Rebuilding Global Fisheries

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After a long history of overexploitation, increasing efforts to restore marine ecosystems and rebuild fisheries are under way. Here, we analyze current trends from a fisheries and conservation perspective. In 5 of 10 well-studied ecosystems, the average exploitation rate has recently declined and is now at or below the rate predicted to achieve maximum sustainable yield for seven systems. Yet 63% of assessed fish stocks worldwide still require rebuilding, and even lower exploitation rates are needed to reverse the collapse of vulnerable species. Combined fisheries and conservation objectives can be achieved by merging diverse management actions, including catch restrictions, gear modification, and closed areas, depending on local context. Impacts of international fleets and the lack of alternatives to fishing complicate prospects for rebuilding fisheries in many poorer regions, highlighting the need for a global perspective on rebuilding marine resources.

Overfishing has long been recognized as a leading environmental and socioeconomic problem in the marine realm and has reduced biodiversity and modified ecosystem functioning (*1*–*3*). Yet, current trends as well as future prospects for global fisheries remain controversial (*3*–*5*). Similarly, the solutions that hold promise for restoring marine fisheries and the ecosystems in which they are embedded are hotly debated (*4*–*6*). Such controversies date back more than a hundred years to the famous remarks of Thomas Huxley on the inexhaustible nature of sea fisheries (*7*) and various replies documenting their ongoing exhaustion. Although management authorities have since set goals for sustainable use, progress toward curbing overfishing has been hindered by an unwillingness or inability to bear the short-term social and economic costs of reducing fishing (*8*). However, recent commitments to adopting an ecosystem approach to fisheries may further influence progress because they have led to a reevaluation of management targets for fisheries and the role of managers in meeting broader conservation objectives for the marine environment (*9*).

In light of this debate, we strive here to join previously diverging perspectives and to provide an integrated assessment of the status, trends, and solutions in marine fisheries. We explore the prospects for rebuilding depleted marine fish populations (stocks) and for restoring the ecosystems of which they are part. In an attempt to unify our understanding of the global fisheries situation, we compiled and analyzed all available data types, namely global catch data (Fig. 1A), scientific stock assessments, and research trawl surveys (Fig. 1B), as well as data on small-scale fisheries (*10*). We further used published ecosystem models (Fig. 1B) to evaluate the effects of exploitation on marine communities. Available data sources are organized hierarchically like a Russian doll: Stock assessments provide the finest resolution but represent only a subset of species included in research surveys, which in turn represent only a small subset of species caught globally. These sources need to be interpreted further in light of historical fisheries before data collection and illegal or unreported fisheries operating today (*11*). We focus on two leading questions: (i) how do changes in exploitation rates impact fish populations, communities, and yields, and (ii) which solutions have proven successful in rebuilding exploited marine ecosystems?

Models. A range of models is available to analyze the effects of changes in exploitation rate on fish populations, communities, and ecosystems. Exploitation rate (u_t) is defined as the proportion of biomass that is removed per year, i.e., $u_t = C_t/B_t$ where C is the catch (or yield) and B is the available biomass in year t. Single-species models are often used to determine the exploitation rate u_{MSY} that provides the maximum sustainable yield (MSY) for a particular stock. Fishing for MSY results in a stock biomass, B_{MSY}, that is substantially (typically 50 to 75%) lower than the unfished biomass (B_0). It has been a traditional fisheries objective to achieve single-species MSY, and most management regimes have been built around this framework. Recently this focus has expanded toward assessing the effects of exploitation on communities and ecosystems (*9*).

Multispecies models can be used to predict the effects of exploitation on species composition, size structure, biomass, and other ecosystem properties. They range from simpler community models to more-complex ecosystem models (*12*). Figure 2 displays equilibrium solutions from a size-based community model, which assumes that fishing pressure is spread across species according to their size and that a subset of species remains unfished (*13*). Results of more-complex ecosystem models across 31 ecosystems and a range of different fishing scenarios were remarkably similar (fig. S1 and table S1). With increasing exploitation rate, total fish catch is predicted to increase toward the multispecies maximum sustainable yield (MMSY) and decrease thereafter. In this example, the corresponding exploitation rate that gives maximum yield u_{MMSY} is ~0.45, and total community biomass B_{MMSY} equilibrates at ~35% of unfished biomass (Fig. 2). Overfishing occurs when u exceeds u_{MMSY}, whereas rebuilding requires reducing exploitation below u_{MMSY}. An increasing exploitation rate causes a monotonic decline in total biomass and average body size, and an increasing proportion of species is predicted to collapse (Fig. 2). We used 10% of unfished biomass as a definition for collapse. At such low abundance, recruitment may be severely limited, and species may cease to play a substantial ecological role. This model suggests that a wide range of exploitation rates ($0.25 < u < 0.6$) yield ≥90% of maximum catch but with very different ecosystem consequences: whereas at $u = 0.6$ almost half of the species are predicted to collapse, reducing exploitation rates to $u = 0.25$ is predicted to rebuild total biomass, increase average body size, and strongly reduce species collapses with little loss in long-term yield (Fig. 2). In addition to reconciling fishery and conservation objectives, setting exploitation rate below u_{MMSY} reduces the cost of fishing and increases profit margins over the long term (*14*). This simple model does not incorporate fishing selectivity; however, in practice the proportion of collapsed species could be reduced further by increasing selectivity through improved gear technology (*15*), by closing areas frequented by vulnerable species, or through offering incentives to improve targeting practices (*16*). Such strategies allow for protection of vulnerable or collapsed species, while allowing for more intense exploitation of others.

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These results suggest that there is a range of exploitation rates that achieve high yields and maintain most species. To test whether current fisheries fall within this range, we evaluated trends in 10 large marine ecosystems for which both ecosystem models and stock assessments were available (*10*). Figure 3A shows exploitation rate and biomass trajectories derived from 4 to 20 assessed fish or invertebrate stocks per ecosystem. These stocks typically represent most of the catch, and we assumed that trends in their exploitation rates represent the community as a whole. Ecosystem models were used to calculate u_{MMSY} (light blue bars) and the exploitation rate at which less than 10% of the fished species are predicted to be collapsed ($u_{conserve}$, dark blue bars). Across the 10 examined ecosystems, MMSY was predicted at multispecies exploitation rates of u_{MMSY} = 0.05 to 0.28 (mean of 0.16), whereas avoiding 10% collapse rates required much lower exploitation rates of $u_{conserve}$ = 0.02 to 0.05 (mean of 0.04).

Up to the 1990s, assessed species in 6 of the 10 ecosystems had exploitation rates substantially higher than those predicted to produce MMSY (Fig. 3A). Only the eastern Bering Sea has been consistently managed below that threshold. Since the 1990s, Iceland, Newfoundland-Labrador, the Northeast U.S. Shelf, the Southeast Australian Shelf, and California Current ecosystems have shown substantial declines in fishing pressure such that they are now at or below the modeled u_{MMSY}. However, only in the California Current and in New Zealand are current exploitation rates predicted to achieve a conservation target of less than 10% of stocks collapsed (Fig. 3A). Declining exploitation rates have contributed to the rebuilding of some depleted stocks, whereas others remain at low abundance. Averaged across all assessed species, biomass is still well below B_{MSY} in most regions. However, biomass has recently been increasing above the long-term average in Iceland, the Northeast U.S. Shelf, and the California Current, while remaining relatively stable or decreasing elsewhere (Fig. 3A).

Scientific stock assessments. Stock assessments quantify the population status (abundance, length, and age structure) of targeted fish or invertebrate stocks. We explored the status of 166 stocks worldwide for which we were able to obtain estimates of current biomass and exploitation rate (Fig. 3B). For about two-thirds of the examined stocks (63%), biomass (B) has dropped below the traditional single-species management target of MSY, that is, $B < B_{MSY}$. About half of those stocks (28% of total) have exploitation rates that would allow for rebuilding to B_{MSY}, that is, $u < u_{MSY}$, whereas overfishing continues in the remainder ($u > u_{MSY}$ in 35% of all stocks). Another 37% of assessed stocks have either not fallen below B_{MSY} or have recovered from previous depletion; most stocks in this category (77%) are in the Pacific. The weight of the evidence, as shown by the kernel density plot in Fig. 3B, indicates that most assessed stocks have

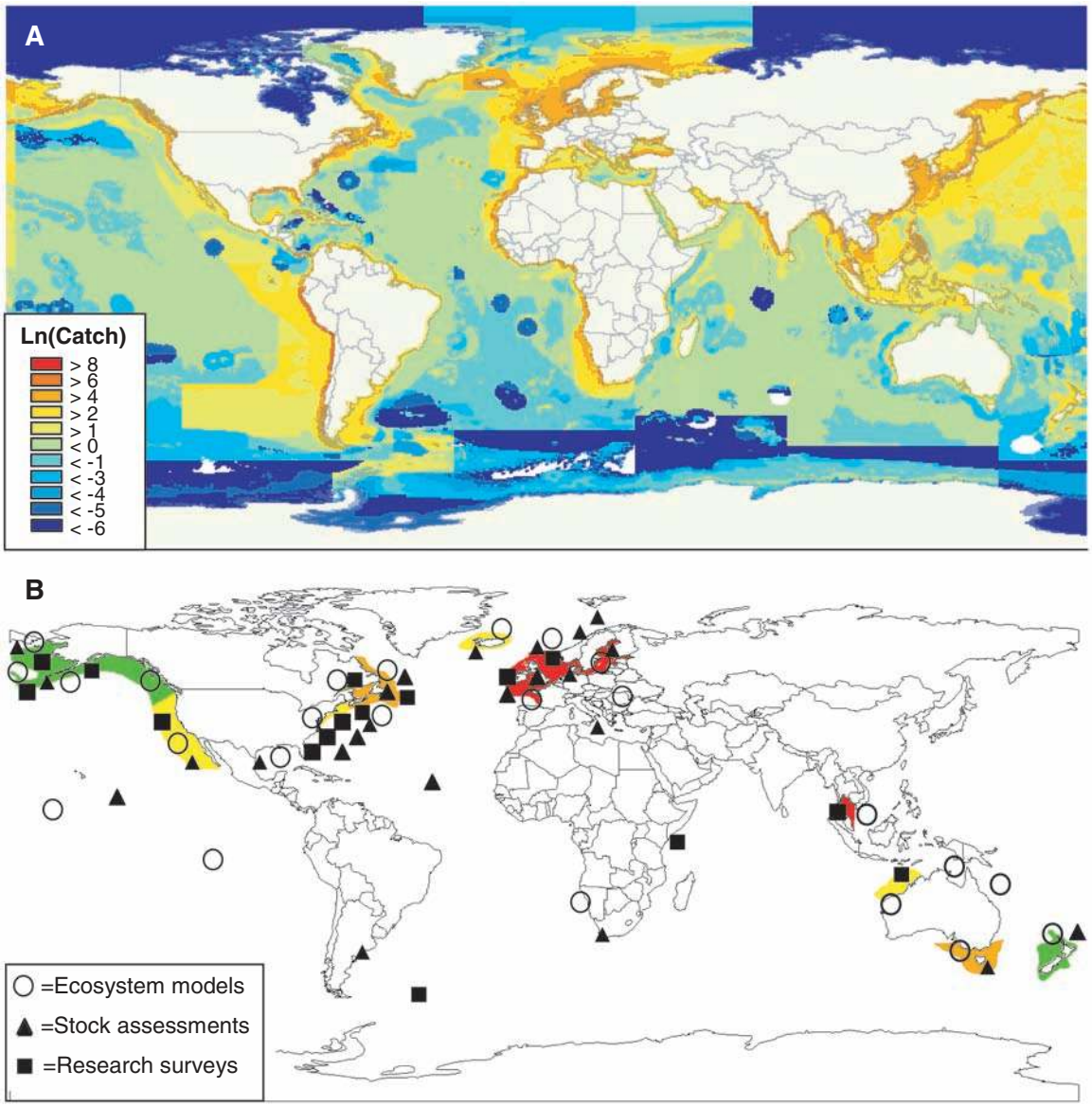

Fig. 1. Data sources used to evaluate global fisheries. (**A**) Global catch data; colors refer to the natural logarithm of the average reported catch (metric ton km^{-2} year^{-1}) from 1950 to 2004). (**B**) Other data: Stock assessments quantify the status of exploited populations; research trawl surveys are used to estimate fish community trends; ecosystem models are used to assess responses to fishing. Ecosystems that were analyzed in some detail are highlighted in green (not overfished), yellow (low exploitation rate, biomass rebuilding from overfishing), orange (low to moderate exploitation rate, not yet rebuilding), or red (high exploitation rate).

fallen below the biomass that supports maximum yield ($B < B_{MSY}$) but have the potential to recover, where low exploitation rates ($u < u_{MSY}$) are maintained. Note that most stock assessments come from intensely managed fisheries in developed countries, and therefore our results may not apply to stocks in many developing countries, which are often not assessed but fished at high exploitation rates and low biomass. Full results are provided in table S2.

When we combined the biomass estimates of stocks assessed since 1977 (n = 144, Fig. 4A), we observed an 11% decline in total biomass. This trend is mostly driven by declines in pelagic (mid-water) species, whereas large declines in demersal (bottom-associated) fish stocks in the North Atlantic were offset by an increase in demersal biomass in the North Pacific after 1977. This shows how a global average can mask considerable regional variation. Although some ecosystems showed relative stability (e.g., the eastern Bering Sea, Fig. 4B), some experienced a collapse of biomass (e.g., eastern Canada, Fig. 4C), whereas others indicated rebuilding of some dominant target species (e.g., Northeast U.S. Shelf, Fig. 4D). These regional examples illustrate different stages of exploitation and rebuilding.

Research trawl surveys. The best sources of information to assess the state of fished communities are repeated scientific surveys that include both target and nontarget species. We analyzed research trawl survey data from 19 ecosystems where such data were available (see Fig. 1B for locations and fig. S2 and table S3 for full data set). We found that community trends averaged across all surveys (Fig. 4E) were broadly similar to the combined biomass trends seen in the recent assessments (Fig. 4A), with similar signatures of stability (Fig. 4F), collapse (Fig. 4G), and recovery (Fig. 4H) in selected regional ecosystems. Few of these surveys, however, reached back to the beginning of large-scale industrial exploitation in the 1950s and early 1960s. Where they did, for example, in the Gulf of Thailand and in Newfoundland, they revealed a rapid decline in total biomass within the first 15 to 20 years of fishing (fig. S2) as predicted by ecosystem models (Fig. 2). These declines were typically most pronounced for large predators such as gadoids (codfishes) and elasmobranchs (sharks and rays). Subsequent to the initial decline, total biomass and community composition have often remained relatively stable (fig. S2), although there may be substantial species turnover and collapses of individual stocks (see below). Across all surveys combined (*10*), we documented a 32% decline in total biomass, a 56% decline in large demersal fish biomass (species ≥90 cm maximum length), 8% for medium-sized demersals (30 to 90 cm), and 1% for small demersals (≤30 cm), whereas invertebrates increased by 23% and pelagic species by 143% (Fig. 4E). Increases are likely due to prey release from demersal predators (*17*, *18*).

The trawl surveys also revealed changes in size structure that are consistent with model predictions: average maximum size (L_{max}) declined by 22% since 1959 when all communities were included (Fig. 4M). However, there were contrasting trends among our focal regions: L_{max} changed little in the eastern Bering Sea over the surveyed time period (Fig. 4N), dropped sharply in the southern Gulf of St. Lawrence, eastern Canada (Fig. 4O), as large demersal stocks collapsed, and increased because of rebuilding of large demersals (particularly haddock) on Georges Bank, Northeast U.S. Shelf (Fig. 4P). These trends included both target and nontarget species and show how changes in exploitation rates affect the broader community. Published analyses of the Gulf of St. Lawrence and adjacent areas in eastern Canada demonstrate that these community shifts involved large changes in predation regimes, leading to ecological surprises such as predator-prey reversals (*19*), trophic cascades (*17*), and the projected local extinction of formerly dominant species (*20*). Research on the Georges Bank closed area (*21*) and in marine protected areas worldwide (*22*) has shown how some of these changes may reverse when predatory fish are allowed to recover. This reveals top-down interactions cascading from fishers to predators and their multiple prey species as important structuring forces that affect community patterns of depletion and recovery (*18*).

Global fisheries catches. The benefits and costs involved in rebuilding depleted fisheries are demonstrated by an analysis of catch data. Global

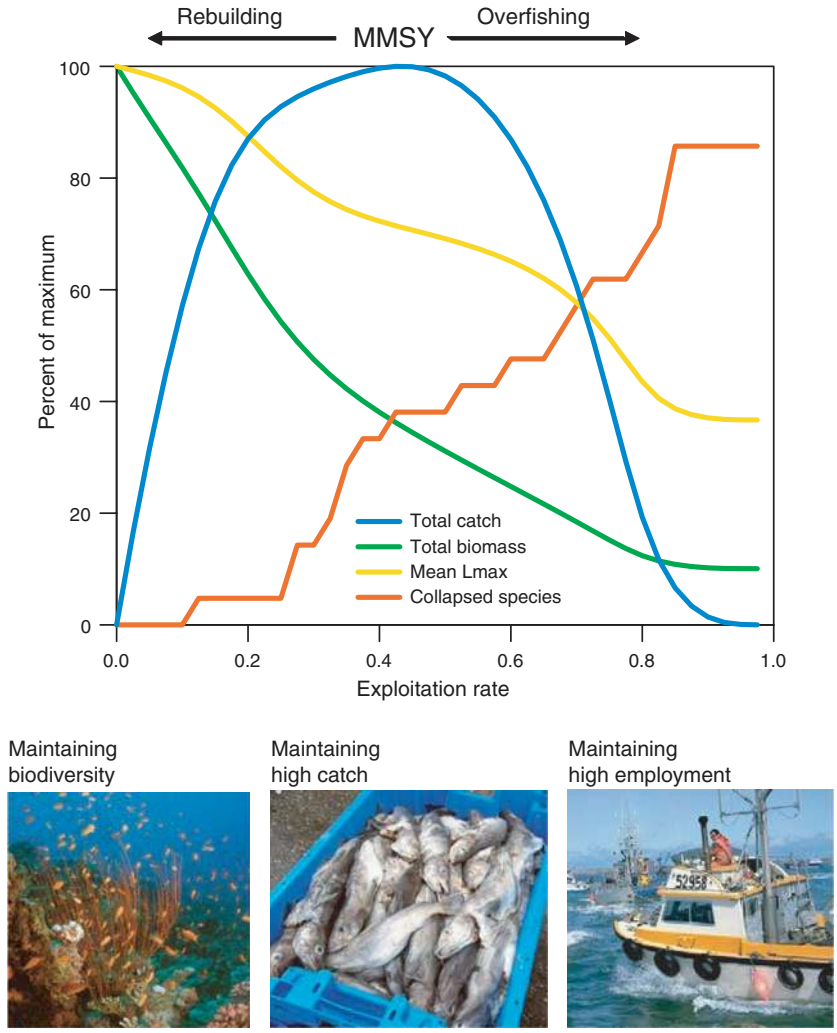

Fig. 2. Effects of increasing exploitation rate on a model fish community. Exploitation rate is the proportion of available fish biomass caught in each year. Mean L_{max} refers to the average maximum length that species in the community can attain. Collapsed species are those for which stock biomass has declined to less than 10% of their unfished biomass. This size-structured model was parameterized for 19 target and 2 nontarget species in the Georges Bank fish community (*13*). It includes size-dependent growth, maturation, predation, and fishing. Rebuilding can occur to the left, overfishing to the right, of the point of maximum catch. Three key objectives that inform current management are highlighted: biodiversity is maintained at low exploitation rate, maximum catch is maintained at intermediate exploitation rate, and high employment is often maintained at intermediate to high exploitation rate, because of the high fishing effort required.

catches have increased ~fivefold since 1950 as total biomass has been fished down (Fig. 4, A and E) then reached a plateau at ~80 million tons in the late 1980s (Fig. 4I). Catch composition with respect to the major species groups has remained relatively stable over time, with the exception of large demersal fishes, which have declined from 23 to 10% of total catch since 1950. Composition with respect to individual species, however, has fluctuated more widely owing to stock collapses (*3*) and expansion to new fisheries (*6*). Individual regions showed very different catch composition and trends, with large- and medium-sized demersal fish being historically dominant in the North Atlantic and North Pacific, small demersals being important in many tropical areas, and pelagic fish dominating the catch from oceanic and coastal upwelling systems (fig. S3). Among our focal regions, the eastern Bering Sea showed a high and stable proportion of large demersal fish (Fig. 4J), the Gulf of St. Lawrence displayed a collapse of the demersal catch and a replacement with small pelagic and invertebrate species (Fig. 4K), and Georges Bank (Fig. 4L) showed a large reduction in catch associated first with declining stocks and then with rebuilding efforts. These examples illustrate that the decline and rebuilding of fished stocks can incur significant costs because of lost catch, whereas sustained management for lower exploitation rates may promote greater stability with respect to both biomass and catches. Part of this stability may arise from the diversity of discrete populations and species that are more likely to persist in fisheries with low exploitation rates (*3*, *23*).

