystem (ICES, 2014a).
D7	Permanent alteration of hydrographical conditions does not adversely affect the ecosystem.
D8	Concentrations of contaminants give no effects.
D9	Contaminants in seafood are below safe levels.
D10	Marine litter does not cause harm.
D11	Introduction of energy (including underwater noise) does not adversely affect the ecosystem.

would be recognised once achieved. It may also be necessary to integrate conflicting signals for an overall picture of ecosystem health, because individual descriptors may be moving in different directions. Several different approaches have been discussed in the literature. These are summarised and discussed later, but broadly speaking there are two facets to attainment of GES. Firstly, the current state of the ecosystem should feature "clean, healthy and productive seas". Secondly, future states that may be achieved as a consequence of the decisions being made now should "safeguard the potential for uses and activities by current and future generations". To date most effort has been focussed on the acceptability of current or projected states because this is easier to monitor than recovery timescales, but Rossberg et al. (2017) in particular have argued that considering the longest timescales of ecosystem recovery is more in keeping with the spirit of GES and desirable for determining what levels of current exploitation are allowable whilst safeguarding the interests of future generations.

The adoption of a methodology for an assessment of change based on longest recovery timescales (hereafter termed LRT) has clear implications for indicator target ranges, attainment of acceptable states, and clarifies what we mean by "recovery". In particular, it might be permissible to harvest some components of the ecosystem more intensively if one is not greatly concerned with near-present status, provided that the component can recover rapidly. Components that would be able to sustain high harvest rates would be those with shorter response timescales or more variable baselines, whilst other slow-responding components would likely need greater protection. A priori it might be expected that this would promote more intense harvesting near the bottom of the food web (e.g. on planktivorous fish species that are typically smaller and relatively quick to mature), and less at higher trophic levels (e.g. on typically larger-bodied piscivorous species), given that the generally longer-lived predatory species might respond more slowly. However, harvesting low trophic level species may lead to subsequent detrimental impacts on predator populations dependent on them (e.g. birds and mammals).

Quantitative study of LRT as a prospective indicator is computationally challenging. Unlike the setting of indicators for target states, which require characterisation of a (small) set of desirable states, an assessment of LRT suitability requires an analysis of the full ecosystem dynamics across a wide range of states. This is because a recovery timescale depends upon both start and end states, whilst a full ecosystem analysis is needed to identify the slowest component of the response. Whilst we can specify a desirable end state as the unfished state given the current environment, we do not know the starting state a priori, and so we have to consider a very wide range of possible starting states to have confidence that recovery will be speedy enough. We also need to ensure that highly depleted starting states should be ruled out by not being recoverable within a suitable time-frame even if fishing ceases.

In the interests of making the computational aspects manageable, we need a full end-to-end model of the ecosystem that nevertheless has modest run time. So for our study we use the StrathE2E2 model (Heath et al., 2021), an end-to-end foodweb model of intermediate complexity which is ideal for answering "big picture" questions about foodweb responses to a variety of forcings, and which takes the mechanistic approach needed to answer questions about response timescales (Thorpe et al., 2022). Achieving the required modest runtimes required a sacrifice of some spatial, taxonomic, and biological granularity in order to span the ecosystem and food web from physics, nutrients, and microbes through to megafauna and fishing fleets (Heath et al, 2020). In this study, therefore, we limit ourselves to considerations of biomass trajectories of broad functional groups (e.g demersal fish, birds, and seals) rather than individual species, so our results have direct relevance only for descriptors D1 and D4 of GES (Table 1). However, ICES (2015) have suggested that a relatively simple breakdown of the ecosystem into functional groups may be sufficient to improve management. By monitoring the biomass of fish, their benthic and pelagic resources, and primary production (or proxies thereof), changes in energy pathways, and imbalances in the functioning of ecosystems can be detected, and their ability to recover can be assessed. This means that a "big picture" model, which represents the end-to-end mechanistic impact of processes including bio-geochemical cycling of resources on broad functional groups can contribute to ecosystem management, even if it does not resolve individual species.

We use StrathE2E2 to investigate the response timescales of the North Sea food web to a wide variety of fishing scenarios, and for three baseline environmental states, the 2003–13 reference state, and for warming of 2 °C (warming that might be expected around 2100 if there is a strong mitigation response) and 4 °C (warming that might be expected around 2100 if there was little further mitigation response). These scenarios are not necessarily realistic climate projections (see Burgess et al. 2023 for an excellent discussion of the use of climate scenarios in environmental modelling). Instead we use them to address two issues, firstly are response timescales sensitive to warming relative to the reference state (is the foodweb stabilised or destabilised by warming?), and secondly, does warming of the environment matter if we fail to recognise that it has happened and thus manage the system to the wrong reference levels (an example of shifting baselines - Pauly, 1995; Papworth et al., 2008; Atmore et al., 2021; Jones et al., 2021)? Specifically, we use our fishing and warming scenarios to address the following questions:

(1) What timescale is necessary to prevent LRT permitting ecosystem states so impacted they should be ruled out on other grounds? Rossberg et al. (2017) suggested 30 years – is this sufficient to protect ecosystem form and function, and is it robust against future warming?
(2) To what extent does variability similar to that observed impact recovery timescales?
(3) Related to Q2) are results very sensitive to the manner in which recovery to an unfished state is defined?
(4) How sensitive are results to assumptions made about the "no fishing" baselines against which recovery is assessed? What might happen if warming shifts the baseline more than is realised at the time?
(5) Would an LRT approach alone produce dramatically different management recommendations, resulting in stakeholder resistance?

In combination, addressing these questions will help us to evaluate LRT (Rossberg et al., 2017) methodology, which recommends the use of recovery timescales to define GES for the North Sea.

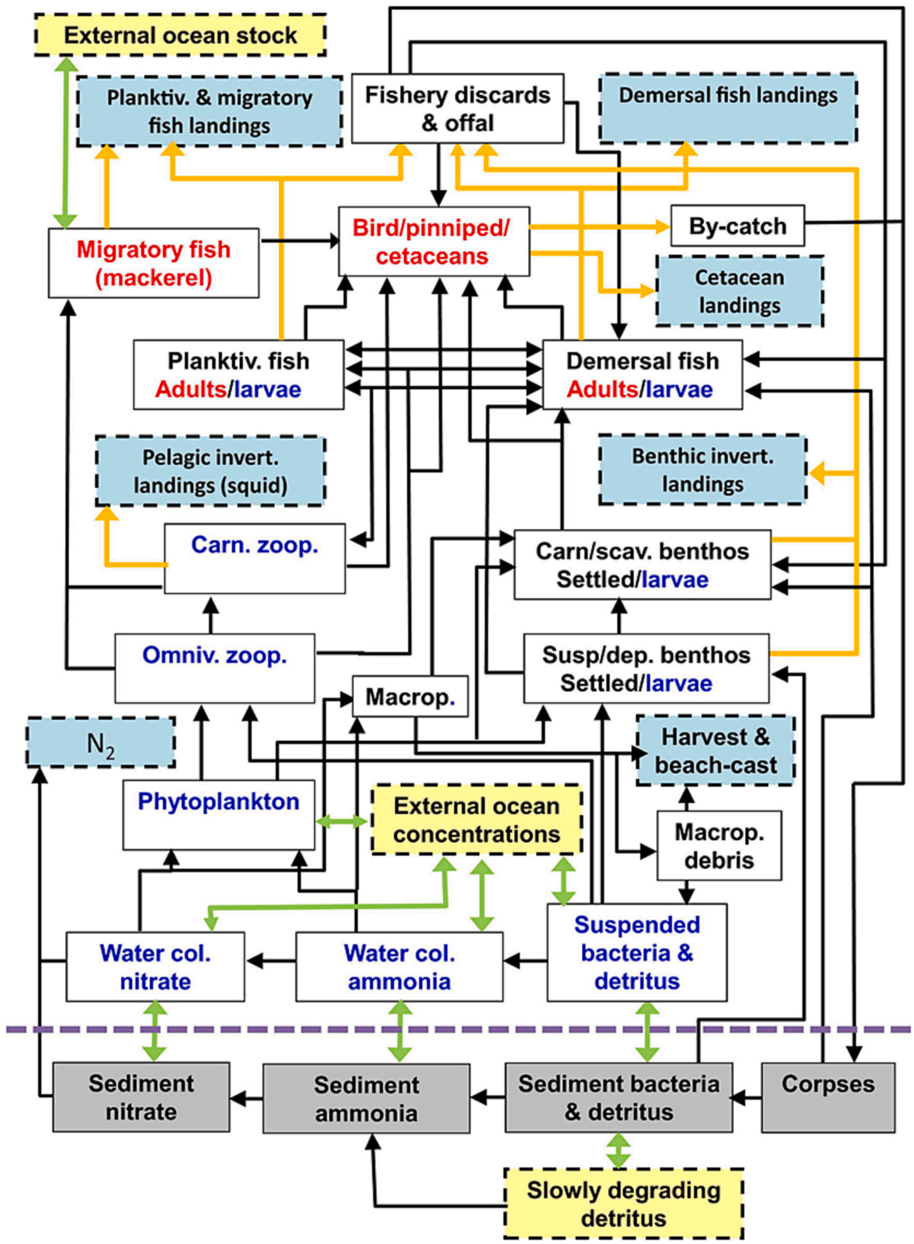

Fig. 1. Schematic of the food web compartments of the StrathE2E2 model. Green arrows represent advection, mixing, and migration, orange arrows represent fishery-related fluxes, black arrows represent biological fluxes. Red labelled components are active migrators while blue are subject to passive advection and mixing, and black are anchored. Pale blue boxes represent quantities that are exported from the model while yellow are imported. The model also includes fluxes from living components to ammonia, detritus, and corpses due to excretion, defecation, and death, but these are not shown for clarity. Also for clarity, birds, pinnipeds, and cetaceans are combined as a single box in the figure, but in the model are treated separately. The abbreviation "Macrop." is shorthand for macrophytes. Diagram reproduced from Heath et al. (2020).

2. Methods

Our study used the StrathE2E2 end-to-end ecosystem model to examine how the biomasses of 18 functional units based on behavioural mechanisms of food acquisition rather than taxonomic groups within the North Sea ecosystem (hereafter referred to as functional groups) are predicted to respond to 3000 scenarios of warming and changes in the intensity of fishing by three different fleet groupings. These included a 'demersal' fleet, (demersal seine and otter); a 'pelagic' fleet (pelagic seines and trawls), and a third group comprised of nine 'other' gear types encompassing static gears, beam and industrial trawlers (longline mackerel, sandeel/sprat gears, demersal beam trawl, demersal longline and gillnets, shrimp trawls, nephrops trawls, creels and pots, mollusc dredges, and Norwegian whalers - Heath et al., 2021; see their Table 13 and adjacent sections). The aim was to explore the dynamics of recovery timescales following the cessation of fishing, and consider the implications of using this as a metric for achievement of Good Environmental Status (GES).

The StrathE2E2 model of the North Sea (Heath et al., 2020) is an intermediate-complexity model which has been designed to look at "big picture" ecological questions, where the emphasis is on the overall response in terms of structure or energy flows, rather than species-specific details. In a development from the prototype (Heath, 2012; Morris et al., 2014), it has separate functional groups to represent seabirds, seals, and cetaceans, more flexibility for users, and comprehensive documentation within an R-package (Heath et al., 2021). The advantages of the model framework are: (1) a comprehensive treatment of bottom-up (often based on hydrodynamic or chemical conditions) and

Table 2
Key questions for evaluating the LRT thesis of Rossberg et al. (2017) alongside the approaches taken to address each of the questions.

