So long, and thanks for all the fish:

The *Sea Around Us*, 1999-2014
A Fifteen-Year Retrospective

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A fifteen year retrospective

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And Wikipedia sayeth: "So Long, and Thanks for All the Fish is the fourth book of the *Hitchhiker's Guide to the Galaxy* trilogy written by Douglas Adams. Its title is the message left by the dolphins when they departed Planet Earth just before it was demolished to make way for a hyperspace bypass."

INTRODUCTION

This report is to celebrate the first 15 years of the *Sea Around Us* and to thank The Pew Charitable Trusts for having initiated and steadfastly supported this activity, which began as the '*Sea Around Us* Project' in mid-1999.

The project arose from a request by Dr. Joshua Reichert, the Director of the Environmental Program of The Pew Charitable Trusts, Philadelphia, that we answer six specific questions about the North Atlantic (and by extension throughout the world):

1. What are the total fisheries catches from the ecosystems, including reported and unreported landings and discards at sea?
2. What are the biological impacts of these withdrawals of biomass for the remaining life in the ecosystems?
3. What would be the likely biological and economic impacts of continuing current fishing trends?
4. What were the former states of these ecosystems before the expansion of large-scale commercial fisheries?
5. How do the present ecosystems rate on a scale from "healthy" to "unhealthy"?
6. What specific changes and management measures should be implemented to avoid continued worsening of the present situation and improve North Atlantic ecosystem's "health"?

In the first five years of its existence (summarized in Pauly, 2005), we documented that we had achieved these aims for the North Atlantic and were on our way to tackle the same issues for the world as a whole. Major steps in this were (*i*) papers in *Nature* (Pauly *et al.,* 2002) and *Science* (Pauly *et al.* 2003), documenting the main trends in global fisheries, and particularly demonstrating that the world catch, which was at the time supposed to be increasing (FAO, 2000), was actually declining, a fact then masked by exaggerated catch data from China (Watson and Pauly, 2001), (*ii*) a book on the state of fisheries and ecosystems in the North Atlantic Ocean (Pauly and Maclean, 2003), and (*iii*) the demonstration that the biomass of large fish in the North Atlantic had radically declined since the onset of industrial fisheries (Christensen *et al.,* 2003).

The successes of the *Sea Around Us*, whose mission was fully articulated in Pauly (2007) continued during the next 5 years, as documented in our 10-year retrospective report (Pauly, 2010). In it, we described the "*Top 10 Accomplishments of the Sea Around Us*" as follows:

Top 10 accomplishments in the first 10 years of the *Sea Around Us*

1. Created the first database in the world that maps catch and derived information, such as catch values onto to the Exclusive Economic Zones of specific countries, and Large Marine Ecosystems (see p. 39). This work has made the *Sea Around Us* website (www.seaaroundus.org) the key source of spatialized fisheries information for the international scientific and environmental communities, used by thousands of users every month;

2. Using the catch maps in (1) to establish that China, by over-reporting its catches, had masked a global decline of fisheries catches that started in the late 1980s. These results, published in the journal *Nature*, and later tacitly endorsed by FAO, provided the background for discussions about the global crisis of fisheries;

3. Debunked, via reports presented at meetings of the International Whaling Commission and a 'policy forum' article in *Science* (Gerber *et al.*, 2009), the assertion promoted by the pro-whaling community that marine mammals and fisheries globally compete for fish, and thus that the culling of whales would make more fish available to fisheries;

4. Estimated the extent of subsidies to the fishing industry on a global basis and by subsidy type (see p. 49). Dr. Rashid Sumaila, in collaboration with *Oceana*, an environmental NGO, was able to introduce these findings into WTO negotiations aimed at eliminating capacity-enhancing subsidies to fisheries;

5. Produced a series of papers investigating the successes and limitations of consumer awareness campaigns. This work, led by then Ph.D. student Jennifer Jacquet, was among the first to question the effectiveness of consumer awareness campaigns on the seafood industry, and highlighted obstacles to these efforts, such as product mislabeling, and lack of metrics for measuring campaign effectiveness;

