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Ecosystems say good management pays off

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Abstract

Understanding the strengths and weaknesses of alternative assessment methods, harvest strategies and management approaches is an important part of operationalising single-species and ecosystem based fisheries management. Simulations run using two variants of a whole of ecosystem model for the Southern and Eastern Scalefish and Shark Fishery (SESSF) area shows that (i) data-rich assessments outperform data-poor assessments for target species and that this performance is reflected in the values of many system-level ecosystem indicators; (ii) ecosystem and multispecies management outperforms single-species management applied over the same domain; (iii) investment in robust science-based fisheries management pays dividends even when there are multiple jurisdictions, some of which are not implementing effective management; and (iv) that multispecies yield-oriented strategies can deliver higher total catches without a notable decline in overall system performance, although the resulting system structure is different to that obtained with other forms of ecosystem based management.

Keywords

Atlantis, ecosystem based management, fisheries, harvest strategies, risk equivalency

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Introduction

Even with a growing list of pressures on marine ecosystems (e.g., IPCC, 2013; Halpern *et al.*, 2015; Wilcox *et al.*, 2015; Hobday *et al.*, 2016; Kunc *et al.*, 2016; Breitburg *et al*., 2017), resource management can be seen as potentially excessively costly in a world where budgets for science and environmental management are under pressure. Ecosystem based management, whether focused solely on fisheries or integrated across multiple marine uses, could add a further regulatory and fiscal burden, particularly given its call for a broader system perspective. Evidence is growing that the portfolio approach provided by an ecosystem perspective provides ecological and financial benefits (Link, 2018). Guidelines (Essington *et al.*, 2016, NOAA 2016) have been created to assist in the implementation of ecosystem based fisheries management (EBFM). Nevertheless, the perceived complexity of the task and its system-specific nature - successful implementation of EBFM requires tailoring broad concepts to local conditions (Trochta *et al*., 2018) - can lead to scepticism and a sense of being overwhelmed. The feasibility of EBFM has been questioned as some have assumed it means expanding the types of management implemented for species targeted by fisheries to a wider range of species, or at the very least adding the tracking of many additional ecosystem indicators.

In contrast to expanding the range of species is the assertion that single-species management, if implemented correctly, should be sufficient to achieve EBFM (NRC 1999). Evidence to support this assertion has been mixed, and largely based on discussions of the weaknesses of single-species management and whether these will be addressed using EBFM approaches (Fogarty, 2014). For example, Hilborn (2011) explained that EBFM is required because single-species management does not account for interactions (trophic or otherwise) amongst system components or for effects on non-target species. Simberloff (1998) also listed the many issues associated with single-species management – even if that management is focused on indicator, umbrella, or flagship species. Specifically, Simberloff (1998) raised concerns around the appropriateness of single species as proxies for other parts of the system; and whether managing particular species incidentally leads to satisfactory outcomes for other species. Simberloff (1998) suggested that managing for keystone species may effectively deliver EBFM. However, it is not clear that each ecosystem has a keystone species. In contrast, several authors (e.g. Jennings, 2006; Hicks *et al*., 2016) have questioned whether a move to an ecosystem approach would address the key drivers behind unsustainable fishing (identified by FAO, 2002) – e.g. inappropriate incentives; market distortions; high demand; poverty; a lack of livelihood alternatives or diversity; information gaps; and weak governance, compliance and enforcement. Of these drivers, EBFM should (by definition) be robust to the need to consider the interactions of all sectors (fisheries and otherwise) that impact the marine environment. However, the realisation that this would not successfully address issues of governance of socioeconomic drivers has recently seen more emphasis on the human dimensions of EBFM (Urquhart *et* al., 2011; Charles, 2014; Andersen *et al*., 2015; Bundy *et al.*, 2017).

Jurisdictional divisions can also lead to fisheries management tensions, particularly when ecosystems span multiple jurisdictions. This has led, for example, to UN agreements regarding straddling stocks (UN, 1995), but has grown to be a larger system-level topic as shifting environmental conditions and variable stock status have challenged fisheries management authorities on either side of national borders or who participate in regional fisheries management organisations (Hátún *et* al., 2009; Spijkers and Boonstra, 2017). Some have questioned whether the additional investment in management is warranted if it is only applied to part of the system (e.g. stock) in discussions of the implementation of EBFM (or even single-species management of straddling stocks) (Gulland, 1980; Munro *et al*., 2004).

Mathematical models are useful for exploring whether single-species management approaches can achieve EBFM objectives given that direct observational evidence regarding ecosystem-level objectives is hard to collect at the scale of entire ecosystems and under controlled conditions. Ecosystem modelling has matured as a scientific discipline over the last thirty years, with hundreds of models developed on scales from local (e.g. single bays) to global, and using a diverse range of modelling platforms and philosophies (from trophically-focused to size- and agent-based). One modelling platform that can be used to explore the implications of fisheries management in an ecosystem context is Atlantis (Fulton *et al*., 2011). Atlantis is a whole-of-ecosystem model that includes the major oceanographic and ecological processes, food webs and human users (Fulton *et al*., 2011). It can be used to explore the ecosystem implications of alternative management strategies (Fulton *et al*., 2014), potential future trajectories under climate and management scenarios (Kaplan *et al.*, 2010; Weijerman *et al*., 2015) and the potential outcomes of the implementation of tiered assessment methods and harvest strategies for target species (Fulton *et al*., 2016; Dichmont *et al*., 2017).

This paper evaluates the incremental ecosystem value of sound fisheries management – single-species and integrated multispecies or ecosystem based management – and its robustness to some key challenges, such as differing management approaches on either side of jurisdictional boundaries. Jurisdictions that lack the capacity to implement sound fisheries management typically do not collect time series data on ecosystem state. Moreover, information facilitating ecosystem-level assessments is not necessarily available even when there has been a strong investment in single-species management (e.g. fisheries-independent surveys do not have a long history or broad coverage in Australia despite significant institutional and industry effort to implement robust fisheries management). This analysis takes a model-based approach, using a model to represent the whole system (interacting ecological components, fisheries and the management methods and processes). This approach means there can be control of what forms of management are implemented (and the levels of compliance and responsiveness) and what data streams are "collected" (i.e. so a full ecosystem perspective can be gained). In this way, this paper allows for consideration of the ecosystem-level outcomes of: (i) implementing data-rich versus data-poor assessment for target species; (ii) implementing assessment methods/harvest strategies on all or only some of the species targeted by fishing; (iii) implementing ecosystem and multispecies management versus single-species management; and (iv) of managing only one part of a multi-jurisdictional marine ecosystem.

Methods

The study region was the Australian Southern and Eastern Scalefish and Shark Fishery (SESSF), a large marine ecosystem that extends across southern Australian from subtropical to subpolar waters (Figure 1).

Terminology

Before describing the models and the simulations run we first clarify some terminology that will be used through the rest of the paper:

- A *target species* is a species that is of primary interest to the fishers, it is central to their decision making regarding where and what to fish. These species include all those species marked with an X in the column "Main target species" in Table S1.
- *Treatment species* are those species managed under a harvest strategy in scenarios P-V (see below). These species include target, by-product and bycatch species.
- *Non-treatment species* are target species that are not being managed using the harvest strategy in that particular scenario. If they are also a total allowable catch (TAC) managed species, they are managed using the 2005 quota levels throughout the run.

- *Non-target species* are all the other groups in the model, which may or may not interact with the fishery, but are not targets of the fishery operations.
- *By-product species* are non-target species with market value that are landed along with the main target species.
- *Bycatch species* are non-target species caught by the fishery that are discarded and not landed.
- *Iconic species* are species of conservation concern – marine mammals, seabirds and large sharks.

There are many species under TAC in the SESSF (listed in the column "EBFM TAC species" in Table S1) – including target species, by-product species and particularly vulnerable bycatch species (such as gulper sharks and school sharks); bycatch species are included in TAC management rules because they may previously have been a target species but are now depleted (e.g. school sharks) or because they are a species of conservation concern (e.g. gulper sharks) where discards are tracked. Discards are accounted for in the assessment and TAC setting process in the SESSF.

The species are not all handled in the same way in each of the management scenarios. This is because the scenarios were defined in response to management questions and so the details do not fit an exhaustive or systematic plan. However, this reflects the complicated nature of managing a multispecies fishery.

Model content

The Atlantis modelling framework was used as the basis of the analysis. Multiple Atlantis models of increasing sophistication have been developed for the SESSF over the past 20 years. Good understanding of both the fishery and the models makes it appropriate for this paper. We first tested the implications of managing only one part of an ecosystem, using Atlantis-AMS, which was previously used by Fulton *et al.*, (2014) to explore alternative management strategies for this fishery. Given the additional data requirements associated with the assessment methods/harvest strategies applied to the individual species in the other part of the study it was necessary to conduct that analysis using Atlantis-RCC, which is very similar to Atlantis-AMS, but includes multiple size-at-age morphs for the assessed species; this model was originally developed to evaluate the efficacy of the use of tiered assessment methods and harvest strategies

(Fulton *et al.*, 2016; Dichmont *et al.*, 2017). While it would have been desirable to have all the simulations directly comparable, by rerunning the simulations conducted using Atlantis-AMS, the computational cost of running Atlantis-RCC made this impracticable given the desire to also consider uncertainty.

Both Atlantis models use the same 71 model regions ("boxes") based on the physical and ecological properties of southeast Australia (Figure 1) – determined primarily by the distribution of the water bodies and the geomorphology of the area as summarized in bioregionalisations (IMCRA, 1998; Butler *et al.*, 2001; Lyne and Hayes, 2005; Fulton *et al.*, 2007). Each of these polygonal boxes has up to five water column layers (dictated by total depth; shallower boxes have fewer layers) and a single sediment layer.

The oceanographic (physical) environment in these Atlantis models includes ocean currents, temperature, salinity, pH, oxygen and nutrient levels. Exchanges (horizontal and vertical) between spatial boxes and layers, as well as temperature and salinity in each layer of each box, were taken from the data-assimilated version of the global "Ocean Forecasting Australia Model" (OFAM; Oke *et al.*, 2005; the database used is available at http://www.bom.gov.au/bluelink/ and SPINUP6 from http://www.marine.csiro.au/ofam1/). During the projection period of each simulation (2005-2050) the OFAM reanalysis was used until 2014, then the patterns of variance in the environmental conditions were looped (from the start of the time series) to complete the projection period; trends in the conditions were maintained in-line with those found in long-term climate projections (as detailed by Fulton and Gorton, 2014).

The model structure in these models is described in Table 1, with minor variations between the two models. Atlantis-RCC includes a few more species than Atlantis-AMS and multiple size-at-age morphs for the species marked as treatment or main target species in Table S1. These morphs represent multiple growth variants for the species, each with its own growth rate and hence multiple size-at-age curves for each species. Both Atlantis-AMS and Atlantis-RCC use the same dynamic growth model formulation, but a single fixed growth rate parameterisation is used per cohort per species in Atlantis-AMS, while the parameters for each cohort are drawn from a distribution in Atlantis-RCC. For all other parameters, one set of biological parameter values (i.e. values for non-predation mortality rates, consumption and growth rates, habitat preferences, movement rates etc.) per species group (or morph) is used for the entire model domain. The exception to this is when a group is defined as having multiple stocks (see Table 1), in which case fecundity, background mortality and diet connection strength varied among stocks.

Food web pathways in Atlantis are defined based on: the maximum potential availability of each prey to each potential predator; the level of physical contact (i.e. spatial overlap within a box given habitat preferences and patchiness); the state of habitat (refugia); and gape limitation (i.e. size of the mouth versus size of the prey given the feeding mode of the predator). Atlantis-AMS uses Heaviside step function-like diet size windows, whereas Atlantis-RCC uses smoother curves (so that realized diets match observed diets when multiple growth morphs are modelled).

Ideally Atlantis should be run with multiple plausible parameterisations, to allow for consideration of uncertainty regarding ecological processes or socioeconomic profiles. All simulations run with Atlantis-AMS (detailed below) were under the alternative parameterisations available for this model (these parameterisations are distinguished in particular by the strength of the trophic interactions). Only a single parameterisation was available for the Atlantis-RCC runs due to the technical difficulty of achieving a stable model state using multiple growth morphs. All the parameter sets used were determined by calibrating the models to available historical biological and catch data (Fulton *et al.*, 2007, 2014) using a pattern-oriented modelling approach (Fulton *et al.*, 2007; Kramer-Schadt *et al.*, 2007), whereby the most uncertain parameters were adjusted according to the following criteria: (i) the predicted spatial distributions and time series of biomasses, age structure, realized diet composition, and catches, must approximate the shape, magnitude and variability of observed time series across the majority of boxes; (ii) observed catches and discards must be sustained without rendering any model group extinct; and (iii) rate parameters must not be adjusted beyond bounds reported in the literature without expert advice from researchers active in the region. In this way, parameters were set to achieve (a) a stable ecosystem, under constant fishing pressure, with biomass and parameter values within the range of biomass values reported for these groups in the literature; and (b) produce time series for the target and surveyed species that matched observed time series and spatial distributions. The parameter pedigree (i.e. the relative uncertainty and reliability associated with each parameter) was set based on the data used to provide the initial parameter values (i.e. whether taken from the local ecosystem, sister species, general ecological theory, etc.) and sensitivity to that parameter as defined in the analyses of Pantus and Dennison (2005) and Fulton *et al*., (2007). In practice, this meant most tuning modifications were made to the diet availabilities, growth and consumption rates, background natural mortalities (especially for the highest trophic levels), fecundity levels and the steepness of the stock-recruitment curve.

The models were initialized for conditions in 1980. Available biomass estimates for the biological groups (e.g. from Morison *et al*. 2012 for assessed species) were used to set the initial 1980 abundances. For all other species, historical fish-down scenarios run by Fulton *et al*. (2007) were used to set relative depletion levels in 2005 versus 1980 and then 1980 biomass levels calculated by dividing estimated 2005 biomasses by the associated depletion levels (e.g. if the relative depletion was 50% then the 1980s biomass was twice the 2005 estimate of biomass).

Both models used the socio-economic effort allocation model of an earlier Atlantis model for the region, Atlantis-SE (Fulton *et al*., 2007), including its price and cost structures. This effort allocation model is largely driven by two main components — a quota trading module and a métier-level space-time dynamic effort module (Fulton et al., 2007; van Putten et al., 2013). However, the model also explicitly models prices (accounting for market distortion and perverse market-driven incentives), as well as different behavioural profiles across fishers, which allows them react to management actions, their social (trading and information) networks and perceived ecosystem state in diverse ways. This dynamically determines which gears are used, which suite of species is targeted by fishers through time (allowing for shifting multi-species targeting), as well as where and at what time of year fishing takes place, and how these patterns change through time. The model also determines whether fishers invest more into the fishery or alternatively choose to leave altogether. In terms of harvesting the ecosystem, this means the model does not assume a catch limit must be taken exactly (i.e. undercatch can occur), while also allowing for non-compliance and imperfect targeting (i.e. accidental overcatch, although this is constrained, as it is in reality).

Simulations

Path dependency of depletion or changing status can be important for determining ecosystem state so each simulation included historical fishing of the system (1980 – 2005) and then a 50-year projection of the system under the conditions of interest. The historical period involved the actual historical catch limits for each target species, as well as the actual values for environmental drivers. The projections (management simulations) were then run from 2005 to 2050 (schematic shown in Figure 2) and the time series for all ecosystem components and fisheries catch and effort per métier stored by time-step and spatial location.