Question	Approach
Q1) What timescale is necessary to prevent LRT permitting ecosystem states so impacted they should be ruled out on other grounds? Rossberg et al. (2017) suggested 30 years – does this preserve ecosystem form and function, and is it robust to future warming?	Consider the lowest biomasses for each group that nevertheless can recover within 30 years (Fig. 4a) or 40 years (Fig. 4b). Differences in response times for 2 °C and 4 °C reflects the sensitivity of response timescales to warming. As these are responses to stopping all fishing, they represent the maximum possible depletion consistent with recovery on that timescale.
Q2) To what extent does typical variability of SST and hence functional group abundance impact recovery timescales?	Consider the case with no variability in SST alongside the variable base case (Fig. 5 vs Fig. 3)
Q3) Related to Q2) are results very sensitive to the manner in which recovery to an unfished state is defined?	Consider several different possible definitions for "recovery", as shown in Table S1 (Figs. 6, 7a).
Q4) How sensitive are results to assumptions made about the "no fishing" baselines against which recovery is assessed? What might happen if warming shifts the climate more than we think?	Consider the effect of assuming the reference baseline when model warming of 2 °C or 4 °C has occurred (Fig. 7b vs 7a, 8)
Q5) Would an LRT approach alone produce dramatically different management recommendations, resulting in stakeholder resistance?	Consider the implications for management of the three fleet groupings, demersal, pelagic, and other fisheries (Figure 9).

top-down (e.g., fishing) control mechanisms alongside each other; (2) a whole-ecosystem approach, setting the functional groups (including the microbial loop (Azam et al., 1983) and benthic groups whose role has sometimes been obscured) in their wider context and allowing their ecosystem impacts to be considered; and (3) modest run times allowing consideration of a large number of climate and fishing scenarios. Whilst it should be noted that the optimum configuration for an end-to-end model will be very context dependent [see Iwasa et al. (1987); Fulton (2010), Giricheva (2015), and Heath et al. (2020)], these features make StrathE2E2 ideal for the current study, where we need to simulate a large number of fisheries scenarios in order to model the dynamics of many possible levels of interim depletion consistent with any given recovery timescale.

Achieving this required a sacrifice of spatial, taxonomic, and size-structured resolution whilst spanning the ecosystem and food web from physics, nutrients, and microbes through to top predators (birds, seals, and cetaceans) and fishing fleets (Heath et al., 2020). So the compromise is that the functional groups are very broad-brush, for example "demersal fish" are represented as a single group rather than resolved at the species-level into cod, haddock, whiting, etc. As a result, the full ecosystem is represented here by just 18 functional groups, compared with e.g. over 60 elements used in the North Sea Ecopath model (Mackinson and Daskalov, 2007). However the functional group approach has the advantage that we do not need to consider the impact of individual species extinctions or invasions resulting from environmental change, unless changes are so fundamental that the entire foodweb energy structure is radically altered (Bartley et al., 2019).

Model state variables represent the nitrogen mass (moles N/m^2 sea surface) of classes of detritus, dissolved inorganic nutrient, plankton, benthos, fish, birds, and mammals (Fig. 1). Dynamics of these variables are simulated in continuous time and output at daily intervals by integrating a set of linked ordinary differential equations (ODEs) describing the key physical, geochemical, and biological processes that occur in the sea and seabed sediments. These include the feeding of living components, and the production, consumption and mineralisation of detritus including fishery discards. Uptake of food is defined by Michaelis-Menten functions for each resource-consumer interaction defined by a preference matrix. Abundances of functional groups through time are determined by ODEs which take account of a variety of biological and physical processes (Appendix S1, Heath et al., 2020). Biological terms describe the balance between gains due to assimilation of food, and losses due to mortality and metabolism. Some components of the food web (planktivorous and demersal fish; suspension/deposit feeding and carnivore/scavenge feeding benthos) are resolved into life stages, and for these the equations also include the balance between gains due to recruitment and losses due to developmental progression or spawning. In addition, each ODE also includes terms representing sinking, advection, mixing and migration flows through the system. Time-dependent external drivers and boundary conditions for the model are harvesting rates of fish and benthos, temperature, sea surface irradiance, suspended sediment, inflow rates of water and nutrients across the external ocean boundaries and from rivers, vertical mixing rates, and atmospheric deposition of nutrients. Outputs from the NEMO-ERSEM model (Butenshon et al., 2016) are used to drive the StrathE2E2 model in terms of temperature, vertical mixing, and currents, and so provide external biogeochemical boundary conditions. Further details can be found in Heath et al. (2020), whilst the prototype model (Heath, 2012) is discussed in the context of policy questions and other modelling approaches in Hyder et al. (2015), and outputs are compared with other North Sea fisheries models in Spence et al. (2018). The model simulates the seasonal cycle, and if there is no change in external forcing scenario, functional group biomasses vary seasonally, but are stable on multiyear timescales.

The twelve fishing fleets have fixed harvest efficiencies for each functional group (see Heath et al., 2021 for details), with catch being a product of effort and efficiency. Key inputs are, for each gear type, the spatial distribution of activity density, catching power, selectivity, discards and at-sea processing rates for each ecology model guild, and contact rate with the seabed (Heath et al., 2020). They are calibrated to the 2003–2013 baseline period such that if they all fish at a relative effort of 1, the estimated 2003–2013 fishing rates are recovered (analogous to the method in Thorpe et al, 2016). Increasing fishing effort reduces the biomasses of target (and bycatch to a lesser extent) functional groups and increases seabed abrasion rates, resulting in a restructuring of energy flows and abundances that tends to increase with effort. Changes in SST affect the vital rates of metabolic processes via Q_{10} terms such that both food uptake and respiration increase with temperature, but the latter more so, such that net productivity is dome-shaped with respect to temperature; increasing below a functional group's thermal optimum and decreasing above it.

Following Thorpe et al. (2022), we considered the reference climate (NEMO-ERSEM reanalysis 2003–2013) and two warming scenarios, a uniform warming of 2 °C and 4 °C relative to this baseline. For each of these we considered an unfished state and 999 fisheries scenarios for the 3 fleet groups (demersal, pelagic, and other) with each fishing fleet independently being allowed to harvest at an intensity between zero and three times the average intensity between 2003 and 2013. A Latin hypercube design (McKay et al., 1979) was used to ensure that this possible space was approximately uniformly sampled. The fisheries scenarios are not designed to be realistic, but rather to allow the exploration of recovery timescales across a wide range of possible fisheries perturbations, given that we do not know what rates might pertain in the future. For each scenario, the designated level of fishing was applied for 100 years, followed by 100 years without fishing to explore the rate of recovery. The initial 100 years was run with a seasonally varying but otherwise constant environment, the recovery portion was run both with a

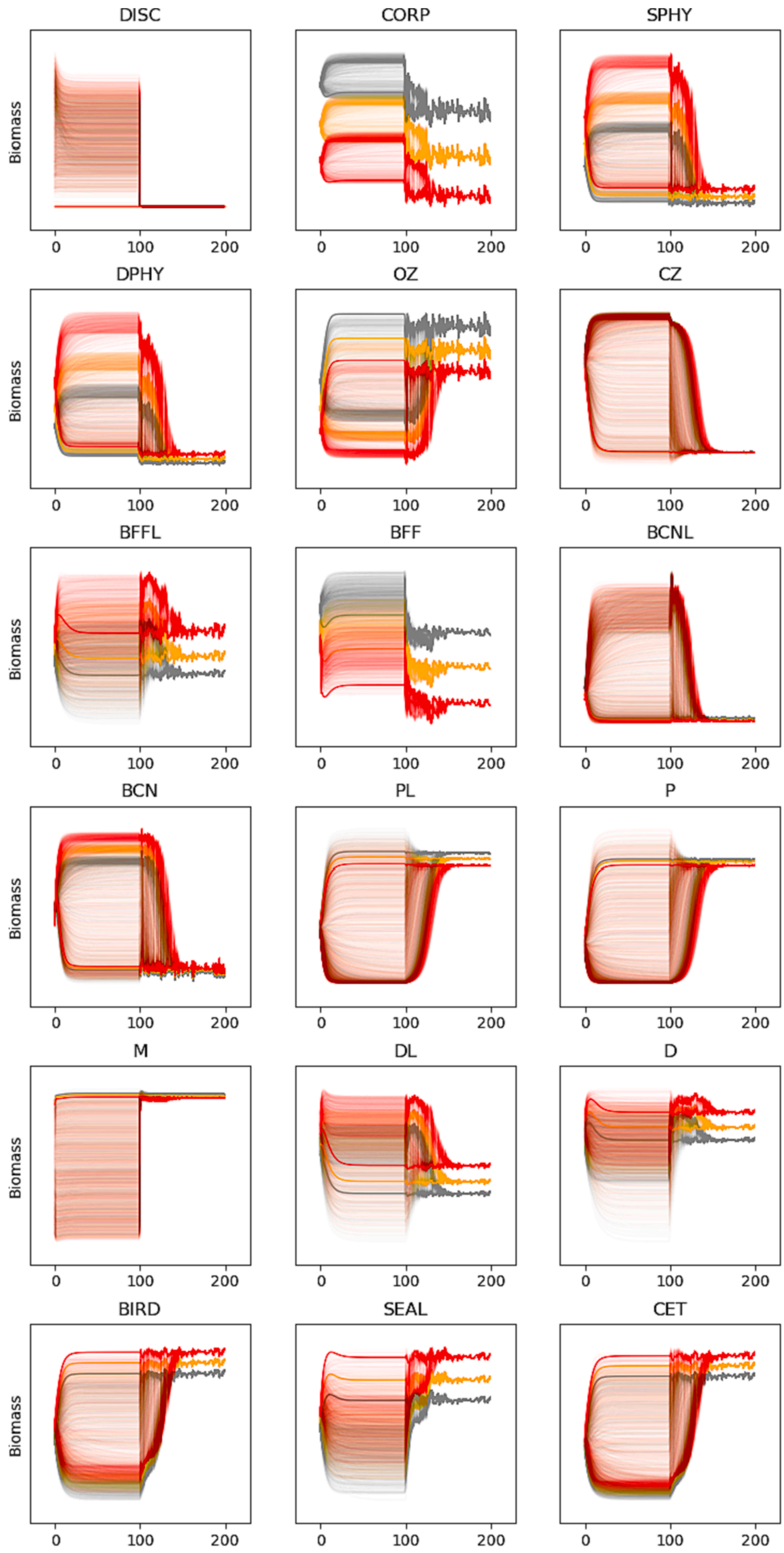

Fig. 2. Relative functional group biomass trajectories for the 1000 fishing and recovery trajectories for the reference (2003–2013) climatology (grey) and for uniform warming of 2 K (orange) and 4 K (red).

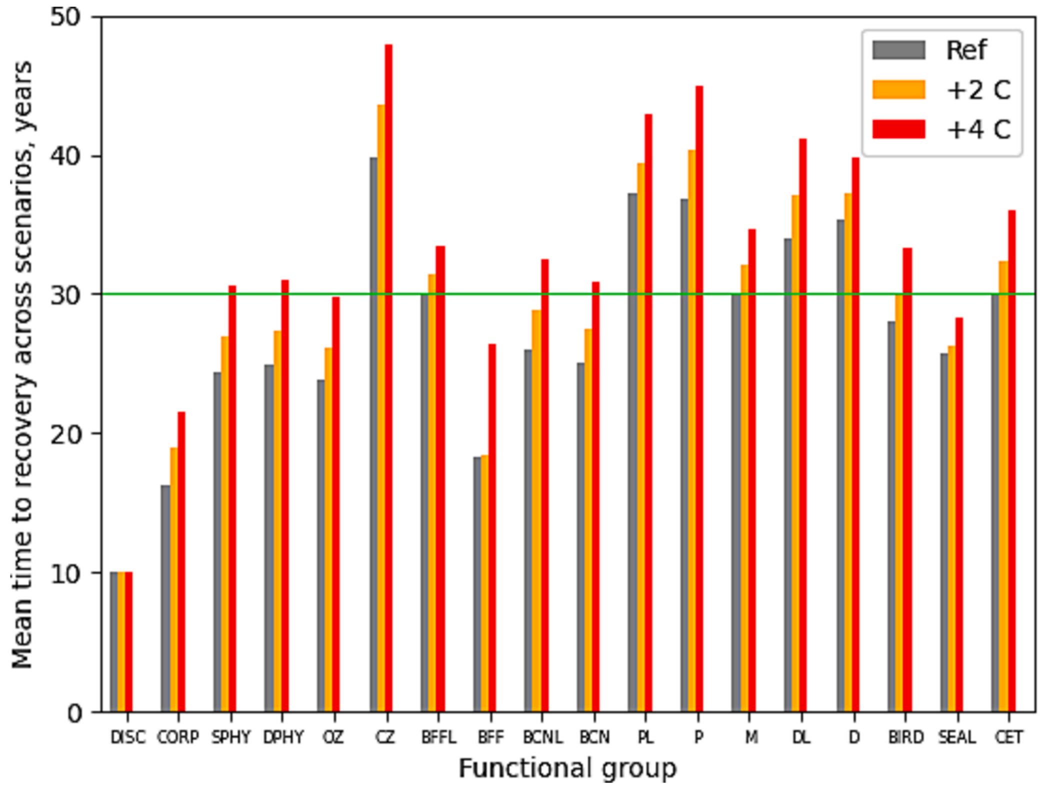

Fig. 3. Mean time to recovery across all fishing scenarios for the reference (2003–2013) climatology (grey) and for uniform warming by 2 K (orange) and 4 K (red). DISC = discards, CORP = corpses, SPHY = surface phytoplankton, DPHY = deep phytoplankton, OZ = omnivorous zooplankton, CZ = carnivorous zooplankton, BFFL = benthic filter feeder larvae, BFF = benthic filter feeders, BCNL = benthic carnivore larvae, BCN = benthic carnivores, PL = pelagic fish larvae, P = pelagic fish, M = migratory fish, DL = demersal fish larvae, D = demersal fish, BIRD = birds, SEAL = seals, CET = cetaceans.