6. Developed and applied a methodology for 'reconstructing' catch statistics from coastal countries (see p. 15), which generally yielded catch estimates much higher than those reported by the FAO (see pp. 109 & 113). Catch reconstructions, led by Dr. Dirk Zeller, have been completed for over 250 countries (or parts of countries, see p. 113). Results typically show that 'small-scale' fisheries contribute far more to the food security of developing countries than previously assumed, highlighting the need for a reassessment of policies that conventionally marginalize such fisheries;

7. Simulated, for the first time, the effect of climate change on fisheries and marine ecosystems on a global scale (see p. 63). Led by Dr. William Cheung, this initiative demonstrated in a continuing series of papers how increases in ocean temperatures may lead to massive shifts in marine biodiversity and estimated 'catch potentials' of coastal countries;

8. Developed, using Ecopath with Ecosim and associated software, a technique for integrating global ecological and fisheries datasets (see p. 71). The development of this "database-driven construction of ecosystem models", led by Dr. Villy Christensen, may represent the most data-intensive integration in marine ecology today;

9. Supported its principal investigator and main spokesperson, Dr. Daniel Pauly, as he became recognized as a leading voice for ocean conservation, as evidenced by his being awarded, e.g., the International Cosmos Prize (Japan, 2005), the Volvo Environment Prize (Sweden, 2006), the Excellence in Ecology Prize (Germany, 2007), the Ramon Margalef Prize in Ecology (Spain, 2008), and numerous honorary doctorates; and

10. Overall, the *Sea Around Us* turned into a respected voice on fisheries science, conservation, and policy, as achieved notably though its numerous articles in top journals (see p. 119) – many highlighted in the media – and other products.

In this period, our 'spatialization' of the catch database maintained and distributed by the Food and Agriculture Organization of the United Nation (FAO), available through our website (www.seaaroundus.org) began to be used by a large number of authors and research groups around the world, which led to numerous insights and publications, notably numerous articles published in *Science* and *Nature*.

Our publications during this period covered all the 6 questions mentioned above, but gradually, and this became stronger in the last decade, we began to realize that Question 1, i.e., "*What are the total fisheries catches from the ecosystems, including reported and unreported landings and discards at sea?*" was the most important of all, because everything else, including elaborating sound management policies depend on accurate catch data, including for fisheries that may be illegal (Belhabib *et al.*, 2014).

We also gradually realized that the 'catch' (actually 'landings') data disseminated by FAO and used more or less uncritically by all researchers working on international fisheries throughout the world are profoundly biased. This is because FAO member countries, which contribute their data on a voluntary basis (Garibaldi, 2012), often do not cover small-scale fisheries (which are not small; Zeller *et al.*, 2014), do not include discarded fish (although they have been caught) and do not attempt to estimate illegal and unreported catches. Rather, problematic fisheries are ignored, and their catches (which are never zero, otherwise they would not take place) are also ignored, and thus set at a figure of *precisely* zero. These data thus do not reflect catch trends, notwithstanding the title of the contribution by Garibaldi (2012).

At the very beginning of the *Sea Around Us*, we had started various attempts at complementing and correcting official catch statistics (see, e.g., contributions in Zeller *et al.*, 2001, 2003). However, it was only in its third phase, from 2009 to 2014, that we fully realized the extent of this bias and the need for us to address it in a systematic fashion.

The way we chose to address this issue was through a "*Global Atlas of Marine Fisheries: Ecosystem Impacts and Analysis*" (Pauly and Zeller, in press) that builds on the bottom-up 'catch reconstructions' that we began formally when the Western Pacific Regional Fishery Management Council, the management body responsible for the management of US fisheries in the Central Pacific, asked us to reconstruct the catch of the US flag territories under its purview (see Zeller *et al.* 2005, 2006, 2007).

The over 200 reconstruction reports (of 10-30 pages each) and datasets that this required, prepared and written by *Sea Around Us* team members and our international network of collaborators, were completed as of December 2014. The material for preparing the "*Global Atlas of Marine Fisheries: Ecosystem Impacts and Analysis*" will be sent to its publisher, Island Press, Washington, D.C., in early 2015.

