Supporting Online Material for

Reconsidering the Consequences of Selective Fisheries

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Materials and Methods for Figure 1

Ecosystem models

We compiled 36 published ecosystem models from 30 systems (Table S1). These models were either ECOPATH with ECOSIM (EwE) (*38*, *39*) or ATLANTIS (*40*) models. They were selected because they spanned systems with differing levels of productivity, degrees of environmental forcing (e.g., by upwellings), temperature profiles (primarily temperate and tropical), and levels of exploitation. In addition, the models were all publicly available, have been properly documented, have had their quality controlled, and have produced nonoscillatory results. [An additional model for Prince William Sound, Alaska (*41*), failed this last criterion and has not been included in this analysis.] For some models, multiple parameterizations existed; these models are marked with a "*" in Table S1. It was computationally impracticable to carry all parameterizations of every model through the whole analysis (as in some cases there were hundreds of parameterizations), so it was decided to use a subset of parameterizations. To select this subset, the parameterizations for each model were first scored (out of 5) against two criteria:

(i) simultaneous goodness of fit to multiple time series (a 5 was given if a parameterization provided a tight fit to multiple time series and spanned multiple functional groups or species, with lower scores given for less comprehensive fits or if fits were only achieved for a minority of groups in the model);

(ii) level of documentation within the model database on the source of the parameters and what motivated the alternative parameterization [a 5 was given if the parameterization was drawn from an alternative or extended data set for that system; a 3 if the parameterization was to cover uncertainty in a single data set (i.e., one parameterization covering upper bounds and another lower); and a 1 if it was purely hypothetical. A zero was given for no documentation].

Then, each parameterization was checked to see whether 100+ year runs (with no fishing, historical levels of fishing, and 10×, 100×, and 1000× historical levels of fishing mortality) did not exhibit oscillations independent of any environmental forcing. Any parameterizations showing any numerical instability that was due to parameterization rather than external forcing was removed from the set to consider.

Finally, the four parameterizations for each model with the highest rank (i.e. closest to a total score of 10) that were also numerically stable were carried through the rest of the analysis. For those models where four or less alternative numerically stable parameter sets existed, all parameterization were run through the analysis detailed below. Note that, after the subsets were selected, the original authors of the models (marked with "#" in Table S1) were contacted regarding which parameterizations they felt best represented their system and its function. In each case, the subset nominated by the original model developers and the subsets selected via the scoring method matched—which was taken as evidence that the selection method was robust. A full list of the scoring (a table would stretch for hundreds of pages) is available on request from the authors (Elizabeth Fulton, beth.fulton@csiro.au).

Note on fishing mortality

Two types of measures of fishing mortality are used in the ecosystem modelling: fishing mortality and average exploitation rate. Instantaneous fishing mortality *F* was applied to each species at the native time step of the model (from 12 hours in Atlantis to 1 month in EwE). For the purposes of Fig 1, we used the exploitation rate at the system level calculated as:

$$U = \frac{\sum_i Y_i}{\sum_i \beta_i}$$

where Y_i is the average annual yield from species *i* (averaged over the final 5 years of the simulation) and β_i is the average (over the final 5 years of the simulation) of the standing biomass in the year plus any production in that year (i.e., due to individual growth plus recruitment). This summation of standing biomass plus production was taken as the annual estimate of total available biomass for the species (given that short-lived species can produce more than their standing biomass in a single year).

Alternative measures of system-level fishing pressure were found unsuitable for a variety of reasons, mostly to do with interpretability or translation into an index that could be feasibly calculated in reality (i.e., outside the models). For example, there is no straightforward equivalent to individual fishing mortality at a system level, and simply using an average (or median) value across species is also problematic because of the issue of how to weight the species included in the composite measure; at one extreme, a simple average increases the weight of species with little biomass to contribute; at the other extreme, if a weighted average was used, then effectively the most biomass-abundant groups dominate the signals.

Analysis steps

For each ecosystem model, we first identified fished groups. For the "conventionally selective fishing" case, this list consisted of all groups historically fished (including whaling, sealing, etc.) in a system—in the majority of cases, given the trophic resolution of the models, this meant simply using the list of groups with nonzero fishing mortalities. Where there were groups (particularly higher trophic levels) that may have been exploited in the past but weren't any longer (e.g., cetaceans and pinnipeds), first, the model documentation was checked (as this often stated whether a group was recovering from a depleted state); then, historical records were checked with the International Whaling Commission (IWC) report series (available from http://iwcoffice.org/); and where necessary, the original model authors were contacted for final confirmation of fishing history for the different model groups. For those models including age structure, only larger size classes (i.e., the marketable size and age classes observed in reality) were identified as "fishable" (i.e., would have fishing mortalities applied in the analysis steps detailed below). For lightly fished systems, where few groups had been fished historically, the list of groups and age classes that could be fished was expanded beyond what had been seen historically to include groups (and sizes) fished in heavily exploited systems of a similar type.