Management simulations were conducted to explore the impact of management on the broader ecosystem; these are described more fully below (and in Tables 2 and 3), but can be grouped into two sets: (i) tiered (data-rich to data-limited) assessment methods/harvest strategies applied to individual species, or combinations of species, using Atlantis-RCC; and (ii) differential management across jurisdictions using Atlantis-AMS. Simulations were conducted for two other management strategies to provide 'bounding results' – (i) unconstrained fishing and (ii) integrated EBFM (defined in more detail below). These strategies were implemented across the entire model domain. Unconstrained fishing used the same effort allocation model as the other simulations, but all forms of fisheries management (i.e. all spatial zoning, gear restrictions, catch limits) were removed at the start of the projection period and the fishery became open access. The initial number of vessels per sector was as of 2005 and after that extra (or less) effort in the form of additional vessels could be introduced into the fishery based on a simple CPUE-based rule following Link *et al.* (2010):

$$V_{t,j} = \begin{cases} \left(1+\alpha_j\right)\cdot V_{t-1,j} & \text{if } CPUE \geq \kappa_H \\ \left(1-\alpha_j\right)\cdot V_{t-1,j} & \text{if } CPUE \leq \kappa_L \\ V_{t-1,j} & \text{otherwise} \end{cases}$$

where $V_{t,j}$ is the number of vessels in fleet (gear type) j during year t; α_j s the rate of growth or contraction for gear type j; and the κ are the CPUE threshold levels (set per métier based on historical fishing patterns in Australia, the USA and Europe; example fits given in Figure S1).

The integrated EBFM approach matched that of Fulton *et al.* (2014). It uses an intentionally multi-faceted set of management methods to handle each of the main objectives and system components, and employs: gear-specific spatial zoning and domain-wide depth and habitat specific closures; seasonal closures of fishing on spawning aggregations or migrations; and regional quotas for 24 of the target groups that shape the fishery's exploitation patterns and economic drivers (listed in Table S1) and groups of conservation concern (e.g. gulper sharks) on an annual cycle (using the first harvest strategy listed in Table 2). Catch limits were set by stock (i.e. were set specific to a region for all species marked with an * in Table 1), accounting for discards; were reconciled on landing; and were adjusted so that no vulnerable companion species was at risk (i.e. catch limits were reduced if a species caught along with the target species could not sustain that level of fishing pressure that would be required to land the full quota of the target species). In addition, there were trip-level catch limits for vulnerable bycatch species, bycatch reduction devices and limits on permissible gears; see Fulton *et al.* (2014) for additional technical details.

Twenty replicates were undertaken for each scenario. Computational speed precluded a larger set of replicates. However, this number was adequate given the deterministic nature of Atlantis (a brief exploration showed that increased numbers of simulations did not materially alter the results). The random deviates governing stochasticity (effort allocation and observation error) were replicate-specific, meaning that each scenario run was compared directly only to matching runs from other scenarios that used an identical set of random deviates. This ensures maximum comparability of the results (i.e. the results are analogous to paired tests). Nevertheless, it is still safest to consider the results in a relative sense. Consequently, the indicators for each scenario are compared to the results under unconstrained fishing.

Harvest strategies

The harvest strategies explored are listed in Table 2. These strategies consist of an assessment method and a decision rule, and included those current in 2014 in the SESSF (Smith *et al.*, 2014) as well as updated versions of data-poor harvest strategies that have been used in other Australian federally-managed fisheries (Zhou *et al.*, 2011; Dowling *et al.*, 2008; Dowling *et al.*, 2016). These harvest strategies are used to determine recommended biological catches (RBCs), which are in turn used to set the total allowable catches (TACs) using the following SESSF meta-rule:

$$TAC_t = \begin{cases} 0.5 \cdot TAC_{t-1} & \text{if } RBC_t < 0.5 \cdot TAC_{t-1} \\ TAC_{t-1} & \text{if } 0.9 \cdot TAC_{t-1} \leq RBC_t \leq 1.1 \cdot TAC_{t-1} \\ 1.5 \cdot TAC_{t-1} & \text{if } RBC_t > 1.5 \cdot TAC_{t-1} \\ RBC & \text{otherwise} \end{cases}$$

These strategies (and resulting TACs) were implemented on an annual cycle (i.e. aggregate annual data were used in the assessments, as that is typical for most fisheries). The data for the assessments were generated using a sampling model, which generated catch length- and age-composition data; catch-per-unit-effort data (by vessel size-class and fishery sector); landings data (and catch species composition) by vessel size-class and fishery sector; and discard data. This sampling model allowed for ageing error, measurement error, variation in catchability, and error when measuring discards, with error levels that were stock-specific (Table S1). Data were generated for each 12-hour Atlantis time-step and aggregated to trip, month, and year. The same approach was applied to a survey design to generate fishery-independent survey data for the monitoring-based strategy discussed in the next section (and in Table 3).

As is the case in actual multispecies fisheries, harvest strategies were not applied to all fished species, but only to the 'treatment species' identified in Table S1; these species represent a range of life histories and have a range of influences on effort dynamics – including major target species (e.g. tiger flathead or blue grenadier), by-product species (e.g. blue warehou) and some bycatch species (e.g. gulper shark). The application of these harvest strategies was conducted in two ways. The first was to apply the same harvest strategy to all treatment species simultaneously (this was done for each harvest strategy in Table 2). This was assumed to be a pragmatic approach to achieve domain-wide multispecies management. A final multispecies scenario involved applying the mix of harvest strategies actually applied in the SESSF, as this is an indication of the kind of pragmatic compromises that are made in fisheries management (the strategy applied per species in this case is listed in Table S1). This scenario is referred to as the "Mixed" scenario in the results section.

The second approach to applying harvest strategies was more single-species focused. This involved applying each of the seven harvest strategies listed in Table 2 to each of the 14 treatment species (as identified in Table S1) individually with the TACs for all other species set to the 2005 level. This meant there were 98 (7x14) combinations run, with the focus on the dynamic management of a single target species with all other species TACs held at 2005 levels. This approach is illustrative of the kind of complexities that might arise around over/underfishing of some species in a multispecies fishery should the focus of management be constrained to a very limited set of species.

Differential management across jurisdictions

While no national boundary exists within the SESSF, there are multiple state boundaries within the ecosystem. There are differences in management actions implemented between state and federal jurisdictions in Australia, but these did not show the desired contrast in terms of types of management. Consequently, an artificial political jurisdictional boundary was drawn within the broader model domain and unconstrained fishing was allowed on one side of the border while fisheries management of specific forms was implemented and enforced on the other side of the border.

Three potential border locations (Figure 1) were used to examine the sensitivity of the results to the location of the border versus the spatial distribution of the ecosystem components (a simple east-west split as shown by border location 2 could be confounded with biogeographic splits in the system due to circulation patterns within the model domain). In addition, projections were undertaken first with the western/southern jurisdiction being the managed area and then another set of projections were undertaken when the eastern/northern jurisdiction was the managed area. The final results were averaged across all these simulations.

Table 3 summarises the management methods that were explored. Unless noted otherwise, those scenarios using a more limited form of management (e.g. only gear modifications or discard minimisation) are subsets of the integrated EBFM strategy.

Indicators

Fourteen indicators (Table 4) were used to assess the ecosystem-level performance of the management actions. Individually, the indicators selected reflected different aspects of ecosystem structure and function or different management and societal objectives for the ecosystem. The ecological indicators were selected based on proven reliability and clear understanding of expected responses to fishing pressure from previous indicator studies (e.g. Fulton *et al.*, 2005; Link 2005; Shin *et al.*, 2010, 2018). The potential social and economic indicators that could be derived from the model output were limited, but an effort was made to capture aspects of the system that are of importance to the fishers and the broader economy (as noted in Table 4). The correlation and redundancy amongst the indicators was checked using Pearson and Spearman correlations – using the R cor() function (R version 3.4.4).

The mean and simulation intervals of each indicator were calculated over the final 10 years of the projections for each management strategy. These indicator values were then normalised against the values for the unconstrained fishing scenario to give the final scores per indicator per strategy. The mean result per indicator was then used to rank the performance of each management strategy; a lower value rank represents a higher value for the indicator. These ranks were used to indicate the effectiveness of the various management styles and geographic extents for those ecosystem aspects.

An overall score per strategy was created based on the median score across all the indicator ranks. These median scores were then themselves ranked to give the final overall rank. These final ranks were calculated across all strategies, regardless of which Atlantis model was used. A principal components analysis - princomp in R (version 3.4.4) - was also run on the indicator scores for all strategies across both models to assess if there were natural groupings in the results.

Tables 3 and 5 list the model version used for each strategy to facilitate consideration of results for a specific Atlantis model version. The following comparisons are based on a single model:

a) the data-rich and data-poor multispecies strategies applied across the entire domain (scenarios A-I) – all using Atlantis-RCC
b) the data-rich and data-poor single-species strategies applied across the entire domain (Scenarios P-V) – all using Atlantis-RCC
c) all strategies (single-species and multispecies) applied across the entire domain – all using Atlantis-RCC
d) the strategies applied only to part of the domain – all using Atlantis-AMS

All scenarios have been compared to the unconstrained fishing scenario to facilitate comparison between scenarios run using the same model, but also to allow for consideration of results across models (i.e. to compare strategies applied across the entire and only part of the domain). The unconstrained fishing scenario run for Atlantis-AMS and Atlantis-RCC produce essentially the same results and so only Atlantis-RCC unconstrained fishing simulation outputs were used in the reported analysis (as noted in Table 3).

Results

The biomass trends in both Atlantis-AMS and Atlantis-RCC for the historical period were similar to each other and to those from formal stock assessments for the SESSF (Figure S2).

The correlations (Figure S3) showed that the forage fish, iconic species, habitat, total catch and employment indicators were not correlated with other indicators. All the rest of the indicators were significantly correlated, though the majority of these were recognizably linear and the correlation coefficients were not particularly strong (i.e. $0.5 \leq |r| \leq 0.75$). Biodiversity had the highest number of significant and strong correlations with other indicators – specifically, target species biomass, demersal:pelagic biomass, total value, foregone value and value per unit effort. Value per unit effort had strong correlations with not only biodiversity, but also foregone catch and value when using the Pearson test. Overall, however, there is sufficient differentiation between the indicators to retain the full set of indicators.

Rank order of performance

Overall, the relative ranks of the approaches (from best to poorest performing) are: (i) EBFM across the entire domain (A); (ii) multispecies management across the entire domain (B-I); (iii) single-species management across the entire domain (P-V); (iv) EBFM across part of the domain (J); (v) multispecies management across part of the domain (K-O); (vi) single-species management only in some jurisdictions (W-Z); and (vii) unconstrained fishing pressure (AA).

Integrated EBFM applied across the full domain (strategy A in Table 5) ranks first (best performer) across the majority of indicators – target species, iconic species, habitats, diversity and demersal:pelagic biomass, total value, value per unit effort and minimisation of foregone catch and value. This form of management has a much lower rank (10-16) for the biomass of forage fish, total catch and the size in the catch. At the other extreme, unconstrained fishing throughout the domain (strategy AA in Table 5) was the poorest performer – with a rank of 27 (worst possible) across all indicators except habitat (rank 25), forage fish biomass (where it had rank 1), total catch (rank 6) and employment (rank 7).

The ranks of the other management strategies are more mixed (Table 5). In general, the multispecies application of the harvest strategies across the entire domain (strategies B-I) perform well, typically outperforming both the application of harvest strategies to only a single target species (P-V) or to management methods only applied to part of the domain (J-O and W-Z). The major exceptions to this pattern are: (i) the habitat state is better (>20% greater area and rugosity) under specific management strategies (e.g. extensive spatial closures) even if only applied in some jurisdictions (N); and (ii) iconic species fare better (with population sizes more than 2-4x higher) under multispecies and EBFM management even if only applied in part of a system (A, J-O). In addition, when implementing the more qualitative strategies (G and H) across the entire domain for all species, personal wellbeing can be lower (more time at sea) and the levels of foregone catch and foregone value can be higher than when using more rigorous quantitative single species management for at least some species (P-R).

Single-species management (strategies P-V) did not consistently outperform multispecies or EBFM approaches (strategies A-O), though it occasionally scored well for individual indicators -

e.g. age structured assessments focusing on key target species that dictate fleet behaviour, such as tiger flathead in the SESSF, size in the catch, Pelagic:Demersal biomass ratio and personal wellbeing (with less time spent at sea). The performance of the management strategies based on data poor assessments (strategies e.g. F-H, T-V) is inferior to the data rich quantitative assessment methods (strategies B-E, P-S). The total catch indicator is less straightforward to interpret than the other indicators because high catch (and thus high rank) could result from either higher catches due to healthier stocks being managed sustainably or higher catches due to less sustainable fishing. Similarly, total value could be high due to a high volume of low-moderate value species or because of a smaller volume of high value product.

The rankings ignore the among-simulation variation in the values for the indicators. Consideration of this variation reveals considerable overlap in indicator values among many of the management strategies. Nevertheless, Figures 3-10 indicate the improved status of the indicators in a managed system versus a system exploited by unconstrained fishing, as outlined in the sections below. Only three indicators are not consistently higher in a managed system- the biomass of forage species (due to the increased abundance of their predators in managed systems), the total catch landed and employment.

Performance of alternative strategies applied across the entire domain

Looking first to the single -species strategies applied across the entire domain (P-V in Table 3), while the results for target species in Figure 3(a) do not reach the high levels of EBFM (the dark grey bar marked A) for any of the harvest strategies tested, they do typically exceed those of the unconstrained fishing (i.e. are >1) for the more quantitative (data rich) harvest strategies (P-S). The more qualitative (data poor) strategies T-V do not outperform unconstrained fishing in terms of the biomass of target species. It is clear from this that there is a direct benefit – to the treatment species and the other species caught and landed with them – of using quantitative harvest strategies. Implementing management strategies based on trigger points or catch composition (U and V) does not substantially increase values of indicators for the treatment species relative to the same indicators under unconstrained fishing (Figure 3a). However, all strategies can have positive benefits for by-product species (e.g. ‘dories and oreos’ or ‘shallow water piscivores’).

Relative performance among these single-species strategies is much less clear for the other ecological and catch indicators (Figure 3b-h). For example, there is little difference in the status of iconic species between the harvest strategies (P-V), with all of them leading to improved performance in comparison to unconstrained fishing (Figure 3c). The lack of a clear pattern for the indicators other than the target species biomasses is due to the variability associated with dynamics that were conditional on the identity of the treatment species, the strategies used and the fleet's response to the resulting management restrictions and quota availability. For instance, while forage fish biomasses are always lower than under unconstrained fishing (Figure 3b), even then nonlinear responses complicate the picture; e.g. the community composition when catch curves are used as the basis for management advice (Q) leads to forage fish biomass levels lower than those under EBFM (Figure 3b).

There was a clear benefit (in most cases) from using more quantitative strategies (P-S) in terms of resulting system dynamics, as expressed by indicators for (i) habitat, (ii) diversity and (iii) the average size of the animals in the catch when only one treatment species is managed using a harvest strategy (Figure 3d, f, g). There may also be some benefit for iconic species (e.g. mammals, seabirds and large sharks), although this improvement is marginal given the variation within strategies (the trend is clearer in Figure S4 where the results are plotted without the EBFM simulation results, so the management strategy results are not as compressed). This is also apparent for treatment species, though when pooling across species the variable nature of the stock status of the treatment species and how they fit into system dynamics more broadly means the simulation intervals in Figure 3a are quite broad. Nonetheless, for target species overall (and the treatment species in particular) the relative biomass is much higher with the most quantitative management strategies, while biomasses are lower (overlapping those under the unconstrained scenarios) when the more qualitative management strategies are used. The non-treatment species vary less among strategies due to the use of time-invariant TACs for those species (so any variation among management strategies is due to indirect ecosystem effects flowing from the treatment species).

The economic and social performance of the different strategies is less clear due to a high level of variability among simulations when implementing the more quantitative strategies (Figure 4). Total value landed was typically higher under the more qualitative strategies T-V (Figure 4a), due to the volume of catch while the improved quality of what product was being landed in the more quantitative strategies (P-S) is clear from the value per unit effort (VPUE) indicator (Figure 4d). The economic losses (opportunity costs) are lower for the quantitative strategies meaning their foregone value performance is stronger than for strategies T-V (Figure 4c). While there is not much to distinguish the strategies in terms of fisher wellbeing (except for strategy V, which leads to high levels of expended effort), there is a clear difference in terms of employment, with strategies U and V having effort levels quite similar to unconstrained fishing.