Our "*Global Atlas*" documents the fisheries of the world in the form of over 250 one-page 'micro-chapters', each summarizing a single reconstruction documenting the fishery and total catches from 1950 to 2010 in the Exclusive Economic Zones (EEZs) of smaller maritime countries, or parts of the EEZs of larger maritime countries, or the EEZs around their island territories. Also, all catches are assigned to fishing sector, being either commercial (i.e., industrial or artisanal) or non-commercial (i.e., subsistence or recreational), described as either landed or discarded, foreign or domestic, and previously reported (e.g., by FAO) or not (i.e., 'IUU'), besides being identified to the finest possible taxonomic composition.

The present 15-year retrospective report presents a random selection of 5 of these micro-chapter summaries (see pp. 113 onwards), along with summaries of 12 of the 14 global chapters that also make up this Atlas, with each summarizing our work on one aspect of marine science and conservation (see p. 9 onwards). These global summaries, however, do not cover the whole range of our work, which is best appreciated by consulting our list of contributions to the scientific literature from the second half of 1999 to the present (see p. 119).

Jointly, these two types of summaries will give an impression of the *Atlas* that will become available in 2015, and which we hope will 'reset' various debates about the scope and status of global marine fisheries.

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We sincerely thank The Pew Charitable Trusts for supporting us for 15 years. The fundamental trust that this support reflects was extremely valuable to us. It made us feel appreciated, and resulted in more effective work. It enabled us to be creative and to think big, i.e., to tackle the long-term global fisheries issues that none of our colleagues could address, all without being monitored using short-term 'metrics'.

We would like to thank Ms. Rebecca Rimel, President and CEO of The Pew Charitable Trusts for the long-term support, Dr. Joshua Reichert for his inspiration (and his friendship with D.P.) and for formulating the six-point mission statement mentioned above, which has been our guiding star throughout, and Dr. Rebecca Goldburg for skillfully mediating between the different styles of an environmental advocacy organization and a university-based research group. We also wish to thank the many dedicated Pew staffers with whom we established excellent relationships throughout the years and with whom we hope to continue collaborating in the future, if under different circumstances.

From 2015 on, the Paul G. Allen Family Foundation will provide the bulk of the support for the *Sea Around Us*, their funding having overlapped with the finishing grant of The Pew Charitable Trusts since July 2014. This has enabled a smooth transition from support by The Pew Charitable Trusts, for which we are extremely thankful.

GLOBAL SUMMARIES

On the importance of fisheries catches [1]

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Fishing must generate a catch, whether it is done by West African artisanal fishers supplying a teeming rural market, by the huge trawler fleets in Alaska that supply international seafood markets, by women gleaning on a reef flat in the Philippines to feed their families, or by an Australian angler bragging about his catch in a bar. There is no point fishing if not for generating a catch (except perhaps when fishing for subsidies; see Sumaila *et al.* this report, p. 49). Indeed, the catch of a fishery and its monetary value both define that fishery and provide the metric by which to assess its importance relative to other fisheries and other sectors of the overall economy. Hence, changes in the magnitude and species composition of catches obviously can and should be used – along with other information (e.g., on life history of the fish that are exploited) – for inferences on the status of fisheries.

The key role of catch data is the reason why the Food and Agriculture Organization of the United Nations (FAO) proceeded, soon after it was founded in 1945, to issue occasional compendia of the world's fishery statistics, as part of the United Nations' attempt to "quantify the world" (Ward, 2004). These compendia turned, in 1950, into the much-appreciated FAO *Yearbook of Fisheries Catch and Landings*, based on annual submissions by its member countries, vetted and harmonized by its staff. Contrary to the situation prevailing for major food crops (rice, wheat, maize, etc.), for which we have numerous international databases, the contents of the *Yearbooks* (now available online; see www.fao.org) have been, to this day, the only global database on wild caught 'fish' (i.e., including invertebrates and other marine groups such as, e.g., edible algae). As such, it is widely cited as the major source for inferences on the status of fisheries in the world (Garibaldi, 2012).