Trends in species collapses. Theory suggests that increases in fishing pressure, even at levels below MMSY, cause an increasing number of target and non-target species to collapse (Fig. 2). Reductions in fishing pressure are predicted to reverse this trajectory, at least partially. By using biomass data from stock assessments compared to estimates of unfished biomass (B_0) (*10*), we found an increasing trend of stock collapses over time, such that 14% of assessed stocks were collapsed in 2007, that is, $B/B_0 < 0.1$ (Fig. 4M). This estimate is in the same range as figures provided by the United Nations Food and Agriculture Organization (FAO), which estimated that 19% of stocks were overexploited and 9% depleted or recovering from depletion in 2007 (*24*). Collapse trends vary substantially by region: The eastern Bering Sea had few assessed fish stocks collapsed (Fig. 4N), whereas collapses strongly increased to more than 60% of assessed stocks in eastern Canada (Fig. 4O) and more than 25% on the Northeast U.S. Shelf (Fig. 4P).

It appears that recent rebuilding efforts, although successful in reducing exploitation rates in several ecosystems (Fig. 3A), have not yet reversed a general trend of increasing depletion of individual stocks (Fig. 4M). This matches the model-derived prediction that reduction of exploitation rate to the level that produces MMSY will still keep a number of vulnerable species collapsed (Fig. 2). Rebuilding these collapsed stocks may require trading off short-term yields for conservation benefits or, alternatively, more selective targeting of species that can sustain current levels of fishing pressure while protecting others from overexploitation.

Small-scale fisheries. Fish or invertebrate stocks that are scientifically assessed (n = 177 in our analysis) or appear in research trawl surveys (n = 1309 taxa-by-survey combinations in fig. S2) constitute only a fraction of fisheries worldwide, which is an important caveat to the above discussion. Moreover they represent a nonrandom sample dominated by valuable industrial fisheries with some form of management in developed countries. The information on other fisheries, particularly small-scale artisanal and recreational fisheries is scarcer, less accessible, and more difficult to interpret. This is because small-scale fisheries are harder to track, with 12 million fishers compared with 0.5 million in industrialized

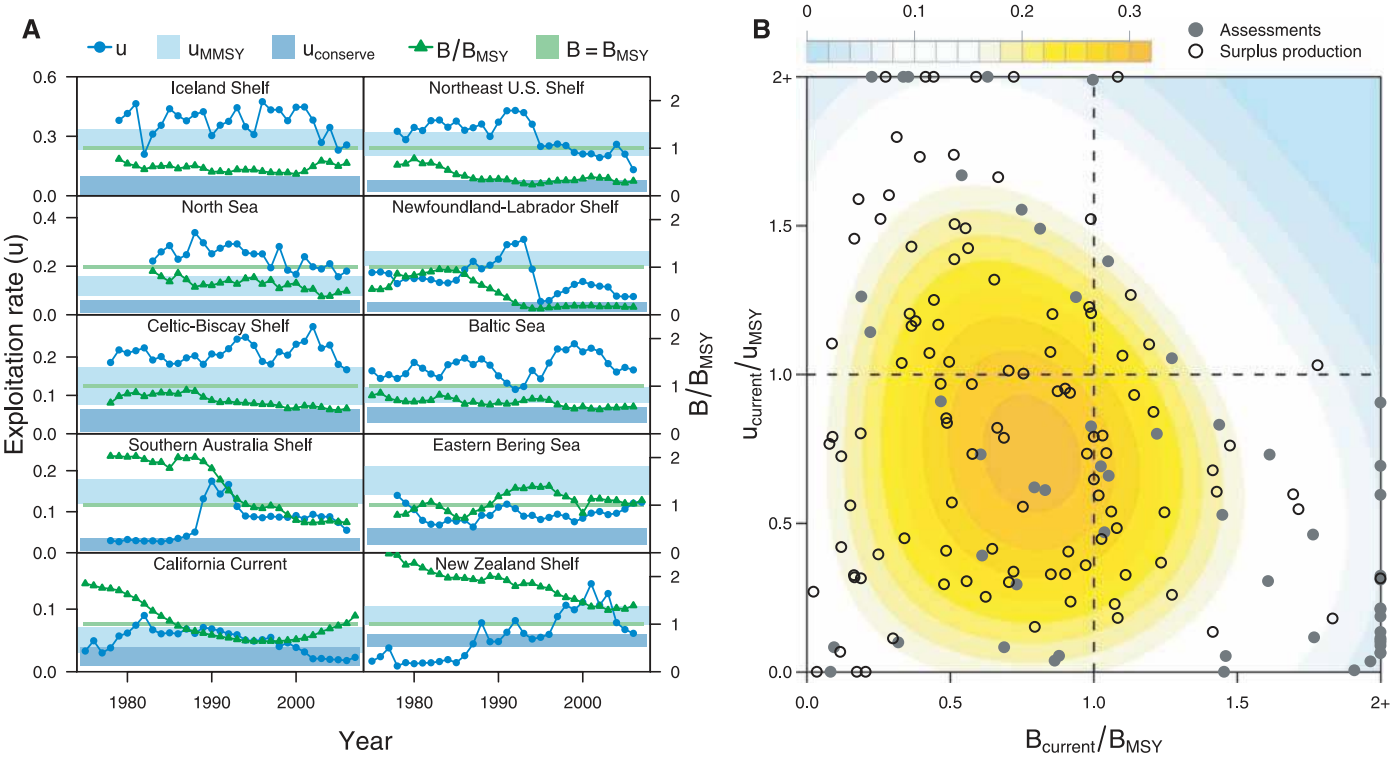

Fig. 3. Exploitation rate and biomass in large marine ecosystems and individual stocks. (**A**) Time trends of biomass (green triangles) are shown relative to the B_{MSY} (green band), exploitation rates (blue circles) relative to the u_{MMSY} (light blue band), and a hypothetical conservation objective at which less than 10% of species are collapsed ($u_{conserve}$, dark blue band). In each ecosystem, stock assessments were used to calculated average biomass relative to B_{MSY} and exploitation rate (total catch divided by total biomass) for assessed species. Reference points were calculated by using published ecosystem models; the width of the bands represents estimated uncertainty (*10*). (**B**) Current exploitation rate versus biomass for 166 individual stocks. Data are scaled relative to B_{MSY} and the exploitation rate (u_{MSY}) that allows for maximum sustainable yield. Colors indicate probability of occurrence as revealed by a kernel density smoothing function. Gray circles indicate that B_{MSY} and u_{MSY} estimates were obtained directly from assessments; open circles indicate that they were estimated from surplus production models (*10*).

fisheries (*25*), and assessments or survey data are often lacking. Small-scale fisheries catches are also poorly reported; the best global estimate is about 21 million tons in 2000 (*25*). Conventional management tools used for industrial fisheries are generally unenforceable in small-scale fisheries when implemented in a top-down manner. More successful forms of governance have involved local communities in a co-management arrangement with government or nongovernmental organizations (*26*). An example is the rebuilding of depleted fish stocks on Kenyan coral reefs (Fig. 5A). A network of closed areas and the exclusion of highly unselective beach seines were implemented in cooperation with local communities and led to a recovery of the biomass and size of available fish (*27*). This translated into steep increases in fishers' incomes, particularly in

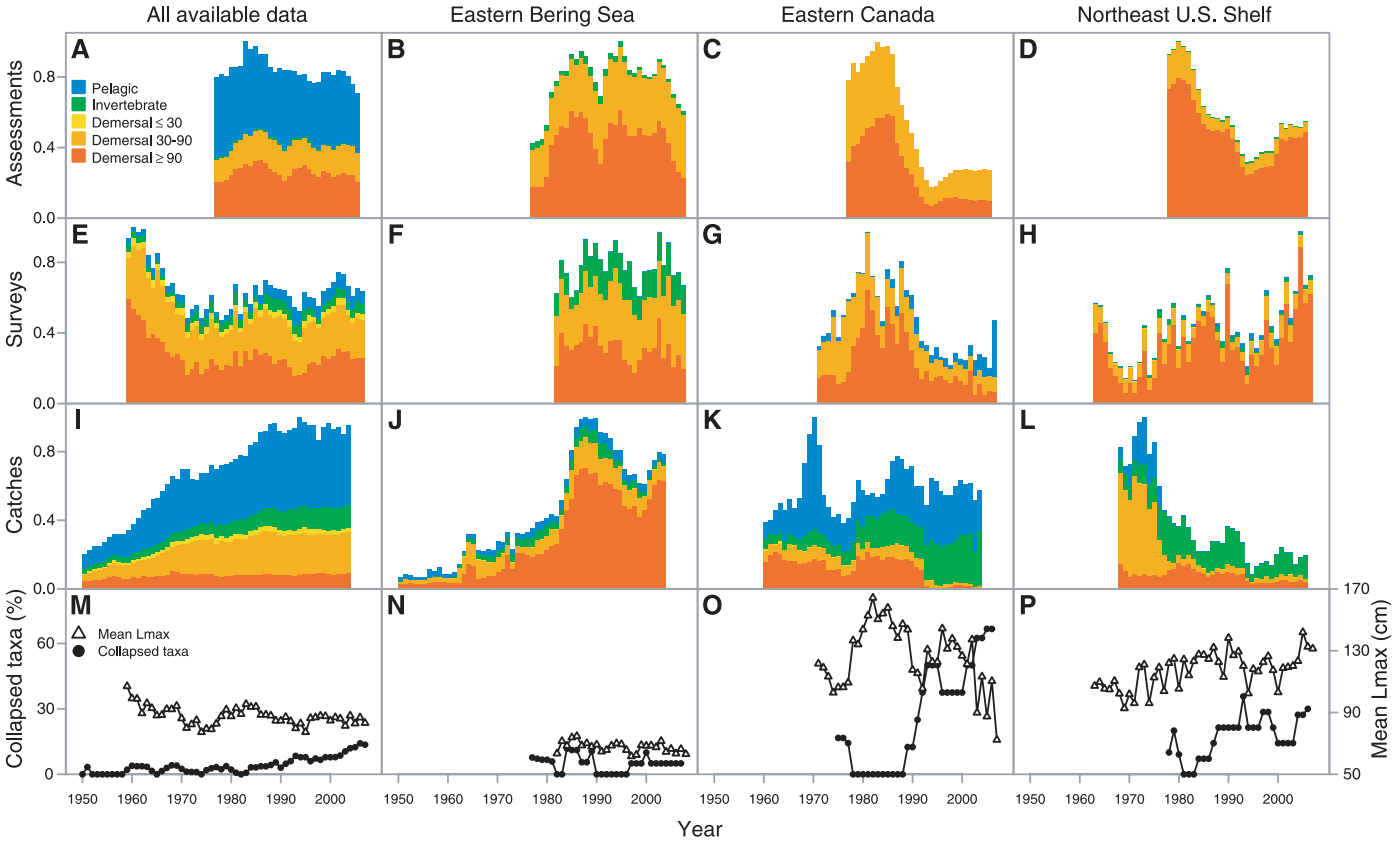

Fig. 4. Global and regional trends in fished ecosystems. Biomass trends computed from stock assessments (**A** to **D**), research surveys (**E** to **H**), as well as total catches (**I** to **L**) are depicted. Trends in the number of collapsed taxa (**M** to **P**, solid circles) were estimated from assessments, and changes in the average maximum size, L_{max} (M to P, open circles), were calculated from survey data (*10*). All data are scaled relative to the time series maximum. (G) and (K) represent the Southern Gulf of St. Lawrence (eastern Canada); (H) and (L), Georges Bank (Northeast U.S. Shelf) only. Collapsed taxa are defined as those where biomass declined to <10% of their unfished biomass. Colors refer to different species groups (demersal fish are split into small, medium, and large species based on the maximum length they can attain).

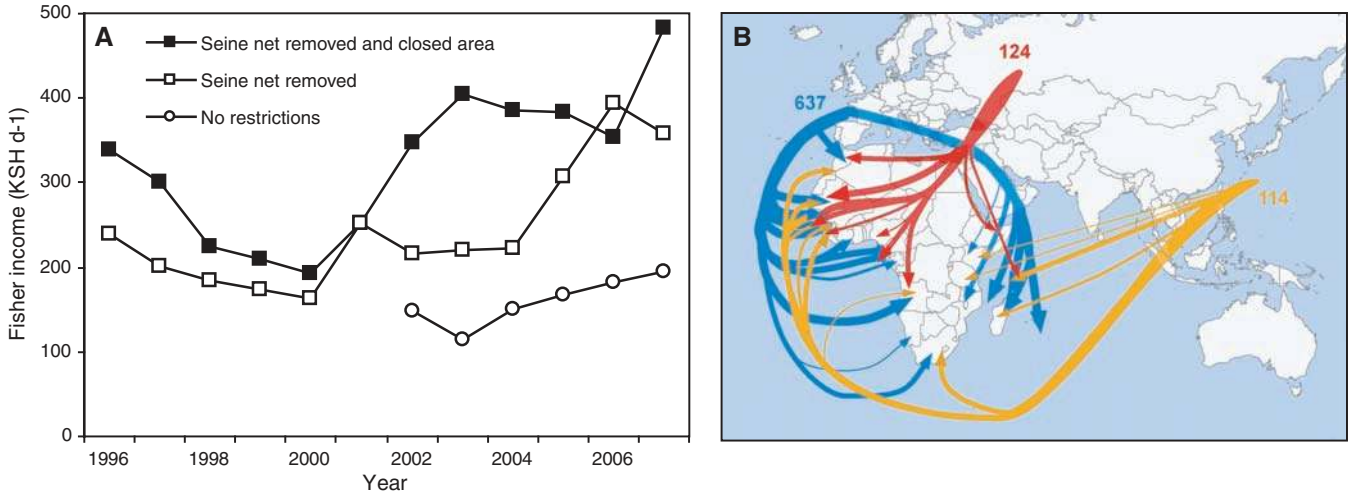

Fig. 5. Problems and solutions for small-scale fisheries. (**A**) Rebuilding of Kenyan small-scale fisheries through gear restrictions and closed area management. Updated, after (*27*). (**B**) Movement of fishing effort from developed nations to Africa in the 1990s. Data indicate total access years in distant-water fishing agreements. Updated, after (*39*).

Table 1. Management tools for rebuilding fisheries. Symbols indicate the contributions of a range of management tools to achieving reductions in exploitation rate: + tool contributed, ++ an important tool, or +++ an essential tool. Note that these examples are for industrialized fisheries, except Kenya, Chile, and Mexico. Ratings were supplied and checked by local experts.

Region	Gear restrictions	Capacity reduced	Total allowable catch reduced	Total fishing effort reduced	Closed areas	Catch shares	Fisheries certification	Community co-management
Bering Sea, Gulf of Alaska	+	++	+++		++	+++	+	+
California Current	+	++	+++		+++			
Northeast U.S. Shelf	+	++		+++	++			
North Sea, Celtic-Biscay	+	+	+++	++	+	+		+
Iceland	+	+	+++			+++		
Southeast Australian Shelf	+	+	+++		++	+++	+	
Northwest Australian Shelf	++				++			
New Zealand	+	+	+++			+++	+	
Kenya (Artisanal)	++				++			+++
Chile and Mexico (Artisanal)	+++				+			+++
Count	10	7	6	2	8	5	3	4
Total score	14	10	18	5	15	13	3	8

regions that had both closed areas and gear restrictions in place (Fig. 5A). Other examples of successful rebuilding come from Latin America, particularly Chile and Mexico, where open-access fisheries for valuable invertebrates were transformed by the establishment of spatial management units that had exclusive access by local fishing organizations (*26*). Despite these successes, rebuilding small-scale fisheries remains a significant challenge in developing countries where most fishers do not have access to alternative sources of food, income, and employment.

Tools for rebuilding. Management actions in a few ecosystems have prevented overfishing or, more commonly, reduced exploitation rates after a period of overfishing (Figs. 3 to 5). Diverse management tools have helped to achieve reductions in exploitation rates (Table 1). The most commonly used tools overall are gear restrictions, closed areas, and a reduction of fishing capacity, followed by reductions in total allowable catch and catch shares. Reductions in fishing capacity and allowable catch directly reduce the exploitation rate of target species by limiting catches. Gear modifications may be used to increase selectivity and reduce by-catch of nontarget species. Closed areas are either fully protected marine reserves (as in the Kenyan example discussed above) or are designed to exclude specific fisheries from certain areas. They can initiate recovery by providing refuge for overfished stocks (*21*, *28*), restoring community structure (*22*) and biodiversity (*3*), protecting important habitat features, and increasing ecosystem resilience (*29*). Assigning dedicated access privileges, such as catch shares or territorial fishing rights, to individual fishers or fishing communities has often provided economic incentives to reduce effort and exploitation rate (*30*) and may also improve compliance and participation in the management process (*31*). Likewise, the certification of sustainable fisheries is increasingly used as an incentive for improved management practices. Realigning economic incentives with resource conservation (rather than overexploitation) is increasingly recognized as a critical component of successful rebuilding efforts (*8*).

We emphasize that the feasibility and value of different management tools depends heavily on local characteristics of the fisheries, ecosystem, and governance system. For example, the most important element of small-scale fisheries success has been community-based management (Table 1), in which local communities develop context-dependent solutions for matching exploitation rates to the productivity of local resources (*26*). A combination of diverse tools, such as catch restrictions, gear modifications, and closed areas, is typically required to meet both fisheries and conservation objectives.

Here we have only identified the proximate tools, not the ultimate socioeconomic drivers that have enabled some regions to prevent or reduce overfishing while others remained overexploited. Yet it is generally evident that good local governance, enforcement, and compliance form the very basis for conservation and rebuilding efforts (*32*). Legislation that makes overexploitation illegal and specifies unambiguous control rules and rebuilding targets has also been critically important, for example, in the United States (*8*, *28*).

Most rebuilding efforts only begin after there is drastic and undeniable evidence of overexploitation. The inherent uncertainty in fisheries, however, requires that agencies act before it comes to that stage (*33*); this is especially true in light of accelerating global change (*34*). We found that only Alaska and New Zealand seemed to have acted with such foresight, whereas other regions experienced systemic overexploitation. The data that we have compiled cannot resolve why inherently complex fish-fisher-management systems (*35*) behaved differently in these cases; possible factors are a combination of abundant resources and low human population, slow development of domestic fisheries, and little interference from international fleets. It would be an important next step to dissect the underlying socioeconomic and ecological variables that enabled some regions to conserve, restore, and rebuild marine resources.

Problems for rebuilding. Despite local successes, it has also become evident that rebuilding efforts can encounter significant problems and

short-term costs. On a regional scale, the reduction of quotas, fishing effort, and overcapacity eliminates jobs, at least in the short term. Initial losses may create strong resistance from fisheries-dependent communities through the political process. For instance in the United States, where 67 overfished stocks have rebuilding plans, 45% of those were still being overfished in 2006, whereas only 3 stocks had been rebuilt at that time (*36*). This problem is exacerbated by the fact that the recovery of depleted stocks can take years or even decades (*28*, *37*), and during this time catches may be dramatically reduced (e.g., Fig. 4L). Furthermore, government subsidies often promote overfishing and overcapacity and need to be reduced against the interests of those who receive them (*38*). Lastly, there is the problem of unreported and illegal fishing, which can seriously undermine rebuilding efforts (*11*). Illegal and unreported catches vary between regions, ranging between an estimated 3% of total catch in the Northeast Pacific to 37% in the East Central Atlantic, with a global average of 18% in 2000–2003 (*11*).

On a global scale, a key problem for rebuilding is the movement of fishing effort from industrialized countries to the developing world (Fig. 5B). This north-south redistribution of fisheries has been accelerating since the 1960s (*39*) and could in part be a perverse side effect of efforts to restore depleted fisheries in the developed world, as some fishing effort is displaced to countries with weaker laws and enforcement capacity. The situation is particularly well documented for West Africa (*39*) and more recently East Africa, where local fisheries have seen increasing competition from foreign fleets operating under national access agreements (Fig. 5B) and where illegal and unreported catches are higher than anywhere else (*11*). Almost all of the fish caught by foreign fleets is consumed in industrialized countries and may threaten regional food security (*39*) and biodiversity (*40*) in the developing world. Clearly, more global oversight is needed to ensure that rebuilding efforts in some regions do not cause problems elsewhere. For example, fishing vessels removed in effort-reduction schemes would ideally be prohibited from migrating to other regions and exacerbating existing problems with overcapacity and overexploitation.

