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# Testing systemic fishing responses with ecosystem indicators

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ABSTRACT

Successful implementation of Ecosystem Based Fisheries Management (EBFM) requires practical methods of translating information on system status into management actions. Threshold values in ecosystem indicators have been demonstrated to provide insight for characterizing change points in marine ecosystems and suggested as reference points for EBFM. We used a guild based multispecies simulation model of the Georges Bank finfish community to quantify tradeoffs and changes among values for proposed ecological indicators given alternative fishing scenarios, and tested the performance of indicator-based approaches for setting system ceilings on annual catches.

Values for ecosystem indicators were sensitive to the exploitation rates on guilds, with total biomass of the community being most sensitive to groundfish exploitation rate. Setting ceilings on system-wide annual catches was successful in constraining values for indicators and revealed levels of system catch associated with indicator change. Community composition indicators showed catch thresholds lower than provided by the total biomass indicator. Ceilings based on community composition indicators more frequently resulted in higher yields and fewer species being overfished than when ceilings were set using total biomass or when no ceiling was in place.

Simulations demonstrated that threshold values in ecosystem indicators could be used to determine reference points in an EBFM context. The broad ranges for threshold values obtained demonstrates the sensitivity of such methods to exploitation history, underscoring the need to both incorporate expert knowledge and relate reference point determination to management objectives.

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## 1. Introduction

Ecosystem-based fisheries management (EBFM) calls for a recognition of ecosystem processes when managing living marine resources, rather than managing species in isolation (e.g. Francis et al., 2007; Larkin, 1996; Murawski, 2007; Pikitch et al., 2004). Practical application of EBFM is conditioned upon the definition of appropriate indicators and associated reference points to reflect the multiple objectives of an ecosystem approach (e.g. Jennings, 2005; Sainsbury et al., 2000). Challenges for fully implementing EBFM include defining the scope and appropriate values for these quantities as well as testing their performance.

Ecosystem indicators offer a synthetic view of a marine system, and can focus on system-scale properties and attributes rather than the status of individual species, typically the focus of traditional fisheries management. Ecosystem indicators provide a means to assess the status of ecosystems relative to the multiple goals associated with EBFM, and as such have been suggested as useful tools for implementation of EBFM (Link et al., 2002). The use of indicators to describe the status of marine ecosystems has received much attention, with a suite of indicators typically recommended as an avenue toward encompassing both human-induced and environmental pressures on marine ecosystems, and the multiple objectives associated with an ecosystem approach (Methratta and Link, 2006; Rice and Rochet, 2005). While it is feasible to calculate a very large number of indicators (e.g. EcoAP, 2012), salient features of ecosystem change can be captured by a relatively small set of indicators that focus on key objective-related properties (e.g. Fulton et al., 2005; Shin et al., 2010a). Thresholds in indicator responses to system pressures (Samhouri et al., 2010) have been used to suggest reference points for use in indicator-based fishery harvest control rules (Fulton et al., 2005; Link, 2005). Successful application of ecosystem indicators as fisheries management tools requires an understanding of how changes in system drivers (including exploitation) translate to both changes in the values for individual indicators, and in the interactions among indicators. Values for ecosystem response indicators have been shown to correlate well with fishing pressure (Link et al., 2010b). Preliminary thresholds in indicators associated with proxies for total system removals have

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been empirically identified for a few marine ecosystems, including the Northeast US Large Marine Ecosystem (LME) (Large et al., 2013), the North Sea (ICES, 2009, 2011a), and the Baltic Sea (ICES, 2011b).

A large range of ecosystem modeling tools have emerged to implement EBFM (e.g. Plagányi, 2007; Townsend et al., 2008), and have been used extensively to explore how values of ecological indicators respond to pressures (such as fishing; c.f. Fulton et al., 2005). Simulation modeling frameworks offer a powerful approach for identifying tradeoffs associated with values for system properties of interest. Modeling the effects of fishing within the ecosystem model, and then evaluating the consequence of these effects for model outputs, including ecosystem indicators of interest, quantifies tradeoffs among conflicting objectives associated with alternative management actions. Such methods serve as powerful tools for addressing issues associated with EBFM, where management objectives extend beyond yield and biomass associated with target fishery components.

In this paper we use a simulation model of a finfish community in the Northeast US LME to explore the impact of alternative fishing scenarios on the responses of ecosystem indicators. We demonstrate the sensitivity of indicators to these scenarios and then evaluate how these indicators change given the imposition of annual limits (ceilings) on total system-wide catches. We use a multispecies biomass production model (MS-PROD, Gamble and Link, 2009), which incorporates competition and predation in addition to fisheries exploitation of target species, to generate time series of ecological indicators. The simulated indicator responses to catch are then used to develop candidate values for ceilings. Finally, we evaluate the consequences for indicators of fishing using these threshold-derived ceilings on system catches.

## 2. Methods

### 2.1. Simulation framework

The application of marine ecosystem models for the Northeast US LME has encompassed a full range of modeling tools (Link et al., 2011). We parameterized a multispecies production model (MS-PROD, Gamble and Link, 2009) to be characteristic of a community of 10 finfish species (or in some cases, species groups) on Georges Bank in the Northeast US LME, a community that is the primary target of commercial fishing (Gaichas et al., 2012). The model incorporated density dependent population growth, the direct effects of fishing, and species interactions via competition and predation. Predation impacts were modeled on small pelagic species by some groundfish and elasmobranch species. The model did not consider positive impacts of predation on predator species as it is expected that predators will have alternative prey available due to the fact that the modeled predator species eat a wide variety of prey species (Garrison and Link, 2000). The dynamics of the model over time are governed by:

$$\frac{dB_i}{dt} = r_i B_i \left(1 - \frac{B_i}{K_i} - \frac{\sum_{j \neq i} \alpha_{ji} B_j}{K_{\mathrm{Tot}} - K_i}\right) - B_i \sum_{j \neq i} P_{ji} B_j - u_{G_i} B_i \qquad (1)$$

where $B_i$ is the biomass of group $i$, $r_i$ is the intrinsic growth rate for group $i$, $K_i$ is a carrying capacity for group $i$, $\alpha_{ji}$ is the effect of competition on group $j$ on group $i$, $K_{\mathrm{Tot}} = \sum K_i$, $P_{ji}$ is the magnitude of predation by group $j$ on group $i$, and $u_{G_i}$ is the exploitation rate experienced by guild $G$ that group $i$ belongs to. Full technical details of the production model and description of the modeled community are given in Gamble and Link (2009) and Gaichas et al. (2012), and parameter values are given in Appendix A.