However, in many countries, particularly in the developing world, the governments' role in monitoring their fisheries seems to end with the annual ritual of filling-in catch report forms and sending them to FAO, as parodied in Marriott (1984). For others, mainly developed countries, collecting catch data from fishing ports and markets is only a start, and the bulk of their fishery-related research is in the form of formal stock assessments.

Stock assessments, which are usually performed annually, are, however, extremely expensive, ranging from an estimated 50,000 USD per stock (assuming 6 expert-months for analyzing existing data) to millions of USD when fisheries-independent data are required (Pauly, 2013).

[1] Abbreviated from: Pauly, D. On the importance of fisheries catches, with a rationale for their reconstruction, p. XX *In:* D. Pauly and D. Zeller (eds.) *Global Atlas of Marine Fisheries: Ecosystem Impacts and Analysis*. Island Press, Washington, D.C. [*in press*]

This is the reason, beside a worldwide scarcity of domain expertise, why at most 20% of the over 200 current maritime countries and associated island territories perform regular stock assessments, which, moreover, deal only with the most abundant or economically most valuable species they exploit. For some countries or territories, this may be one species, a dozen, or about two hundreds, as in the USA. In all cases, however, this is only a small fraction of the number of species that are exploited, if only as unintended 'by-catch' and often discarded.

This situation can be partly mitigated by detailed analyses of the results of 'catch reconstructions', for which I provided the rationale just before the *Sea Around Us* originated (Pauly, 1998), and from which the following is adapted:

A frequent response [to the lack of information that this situation generates] has been to set up intensive, but relatively short-term projects devoted to improving national data reporting systems. Their key products are detailed statistics covering the (few) years of the project. However, these data are usually hard to interpret, given the frequent absence of data from previous periods, from which changes could be evaluated. This is a major drawback, as it is the changes occurring within a contrast-rich dataset which provide the basis from which trends in the status of the resources supporting various fisheries can be determined.

It is thus evident that reconstructing past catches and catch compositions is a crucial activity for fisheries scientists and officers in the Caribbean or the Pacific, and that such activity is required to fully interpret the data emanating from current data collection projects. This may be illustrated as follows: suppose that the Fisheries Department of Country A establishes, after a large and costly sampling project, that its reef fishery generated catches of 5 and 4 $t{\cdot}km^{-2}{\cdot}year^{-1}$ for the years 1995 and 1996, respectively. The question now is: are these catch figures high values, allowing an extension of the fishery, or low values, indicative of an excessive level of effort?

Clearly, one approach would be to compare these figures with those of adjacent Countries B and C. However, these countries may lack precise statistics, or have fisheries which use a different gear. Furthermore, A's Minister in Charge of Fisheries may be hesitant to accept conclusions based on comparative studies, and require local evidence before taking important decisions affecting her country's fisheries. One approach to deal with this very legitimate requirement is to reconstruct and analyze time series, covering the years preceding the recent period for which detailed data are available, and going as far back in time as possible (e.g., to the year 1950, when the above-mentioned annual FAO statistics begin). With such data covering the early period of fisheries, it is then possible to quickly evaluate the status of fisheries and their supporting resources, and to evaluate whether further increases of effort will be counterproductive or not.

Basic methodology for catch and effort reconstruction

The key part of the methodology proposed here is psychological: one must overcome the notion that "no information is available", which is the wrong default setting if dealing with an industry such as fisheries. Rather, one must realize that fisheries are social activities, bound to throw large 'shadows' onto the societies in which they are conducted. Hence, records usually exist that document some aspects of these fisheries. All that is required is to find them and to judiciously interpret the data they contain.

Important sources for such an undertaking are:

1. Old files of the Department of Fisheries;
2. Peer-reviewed journal articles;
3. Theses, scientific and travel reports, accessible in departmental or local libraries or branches of the University of the West Indies or the University of the South Pacific, or through regional databases;
4. Records from harbor master and other maritime authorities with information on number of fishing crafts (small boats by type; large boats by length class and/or engine power);
5. Records from the cooperative or private sectors (companies exporting fisheries products, processing plants, importers of fishing gear, etc.);
6. Old aerial photos from geographic or other surveys (to estimates numbers of boats on beaches and along piers); and last, but not least
7. Interviews with old fishers.