Open questions. Rebuilding efforts raise a number of scientific questions. Recovery of depleted stocks is still a poorly understood process, particularly for demersal species (*37*). It is potentially constrained by the magnitude of previous decline (*37*), the loss of biodiversity (*3*, *23*), species life histories (*37*), species interactions (*17*, *18*, *20*), and climate (*28*, *34*). Yet, many examples of recovery exist, both in protected areas (*3*, *21*, *22*) and in large-scale ecosystems where exploitation was substantially reduced (Fig. 3A). A better understanding of how to predict and better manage for recovery will require insight into the resilience and productivity of individual populations and their communities. This could be gained by more widespread spatial experimentation, involving proper controls, good monitoring, and adaptive management. Some of the most spectacular rebuilding efforts, such as those undertaken in California (*41*), the northeast United States (*21*), and northwest Australia (*42*), have involved bold experimentation with closed areas, gear and effort restrictions, and new approaches to catch allocation and enforcement. Science has a key role to play in guiding such policies, analyzing the effects of changes in management and advancing toward more general rules for rebuilding.

A second area of inquiry relates to the question of how to avoid contentious trade-offs between allowable catch and the conservation of vulnerable or collapsed species. Recovering these species while maintaining global catches may be possible through improved gear technology and a much more widespread use of ocean zoning into areas that are managed for fisheries benefits and others managed for species and habitat conservation. Designing appropriate incentives for fishers to avoid the catch of threatened species, for example, through tradable catch and by-catch quotas, has yielded good results in some regions (*16*). Temporary area closures can also be effective but require detailed mapping of the distribution of depleted populations and their habitats.

Conclusions. Marine ecosystems are currently subjected to a range of exploitation rates, resulting in a mosaic of stable, declining, collapsed, and rebuilding fish stocks and ecosystems. Management actions have achieved measurable reductions in exploitation rates in some regions, but a significant fraction of stocks will remain collapsed unless there are further reductions in exploitation rates. Unfortunately, effective controls on exploitation rates are still lacking in vast areas of the ocean, including those beyond national jurisdiction (*6*, *8*, *32*). Ecosystems examined in this paper account for less than a quarter of world fisheries area and catch, and lightly to moderately fished and rebuilding ecosystems (green and yellow areas in Fig. 1B) comprise less than half of those. They may best be interpreted as large-scale restoration experiments that demonstrate opportunities for successfully rebuilding marine resources elsewhere. Similar trajectories of recovery have been documented in protected areas around the world (*3*, *21*, *22*), which currently cover less than 1% of ocean area. Taken together, these examples provide hope that despite a long history of overexploitation (*1*, *2*) marine ecosystems can still recover if exploitation rates are reduced substantially. In fisheries science, there is a growing consensus that the exploitation rate that achieves maximum sustainable yield (u_{MSY}) should be reinterpreted as an upper limit rather than a management target. This requires overall reductions in exploitation rates, which can be achieved through a range of management tools. Finding the best management tools may depend on the local context. Most often, it appears that a combination of traditional approaches (catch quotas, community management) coupled with strategically placed fishing closures, more selective fishing gear, ocean zoning, and economic incentives holds much promise for restoring marine fisheries and ecosystems. Within science, a new cooperation of fisheries scientists and conservation biologists sharing the best available data, and bridging disciplinary divisions, will help to inform and improve ecosystem management. We envision a seascape where the rebuilding, conservation, and sustainable use of marine resources become unifying themes for science, management, and society. We caution that the road to recovery is not always simple and not without short-term costs. Yet, it remains our only option for insuring fisheries and marine ecosystems against further depletion and collapse.

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Supporting Online Material
www.sciencemag.org/cgi/content/full/325/5940/578/DC1
Materials and Methods
Figs. S1 to S6
Tables S1 to S7
References and Notes

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Pre-Target Axon Sorting Establishes the Neural Map Topography

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Sensory information detected by the peripheral nervous system is represented as a topographic map in the brain. It has long been thought that the topography of the map is determined by graded positional cues that are expressed by the target. Here, we analyzed the pre-target axon sorting for olfactory map formation in mice. In olfactory sensory neurons, an axon guidance receptor, Neuropilin-1, and its repulsive ligand, Semaphorin-3A, are expressed in a complementary manner. We found that expression levels of Neuropilin-1 determined both pre-target sorting and projection sites of axons. Olfactory sensory neuron–specific knockout of Semaphorin-3A perturbed axon sorting and altered the olfactory map topography. Thus, pre-target axon sorting plays an important role in establishing the topographic order based on the relative levels of guidance molecules expressed by axons.

In the vertebrate nervous system, sensory information is spatially encoded in the brain, forming topographic maps that are fundamental for cognition and higher-order processing of sensory information (*1*, *2*). Molecular mechanisms of topographic map formation have been extensively studied in the visual system. The visual image on the retina is roughly preserved in the tectum, which receives retinal ganglion cell axons. Nearly 50 years ago, Sperry proposed the "chemoaffinity hypothesis," in which target cells present chemical cues to guide axons to their destinations (*3*). Axonal projection of retinal ganglion cells is instructed by several pairs of axon guidance molecules that demonstrate graded expression in the retina and tectum (*1*, *2*).

Olfactory information is also encoded in a topographic map formed on the olfactory bulb (OB), a part of the forebrain. In rodents, odors are detected with ~1000 types of odorant receptors (ORs) expressed in olfactory sensory neurons (OSNs) in the olfactory epithelium (*4*). Each OSN expresses only one functional OR gene (*5*, *6*). Furthermore, OSNs expressing a given type of OR converge their axons to a specific glomerulus on each glomerular map in the OB (*7*–*9*). During olfactory development, OSN axons are guided to approximate locations in the OB by the combination of dorsal-ventral patterning, based on anatomical locations of OSNs in the olfactory epithelium (*10*), and anterior-posterior patterning, regulated by OR-derived cyclic adenosine monophosphate (cAMP) signals (*11*, *12*). The glomerular arrangement along the dorsal-ventral axis appears to be determined by axon guidance molecules expressed in a graded manner along the dorsomedial-ventrolateral axis in the olfactory epithelium, such as Robo-2 (*13*) and Neuropilin-2 (*14*). Unlike dorsal-ventral positioning, anterior-posterior positioning of glomeruli is independent of positional information in the olfactory epithelium. Instead, OR-specific cAMP signals determine the expression levels of Neuropilin-1 (Nrp1) in OSN axon termini, forming a gradient of Nrp1 (*11*). Thus, the olfactory system also uses gradients of axon guidance molecules to form the topographic map.

How then do guidance molecules regulate topographic map formation? Does map formation solely depend on axon-target interaction? Topographic order emerges in axon bundles, well before they reach the target (*15*, *16*). Here, we studied the pre-target sorting of OSN axons and its role in topographic map formation in the mouse olfactory system.

Nrp1 regulates axonal projection of OSNs along the anterior-posterior axis. OR-derived cAMP signals regulate the axonal projection of OSNs along the anterior-posterior axis in the OB; low cAMP leads to anterior positioning and high cAMP leads to posterior positioning (*11*). Furthermore, the levels of Nrp1 in OSN axon termini correlated with the level of cAMP signals (*11*).

We found that the Nrp1 levels determine the glomerular positioning along the anterior-posterior axis. When Nrp1 was overexpressed in OR-I7–expressing OSNs (fig. S1), projection sites shifted posteriorly relative to the control (Fig. 1A and fig. S2). In contrast, when Nrp1 was knocked out specifically in I7 OSNs, the projection sites shifted anteriorly relative to the control (Fig. 1A and fig. S2). In the pan-OSN Nrp1 knockout, however, projection sites for I7 often split into anterior and posterior areas (fig. S3). If absolute Nrp1 levels determine glomerular positioning, all glomeruli should form in the anterior OB in the pan-OSN knockout, and the results for I7 OSNs should be the same between the I7-specific knockout and pan-OSN knockout. These results indicate that the relative Nrp1 levels among axons determine the OSN projection sites.

Pre-target axon sorting in the bundle. How do the relative levels of Nrp1 determine the anterior-posterior positioning of glomeruli in the axonal projection of OSNs? To determine where the organization occurs for the olfactory map topography, we analyzed the axon bundles of dorsal-zone (D-zone) OSNs that project to the dorsal domain (D domain) of the OB. The D domain OB comprises two regions, DI and DII; DI is represented by class I ORs, and DII is represented by class II ORs. Class I and class II ORs are phylogenetically distinct and their glomeruli are segregated in the OB (*17*). We subdivided DII into two areas on the basis of Nrp1 expression level (*18*): DII-P is the posterior portion innervated by Nrp1-high axons, and DII-A is the anterior region innervated by Nrp1-low axons. Thus, the D domain can be divided into three areas: DI, DII-A, and DII-P (Fig. 1B).

Axon bundles that project to the D-domain OB were analyzed in neonatal mice by staining serial coronal sections from the anterior olfactory epithelium through the OB. Within the bundle,

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Supporting Online Material for

Rebuilding Global Fisheries

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Rebuilding global fisheries: Supporting Online Material

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Methods and Data Sources

In the following, we describe in detail the methods and data sources supporting our analyses of (i) ecosystem models, (ii) stock assessment data, (iii) trawl survey data, (iv) catch data, and (v) fishing access data. Data sources are described in Tables S1-S3 and supporting data are provided in Tables S4-S7 and Figures S1-S6.

Ecosystem models

We compiled 37 published ecosystem models from 31 systems (Table S1, mapped in Fig. 1). These models were either ECOPATH with ECOSIM (*S1, S2*) or ATLANTIS (*S3*) models. They were selected because they were publicly available, have been properly documented and quality controlled, and covered all systems that we examined empirically using the stock assessment and trawl survey data. For each ecosystem model, we did the following:

1) For each fished group in the model the exploitation rate u (defined as the proportion of biomass that is removed per year, i.e. $u_t = C_t / B_t$ where C is the catch (or 'yield') and B is the available biomass in year t) was incrementally increased and decreased, holding u 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 exploitation rate that produced maximum sustainable yield u_{MSY} for each species.

2) The u for the fished groups in the model were set to $k \times u_{MSY}$ from step 1. Long-term runs (1000 years) were then run with k incrementing from $k = 0$ to $k = 20$. 1000 years was selected because EwE models are effectively equilibrium models and the simulations had returned to a stable state by 1000 years.

3) While predation was allowed to vary during step 1, in reality u would also vary across species through time (with changing targeting and gear use). Also, given ecosystem interactions can be complex it was felt that it would be beneficial if additional runs were done to try and further fill out the phase space of possible outcomes when fishing across an entire ecosystem. Consequently, to complement the *MSY* based exploration, a set of fisheries policy searches was

performed. The objective function used in the search combines economical, ecological and potentially social terms. The set of searches incremented 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). Levels of mandated rebuilding from base ECOPATH levels were set based on how the base ECOPATH model biomasses compared with 40% of the unfished biomass levels (taken from a long-term run of the ECOSIM model with all u set to 0). Values relative to the unfished run were used, because many of the ECOPATH models 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). Ideally social (employment) considerations would have also been included in this policy search. Unfortunately, employment information was not readily available for many of the modeled fleets and the same fleet resolution was not available in all models, so social contributions to the objective function were not considered.

4) The results from the different analyses per system (i.e. the results of steps 2 and 3) were then combined to produce aggregate plots of catch, available biomass, size, and the number of groups that have dropped below 10% of their unfished levels (defined as 'collapsed') against the system-level exploitation rate (calculated as the catch / total available biomass).

5) Using the catch-exploitation rate plot for each system, the point of MMSY was defined as the peak of the plot, with the bands of uncertainty defined as the exploitation rates producing 90% of that peak catch. Similarly the conservation reference point $u_{conserve}$ (10% of stocks collapsed) and associated uncertainty band was read off the model plots, with the band defined as the u giving 9-11% of the stocks collapsed. Where alternative parameterizations for the model existed (e.g. for the Benguela and SE Australia) all steps 1-4 and calculation of MMSY and conservation reference points were repeated and in all cases the resulting values from the alternative parameterizations fell within the uncertainty bands defined from the original plots. Where there was any difference in the uncertainty bands produced by the alternative parameterizations these were combined to give the final uncertainty band used in Fig 3A. In addition, in some cases it was necessary to combine the results of multiple models to produce results at an LME or regional scale for Fig 3A, such as the Newfoundland-Labrador Shelf LME which includes the Northern Gulf St Lawrence and Grand Banks models, and the Celtic-Biscay LME which includes the Bay of Biscay, Irish Sea, and Western English Channel models. In those cases the final bands were created using an area-based weighted average (so a model covering a smaller section of the overall area contributed less to the average).

6) An overall plot (Fig. S1) was created by averaging the results in step 4 across all ecosystems, the confidence bounds in the plot mark one standard error from the mean.

Comparing ecosystem and single-species MSY

In some cases (e.g. Georgia Strait, Southeast Australian shelf) the sum of the predicted single-species *MSY* did approximate (within 2%) the system-level catches (*MMSY*). (*S4*) found this from some systems, even when at an individual level the realized catches of species within those systems could be strongly different to the predicted single-species *MSY* and often at the expense

of top predators. However, it would be inappropriate to use this result as a "rule of thumb" to predict what *MMSY* would be for a system, as the sum of single-species *MSY* is generally a poor predictor of multi-species *MSY*. This is because it's hard to say *a priori* whether depensatory or compensatory responses to fishing will occur. These responses will lead to divergence between yields at the system level and those predicted by single-species assessments as seen in (*S5*). The sum of predicted single-species *MSY* differed from system-level *MMSY* by more than 20% in 42% of the systems and by more than 50% in 18% of the systems.

Across the modeled systems there were examples of systems for which the sum of single-species *MSY* exceeded *MMSY* (e.g. Great Barrier Reef, North Sea); however there were also cases (e.g. Central North Pacific, Bay of Biscay) where the sum of single-species *MSY* was less than *MMSY* (but at a significant cost to top and even medium level predators). Looking at system type (inshore versus shelf or open ocean; temperate versus tropical), there is no consistent pattern regarding how *MMSY* at a system level will compare with the sum of single-species *MSY*. In contrast, patterns of response are clearer at an individual group level. For individual groups, across all systems, it was found that *MMSY* is never significantly greater than *MSY* at the higher trophic levels, whereas this is often the case at lower trophic levels. In the majority (61-71%) of cases the catch, when the system was fished at $u_{MMSY,}$ was greater than the catch predicted from single-species models for groups of trophic level 1 or 2. For trophic level 3, there was an even split in the number of cases in which the catch of a group under u_{MMSY} exceeded the expected *MSY* and vice versa (i.e. *MSY* > *MMSY* for that group in 50% of cases). For the highest trophic levels (4+) in the majority of cases (66-84%) the group-level catch under u_{MMSY} is much less than *MSY*. For those cases in which *MMSY* is not less than *MSY* then they are effectively identical (differing by less than 4%).

Creating the exploitation-rate plots

To give insight into the history of the exploitation of a range of systems from around the world, reference points were taken from the ecosystem modeling analysis (see step 5 above) and compared with the history of overall exploitation rates observed (calculated from catch and biomass for assessed species) in those same systems (Fig. 3A, Table S4, see stock assessment data). The reference points were (i) u_{MMSY}: the band of exploitation rates that produced the maximum system-level catch (uncertainty associated with the models and also the shape of the curves meant there was no single peak exploitation rate, but rather a band of potential rates) and (ii) $u_{conserve}$: the band of exploitation rates where 10% of the groups in the model fell below 10% of their unfished biomass levels ($u = 0$ for all system components). This latter reference point was chosen as a hypothetical conservation objective; as there is no easily defined conservation equivalent of an *MSY* concept, the use of other levels of extirpation could be substituted.

In Fig. 3A, the ratio of biomass B to B_{MSY} was calculated for each of these systems, from corresponding units, the ratio of spawning biomass to the spawning biomass that would be present in a system producing *MSY*, but sometimes total biomass for both. Where available, these ratios were obtained from the stock assessments, otherwise B_{MSY} was calculated using the surplus production model approached outlined below.

Stock assessment data

We gathered time series of recorded catch, model-estimated biomass and fishing mortality rates, and reference points (B_{MSY} and u_{MSY}, the biomass and exploitation rate, respectively, that result in maximum sustainable yield) from published stock assessments of exploited marine fish and invertebrate stocks and from personal communications with stock assessment scientists (see Table S2). We attempted to gather the most recent available assessments for stocks around the world in which a population model was applied to estimate a time series of biomass and exploitation rates. While we compiled the majority of assessments we have identified, the data set is not complete and is evolving. Many regions are under-represented either because of analytical uncertainties associated with recent attempts to assess the stocks or because assessments for previously over-exploited stocks are highly uncertain or are no longer conducted. A prime example is that of northern cod off Newfoundland and Labrador, a stock estimated to have declined 99% between 1962 and 1992, for which the offshore segment of the stock has not been assessed since shortly after it collapsed in 1992.

Where assessment estimates of B_{MSY} and u_{MSY} (or the instantaneous fishing mortality rate F_{MSY} which was provided for some stocks) were available ($N = 41$ stocks for u_{MSY} or F_{MSY} and $N = 54$ stocks for B_{MSY}), these were used to determine stock status (Fig. 3B and Fig. 4M-P). Where these reference points were unavailable, we fit a surplus-production model to time series of annual total biomass B_t and total catch or landings (where catch was unavailable) C_t from the assessments. The surplus-production model was only applied where ≥20 years of catch or landings and biomass data were available. One hundred and sixty-six of the 239 stock assessments that we gathered either had B_{MSY} and u_{MSY} reference points provided or had ≥20 years of catch or landings and biomass data.

Overall biomass trends computed from stock assessments (Fig. 4A-D) are provided in Table S5.

Surplus-production model

Surplus production in year t, P_t, a commonly-used measure of stock productivity has been used previously (*S6, S7, S8*) where surplus production in year t, P_t, can be calculated as:

$$(1) \qquad P_t = B_{t+1} - B_t + C_t$$

where B_t is the biomass at time t and C_t is the catch at time t.

We fit a Schaefer surplus-production model, which is based on a logistic model of population growth (*S9*) to the catch and biomass time series data. The predicted surplus production in each year in the Schaefer model is given by:

$$(2) \qquad \hat{P}_t = \frac{4mB_t}{K} - 4m\left(\frac{B_t}{K}\right)^2$$

where m is the maximum sustainable yield and K is the carrying capacity or equilibrium biomass in the absence of fishing.

We estimated the model parameters (m and K) using maximum likelihood in AD Model Builder (http://admb-project.org) assuming that the residuals $\left(\varepsilon_t = P_t - \hat{P}_t\right)$ were normally distributed. For the Schaefer model, B_{MSY} is simply $0.5K$, and the harvest rate that results in maximum sustainable yield, u_{MSY}, is m/B_{MSY}. Carrying capacity was constrained to be less than twice the maximum observed biomass. Thirty-eight percent of stocks were affected by this constraint. For five Eastern Bering Sea crab stocks, only B_{MSY} reference points were available, not u_{MSY} reference points and surplus production model fits could not be obtained, therefore the total number of stocks for which we were able to obtain estimates of $B_{current}/B_{MSY}$ and $u_{current}/u_{MSY}$ was 160 (Fig. 3B).

We compared the surplus production model estimates of $B_{current}/B_{MSY}$ and $u_{current}/u_{MSY}$ to the value of these ratios obtained from the assessments for all stocks which had assessment-based reference points. After replacing values of these ratios that were greater than 2 with a value of 2 (as was done in Fig. 3B), the Pearson correlation between the harvest rate ratios from the surplus production model and the harvest rate ratios from the assessments was $r = 0.62$. The equivalent correlation for the biomass ratios was $r = 0.65$. All stocks used in this analysis and their estimated $B_{current}/B_{MSY}$ and $u_{current}/u_{MSY}$ are shown in Table S2.

Caveats on MSY-related calculations

Despite the fact that the concept of *MSY* has been prominent in the fisheries science and management literature for about 5 decades, it is not uniformly defined or estimated. Numeric values of B_{MSY} and u_{MSY} are dependent on the vulnerability of different ages or sizes of fish to fishing gear; whether B_{MSY} is defined in terms of spawning biomass, available (exploitable) biomass, or total biomass; the harvest strategy used or assumed (e.g. a constant catch strategy or a constant fishing mortality strategy); and the model or method used for estimation. The Schaefer surplus-production model used here when the management agency did not provide its own estimates of B_{MSY} assumes a symmetric relationship between sustainable yield and biomass (with B_{MSY} being half of the carrying capacity) whereas the more common result is that B_{MSY} is less than half of carrying capacity, usually in the range 25-40%. If exploitation can be delayed until several years after the onset of maturity, then this range will be even lower.