### 2.2. Base–case scenario

The model was initialized with the current biomass estimates for each species (Table A.1), hereafter referred to as the base–case scenario. The model was then projected forward for 50 years, under a fixed fishing pattern. Fishing patterns consisted of pre-specified exploitation rates for three distinct guilds: groundfish, pelagic, and elasmobranch species (each species within a guild was subject to the same exploitation rate). A fully crossed set of simulations were conducted with guild-specific exploitation rates ranging from 0% to 90% (in increments of 10%), resulting in a set of 1000 simulations for each initial biomass vector.

The time trajectories of species biomass and catches from the model output were used to compute time series for a set of ecosystem indicators:

1. TotBio: total system biomass (summed over all species),
2. TotCat: total system catch (summed over all species),
3. Cat/Bio: system exploitation rate (i.e. TotCat/TotBio),
4. PropOF: the proportion of species that are overfished (proportion of species with biomass below 50% of the current [single-species] estimate of $B_{\mathrm{MSY}}$),
5. PropPel: the proportion of total biomass that is made up of pelagic species (Table 1),
6. PropPred: the proportion of total biomass that is comprised by predatory species (Table 1),
7. MTLcat: mean trophic level of the catch (MTL weighted by the species composition in the catches), and
8. MTLbio: mean trophic level of the community (MTL weighted by the system species biomass composition).

The final values of indicators after 50 years were summarized for each of the 1000 fishing patterns tested.

Indicators 2–4 reflect the highly visible conflicting objectives of fisheries management, yield and biodiversity. Indicators 5–8 summarize community composition. We used the proportion of pelagic biomass rather than the more commonly used pelagic to demersal ratio (P:D) because for some combinations of exploitation rates this ratio could be very large (for example, when simulated exploitation on groundfish and elasmobranchs was sufficient to result in extirpation of these groups). Mean trophic level was calculated based on fixed values for MTL for individual species (Table 1). Consequently, correlation among the values for indicators 5–8 can be expected, though each indicator is measuring something slightly different.

The set of indicators mirrors those selected by the IndiSeas project for comparing the effects of fishing across marine ecosystems (Coll et al., 2010; Shin et al., 2010b). We did not use the variability in system biomass, an additional indicator used by the IndiSeas group, because our simulations are based on a deterministic model that leads to equilibrium; the value of this indicator would tend toward zero and largely depend on the length of the simulation period rather than the particular fishing scenario.

### 2.3. Sensitivity to initial conditions

Sensitivity of indicator values to the initial biomass for each species was evaluated by running three additional scenarios, that increased (or decreased) the biomass of species groups relative to the total biomass. For each of the three species groups (groundfish, pelagics, elasmobranchs), we conducted simulations that increased the biomass of the group of interest by 50% over the current values (Table 1), and also reduced the biomass of the other groups by 50%. While simplistic, this approach highlights effects of large-scale changes in the community species composition on the responses of indicators to fishing effects. Alternative approaches could include randomly sampling an initial biomass for each species from the

**Table 1**
Species groups and specifications used in the modeled Georges Bank community. Single-species $B_{MSY}$ values taken from Gaichas et al. (2012). Mean trophic level values derived from the NMFS Northeast Fisheries Science Center Food Web Dynamics Program food habits database (Link et al., 2006; Smith and Link, 2010).

| Species group | Single-species $B_{MSY}$ ($t$) | Mean trophic level | Guild | Predator? |
|---|---|---|---|---|
| Cod | 148,084 | 4.4 | Groundfish | Yes |
| Haddock | 158,000 | 4.1 | Groundfish | No |
| Herring | 134,120 | 3.2 | Pelagics | No |
| Mackerel | 38,800 | 3.7 | Pelagics | No |
| Redfish | 54,200 | 3.8 | Groundfish | Yes |
| Skates | 211,146 | 4 | Elasmobranchs | Yes |
| Spiny dogfish | 20,000 | 4.3 | Elasmobranchs | Yes |
| Winter flounder | 16,000 | 4.2 | Groundfish | No |
| Yellowtail flounder | 43,200 | 3.2 | Groundfish | No |
| Windowpane flounder | 5599 | 4.2 | Groundfish | No |

current point estimate (with an assumed observation error variance), or by using the historical biomass estimates (to obtain cross-covariance among species that is reflective of actual species composition).

### 2.4. System-wide ceilings on catch

The effect of imposing a system-wide ceiling on total catches (to reflect limits on system productivity) was explored by pre-specifying a value for the ceiling at the start of the simulations, proceeding as above (Section 2.2) for the different fishing patterns, but reducing exploitation rates in years where the predicted system-wide catch would be above the ceiling. Exploitation rates were scaled down equally among guilds to result in an annual catch equal to the ceiling. Indicators were then calculated once the 50 years projections with these ceilings were completed.

### 2.5. Indicator thresholds

Threshold values of system-wide catches that resulted in a response in indicators during simulations were determined by applying a generalized additive modeling (GAM) approach (Large et al., 2013) to the 1000 time series of indicators and catches obtained during the base–case simulations (Fig. 1). Our goal was to identify values for the total catch (TotCat) that were associated with changes in the values for response indicators. We fit GAM models of the form Indicator ~ s(TotCat) for each simulation (where $s(x)$ is a non-linear smoothing function), and selected GAM models that resulted in a fit to the data that were significant compared to a linear approach (Large et al., 2013). We then defined threshold catches for indicators by obtaining the mean of the range of catches predicted by the GAM to be associated with a significant first derivative of the smoothing function (Fig. 1c).

Finally, we used the median of the threshold distribution for each indicator as candidates for ceilings on system-wide catches and re-ran the simulation procedure in Section 2.4 with ceilings set at these values. We then calculated values for indicators as before.

## 3. Results

### 3.1. Initial conditions

The final values for indicators were generally not sensitive to the initial biomass, with few instances of indicators differing from the