Estimating catches

Analysis of the scattered data obtained from (1)-(7) should be based on the simple notion that catch in weight (Y) is the product of catch/effort (U; also known as 'CPUE') times effort (f), or

$$Y = U \cdot \mathrm{f}$$

This implies that one should obtain from (1)-(7) estimates of the effort (how many fishers, boats or trips) of each gear type, and multiply it with the mean catch/effort of that gear type (e.g., mean annual catch per fisher, or mean catch per trip). As the catch/effort of small boats and of individual fishers will differ substantially from that of the larger boats, it is best to estimate annual catches by gear or boat type, with the total catch estimates then obtained by summing over all gear or boat types.

Moreover, as catch/effort usually varies with season, estimation of Y should preferably be done on a monthly basis whenever possible, by applying the above equation separately for every month, then adding the monthly catch values to obtain an annual sum. This should be repeated for every component of the fishery, e.g., for the small-scale and industrial components.

Once all quantitative information has been extracted from the available records, linear interpolations can be used to ‘fill in’ the years for which estimates are missing. For example, if one has estimated 1000 t as annual reef catch for 1950 and 4000 t for 1980, then it is legitimate to assume, in the absence of any information to the contrary, that the catches were about 2000 t in 1960 and 3000 t in 1970. This interpolation procedure may appear too daring; however, the alternative to this is to leave blanks (i.e., the all too common ‘no data’), which later will invariably be interpreted as catches of *zero*, which is a far worse estimate than interpolated values.

Estimating catch composition

Once catch time series have been established for distinct fisheries (nearshore/reef, shelf, oceanic, etc.), the job is to split these catches into their components, i.e., into distinct species or species groups. Comprehensive information on catch composition allowing this to be done will usually be lacking. Therefore, the job of splitting up the catches must be based on fragmentary information, such as the observed catch composition of a few, hopefully representative fishing units. Still, combining within periods of say five years all available anecdotal information on the catch composition of a fishery (i.e., observed composition of scattered samples) should allow the estimation of reasonable estimates of mean composition if use is made of the principle that, in the absence of further information on their relative contributions, equal probabilities can be assigned to the items jointly contributing to a whole. Thus, a report stating, say, that "the catches consisted of groupers, snappers, grunts and other fish" can be turned into 25% groupers, 25% snappers, 25% grunts, and 25% other fish as a reasonable first approximation.

A number of such approximations of catch composition can then be averaged into a representative set of percentages, which can be applied to the catches of the relevant period. These percent catch compositions can be interpolated in time, e.g., for 1950-1954 with a composition of 40% groupers, 20% snappers, 10% grunts and 30% other fish, and 10%, 10%, 20% and 60%, respectively for these same groups in 1960-1964. In this case, the values for the intermediate period (1955-1959) can be interpolated as 25% groupers, 15% snappers, 15% grunts, and 45% other fish.

Conclusions

Estimating catches from the catch/effort of selected gear and fishing effort is a standard method for fisheries management. Reconstruction of historic catches and catch compositions series may require interpolations and other bold assumptions, justified by the unacceptability of the alternative (i.e., accepting catches to be zero, or otherwise known to be incompatible with empirical data and historic records).

There is obviously more to reconstructing catch time series than outlined above, and some of the available methods are rather sophisticated (see Zeller *et al.*, this volume, p. 15). The major impediment to applying this methodology is that colleagues initially do not trust themselves to make the bold assumptions required to reconstruct unseen quantities such as historic catches. Yet it is only by making bold assumptions that we can obtain the historic catches required for comparisons with recent catch estimates, and thus infer major trends in fisheries.