Use of asymmetric models will undoubtedly give different estimates of the MSY related ratios. This coupled with our decision to use the estimates provided with assessments in preference means that the ratio estimates may not be fully consistent. However, we believe that our analysis represents a first attempt to characterize the status of a large number of fisheries worldwide in terms of both $B_{current}/B_{MSY}$ and $u_{current}/u_{MSY}$. Detailed analysis of individual stocks will undoubtedly come to different conclusions if other models or data are used.

Finally, catch time series presented in the assessments should match time series of exploitation estimates derived from the instantaneous fishing mortality rates. In cases where these differ, for example when unaccounted discards or misreporting lead to commercial data being omitted or down-weighted in the assessment, the ratios of exploitation rates to the reference exploitation rate will also differ. In a preliminary investigation of ICES stocks, for example, differences between ratios based on exploitation estimates from catches and mortality rates were typically small, with the important exceptions of West of Scotland cod and haddock, and Irish Sea cod.

For these stocks the exploitation ratios were higher when based on converted fishing mortality rates than when using catches or landings. The differences follow from our decision to use a consistent method for describing exploitation status, but they do not affect our main conclusions at regional and global scales. However, the differences do highlight the importance of consulting the original assessments and assessment scientists when seeking information on the status of individual stocks.

Methods to obtain trends in relative biomass and exploitation rates

The biomass trends in Fig. 3A for each Large Marine Ecosystem were obtained by taking the geometric mean of the B/B_{MSY} ratios from assessed stocks in each year (Table S4). The geometric mean was preferred to an arithmetic mean since these are averages of ratios. The exploitation rate trends were obtained by calculating the ratio of total catch to total assessment biomass in each year. Three species were excluded from the trajectories of biomass and exploitation rate in Fig. 3A, these species were pelagic species with dominant catches but these catches fell mostly outside the Large Marine Ecosystems examined. The excluded species were: Pacific hake (*Merluccius productus*) for the California Current, Atlantic menhaden (*Brevoortia tyrannus*) in the Northeast U.S. Shelf, and blue whiting (*Micromesistius poutassou*) in the Celtic-Biscay Shelf.

Methods to calculate trends in collapsed taxa

To calculate trends in the proportion of total stocks collapsed (Fig. 4M-P, Table S7), we compared time series of biomass B to B_{MSY}. The proportion of collapsed stocks in any year is simply the number of collapsed stocks divided by the total number of stocks for which an estimate of biomass was available in that year. A stock was defined as collapsed in any year if the biomass in that year was less than 20% of B_{MSY}. For a population growing according to the logistic growth function, this is equivalent to 10% of carrying capacity.

Trawl survey data

We compiled data from 20 long-term research trawl surveys (Fig. S2, Table S3), from a variety of regions around the world, but dominated by the Northern Hemisphere (n = 17) and especially the Northwest Atlantic (n = 10). The surveys each spanned at least 18 years from earliest to the latest and contained at least ten annual surveys. Surveys were typically obtained directly from the agency responsible for the surveys, but also from published sources.

Each taxon in each survey was allocated to a category: invertebrate, pelagic fish and demersal fish. The demersal fish were further subdivided into small (≤30 cm), medium (30-90 cm) and large (≥90 cm) (as plotted in Fig. S2) categories based on the maximum length (L_{max}) recorded in the online database FishBase (*S10*), or SeaLifeBase (*S11*), where available. Where L_{max} was missing for a particular species, these were assumed to be the average for that genus or where that was not possible, from family or higher-level taxon. In the rare instances where all of these methods failed in obtaining L_{max} values (<1% of the total), they were obtained from a variety of grey literature and internet sites. Allocation to demersal or pelagic were based on the habitat categories in FishBase with categories of bathydemersal, benthopelagic, and reef-associated all

assumed to be "demersal", and categories of bathypelagic, pelagic-neritic and pelagic-oceanic all assumed to be "pelagic". Although the default option for benthopelagic species was to assume they were demersal, the following species of obviously pelagic nature (Atlantic herring *Clupea harengus*, American butterfish *Peprilus triacanthus*, black mackerel *Scombrolabrax heterolepis*, oxeye herring *Megalops cyprinoides*, red tailed round scad *Decapterus russelli*, and jack mackerel *Trachurus declivis*) were categorized as pelagic.

The survey data typically came from gears designed to adequately sample medium to large demersal species. While recognizing this limitation for interpreting trends in other categories, catch trends were included if deemed a reliable index by the agency that supplied the data.

Invertebrates
Invertebrates for a given survey were classified as "not recorded" if a data provider supplied finfish data only (Table S3). This was the case for: St Pierre Bank, Newfoundland; Southern Grand Banks, Newfoundland; Southern Gulf of St. Lawrence; Northern Gulf of St. Lawrence; Celtic Sea; and the North Sea IBTS surveys. In the Scotian Shelf survey data invertebrates are represented by one species (*Illex illecebrosus*). No other invertebrate species were recorded in the data provided. A separate invertebrate survey dataset for this region is only available from the year 2000, when invertebrates were consistently recorded. Given the short length of this series, it was not included in the analyses.

Demersal ≤30cm
When the data providers deemed catch trends for small demersals unreliable, "not recorded" values were assigned. This was the case for: St Pierre Bank, Newfoundland; Northern Gulf of St Lawrence; and Eastern Bering Sea surveys. In the Eastern Bering Sea survey grouped categories like "Sculpins" might have contained some component species that were small demersals, but the groups as a whole were all >30 cm. True zeroes were provided for small demersals in surveys where the biomass of small demersal species contributed to less than 2% of the total biomass sampled per year. This was the case for: URI Whale Rock and URI Fox Island surveys.

Pelagic
Similar to small demersals, pelagic species sampling is limited by the demersal nature of the sampling gear. Here again, if trends in pelagic species were deemed a reliable index by the agency that sampled them, they were provided and are included in the analyses. Pelagic species were assigned "not recorded" values because they were not provided in the following surveys: St Pierre Bank, Newfoundland and Northern Gulf of St Lawrence.

Methods to obtain overall trends in survey biomass

Overall survey biomass trends (Fig. 4E) were based on 19 of the surveys in Table S3 and are shown in Table S6. We did not include the South Georgia series in the analysis because only five commercially important species were reported in the source document and these were not representative of the ecosystem as a whole. We also excluded years where fewer than four surveys were represented in the data set.

Data standardization

The survey biomass index (biomass, biomass per unit area/tow-time, depending on the survey) of taxon *i* in survey *s* in year *t* was given by $B_{i,s,t}$. The *i* taxa (usually identified to species but occasionally only identified to genus or family) were grouped into one of $c = 5$ categories as described above (Pelagic fish, Invertebrate, Demersal fish ≤30 cm, Demersal fish 30-90 cm, Demersal fish ≥90 cm). The biomass of category *c* in survey *s* in year *t* was given by

(3) $$B_{c,s,t} = \sum_{i=1}^{n_{c,s}} B_{i,s,t}$$

where $n_{c,s}$ is the number of taxa in category *c* in survey *s*.

As the data come in a variety of different units, it was necessary to standardize before analyzing the trends. A standardization method was required that maintained the strength of the category within a survey but also allowed the categories to be combined across surveys. Assuming that the biomass indices from a given survey are lognormally distributed within a year, the index was log transformed:

(4) $$b_{c,s,t} = \ln(B_{c,s,t} + 1)$$

The data were standardized by subtracting the survey mean (on the log scale), that is:

(5) $$\delta_{c,s,t} = b_{c,s,t} - \bar{b}_s$$

where $\bar{b}_s$ is the mean across category and time of the log biomass in survey *s*.

As the overall mean of the survey is subtracted, the relative strength of each category within a survey was maintained but the data are now in a standardized format. Plots of the standardized indices by category are presented in Fig. S4.

Analysis

The goal of the analysis was to obtain an overall trend by category over time (Fig. 4E). The data were non-independent at the survey level so a hierarchical approach was adopted that accounts for within-survey correlation.

Linear Mixed Effects with continuous first order within-group correlation

A linear mixed effects model describing the trend for a given category (e.g. Pelagic) was given by:

(6) $$\delta_{s,t}^{Pelagic} = \mu_t + a_s + \varepsilon_{s,t}$$

where μ_t was the yearly fixed effect mean, a_s was the random effect deviation from μ_t by survey s, which were distributed normally $a_s \sim \mathrm{N}(0,\sigma_a^2)$, and $\varepsilon_{st} \sim \mathrm{N}(0,\sigma_\varepsilon^2)$ were the normally distributed residual errors.

The model described in Equation 6 assumes that the within-group observations are considered exchangeable. To account for the fact that longitudinal data generally have an autocorrelated structure, this basic model was extended to include autocorrelation in the residuals, i.e.

(7) $$\mathrm{Corr}(\varepsilon_{s,t_1},\varepsilon_{s,t_2}) = \varphi^{-|t_2-t_1|}$$

where ϕ was the autocorrelation coefficient.

First-order autocorrelation (AR(1) or a Markov process) occurs when adjacent years are non-independent, with the strength of the dependence decaying with increasing lag time. Note that the correlation structure depends on the time distance between the observations, not on their being strictly consecutive (usual assumption). This maintains the AR(1) correlation structure when missing data values are present. The estimated fixed effects, confidence intervals, and residuals are plotted in Figure S5.

For presentation as a stacked barplot, the standardized trends by category were converted back into positive standardized biomass units using the expected value of a lognormal distribution, including the random effects variance:

(8) $$\mathrm{E}[\Delta_t^{Pelagic}] = e^{\mu_t + (\sigma_\varepsilon^2 + \sigma_a^2)/2}$$

where $\Delta_t^{Pelagic}$ is the antilog of $\delta_t^{Pelagic}$. The bias correction for a lognormal variable X when transforming to the original scale is $\mathrm{Var(X)}/2$, here given by $\mathrm{Var}(\Delta_{s,t})/2 = (\sigma_\varepsilon^2 + \sigma_a^2)/2$, assuming the fixed effects values are constant.

Methods to obtain overall trends in $\overline{L}_{\max,t}$

Overall trends in L_{max} (Fig. 4M-P, Table S6) were based on 19 of the surveys in Table S3 and shown in Table S6. Mean maximum length $L_{\max}$ for each survey-year combination was calculated from:

(9) $$\overline{L}_{\max,s,t} = \frac{\sum_{i=1}^{n_{s,t}} L_{i,s} B_{i,s,t}}{\sum_{i=1}^{n_{s,t}} B_{i,s,t}}$$

where $L_{i,s}$ is the $L_{\max}$ for taxon *i* in survey region *s*, $B_{i,s,t}$ is the biomass estimate for taxon *i* in survey *s* in year *t* and $n_{s,t}$ is the number of taxa in survey *s* in year *t*. Trends in $\overline{L}_{\max,s,t}$ are presented in Fig. S6 (panel A).

A similar modeling framework to that used for analyzing the standardized biomass indices was implemented to obtain the combined trend of $L_{\max}$ over time. The linear mixed effects model was given by

(10) $$\overline{L}_{\max,s,t} = L_{\max,t} + b_s + \varepsilon_{s,t}$$

where $L_{\max,t}$ is the yearly average, b_s is the deviance from the yearly average by survey s with $b_s \sim \mathrm{N}(0,\sigma_b^2)$ and $\varepsilon_{st} \sim \mathrm{N}(0,\sigma_\varepsilon^2)$ were the residual errors.

The within-group correlation structure was again assumed AR(1). The average fixed effects, fitted values to each survey and residuals are plotted in Fig. S6 (panels B and C).

The South Georgia Island surveys were not included in the analyses because only five commercially exploited fish species were reported in the source document (*S12*). Analyses included the Gulf of Thailand surveys only for those years where data were separated out to the level of taxon (18 years, 1963 and 1966-1982). Analyses were based on biomass estimates which were reported for all surveys except for the two University of Rhode Island surveys (Fox Island and Whale Rock), which only reported numbers per survey tow. To avoid over-representing numerous but small taxa for these two surveys, we converted the reported values to a biomass index by multiplying the numbers for all years by the average weight for these species during the later years 1994-2005 when weight data were available. These mean weights were obtained separately for the two University of Rhode Island surveys.

Survey data analyses were conducted in the R statistical programming environment (*S13*).

Catch data

Mapped global catch-rates (tonnes km^{-2} yr^{-1}) used in Figs. 1 and 4 and summarized by LME and species group in Fig. S3 were constructed with rule-based procedures developed by the Sea Around Us project (SAUP) based at the Fisheries Centre of the University of British Columbia, Canada (*S14*) (www.seaaroundus.org, contact Reg Watson) Available fisheries data were harmonized from a wide range of sources including the Food and Agriculture Organization of the UN (FAO) and its regional bodies, the International Council for the Exploration of the Sea (ICES), the Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR), the Northwest Atlantic Fisheries Organization (NAFO), and many reconstructed national datasets (*S15*) to produce a representative database of global fisheries landings (see also http://www.seaaroundus.org/doc/saup_manual.htm#13). Using additional databases of fishing access arrangements and/or observed national fleet fishing patterns, and extensive information on the distribution and harvest patterns of commercial marine species developed by SAUP, the spatially coarse fisheries landings data records were assigned to a grid of 30-minute latitude × 30-minute longitude spatial cells from 1950 to 2004. Many subsequent associations were then made possible with this mapped data including global fishing gear use.

Fishing access data

The total years of fishing access by regions (Europe, Russia, Asia) calculated and presented for the 1990s in Fig. 5B were computed by adding the number of years a foreign country had access to EEZs of countries in Africa through a bilateral fishing agreement, as defined by (*S16*). Where the European Union (EU) entered into an agreement on behalf of its members, the total years were calculated for each member country that gained access rights under the agreement. Details of international fishing agreements were obtained from the Sea Around Us Project (www.seaaroundus.org, contact Dirk Zeller) fishing agreement database, which is based on the FAO FARISIS database corroborated and supplemented using inter-governmental and governmental sources (e.g. EUR-Lex) and other references. Where the terms of an agreement were unknown, duration of one year was used as default. Thus, the information presented here is conservative and excludes illegal access and traditional access that are not formalized through bilateral agreements.

Supporting Tables

Table S1: List of 37 ecosystem models for 31 systems and their sources used to explore multi-species *MSY*. For some systems two EwE models from different time periods were used.

System	Model type	Notes and Source(s)
Alaska Prince William Sound	EwE	(*S17*)
Aleutians	EwE	(*S18*)
Australia Darwin Harbour	EwE	(*S19*)
Baltic	EwE	Database for (*S4*)
Bay of Biscay	EwE	(*S20*)
Benguela	2 EwE	Database for (*S4*)
Black Sea	EwE	Database for (*S4*)
California Current	2 EwE and 1 Atlantis	(*S21, 22*)
Canada - Nth Gulf St Lawrence	EwE	(*S23*)
Central Nth Pacific	EwE	Database for (*S4*)
Chesapeake	EwE	Database for (*S4*)
Eastern Bering Sea	EwE	Database for (*S4*)
Eastern Tropical Pacific	EwE	Database for (*S4*)
Great Barrier Reef	EwE	(*S24*)
Georges Bank	EwE	(*S25*)
Georgia Strait	EwE	Database for (*S4*)
Gironde Estuary	EwE	(*S26*)
Grand Banks	EwE	(*S27*)
Gulf Mexico	EwE	(*S28*)
Gulf Thailand 1973	EwE	Database for (*S4*)
Irish Sea	EwE	(*S29*)
New Zealand	EwE	(*S30*)
North Sea	EwE	Database for (*S4*), (*S31*)
North West Shelf	EwE	(*S32*)
Port Phillip Bay	EwE	(*S33*)
SE Alaska 1963	EwE	(*S18*)
SE Australia	2 EwE and 1 Atlantis	(*S34, 35*)
Tampa Bay	EwE	Database for (*S4*)
West Coast Vancouver Island	EwE	Database for (*S4*)
Western English Channel	2 EwE	(*S36*)
West Florida Shelf	EwE	(*S37*)

Table S2. Summary of all stock assessments and their sources used in this analysis and their estimated ratios of current biomass to the equilibrium biomass when harvested at maximum sustainable yield ($B_{current}/B_{MSY}$) and current harvest rate (or fishing mortality rate) to the harvest rate that results in maximum sustainable yield ($u_{current}/u_{MSY}$). The reference ratios were either obtained directly from stock assessments ("Yes") or from surplus production model fits ("No"); where reference ratios could not be obtained (N/A), the stocks were not plotted in Fig. 3B, but were included in the other analyses.

Large Marine Ecosystem	**Scientific name**	**Fisheries stock**	**Current year**	$B_{current}/B_{MSY}$	$u_{current}/u_{MSY}$	**From assessment?**	**Source**
Atlantic High Seas	*Thunnus alalunga*	Albacore tuna North Atlantic	2005	0.81	1.49	Yes	*(S38)*
Atlantic High Seas	*Thunnus thynnus*	Bluefin tuna Eastern Atlantic	2007	0.34	9.38	Yes	*(S39)*
Baltic Sea	*Gadus morhua*	Atlantic cod Baltic Areas 22 and 24	2006	0.36	1.43	No	*(S40)*
Baltic Sea	*Gadus morhua*	Atlantic cod Baltic Areas 25-32	2006	0.16	1.46	No	*(S40)*
Baltic Sea	*Clupea harengus*	Atlantic herring ICES 25-32	2006	0.69	0.79	No	*(S40)*
Baltic Sea	*Clupea harengus*	Atlantic herring ICES 30	2006	1.19	1.10	No	*(S40)*
Baltic Sea	*Clupea harengus*	Atlantic herring ICES 31	2006	0.29	1.60	No	*(S40)*
Baltic Sea	*Clupea harengus*	Atlantic herring ICES 28	2006	1.21	0.87	No	*(S40)*
Baltic Sea	*Sprattus sprattus*	Sprat ICES Baltic Areas 22-32	2006	1.13	1.27	No	*(S40)*
Barents Sea	*Gadus morhua*	Atlantic cod Northeast Arctic	2006	0.56	1.42	No	*(S41)*
Barents Sea	*Mallotus villosus*	Capelin Barents Sea	2006	0.17	0.00	No	*(S41)*
Barents Sea	*Reinhardtius hippoglossoides*	Greenland halibut Northeast Arctic	2006	0.36	1.20	No	*(S41)*
Barents Sea	*Melanogrammus aeglefinus*	Haddock Northeast Arctic	2006	1.10	1.06	No	*(S41)*
Barents Sea	*Pollachius virens*	Saithe Northeast Arctic	2006	1.70	0.60	No	*(S41)*
Benguela Current	*Engraulis encrasicolus*	Anchovy South Africa	2006	0.97	0.36	No	*(S42)*
Benguela Current	*Trachurus capensis*	Cape horse mackerel South Africa South Coast	2007	1.47	0.76	No	*(S43)*
Benguela Current	*Sardinops sagax*	Sardine South Africa	2006	0.75	0.55	No	*(S44)*
Benguela Current	*Palinurus gilchristi*	Southern spiny lobster South Africa South Coast	2008	0.51	1.50	No	*(S45)*
California Current	*Reinhardtius stomias*	Arrowtooth flounder Pacific Coast	2007	3.81	0.21	Yes	*(S46)*
California Current	*Sebastes melanops*	Black rockfish Northern Pacific Coast	2006	1.45	0.53	Yes	*(S47)*
California Current	*Sebastes melanops*	Black rockfish Southern Pacific Coast	2007	2.23	0.19	Yes	*(S48)*
California Current	*Sebastes mystinus*	Blue rockfish California	2007	0.75	1.55	Yes	*(S49)*
California Current	*Sebastes paucispinis*	Bocaccio Southern Pacific Coast	2006	0.32	0.10	Yes	*(S50)*
California Current	*Sebastes pinniger*	Canary rockfish Pacific Coast	2007	0.86	0.04	Yes	*(S51)*
California Current	*Sebastes goodei*	Chilipepper Southern Pacific Coast	2007	1.96	0.03	Yes	*(S52)*
California Current	*Sebastes levis*	Cowcod Southern California	2007	0.09	0.08	Yes	*(S53)*