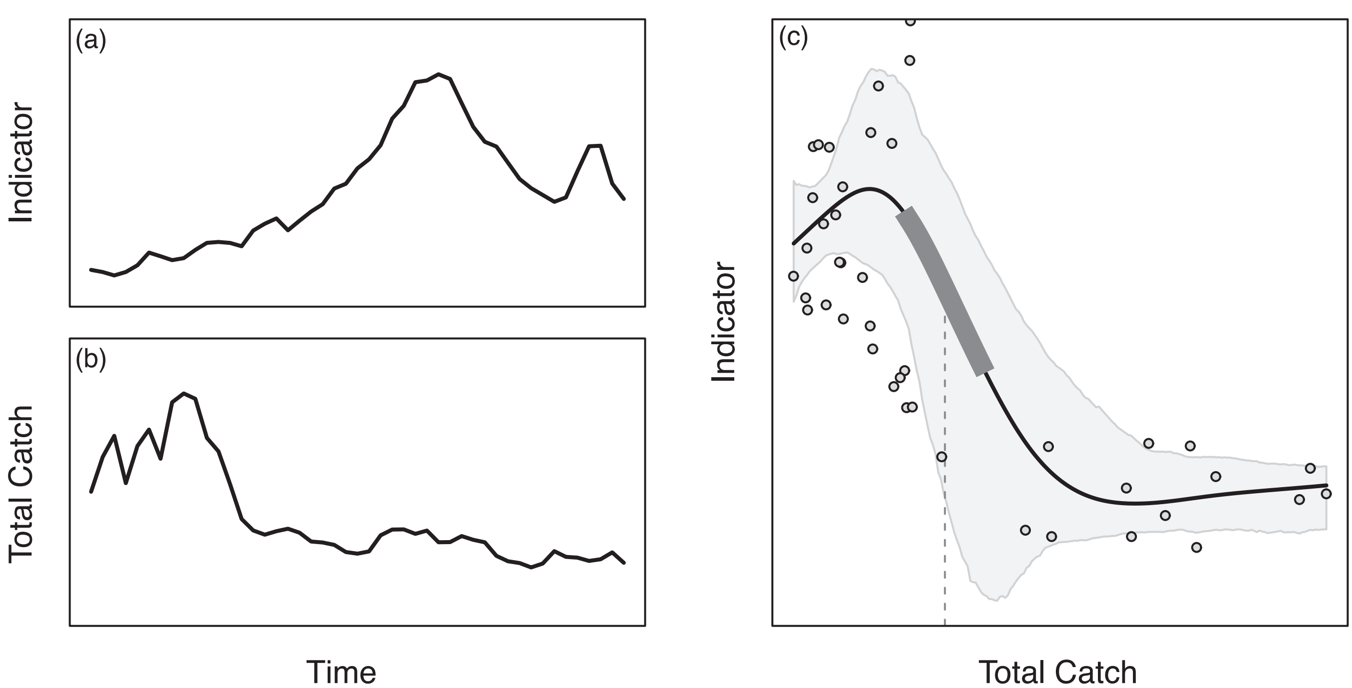


**Fig. 1.** Summary of the method used to calculate indicator thresholds. Simulated time series of indicators (a) and total catch (b) from the multispecies fishery model were regressed in a pressure/response framework using a GAM (c). Bootstrapped confidence intervals of the GAM (gray shaded area) identified values for which the estimated smoothing function had a significant first derivative (solid thick line). The threshold value of the total catch for a given indicator for that simulation was then calculated as the mean of these values (dashed vertical line). For full explanation and details of the method, see Large et al. (2013).

**Table 2**
Percentage of fishing pattern simulations (Georges Bank) that resulted in indicator values different than the base scenario by at least 5% when changing the initial species biomass composition. Values for proportion-based indicators (PropOF, PropPel, and PropPred) reflect the percentage of simulations where the absolute change was greater than 5%. Row denotes species group of which initial biomass was increased by 50% over the base scenario values in Table A.1.

| Increased initial biomass group | Percentage of simulations for which final indicators varied >5% from base scenario | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | TotBio | TotCat | Cat/Bio | PropOF | PropPel | PropPred | MTLcat | MTLbio |
| Groundfish | 6% | 10% | 6% | 0% | 1% | 1% | 0% | 0% |
| Pelagics | 4% | 10% | 6% | 0% | 2% | 0% | 0% | 0% |
| Elasmobranchs | 5% | 11% | 6% | 0% | 0% | 1% | 0% | 0% |

base–case scenario (Table 2). Instances where indicators differed from the base–case scenario value by at least 5% (or an absolute value of 0.05 for the proportion-based indicators), for example for TotBio and TotCat (Table 2), were generally for simulations where the base–case values for these indicators were low due to system collapse (e.g. associated values for PropOF were close to 1). This result was consistent despite the guild selected for higher initial biomass (Table 2).