One example may be given here. The FAO catch statistics for Trinidad & Tobago for the years 1950-1959 start at 1,000 t (1950-1952), then gradually increase to 2,000 t in 1959. Of this, 500-800 t was contributed by 'Osteichtyes', 300-500 t by '*Scomberomorus maculatus*' (now known as *S. brasiliensis*), 100-200 t by '*Penaeus* spp.', and 0-100 t by 'Perciformes' (presumably reef fishes). On the other hand, the same statistics report, for 1950-1959, catches of zero for *Caranx* spp.; Clupeoidae; *Katsuwonus pelamis*; *Sarda sarda* (not surprising, since it does not occur in Trinidad & Tobago, though it reported yielded catches of 21-35 t in 1983 and 1984); *Scomberomorus cavalla*; *Scomberomorus* spp.; *Thunnus alalunga*; and *T. albacares*.

Despite their obvious deficiencies, these and similar data from other Caribbean countries are commonly used to illustrate fisheries trends from the region. Fortunately, it is very easy to improve on this. Thus, Kenny (1955) estimated, based on detailed surveys at the major market (Port of Spain), and a few, quite reasonable assumptions, that the total catch from the island of Trinidad was in the order of 13 million pounds (2,680 t) in 1954/1955, i.e., about two times the FAO estimate, at this time, for both Trinidad & Tobago. Moreover, King-Webster and Rajkumar (1958) provide details of the small-scale fisheries existing on Tobago, from which fishing effort and a substantial catch can be estimated, notably of 'carite' (*Scomberomerus regalis*). Further, both of these sources include detailed catch compositions as well, indicating that several of the categories with entries of zero in the FAO statistics (e.g., the clupeoids) actually generated substantial catches in the 1950s. Other early sources exist which can be used to corroborate this point. Similar data sets exist in other Caribbean countries, and I look forward to their analysis by different colleagues.

The text of Pauly (1998) ended here, and this introductory summary chapter also will do so, as the global and country summaries presented in this report address the analyses I was hoping would be done in the Caribbean - and the rest of the world.

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Marine fisheries catch reconstruction: how to do it [2]

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It is now well-established that official fisheries catch data, for perfectly legitimate reasons have historically ignored certain sectors (e.g., the subsistence, or the recreational sectors) as well as fisheries discards, notably because landings data were collected in many cases for purposes of taxation or for development accounting under market economics.

Nowadays, however, when fisheries need to be managed in the context of the ecosystems in which they are embedded (Pikitch *et al.* 2004); less than full accounting for all withdrawals or mortality from marine ecosystems is insufficient. This contribution documents an effort to provide a time-series of all marine fisheries catches from 1950, the first year that the Food and Agriculture Organization of the United Nations (FAO) produced its annual compendium of global fisheries statistics to 2010, i.e., 61 years with sharply contrasting economic, political and environmental conditions.

This contribution deals with catches in Exclusive Economic Zones (EEZ), i.e., in about 40% of the world ocean (Figure 1), while the catches (mainly of tuna and other large pelagic fishes) made in the high seas, are dealt with by Le Manach *et al.* (this volume, see p. 25).

Methods and definitions, with emphasis on domestic catches

The fisheries catch reconstructions whose summaries form the core of the "*Global Atlas of Marine Fisheries: Ecosystem Impacts and Analysis*" (Pauly and Zeller, in press) are based on the concepts in Pauly (1998; see also this volume p. 9) and the methodology detailed by Zeller *et al.* (2007). The former contribution asserted (*i*) there is no fishery with 'no data' because fisheries, as social activities throw a shadow unto the other sectors of the economy in which they are embedded, and (*ii*) it is always worse to put a value of 'zero' for the catch of a poorly documented fishery than to estimate its catch, even roughly, because subsequent users of one's statistics will interpret the zeroes as 'no catches', rather than 'catches unknown'.