California Current	*Sebastes crameri*	Darkblotched rockfish Pacific Coast	2007	0.73	0.29	Yes	*(S54)*
California Current	*Parophrys vetulus*	English sole Pacific Coast	2007	6.42	0.06	Yes	*(S55)*
California Current	*Raja rhina*	Longnose skate Pacific Coast	2007	1.76	0.46	Yes	*(S56)*
California Current	*Merluccius productus*	Pacific hake Pacific Coast	2008	1.61	0.73	Yes	*(S57)*
California Current	*Sebastes alutus*	Pacific ocean perch Pacific Coast	2007	0.69	0.08	Yes	*(S58)*
California Current	*Anoplopoma fimbria*	Sablefish Pacific Coast	2007	1.02	0.69	Yes	*(S59)*
California Current	*Sebastes entomelas*	Widow rockfish Pacific Coast	2006	0.88	0.05	Yes	*(S60)*
California Current	*Sebastes ruberrimus*	Yelloweye rockfish Pacific Coast	2006	0.83	0.61	Yes	*(S61)*
Celtic-Biscay Shelf	*Gadus morhua*	Atlantic cod Irish Sea	2006	0.15	0.56	No	*(S62)*
Celtic-Biscay Shelf	*Gadus morhua*	Atlantic cod West of Scotland	2006	0.12	0.42	No	*(S62)*
Celtic-Biscay Shelf	*Micromesistius poutassou*	Blue whiting Northeast Atlantic	2006	0.67	1.66	No	*(S63)*
Celtic-Biscay Shelf	*Solea vulgaris*	Common European sole Bay of Biscay	2006	0.75	1.00	No	*(S64)*
Celtic-Biscay Shelf	*Solea vulgaris*	Common European sole Irish Sea	2006	0.36	1.16	No	*(S62)*
Celtic-Biscay Shelf	*Solea vulgaris*	Common European sole ICES VIId	2006	1.41	0.68	No	*(S65)*
Celtic-Biscay Shelf	*Solea vulgaris*	Common European sole Celtic Sea	2006	0.90	0.95	No	*(S66)*
Celtic-Biscay Shelf	*Solea vulgaris*	Common European sole Western English Channel	2006	0.51	1.74	No	*(S66)*
Celtic-Biscay Shelf	*Pleuronectes platessa*	European plaice Irish Sea	2006	1.07	0.23	No	*(S62)*
Celtic-Biscay Shelf	*Pleuronectes platessa*	European plaice ICES VIIe-k	2006	0.65	0.41	No	*(S66)*
Celtic-Biscay Shelf	*Pleuronectes platessa*	European plaice ICES VIIe	2006	0.51	1.39	No	*(S66)*
Celtic-Biscay Shelf	*Melanogrammus aeglefinus*	Haddock West of Scotland	2006	0.58	0.73	No	*(S62)*
Celtic-Biscay Shelf	*Merluccius merluccius*	Hake Northeast Atlantic North	2006	1.04	0.74	No	*(S64)*
Celtic-Biscay Shelf	*Clupea harengus*	Atlantic herring Northern Irish Sea	2006	0.72	0.34	No	*(S67)*
Celtic-Biscay Shelf	*Clupea harengus*	Atlantic herring ICES VIa	2006	0.18	1.59	No	*(S67)*
Celtic-Biscay Shelf	*Clupea harengus*	Atlantic herring ICES VIa-VIIb-VIIc	2000	0.50	1.04	No	*(S67)*
Celtic-Biscay Shelf	*Scomber scombrus*	Mackerel ICES Northeast Atlantic	2006	0.98	0.73	No	*(S68)*
Celtic-Biscay Shelf	*Merlangius merlangus*	Whiting ICES VIIe-k	2006	0.44	1.25	No	*(S66)*
Eastern Bering Sea	*Pleuronectes quadrituberculatus*	Alaska plaice Bering Sea and Aleutian Islands	2007	2.20	0.06	Yes	*(S69)*
Eastern Bering Sea	*Reinhardtius stomias*	Arrowtooth flounder Bering Sea and Aleutian Islands	2008	2.70	0.31	No	*(S70)*
Eastern Bering Sea	*Pleurogrammus monopterygius*	Atka mackerel Bering Sea and Aleutian Islands	2008	1.71	0.55	No	*(S71)*
Eastern Bering Sea	*Hippoglossoides elassodon*	Flathead sole Bering Sea and Aleutian Islands	2008	1.83	0.18	No	*(S72)*
Eastern Bering Sea	*Reinhardtius hippoglossoides*	Greenland turbot Bering Sea and Aleutian Islands	2007	1.46	0.05	Yes	*(S73)*
Eastern Bering Sea	*Lepidopsetta polyxystra*	Northern rock sole Eastern Bering Sea and Aleutian Islands	2007	3.02	0.21	Yes	*(S74)*
Eastern Bering Sea	*Sebastes polyspinis*	Northern rockfish Bering Sea and Aleutian Islands	2008	1.42	0.13	No	*(S75)*

Eastern Bering Sea	*Gadus macrocephalus*	Pacific cod Bering Sea and Aleutian Islands	2007	1.14	0.93	No	*(S76)*
Eastern Bering Sea	*Sebastes alutus*	Pacific ocean perch Eastern Bering Sea and Aleutian Islands	2008	1.27	0.26	No	*(S77)*
Eastern Bering Sea	*Chionoecetes opilio*	Snow crab Bering Sea	2008	0.55	1.49	No	*(S78)*
Eastern Bering Sea	*Chionoecetes bairdi*	Tanner crab Bering Sea	2007	0.79	0.15	No	*(S78)*
Eastern Bering Sea	*Theragra chalcogramma*	Walleye pollock Eastern Bering Sea	2007	0.92	0.94	No	*(S79)*
Eastern Bering Sea	*Limanda aspera*	Yellowfin sole Bering Sea and Aleutian Islands	2007	2.00	0.69	Yes	*(S80)*
Eastern Bering Sea	*Paralithodes platypus*	Blue king crab Pribilof Islands	2008	0.08	0.00	Yes	*(S78)*
Eastern Bering Sea	*Paralithodes camtschaticus*	Red king crab Norton Sound	2008	1.47	NA	Yes	*(S78)*
Eastern Bering Sea	*Paralithodes platypus*	Blue king crab St. Matthew Island	2008	1.45	NA	Yes	*(S78)*
Eastern Bering Sea	*Paralithodes camtschaticus*	Red king crab Pribilof Islands	2009	1.44	NA	Yes	*(S78)*
Eastern Bering Sea	*Paralithodes camtschaticus*	Red king crab Bristol Bay	2008	1.27	1.05	Yes	*(S78)*
Eastern Bering Sea	*Lithodes aequispinus*	Golden king crab Aleutian Islands Eastern Stock	2007	0.61	NA	Yes	*(S78)*
Eastern Bering Sea	*Lithodes aequispinus*	Golden king crab Aleutian Islands Western Stock	2007	0.53	NA	Yes	*(S78)*
Faroe Plateau	*Gadus morhua*	Atlantic cod Faroe Plateau	2006	0.26	1.52	No	*(S81)*
Faroe Plateau	*Melanogrammus aeglefinus*	Haddock Faroe Plateau	2006	0.85	1.07	No	*(S81)*
Faroe Plateau	*Pollachius virens*	Saithe Faroe Plateau	2006	0.99	1.52	No	*(S81)*
Gulf of Alaska	*Parophrys vetulus*	English sole Hecate Strait	2001	1.23	0.37	No	*(S82)*
Gulf of Alaska	*Gadus macrocephalus*	Pacific cod Hecate Strait	2004	1.08	0.18	No	*(S83)*
Gulf of Alaska	*Gadus macrocephalus*	Pacific cod West Coast of Vancouver Island	2001	1.04	0.47	Yes	*(S84)*
Gulf of Alaska	*Clupea pallasii*	Pacific herring Central Coast	2007	0.30	0.11	No	*(S85)*
Gulf of Alaska	*Clupea pallasii*	Pacific herring Prince Rupert District	2007	0.16	0.32	No	*(S85)*
Gulf of Alaska	*Clupea pallasii*	Pacific herring Queen Charlotte Islands	2007	0.20	0.00	No	*(S85)*
Gulf of Alaska	*Clupea pallasii*	Pacific herring Strait of Georgia	2007	0.91	0.40	No	*(S85)*
Gulf of Alaska	*Clupea pallasii*	Pacific herring West Coast of Vancouver Island	2007	0.03	0.00	No	*(S85)*
Gulf of Alaska	*Lepidopsetta bilineata*	Rock sole Hecate Strait	2001	1.03	0.45	No	*(S82)*
Gulf of Alaska	*Anoplopoma fimbria*	Sablefish Eastern Bering Sea / Aleutian Islands / Gulf of Alaska	2007	1.05	0.66	Yes	*(S86)*
Gulf of Mexico	*Mycteroperca microlepis*	Gag Gulf of Mexico	2004	1.00	1.99	Yes	*(S87)*
Gulf of Mexico	*Brevoortia patronus*	Gulf menhaden Gulf of Mexico	2004	1.08	0.48	No	*(S88)*
Iberian Coastal	*Lepidorhombus boscii*	Fourspotted megrim ICES VIIIc-IXa	2006	0.70	1.01	No	*(S64)*
Iberian Coastal	*Lepidorhombus whiffiagonis*	Megrim ICES VIIIc-IXa	2006	0.43	1.07	No	*(S64)*
Iceland Shelf	*Gadus morhua*	Atlantic cod Iceland	2006	0.46	1.17	No	*(S81)*
Iceland Shelf	*Mallotus villosus*	Capelin Iceland	2006	0.49	0.85	No	*(S81)*

Iceland Shelf	*Melanogrammus aeglefinus*	Haddock Iceland	2007	0.98	1.23	No	*(S81)*
Iceland Shelf	*Clupea harengus*	Atlantic herring Iceland (summer spawners)	2006	1.00	0.79	No	*(S81)*
Mediterranean Sea	*Xiphias gladius*	Swordfish Mediterranean Sea	2005	0.94	1.26	Yes	*(S89)*
New Zealand Shelf	*Genypterus blacodes*	Ling New Zealand Areas LIN3 and LIN4	2007	3.07	0.09	Yes	*(S90)*
New Zealand Shelf	*Genypterus blacodes*	Ling New Zealand Areas LIN5 and LIN6	2007	3.96	0.10	Yes	*(S90)*
New Zealand Shelf	*Genypterus blacodes*	Ling New Zealand Area LIN6b	2006	2.19	0.11	Yes	*(S91)*
New Zealand Shelf	*Genypterus blacodes*	Ling New Zealand Area LIN7CK	2007	2.49	0.32	Yes	*(S90)*
New Zealand Shelf	*Genypterus blacodes*	Ling New Zealand Area LIN7WC	2008	2.21	0.13	Yes	*(S91)*
New Zealand Shelf	*Allocyttus niger*	Black oreo west end of Chatham Rise	2007	0.99	0.82	Yes	*(S91)*
New Zealand Shelf	*Haliotis iris*	Paua New Zealand Area PAU5A	2006	0.72	2.83	No	*(S92)*
New Zealand Shelf	*Haliotis iris*	Paua New Zealand Area PAU5B	2007	1.02	0.59	No	*(S93)*
New Zealand Shelf	*Haliotis iris*	Paua New Zealand Area PAU5D	2006	0.44	2.10	No	*(S92)*
New Zealand Shelf	*Haliotis iris*	Paua New Zealand Area PAU7	2008	0.87	0.94	No	*(S94)*
New Zealand Shelf	*Rexea solandri*	Common gemfish	2006	1.61	0.30	Yes	*(S95)*
New Zealand Shelf	*Macruronus novaezelandiae*	Hoki Eastern New Zealand	2007	1.11	0.33	No	*(S96)*
New Zealand Shelf	*Macruronus novaezelandiae*	Hoki Western New Zealand	2007	0.51	0.57	No	*(S96)*
New Zealand Shelf	*Chrysophrys auratus*	New Zealand snapper New Zealand SNA8	2005	0.35	2.50	Yes	*(S97)*
New Zealand Shelf	*Pseudocyttus maculatus*	Smooth oreo west end of Chatham Rise	2004	1.06	0.54	No	*(S91)*
New Zealand Shelf	*Micromesistius australis*	Southern blue whiting Campbell Island Rise	2006	0.86	1.20	No	*(S98)*
New Zealand Shelf	*Merluccius australis*	Southern hake Chatham Rise	2006	1.77	0.12	Yes	*(S99)*
New Zealand Shelf	*Merluccius australis*	Southern hake Sub-Antarctic	2007	2.91	0.11	Yes	*(S100)*
New Zealand Shelf	*Pseudocaranx dentex*	Trevally New Zealand Area TRE7	2005	1.44	0.83	Yes	*(S101)*
Newfoundland-Labrador Shelf	*Hippoglossoides platessoides*	American plaice NAFO 23K	2003	0.12	0.07	No	*(S102)*
Newfoundland-Labrador Shelf	*Hippoglossoides platessoides*	American plaice NAFO 3LNO	2006	0.08	0.77	No	*(S103)*
Newfoundland-Labrador Shelf	*Gadus morhua*	Atlantic cod NAFO 3Ps	2004	0.48	0.41	No	*(S104)*
Newfoundland-Labrador Shelf	*Gadus morhua*	Atlantic cod NAFO 3Pn4RS	2006	0.09	0.79	No	*(S105)*
Newfoundland-Labrador Shelf	*Gadus morhua*	Atlantic cod NAFO 3NO	2006	0.02	0.27	No	*(S106)*
Newfoundland-Labrador Shelf	*Reinhardtius hippoglossoides*	Greenland halibut NAFO 23KLMNO	2006	0.39	1.73	No	*(S107)*
Newfoundland-Labrador Shelf	*Redfish species*	Redfish species NAFO 3LN	2006	1.91	0.01	Yes	*(S108)*
North Sea	*Gadus morhua*	Atlantic cod Kattegat	2006	0.19	0.31	No	*(S40)*
North Sea	*Gadus morhua*	Atlantic cod North Sea	2006	0.19	0.80	No	*(S65)*
North Sea	*Solea vulgaris*	Common European sole ICES Kattegat and Skagerrak	2006	1.25	0.54	No	*(S40)*
North Sea	*Melanogrammus aeglefinus*	Haddock ICES IIIa and North Sea	2006	0.62	0.25	No	*(S65)*

North Sea	*Clupea harengus*	Atlantic herring North Sea	2006	0.65	1.32	No	*(S67)*
North Sea	*Trisopterus esmarkii*	Norway pout North Sea	2006	0.90	0.33	No	*(S65)*
North Sea	*Pollachius virens*	Saithe ICES IIIa, VI and North Sea	2006	0.57	0.97	No	*(S65)*
North Sea	*Ammodytes marinus*	Sandeel North Sea	2007	0.92	0.24	No	*(S65)*
North Sea	*Merlangius merlangus*	Whiting ICES IIIa, VIId and North Sea	2006	0.33	1.04	No	*(S65)*
Northeast U.S. Shelf	*Homarus americanus*	American lobster Rhode Island	2006	0.61	0.73	Yes	*(S109)*
Northeast U.S. Shelf	*Hippoglossoides platessoides*	American plaice NAFO 5YZ	2007	0.70	0.30	No	*(S110)*
Northeast U.S. Shelf	*Gadus morhua*	Atlantic cod NAFO 5Zjm	2002	0.34	0.45	No	*(S111)*
Northeast U.S. Shelf	*Gadus morhua*	Atlantic cod Georges Bank	2007	0.12	0.72	No	*(S110)*
Northeast U.S. Shelf	*Gadus morhua*	Atlantic cod Gulf of Maine	2007	0.63	2.40	Yes	*(S110)*
Northeast U.S. Shelf	*Melanogrammus aeglefinus*	Haddock NAFO 4X5Y	2003	0.85	0.33	No	*(S112)*
Northeast U.S. Shelf	*Melanogrammus aeglefinus*	Haddock NAFO 5Zejm	2002	1.00	0.65	No	*(S113)*
Northeast U.S. Shelf	*Melanogrammus aeglefinus*	Haddock NAFO 5Y	2007	0.99	1.21	No	*(S110)*
Northeast U.S. Shelf	*Pollachius virens*	Pollock NAFO 4VWX5Zc	2006	0.56	0.30	No	*(S114)*
Northeast U.S. Shelf	*Tautoga onitis*	Tautog Rhode Island	2006	0.79	0.62	Yes	*(S109)*
Northeast U.S. Shelf	*Pseudopleuronectes americanus*	Winter flounder Southern New England-Mid Atlantic	2007	0.09	1.10	No	*(S110)*
Northeast U.S. Shelf	*Pseudopleuronectes americanus*	Winter flounder Rhode Island	2006	0.23	2.02	Yes	*(S109)*
Northeast U.S. Shelf	*Limanda ferruginea*	Yellowtail flounder Georges Bank	2007	0.22	1.14	Yes	*(S110)*
Northeast U.S. Shelf	*Brevoortia tyrannus*	Atlantic menhaden	2005	0.47	0.97	No	*(S115)*
Norwegian Sea	*Gadus morhua*	Atlantic cod coastal Norway	2006	0.27	2.17	No	*(S41)*
Pacific High Seas	*Thunnus alalunga*	Albacore tuna South Pacific Ocean	2006	2.46	0.91	Yes	*(S116)*
Pacific High Seas	*Thunnus obesus*	Bigeye tuna Western Pacific Ocean	2006	1.05	1.38	Yes	*(S117)*
Pacific High Seas	*Katsuwonus pelamis*	Skipjack tuna Central Western Pacific	2006	4.38	0.31	Yes	*(S118)*
Pacific High Seas	*Thunnus albacares*	Yellowfin tuna Central Western Pacific	2005	1.22	0.80	Yes	*(S119)*
Patagonian Shelf	*Merluccius hubbsi*	Argentine hake Northern Argentina	2007	0.19	1.26	Yes	*(S120)*
Patagonian Shelf	*Merluccius hubbsi*	Argentine hake Southern Argentina	2007	0.54	1.67	Yes	*(S121)*
Patagonian Shelf	*Macruronus magellanicus*	Patagonian grenadier Southern Argentina	2006	2.15	0.60	Yes	*(S122)*
Patagonian Shelf	*Micromesistius australis*	Southern blue whiting Southern Argentina	2007	0.38	1.18	No	*(S123)*
Scotian Shelf	*Gadus morhua*	Atlantic cod NAFO 4TVn	2006	0.17	0.32	No	*(S124)*
Southern Australian Shelf	*Genypterus blacodes*	Ling Great Australian Bight	2007	1.08	8.98	No	*(S125)*
Southern Australian Shelf	*Genypterus blacodes*	Ling Southeast Australia	2007	0.59	2.20	No	*(S125)*
Southern Australian Shelf	*Seriolella brama*	Blue warehou Great Australian Bight	2006	0.41	2.04	No	*(S126)*
Southern Australian Shelf	*Seriolella brama*	Blue warehou Southeast Australia	2006	0.49	0.84	No	*(S126)*

Southern Australian Shelf	*Rexea solandri*	Common gemfish Southeast Australia	2007	0.25	0.39	No	*(S127)*
Southern Australian Shelf	*Platycephalus conatus*	Deepwater flathead Southeast Australia	2006	1.43	0.61	No	*(S128)*
Southern Australian Shelf	*Nemadactylus macropterus*	Jackass morwong Southeast Australia	2007	0.31	1.80	No	*(S129)*
Southern Australian Shelf	*Hoplostethus atlanticus*	Orange roughy Southeast Australia	2006	0.48	0.29	No	*(S130)*
Southern Australian Shelf	*Sillago flindersi*	School whiting Southeast Australia	2007	0.66	0.82	No	*(S131)*
Southern Australian Shelf	*Seriolella punctata*	Silverfish Southeast Australia	2006	1.03	0.79	No	*(S132)*
Southern Australian Shelf	*Neoplatycephalus richardsoni*	Tiger flathead Southeast Australia	2006	1.78	1.03	No	*(S133)*
Southeast U.S. Shelf	*Pagrus pagrus*	Red porgy Southern Atlantic coast	2004	0.61	0.39	Yes	*(S134)*
Southeast U.S. Shelf	*Scomberomorus maculatus*	Spanish mackerel Southern Atlantic Coast	2007	0.47	0.91	Yes	*(S135)*

Table S3. Summary of the trawl surveys compiled for analysis, the number of years in which surveys were conducted, the time span of the surveys, the number of taxa included in the analysis, how many of these taxa were identified to species, and the number of taxa that were invertebrate, pelagic (mid-water fish species) and demersal (bottom-dwelling fish species). NR = taxa not reported in a given survey, URI = University of Rhode Island, MLI = Maurice Lamontagne Institute, NOAA = National Oceanic and Atmospheric Administration, CEFAS = Centre for Environment, Fisheries and Aquaculture Science, CSIRO = Commonwealth Scientific and Industrial Research Organisation, ADF&G = Alaska Department of Fish and Game.