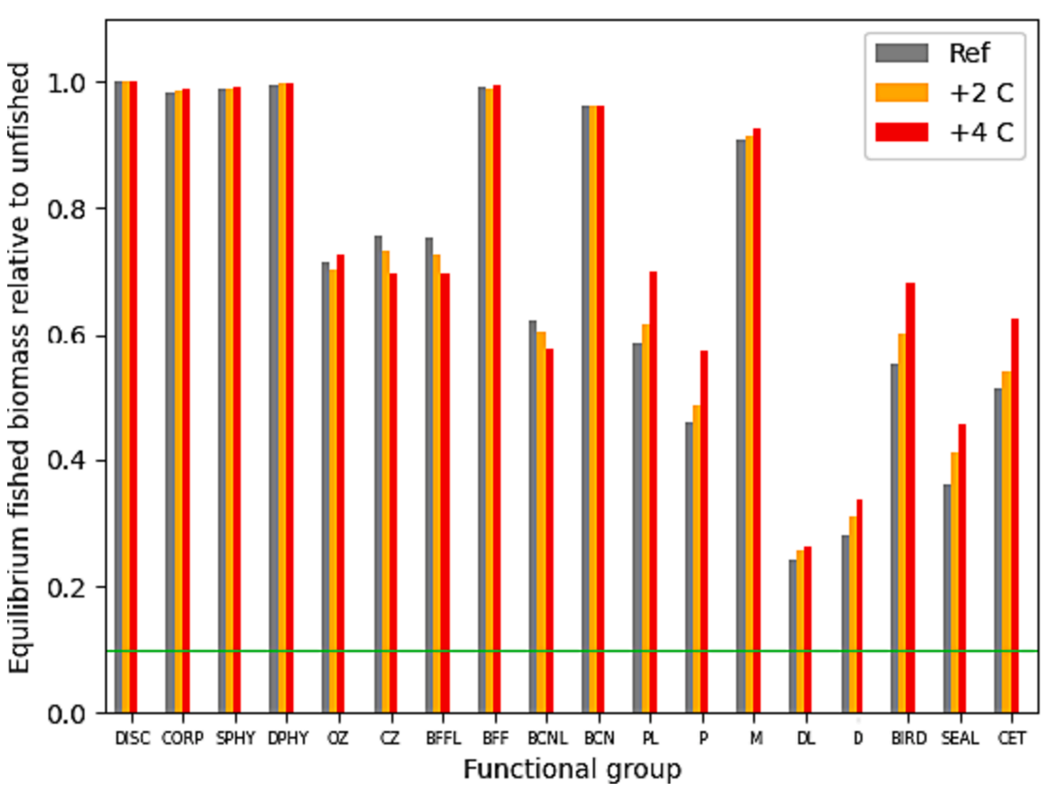

Fig. 4a. Greatest biomass depletion of functional groups relative to the unfished state for all scenarios that recover their unfished levels within 30 years after fishing stops. Grey bars are for the reference climate, orange for 2 K warming, and red for 4 K warming. The green line represents the 90 % depletion level associated with "collapsed" stocks in Worm et al. (2009). DISC = discards, CORP = corpses, SPHY = surface phytoplankton, DPHY = deep phytoplankton, OZ = omnivorous zooplankton, CZ = carnivorous zooplankton, BFFL = benthic filter feeder larvae, BFF = benthic filter feeders, BCNL = benthic carnivore larvae, BCN = benthic carnivores, PL = pelagic larvae, P = pelagic fish, M = migratory fish, DL = demersal larvae, D = demersal fish, BIRD = birds, SEAL = seals, CET = cetaceans.

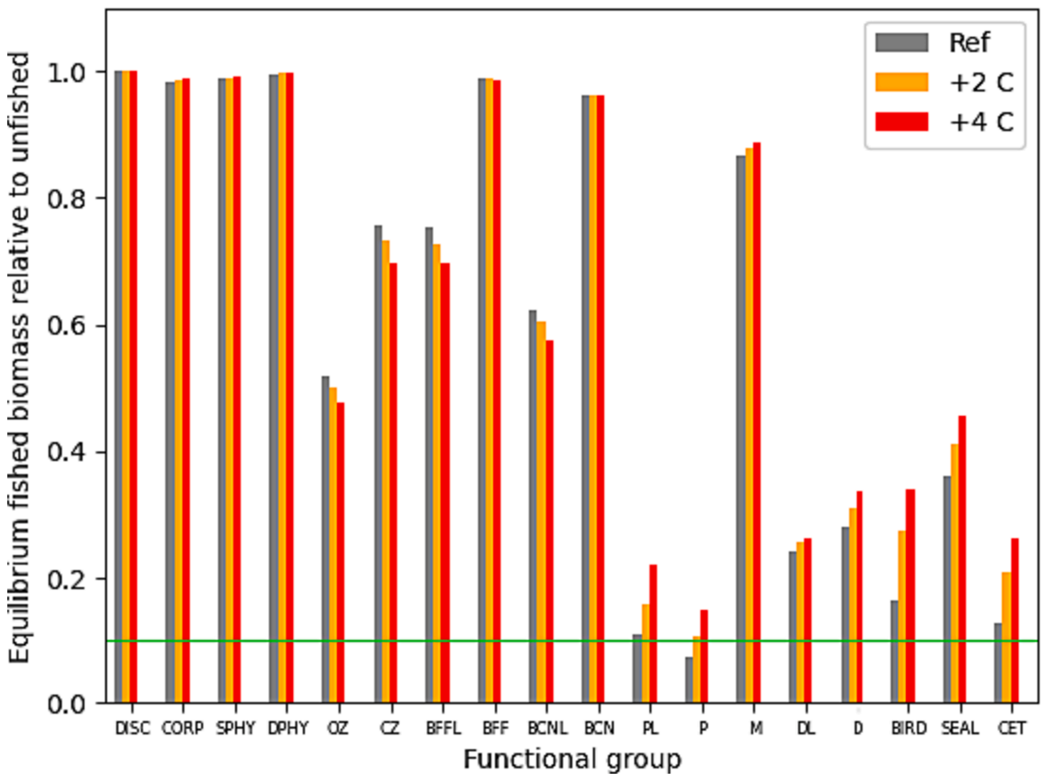

Fig. 4b. Greatest biomass depletion of functional groups relative to the unfished state for all scenarios that recover their unfished levels within 40 years after fishing stops. Grey bars are for the reference climate, orange for 2 K warming, and red for 4 K warming. The green line represents the 90 % depletion level associated with "collapsed" stocks in Worm et al. (2009). DISC = discards, CORP = corpses, SPHY = surface phytoplankton, DPHY = deep phytoplankton, OZ = omnivorous zooplankton, CZ = carnivorous zooplankton, BFFL = benthic filter feeder larvae, BFF = benthic filter feeders, BCNL = benthic carnivore larvae, BCN = benthic carnivores, PL = pelagic larvae, P = pelagic fish, M = migratory fish, DL = demersal larvae, D = demersal fish, BIRD = birds, SEAL = seals, CET = cetaceans.

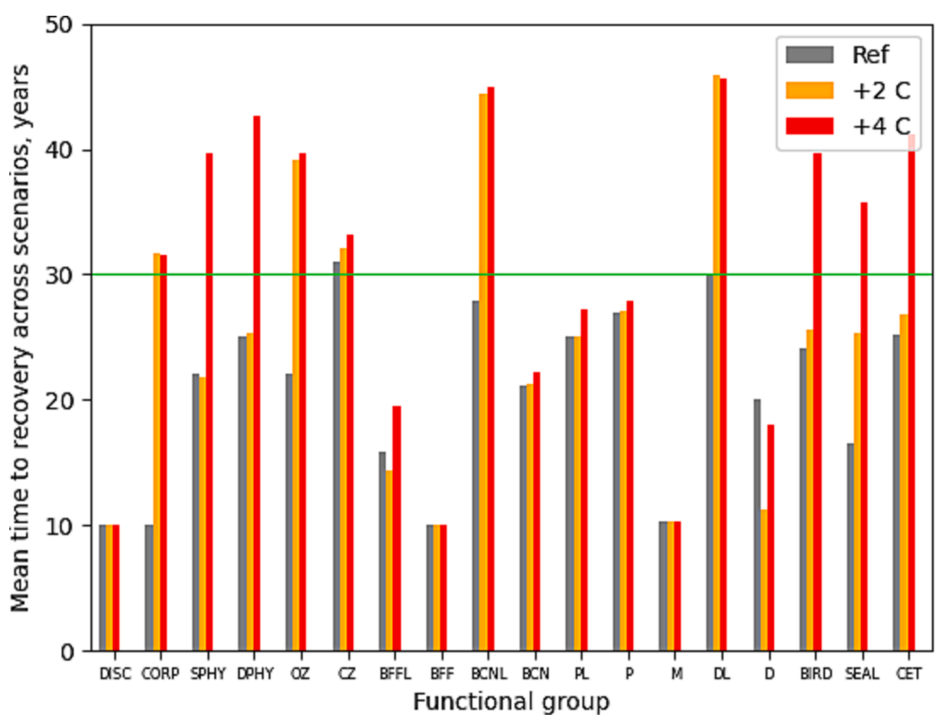

Fig. 5. Mean time to recovery across all fishing scenarios for the reference (2003–2013) climatology (grey) and for uniform warming by 2 K (orange) and 4 K (red) in the case of constant SST. DISC = discards, CORP = corpses, SPHY = surface phytoplankton, DPHY = deep phytoplankton, OZ = omnivorous zooplankton, CZ = carnivorous zooplankton, BFFL = benthic filter feeder larvae, BFF = benthic filter feeders, BCNL = benthic carnivore larvae, BCN = benthic carnivores, PL = pelagic fish larvae, P = pelagic fish, M = migratory fish, DL = demersal fish larvae, D = demersal fish, BIRD = birds, SEAL = seals, CET = cetaceans.