The benefits of using quantitative approaches were larger when the TACs for all target species were updated annually (Figure 5 vs Figure 3, and Figure 6 vs Figure 4) – i.e. when multispecies and EBFM management approaches were used (strategies B-I in Table 3). However, there were exceptions such as for the iconic species indicator, which differs little between the two cases (see strategies of the same colour in Figures 3c, 5c; Figures S4c, S5c). The benefits of implementing a management strategy are greatest for the target species when integrated age-structured assessments and management strategies (strategy B) are used (preferably for as many species as possible), followed by the other quantitative approaches (strategies C, D and E), then the more qualitative approaches (strategies F, G and H in Figure 5a; also compare strategies with the same colour in Figure 3a, Figure 5a). This performance improvement occurs not just for the fished species, which indicates that the management footprint extends beyond the target species and their direct predators or prey. This is also reflected in the total value (Figure 4a vs Figure 6a), which shows less difference between the strategies when management of all species is updated annually, as the improvement in the stock status compensates for any catch constraints due to management, also reducing the degree of variability between scenarios. Opportunity costs are smaller for the quantitative scenarios B-E, as catch foregone due to poor stock status is lower (meaning the performance of these strategies is much better for the foregone catch and value indicators). The value per unit effort is also much higher for scenarios B-E. The employment outcomes (Figure 6e) are much more mixed and variable, however, and depend on how the costs of access and management play out against profits.

The Mixed strategy (I in Table 5) involves applying the actual harvest strategies for each species, and this is reflected by its results, which are amongst those for the more quantitative strategies (of which it is made up), although at the lower end of that group of strategies (Figure 5). The aggregate performance of strategy I across all indicators together ranks it at around the same level as when catch curves or CPUE-based strategies are applied to all the treatment species simultaneously (Table 5). The performance of individual indicators for the Mixed strategy is variable (Figures 5 and 6), and indicator responses under strategy I do not match those of any one of the individual strategies that contribute to this aggregate strategy. Age-structured strategies are used for many species in the Mixed strategy, which explains the strong performance of this strategy for treatment species (Figure 5a) – and for the demersal:pelagic biomass ratio, size in catch (Figure 5e,g) and foregone catch and value indices (Figure 6b,c). The Mixed strategy also performs well for habitats and iconic species, but leads to a different system structure compared to when one strategy is applied to all treatment species. This is why the forage indicator (Figure 5b) is much higher than for the other quantitative strategies and the diversity (Figure 5e) and value per unit effort (Figure 6d) are lower.

In general, the improved ecological status of the quantitative strategies comes at the cost of lower landed catches (Figure 3h, 5h). However, total values (Figure 4a, 6a) are not as strongly differentiated (with values varying by <10-15% across the various options). This is because some of the highest value species benefit the most from the quantitative harvest strategies. The benefit of the investing in more holistic approaches to management is clear from the reduction in opportunity costs (improved foregone value score) under EBFM and when all target species are managed using quantitative strategies. These forms of management lead to a sufficient increase in production and stock status for improved, if constrained, catches over the longer term.

Some of the more qualitative management strategies lead to catches similar to, or higher than, those from unconstrained fishing while still achieving an ecological status that outperforms the unconstrained case. This is in part an artefact of the projection period because there is a declining trend in biomass for the more qualitative management strategies (reflected in their lower ecological performance in comparison to the more quantitative strategies), indicating that the simple management rules are insufficient at a system scale, but that declines are not as rapid as in the unconstrained state (e.g. Figure S6). A small number of much longer simulations indicated that, while these more qualitative tiers avoid the worst of the reductions in biomass of unconstrained fishing, they are insufficient to avoid the system entering an undesirable state where at least some of the main target species (e.g. pink ling, blue-eye trevalla) have dropped below the limit reference point of 20% of their unfished biomasses or failed to recover from past over exploitation (Figure S7).

Impact of managing only part of the system

Ecological status is typically better (higher biomass of target and iconic species, broader habitat extents and forage fish levels closer to those under EBFM) at the entire system level when some form of management is implemented than when catch is unconstrained in all regions; this is true even when management only occurs in a part of the system (Table 5 options W-Z; Figure 7). However, improvement is low (<50% increase in biomass beyond levels seen under system-wide unconstrained fishing) for target species. Improvements were also low for some other indicators under certain strategies. For example, there is very little improvement in the state of the habitat, or the abundance of iconic species when management relies solely on discard controls in one jurisdiction (strategy X; Figure 7c, d). Of the strategies only applied to one jurisdiction, the discards strategy (strategy X) also had amongst the highest abundances of forage fish, due to predation release (Figure 7b).

Of those strategies applied to only part of the entire domain, integrated EBFM within a single jurisdiction (strategy J) led the highest levels of target biomass, as well as relatively high mean levels of iconic species, diversity, average size of capture (Figures 7a, 7c, 7f, 7g), value per unit effort and wellbeing (Figure 8d, 8f). It can also minimise levels of foregone catch, improving long-term yield performance (Figure 8b). Spatial management (with extensive closures of 30% of the fishable area in one jurisdiction; strategy N) led to significant improvements in target biomass there, but that may be an artefact of the focus of this index in this ecosystem on demersal, less mobile species. Using ITQs as the only means of fisheries management in the managed portion of the system (strategy Z) does not necessarily lead to higher target species biomass at the system level, but ITQs are associated with lower variance in the outcomes than other management strategies that focus on fisheries targeting or technology – i.e. the simulation intervals for ITQs (strategy Z) in Figure 7a are much tighter than for the other management strategies. The use of ITQs also leads to some improvement in value per unit effort (Figure 8d).

With management limited to only part of the system, the abundance of the iconic species (mammals, seabirds and large sharks) was sensitive to the form of management used (Figure 7c); benefiting most from specific bans on interactions with them (strategy M), use of gear that minimised interactions with them (strategy W), management based on simple ecological indicators that included iconic species status directly into the decision making (strategy L) and integrated EBFM (strategy J). Habitats also showed clear benefits of management strategies that either simply avoided impacts on ecologically valuable habitats (via gear modifications, strategy W) or by recognising them in management processes (strategies L and N).

Management focused on multispecies yield (strategy K) not only leads to a higher catch overall (Figure 7h), especially in the managed part of the system (Figure 9h), but also to higher total value (Figure 8a) and the lowest levels of opportunity costs (and thus the strongest foregone value score; Figure 8c). The species mix is much broader when focusing on multispecies yield, leading to a lower overall average size in the catch (Figure 7g), with sizes in both regions of about the same level (Figure 9g), without resulting in a strong reduction in the typical medium- to large-sized target fish species in the managed region (Figure 9a). This strategy also leads to some of the highest wellbeing and value per unit effort scores of the strategies applied to only part of the domain (Figure 8).

When comparing the managed and unmanaged regions there is a clear improvement in terms of ecosystem structure (as captured by the ratio of demersal:pelagic biomass) and social and economic performance in the managed region (Figure 9e). While the extent of this benefit can be quite variable, integrated EBFM (strategy J) clearly outperformed other management methods (Figure 9e). Interestingly, it was the multispecies-focused management (strategy K) that had a markedly improved biodiversity in the managed versus unmanaged regions (Figure 9f). There was little difference between the outcomes under the other management strategies; and while there was a biodiversity benefit within the managed area (vs the unmanaged area) this was diluted at the whole of system level (thus the small effect size in Figure 7f). In contrast, there is little difference between forage fish levels in managed and unmanaged jurisdictions within the one ecosystem (Figure 9b), despite a clear ecosystem level signature of management in the forage fish indicator (Figure 7b). This is because the relatively high mobility of the forage fish groups, which move across large parts of the modelled domain. There were clear social and economic benefits to having some form of management in place – with all the social and economic indicators being higher in the managed area, except for employment levels, (Figure 10).

Multivariate Patterns

The principal components analysis clearly identifies the strategies applied across the entire domain from those applied in only part of the domain (Figure S8). Moreover, the multispecies strategies are located apart from the single-species strategies; and the quantitative strategies are separate from the more qualitative approaches. The mixed strategy (I) actually in use in the fishery (2014) clusters quite closely with the quantitative strategies. In contrast, there is no simple ordering to the strategies applied to only part of the domain (in this bi-plot the bulk of the single-species and multispecies strategies are co-located). The EBFM and unconstrained fishing scenarios are also separated from the rest of the strategies; between them bounding the space occupied by the other strategies. The multispecies-focused harvest strategy is particularly different; it does not locate with the other strategies applied only to part of the system because it has much higher total catches for less of an ecosystem footprint. However, its overall position is also quite distinct from the EBFM strategies.

In terms of what is structuring the principal components, the results lend weight to the correlation analysis, suggesting that indicators of iconic species, forage fish, employment, catch size, total value and total catch (and perhaps also the demersal:pelagic biomass ratio) may have been sufficient to characterise the relative performance of the different strategies.

Discussion

EBFM has been an internationally recommended approach to fisheries management for 15 years (FAO, 2003) and is being adopted in fisheries legislation by an increasing number of nations. Approaches such as the Ecological Risk Assessment of the Effects of Commercial Fishing (Hobday *et al.*, 2011), Integrated Ecosystem Assessments (Levin *et al.*, 2013; DePiper *et al.*, 2017) and the delivery of ecosystem status reports to fisheries management councils (as is done in the North Pacific; e.g. Zador and Yasumiishi, 2017; Slater *et al.*, 2017) all represent useful steps towards delivering EBFM. However, despite considerable advances, fisheries continue to face considerable challenges around operationalising EBFM and achieving its goals.

The failure of single-species management to account for feedbacks and trade-offs within fished systems has been used repeatedly as an argument for EBFM (Pikitch *et al.*, 2004; Leslie and McLeod, 2007; Marasco *et al.*, 2007; Möllmann *et* al., 2014; Fogarty, 2014). However, those familiar with the inertia and other realities of the decision-making processes associated with fisheries have questioned whether an ecosystem-based approach is any more politically robust than single-species management (Jennings, 2006; Rice, 2011). For the management authorities struggling under fisheries legislation calling for EBFM and a reduction in the number of overfished stocks (e.g. in USA, Europe and Australia), the first reaction has been to simply expand the number of stocks assessed to encompass all the major target species (e.g. Australia regularly assesses 94 stocks (Patterson *et al.*, 2017), Canada assesses 159 (ECCC, 2017); the USA periodically assesses up to 316 stocks (NOAA, 2017) and the European Union at least 50 (based on the number of reports listed per year in the ICES stock assessment repository; http://standardgraphs.ices.dk/stockList.aspx)) and to argue that this is a first step to EBFM. Realistically, EBFM cannot follow this path *ad infinitum*; the simple mental exercise of extrapolating single assessment decision processes (and expenses) to the hundreds of species that a mixed fishery, such as the trawl fishery of south eastern Australia, interacts with shows how expensive that approach would be in the extreme. Moreover, such an “ecosystem approach” would be open to many of the same flaws as single-species management, but at greater expense. However, as expanding the number of assessments has been the pattern in the developed world it would be beneficial to know what advantages it does convey. So the questions remain, societal and political complexities aside, i) what are the benefits of using more quantitative methods over data-poor methods that could be implemented more rapidly over broader sets of species at lower cost? ii) would moving to the formal assessment (and direct management) of more species lead to better system level outcomes, as a useful step toward EBFM? and iii) in cases where a country does not have sole control of an entire ecosystem, is the institutional and scientific effort associated with fisheries management worth it if the neighbouring jurisdiction is not doing likewise? While these seem to be fairly rudimentary, even obvious, questions to ask, there are few published examples addressing them.

The results presented here provide some model-based input into this discussion. The ranks in Table 5 indicate that, while expanding the number of annually assessed species and thereby adopting a more multispecies management form is not the same as fully fledged EBFM, it is a positive step in that direction. Well-enforced quantitative single-species management focused on a small number of species, implemented over the entire ecosystem domain, has substantial positive outcomes in terms of target species, habitats, iconic species, ecosystem structure, diversity, economic value and fisher wellbeing. This form of management can even out-perform less quantitative multispecies-oriented approaches applied across the same domain (e.g. strategies P-S outperform strategies F-H for several indicators, Table 5). Nevertheless, there are major benefits at the ecosystem level of using integrated rather than single-species oriented management. This confirms arguments in favour of EBFM (e.g. Pikitch *et al*., 2004; Hilborn, 2011; Fogarty, 2014; Möllmann *et al*., 2014). The result also aligns with earlier work by Fulton and Gorton (2014), who found that taking an integrated approach to the management of fisheries and aquaculture in southeastern Australia was necessary if the industry is to be as robust as possible to the worst effects of global change – both climate-driven shifts, but also expanding pressure from other uses of ocean and coastal zones. It is also evident that improvements in ecosystem outcomes may be made without sacrificing catches. It is already widely discussed in the literature that improved stock status leads to higher catches (Costello *et al*., 2016; Hilborn and Costello, 2018). The same principle applies at the ecosystem level.

It is possible to go further still and move to fisheries practices more oriented to deliver on sustainable multispecies yields (Garcia *et al.*, 2012; Jacobsen *et al*., 2014). While this is contentious (Burgess *et al*., 2016; Froese *et* al., 2016; Law *et al*., 2016; Pauly *et al.*, 2016), many of the fisheries in developing nations face the compound problem of: struggling with increasing populations and food insecurity (Blanchard *et al.,* 2017); relying on mixed fisheries that land hundreds of species spanning the highest through to the lowest trophic levels; and being data-poor with high levels of illegal or unreported fishing. The performance of the multispecies yield-oriented approach (strategy K in Figures 5 and 6) indicates that total catches can be much higher under this strategy without a notable decline in performance (compared to the other management strategies) for most of the other indicators. The mean values for the ecological indicators may have been lower (leading to poor rank in absolute terms), but the range of possible values overlapped those of the other multispecies strategies. Simultaneously many of the social and economic scores were much improved on the other strategies trialled. While the final multivariate result was located apart from EBFM (Figure S8), the ability to deliver to society without causing the level of degradation seen under unconstrained fishing indicates that it deserves further attention in those nations struggling to deal with complex fisheries and food security issues. Farcas and Rossberg (2016) also found that strategies focused on multispecies harvest sustainably yielded more than single-species-oriented controls, due to improved ecosystem state.

Discussion of the objectives across all interested parties and relevant legislative directives will be a key step in implementing EBFM. As we have not undertaken such a discussion for this study we have chosen not to weight the individual indicators here, instead reporting on them with equal weight. Such an approach may not be appropriate in individual systems. For instance, some groups may up-weight environmental status, while others may prefer social and economic outcomes; still others may look for consistency in performance across indicators. In the latter instance, care will be needed to distinguish between strategies that do moderately well across all indicator categories (e.g. managing based on ecological indicators or quantitative single-species strategies versus those that are simply universally poor, such as unconstrained management of fairly qualitative approaches). Importantly, for those who chose to embrace integrated management this will mean acknowledging that it may involve some strong trade-offs – for instance, between system structure and function and employment (Table 5).

There are lessons to be learnt around the kinds of assessment tools employed even without such radical changes in fisheries and management approaches. Fulton *et al.* (2016) and Dichmont *et al.* (2017) have considered the implications of data-rich versus data-poor management strategies (and assessment methods) in terms of the risk to the resource and the catch-cost-risk trade-off. The results presented here consider the ecosystem aspects of that discussion. Fortunately, moving to the ecosystem perspective has not overly complicated the general conclusions. As discussed in Fulton *et al.* (2016) and Dichmont *et al*. (2017) – and shown here in Figures 3 and 5 – individual stock status is lower (and thus risk is higher) when data-poor methods are used. This is not simply because fewer data are available, but also because of biases in the assessments and slow response times to unexpected declines in resource status (Dichmont *et al.*, 2017). Importantly, the same pattern extends beyond the species directly assessed to other species caught in the fisheries (i.e. "non-treatment" target species) and to iconic species, habitats, system structure and diversity. Use of data-poor methods also has implications for economic and social outcomes – the absolute catch and value landed may have the potential to be high (with fewer constraints in place), but this comes at the cost of lower value per unit effort, higher opportunity costs and poorer outcomes for individual wellbeing. In contrast, effective data-rich single species management can deliver towards ecosystem outcomes; although, the magnitude of delivery is far greater when more species are actively managed (quantitatively assessed with relative short assessment intervals). The biomass of fished species was 45-120% higher when all major target species were managed using harvest strategies. Notably, such multispecies management also saw improved annual returns (with value per unit effort increasing by > 40%), lower opportunity costs, 20-30% higher aggregate landings (i.e. lower levels of foregone catch) and even higher employment levels, as the improved stock status saw more vessels remain in the fishery long term.