Survey name	Years	Year range	Taxa	Species	Invertebrates	Pelagics	Demersals	Source or analyst
St. Pierre Bank, Newfoundland	40	1951-1995	27	26	NR	1	26	(*S136*)
Southern Grand Banks, Newfoundland	41	1952-1995	20	19	NR	1	19	(*S136*)
Northern Gulf of St. Lawrence	18	1990-2007	10	9	NR	1	9	Diane Archambault, MLI
Southern Gulf of St. Lawrence	37	1971-2007	52	49	NR	7	45	(*S137*)
Scotian Shelf and Bay of Fundy	36	1970-2006	49	48	1	6	42	(*S138*)
Gulf of Maine	45	1963-2007	34	34	3	4	27	Michael Fogarty, NOAA
Georges Bank	45	1963-2007	40	40	3	1	36	Michael Fogarty, NOAA
URI Fox Island	47	1959-2005	25	23	7	2	16	(*S139*)
URI Whale Rock	47	1959-2005	25	23	7	2	16	(*S139*)
Mid-Atlantic Bight	41	1967-2007	40	40	3	4	33	Michael Fogarty, NOAA
South Georgia, Subantarctic	23	1970-1992	5	5	NR	1	4	(*S12*)
Celtic Sea	18	1987-2004	55	53	NR	9	46	Simon Jennings, CEFAS
North Sea	28	1980-2007	49	53	NR	4	45	Simon Jennings, CEFAS
North-west Australia	13	1978-1997	542	538	5	39	498	Beth Fulton, CSIRO
Gulf of Thailand	35	1961-1995	38	5	6	6	26	(*S140, S141*)
Eastern Bering Sea	27	1982-2008	32	12	13	2	17	Robert Lauth, NOAA
Aleutian Islands	10	1980-2006	66	58	2	2	62	Mark Wilkins, NOAA
Gulf of Alaska small mesh	36	1972-2007	24	13	11	3	10	Aaren Ellsworth, ADF&G
Gulf of Alaska trawl	10	1984-2007	118	105	6	8	104	Mark Wilkins, NOAA
US West Coast	10	1977-2004	58	57	1	9	48	Mark Wilkins, NOAA

Table S4. Ecosystem exploitation rates ($u_{ave}=C_{tot}/B_{tot}$) and average ratio of biomass to B_{MSY} ($B_{ave} = B/B_{MSY}$) for ecosystems plotted in Figure 3A, based on stock assessments. The average B/B_{MSY} ratio is the geometric mean of the ratios for individual fishery stocks. For this analysis, Pacific hake, Atlantic menhaden, and blue whiting were excluded from the California Current, Northeast U.S. Shelf, and Celtic-Biscay Shelf, respectively (see above text for explanation).

	Iceland Shelf		North Sea		Celtic-Biscay Shelf		S. Australia Shelf		California Current		Northeast U.S. Shelf		Newfoundland-Labrad.		Baltic Sea		Eastern Bering Sea		New Zealand Shelf	
Year	u_{ave}	B_{ave}	u_{ave}	B_{ave}	u_{ave}	B_{ave}	u_{ave}	B_{ave}	u_{ave}	B_{ave}	u_{ave}	B_{ave}	u_{ave}	B_{ave}	u_{ave}	B_{ave}	u_{ave}	B_{ave}	u_{ave}	B_{ave}
1975									0.033	1.853			0.174	0.538	0.164	0.803			0.017	2.728
1976									0.050	1.810			0.176	0.532	0.144	0.857			0.024	2.641
1977									0.030	1.760			0.169	0.574	0.154	0.744			0.038	2.479
1978					0.186	0.649	0.030	2.030	0.038	1.746	0.325	0.647	0.128	0.861	0.144	0.700	0.140	0.792	0.009	2.451
1979	0.380	0.771			0.218	0.787	0.027	2.029	0.057	1.707	0.283	0.672	0.151	0.800	0.156	0.676	0.121	0.813	0.015	2.246
1980	0.415	0.673			0.211	0.841	0.032	2.013	0.062	1.623	0.345	0.783	0.149	0.829	0.186	0.672	0.105	0.922	0.013	2.284
1981	0.464	0.633			0.216	0.871	0.030	2.029	0.074	1.543	0.329	0.678	0.150	0.877	0.171	0.693	0.079	1.020	0.014	2.185
1982	0.210	0.555			0.224	0.785	0.029	1.964	0.090	1.426	0.379	0.690	0.144	0.904	0.156	0.703	0.070	1.067	0.014	2.098
1983	0.311	0.614	0.221	0.917	0.192	0.836	0.029	1.900	0.067	1.278	0.382	0.628	0.133	0.954	0.146	0.817	0.068	0.977	0.016	2.046
1984	0.355	0.623	0.259	0.804	0.201	0.863	0.027	1.902	0.061	1.169	0.346	0.540	0.131	0.941	0.180	0.773	0.079	0.868	0.019	1.973
1985	0.439	0.634	0.285	0.697	0.181	0.856	0.030	1.777	0.062	1.069	0.377	0.448	0.142	0.931	0.187	0.729	0.077	0.744	0.015	1.981
1986	0.403	0.571	0.229	0.874	0.183	0.837	0.035	2.002	0.061	0.963	0.330	0.417	0.186	0.852	0.197	0.624	0.080	0.730	0.026	1.966
1987	0.378	0.611	0.247	0.742	0.198	0.918	0.040	1.981	0.068	0.893	0.343	0.361	0.220	0.716	0.180	0.668	0.063	0.854	0.045	1.942
1988	0.411	0.635	0.338	0.583	0.203	0.894	0.050	2.010	0.061	0.852	0.362	0.338	0.189	0.624	0.195	0.625	0.091	0.920	0.078	1.907
1989	0.424	0.581	0.297	0.632	0.181	0.772	0.132	1.918	0.071	0.810	0.299	0.344	0.204	0.550	0.179	0.610	0.091	1.007	0.048	2.007
1990	0.304	0.503	0.251	0.617	0.208	0.704	0.175	1.771	0.069	0.772	0.371	0.351	0.229	0.461	0.151	0.652	0.111	1.183	0.048	1.989
1991	0.356	0.512	0.274	0.670	0.204	0.660	0.143	1.535	0.065	0.739	0.429	0.340	0.289	0.358	0.132	0.627	0.119	1.255	0.063	1.920
1992	0.375	0.491	0.291	0.725	0.222	0.679	0.166	1.304	0.061	0.712	0.430	0.297	0.293	0.243	0.116	0.649	0.108	1.361	0.082	1.796
1993	0.444	0.551	0.292	0.646	0.246	0.657	0.113	1.129	0.059	0.678	0.419	0.255	0.311	0.175	0.123	0.701	0.089	1.351	0.061	1.856
1994	0.348	0.545	0.256	0.761	0.252	0.649	0.090	1.018	0.051	0.654	0.361	0.233	0.188	0.128	0.165	0.733	0.091	1.394	0.052	1.860
1995	0.309	0.547	0.251	0.786	0.230	0.636	0.088	0.952	0.049	0.644	0.249	0.257	0.055	0.127	0.143	0.732	0.081	1.379	0.055	1.786
1996	0.475	0.514	0.251	0.645	0.188	0.619	0.085	0.932	0.051	0.643	0.252	0.290	0.059	0.150	0.184	0.706	0.086	1.401	0.060	1.758
1997	0.433	0.489	0.194	0.726	0.181	0.618	0.089	0.987	0.055	0.643	0.261	0.314	0.087	0.163	0.221	0.611	0.094	1.242	0.088	1.634
1998	0.434	0.484	0.281	0.553	0.214	0.593	0.087	0.932	0.040	0.644	0.253	0.321	0.100	0.176	0.219	0.559	0.089	1.178	0.106	1.561
1999	0.379	0.485	0.184	0.633	0.205	0.531	0.086	0.795	0.046	0.659	0.219	0.322	0.125	0.188	0.234	0.526	0.076	1.100	0.099	1.525
2000	0.447	0.457	0.167	0.667	0.233	0.544	0.091	0.685	0.039	0.678	0.211	0.357	0.137	0.188	0.214	0.570	0.084	0.833	0.111	1.454
2001	0.448	0.511	0.239	0.529	0.241	0.586	0.083	0.636	0.032	0.708	0.212	0.397	0.124	0.192	0.223	0.525	0.097	1.131	0.141	1.366
2002	0.382	0.618	0.198	0.540	0.279	0.580	0.094	0.632	0.021	0.764	0.193	0.374	0.118	0.172	0.214	0.510	0.100	1.134	0.106	1.381

2003	0.269	0.733	0.189	0.382	0.223	0.558	0.088	0.645	0.021	0.838	0.203	0.371	0.114	0.176	0.183	0.550	0.093	1.104	0.125	1.295
2004	0.345	0.708	0.210	0.391	0.231	0.509	0.088	0.670	0.020	0.907	0.260	0.284	0.077	0.178	0.161	0.548	0.096	1.100	0.078	1.336
2005	0.230	0.623	0.157	0.468	0.181	0.496	0.074	0.649	0.019	0.965	0.210	0.273	0.075	0.163	0.173	0.562	0.106	1.038	0.067	1.318
2006	0.256	0.687	0.180	0.499	0.167	0.530	0.055	0.638	0.018	1.016	0.132	0.310	0.075	0.163	0.166	0.568	0.121	1.032	0.061	1.391
2007								1.176	0.023								0.123			
n stocks	4		9		17		11		16		13		7		7		20		19	
u_{MMSY}	0.23-0.34		0.08-0.16		0.08-0.17		0.12-0.18		0.03-0.07		0.20-0.32		0.20-0.26		0.08-0.12		0.14-0.21		0.08-0.11	
$u_{conserve}$	0.005-0.10		0.01-0.06		0.006-0.065		0.005-0.035		0.01-0.04		0.02-0.08		0.01-0.05		0.03-0.07		0.02-0.06		0.04-0.06	

Table S5. Annual total biomass (metric tons) from stock assessments for all data and for the three focal regions, corresponding to Figs 4A-D. This is based on stocks with assessment biomass data for at least 25 years within 1977-2006.

Region	Year	Demersal ≥90	Demersal 30-90	Demersal ≤30	Invertebrate	Pelagic	*n* stocks
All data	1977	17,816,886	10,933,201	-	886,431	39,198,859	144
All data	1978	18,148,804	11,277,329	-	819,442	40,085,956	144
All data	1979	18,621,668	11,417,867	-	795,435	38,718,873	144
All data	1980	20,975,601	12,076,138	-	772,254	39,829,121	144
All data	1981	24,117,281	13,029,132	-	697,201	36,062,916	144
All data	1982	24,681,175	13,750,325	-	554,708	35,554,985	144
All data	1983	26,628,867	14,669,171	-	536,099	44,591,334	144
All data	1984	25,715,680	14,709,692	-	544,492	42,056,401	144
All data	1985	27,878,462	15,070,862	-	557,139	40,471,702	144
All data	1986	28,373,242	14,926,243	-	705,275	36,382,584	144
All data	1987	26,844,578	15,595,944	-	865,234	36,826,942	144
All data	1988	23,972,032	15,064,374	-	1,015,574	33,677,846	144
All data	1989	23,091,704	15,188,085	-	1,123,676	32,101,630	144
All data	1990	21,028,206	14,909,796	-	1,141,282	36,447,263	144
All data	1991	18,285,742	14,697,201	-	990,974	38,417,689	144
All data	1992	20,657,921	14,625,057	-	841,230	36,519,227	144
All data	1993	24,018,728	14,550,552	-	721,690	32,644,692	144
All data	1994	24,750,393	13,897,749	-	691,168	30,501,946	144
All data	1995	26,049,168	13,372,661	-	696,255	30,510,852	144
All data	1996	24,255,480	12,882,579	-	692,830	29,314,150	144
All data	1997	21,573,705	12,842,952	-	601,987	31,217,521	144
All data	1998	20,356,689	12,584,281	-	431,302	32,884,159	144
All data	1999	23,209,113	12,580,708	-	307,177	35,375,669	144
All data	2000	22,251,220	13,317,021	-	276,214	35,606,766	144
All data	2001	20,031,195	13,610,095	-	263,017	35,958,787	144
All data	2002	21,032,807	14,482,568	-	265,593	36,687,404	144
All data	2003	21,976,779	14,545,071	-	283,830	34,845,474	144
All data	2004	21,470,974	14,433,304	-	312,984	33,391,206	144
All data	2005	21,301,002	14,068,621	-	351,668	30,066,567	144
All data	2006	17,885,573	13,890,451	-	410,944	29,066,058	144
Eastern Bering Sea	1977	4,182,241	4,828,054	-	875,574	-	15
Eastern Bering Sea	1978	4,101,016	5,216,561	-	809,077	-	15
Eastern Bering Sea	1979	4,142,030	5,538,150	-	785,558	-	15
Eastern Bering Sea	1980	5,526,030	6,001,714	-	762,856	-	15
Eastern Bering Sea	1981	9,636,570	6,271,765	-	687,900	-	15
Eastern Bering Sea	1982	11,059,670	6,474,280	-	545,966	-	15
Eastern Bering Sea	1983	12,222,520	6,673,643	-	527,850	-	15
Eastern Bering Sea	1984	11,921,580	6,868,758	-	536,373	-	15
Eastern Bering Sea	1985	14,165,850	6,897,416	-	549,137	-	15
Eastern Bering Sea	1986	13,368,060	6,893,876	-	696,990	-	15
Eastern Bering Sea	1987	13,994,680	7,114,904	-	856,804	-	15
Eastern Bering Sea	1988	13,192,110	7,187,231	-	1,006,658	-	15
Eastern Bering Sea	1989	11,312,030	7,430,937	-	1,113,805	-	15
Eastern Bering Sea	1990	9,107,410	7,763,384	-	1,130,892	-	15

Eastern Bering Sea	1991	7,153,070	7,988,098	-	980,816	-	15
Eastern Bering Sea	1992	10,416,780	8,428,810	-	831,412	-	15
Eastern Bering Sea	1993	12,634,860	8,468,827	-	711,773	-	15
Eastern Bering Sea	1994	12,310,360	8,410,467	-	681,495	-	15
Eastern Bering Sea	1995	14,293,790	8,316,493	-	686,104	-	15
Eastern Bering Sea	1996	12,398,500	8,207,331	-	682,239	-	15
Eastern Bering Sea	1997	10,918,220	8,053,359	-	592,024	-	15
Eastern Bering Sea	1998	10,916,180	7,768,184	-	421,909	-	15
Eastern Bering Sea	1999	11,901,510	7,644,098	-	298,740	-	15
Eastern Bering Sea	2000	11,098,780	7,523,617	-	268,622	-	15
Eastern Bering Sea	2001	10,816,550	7,501,460	-	256,057	-	15
Eastern Bering Sea	2002	11,256,650	7,549,053	-	259,051	-	15
Eastern Bering Sea	2003	12,889,180	7,716,193	-	277,267	-	15
Eastern Bering Sea	2004	11,904,540	8,005,913	-	306,161	-	15
Eastern Bering Sea	2005	9,914,060	8,093,434	-	344,444	-	15
Eastern Bering Sea	2006	7,671,050	8,192,378	-	403,452	-	15
Eastern Bering Sea	2007	6,180,190	8,348,116	-	443,414	-	15
Eastern Bering Sea	2008	5,297,950	8,303,982	-	426,422	-	15
Eastern Canada	1977	716,384	1,033,571	-	-	-	8
Eastern Canada	1978	918,804	1,043,653	-	-	-	8
Eastern Canada	1979	939,788	905,712	-	-	-	8
Eastern Canada	1980	1,046,927	916,708	-	-	-	8
Eastern Canada	1981	1,144,446	906,104	-	-	-	8
Eastern Canada	1982	1,171,863	939,001	-	-	-	8
Eastern Canada	1983	1,276,854	961,264	-	-	-	8
Eastern Canada	1984	1,272,302	888,437	-	-	-	8
Eastern Canada	1985	1,329,463	853,920	-	-	-	8
Eastern Canada	1986	1,301,265	774,311	-	-	-	8
Eastern Canada	1987	957,556	701,209	-	-	-	8
Eastern Canada	1988	791,766	645,418	-	-	-	8
Eastern Canada	1989	639,677	604,011	-	-	-	8
Eastern Canada	1990	556,974	513,773	-	-	-	8
Eastern Canada	1991	402,520	472,523	-	-	-	8
Eastern Canada	1992	274,751	378,716	-	-	-	8
Eastern Canada	1993	183,943	299,782	-	-	-	8
Eastern Canada	1994	166,161	229,391	-	-	-	8
Eastern Canada	1995	191,785	215,451	-	-	-	8
Eastern Canada	1996	227,501	242,546	-	-	-	8
Eastern Canada	1997	256,522	264,043	-	-	-	8
Eastern Canada	1998	261,392	306,123	-	-	-	8
Eastern Canada	1999	269,643	341,165	-	-	-	8
Eastern Canada	2000	247,759	366,462	-	-	-	8
Eastern Canada	2001	248,661	377,957	-	-	-	8
Eastern Canada	2002	233,626	368,800	-	-	-	8
Eastern Canada	2003	218,810	377,914	-	-	-	8
Eastern Canada	2004	233,618	373,657	-	-	-	8
Eastern Canada	2005	235,779	382,918	-	-	-	8
Eastern Canada	2006	226,761	387,477	-	-	-	8
Northeast U.S. Shelf	1978	399,970	104,135	-	1,333	-	14
Northeast U.S. Shelf	1979	414,797	106,430	-	1,077	-	14

Northeast U.S. Shelf	1980	435,759	108,736	-	992	-	14
Northeast U.S. Shelf	1981	428,435	102,113	-	1,530	-	14
Northeast U.S. Shelf	1982	415,449	99,468	-	1,555	-	14
Northeast U.S. Shelf	1983	367,021	92,828	-	1,531	-	14
Northeast U.S. Shelf	1984	324,101	74,021	-	1,780	-	14
Northeast U.S. Shelf	1985	290,270	52,433	-	1,849	-	14
Northeast U.S. Shelf	1986	276,710	42,763	-	2,152	-	14
Northeast U.S. Shelf	1987	271,898	37,210	-	2,270	-	14
Northeast U.S. Shelf	1988	274,747	32,278	-	2,834	-	14
Northeast U.S. Shelf	1989	265,998	32,219	-	3,908	-	14
Northeast U.S. Shelf	1990	276,399	34,031	-	4,576	-	14
Northeast U.S. Shelf	1991	247,255	34,990	-	4,473	-	14
Northeast U.S. Shelf	1992	203,518	33,527	-	4,257	-	14
Northeast U.S. Shelf	1993	160,959	31,056	-	4,476	-	14
Northeast U.S. Shelf	1994	133,506	35,092	-	4,311	-	14
Northeast U.S. Shelf	1995	137,519	33,451	-	4,909	-	14
Northeast U.S. Shelf	1996	150,559	32,780	-	5,543	-	14
Northeast U.S. Shelf	1997	157,528	40,337	-	5,172	-	14
Northeast U.S. Shelf	1998	158,440	43,270	-	4,852	-	14
Northeast U.S. Shelf	1999	160,913	42,567	-	4,046	-	14
Northeast U.S. Shelf	2000	198,862	45,953	-	3,239	-	14
Northeast U.S. Shelf	2001	244,508	45,306	-	2,541	-	14
Northeast U.S. Shelf	2002	237,761	39,697	-	2,053	-	14
Northeast U.S. Shelf	2003	252,053	37,429	-	2,019	-	14
Northeast U.S. Shelf	2004	250,128	30,942	-	2,220	-	14
Northeast U.S. Shelf	2005	252,222	26,540	-	2,605	-	14
Northeast U.S. Shelf	2006	265,773	32,384	-	2,920	-	14

Table S6. Survey biomass estimates and mean maximum length (cm) from all surveys combined and for each of the three focal regions (data plotted in Figs 4E-H, M-P). An asterisk next to the year indicates there were too few surveys conducted in that year for the "All data" values to be meaningful.