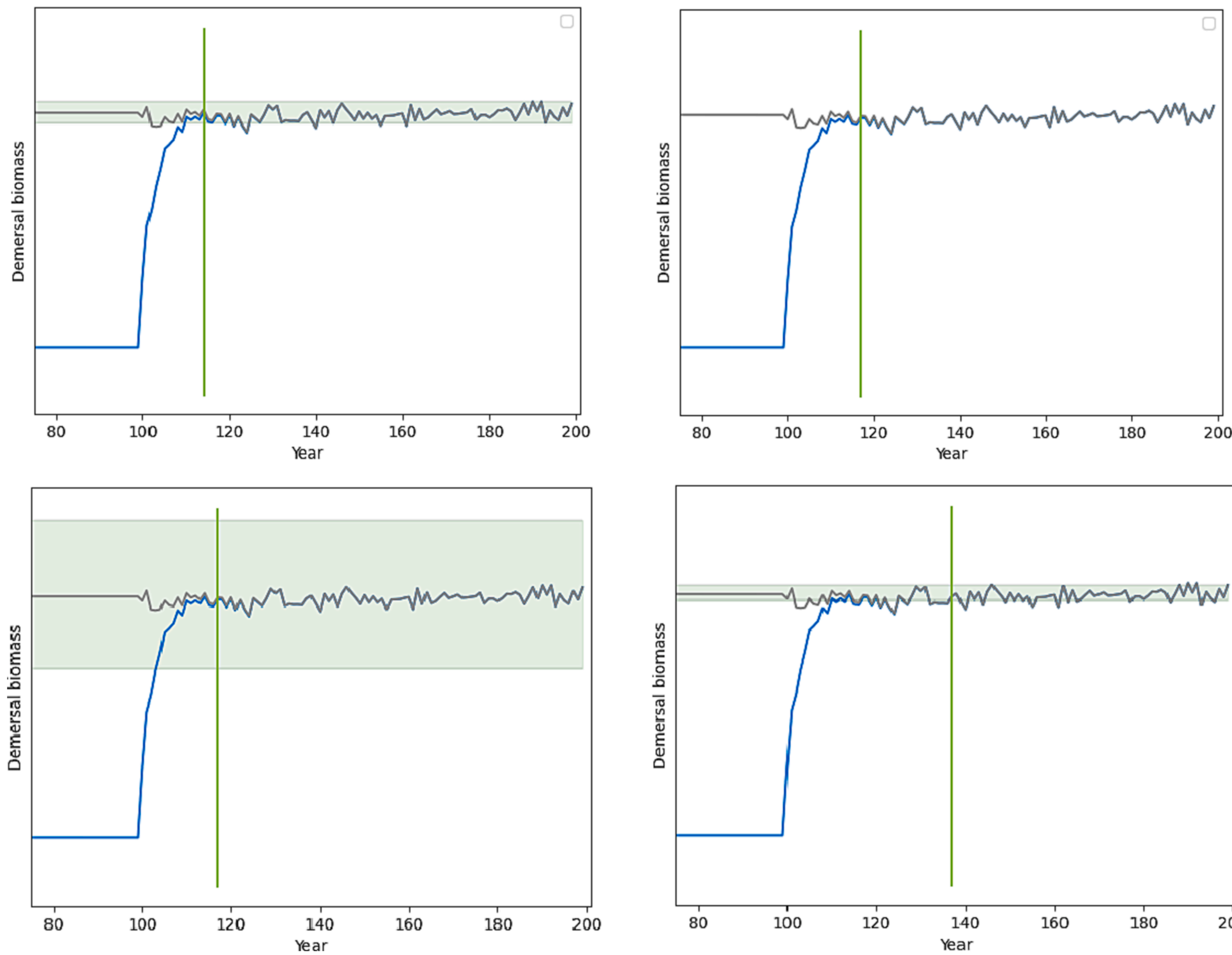

Fig. 6. Recovery trajectories for a reference climate scenario with high levels of demersal fishing, showing the point of recovery for a) the standard recovery definition, with 4 years in 10 inside the last decade of the unfished scenario, b) recovery once rates of change fall to less than 10 % of maximum, c) recovery to within 10 % of unfished abundances, d) recovery to within 1 % of unfished abundances. The grey trajectory is without fishing, and the blue trajectory is with fishing until year 100. The green shaded region represents the abundances deemed consistent with recovery (absent in Fig. 6b because this is trend-based), and the vertical green line represents the point in time at which recovery is deemed to have happened. This is lagged relative to the point at which abundances appear to have recovered because some time has to elapse before recovery can be confirmed.

seasonally varying but otherwise constant environment and with SSTs allowed to vary annually from the NEMO-ERSEM baseline (or the 2 °C or 4 °C warming scenarios) lognormally with a variance of 0.5 °C (Fig. S1). Changes in SST drive variations in metabolic scope of functional groups and hence their biomasses, allowing us to explore more realistic levels of ecosystem variability on the definition of "recovery". Given that cessation of fishing is the most dramatic possible management intervention, we would expect our study to result in the minimum possible ecosystem recovery times (and the maximum possible current depletion consistent with eventual recovery on this timescale).

We monitored the biomasses of 18 ecosystem groups through time and investigated the consequences of using the timescale for recovery of the slowest components to define a GES-compatible space. We addressed our five key study questions as shown in Table 2. By answering these questions we can evaluate the suitability of the recommendation by Rossberg et al. (2017) to use LRT for ecosystem management.

3. Results

Biomass trajectories for the fishing and recovery scenarios are presented in Fig. 2. During the fishing phase of the simulations (i.e. the initial 100 years), biomass trajectories are predicted to be scenario-dependent, reflecting the wide range of fishing mortality modelled across the three fleet groups. Once fishing ceases (i.e. for the following 100 years), differences between scenarios then decrease with time as the ecosystem recovers towards an unfished state. As expected, timescales of recovery following cessation of fishing depend upon both the pattern and intensity of exploitation. For similar patterns of exploitation, recovery is fastest following light pressure, but can be in excess of 50 years following high levels of fishing (particularly for the pelagic fleets), and for slow-response functional groups. Despite the severe reductions in biomass for some functional groups (particularly pelagic fish) in the scenarios with high levels of fishing (Fig. S2), all scenarios presented here eventually recovered to their unfished states. There was only one unfished state for each warming scenario, so there was no evidence of hysteresis.

Perhaps surprisingly, we found that seals recovered moderately quickly, despite being long-lived and at the top of the food chain (Fig. 3). Birds and cetaceans were predicted to be much slower to recover. The longest recovery timescales were associated with carnivorous zooplankton, pelagic fish and larvae, and demersal fish and larvae (Fig. 3), Thus recovery timescales in response to cessation of fishing did

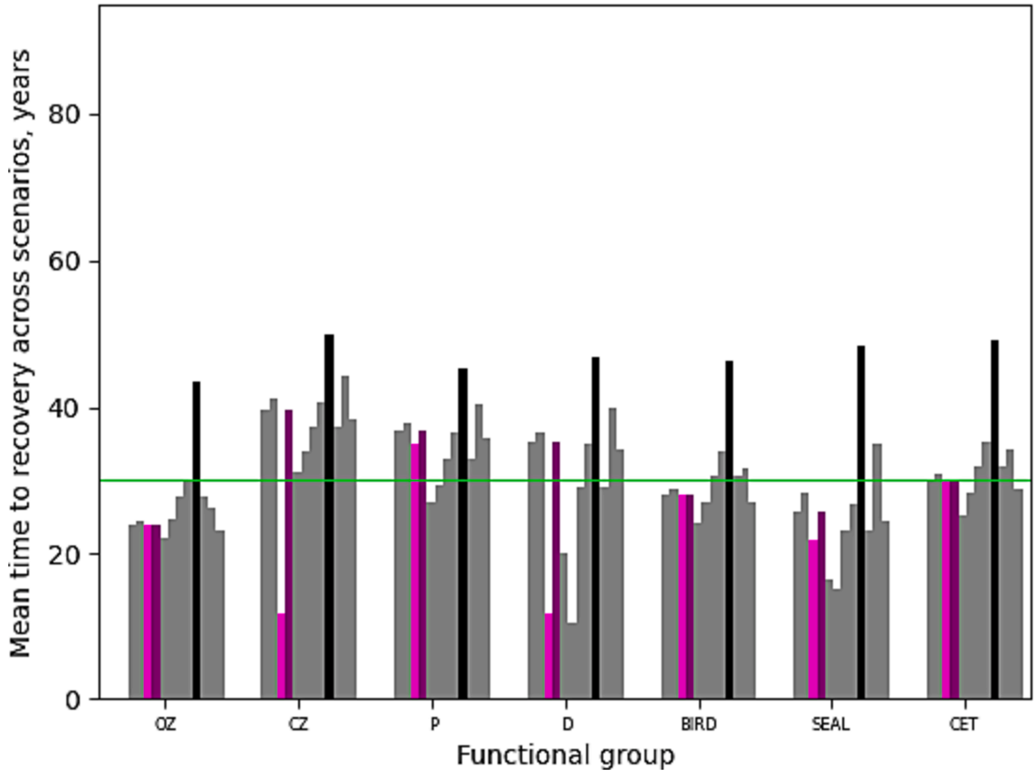

Fig. 7a. Mean time to recovery across all fishing scenarios for the reference (2003–2013) climatology (grey) for different ways of defining the moment of recovery (see Table S1). The requirement for recovery within 1 % of biomass is in black, and methods involving recovery to the reference climate are in magenta for being above the minimum reference climate biomass, and purple for being within the reference biomass range. OZ = omnivorous zooplankton, CZ = carnivorous zooplankton, P = pelagic fish, D = demersal fish, BIRD = birds, SEAL = seals, CET = cetaceans.

not simply increase with position higher up the foodweb but rather were longest in the middle. For all functional groups, recovery responses tended to be slower with warming, suggesting that environmental change might reduce recovery potential.

3.1. Q1 – does an LRT below 30 years permit severe depletion?

Given that intense fishing is predicted to lead to extreme depletion of at least some functional groups (Fig. S2), an LRT must be short enough for severe short-term depletion to be avoided. Figs. 4a,b shows the most extreme level of depletion by functional group consistent with recovery within a) 30 years, and b) 40 years. To achieve an LRT of 30 years (Rossberg et al., 2017) demersal fish biomasses would always have to be well above the 10 % of their unfished state that is often taken as a proxy for collapsed stocks (Thorpe et al., 2016; Thorpe and De Oliveira, 2019; Worm et al., 2009) and indeed would have to stay above the more cautious alternative of 20 % (Smith et al. 2009). Pelagic fish abundance would be more constrained, being above 40 % of their unfished state and hence would leave more than "one third for the birds" (Cury et al., 2011). Based purely on avoiding highly depleted states, a recovery timescale of 30 years does seem adequate. Conversely, an LRT of 40 years would permit severe depletion (Fig. 4b). The generally slower recovery responses with warming are associated with higher levels of biomass consistent with achieving acceptable LRTs (as expected), but the choice of a 30-year timescale as opposed to one of 40 years is not affected by warming of up to 4 °C.

3.2. Q2 - To what extent does SST variability similar to that observed impact recovery timescales?

We assessed the implications of typical SST variability for recovery timescales, by comparing the standard experiments in which SST during the recovery phase varied according to a lognormal distribution with a variance of 0.5 K (Fig. S1) with an otherwise identical experiment set without this variability. This showed that in the absence of SST variability, some groups appeared to recover faster (particularly benthic filter feeders, migratory fish, pelagics and demersal adults), especially for the reference climate, leading to a modest reduction in LRT overall. Though the balance of recovery timescales across functional groups did shift modestly, there was little change in overall LRT for the warming experiments (Fig. 5). Given that removing SST variability is unrealistic and extreme, the modest changes here are consistent with the discussion in Rossberg et al. (2017) suggesting that interannual variability will not be a major problem for the LRT approach.

3.3. Q3 - Are results very sensitive to the manner in which recovery to an unfished state is defined?

Recovery can be defined in a number of ways, including i) a slowing in biomass trends through time as the system approaches the unfished state, or ii) convergence towards that state (if known) where a certain fraction of years must fall within the standard variability pertaining to the unfished state, or iii) achievement of a target level that is based on the unfished state, or the difference between it and either the current or worst affected state. For the idealised case without SST variability, the second approach is not practical, so in this case we defined recovery to be within X% of the unfished state, for variable SST all three types of definition were considered (Table S1).