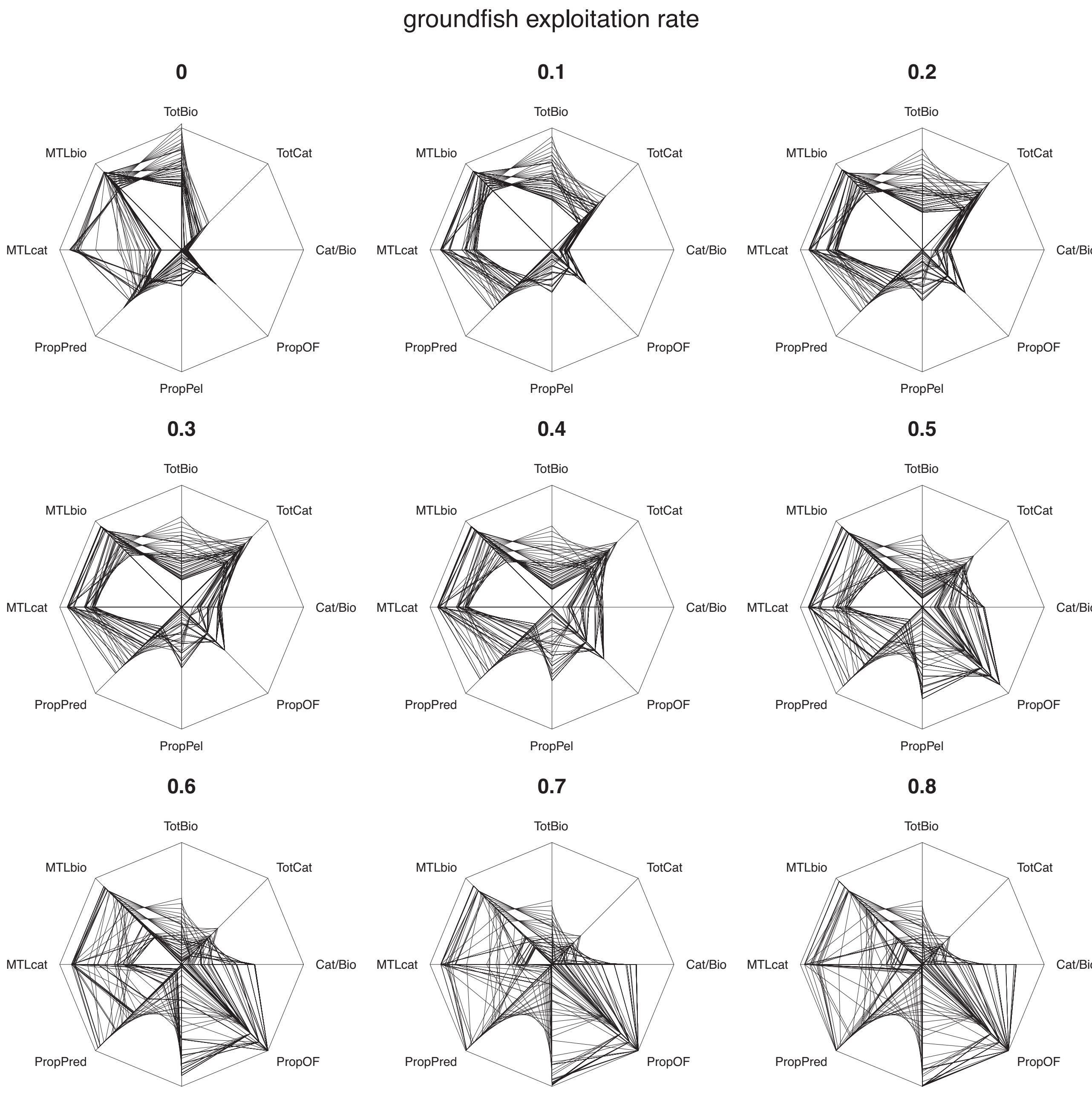


**Fig. 2.** Sensitivity of ecosystem indicators to exploitation rate of groundfish species. Each simulation (guild-specific fishing pattern) is represented by a single polygon, multiple polygons within each plot represent different values for exploitation on the other guilds (pelagics and elasmobranchs). Indicator values are scaled proportional to either their maximum value (TotBio, TotCat), a fixed value (5 for MTLcat, MTLbio), or to one for proportional data (Cat/Bio, PropOF, PropPel, PropPred).

*3.2. Base–case results*

Spider plots of final values for the indicators given different values for the exploitation rate for the three guilds reveal tradeoffs among indicators, and how these change based on exploitation rate (Figs. 2–4). These plots reveal typical tradeoffs such as those associated with biomass and yield as a function of exploitation rate, but also demonstrate the sensitivity of indicators to the exploitation rate of other system components.

Biomass-based indicators were most sensitive to groundfish exploitation rate (Fig. 2), with the distribution of total biomass (TotBio) shifting from high values to low values with increasing groundfish exploitation rate (Fig. 2), in contrast to the full range of observed values being obtained when the simulations were partitioned as functions of the exploitation rate on pelagics (Fig. 3) and elasmobranchs (Fig. 4). High values for PropPel and PropOF were obtained with high exploitation rates on groundfish. Intermediate values for PropPred were obtained at low levels of groundfish exploitation, whereas both low and high values were obtained when groundfish exploitation rate was high (Fig. 2). This reflects that cod and redfish in the groundfish guild are predators, and that there is competition within the model among groundfish species and the elasmobranch groups (spiny dogfish and skate), which are also predators.

Increasing the exploitation rate on the pelagics resulted in higher values for PropPred, and lower values for PropPel, though the total system biomass was fairly insensitive to these rates (Fig. 3). Similarly, PropPel and PropPred were sensitive to elasmobranch exploitation rate (though in the other direction, Fig. 4). The mean trophic level of the catch (MTLcat) was more variable in simulations with high elasmobranch exploitation rates, presumably allowing the relative biomass of pelagic species to have an effect on this