It is critical to understand the strengths and weaknesses of any method used, whether data-rich or data-poor. As discussed in Dichmont *et al.* (2017), the performance of catch curves in this modelled system was mixed and they were not always as precautionary as CPUE-based methods. This translated into performance that was sensitive to the life history of the managed species and a greater sensitivity to the history of depletion of a stock. In turn, stock status influenced performance in terms of the broader fish community and in combination with technological interactions and fleet responses to quota allocations could affect other indicators. Ultimately however, the differences at the system level amongst the more quantitative methods were less than the declines in performance as increasingly qualitative methods were employed. This does not completely invalidate the use of such data-poor methods, but would argue for their use to be constrained to systems that are only lightly fished (and so with little residual risk) – noting that many of these data-poor methods were never intended for use in fisheries receiving as much directed pressure as simulated here (Dowling *et al.,* 2008, 2013).

Results were sometimes complex across the indicators, where there was no simple pattern, but rather results could be non-linear and conditional on the identity of the treatment species and how fishers responded to the management strategy in place. This complexity further reinforces (a) that a suite of indicators is required to track overall structure and function of the socioecological system (Fulton *et al.,* 2005; Rice and Rochet, 2005); and (b) that the nature of EBFM will differ among locations and will likely also need to evolve through time as conditions (and even expectations) change (Shannon *et al.,* 2014; Trochta *et al.*, 2018).

One of the important steps in transitioning to EBFM is to define ecosystem-relevant reference points and control rules for non-target ecosystem components. The form of these rules has been the subject of much discussion, but one of the clearest statements on the topic was made by Link (2005), who identified "warning" and "limit" reference points for a number of ecological indicators including the biomass of specific functional groups (gelatinous, forage, target, habitat and iconic species), the slope of the biomass size spectrum, diversity indices and total fisheries removals (amongst others). Link's rules were defined based on empirical observations from the Georges Bank-Gulf of Maine ecosystem and were applied unmodified in the "simple ecological indicators based" strategy applied in this study. Despite only being applied in part of the domain (and not being modified to best suit the ecosystem of interest) these rules performed remarkably well. They delivered some of the best scores across the board for iconic species and habitat status. In terms of strategies only applied to a single jurisdiction, the ecological indicator-based strategy (L) was one of the highest-ranked strategies and was second only to full integrated management (EBFM) in terms of the target species stock status (clearly outperforming the single species management strategies applied over the same domain). The indicator-based strategy did not perform as well for some of the size and diversity indices, but other work has shown that indicator performance is system dependent and so rules really need to be tailored to the system in question (Shannon *et al.*, 2014; Shin *et al.*, 2018). Consequently, it is very likely that overall performance of this approach would be even better once tailored to the SESSF, likely mitigating the strong catch constraints imposed under this strategy (which had quite strong impacts on its economic performance). Nevertheless, the ecological performance of this strategy in the simulations provides strong support for further exploration of this approach, as it has the potential to progress fisheries science and management by implementing ecosystem relevant control rules for a suite of relatively straightforward ecological indicators.

Fisheries management, EBFM or single-species focused, that is constrained to only part of an ecosystem is not as effective as when it is implemented over the entire ecosystem, but is still much better than if fishing is unconstrained (both in terms of the overall state of the ecosystem and the status of groups within the managed portion of the ecosystem). Naturally, the more of an ecosystem that can be managed the better the outcomes. For example, Figure 11 shows that when managing only part of an ecosystem the best performance for the demersal:pelagic biomass (a proxy for ecosystem structure) is seen when management is applied to 50% or more of the ecosystem's area. Moreover, it is due to the loss in performance of managing less than 50% of a system that saw single-species approaches applied across an entire ecosystem outrank more ecosystem-oriented approaches limited to just part of the system for many indicators (Table 5).

Nevertheless, management that conserves stocks and improves habitats and other ecosystem components on one side of the boundary subsidises the neighbouring jurisdiction. For example, highly mobile species – such as large pelagics – will move between jurisdictions, but this is insufficient to undermine management altogether. While movement between jurisdictions also occurs for the iconic species (mammals, seabirds and large sharks) and the unmanaged jurisdiction does benefit from the efforts of the other jurisdiction, the status of iconic species is sensitive to the forms of management used, with quite strong differences in indicator values and variability across the various management strategies. In some instances, the pressures in the unmanaged jurisdiction cannot be compensated for by management applied in the other jurisdiction, and the overall status of the iconic species declines towards the case under unconstrained fishing (e.g. when the managed jurisdiction relies solely on spatial management, discard controls or catch quotas). The variability in particular was because of the confounding effects of mobility and feeding behaviours. Increased prey fields were of direct benefit, but this was diluted by the ability of (some) iconic species to move or switch prey if there were insufficient local resources.

For managers concerned with the status of iconic species and habitats who do not have control over entire ecosystems, Figures 7 and 8 indicate that there are clear benefits of either simply avoiding impacts on those species and habitats (via gear modifications or spatial zoning) or by recognising them in management processes (e.g. via including information on their status in the decision-making process via the inclusion of appropriate ecological indicators). Given that 59% of all the large marine ecosystems and all the high seas FAO areas are under shared management, and there are already concerns over transboundary species (e.g. Thornton *et al.*, 2017), these kinds of understandings will be important for managers located on one side or another of a jurisdictional divide. This will be particularly important given that it is likely that there will be jurisdictional differences in terms of food security (Blanchard *et al.*, 2017), trade policy (Watson *et al.*, 2017), research capacity (as captured by UNESCO statistics on the Researchers in R&D per million people; https://data.worldbank.org/indicator/SP.POP.SCIE.RD.P6?view=map), societal valuation of conservation (Schultz *et al.*, 2005; Balmford *et al.*, 2009; Snyman, 2012; do Paço *et al.*, 2013), etc. Such differences may well even lead to tension or open conflict (McClanahan *et al.*, 2015). Consequently, understanding what is possible given the constraints in place will be important. Nonetheless, rather than abandon action, the results presented here suggest that some forms of management are effective even when only applied to part of a system.

It is important to recognise that this is a modelling study. We have endeavoured to (i) include the kinds of processes and data imperfections that real world assessment, management strategies and fisheries management agencies face (including inappropriate incentives, market distortions, information gaps and enforcement issues that can lead to divergence between the intent and outcome of specific management actions); and (ii) address some aspects of system uncertainty by including multiple parameterisations, where possible. Ultimately, however, this is but one modelled system and one where the social and economic aspects of the model were conditioned on a system where food security, poverty and a lack of livelihood alternatives are not crippling concerns, and thus not explicitly considered in the model. Moreover, while the treatment species for the assessments span the majority of those assessed in the main fishery in the region (the SESSF) they are not exhaustive, as they do not include herbivores, short-lived or sedentary invertebrates, or forage fish. The individual species level results are consistent with results from single-species MSE testing of the data-rich management strategies (e.g., Wayte and Klaer, 2010; Fay *et al.*, 2011; Little *et al.*, 2011; Klaer *et al.*, 2012). Nonetheless, confidence in these results would be much greater if repeated using other modelling frameworks, more socioecological systems of different types (so not just different ecosystem structure, but systems with alternative cultural expectations, demographics, livelihood make up etc), or if complemented with observational datasets.

Conclusions

It has been a decade since Murawski (2007) discussed the ten myths of an ecosystem approach to management. The efforts since then have confirmed that the means of operationalising EBFM have remained vague as has the exact nature of the science needed in support of it. However, the approach continues to evolve regardless, as Murawski (2007) said it would. Part of that evolution is concluding the discussion that there is actual benefit in management that is well enforced and actively conserves stocks and maintains viable ecosystem structure and function. The results presented here indicate that, while management may appear costly, it has real benefits far beyond the immediate target species, and that where possible the effort should be put into science and management, even if all jurisdictions are not cooperating. Shifting management to larger geographic or ecological proportions of the ecosystem and supporting application of data-rich harvest strategies clearly improved outcomes in terms of improved system state.

Authors' Contributions

EAF performed the simulation experiments with the technical support of RG. The overall concept and plans for the work were co-developed by all authors. All authors also contributed to the writing and revision of the paper.

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Tables

Table 1: Species groups included in the models. Species in bold have multiple size-at-age growth morphs in Atlantis-RCC (all groups only have one growth morph in Atlantis-AMS). The species marked with an asterisk have multiple stocks. Seabirds and baleen whales migrate outside the model domain and return annually.

Model Component	Group Composition
Pelagic invertebrates	
Large phytoplankton	Diatoms
Small phytoplankton	Picophytoplankton
Small zooplankton	Heterotrophic flagellates
Mesozooplankton	Copepods
Large zooplankton	Krill and chaetognaths
Gelatinous zooplankton	Salps (pryosomes), coelenterates
Pelagic bacteria	Pelagic attached and free-living bacteria
Squid	*Sepioteuthis australis* (Loliginidae), *Notodarus gouldi* (Ommastrephidae)
Benthic invertebrates	
Sediment bacteria	Aerobic and anaerobic bacteria
Carnivorous infauna	Polychaetes
Deposit feeders	Holothurians, echinoderms, burrowing bivalves
Deep water filter feeders	Sponges, corals, crinoids, bivalves
Shallow water filter feeders	Mussels, oysters, sponges, corals
Scallops	*Pecten fumatus* (Pectinidae)

Model Component	Group Composition
Herbivorous grazers	Urchins, *Haliotis laevigata* (Haliotidae), *Haliotis rubra* (Haliotidae), gastropods
Deep water megazoobenthos	Crustacea, asteroids, molluscs
Shallow water megazoobenthos	Stomatopods, octopus, seastar, gastropod, and non-commercial crustaceans
Rock lobster	*Jasus edwardsii* (Palinuridae), *Jasus verreauxi* (Palinuridae)
Meiobenthos	Meiobenthos
Macroalgae	Kelp
Seagrass	Seagrass
Prawns	*Haliporoides sibogae* (Solenoceridae)
Giant crab	*Pseudocarcinus gigas* (Menippidae)
Fin-fish	
Small pelagics*	*Sardinops* (Clupeidae), sprat, *Engraulis* (Clupeidae)
Redbait	*Emmelichthys nitidus*, Emmelichthyidae
Mackerel*	*Trachurus declivis* (Carangidae), *Scomber australisicus* (Scombridae)
Migratory mesopelagics	Myctophids
Non-migratory mesopelagics	Sternophychids, cyclothene (lightfish)
School whiting*	*Sillago* (Sillaginidae)
Shallow water piscivores	*Arripis* (Arripidae), *Thyrsites atu* (Gempylidae), *Seriola* (Carangidae), leatherjackets
Blue warehou*	*Seriolella brama* (Centrolophidae)
Spotted warehou	*Seriolella punctata* (Centrolophidae)
Tuna and billfish*	*Thunnus* (Scombridae), *Makaira* (Istiophoridae), *Tetrapturus* (Istiophoridae), *Xiphias* (Xiphiidae)
Gemfish*	*Rexea solandri* (Gempylidae)
Shallow water demersal fish*	Flounder, *Pagrus auratus* (Sparidae), Labridae, *Chelidonichthys kumu* (Triglidae), *Pterygotrigla* (Triglidae), *Sillaginoides punctate* (Sillaginidae), *Zeus faber* (Zeidae)
Flathead*	*Neoplatycephalus richardsoni* (Platycephalidae), *Platycephalus* (Platycephalidae)
Redfish*	*Centroberyx* (Berycidae)
Morwong*	*Nemadactylus* (Latridae)
Pink ling*	*Genypterus blacodes* (Ophidiidae)
Blue grenadier	*Macruronus novaezelandiae* (Merlucciidae)
Blue-eye trevalla	*Hyperoglyphe Antarctica* (Centrolophidae)
Ribaldo	*Mora moro* (Moridae)
Orange roughy*	*Hoplostethus atlanticus* (Trachichthyidae)
Dories and oreos*	Oreosomatidae, Macrouridae, *Zenopsis* (Zeidae)
Cardinalfish	Epigonidae

Model Component	Group Composition
Sharks	
Gummy shark*	*Mustelus antarcticus* (Triakidae)
School shark*	*Galeorhinus galeus* (Triakidae)
Demersal sharks	*Heterodontus portusjacksoni* (Heterodontidae), Scyliorhinidae, Orectolobidae
Pelagic sharks	*Prionace glauca* (Carcharhinidae), *Isurus oxyrunchus* (Lamnidae), *Carcharodon carcharias* (Lamnidae), *Carcharhinus* (Carcharhinidae)
Dogfish	Squalidae
Gulper sharks	*Centrophorus* (Centrophoridae)
Skates and rays	Rajidae, Dasyatidae
Top predators	
Seabirds	Albatross (Diomedeidae), shearwater (Procellariidae), gulls and terns (Laridae), gannets)Sulidae)
Seals	*Arctocephalus pusillus doriferus* (Otariidae), *Arctocephalus forsteri* (Otariidae)
Sea lion	*Neophoca cinereal* (Otariidae)
Dolphins	Delphinidae
Orcas	*Orcinus orca* (Delphinidae)
Baleen whales	*Megaptera novaeangliae* (Balaenopteridae), *Balaenoptera* (Balaenopteridae), *Eubalaena australis* (Balaenidae)

Table 2: Summary of the harvest strategies (assessment methods and decision rules) used with the Atlantis-RCC model. The continuum of quantitative to semi-quantitative (more qualitative methods) is also shown – the dashed line marks the division between those methods considered quantitative and those considered semi-quantitative.

Assessment method	Decision rule	Type
Integrated quantitative age-structured population model used to estimate biomass (B)	B_{20}:B_{35}:B_{48} “broken stick” strategy. Vessel level catch and effort data are aggregated based on the gear used (this allows for fleet-specific parameterisation of selectivity).	Quantitative
Catch curves used to estimate current fishing mortality, F (F_{CUR})	Broken stick-like strategy used to calculate F_{RBC} (see Wayte and Klaer 2010) and the final recommended catch is given by:	

	$$RBC = max\left(\frac{1-e^{-F_{RBC}}}{1-e^{-F_{CUR}}}, 3\right) C_{CUR}$$ where C_{CUR} is current catch.	
CPUE-based	Recommended catch is given by: $$RBC = C_T max\left(\frac{\overline{CPUE} - CPUE_L}{CPUE_T - CPUE_L}, 0\right)$$ where C_T is the catch target, $CPUE_L$ is the limit CPUE, $\overline{CPUE}$ is the average CPUE over the most recent four years and $CPUE_T$ is the target CPUE (average over the period 1996-2005 by default, but set to the more conservative 1986-1996 period for a subset of species as described in Dichmont *et al.*, 2017).	
F estimated from lengths	F_{CUR} based on observed average length in catch versus expected lengths – as a function of fishing mortality from a yield-per-recruit calculation (Haddon *et al.*, 2015). This F_{CUR} is then used in Tier 3 harvest strategy.	
F estimated from the fishery footprint	F_{CUR} is calculated as: $$F_i = \frac{q_i^h \cdot q_i^{\lambda} \cdot (1 - S_i) \cdot \sum_t a_{t,i} \cdot E_t}{A_i}$$ where q^h is the overlap of the species distribution and the fisheries' spatial footprint, q^{λ} is the size- and behaviour-dependent gear selectivity, S is the discard survival rate, a_t is the area covered in time step t, E_t is the effort applied in time step t and A_i is the area the species occupies. F_i is compared with a reference F (as defined in Zhou *et al*., 2011) to give the final RBC.	
Trigger based on catch versus Historical Maximum Catch	Current catch (C_{CUR}) is compared with the historical maximum catch (HMC). If C_{CUR} < 50% HMC then the fishery continues without restriction, otherwise restrictions (e.g. closure if C_{CUR} > 200% HMC) and more quantitative assessments are triggered (F from lengths if < HMC, otherwise a catch curve based assessment is triggered).	
Trigger based on catch composition	Catch composition, individual and aggregate catch, CPUE and the area fished are all compared to historical conditions. As for the other trigger-based method, limited change does not trigger a response, but a moderate change triggers a footprint-based assessment and larger changes trigger a catch curve based assessment.	Semi-quantitative (more qualitative)

Table 3: Summary of the scenarios. Where the jurisdiction is marked as "single", the management method is applied in one jurisdiction only (the other jurisdiction has unconstrained fishing) but the data used in the management strategy is drawn from both jurisdictions. IDs have been assigned to each strategy for each geographic extent to assist in reporting the results.