Given that there are several alternative definitions of "recovery", and it is not obvious that there is an objectively best one, it is important that LRT is not too sensitive to the definition of "recovery" which is used. Subject to the additional requirement that any definition has to include the unfished scenario at all times, we found that choosing a definition within the subset of possibilities had modest impacts on the recovery

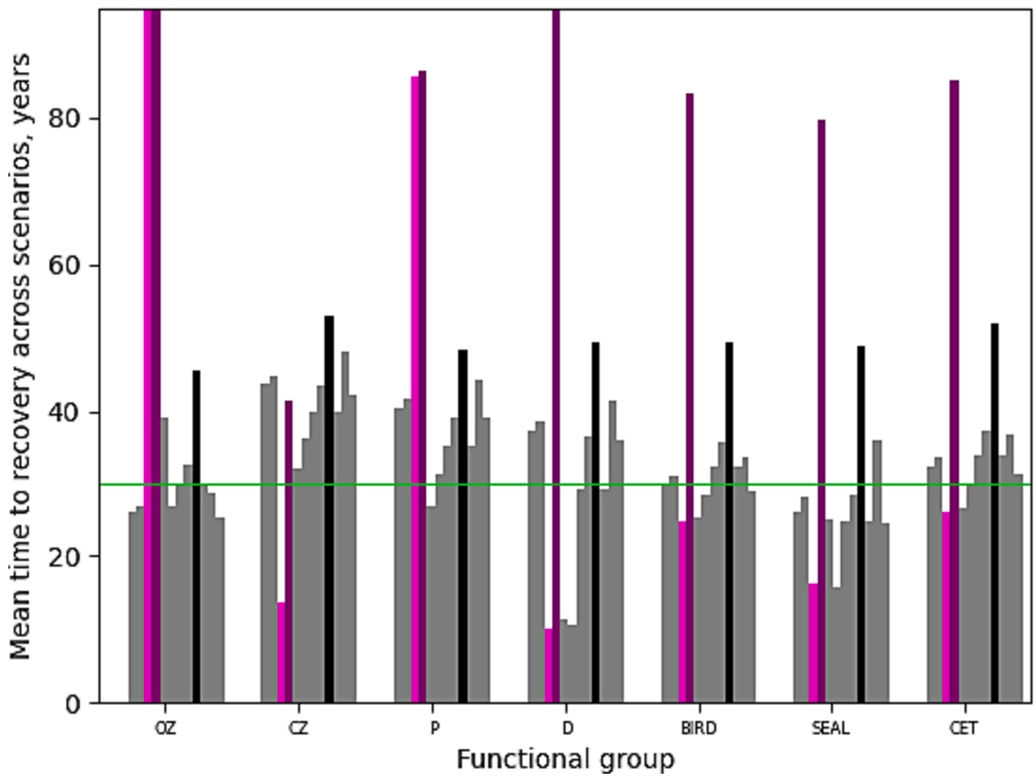

Fig. 7b. Mean time to recovery across all fishing scenarios for a warming of 2 °C for different ways of defining the moment of recovery (see Table S1). The requirement for recovery within 1 % of biomass is in black, and methods involving recovery to the reference climate are in magenta for being above the minimum reference climate biomass, and purple for being within the reference biomass range Changes in the magenta and purple bars relative to Fig. 8a show the impact of shifting baselines for a 2 °C warming. OZ cannot be recovered to the minimum reference climate threshold under any management scenario. OZ = omnivorous zooplankton, CZ = carnivorous zooplankton, P = pelagic fish, D = demersal fish, BIRD = birds, SEAL = seals, CET = cetaceans.

timescale, and did not change the qualitative picture. This is illustrated in Fig. 6, for a single functional group and climate/warming scenario, where 6a-6c give similar results. Whilst 6d suggests a longer recovery timescale, it (along with 6b) is not consistent with the requirement that the unfished state is always "recovered". Fig. 7a shows the average recovery time across the 100 fishing scenarios for 12 different methods of determining when recovery is attained (fixed baseline experiments 5,6, and 7 – Table S1 - not presented for clarity of figure, but are consistent with those shown). Results are qualitatively similar across a wide spectrum of options, the only obvious exceptions being cases where the definition is inconsistent with recovery in the unfished state (black in Fig. 7a).

3.4. Q4) How sensitive are results to assumptions made about the "no fishing" baselines against which recovery is assessed?

Next, we considered recovery timescales as a function of the reference climate, for all three warming scenarios, Fig. 7b shows the average functional group recovery time across the 1000 fishing scenarios for the warming of 2 °C for the same methods. Those in magenta and purple used biomass thresholds associated with the reference climate and show the impact of failing to take account of shifting baselines. In this case, some functional groups cannot be recovered to reference abundances under any management approach (mean time > 95 years: see also Fig. S7). Fig. 8 shows the impact of a 2 °C warming on assessment of recovery for the same fishing scenario in Fig. 6 if we assume that the baselines of the reference (2003–2013) climate still apply, illustrating the general point that the impact of environmental change on reference levels is much greater than for defining whether the agreed reference level has been reached. Although the trends-based method is relatively insensitive to the shifted baseline, unfortunately it does not work for light fishing pressure or the unfished state. Both for individual scenarios and overall, even for a warming of 2 °C, the impacts of shifting baselines (Currie et al. 2020) were much greater than those associated with natural variability or methodology for determining that recovery had been achieved. Our results show the vital importance of ascertaining the expected recovery state in a changing world and indicate that recovery targets for states must avoid biomasses that are too high as well as ones that are too low (illustrated by the failure of OZ to recover in the magenta scenario in Fig. 7b).

3.5. Q5) Would an LRT approach alone produce dramatically different management recommendations, resulting in stakeholder resistance?

Finally, we considered how different levels of fleet effort might impact the LRT. We did so assuming the other fleets were either inactive (Fig. 9a), or fished at their average 2003–2013 effort levels (Fig. 9b). Thus the different responses in Figs. 9a and 9b are caused by the higher levels of fishing of the two fleets whose effort is not being varied in each of the 3 sub-panels in Fig. 9b. In both cases we found that the pelagic fleet makes by far the largest contribution to the timescale of recovery and that warming increases this relative sensitivity. But importantly, we also found that the impact of the demersal fleet strongly depends on the level of pelagic fishing. If pelagic fishing is very light (F ~ 0; first and third panels of Fig. 9a), the demersal fleet has a modest negative impact on recovery timescales. If on the other hand, pelagic fishing is at or near 2003–2013 levels (Fig. 9b, first and third panels), higher demersal fishing increases the speed of recovery. Whilst the 2003–2013 period was not consistent with GES in this study (yellow circles in Fig. 9a), higher levels of effort can be if the effects of demersal and pelagic harvesting cancel out. Other fleets have less impact on recovery timescales. The reason for the high sensitivity to pelagic fishing, and its partial offset

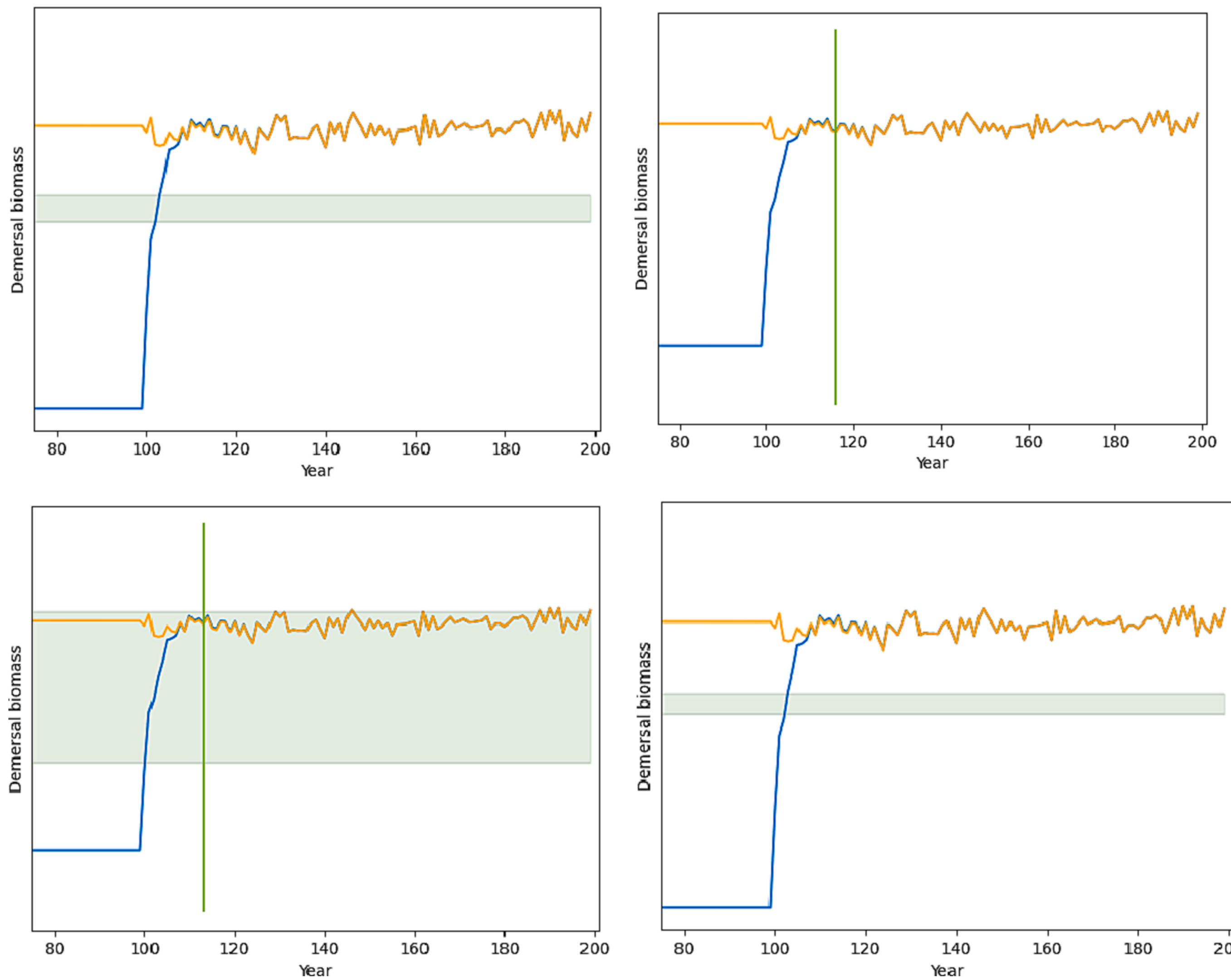

Fig. 8. Recovery trajectories for a 2 °C warming scenario with high levels of demersal fishing, showing the point of recovery for a) the standard recovery definition, with 4 years in 10 inside the last decade of the unfished scenario, b) recovery once rates of change fall to less than 10 % of maximum, c) recovery to within 10 % of unfished abundances, d) recovery to within 1 % of unfished abundances. The grey trajectory is without fishing, and the blue trajectory is with fishing until year 100. The green shaded region represents the abundances deemed consistent with recovery if we assume that the reference climate baseline still applies (again absent in Fig. 6b because this is trend-based). The vertical green line represents the point in time at which recovery is deemed to have happened. This is lagged relative to the point at which abundances appear to have recovered because some time has to elapse before recovery can be confirmed. There is no vertical green line in panels a) and d) because the reference baseline is not recovered according to these definitions of recovery given the warming of 2 °C.

by demersal fishing is the trophic cascade identified in Thorpe et al. (2022), which can lead to a large shift in the balance between different types of zooplankton, demersal, and pelagic stocks. In the absence of pelagic fishing (Fig. 9a), there is no trophic cascade, and LRT increases with the rate of demersal fishing. Higher levels of pelagic fishing in Fig. 9b reduce pelagic stocks and are enough to trigger a trophic shift in the absence of demersal fishing, leading to increased LRT. In this situation, demersal fishing acts to make the cascade less likely by removing key predators of the pelagic fish, and hence can offset the impacts of the pelagic fleet and result in a reduced LRT. Whilst this study's sensitivity to pelagic fishing is a consequence of model structure and parameterisation, the result illustrates the potential of new methodology to dramatically impact management advice potentially leading to stakeholder resistance to implementation.