Survey name	Year	Demersal ≥90	Demersal 30-90	Demersal ≤30	Invertebrate	Pelagic	Mean Lmax	*n* surveys
All data	1951*	60.20	20.86	NA	NA	NA	80.7	1
All data	1952*	43.20	19.97	1.04	NA	0.59	98.0	1
All data	1953*	57.32	16.76	1.07	NA	1.16	96.9	2
All data	1954*	74.43	4.38	1.07	NA	0.61	115.5	2
All data	1955*	23.68	2.46	1.30	NA	0.62	127.5	2
All data	1956*	43.59	12.27	1.19	NA	0.56	108.0	2
All data	1957*	29.39	20.84	1.27	NA	0.55	94.8	2
All data	1958*	30.74	20.20	5.04	NA	0.56	97.2	2
All data	1959	24.00	9.53	1.44	1.70	1.20	107.0	4
All data	1960	21.89	14.26	1.81	1.72	0.84	99.2	4
All data	1961	20.45	14.93	1.49	1.59	0.88	98.9	3
All data	1962	19.31	16.37	1.51	1.38	1.55	89.5	4
All data	1963	15.01	13.72	1.57	0.89	2.81	96.0	7
All data	1964	15.95	10.70	1.71	0.87	1.80	92.8	6
All data	1965	15.03	15.81	1.22	0.61	1.83	88.1	6
All data	1966	14.03	13.08	1.40	1.01	1.57	88.4	6
All data	1967	11.92	11.38	1.33	0.76	1.60	92.2	8
All data	1968	10.67	12.69	1.32	0.99	1.97	92.1	8
All data	1969	11.32	9.32	1.38	1.35	1.68	94.2	7
All data	1970	10.10	10.16	1.60	1.32	1.67	86.2	9
All data	1971	6.62	8.53	1.33	1.53	1.44	79.9	9
All data	1972	7.89	8.95	1.60	1.76	1.53	82.3	11
All data	1973	9.60	8.16	1.26	2.17	1.30	85.0	11
All data	1974	6.83	7.76	1.39	2.19	1.53	77.3	10
All data	1975	7.99	8.13	1.42	2.07	1.96	79.5	10
All data	1976	10.40	9.20	1.21	2.67	1.29	79.1	11
All data	1977	8.77	8.66	1.09	1.38	1.12	82.7	12
All data	1978	7.76	7.65	1.15	1.44	1.05	87.6	12
All data	1979	8.81	7.26	0.96	2.05	1.33	91.9	12
All data	1980	8.64	8.26	1.10	2.14	1.45	87.5	15
All data	1981	12.16	9.21	1.38	2.87	1.66	92.8	12
All data	1982	8.76	7.71	1.01	1.90	1.21	89.0	14
All data	1983	12.09	9.09	0.99	1.72	1.58	95.4	14
All data	1984	10.00	8.31	1.09	1.72	1.79	93.6	13
All data	1985	12.69	9.26	1.29	1.59	2.42	93.8	12
All data	1986	11.22	8.81	1.41	1.59	1.86	88.5	15
All data	1987	11.72	9.95	1.36	2.14	3.02	88.6	15
All data	1988	11.00	9.32	1.20	3.08	2.50	88.0	14
All data	1989	9.02	10.76	1.25	3.04	2.38	84.7	15
All data	1990	10.15	10.00	1.33	2.49	2.19	84.7	16
All data	1991	10.43	8.97	1.30	2.61	2.55	87.1	16
All data	1992	8.42	9.12	1.69	2.45	2.88	84.3	15
All data	1993	6.40	8.10	1.52	2.18	3.84	79.8	15

All data	1994	6.65	7.46	1.27	2.62	3.26	82.9	15
All data	1995	7.02	10.34	1.70	2.47	4.05	77.3	16
All data	1996	9.21	9.47	1.63	1.88	2.18	86.4	13
All data	1997	9.59	9.12	1.28	1.62	2.67	86.8	14
All data	1998	8.86	8.41	1.11	1.34	2.69	87.9	13
All data	1999	10.93	8.85	1.28	1.57	3.02	87.6	13
All data	2000	10.37	10.23	1.26	1.65	2.86	84.8	13
All data	2001	11.13	9.88	1.53	2.10	3.16	86.9	14
All data	2002	12.58	10.21	1.48	2.33	3.56	86.1	13
All data	2003	12.01	10.57	1.56	2.05	3.56	81.3	13
All data	2004	12.16	8.22	1.19	2.05	3.22	88.0	13
All data	2005	10.21	8.25	1.34	1.90	2.83	82.9	12
All data	2006	10.54	9.12	1.10	2.94	2.81	86.8	10
All data	2007	10.53	8.75	1.43	2.10	2.91	83.2	9
All data	2008*	7.20	8.96	NA	2.19	4.27	80.8	1
Eastern Bering Sea	1982	88.54	117.18	0.00	52.44	0.20	63.5	1
Eastern Bering Sea	1983	166.71	125.08	0.00	42.57	1.74	71.5	1
Eastern Bering Sea	1984	134.92	119.58	0.00	50.53	0.58	68.5	1
Eastern Bering Sea	1985	126.70	94.39	0.00	26.24	0.78	73.9	1
Eastern Bering Sea	1986	137.42	95.13	0.00	32.83	0.38	74.7	1
Eastern Bering Sea	1987	143.48	116.31	0.00	62.13	0.23	68.6	1
Eastern Bering Sea	1988	185.86	125.62	0.00	70.75	3.36	70.1	1
Eastern Bering Sea	1989	156.88	116.01	0.00	71.87	0.20	67.9	1
Eastern Bering Sea	1990	182.87	117.44	0.00	75.28	0.22	69.2	1
Eastern Bering Sea	1991	133.93	123.41	0.00	78.68	0.93	64.7	1
Eastern Bering Sea	1992	117.94	125.31	0.00	65.46	0.39	65.7	1
Eastern Bering Sea	1993	146.45	141.37	0.00	64.54	3.66	68.6	1
Eastern Bering Sea	1994	150.61	166.26	0.00	64.71	0.90	70.0	1
Eastern Bering Sea	1995	150.86	131.41	0.00	69.07	1.34	69.4	1
Eastern Bering Sea	1996	101.72	139.62	0.00	70.12	0.62	65.7	1
Eastern Bering Sea	1997	90.59	153.41	0.00	87.33	0.97	61.7	1
Eastern Bering Sea	1998	71.56	135.34	0.00	64.90	0.44	62.5	1
Eastern Bering Sea	1999	101.51	95.01	0.00	53.04	0.57	68.9	1
Eastern Bering Sea	2000	132.27	110.04	0.00	66.87	0.83	68.5	1
Eastern Bering Sea	2001	120.40	128.48	0.00	62.83	1.18	68.7	1
Eastern Bering Sea	2002	129.22	119.00	0.00	64.75	0.39	67.5	1
Eastern Bering Sea	2003	200.42	133.08	0.00	67.07	1.17	71.5	1
Eastern Bering Sea	2004	106.54	145.39	0.00	70.08	2.10	64.5	1
Eastern Bering Sea	2005	139.98	159.51	0.00	75.00	2.55	66.5	1
Eastern Bering Sea	2006	86.82	141.53	0.00	70.82	0.59	63.4	1
Eastern Bering Sea	2007	113.60	128.18	0.00	64.77	0.70	66.3	1
Eastern Bering Sea	2008	81.95	128.88	0.00	65.65	1.80	63.1	1
Southern Gulf of St Lawrence	1971	55.63	61.52	0.00	0.00	8.52	121.6	1
Southern Gulf of St Lawrence	1972	63.78	73.05	0.00	0.00	12.16	119.0	1
Southern Gulf of St Lawrence	1973	63.31	74.95	0.00	0.00	15.97	113.3	1
Southern Gulf of St Lawrence	1974	63.76	127.47	0.04	0.00	14.52	102.9	1
Southern Gulf of St Lawrence	1975	42.96	95.34	0.00	0.00	7.23	106.3	1
Southern Gulf of St Lawrence	1976	49.45	141.25	0.00	0.00	2.54	106.3	1
Southern Gulf of St Lawrence	1977	70.35	154.61	0.02	0.00	4.39	109.4	1
Southern Gulf of St Lawrence	1978	127.53	107.72	0.00	0.00	6.61	136.6	1

Southern Gulf of St Lawrence	1979	146.36	141.06	0.00	0.00	1.36	134.1	1
Southern Gulf of St Lawrence	1980	166.62	119.09	0.00	0.00	1.69	143.7	1
Southern Gulf of St Lawrence	1981	251.21	125.42	0.00	0.00	1.50	152.7	1
Southern Gulf of St Lawrence	1982	206.59	72.52	0.01	0.00	1.91	164.2	1
Southern Gulf of St Lawrence	1983	150.67	84.87	0.00	0.00	0.76	150.4	1
Southern Gulf of St Lawrence	1984	123.79	47.04	0.01	0.00	10.41	154.1	1
Southern Gulf of St Lawrence	1985	212.80	61.88	0.01	0.00	20.03	157.8	1
Southern Gulf of St Lawrence	1986	174.56	80.95	0.01	0.00	18.32	145.9	1
Southern Gulf of St Lawrence	1987	132.32	61.62	0.02	0.00	20.63	138.2	1
Southern Gulf of St Lawrence	1988	199.90	98.82	0.02	0.00	13.78	147.2	1
Southern Gulf of St Lawrence	1989	158.67	77.43	0.02	0.00	13.60	143.7	1
Southern Gulf of St Lawrence	1990	118.25	90.34	0.04	0.00	30.64	117.6	1
Southern Gulf of St Lawrence	1991	78.36	75.93	0.06	0.00	37.27	115.4	1
Southern Gulf of St Lawrence	1992	54.36	67.96	0.04	0.00	15.36	105.5	1
Southern Gulf of St Lawrence	1993	72.40	44.62	0.08	0.00	11.42	130.8	1
Southern Gulf of St Lawrence	1994	49.24	41.49	0.08	0.00	15.72	122.8	1
Southern Gulf of St Lawrence	1995	59.35	38.62	0.11	0.00	25.22	122.7	1
Southern Gulf of St Lawrence	1996	61.18	36.41	0.08	0.00	5.15	144.4	1
Southern Gulf of St Lawrence	1997	53.46	27.67	0.11	0.00	17.77	131.3	1
Southern Gulf of St Lawrence	1998	46.22	30.80	0.09	0.00	7.64	137.8	1
Southern Gulf of St Lawrence	1999	61.69	31.76	0.10	0.00	17.88	132.5	1
Southern Gulf of St Lawrence	2000	49.13	36.60	0.07	0.00	12.91	126.7	1
Southern Gulf of St Lawrence	2001	42.19	33.51	0.05	0.00	18.67	121.4	1
Southern Gulf of St Lawrence	2002	73.38	31.24	0.12	0.00	22.94	137.3	1
Southern Gulf of St Lawrence	2003	19.57	40.55	0.09	0.00	34.04	89.9	1
Southern Gulf of St Lawrence	2004	42.65	32.04	0.09	0.00	33.60	113.3	1
Southern Gulf of St Lawrence	2005	20.84	37.59	0.22	0.00	41.88	87.4	1
Southern Gulf of St Lawrence	2006	27.34	34.86	0.13	0.00	16.00	110.3	1
Southern Gulf of St Lawrence	2007	27.00	31.43	0.16	0.00	124.62	72.3	1
Georges Bank	1963	126.86	51.91	0.07	0.02	1.51	107.2	1
Georges Bank	1964	144.93	28.28	0.01	0.35	1.59	109.7	1
Georges Bank	1965	111.44	32.61	0.03	0.86	0.87	105.4	1
Georges Bank	1966	64.16	25.32	0.25	0.06	1.45	105.0	1
Georges Bank	1967	49.20	22.87	0.24	0.63	1.23	110.4	1
Georges Bank	1968	37.96	26.89	0.10	0.71	1.28	102.2	1
Georges Bank	1969	19.01	24.43	0.22	1.27	0.46	92.7	1
Georges Bank	1970	37.89	27.13	0.13	1.53	0.33	101.8	1
Georges Bank	1971	19.65	19.74	0.29	1.53	1.43	96.0	1
Georges Bank	1972	48.18	23.36	0.31	1.35	1.56	119.3	1
Georges Bank	1973	94.09	40.32	0.14	4.83	0.46	121.0	1
Georges Bank	1974	24.79	21.39	0.45	2.37	1.12	96.1	1
Georges Bank	1975	46.42	31.01	0.34	2.71	1.38	112.7	1
Georges Bank	1976	126.31	27.60	0.03	11.58	2.25	119.2	1
Georges Bank	1977	71.73	38.46	0.13	3.90	0.46	103.8	1
Georges Bank	1978	106.70	34.12	0.67	8.39	0.79	122.0	1
Georges Bank	1979	141.93	28.70	0.07	8.07	1.24	124.8	1
Georges Bank	1980	63.44	33.44	0.23	4.73	0.87	105.4	1
Georges Bank	1981	118.55	31.20	0.22	3.40	3.85	124.4	1
Georges Bank	1982	75.16	31.87	0.20	2.82	1.26	114.1	1
Georges Bank	1983	116.65	19.62	0.04	3.08	3.02	123.2	1

Georges Bank	1984	146.96	26.25	0.06	3.61	1.02	127.6	1
Georges Bank	1985	139.56	17.30	0.17	3.47	4.74	127.3	1
Georges Bank	1986	164.11	20.37	0.22	6.09	1.92	124.7	1
Georges Bank	1987	155.61	20.43	0.10	1.48	1.60	132.1	1
Georges Bank	1988	112.97	15.83	0.06	7.73	5.94	122.6	1
Georges Bank	1989	75.12	25.40	0.18	8.98	0.87	113.1	1
Georges Bank	1990	213.32	21.00	0.50	3.85	3.16	138.3	1
Georges Bank	1991	96.57	14.07	0.06	8.16	2.50	127.0	1
Georges Bank	1992	101.35	17.62	0.13	4.52	4.75	129.5	1
Georges Bank	1993	76.39	16.47	0.20	6.19	7.80	120.2	1
Georges Bank	1994	35.11	17.33	0.09	7.39	5.47	102.4	1
Georges Bank	1995	87.50	23.96	0.04	3.30	10.02	118.1	1
Georges Bank	1996	67.63	22.92	0.17	1.27	6.18	116.6	1
Georges Bank	1997	86.94	26.06	0.42	2.42	5.46	122.6	1
Georges Bank	1998	149.64	43.51	0.08	2.33	6.17	126.4	1
Georges Bank	1999	99.20	26.13	0.49	8.79	3.41	117.4	1
Georges Bank	2000	69.82	34.02	0.07	6.68	3.97	103.1	1
Georges Bank	2001	117.70	40.81	0.21	3.05	4.52	118.8	1
Georges Bank	2002	177.88	39.65	0.07	5.77	3.20	119.4	1
Georges Bank	2003	110.91	29.37	0.11	4.32	8.16	120.2	1
Georges Bank	2004	169.92	25.50	0.20	1.70	2.85	123.3	1
Georges Bank	2005	280.75	20.23	0.14	3.57	1.96	141.9	1
Georges Bank	2006	179.32	24.69	0.86	4.42	2.92	132.6	1
Georges Bank	2007	195.69	25.95	0.37	3.71	3.09	131.3	1

Table S7. Number of stock assessments included in the collapse analyses, and the number of these that were "collapsed", i.e. with biomass less than 20% of B_{MSY} (data plotted in Fig. 4M-P). Where surplus production model fits were used to obtain B_{MSY}, this definition of collapse corresponds to total biomass falling below 10% of pre-exploitation biomass.

	All data		Eastern Bering Sea		Eastern Canada		Northeast U.S. Shelf	
Year	Stocks	Collapsed	Stocks	Collapsed	Stocks	Collapsed	Stocks	Collapsed
1950	24	0	-	-	-	-	-	-
1951	30	1	-	-	-	-	-	-
1952	33	0	-	-	-	-	-	-
1953	33	0	-	-	-	-	-	-
1954	33	0	-	-	-	-	-	-
1955	35	0	-	-	-	-	-	-
1956	38	0	-	-	-	-	-	-
1957	39	0	-	-	-	-	-	-
1958	41	0	-	-	-	-	-	-
1959	47	1	-	-	-	-	-	-
1960	51	2	-	-	-	-	-	-
1961	54	2	-	-	-	-	-	-
1962	54	2	-	-	-	-	-	-
1963	58	2	-	-	-	-	-	-
1964	65	1	-	-	-	-	-	-
1965	65	0	-	-	-	-	-	-
1966	68	1	-	-	-	-	-	-
1967	69	2	-	-	-	-	-	-
1968	73	3	-	-	-	-	-	-
1969	75	3	-	-	-	-	-	-
1970	80	2	-	-	-	-	-	-
1971	83	1	-	-	-	-	-	-
1972	93	1	-	-	-	-	-	-
1973	99	1	-	-	-	-	-	-
1974	102	0	-	-	-	-	-	-
1975	106	2	-	-	6	1	-	-
1976	109	3	-	-	6	1	-	-
1977	121	4	13	1	7	1	-	-
1978	131	3	14	1	8	0	10	1
1979	134	5	15	1	8	0	10	2
1980	139	3	15	1	8	0	11	1
1981	143	1	17	1	8	0	12	0
1982	147	0	17	0	8	0	14	0
1983	149	1	17	0	8	0	14	0
1984	156	5	17	2	8	0	14	1
1985	158	5	18	2	8	0	14	1
1986	163	6	18	2	8	0	14	1
1987	164	6	18	1	8	0	14	2
1988	164	7	18	1	8	0	14	3
1989	165	9	19	2	8	1	14	3
1990	166	5	20	0	8	1	14	3
1991	166	8	20	0	8	2	14	3
1992	166	10	20	0	8	3	14	3

1993	166	14	20	0	8	4	14	5
1994	166	13	20	0	8	4	14	3
1995	166	13	20	0	8	4	14	3
1996	166	10	20	0	8	3	14	3
1997	166	12	20	1	8	3	14	4
1998	166	11	20	1	8	3	14	4
1999	166	13	20	1	8	3	14	3
2000	166	13	20	2	8	3	14	2
2001	165	13	20	1	8	4	14	2
2002	163	14	20	1	8	4	14	2
2003	162	17	20	1	8	5	14	2
2004	159	19	20	1	8	5	11	3
2005	153	19	20	1	6	4	11	3
2006	148	21	20	1	6	4	10	3
2007	81	11	20	1	-	-	-	-

Supporting Figures

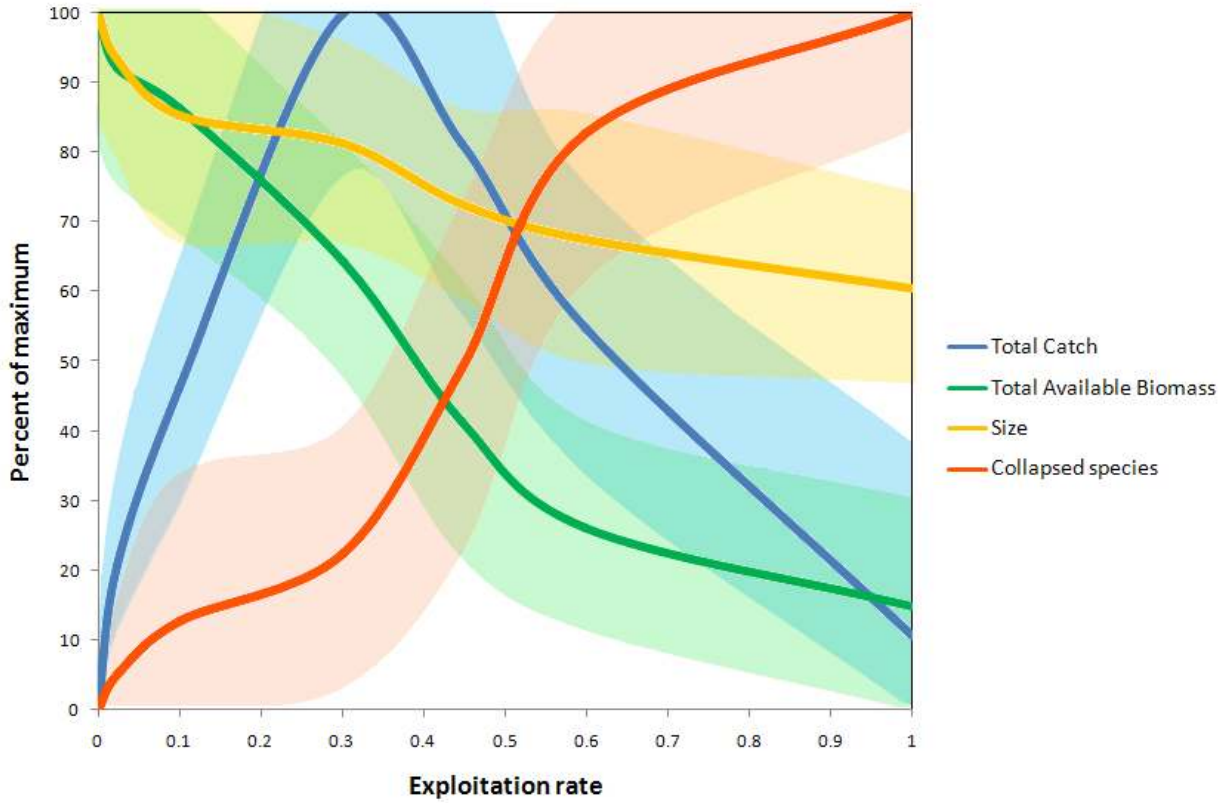

Figure S1: Effects of increasing exploitation rate on 31 model fish communities. Averaged results of ECOSIM models are displayed, shades refer to 95% confidence bounds.