ID	Model	Management Strategy (Scenario)	Details of implementation	Jurisdiction
		Multispecies & EBFM management, entire domain		
A	RCC	Integrated management (EBFM)	Integrated EBFM (as defined in Fulton *et al.*, 2014) – includes a mix of ITQs, limited entry, gear controls, spatial management.	All
B	RCC	All treatment species – age structured assessments	TACs are applied to all treatment species (listed in Table S1) are all are calculated annually using an integrated age- structured population model and the associated decision rule as outlined in Table 2. This TAC is allocated as quota to individual vessels (with the allocation based on the proportion of TAC owned in the previous year). This quota may be traded among vessels. The effort allocation model attempts to avoid species where no TAC is available (avoidance is not always possible due to the multispecies nature of the fishery).	All
C	RCC	All treatment species – catch curves	As for scenario B, but with assessment using a catch curve (and the associated decision rule) as defined in Table 2.	All
D	RCC	All treatment species – CPUE based rule	As for scenario B, but with assessment using a CPUE-based assessment method (and the associated decision rule) as defined in Table 2.	All
E	RCC	All treatment species – F estimated from lengths	As for scenario B, but with assessment using F estimated from lengths (and the associated decision rule) as defined in Table 2.	All
F	RCC	All treatment species – F from fishery footprint	As for scenario B, but with assessment using F estimated from the fishery footprint (and the associated decision rule) as defined in Table 2.	All
G	RCC	All treatment species – Hist. max catch trigger based	As for scenario B, but with assessment using catch triggers versus Historical Maximum Catch (and the associated decision rule) as defined in Table 2.	All
H	RCC	All treatment species – catch composition based	As for scenario B, but with assessment using catch composition trigger (and the associated	All

ID	Model	Management Strategy (Scenario)	Details of implementation	Jurisdiction
			decision rule) as defined in Table 2.	
I	RCC	All treatment species – mixed strategies	As for scenario B, but with a combination of the assessments and decision rules that reflects the set of strategies used in reality in the SESSF (the rules used per species is given in Table S1).	All
		Multispecies & EBFM management, part of domain		
J	AMS	Integrated management (EBFM)	Integrated EBFM (as defined in Fulton *et al.*, 2014) – includes a mix of ITQs, limited entry, gear controls, spatial management.	Single
K	AMS	Multispecies yield-focused management	The take of all fished species is in proportion to productivity (within the constraints imposed by the existing mix of gears and their selectivities); implemented through differential effort levels across different fleet sectors. The realised effort levels result from the TACs set for key species (listed in Table S1), which are calculated annually using age-structured population models fitted to fishery-dependent and fishery-independent data with the acceptable fishing mortality rates and biomass reference points set in proportion to productivity. For ease of implementation, species bans are implemented for some gear types so that keeping F at the acceptable levels is easier (either because it reduces the number of gears or interactions to consider or because a companion species caught by that gear would be over-exploited as a result of allowing this gear target the species if interest).	Single
L	AMS	Simple ecological indicators based	TACs for main target species (listed in Table S1) are calculated annually using survey-based ecological indicators vs historical baselines – including the relative biomass of gelatinous, forage, target, habitat and iconic species; biomass ratios for demersal:pelagic and planktivore:piscivores; mean fish length; slope of biomass size spectrum; Reyni diversity index; total removals; and large fish indicator (reference points for these indicators are as defined in Link 2005).	Single
M	AMS	Forage and iconic species catch ban	2005 TACs in place for all species, but the landing of all forage fish and iconic species (large sharks, pinnipeds, cetaceans, sea birds) are banned - i.e. any caught must be	Single

ID	Model	Management Strategy (Scenario)	Details of implementation	Jurisdiction
			discarded. Bycatch reduction devices are used to minimise interactions with these groups and the effort allocation model attempts to avoid these species (i.e. penalises locations where such species had been caught previously in the simulation).	
N	AMS	Spatial management	30% closure of all habitat types (shelf, slope, deep ocean). 2005 quotas in place for all species, but with no fishing in the closed areas (100% compliance assumed).	Single
O	AMS	High levels of monitoring informing management	Monitoring (spatially and temporally) informs quota setting (including for non-target & conservation species). TACs for key species (listed in Table S1) are calculated annually using age-structured population models based on fishery-dependent and -independent data.	Single
			Single species management, entire domain	
P	RCC	Single treatment species – Age structured assessments	As for scenario B, but with the harvest strategy and decision rule only applied for a single treatment species at a time (instead of all at once). 2005 TACs in place for all other species.	All
Q	RCC	Single treatment species – catch curves	As for scenario C, but with the harvest strategy and decision rule only applied for a single treatment species at a time (instead of all at once). 2005 TACs in place for all other species.	All
R	RCC	Single treatment species – CPUE based rule	As for scenario D, but with the harvest strategy and decision rule only applied for a single treatment species at a time (instead of all at once). 2005 TACs in place for all other species.	All
S	RCC	Single treatment species – F estimated from lengths	As for scenario E, but with the harvest strategy and decision rule only applied for a single treatment species at a time (instead of all at once). 2005 TACs in place for all other species.	All
T	RCC	Single treatment species – F from fishery footprint	As for scenario F, but with the harvest strategy and decision rule only applied for a single treatment species at a time (instead of all at once). 2005 TACs in place for all other species.	All
U	RCC	Single treatment species – Hist. max catch trigger based	As for scenario G, but with the harvest strategy and decision rule only applied for a single treatment species at a time (instead of all at once). 2005 TACs in place for all other species.	All
V	RCC	Single treatment species – catch composition based	As for scenario H, but with the harvest strategy and decision rule only applied for a single treatment species at a time (instead of all at once). 2005 TACs in place for all other species.	All
			Single species management, part of domain	
W	AMS	Gear modification	2005 TACs in place for all species with gear restrictions (i.e. larger mesh sizes) and bycatch	Single

ID	Model	Management Strategy (Scenario)	Details of implementation	Jurisdiction
			reduction devices to minimise interactions with bycatch, habitats and iconic species. Effort allocation model also attempts to avoid bycatch or incidental catch of all species (i.e. penalises locations where these had been caught previously in the simulation).	
X	AMS	Discard controls	2005 TACs in place for all species, with spatial effort allocation conditioned on economic incentives (penalties) on discards so that discards are minimised (by avoidance, shifting gears to minimise interactions etc.) – see Hutton *et al.*, (2010) for further details on the incentives.	Single
Y	AMS	Catch quotas	TACs for major target species (listed in Table S1) are calculated annually using an age-structured population model. All vessels begin fishing January 1 of the calendar year and landing of the species continues until the TAC for that species is full. After that point all further catch for the species must be discarded. The effort allocation model attempts to avoid species where no TAC is available (avoidance is not always possible due to the multispecies nature of the fishery).	Single
Z	AMS	Individual transferable quotas	TAC species list and calculations as for scenario Y, but the quotas can be traded.	Single
AA	RCC*	*Unconstrained fishing*	No restrictions on fishing (as described in the main text).	All

* The results obtained for this scenario using Atlantis-AMS are very similar to those for Atlantis-RCC and do not lead to any change in the pattern of results reported here.

Table 4: List of indicators used to summarise the ecosystem-level outcomes of the simulations. All have been structured so that a high value is typically desirable (non-linearities can complicate matters – see the discussion of forage fish index in the main text). Values are reported relative to the value of the indicator in the unconstrained fishing simulations.

Indicator	Definition	Notes
Target species biomass	Relative biomass averaged across all species (treatment and non-treatment) targeted by fishing	Measure of the direct effect on the fished ecosystem. As the fishing pressure considered in the model was sufficient to deplete species to around the target reference point of the harvest strategies (and beyond), a higher value (i.e. one close to the target reference point or a little higher) was considered desirable.
Forage fish biomass	Biomass summed across sardines, anchovy and mackerel	Prey biomass field; incidentally in combination gives some insight into size-composition of the ecosystem. Avoiding depletion of prey fields is a driving motivation of calls for precautionary reference points for forage fish (Smith *et al.*, 2011), consequently a higher value was judged to be desirable for this indicator (noting that it can also be high under predator

		release, but such a situation should be flagged by other indicators in the suite).
Iconic species abundance	Relative biomass averaged across marine mammals, seabirds and large sharks	Species of conservation concern; vulnerable (slow growing) species; species that synthesise dynamics over large spatiotemporal scales. Higher scores for this indicator conveys that the system structure has not been distorted by the removal of these vulnerable species (Fulton *et al*., 2005, Link 2005).
Habitat coverage	Proportional cover by habitat forming species groups (e.g. seagrass, algae, filter feeders)	Health of habitat in the ecosystem. Higher values for this indicator shows that habitats relied upon by other species (e.g. as nursery habitat or refugia) are in good condition (Fulton *et al*., 2005, Link 2005).
Demersal:Pelagic biomass	Ratio of the total biomass of demersal:pelagic fish species groups	Provides an index of structure of the ecosystem and typically decreases under intensive fishing pressure or disturbance (Caddy 2000, Fulton *et al*., 2005)
Q90 Diversity index	The Q-90 diversity statistic (Ainsworth and Pitcher 2006):	Index of biodiversity. Expected to decrease under intense fishing pressure (Link 2005); higher levels considered more desirable by

	$$Q90 = \frac{0.8 \cdot S}{log\left({R_2}/{R_1}\right)}$$ where only those groups > 10% of their unfished values are included in the calculation (as recommended for ecosystem models by Ainsworth and Pitcher (2006); *S* is the total number of functional groups included in the calculation; R_1 and R_2 are the representative biomass values of the 10^{th} and 90^{th} percentiles in the cumulative abundance distribution across the functional groups	implication in legislation.
Size in catch	Average size of individuals in the catch across all fisheries sectors.	Initially an index of footprint of fishery, but ultimately can also reflect stock and system structure (Rochet and Trenkel 2003). Also indicates relative value (although market-dependent, while small finfish are typically worth less than larger, invertebrates can be high value).
Total catch	Total landed catch summed over all fisheries sectors and fished species	Food security index (in simple terms, more is better than less).
Total value	Total value of landed catch summed over all fisheries sectors and fished species	Gross economic contribution index (considered desirable in terms of total contribution to the broader economy).

Foregone catch index	$$L_c = \frac{1}{\sum_y \sum_f \sum_s (C_{b,s,f,y} - C_{s,f,y})\, e^{-\delta y}}$$ where $C_{b,s,f,y}$ is the landed catch of species *s* under theoretical "optimal" management[1] in fishery *f* in year *y*; $C_{s,f,y}$ is the landed catch under the harvest strategy; and δ is the is the economic discount rate (0.05). Note that discounted catches are used given each species started from a different biomass relative to $0.4B_0$ (the assumed target reference biomass level).	Index of loss of food provision. Minimising losses (i.e. high value for this index) is widely stated as desirable – as evidenced by the FAO's global initiative on food loss and waste reduction (FAO 2015). Achieving this goal minimises the denominator so will maximise this index.
Foregone value index	$$L_V = \frac{1}{\sum_y \sum_f \sum_s p_s (C_{b,s,f,y} - C_{s,f,y})\, e^{-\delta y}}$$ where the terms are as for foregone catch and p_s is the price of species *s* (held constant through time).	Index of economic losses. Minimising opportunity costs (i.e. high value for this index) is a fundamental economic principle. Achieving this goal minimises the denominator so will maximise this index.
Value per unit effort	Average over fisheries of (value of catch / effort expended)	Profitability index. Maximising profits is another fundamental economic principle.
Employment	Total number of crew members employed across vessels in all fisheries	Typical indicator assumed to be an index of social value (wellbeing) – with the inherent assumption that more is better as productive employment is correlated with poverty reduction, and other positive outcomes such as access to services, social inclusion etc

		(Fischer 2014).
Personal social wellbeing index	$$S = \frac{1}{\sum_f E_f}$$ Where E_f is the effort in fleet f.	Index of minimisation of time away from family[2] and exposure to at-sea risks (the higher the score, the less time away). Achieving this goal minimises the denominator so will maximise this index.

1. The theoretical "optimal" catch here was given by a "bang-bang" harvest strategy as described in Dichmont *et al.*, (2017) – to summarise: using perfect knowledge of the fished stocks, biomass above the target level is removed via the following protocol: targeted fishing of a species is eliminated for N_1 years if $B < 0.48B_0$, while large catches are allowed for N2 years if $B > 0.48B_0$. N_1 and N_2 were selected iteratively for each species as analytical determination was not possible due to the use of the dynamic effort allocation model (which allowed for implementation error and incidental catches of species under moratorium).
2. We appreciate that some fishers prefer to be at sea and do not perceive a "loss" from being at sea.

Table 5: Rank of the performance of each management strategy for each indicator. The strategies have been grouped based on their focus (ecosystem/multispecies versus single species) and geographic extent (full vs partial domain coverage). IDs and management strategies are as defined in Table 3. The Atlantis operating model (OM) used in each instance is given for reference.

ID	**Model**	**Management Strategy (Scenario)**	Target species	Forage fish	Iconic species	Habitat	Demersal:Pelagic B	Diversity	Size in catch	Total catch	Total value	Foregone catch	Foregone value	Value per unit effort	Employment	Wellbeing index	Overall rank
		Multispecies & EBFM management, entire domain															
A	RCC[1]	Integrated management (EBFM)	1	16	1	1	1	1	10	16	1	1	1	1	20	2	1
B	RCC	All target species – age structured assessments	2	5	12	7	16	7	2	10	10	2	2	3	23	4	2
C	RCC	All target species – catch curves	6	24	14	9	2	4	4	10	11	5	6	4	13	12	4
D	RCC	All target species – CPUE based rule	4	24	16	8	8	3	6	2	3	3	4	2	16	5	3
E	RCC	All target species – F estimated from lengths	3	24	18	12	6	5	4	13	13	7	7	5	6	14	6
F	RCC	All target species – F from fishery footprint	5	24	23	9	5	6	3	4	5	16	11	11	5	17	8
G	RCC	All target species – Hist. max catch trigger based	7	2	24	18	13	2	15	5	4	22	15	8	8	16	9

H	RCC	All target species – catch composition based	9	2	26	22	15	9	16	2	9	24	18	9	4	18	13
I	RCC	All target species – mixed strategies	8	2	19	9	9	8	7	12	12	4	3	12	21	13	5
		Multispecies & EBFM management, part of domain															
J	AMS	Integrated management (EBFM)	14	20	4	6	17	18	12	20	22	8	23	6	18	1	12
K	AMS	Multispecies yield-focused management	21	23	6	26	20	13	26	1	2	6	9	10	17	3	11
L	AMS	Simple ecological indicators based	15	11	3	3	18	23	22	26	25	14	25	25	2	19	19
M	AMS	Forage and iconic species catch ban	18	13	5	27	22	26	20	19	18	26	22	22	3	20	25
N	AMS	Spatial management	16	13	9	2	23	21	19	18	21	23	21	26	12	24	21
O	AMS	High levels of monitoring informing management	17	12	7	21	23	22	18	17	20	13	20	23	19	23	24
		Single species management, entire domain															
P	RCC	Treatment species – Age structured assessments	10	9	13	12	3	10	1	22	16	12	5	7	25	7	7
Q	RCC	Treatment species – catch curves	12	19	17	15	11	12	17	25	17	17	10	13	24	8	15
R	RCC	Treatment species – CPUE based rule	11	17	14	14	4	11	11	21	15	15	14	16	27	6	10
S	RCC	Treatment species – F	13	21	19	17	10	14	9	23	14	19	8	15	26	10	16

		estimated from lengths															
T	RCC	Treatment species – F from fishery footprint	23	22	22	19	6	15	8	7	7	21	16	20	22	8	18
U	RCC	Treatment species – Hist. max catch trigger based	25	8	19	20	12	16	13	9	8	11	13	14	9	10	14
V	RCC	Treatment species – catch composition based	26	5	25	22	14	17	14	8	6	9	12	19	10	15	17
		Single species management, part of domain															
W	AMS	Gear modification	20	10	2	4	25	23	25	15	19	18	19	21	15	21	20
X	AMS	Discard controls	24	7	11	24	26	20	23	14	23	25	17	24	11	25	26
Y	AMS	Catch quotas	19	15	8	16	21	25	21	24	24	10	24	18	1	26	22
Z	AMS	Individual transferable quotas	22	18	10	5	19	19	24	27	25	20	26	17	14	22	23
AA	RCC[1]	*Unconstrained fishing*	27	1	27	25	27	27	27	6	27	27	27	27	7	27	27

1. The same rankings were obtained if the Atlantis-AMS model was used instead of the Atlantis-RCC model for these management strategies.