For the "unselective fishing," all nonmicrofauna that have been exploited somewhere in the world were included (e.g., jellyfish, macroalgae, and small-bodied pelagic

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invertebrate groups such as krill), as well as a broader set of size and age classes. That is, all post establishment size and age classes were marked as fishable, but any larval age classes listed in the EwE models and the very smallest age classes in Atlantis, which represent tiny settlers, were not identified as fishable unless fish of that size for that species or functional group had been recorded as fished somewhere in the world. Hence, the "unselective fishing" includes a wider range of species and size groups than the "selective fishing."

With the list of "fishable" groups identified for the selective and unselective fishing cases, the following steps were then carried out for each parameterization used for each ecosystem. Note that, although the analyses were done at one time, a nonnormalized form of the selective fishing scenario results have been presented previously in (figure S1of (*2*). Also, for the remainder of this description, please treat the word ecosystem as meaning a specific parameterization of an ecosystem model. Hence, in the production of the final Fig. 1, averaging, etc., was done treating each parameterization as an individual case. Although this handling does mean that some systems were represented up to four times and that those replicates did not have independent model structures, the different parameterizations lead to as much variance across outcomes for a single system as that seen between systems. Consequently, more complicated statistical handling was not undertaken, as the replicates did not introduce significant bias in the form of the overall outcomes (i.e., the normalized results for a single system did not all clump high or low compared with other ecosystems).

1) For each fished group in the model, we incrementally increased and decreased the fishing mortality rate F_i (by 0.01, with $F_{min} = 0$ and $F_{max} =$ the F that causes the biomass of the group to decline to zero), while holding F constant for all other groups in the model, but allowing for full dynamic responses due to trophic interactions and direct fisheries extractions. This step produced an estimate of the fishing mortality that produced maximum sustainable yield $F_{MSY,i}$ for each fished group.

2) To allow for easy exploration of a wide range of fishing mortalities, the F for the fished groups in the model was set to $k \times F_{MSY,i}$ from step 1 (where k is a simple fishing mortality multiplier). Long-term runs (200–1000 years) were then run with k increasing incrementally from $k = 0$ (i.e., no fishing) to $k = 20$ (which was sufficient to cause all groups to drop to extinction in the selective fishing case; in the case of the unselective fishing, it was necessary to take this to $k = 100$ to reach the point where system-level annual removals reached 100% in all systems). For EwE, 1000 years was selected because these models are effectively equilibrium models, and the simulations had returned to a stable state (flat biomass trajectories through time) by 1000 years. Atlantis is not an equilibrium model and so never settles to one value—environmental variation from climate drivers interacting with ecological interactions means that there is always year-to-year variation, but under constant drivers (or repeating time series drawn from historical oceanographic observations), the biomass trajectories of the groups in the model do settle to vary within a band of values (with the very occasional rare event falling outside the band because of biophysical interactions); these bands are obvious if plotted as a time series or if maximum and minimum values are tracked over 5- to 10-year intervals through the course of a run. Initial investigatory runs showed that Atlantis runs settled into a band of values representative of their 1000 year state by 200 years. Consequently, given that Atlantis takes many days to run for

such long simulation periods (versus a few minutes for EwE), the full set of runs were only run for 200 years each in Atlantis.

3) Although predation was allowed to vary during step 1, in reality, F would also vary across species through time (with changing targeting and gear use). Also, given that ecosystem interactions can be complex, it was necessary to do additional runs to try and further fill out the phase space of possible outcomes when fishing across an entire ecosystem. In other words, because of trophic and other interactions iterating F_{MSY} across species does not cover a sufficiently large range of outcomes to paint a clear picture of even the general shape of ecosystem responses across a broad range of combinations of F applied across species. Consequently, to complement the MSY-based exploration, we developed a wide range of alternative F combinations by performing a set of fisheries policy searches using EwE to maximize a policy goal (represented by an objective function)—using the method detailed in the EwE user guide (available from http://www.ecopath.org/). The objective function used in the search combines economical, ecological, and potentially social terms. The set of searches was increased incrementally (in steps of 0.01) from the policy, with the economic contribution to the objective function set to 1 (effectively maximizing catch from the system, as there was little actual economic information included in the model formulations), and the ecological contribution (ecosystem structure and mandated rebuilding) set to 0 through to the opposite weightings (economic weighting = 0, ecological = 1).