4. Discussion

Concentrating on the longest recovery timescales (LRT) of ecosystems to applied pressures, as advocated by Rossberg et al. (2017) has the advantage of consistency with the Marine Strategy Framework Directive (MSFD; EC 2008), by focussing on preservation of options for future generations. This contrasts with and complements existing approaches (some of which are summarised in Table 3) which may focus more on the characteristics of current states rather than the extent to which any future uses are precluded (literature examples of this are shaded blue in Table 3). Whilst the traditional Maximum Sustainable Yield (MSY) approach (Mesnil, 2012; Thorpe, 2019) for fisheries opts for sustainable patterns of use of marine resources, it does not consider either the ecological impact on the whole ecosystem, or indeed whether any future states are ruled out by the approach, even if it is sustainable as narrowly defined (Rossberg et al., 2017). By way of contrast, still other approaches (an example of which is shaded gold in Table 3) are less

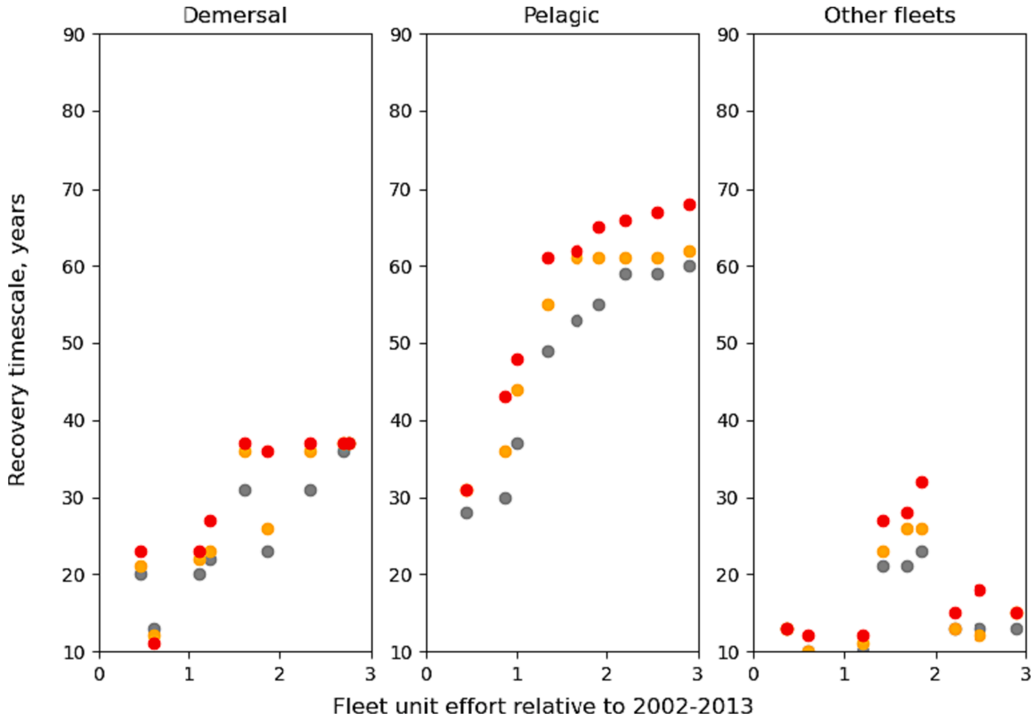

Fig. 9a. LRT as a function of fleet effort (where one unit of effort represents the average for 2003–2013), in cases where there is little effort (F ~ 0) from the other fleets. Grey circles are for the reference climate, orange for 2 K warming, and red for 4 K warming. Each dot represents one fishing scenario for a given level of warming, and the recovery timescale is set by the last functional group to recover.

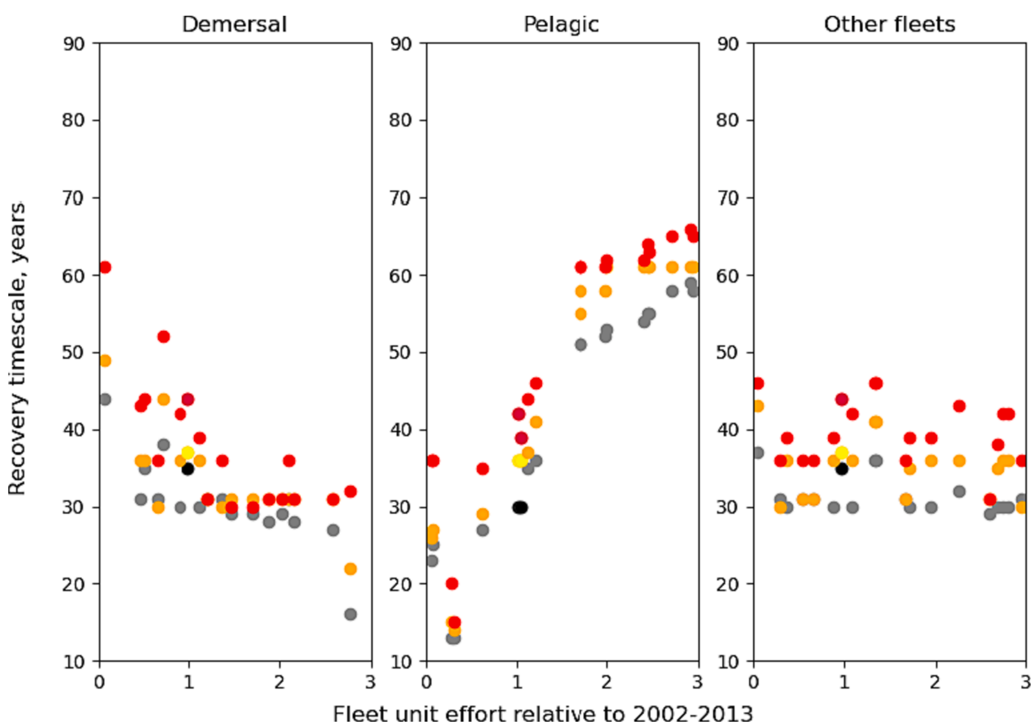

Fig. 9b. Recovery times as a function of fleet effort (where one unit of effort represents the average for 2003–2013), in cases where the other fleets fish near their average 2003–2013 intensities (F ~ 1). Grey circles are for the reference climate, orange for 2 K warming, and red for 4 K warming. Each dot represents one fishing scenario for a given level of warming, and the recovery timescale is set by the last functional group to recover. Yellow circles correspond with scenarios close to the 2003–13 period, showing that the recent past was not achieving GES on the Rossberg et al. (2017) methodology.

ambitious in that they concentrate on stabilising current states or preventing their further deterioration.

The LRT methodology is highly complementary with existing approaches, and fills a gap in determining whether the legal requirements of the MSFD and UK marine strategy (UK Government, 2018) are being met, because assessing the current state may not be sufficient in itself to determine whether GES is being achieved. In this study we have evaluated the ideas in the case of the North Sea using the StrathE2E2 "big

Table 3
Consistency of goals with legislative mandate for a variety of documented approaches. Green shading indicated use of LRT, studies shaded blue focus on current states, those in yellow focus on avoiding further deterioration (future states should not be worse than the present or recent past). Whilst there is a future-state component to MSY, it is defined in terms of being sustainable indefinitely from today's starting point, and is often informed by current stock status or risk level (e.g. Thorpe and De Oliveira 2019), so is coloured blue in the table.

Goal	Example
"provide ecologically diverse and dynamic oceans and seas which are clean, healthy and productive within their intrinsic conditions, and the use of the marine environment is at a level that is sustainable, thus safeguarding the potential for uses and activities by current and future generation"	As defined by MSFD (EU, 2008)
Minimising impact on ecosystem (relative to pristine)	Cochrane et al. (2010)
Ecosystem function is not degraded	ICES (2014a), Thorpe et al.2022
Abundances that can recover from perturbation *or* are historically stable	Rogers et al. (2010); ICES (2014b)
Fish stocks are within "safe biological limits" where they can achieve MSY.	Piet et al. 2010; Mesnel 2012, Thorpe and De Oliveira, 2019; Thorpe 2019.
Timescale of ecosystem recovery.	Rossberg et al. 2017; this study

picture" model (Heath et al., 2020, 2021) which is ideally suited to the task. In addition to the question concerning stakeholder acceptance, we asked four questions about what time horizon would be needed to prevent serious short-term degradation, and whether the approach was practical in view of uncertainties about how to define baselines, how variable these baselines might be, and when we might say that any recovery to baseline levels had been achieved. We found that variability of chosen baselines, and the methods for ascertaining whether recovery had occurred were not first order issues; however the definition of appropriate environmental baselines was critical, with recovery proving difficult or impossible if the environment shifted (e.g. due to climate change) but the baselines used to assess recovery did not. This finding is consistent with the increasing use of the "dynamic B0" (the spawning stock biomass consistent with the current environment in the absence of fishing) approach within fisheries management (MacCall et al., 1985; Punt et al., 2014; Plaganyi et al., 2019, Bessell-Browne et al., 2022), and requires the ability to set realistic baselines for recovery if LRT methodology is to be fully effective.

Our results also shed light on whether recovery timescales are a sufficient condition for attainment of GES or whether we additionally have to ensure that current states are not too degraded. Within our study, we found that the requirement for recovery within 30 years was sufficient to prevent serious degradation of any ecosystem component in the short term, if recovery precluded biomasses that were too high as well as too low. However, we also found that application of recovery timescale alone would permit very different harvesting strategies from those existing in today's (different) management regimes, with severe constraints on pelagic fishing fleets whilst demersal fleets were only weakly constrained. This shift in management might be socially unacceptable (see e.g. Schuch et al., 2021), suggesting that it might be necessary to combine constraints on management pathways as well as recovery timescales in order to give the approach necessary stakeholder legitimacy. Recommending any management approach that was very different from today's would set a high bar for the supporting evidence base, models, and methodology; see the subsequent section on caveats for more discussion of this.

The review of Smit et al. (2021) considers the utility of approaches for ensuring GES in some detail, and it is useful to cast our findings in their framework (see Table 4).

Table 4
Summary of study findings showing that LRT possesses many desirable indicator properties. Assessment is based on the methodology of Smit et al. (2021) (adapted from their Table 3). Smit et al.'s review draws on the work of Rice and Rochet, 2003, McField and Kramer, 2007, and Hayes et al., 2015 and addresses the question as to what makes an indicator good.

Desirable properties for indicators	Description of desirable property	Our findings
Sensitive	Can provide early warnings and early detection and accurately reflects the condition of the environment.	By linking future state outcomes to possible evolutions through time, the framework of Smit et al. 2021 a) may give warning that observed states imply an inability to recover desirable states on sensible timescales, and b) can indicate that environmental shifts are happening provided the modelling component has credible time-dynamics and suitable reference baselines can be modelled.
Representative/ transferrable	Can be broadly applied at different spatial and temporal scales, across regions and potentially across different habitat types.	A strength of the LRT method is that it takes a holistic approach to the ecosystem, and focusses on the response timescales of the slowest components. Method is generalisable across ecosystems.
Responsive	Can establish priorities for management and inform decision-making in a reasonable time.	Management can immediately change reference state or timescale of recovery, and see the modelled implications *NOW*. Key challenge is representing the short-term impacts of these changes in long-term target state. Having credible representations of time-dependent responses to key processes is important.
Ecologically meaningful	Can be understood and interpreted and can distinguish between natural and anthropogenic drivers of change by incorporating sound ecological theory.	Recovery time to a reference (unfished) state is easy to understand and interpret. Fishing can be disentangled from other causes of change by use of the "dynamic B0" concept. Key will be establishment of sensible baselines that exclude the pressure to be managed.
Measurable	Should provide the necessary tools and methods for management, and its effectiveness should be relatively independent of sample size.	Biomasses of key functional groups can be estimated through time.
Easy and cost effective	Is easy to use and interpret and data collection costs should be minimised.	Biomasses of functional groups are easy to understand and interpret and collection costs should be modest.
Able to set reference points	Should include the necessary data and methods to set baselines and establish thresholds for conservation purposes.	Reference points for GES can be set in terms of time taken for the last ecosystem component to recover to the unfished state at the prevailing environmental conditions (See Rossberg et al., 2017; Thorpe et al., 2022). Studies such as this can be used to set appropriate recovery times that reduce risk of catastrophic short-term states. The suggested time of 30 years by Rossberg et al. is supported as reasonable by this study.
Able to create awareness	Should aim to improve environmental understanding and awareness to engage effectively with various stakeholders.	Recovery time is intuitive, consistent with GES legislation and e.g. UK government priorities (UK Government, 2020) so awareness should be easy to generate.

5. Caveats

Since the methodology places importance on modelling the hypothetical "pressure free state" and response timescales rather than ability to reproduce today; and because models are typically calibrated on their abilities to produce states pertaining to today and the immediate past, this has important implications for model tuning and fidelity of mechanisms.

Using the recovery timescale methodology of Rossberg et al. (2017), determination of GES will be sensitive to the rates of response of ecosystem elements to perturbation, and not their absolute abundances, so we need to be sure that recovery timescales are correctly represented in models. Consequently there should be a special focus on the representation of key processes in models to ensure their timescales of response are credible. Related to this is the issue of possible hysteresis behaviour (Lewontin, 1969), which effectively represents an infinite timescale of recovery, since return to the previous state is no longer possible. Any trajectory that led to hysteresis would then be excluded from acceptability. StrathE2E2 does not display hysteresis in these scenarios. The extent to which this is realistic is not known, though it is typical of fisheries models (including LeMans (Hall et al., 2006; Thorpe et al., 2015; Spence et al., 2020), mizer (Scott et al., 2014), and the ensemble of Spence et al. (2018)), and so the risk of being unable to attain GES by this mechanism remains to be attained. If the approach of Rossberg et al (2017) is to be followed in the future the study of mechanisms that could induce hysteresis should be a research priority.