Figure Captions

Figure 1: Map of the model domain showing the polygonal box structure used in the model and the jurisdictional boundary locations used in the second set of simulations (white dashed lines).

Figure 2: Schematic of how the simulations were implemented.

Figure 3: Relative value of ecological and fisheries indicators (compared to the case with unconstrained fishing pressure) for the simulations where individual species were managed using one of the assessment methods/harvest strategies, while the rest of the system was held at 2005 TAC levels. The codes from Table 3 are used to identify the strategies, with full names of these strategies are also given in the key (e.g. strategy P uses age structured assessments). In the first panel – 'target species' – there is a triplet for each tier: the left most symbol for each triplet (solid lines) are the overall results; the middle symbol (large dashed lines) indicate treatment species (species listed in Table S1); and the rightmost symbol of each triplet (short dashes) are the non-treatment target species (all other fished species). The light grey bar with black dashed central line indicates the levels for unconstrained fishing (AA in Table 3); the dark grey line marked with an A indicates the level under EBFM across the entire domain (A in Table 3). The vertical line between scenarios S and T demarcates quantitative from more qualitative harvest strategies.

Figure 4: As for Figure 3, but for the relative value of economic and social indicators.

Figure 5: Relative value of ecological and fisheries indicators (compared to the case with unconstrained fishing pressure) for the simulations where all target treatment species were managed simultaneously using one of the assessment methods/harvest strategies. The codes from Table 3 are used to identify the strategies, with full names also given in the key (e.g. strategy P uses age structured assessments). The light grey bar with black dashed central line indicates the levels for unconstrained fishing (AA); the dark grey line marked with an A indicates the level under EBFM across the entire domain (A in Table 3). The vertical line between scenarios E and F demarcates quantitative from more qualitative harvest strategies, another line separates the "mixed strategy" I from the rest.

Figure 6: As for Figure 5, but for the relative value of economic and social indicators.

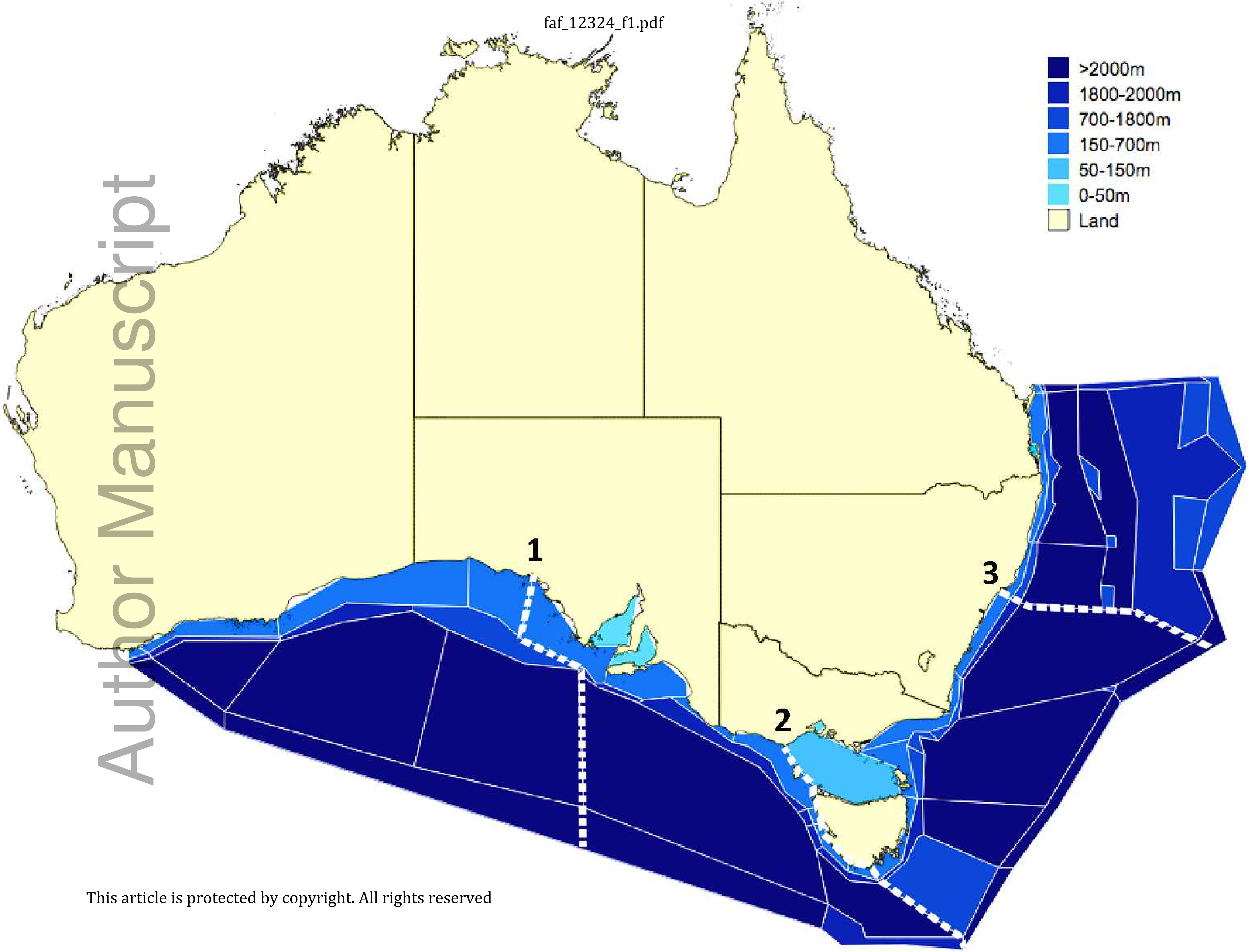
Figure 7: Relative value of ecological and fisheries indicators (compared to the case with unconstrained fishing pressure) calculated at the overall ecosystem level for the simulations where management rules were only applied to one half of the model domain (with fishing unconstrained in the other half). The codes from Table 3 are used to identify the strategies, with full names also given in the key (e.g. strategy J is Integrated management (EBFM)). The light grey bar with black dashed central line indicates the levels for unconstrained fishing (AA); the dark grey line marked with an A indicates the level under EBFM across the entire domain (A in Table 3). The vertical line between scenarios O and W demarcates EBFM/multispecies management strategies from single species strategies.

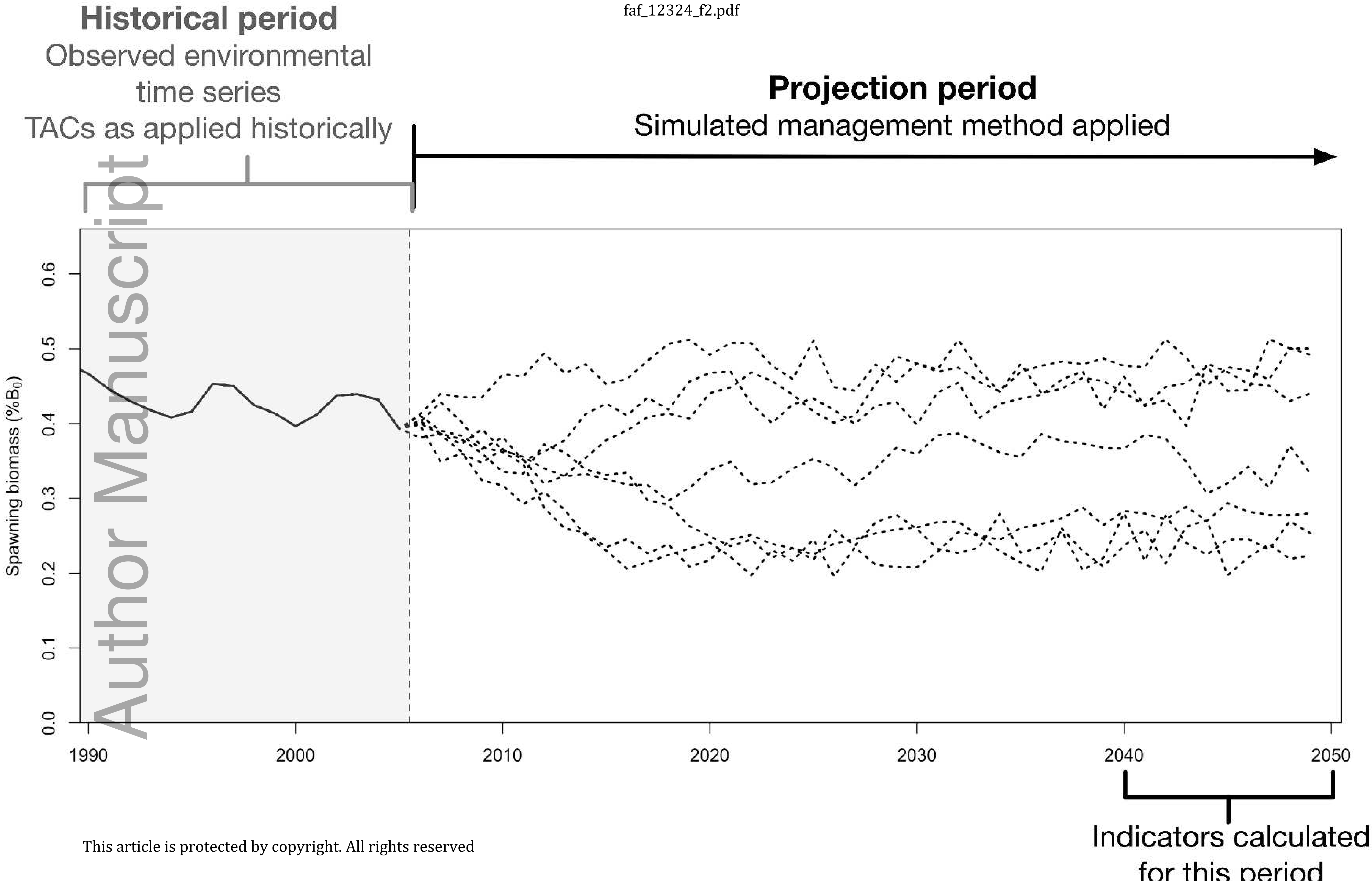
Figure 8: As for Figure 7, but for the relative value of economic and social indicators.

Figure 9: Relative value of ecological and fisheries indicators for the managed half of the model domain in comparison to the values in the region of the model with unconstrained fishing. The codes from Table 3 are used to identify the strategies, with full names also given in the key (e.g. strategy J is Integrated management (EBFM)). The light grey bar with dashed central line indicates the levels for the part of the domain with unconstrained fishing (AA in Table 3). The vertical line between scenarios O and W demarcates EBFM/multispecies management strategies from single species strategies.

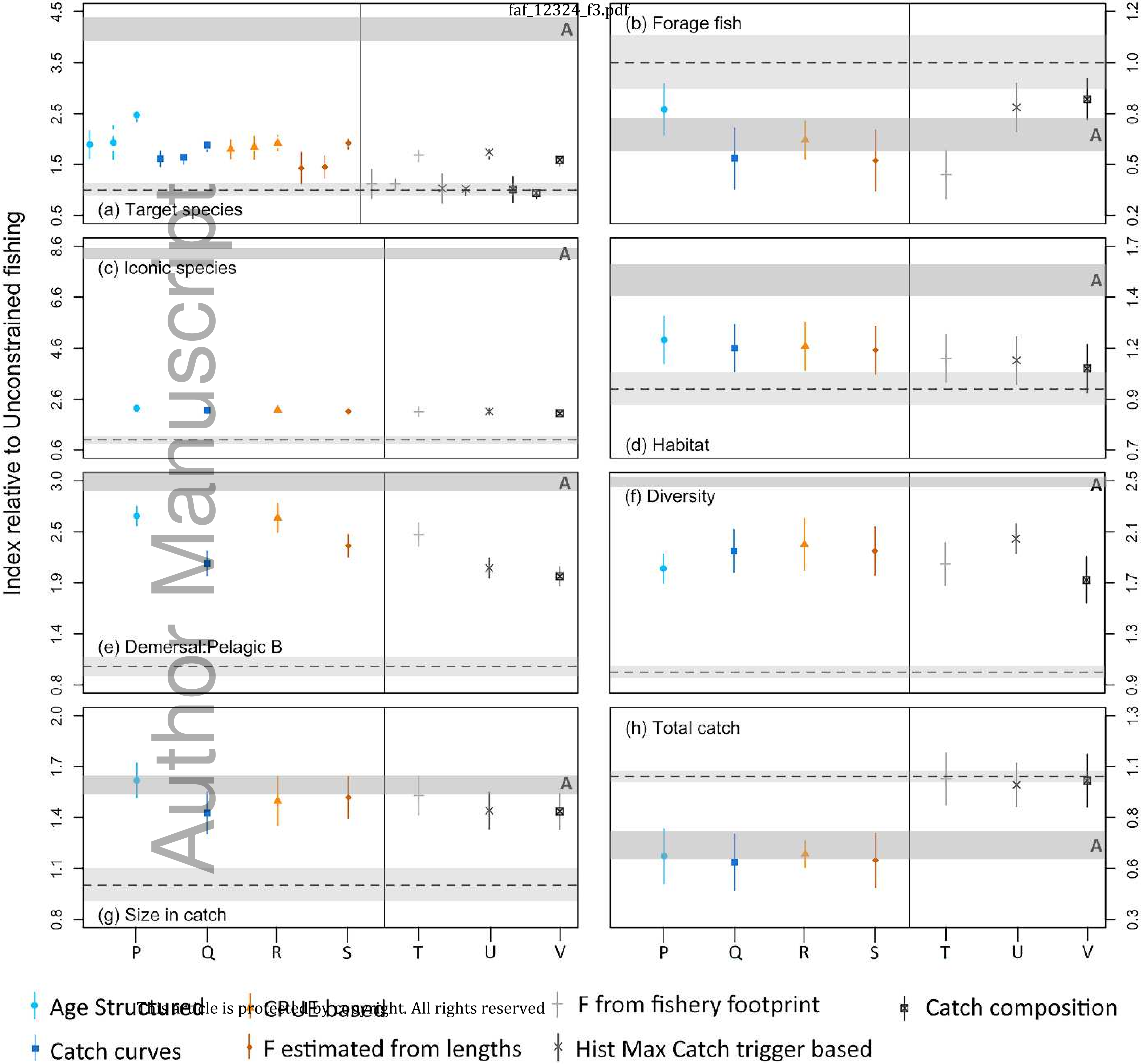
Figure 10: As for Figure 9, but for the relative value of economic and social indicators.