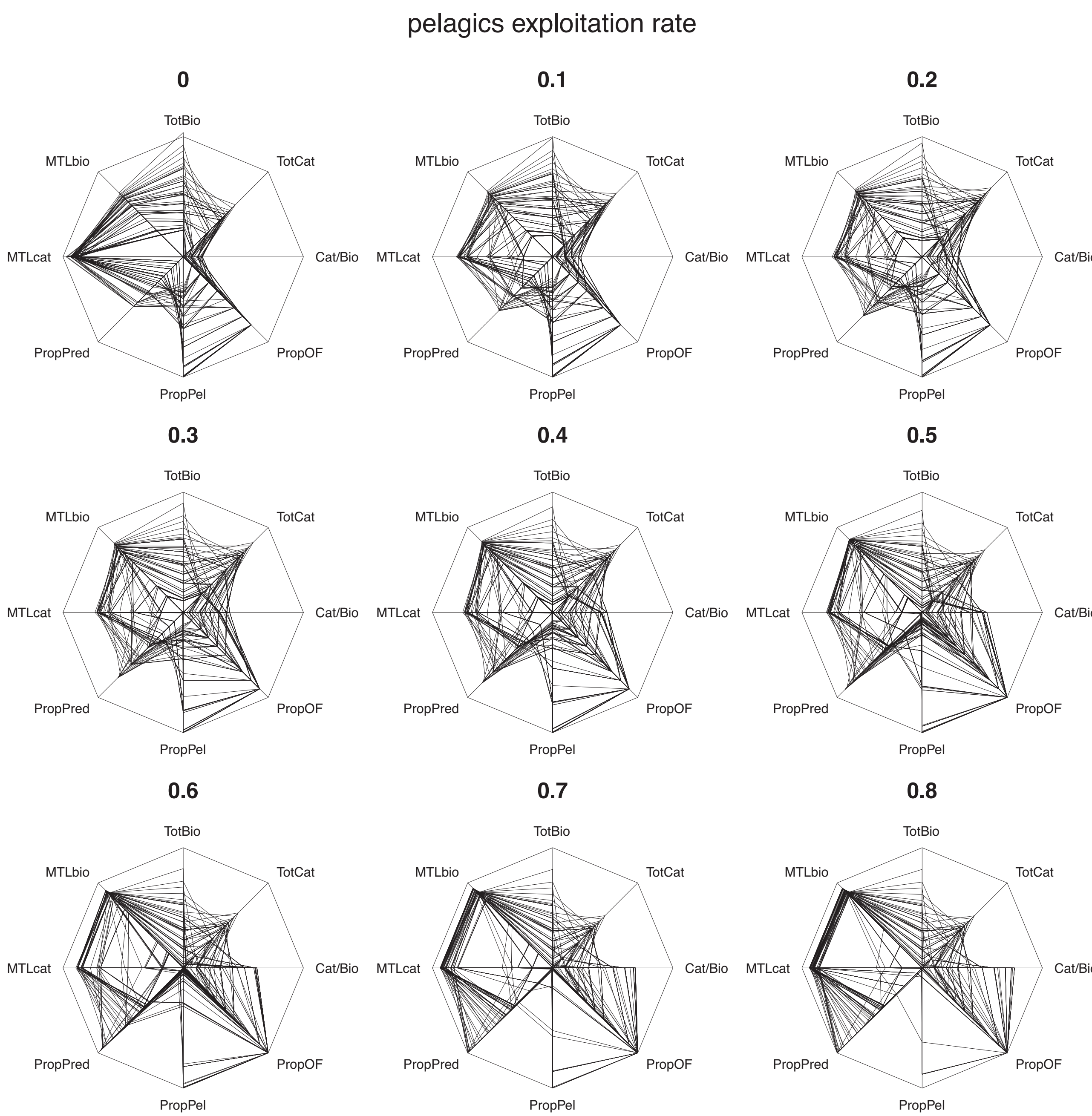


**Fig. 3.** Sensitivity of ecosystem indicators to the exploitation rate of pelagic species. Plot as for Fig. 2.

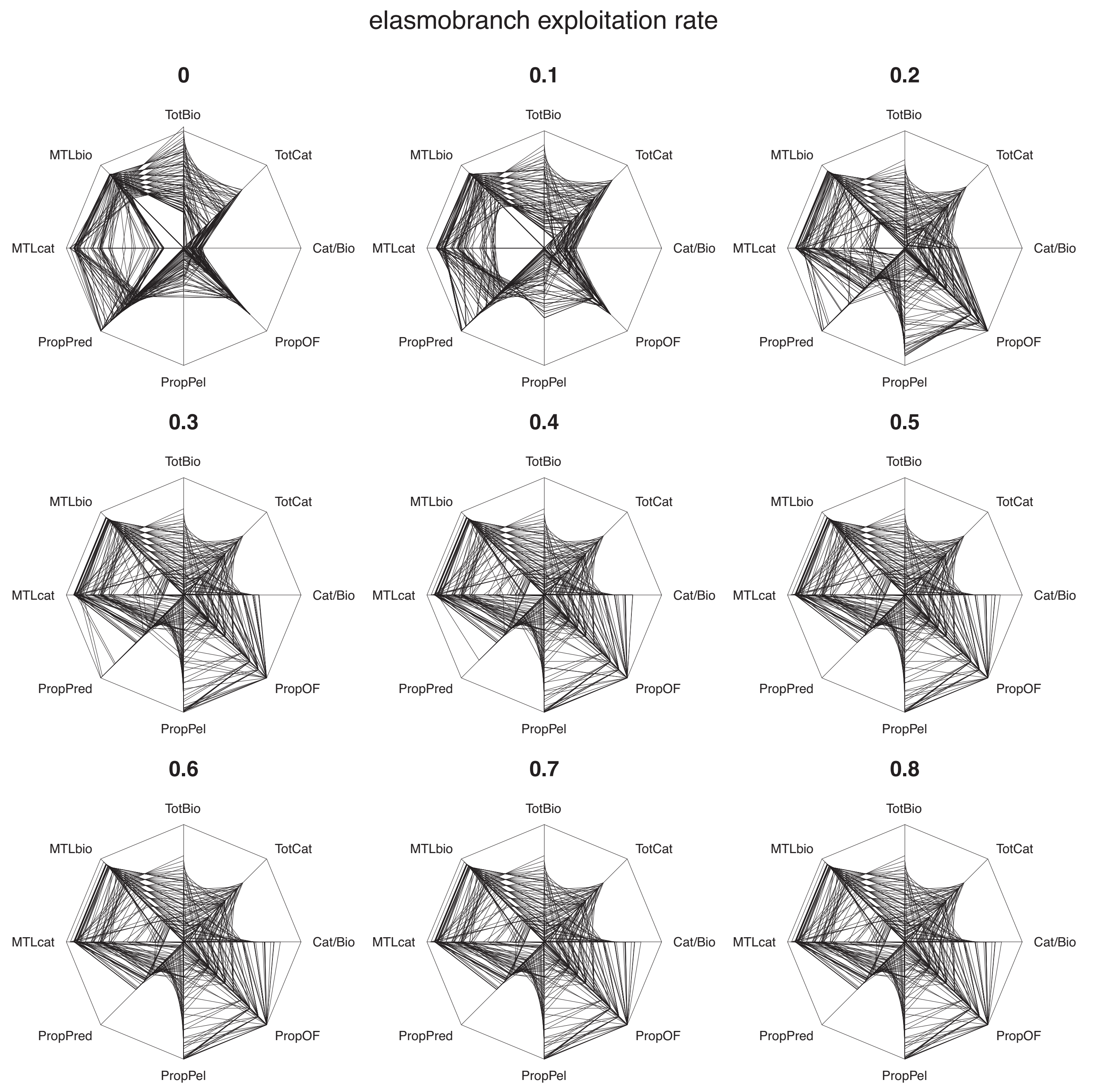


**Fig. 4.** Sensitivity of ecosystem indicators to the exploitation rate of elasmobranch species. Plot as for Fig. 2.

indicator when biomass of the elasmobranch groups was low. Indicators were relatively insensitive to increases in elasmobranch exploitation rate above 20% (Fig. 4), reflecting the low capacity of this group to withstand fishing pressure due to lower intrinsic growth rates.

Summarizing the set of indicators as functions of the final system exploitation rate (Cat/Bio) revealed an inability for biomass persistence at system exploitation rates above 50%, despite the specific relative fishing pattern on the different guilds (Fig. 5). Maximum values for yield from the community were obtained when the system exploitation rate was between 20% and 30%, though system collapse still occurred for some fishing patterns at these exploitation rates (Fig. 5b). While PropOF and PropPred showed clear relationships with the system exploitation rate (Fig. 5c and e), remaining indicators (e.g. MTLcat, PropPel) were less correlated with this quantity, reflecting their dependence on individual guild-specific exploitation rates (Fig. 5). The response of the mean trophic level of the landings to fishing pressure has been shown to be highly dependent on the fishing pattern among species (Branch et al., 2010).

### 3.3. Ceilings on system-wide catch

Setting ceilings on system-wide annual catches constrained values for indicators (Fig. 6). These ceilings revealed levels at which further increases in annual ceilings no longer resulted in changes in values for indicators. Generally, ceilings above 150,000*t* had no impact on the distributions for indicator values, although