To allow for representation of external pressure (or legal requirements) that force policy-makers to preserve or rebuild the population of a specific species, the objective function also includes a mandated rebuilding term (as described in the EwE user guide available from http://www.ecopath.org/). Rebuilding is not a universal requirement in all jurisdictions, but to facilitate a single analysis across all systems, a single rule was applied in all cases. In line with the majority of legal frameworks, harvest control rules, and fisheries science conventions on proxies for biomass supporting MSY versus unfished biomass (*42*), a mandated population size of 40% of unfished biomasses was chosen. In implementing these levels in the models, first, unfished levels needed to be determined. This was necessary because the base ECOPATH models did not represent unfished systems, and they included heavily exploited groups that were in an already depleted state, and so, simply setting rebuilding to 0.4 of ECOPATH values would be inappropriate. To find unfished levels, a long-term run of each ECOSIM model was done with all fishing mortality rates set to 0. Then, within the optimizations, the levels of mandated rebuilding from base ECOPATH levels were set according to how the base ECOPATH model biomasses compared with 40% of the unfished biomass levels.

Ideally social (employment) considerations would have also been included in this policy search. Unfortunately, employment information was not readily available for many of the modelled fleets, and the same fleet resolution was not available in all models, so social contributions to the objective function were not considered. Run times precluded also doing an optimality search directly with Atlantis. Consequently, the search was only run in EwE, and then, the fishing mortalities per group were applied in both models (EwE and Atlantis) to check for consistency in outcome across the models.

4) The results from the different analyses per system (i.e., the results of steps 2 and 3 for each parameterization of each model) were then combined (i.e., all simulations were plotted on the same plot) to produce aggregate plots of catch, available biomass (both standing stock and production in the year), and the number of groups that have dropped below 10% of their unfished levels (defined as "extirpated") against the system-level exploitation rate.

Finally, the overall plot comparing the selective and unselective cases (Fig. 1) was created by (i) normalizing the results for the system in step 4 relative to the maximum value observed across the two fishing cases (selective and unselective) for each metric (catch, biomass, and extirpations) in any simulation for that system (regardless of the kind of fishing used) and then (ii) combining all of the normalized results (i.e., across all parameterizations of all the systems) on a single plot. The solid line (in Fig. 1) marks the average across all ecosystems (and parameterizations) per fishing type (selective or unselective), and the confidence bounds mark the 5th and 95th percentiles across all ecosystems (and parameterizations), as generated by combining uncertainties from the various ecosystems (and parameterizations) by using standard error propagation methods (*43*).

With respect to the modelling, *balanced harvesting scenarios* were defined as those scenarios where all nonmicrofauna (and postlarval stages, see the definition of unselective scenarios at the start of this section) in a system are harvested in proportion to their productivity per unit of biomass (i.e., those fishing mortality scenarios produced in the steps outlined above that had a distribution of relative fishing mortalities that matched the distribution of relative productivity across all fished species were tagged as being a case of "balanced harvesting"). Although the identity and results of these scenarios was tracked through the analysis process, instead of individually marking them on Fig. 1 (which would make it difficult to read), the area encompassing the majority (over 90%) of these scenarios across the systems and parameterizations is marked in dark blue in Fig. 1. There is no pattern to the remainder of the cases except (i) none fall below the 5th percentile for total yield or available biomass and (ii) all but two are below the mean number of extirpations for selective fishing, and even those two are below the mean for unselective fishing.