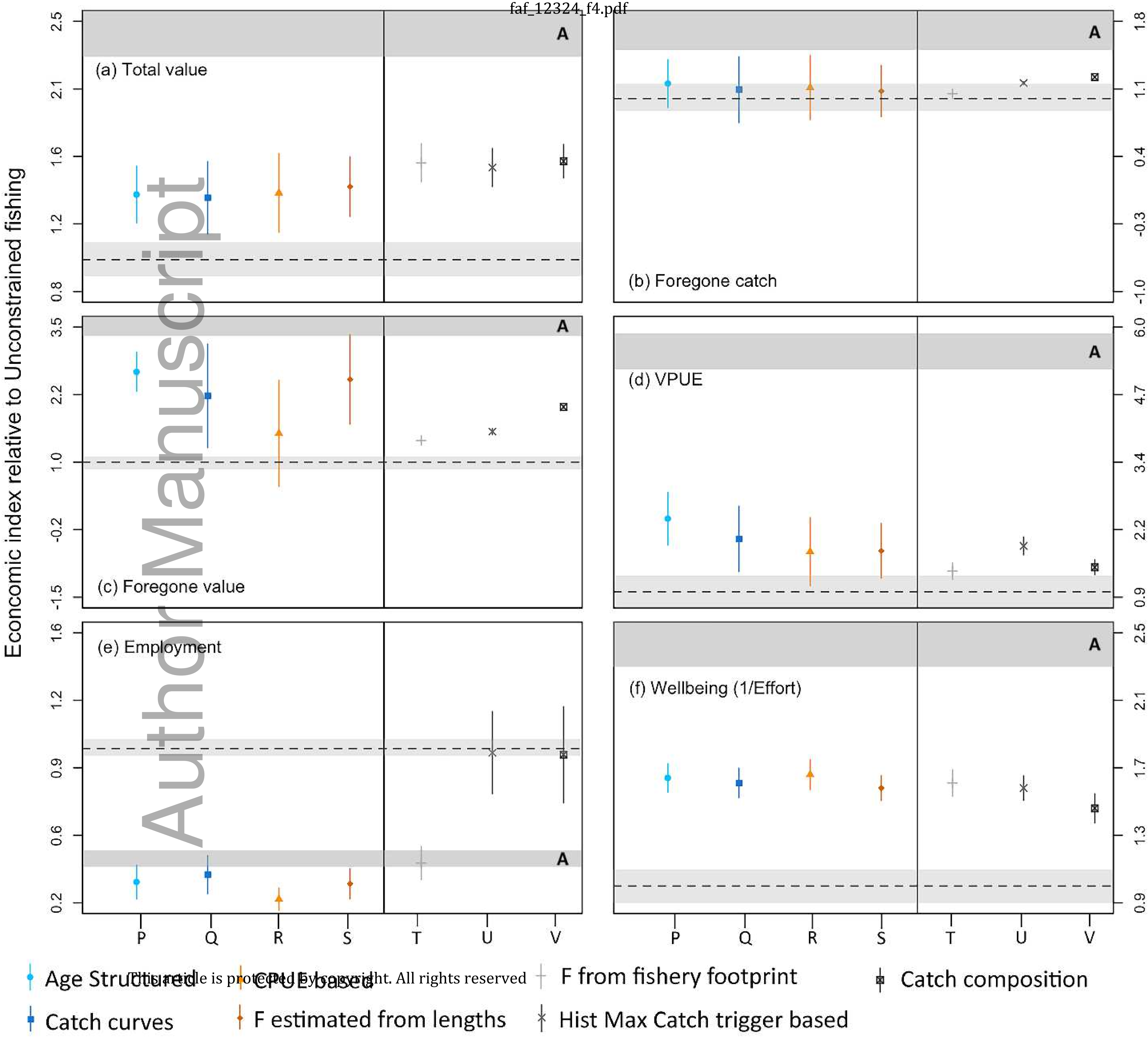
Figure 11: Example explanation (using the overall demersal:pelagic biomass ratio) of the contribution of the different geographic jurisdictional arrangements to the overall results per indicator.

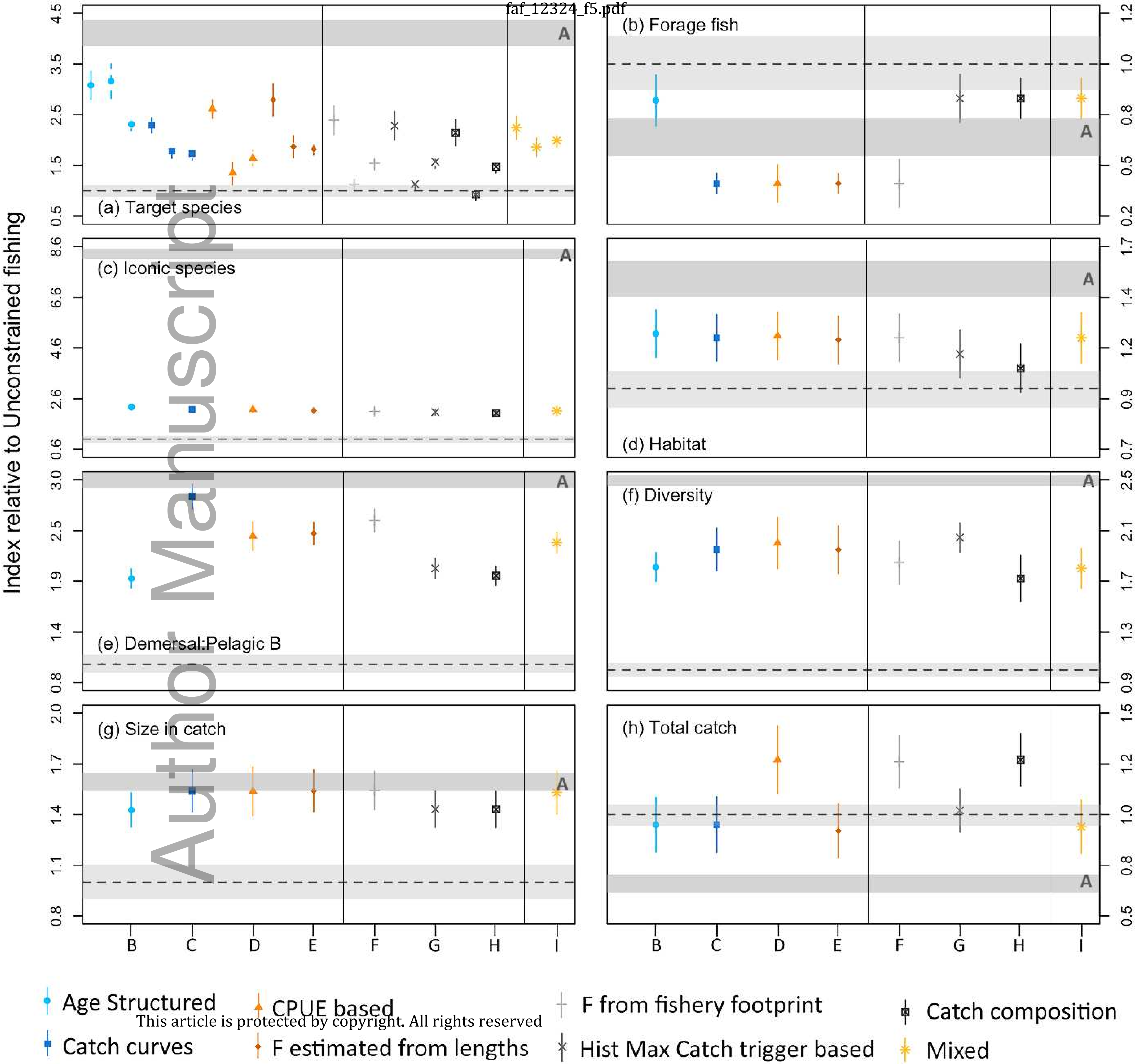

>2000m
1800-2000m
700-1800m
150-700m
50-150m
0-50m
Land
1
2
3



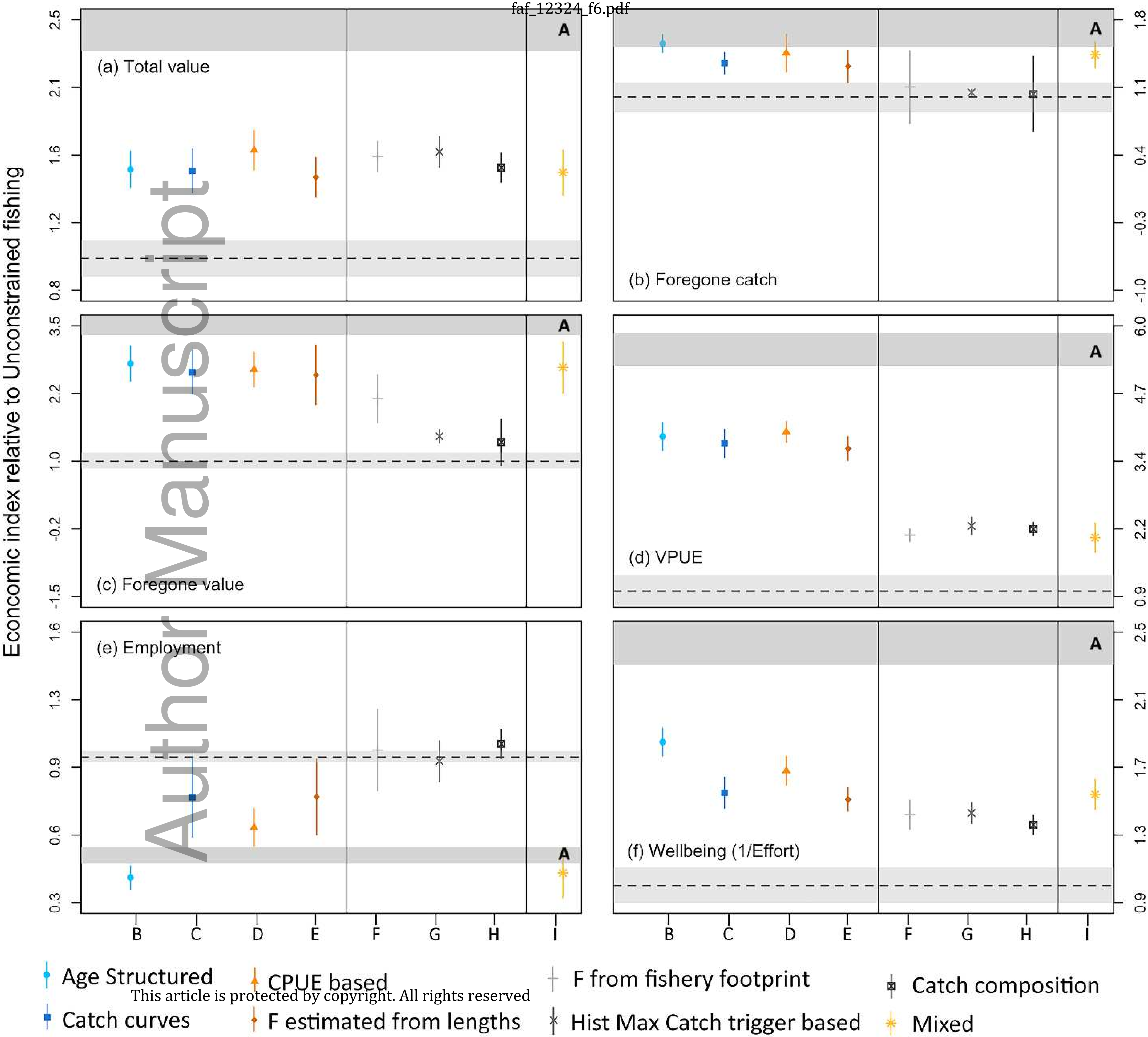
Historical period
Observed environmental time series
TACs as applied historically
Projection period
Simulated management method applied
Spawning biomass (%B_0)
0.0
0.1
0.2
0.3
0.4
0.5
0.6
1990
2000
2010
2020
2030
2040
2050
Indicators calculated for this period

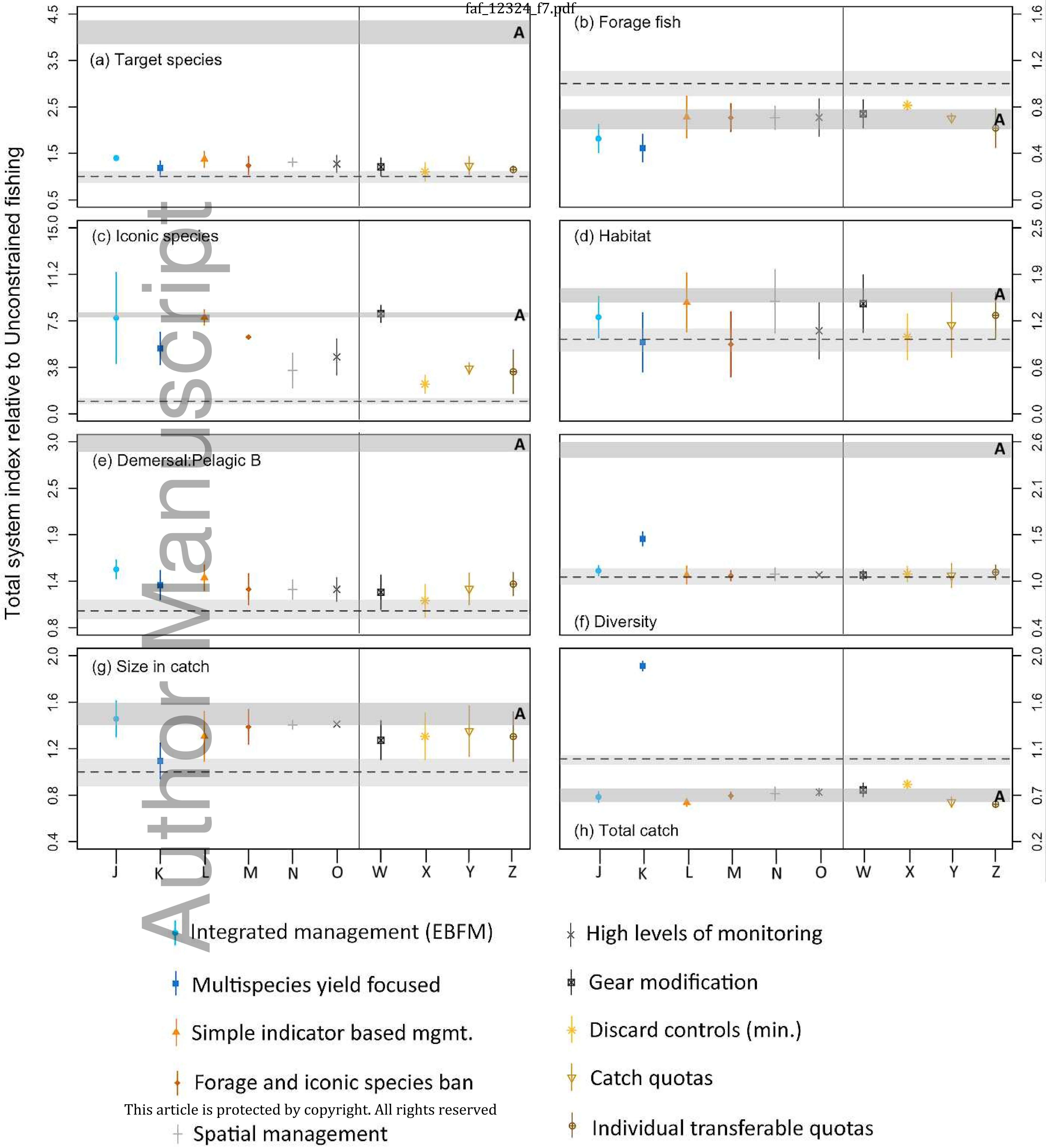

Index relative to Unconstrained fishing
(a) Target species
(b) Forage fish
(c) Iconic species
(d) Habitat
(e) Demersal:Pelagic B
(f) Diversity
(g) Size in catch
(h) Total catch
A
P
Q
R
S
T
U
V
Age Structured
CPUE based
F from fishery footprint
Catch composition
Catch curves
F estimated from lengths
Hist Max Catch trigger based