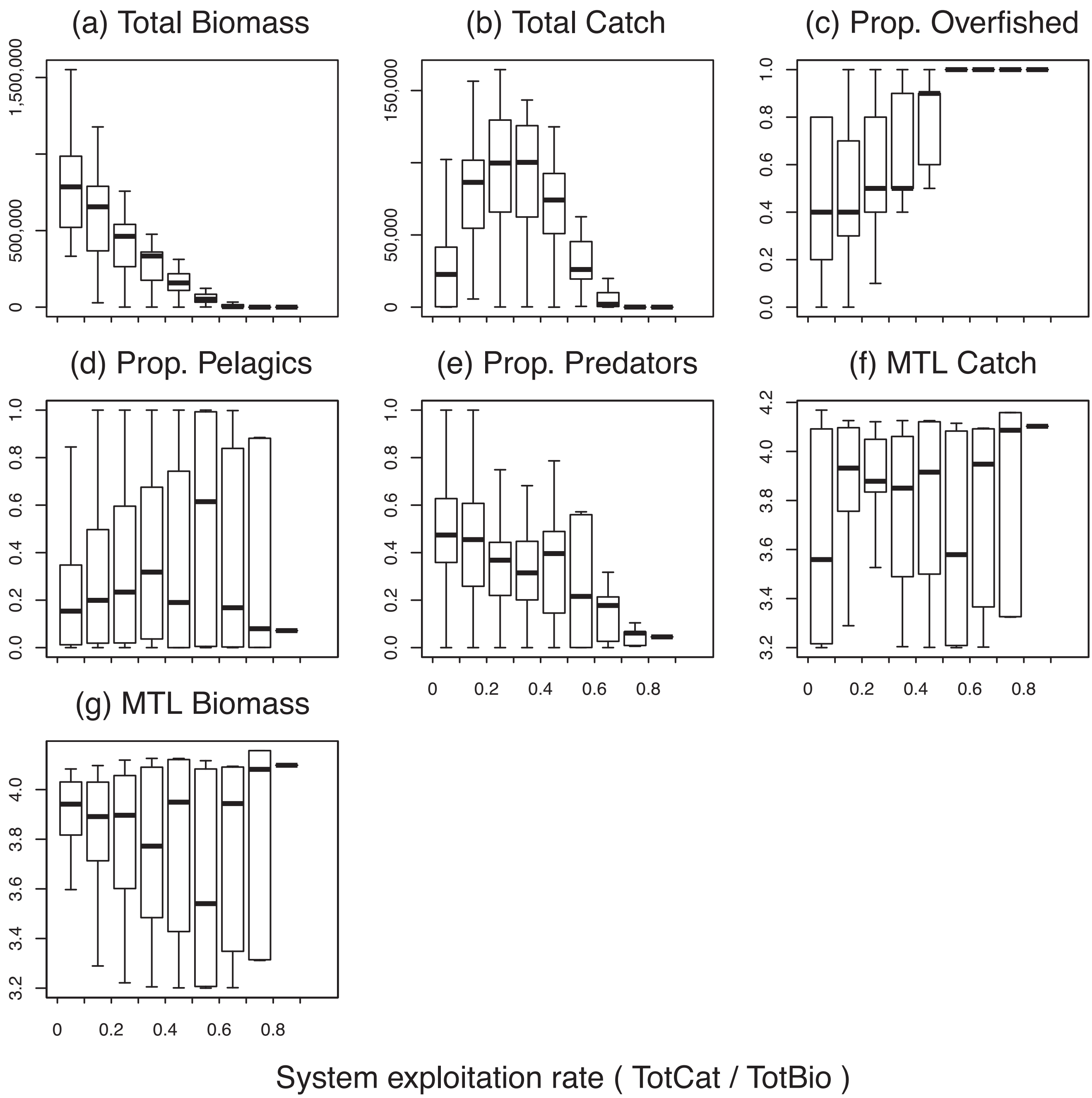


**Fig. 5.** Indicator values from simulations as a function of system exploitation rate (Cat/Bio). Boxes represent simulations where value for Cat/Bio ranged from 0% to 10%, 10–20%, etc.

some indicators showed responses at ceilings below this value. Indeed, there were shifts in the variance of the final values for many indicators at ceilings between 90,000*t* and 100,000*t*. The proportion of species that were overfished (PropOF) showed two general increases, with values constrained below 0.2 at ceilings less than or equal to 100,000*t*, and constrained below 0.4 below 150,000*t* (Fig. 6d). Ceilings on system catches were able to constrain system-wide exploitation rates, and consequently more frequently maintain higher proportions of predators in the system and lower proportions of pelagic biomass. Ceilings greater than 140,000*t* resulted in a sharp increased variability in the MTL of the catch, however the MTL of the community (MTLbio) changed more gradually, but showed increased variability at lower levels of total system removals (Fig. 6g and h).

### 3.4. Using thresholds in indicators to set ceilings

The distributions for the mean value of catch that resulted in significant responses in indicators under the base–case scenario were broad (Fig. 7). This is unsurprising, as the simulations used a wide range of guild-specific exploitation rates, and detectable changes in indicator values are restricted to the range of catches observed during the time series. However, modes in the distribution of these values were observed, with these being different depending on the indicator used. Community composition indicators typically showed a catch threshold lower than that provided by biomass (Fig. 7), suggesting that these indicators respond to the effects of fishing at lower levels of system catch.