Fisheries management – with its focus on short-term stock assessments, tactical quota-setting, and with data horizons that typically do not stretch back to a low-fishing past – is vulnerable to the problem of shifting baselines (Currie et al., 2020), where we manage to an incorrectly specified target state. One way of trying to address this has been through the concept of "dynamic B0" (Bessell-Browne et al., 2022), an estimate (model-based) of the biomasses that would pertain now if there was no fishing but the environment was identical to today. The recovery methodology of Rossberg et al. (2017) has the advantage of linking with dynamic B0.

Our results assume that the stochastic variability of StrathE2E2 is reasonable as far as broad functional group responses are considered. This variability is driven by stochastic SST in the model (Figure S1). We have assumed that the North Sea warms uniformly across the year, with no change in seasonality or monthly SST variability. These assumptions are supported by the CMIP5 ensemble results for the North Sea (IPCC, 2013; Alexander et al., 2018 – their figure 11), although the CESM-LENS large initial condition ensemble (Kay et al., 2015) of the CESM model (Hurrell et al., 2013) suggests warming may be focussed in the warmer seasons. The potential impact of this could be investigated in the future. We have further assumed that i) stochastic response is primarily forced by monthly SST variability rather than other processes, and that the StrathE2E2 response to a given level of SST forcing is realistic. Both assumptions remain to be confirmed by other studies.

It should be noted that StrathE2E2 does not resolve size structure within individual functional groups. Some studies have suggested systematic shifts in size structuring with warming (Foster and Hirst, 2012; Cheung et al., 2013; Queirós et al., 2018; Audzijonyte et al., 2019b), changes that would have consequences for predicted recovery timescales. Ideally this would be investigated with alternative model structures, perhaps combined using the information-integrating ensemble approach of Spence et al. (2018, 2022) to produce a best estimate of recovery timescales using combined information from these various models.

6. Summary/Conclusions

In this study we have used the "big picture" StrathE2E2 end-to-end model to look at recovery timescales as a means of defining GES, following the suggestion of Rossberg et al. (2017). We have illustrated the method for a wide range of fisheries scenarios under different temperature conditions, and considered some key questions for its applicability to policy. In relation to these questions we found:

(1) a recovery timescale of 30 years is acceptable but 40 may be too long as it permits severely depleted states. This finding is not sensitive to warming of up to 4 °C, even though warming does tend to slow recovery responses.
(2) typical variability associated with SST had only modest impacts on timescales of recovery (confirming the discussion in Rossberg et al. (2017)).
(3) whilst results were somewhat sensitive to the definition of "recovery", this sensitivity could be greatly reduced in practice by reference to the unfished state, because any definition that does not accept the unfished state as recovered should be rejected. We also found that any sensitivity to this definition was small relative to the effects of environmental change.
(4) Results were very sensitive to the assumption of what constitutes the appropriate baseline, confirming that recovery scenarios must be considered relative to baselines in which the impacts of other factors such as the environment are catered for, and supporting the concept of dynamic B0 in fisheries management. Definition of recovery has to be sensitive to biomasses that are too high as well as too low, otherwise in the case of an ecosystem whose productivity is increasing (as here under warming), timely recovery may be insufficient to rule out interim states that have some highly impacted functional groups.
(5) LRT may provide a necessary but not sufficient constraint. In this study we found it would permit very high levels of demersal fishing if used alone which may not be acceptable to the stakeholder community.

In conclusion, our work supports the use of a recovery timescale methodology (Rossberg et al., 2017) for determination of GES, provided that it is used alongside other measures that address current or near future states (which in this case would rule out scenarios of heavy demersal fishing), and provided that we are able to estimate a suitable baseline taking account of other key factors such as climate warming. We therefore see LRT as a valuable addition to the toolkit for assessing GES, but not as a replacement for existing indicator sets which will remain essential for determination of near future states.

CRediT authorship contribution statement

Robert B. Thorpe: Conceptualization, Methodology, Formal analysis, Investigation, Visualization, Funding acquisition. **Michael Heath:** Methodology, Software, Validation. **Christopher P. Lynam:** Conceptualization, Investigation, Resources, Visualization, Funding acquisition.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

Acknowledgements

Funding for the work has been provided by the European Union (Horizon Project SEAwise) and the UK Department for Environment, Food, and Rural Affairs (DEFRA – grants MA016 and C8504). Supercomputing support has been provided in association with the UEA high performance computing group. The manuscript has greatly benefited from discussions with colleagues, particularly Georg Engelhard and Karen van der Wolfshaar, and comments from four anonymous referees, whose feedback has improved presentation of the rationale for the study.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecolind.2023.110984.

References

M.A. Alexander J.D. Scott K.D. Friedland K.E. Mills J.A. Nye A.J. Pershing A.C. Thomas J. W. Deming E.C. Carmack Projected sea surface temperatures over the 21st century: Changes in the mean, variability and extremes for large marine ecosystem regions of Northern Oceans Elementa: Science of the Anthripocene 6 2018 2018 10.1525/elementa.191.

Atmore, L.M., Aiken, M., Furni, F., 2021. Shifting baselines to thresholds: reframing exploitation in the marine environment. Front. Mar. Sci. 8 https://doi.org/10.3389/fmars.2021.742188.

Audzijonyte, A., Pethybridge, H., Porobic, J., Gorton, R., Kaplan, I., Fulton, E.A., 2019b. Atlantis: a spatially explicit end-to-end marine ecosystem model with dynamically integrated physics, ecology and socio-economic modules. Methods Ecol. Evol. 10, 1814–1819. https://doi.org/10.1111/2041-210X.13272.

Azam, F., Fenchel, T., Field, J.G., Gray, J.S., Meyer-Reil, L.A., Thingstad, F., 1983. The ecological role of water column microbes in the sea. Mar. Ecol. Prog. Ser. 10, 257–263.

Bartley, T.J., McCann, K.S., Bieg, C., Cazelles, K., Granados, M., Guzzo, M.M., MacDougall, A.S., Tunney, T.D., McMeans, B.C., 2019. Food web rewiring in a changing world. Nat. Ecol. Evol. 3 (3), 345–354.

Bessell-Browne, P., Punt, A.E., Tuck, G.N., Day, J., Klaer, N., Penney, A., 2022. The effects of implementing a 'dynamic B0' harvest control rule in Australia's Southern and Eastern Scalefish and Shark Fishery. Fish. Res. 252.

Borgström, S., Bodin, Ö., Sandström, A., Crona, B., 2015. Developing an analytical framework for assessing progress toward ecosystem-based management. Ambio 44, 357–369. https://doi.org/10.1007/s13280-015-0655-7.

Burgess, M.G., Becker, S.L., Langendorf, R.E., Fredston, A., Brooks, C.M., 2023. Climate change scenarios in fisheries and aquatic conservation research. ICES J. Mar. Sci.

Butenshon, M., Clark, J., Aldridge, J.N., Allen, J.I., Artioli, Y., Blackford, J., et al., 2016. ERSEM 15.06: a generic model for marine biogeochemistry and the ecosystem dynamics of the lower trophic levels. Geosci. Model Dev. 9, 1293–1339. https://doi.org/10.5194/gmd-9-1293-2016.

CBD (2014). Ecosystem Approach Sourcebook. Available online at: https://www.cbd.int/ecosystem/sourcebook/default.shtml.

Cheung, W.W.L., Sarmiento, J.L., Dunne, J., Frölicher, T.L., Lam, V.W.Y., Deng Palomares, M.L., Watson, R., Pauly, D., 2013. Shrinking of fishes exacerbates impacts of global ocean changes on marine ecosystems. Nat. Clim. Chang. 3 (3), 254–258.

Christensen, N.L., Bartuska, A.M., Brown, J.H., Carpenter, S., D'Antonio, C., Francis, R., Franklin, J.F., MacMahon, J.A., Noss, R.F., Parsons, D.J., Peterson, C.H., Turner, M. G., Woodmansee, R.G., 1996. The report of the ecological society of America committee on the scientific basis for ecosystem management. Ecol. Appl. 6 (3), 665–691.

Currie, J.C., Atkinson, L.J., Sink, K.J., Attwood, C.G., 2020. Long-term change of demersal fish assemblages on the inshore Agulhas bank between 1904 and 2015. Front. Mar. Sci. 7, 1–16. https://doi.org/10.3389/fmars.2020.00355 (shifting baselines trap).

Cury, P.M., Boyd, I.L., Bonhommeau, S., Anker-Nilssen, T., Crawford, R.J.M., Furness, R. W., et al., 2011. Global seabird responses to forage fish depletion – one-third for the birds. Science 334, 1703–1706. https://doi.org/10.1126/science.1212928.

Department of Environmental Affairs, 2015. South Africa's 2nd National Biodiversity Strategy and Action Plan. Government of South Africa, Pretoria.

Driver, A., Sink, K. J., Nel, J. N., Holness, S., Van Niekerk, L., Daniels, F.,et al. (2011). NBA (National Biodiversity Assessment) 2011: An Assessment of South Africa's Biodiversity and Ecosystems. Synthesis Report. Pretoria: South African National Biodiversity Institute.

Eu., 2008. Directive 2008/56/EC of the European Parliament and of the Council of 17 June 2008 establishing a framework for community action in the field of marine environment policy (marine stategy framework directive). Off. J. Eur. Union L 164 (2008), 19–40.

Foster, J., Hirst, A.G., 2012. The temperature-size rule emerges from ontogenetic differences between growth and development rates. Funct. Ecol. 26, 483–492. https://doi.org/10.1111/j.1365-2435.2011.01958.x.

Fulton, E.A., 2010. Approaches to end-to-end ecosystem models. J. Mar. Syst. 81, 171–183. https://doi.org/10.1016/j.jmarsys.2009.12.012.

Giricheva, E., 2015. Aggregation in ecosystem models and model stability. Prog. Oceanogr. 134, 190–196. https://doi.org/10.1016/j.pocean.2015.01.016.

Hall, S.J., Collie, J.S., Duplisea, D.E., Jennings, S., Bravington, M., Link, J., 2006. A length-based multispecies model for evaluating community responses to fishing. Can. J. Fish. Aquat. Sci. 63, 1344–1359.

Hayes, K.R., Dambacher, J.M., Hosack, G.R., Bax, N.J., Dunstan, P.K., Fulton, E.A., Thompson, P.A., Hartog, J.R., Hobday, A.J., Bradford, R., Foster, S.D., Hedge, P., Smith, D.C., Marshall, C.J., 2015. Identifying indicators and essential variables for marine ecosystems. Ecol. Ind. 57, 409–419. https://doi.org/10.1016/j.ecolind.2015.05.006.

Heath, M.R., 2012. Ecosystem limits to food web fluxes and fisheries yields in the North Sea simulated with an end-to-end food web model. Prog. Oceanogr. 102, 42–66. https://doi.org/10.1016/j.pocean.2012.03.004.

Heath, M. R., Speirs, D. C., McDonald, A., and Wilson, R. (2021). StrathE2E2 Version 3.3.0: Implementation for the North Sea. Available online at: https://marineresourcemodelling.gitlab.io/ (accessed January 31, 2022).

Heath, M.R., Speirs, D.C., Thurlbeck, I., Wilson, R., 2020. StrathE2E2: an R package for modelling the dynamics of marine food webs and fisheries. Methods Ecol. Evol. 12, 280–287. https://doi.org/10.1111/2041-210X.13510.

Hurrell, J.W., Holland, M.M., Gent, P.R., Ghan, S., Kay, J.E., Kushner, P.J., Lamarque, J.-F., Large, W.G., Lawrence, D., Lindsay, K., Lipscomb, W.H., Long, M.C., Mahowald, N., Marsh, D.R., Neale, R.B., Rasch, P., Vavrus, S., Vertenstein, M., Bader, D., Collins, W.D., Hack, J.J., Kiehl, J., Marshall, S., 2013. The community earth system model: a framework for collaborative research. Bull Amer Meteor Soc 94 (9), 1339–1360.