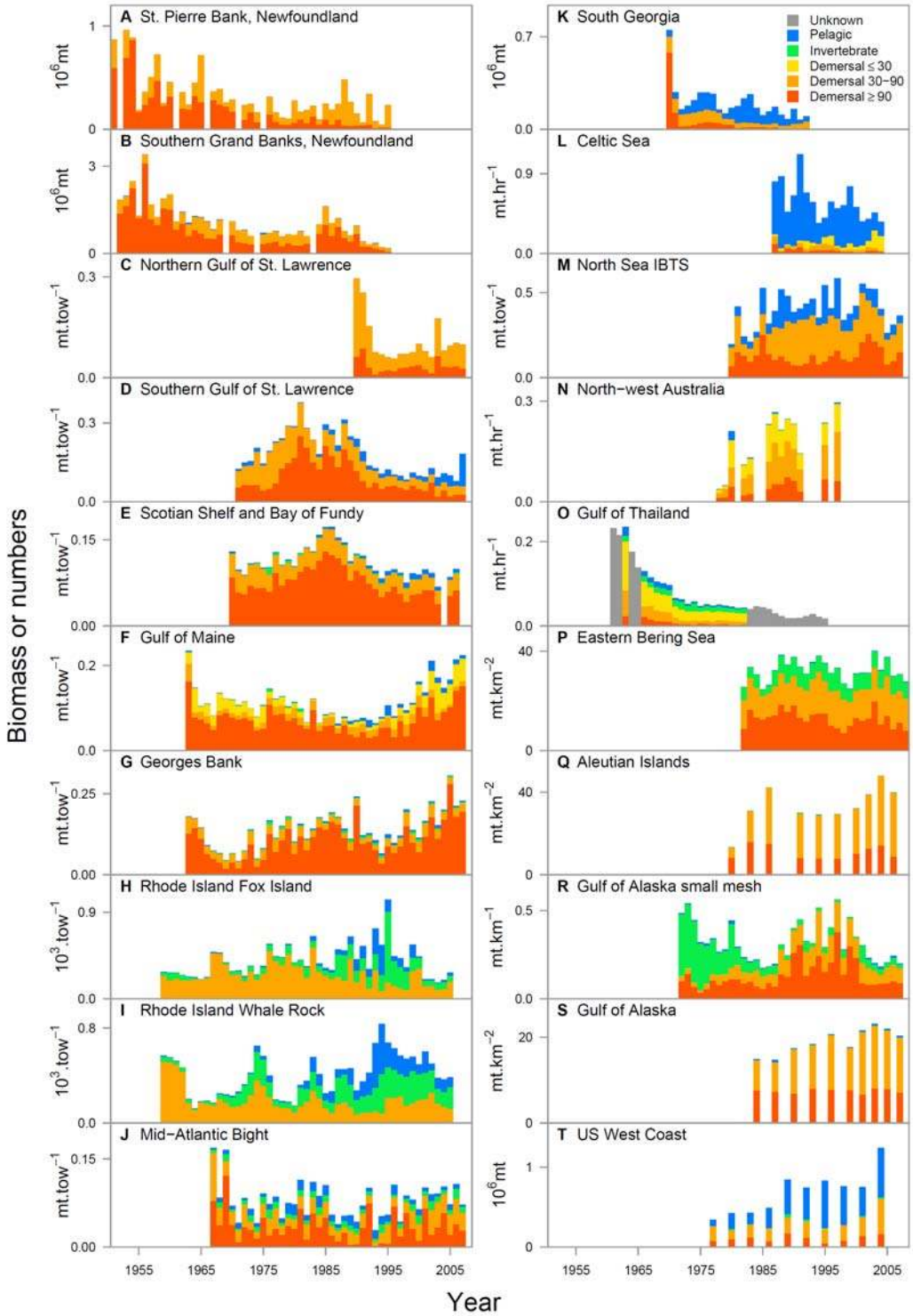

Figure S2. Available survey biomass estimates from 20 ecosystems. Data are grouped into five categories: invertebrate, pelagic (midwater species), and demersal taxa with maximum lengths of ≤30 cm, 30-90 cm, and ≥90 cm.

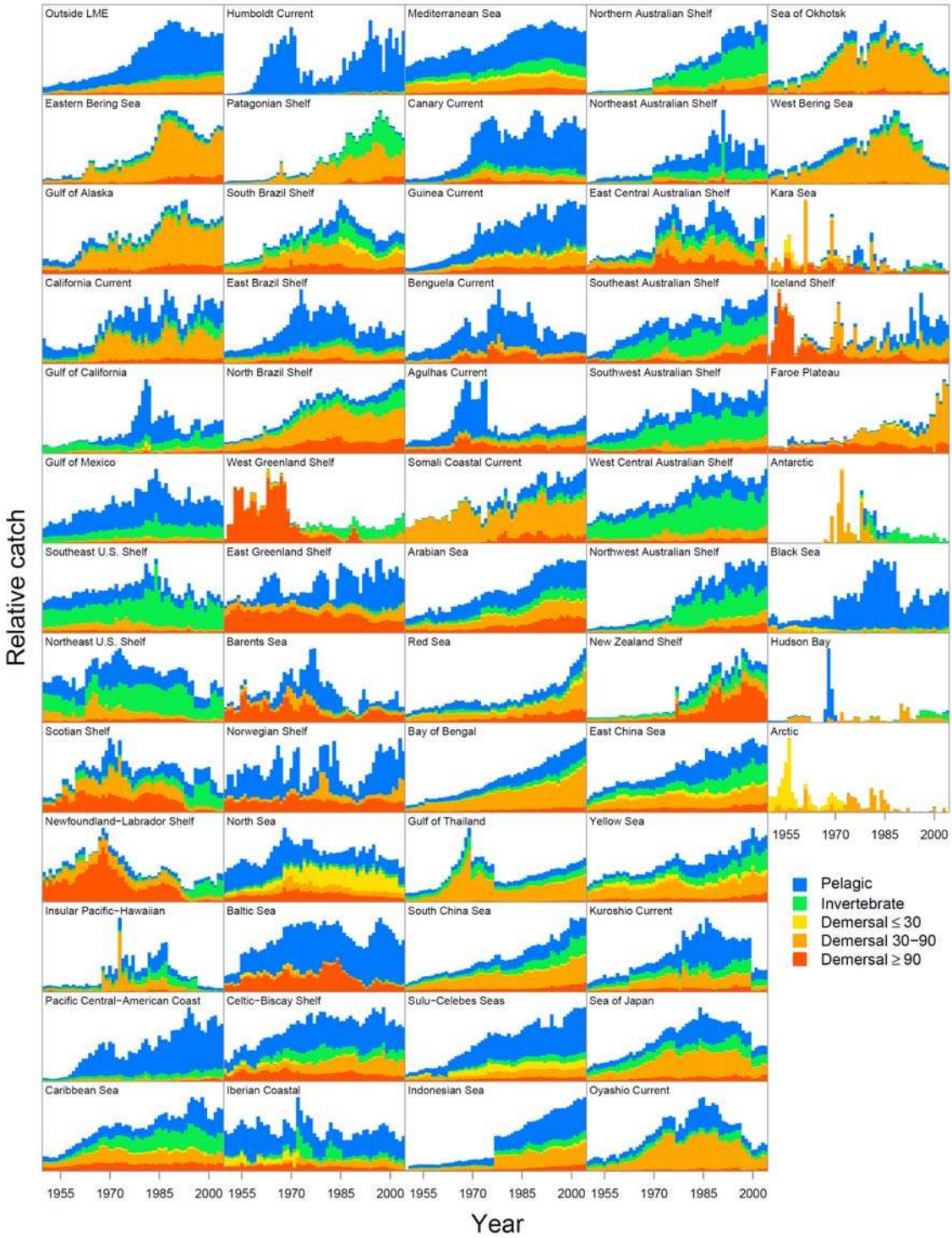

Figure S3: Global catches from the Sea Around Us database, reported by Large Marine Ecosystem and species group.

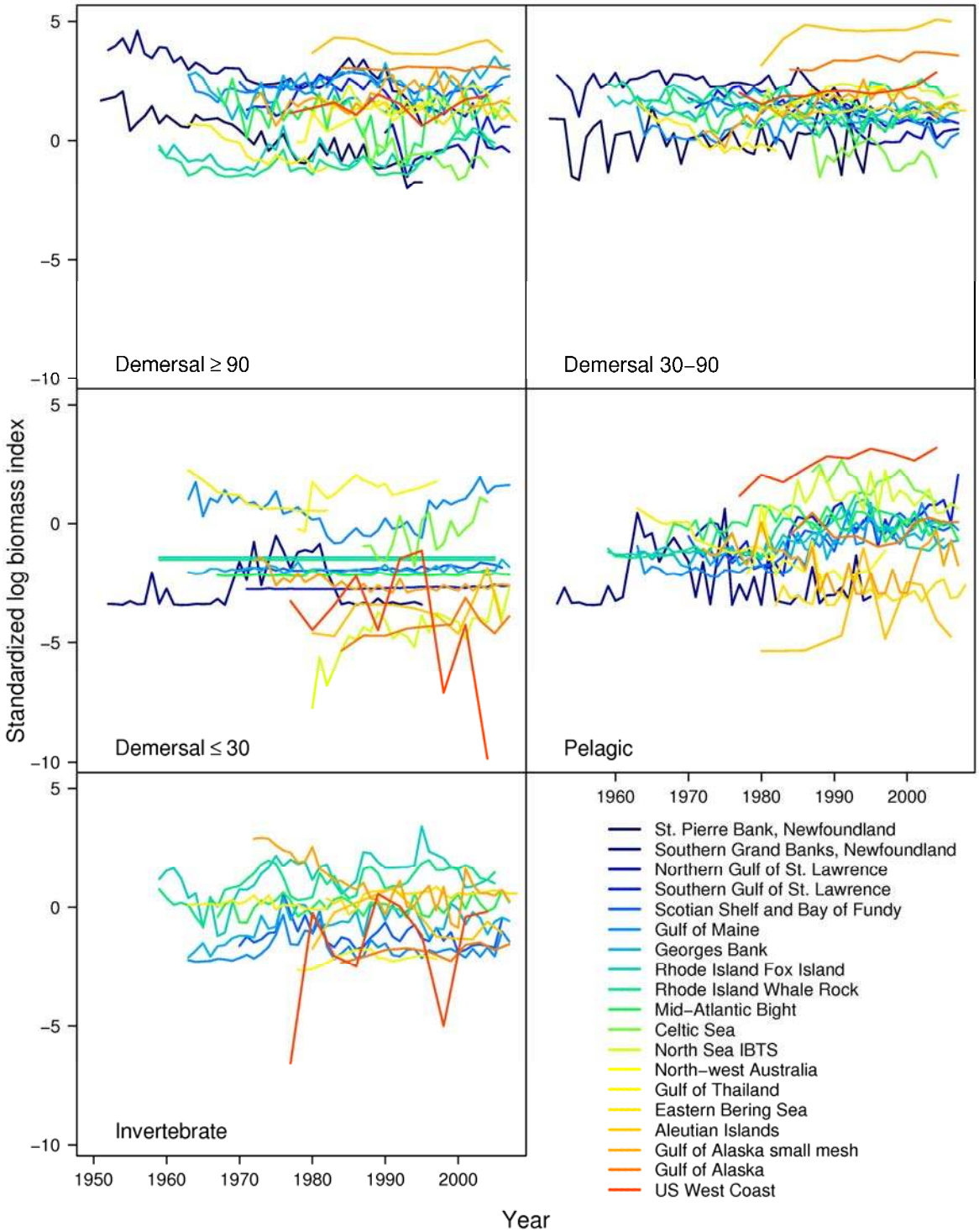

Figure S4: Standardized log biomass indices for each survey and category over time.

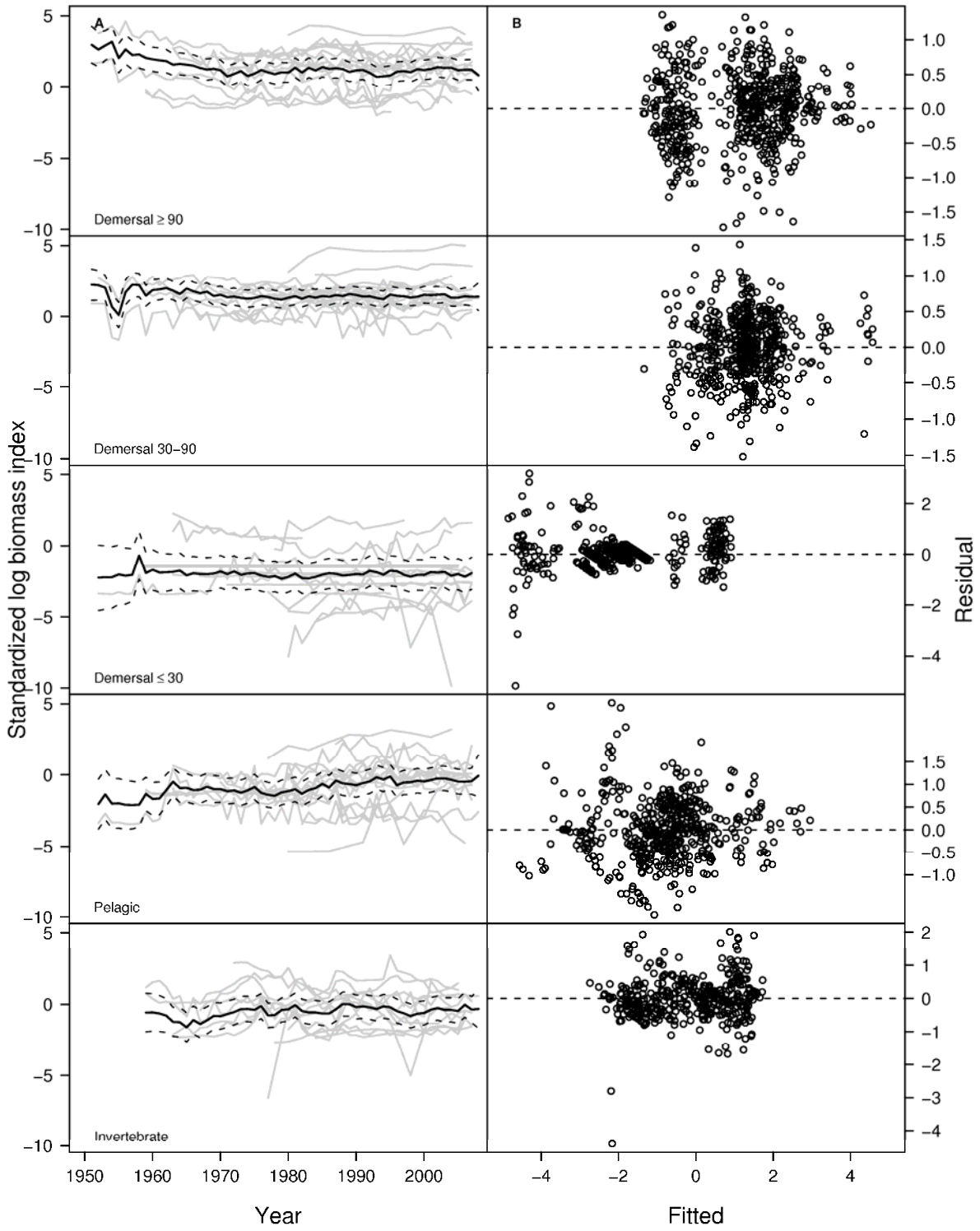

Figure S5: Average trend in survey biomass trends as estimated using a linear mixed effects analysis with a continuous AR(1) within-group correlation structure (Equation 6). Solid and dashed black lines in panel (**A**) indicate overall fixed effect trends and 95% confidence intervals, respectively; grey lines represent individual survey trends. The fitted values and residuals are plotted in panel (**B**).

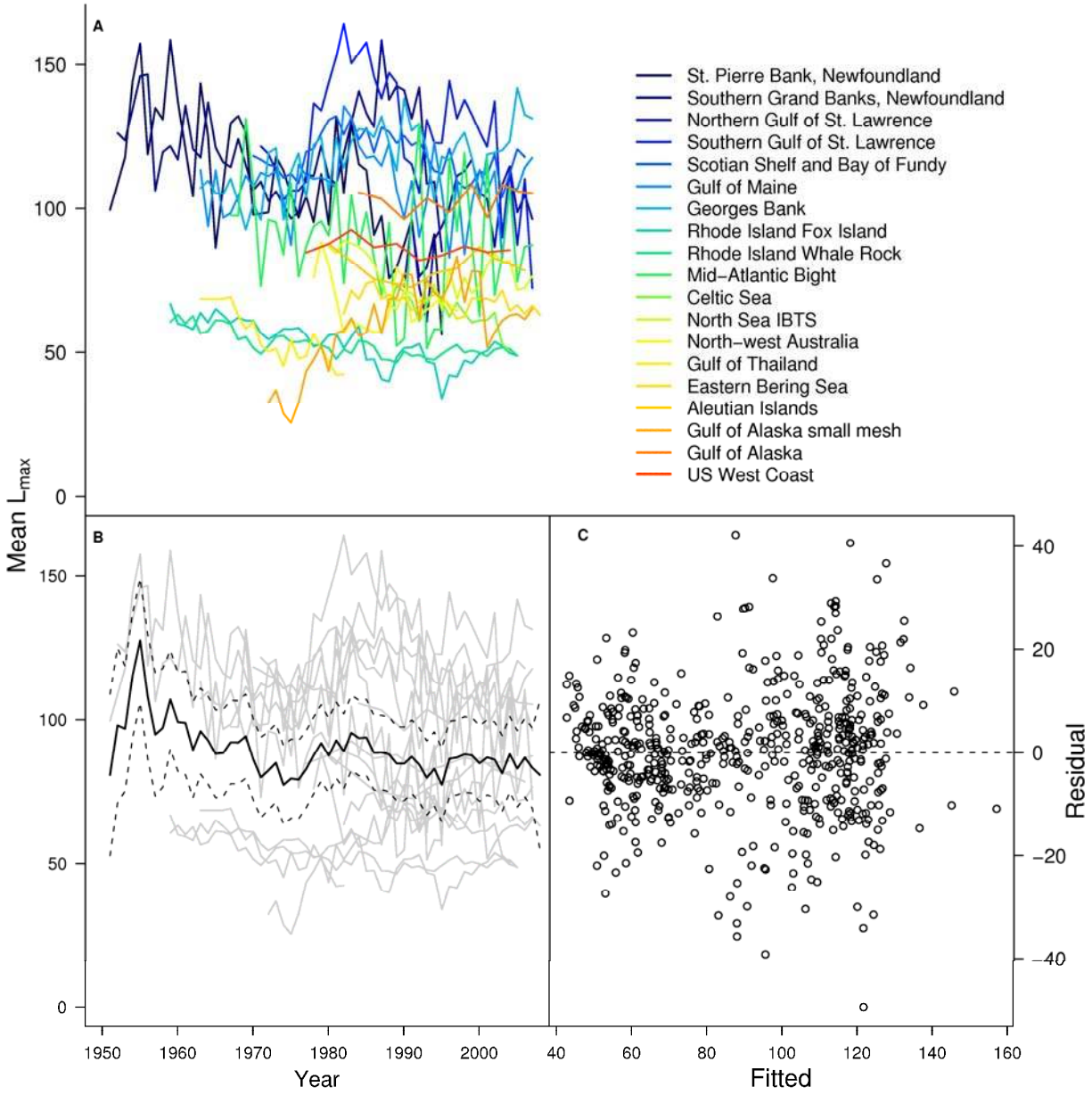

Figure S6. Average trend in $L_{\max}$, as estimated using a linear mixed effects analysis with a continuous AR(1) within-group correlation structure (Equation 10). Individual survey trends are presented in Panel (**A**). Solid and dashed black lines in panel (**B**) indicate overall fixed effects trends and 95% confidence intervals, respectively; grey lines represent individual survey trends.. The fitted values and residuals are plotted in panel (**C**).

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