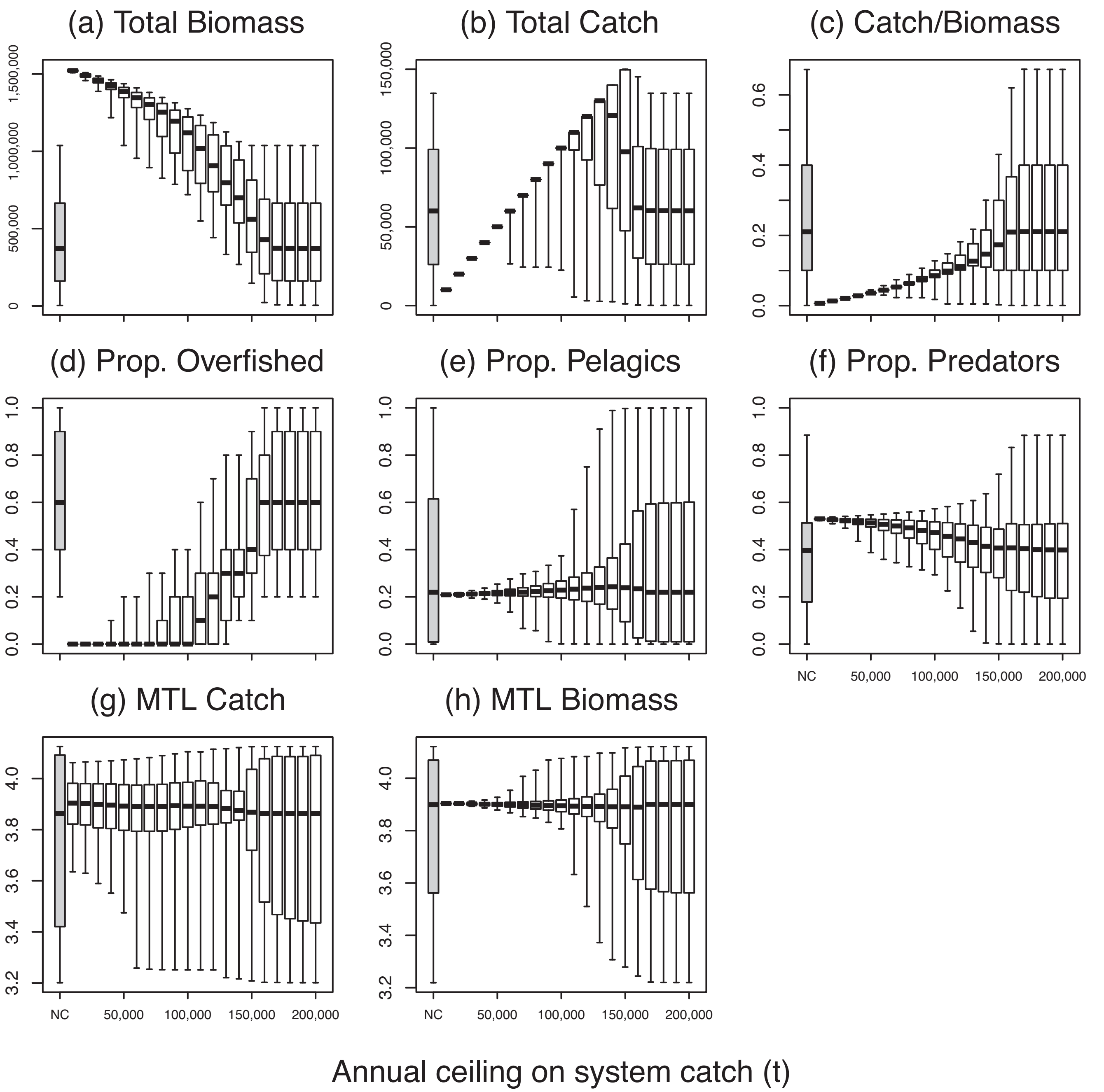


**Fig. 6.** Distribution of final values for Indicators given ceilings on the annual total system catch. Gray boxplot 'NC' shows distribution obtained when no ceiling was in place.

We used the median of the distributions in Fig. 7 to set ceilings on system catch and then compared final values for indicators based on the indicator used to set that ceiling (Fig. 8). In general, ceilings set using the community composition indicators resulted in higher biomass, higher yields, and fewer species overfished than say, ceilings set using total biomass. These results are consistent with those presented in Fig. 6, as the median landings threshold based on total biomass (Fig. 7) was above the value at which ceilings constrained the final values for indicators.

## 4. Discussion

Placing ceilings, or limits, on the total amount of removals from a marine ecosystem is consistent with moves toward EBFM, as this implies recognition that there are limits to aggregate system productivity, some of which fishing activities can exploit (e.g. Fogarty and Murawski, 1998; Pikitch et al., 2004). Indeed, calls for and adoption of precautionary limits on fishery removals of lower trophic level species in many marine ecosystems (e.g. Constable et al., 2000; Smith et al., 2011) are inherently based on recognition of such limits to system productivity. Production models have been used extensively to identify aggregate multispecies/ecosystem yields, with consistent results that maximum sustainable yield estimated from aggregated models (multi-species maximum sustainable yield) is less than the sum of individual single-species MSYs (e.g. Brown et al., 1976; Link et al., 2010a; Meuter and Megrey, 2006). These aggregate quantities (and others much like them) have been suggested as alternative reference points (whether as

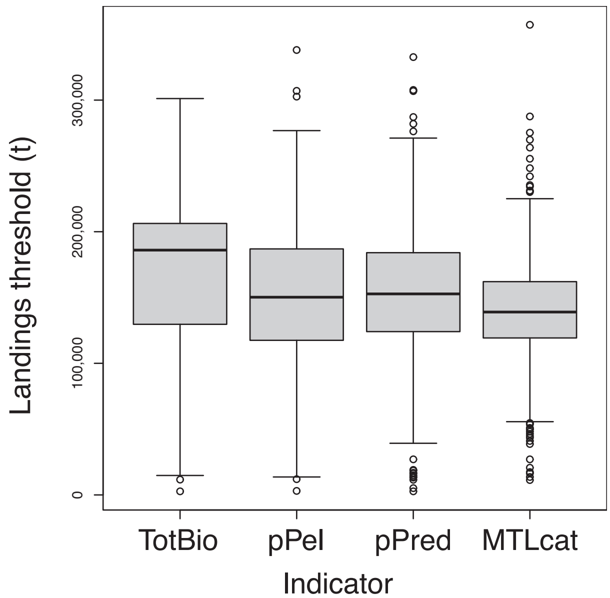


**Fig. 7.** Threshold values of catch associated with Indicators obtained from Generalized Additive Modeling of simulated indicator time series.

targets or limits) for fisheries management under an ecosystem approach (Link et al., 2012; Sainsbury and Sumaila, 2003).

Rather than exploring the consequences of production modeling for single and multiple species yields alone, we used the response of ecosystem indicators to fishing pressure as a means to understand where change-points can be expected to occur, and suggest that these changes in indicators can be used as reference points in harvest control rules in much the same way as limits or targets based on MSY (or MMSY). Our results show that ceilings on system-wide catches can indeed be used to constrain values for indicators. This also suggests the converse, that objectives associated with managing a multi-species fishery can be achieved by adjusting fishing pressure to constrain (through monitoring) the values for indicators. We chose to apply these constraints through the use of system-wide ceilings on annual catch, with reduction in exploitation rate across all species groups. A more adaptive approach may be to adjust exploitation on specific species groups given the direction of change and expected values for certain indicators, perhaps with differential management responses depending on the values for other indicators relative to reference points (e.g. Link, 2005).

The values for catch and exploitation rate levels that resulted in ecosystem indicator change were typically lower than those estimated for this modeled system to achieve single-species maximum sustainable yield, though maximum yields from the base–case simulations (Fig. 5b) were obtained at system exploitation rates (TotCat/TotBio) similar to those estimated to produce maximum sustainable yield at the aggregated level (Gaichas et al., 2012). In practice, these composite yields can be used to manage, or at least constrain, a portfolio of fisheries toward improved sustainability and profitability, as demonstrated in Alaskan marine ecosystems (e.g. Livingston et al., 2011; Schindler et al., 2009; Witherell et al., 2000), and the Antarctic (Constable et al., 2000; Constable, 2011).

We determined threshold values of total catch from the time series of indicators within simulations using a GAM approach (e.g. Large et al., 2013). There are many ways to calculate thresholds in pressure–response relationships. In addition to the GAM method presented, we also performed piece-wise regressions on the catch and indicator time series, with qualitatively similar results to the GAMs when the estimated breakpoints in the regressions were used as threshold values (results not shown). Our approach is different from that used by Samhouri et al. (2010), who compared equilibrium (final) values of indicators among simulations with system-wide exploitation rates. Obtaining changes in indicators from values obtained from a time series of model output more appropriately reflects empirical analyses of determining indicator response to pressures. However the fact that we used a deterministic model with fixed exploitation rates and no observation error meant that regressions of indicators against catch had substantial autocorrelation in residuals. Analyzing the final values for indicators (sensu Samhouri et al., 2010) provides the opportunity to look more broadly at indicator response over a full range of simulations. Using equilibrium values for indicators may obscure some relationships however, as some indicators will respond to pressures faster than others (e.g. McClanahan et al., 2012).

Comparing the performance of alternative ceilings based on model output-derived indicator thresholds provides a transparent way of examining tradeoffs between different approaches. Our analyses can be seen as a step toward implementing a Management Strategy Evaluation approach (c.f. Bunnefeld et al., 2011), though without internal feedback from the model simulations. Extensions to the analyses that use a historical period of fishing to condition the ceilings and then implement the ecosystem-based control rule (limits on system catch) within the same modeled system, with comparisons of performance among alternative methods for setting ecosystem-based reference points, are clear avenues for further work. As the focus of our analyses was on understanding system and indicator behaviors given different fishing patterns, we did not include the effects of observation error. However this and other sources of uncertainty could easily be implemented in such an expanded Management Strategy Evaluation framework.