Hyder, K., Rossberg, A.G., Allen, J.I., Austen, M.C., Barciela, R.M., Bannister, H.J., Blackwell, P.G., Blanchard, J.L., Burrows, M.T., Defriez, E., Dorrington, T., Edwards, K.P., Garcia-Carreras, B., Heath, M.R., Hembury, D.J., Heymans, J.J., Holt, J., Houle, J., Jennings, S., Mackinson, S., Malcolm, S.J., McPike, R., Mee, L., Mills, D.K., Montgomery, C., Pearson, D., Pinnegar, J.K., Pollicino, M., Popova, E.E., Rae, L., Rogers, S.I., Speirs, D., Spence, M.A., Thorpe, R., Turner, R.K., van der Molen, J., Yool, A., Paterson, D.M., 2015. Making modelling count - increasing the contribution of shelf-seas community and ecosystem models to policy development and management. Mar. Policy 61, 291–302.

ICES, 2014a. Report of the Workshop to review the 2010 Commission Decision on criteria and methodological standards on good environmental status (GES) of marine waters; Descriptor 6 Copenhagen.

ICES, 2014b. Report of the Workshop to review the 2010 Commission Decision on criteria and methodological standards on good environmental status (GES) of marine waters; Descriptor 4 Foodwebs Copenhagen.

ICES 2015. Report of the Workshop on Guidance for the Review of MSFD Decision Descriptor 4 –Foodwebs II (WKGMSFDD4-II). ICES Document CM 2015\ACOM:49. Copenhagen: ICES.

IPCC 2013 Summary for Policymakers. In: Climate Change 2013 The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change Stocker, TF , Qin, D , Plattner, G-K , Tignor, M , Allen, SK , Boschung, J , Nauels, A , Xia, Y , Bex, V and Midgley, PM (eds.). Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Iwasa, Y., Andreasen, V., Levin, S., 1987. Aggregation in model ecosystems I. Perfect aggregation. Ecol. Modell. 37, 287–302. https://doi.org/10.1016/0304-3800(87)90030-5.

Jones, L.P., Turvey, S.T., Papworth, S.K., 2021. Is there evidence of shifting baseline syndrome in environmental managers? An assessment using perceptions of bird population targets in UK nature reserves. J. Environ. Manage. 297, 113308 https://doi.org/10.1016/j.jenvman.2021.113308.

Kay, J.E., Deser, C., Phillips, A., Mai, A., Hannay, C., Strand, G., Arblaster, J.M., Bates, S.C., Danabasoglu, G., Edwards, J., Holland, M., Kushner, P., Lamarque, J.-F., Lawrence, D., Lindsay, K., Middleton, A., Munoz, E., Neale, R., Oleson, K., Polvani, L., Vertenstein, M., 2015. The Community Earth System Model (CESM) large ensemble project: a community resource for studying climate change in the presence of internal climate variability. Bull Amer Meteor Soc 96 (8), 1333–1349.

Kidd, S., Maltby, E., Robinson, L., Barker, A., 2011. The ecosystem approach and planning and management of the marine environment. In: Kidd, S., Plater, A., Frid, C. (Eds.), The Ecosystem Approach to Marine Planning and Management. Earthscan, London, pp. 1–33.

Leslie, H.M., McLeod, K.L., 2007. Confronting the challenges of implementing marine ecosystem-based management. Front. Ecol. Environ. 5, 540–548. https://doi.org/10.1890/060093.

Lewontin, R.C., 1969. The meaning of stability. Brookhaven Symp. Biol. 22, 13–23. PMID 5372787.

C.P. Lynam L. Uusitalo J. Patricio C. Piroddi A.M. Queiros H. Teixeira et al. Use of Innovative Modelling Tools within the Implementation of the Marine Strategy Framework Directive 2016 Mar. Sci Front 10.3389/fmars.2016.00182.

A MacCall, A.D., Klingbeil, R.A., Methot, R.D., 1985. Recent increased abundance and potential productivity of pacific mackerel (Scomber japonicas). CalCOFI Report 26CalCOFI, La Jolla, CA, pp. 119–129.

Mackinson, S., Daskalov, G., 2007. An ecosystem model of the North Sea to support an ecosystem approach to fisheries management: description and parameterisation. Sci. Ser. Tech. Rep. 142, 196.

Mcfield, M., Kramer, P., 2007. Healthy Reefs for Healthy People: A Guide to Indicators of Reef Health and Social Well-being in the Mesoamerican Reef Region. With contrbutions by M. Gorrez M. McPherson 208pp.

McKay, M.D., Conover, W.J., Beckman, R.J., 1979. A comparison of three methods for selecting values of input variables in the analysis of output from a computer code. Technometrics 21, 239–245.

Mesnil, B., 2012. The hesitant emergence of maximum sustainable yield (MSY) in fisheries policies in Europe. Mar. Policy 36, 473–480.

Morris, D., Speirs, D., Cameron, A., Heath, M., 2014. Global sensitivity of and end-to-end marine ecosystem model of the North Sea: factors affecting the biomass of fish and benthos. Ecol. Model. 273, 251–263. https://doi.org/10.1016/j.ecolmodel.2013.11.019.

O'Higgins, T.G., Lago, M., DeWitt, T.H., 2020. Ecosystem-Based Management, Ecosystem Services and Aquatic Biodiversity, Theory, Tools, and Applications. Springer Link Open Access e-Book. https://link.springer.com/book/10.1007/978-3-030-45843-0.

Papworth, S.K., Rist, J., Coad, L., Milner-Gulland, E.J., 2008. Evidence for shifting baseline syndrome in conservation. Conserv. Lett. 2 (2), 93–100.

Pauly, D., 1995. Anecdotes and the shifting baseline syndrome of fisheries. Trends Ecol. Evol. 10 (10), 430.

Piroddi, C., Teixeira, H., Lynam, C.P., Smith, C., Alvarez, M.C., Mazik, K., Andonegi, E., Churilova, T., Tedesco, L., Chifflet, M., Chust, G., Galparsoro, I., Garcia, A.C., Kämäri, M., Kryvenko, O., Lassalle, G., Neville, S., Niquil, N., Papadopoulou, N., Rossberg, A.G., Suslin, V., Uyarra, M.C., 2015. Using ecological models to assess ecosystem status in support of the European marine strategy framework directive. Ecol. Ind. 58, 175–191.

Plaganyi, E.E., Haywood, M.D.E., Gorton, R.J., Siple, M.C., Deng, R.A., 2019. Management implications of modelling fisheries recruitment. Fish. Res. 217, 169–184.

Punt, A.E., A'mar, T., Bond, N.A., Butterworth, D.S., de Moor, C.L., De Oliveira, J.A.A., Haltuch, M.A., Hollowed, A.B., Szuwalski, C., 2014. Fisheries management under climate and environmental uncertainty: control rules and performance simulation. ICES J. Mar. Sci. 71 (8), 2208–2220.

Queirós, A.M., Fernandes, J., Genevier, L., Lynam, C.P., 2018. Climate change alters fish community size-structure, requiring adaptive policy targets. Fish Fish. 19, 613–621. https://doi.org/10.1111/faf.12278.

Rice, J.C., Rochet, M.J., 2003. A framework for selecting a suite of indicators for fisheries management. Ocean Coast. Manag. 46 (2003), 235–259. https://doi.org/10.1016/j.icesjms.2005.01.003.

Rossberg, A.G., Uusitalo, L., Berg, T., Zaiko, A., Chenuil, A., Uyarra, M.C., Borja, A., Lynam, C.P., 2017. Quantitative criteria for choosing targets and indicators for sustainable use of ecosystems. Ecol. Ind. 72, 215–224. https://doi.org/10.1016/j.ecolind.2016.08.005.

M.A. Rudd M. Dickey-Collas J. Ferretti E. Johannesen N.M. Macdonald R. McLaughlin M. Rae T. Thiele J.S. Link Ocean Ecosystem-Based Management Mandates and Implementation in the North Atlantic 2018 Mar. Sci Front 10.3389/fmars.2018.00485.

Schuch, E., Gabbert, S., Richter, A.P., 2021. Institutional inertia in European fisheries – Insights from the Atlantic horse mackerel case. Mar. Policy 128, 104464. https://www.sciencedirect.com/science/article/pii/S0308597X21000750.

Scott, F., Blanchard, J.L., Andersen, K.H., 2014. mizer: an R package for multispecies, trait-based and community size spectrum ecological modelling. Methods Ecol. Evol. 5 (10), 1121–1125. https://besjournals.onlinelibrary.wiley.com/doi/10.1111/2041-210X.12256.

Shin, Y.-J., Bundy, A., Shannon, L.J., Blanchard, J.L., Chuenpagdee, R., Coll, M., Knight, B., Lynam, C., Piet, G., Richardson, A.J., 2012. Global in scope and regionally rich: an IndiSeas workshop helps shape the future of marine ecosystem indicators. Rev. Fish Biol. Fish. 22 (3), 835–845.

Smit, K.P., Bernard, A.T.F., Lombard, A.T., Sink, K.J., 2021. Assessing marine ecosystem condition: A review to support indicator choice and framework development. Ecol. Ind. 121.

Smith, D., Punt, A., Dowling, N., Smith, A., Tuck, G., Knuckey, I., 2009. Reconciling approaches to the assessment and management of data-poor species and fisheries with Australia's Harvest Strategy Policy. Marine and Coastal Fisheries: Dynamics, Management and Ecosystem Science 1 (1), 244–254.

Spence, M.A., Blanchard, J.L., Rossberg, A.G., Heath, M.R., Heymans, J.J., Mackinson, S., et al., 2018. A general framework for comparing ecosystem models. Fish Fish. 19, 1031–1042. https://doi.org/10.1111/faf.12310.

Spence, M.A., Bannister, H.J., Ball, J.E., Dolder, P.J., Griffiths, C.A., Thorpe, R.B., Tsikliras, A.C., 2020. LeMaRns: A Length-based Multi-species analysis by numerical simulation in R. PLoS ONE 15 (2). https://doi.org/10.1371/journal.pone.0227767.

Spence, M.A., Lynam, C.P., Thorpe, R.B., Heneghan, R.F., Dolder, P.J., 2022. Synthesising Empirical and Modelling Studies to Predict Past and Future Primary Production in the North Sea. Front. Mar. Sci. https://doi.org/10.3389/fmars.2022.828623.

Thorpe, R.B., 2019. What is multispecies MSY? A Worked Example from the North Sea. Journal of Fish Biology 94 (6), 1011–1018.

Thorpe, R.B., De Oliveira, J.A.A., 2019. Comparing conceptual frameworks for a fish community MSY (FCMSY) using management strategy evaluation – an example from the North Sea. ICES J. Mar. Sci. 76 (4), 813–823. https://doi.org/10.1093/icesjms/fsz015.

Thorpe, R.B., Le Quesne, W.J.F., Luxford, F., Collie, J.S., Jennings, S., 2015. Evaluation and management implications of uncertainty in a multispecies size-structured model of population and community responses to fishing. Methods Ecol. Evol. 6, 49–58.

Thorpe, R.B., Dolder, P.J., Reeves, S., Robinson, P., Jennings, S., 2016. Assessing fishery and ecological consequences of alternate management options for multispecies fisheries. ICES J. Mar. Sci. 73 (6), 1503–1512.

Thorpe, R.B., Arroyo, N.L., Safi, G., Niquil, N., Preciado, I., Heath, M., Pace, M.C., Lynam, C.P., 2022. The Response of North Sea Ecosystem Functional Groups to Warming and Changes in Fishing Front. Mar. Sci. 9 https://doi.org/10.3389/fmars.2022.841909.

UK Government 2018. European Union (Withdrawal) Act. https://www.legislation.gov.uk/ukpga/2018/16/contents/enacted/data.htm.

UK Government Fisheries Act 2020 https://www.legislation.gov.uk/ukpga/2020/22/contents/enacted.

UNEP, 1998. Report of the workshop on the ecosystem approach", in Conference of the Parties to the Convention on Biological Diversity. UNEP, Lingongwe.

Worm, B., Hilborn, R., Baum, J.K., Branch, T.A., Collie, J.S., Costello, C., Fogarty, M.J., Fulton, E.A., Hutchings, J.A., Jennings, S., Jensen, O.P., Lotze, H.K., Mace, P.M., McClanahan, T.R., Minto, C., Palumbi, S.R., Parma, A.M., Ricard, D., Rosenberg, A. A., Watson, R., Zeller, D., 2009. Rebuilding global fisheries. Science 325 (5940), 578–585.