The full set of results is available on request from the authors (Elizabeth Fulton, beth.fulton@csiro.au)
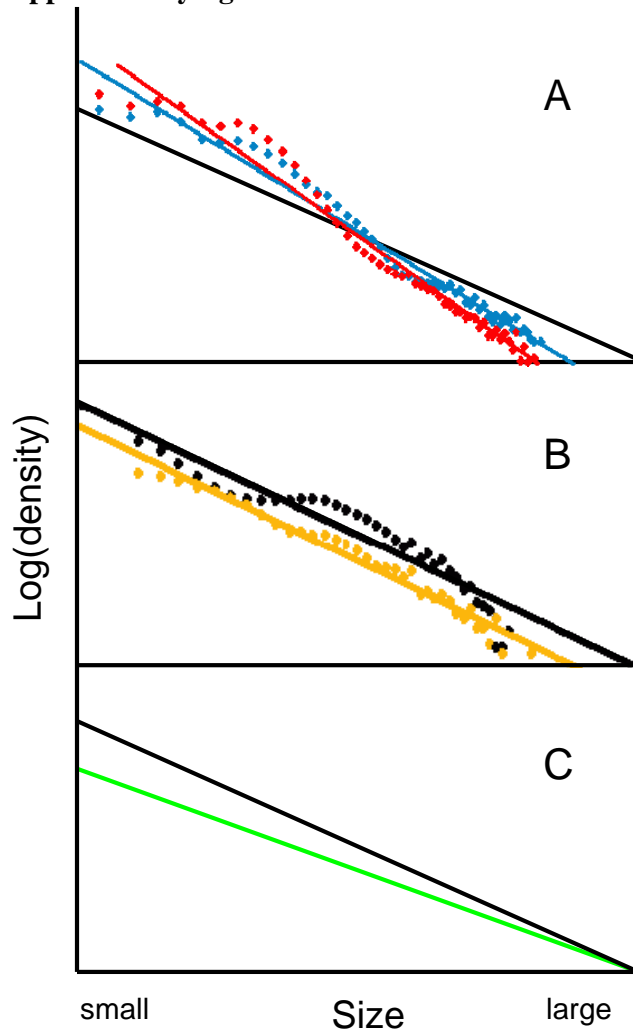
Supplementary figure S1

Fig. S1. The different community size spectra resulting from alternative harvesting strategies. Black lines/symbols are before fishing, colored lines/symbols are after fishing. (**A**) Conventional selective fishing: North Sea before fishing[1] and from two later periods, 1983–1987 (blue symbols and regression line), and after more accumulated fishing impact 1998–2002 (red symbols and regression line). (**B**) Unregulated weakly selective fishing: Lake Kariba, Africa, from an unfished (black symbols and regression line) and a heavily fished area (yellow symbols and regression line). (**C**) Conceptual model of balanced harvest (green line) with exploitation proportional to natural productivity (black line); we are not aware of an empirical example for this strategy.

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[1]The unfished line for the North Sea (A) is hypothetical based on (*64*) and data are numbers per km^2 from the North Sea International Bottom Trawl Survey (*65*), extracted from ICES online trawl survey database http://datras.ices.dk/Home/Default.aspx. Data from Lake Kariba (B) are all numbers per 45-m net set of experimental gillnets averaged over 1980 to 1994 [see (*66*) for details]. Size is binned to 2-cm-length groups.

Supporting Table

Table S1. List of 36 ecosystem models for 30 systems and their sources used to explore multispecies MSY. For some systems, two EwE models from different time periods were used. Those with a * had multiple parameterizations included in the analysis. Authors of the models marked with "#"were contacted regarding which parameterizations they felt best represented their system and its function. In each case, the subset nominated by the original model developers and the subsets selected via the scoring method described earlier matched—which was taken as evidence that the selection method was robust.

System	Model type	Notes and Source(s)
Aleutians	EwE	(*44*)
Australia Darwin Harbour	EwE	(*45*)
Baltic	EwE	Database for (*46*)
Bay of Biscay	EwE	(*47*)
Benguela*#	2 EwE	Database for (*46*)
Black Sea	EwE	Database for (*46*)
California Current*#	2 EwE and 1 Atlantis	(*48*, *49*)
Canada - Nth Gulf St Lawrence	EwE	(*50*)
Central Nth Pacific	EwE	Database for (*46*)
Chesapeake*	EwE	Database for (*46*)
Eastern Bering Sea	EwE	Database for (*46*)
Eastern Tropical Pacific	EwE	Database for (*46*)
Great Barrier Reef*#	EwE	(*51*)
Georges Bank	EwE	(*52*)
Georgia Strait	EwE	Database for (*46*)
Gironde Estuary	EwE	(*53*)
Grand Banks	EwE	(*54*)
Gulf Mexico*	EwE	(*55*)
Gulf Thailand 1973	EwE	Database for (*46*)
Irish Sea	EwE	(*56*)
New Zealand	EwE	(*57*)
North Sea*	EwE	Database for (*46*), (*58*)
North West Shelf*#	EwE	(*59*)
Port Phillip Bay*#	EwE	(*60*)
SE Alaska 1963	EwE	(*44*)
SE Australia*#	2 EwE and 1 Atlantis	(*61*)
Tampa Bay	EwE	Database for (*46*)
West Coast Vancouver Island	EwE	Database for (*46*)
Western English Channel*	2 EwE	(*62*)
West Florida Shelf	EwE	(*63*)

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