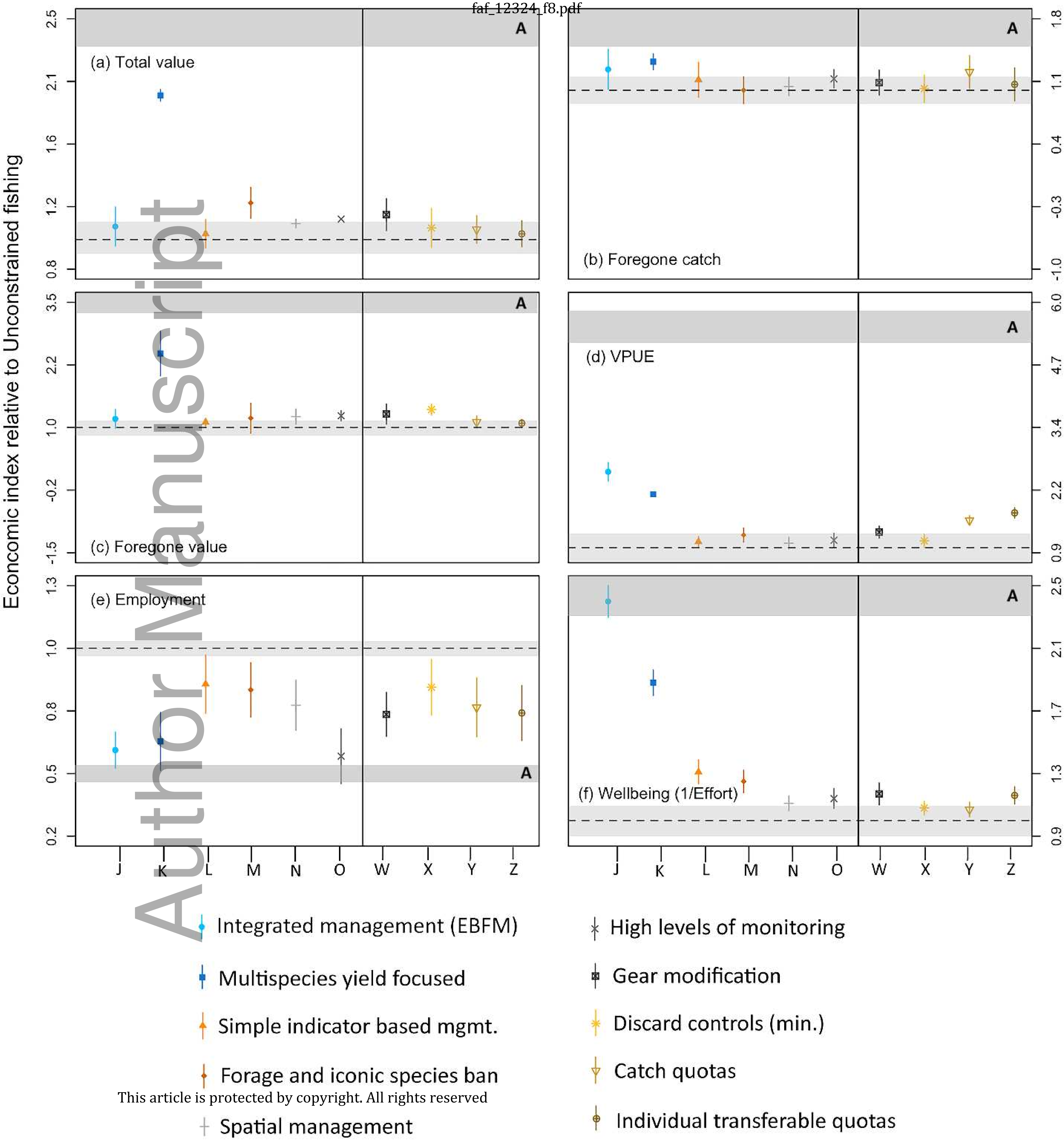

Econcomic index relative to Unconstrained fishing
(a) Total value
(b) Foregone catch
(c) Foregone value
(d) VPUE
(e) Employment
(f) Wellbeing (1/Effort)
A
P
Q
R
S
T
U
V
Age Structured
CPUE based
F from fishery footprint
Catch composition
Catch curves
F estimated from lengths
Hist Max Catch trigger based

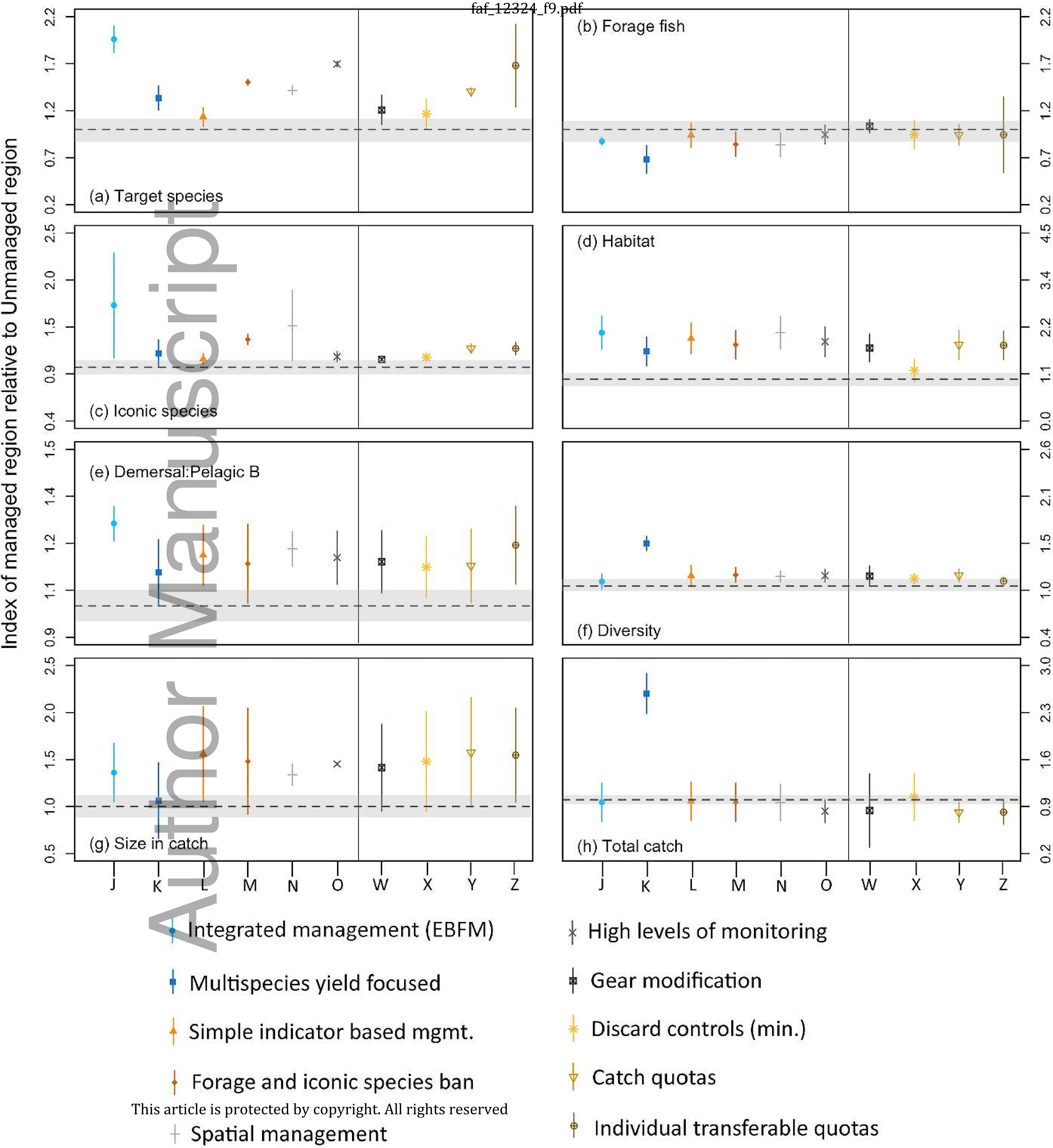


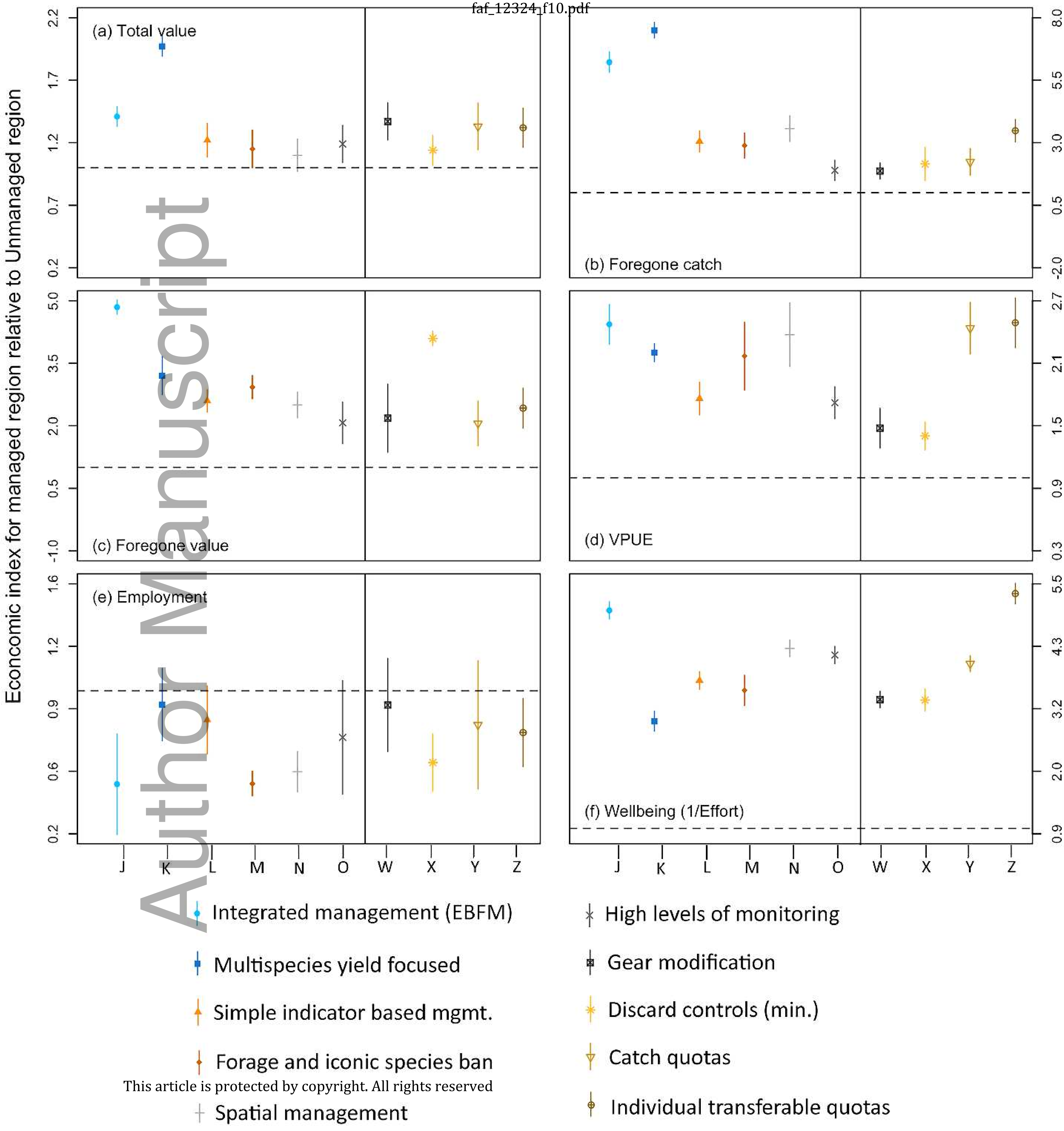
Index relative to Unconstrained fishing
(a) Target species
(b) Forage fish
(c) Iconic species
(d) Habitat
(e) Demersal:Pelagic B
(f) Diversity
(g) Size in catch
(h) Total catch
A
B
C
D
E
F
G
H
I
Age Structured
Catch curves
CPUE based
F estimated from lengths
F from fishery footprint
Hist Max Catch trigger based
Catch composition
Mixed


Economic index relative to Unconstrained fishing
(a) Total value
(b) Foregone catch
(c) Foregone value
(d) VPUE
(e) Employment
(f) Wellbeing (1/Effort)
A
B
C
D
E
F
G
H
I
Age Structured
Catch curves
CPUE based
F estimated from lengths
F from fishery footprint
Hist Max Catch trigger based
Catch composition
Mixed

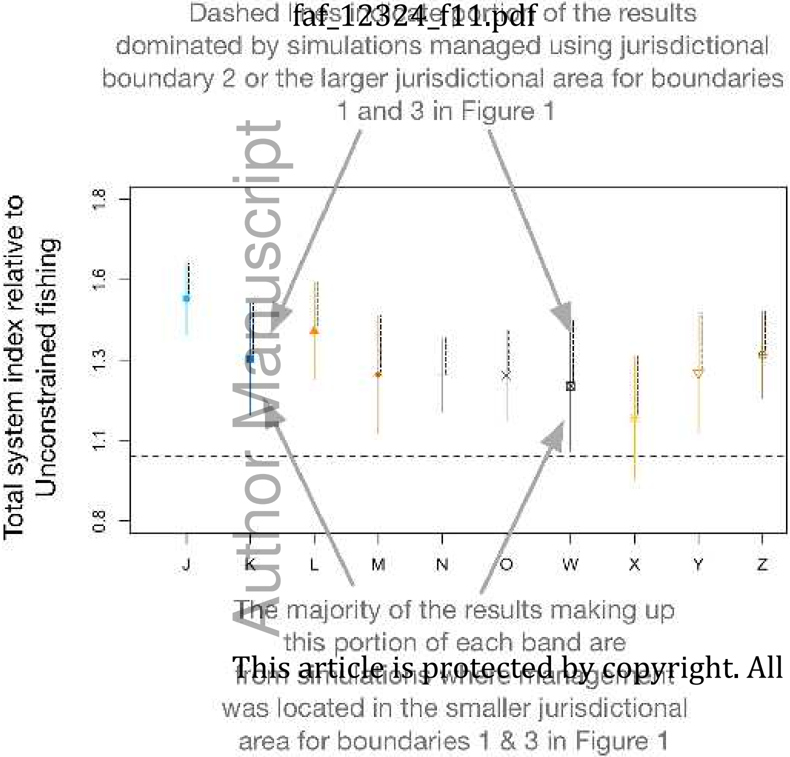

Total system index relative to Unconstrained fishing
(a) Target species
(b) Forage fish
(c) Iconic species
(d) Habitat
(e) Demersal:Pelagic B
(f) Diversity
(g) Size in catch
(h) Total catch
A
J K L M N O W X Y Z
Integrated management (EBFM)
Multispecies yield focused
Simple indicator based mgmt.
Forage and iconic species ban
Spatial management
High levels of monitoring
Gear modification
Discard controls (min.)
Catch quotas
Individual transferable quotas


Economic index relative to Unconstrained fishing
(a) Total value
(b) Foregone catch
(c) Foregone value
(d) VPUE
(e) Employment
(f) Wellbeing (1/Effort)
A
J
K
L
M
N
O
W
X
Y
Z
Integrated management (EBFM)
Multispecies yield focused
Simple indicator based mgmt.
Forage and iconic species ban
Spatial management
High levels of monitoring
Gear modification
Discard controls (min.)
Catch quotas
Individual transferable quotas


Index of managed region relative to Unmanaged region
(a) Target species
(b) Forage fish
(c) Iconic species
(d) Habitat
(e) Demersal:Pelagic B
(f) Diversity
(g) Size in catch
(h) Total catch
J
K
L
M
N
O
W
X
Y
Z
Integrated management (EBFM)
Multispecies yield focused
Simple indicator based mgmt.
Forage and iconic species ban
Spatial management
High levels of monitoring
Gear modification
Discard controls (min.)
Catch quotas
Individual transferable quotas



Econcomic index for managed region relative to Unmanaged region
(a) Total value
(b) Foregone catch
(c) Foregone value
(d) VPUE
(e) Employment
(f) Wellbeing (1/Effort)
J
K
L
M
N
O
W
X
Y
Z
Integrated management (EBFM)
Multispecies yield focused
Simple indicator based mgmt.
Forage and iconic species ban
Spatial management
High levels of monitoring
Gear modification
Discard controls (min.)
Catch quotas
Individual transferable quotas


of the results
dominated by simulations managed using jurisdictional
boundary 2 or the larger jurisdictional area for boundaries
1 and 3 in Figure 1
Total system index relative to
Unconstrained fishing
1.8
1.6
1.3
1.1
0.8
J
K
L
M
N
O
W
X
Y
Z
The majority of the results making up
this portion of each band are
was located in the smaller jurisdictional
area for boundaries 1 & 3 in Figure 1