The wide distribution for threshold values obtained demonstrates the sensitivity of threshold determination methods to exploitation history, and underscores the need to incorporate expert knowledge and system behavior information when defining reference points. This is not solely a problem for ecosystem approaches, the challenge of defining reference points conditioned upon these initial conditions is well documented in single-species fisheries management (e.g. Restrepo and Powers, 1999), and is often of particular concern when defining empirical reference points, such as those often applied to data-limited fisheries (e.g. Dowling et al., 2008). Values for indicators associated with our indicator-based ceilings do not seem at odds with the historical values for these indicators based on survey data (Fig. 8), except perhaps for the proportion of predators (PropPred). Simulation modeling exercises like those presented in this paper allow for an exploration of how modeled systems can be expected to respond given different drivers. Ultimately, reference points for given indicators should be determined not only from analyses that demonstrate system behavior, but also from an understanding of what particular system traits are deemed to be preferable. This implies a weighting of system characteristics according to management objectives.

Our analyses focused on fishing effects on indicators, in that population dynamics were not subject to environmental factors, nor was there nonstationarity in trophodynamics and competitive effects, which would result in hysteresis within the system. As such it was only possible to focus on a small set of indicators. However, the model framework could be used to assess the impact of including regime shift dynamics, such as a change in the carrying capacity of a pelagic species (representing time shifts in mean productivity of these groups, e.g. de Moor et al., 2011). The impact of additional pressures and drivers besides fishing effects on indicators is also of interest. Gamble and Link (2012) used a similar model framework

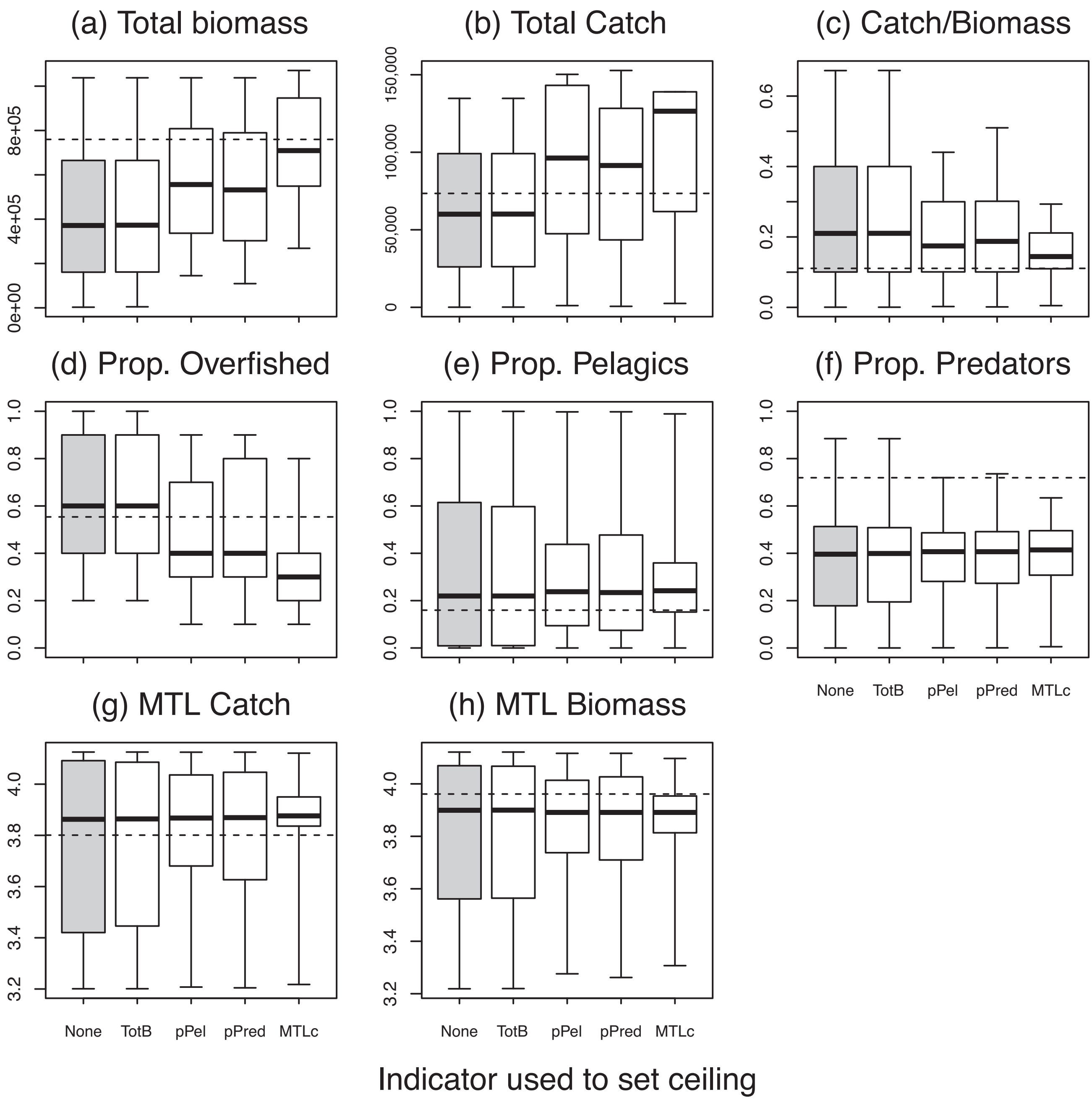


**Fig. 8.** Distributions for indicators obtained when the median landings threshold from GAM models were used to set ceilings on annual system catch, compared among indicators used to set the value for the ceiling. Horizontal dashed lines correspond to the mean values for indicators observed from survey data (1967–2007).

to investigate the impact on simulations of reducing the intrinsic growth rate of groundfish, as a hypothesized response to climate change. Extending analyses beyond production modeling to end-to-end system models (e.g. Fulton et al., 2011) would allow a more complete range of indicators and interactions among ecosystem components to be evaluated.

This paper demonstrates how ecosystem indicators can respond to fishing pressures, and how these responses can be used to determine system level reference points (e.g. ceilings on system catches) under an ecosystem approach. Although the simulation model used to represent the dynamics of the modeled community was relatively simple and focused on a small component of the Northeast US large marine ecosystem (i.e. solely the commercially important finfish community of Georges Bank), the scenarios conducted provide pathways to potential methods of using indicators to operate as reference points in ecosystem-based fishery control rules.

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## Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.ecolmodel.2013.05.016.

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