Zeller *et al.* (2007) developed a six-step approach for implementing these concepts, as follows:

1. Identification, sourcing and comparison of baseline catch times series, i.e., a) FAO (or other international reporting entities) reported landings data by FAO statistical areas (Figure 1), taxon and year; and b) national data series by area, taxon and year;
2. Identification of sectors (e.g., subsistence, recreational), time periods, species, gears etc., not covered by (1), i.e., missing data components. This is conducted via extensive literature searches and consultations with local experts;

[2] Abbreviated from: D. Zeller and D. Pauly. Marine fisheries catch reconstruction: definitions, sources, methodology and challenges, p. XX *In:* D. Pauly and D. Zeller (eds.) *Global Atlas of Marine Fisheries: Ecosystem Impacts and Analysis*. Island Press, Washington, D.C.[*in press*]

3. Sourcing of available alternative information sources on missing data identified in (2), via extensive searches of the literature (peer-reviewed and grey, both online and in hard copies) and consultations with local experts. Information sources include social science studies (anthropology, economics, etc.), reports, data sets and expert knowledge;
4. Development of data 'anchor points' in time for each missing data item, and expansion of anchor point data to country-wide catch estimates;
5. Interpolation for time periods between data anchor points, either linearly or assumption-based for commercial fisheries, and generally via *per capita* (or per-fisher) catch rates for non-commercial sectors; and
6. Estimation of total catch times series, combining reported catches (1) and interpolated, country-wide expanded missing data series (5).

Since these 6 points were originally proposed, a 7th point has come to the fore which cannot be ignored, and has been first applied in Zeller *et al.* (2014):

7. Quantifying the uncertainty associated with each reconstruction.

We briefly expand on each of these seven steps, based on the experience accumulated during the last eight to ten years, while completing or guiding the reconstruction work conducted by the *Sea Around Us*:

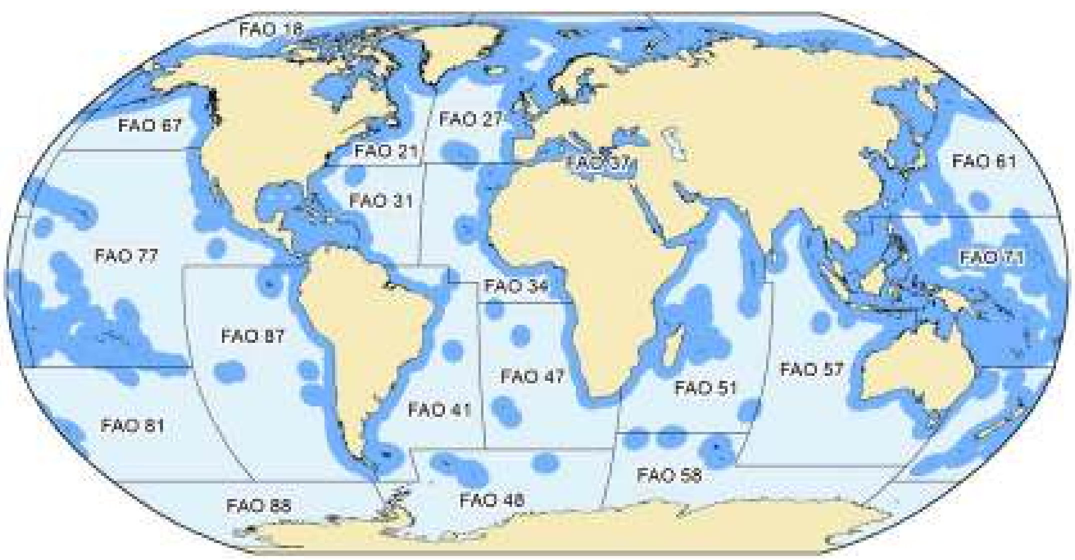

Figure 1. Extent and delimitation of countries' Exclusive Economic Zones (EEZs), as declared by individual countries, or as defined by the *Sea Around Us* based on the fundamental principles outlined in UNCLOS (i.e., 200 nautical miles or mid-line rules), and the FAO statistical areas by which global catch statistics are reported. Note that for several FAO areas some data exist by sub-areas as provided through regional organizations (e.g., ICES for FAO Area 27). The *Sea Around Us* makes use of these spatially refined data to improve the spatial allocation of catch data as described in Lam *et al.* (this volume, p. 39).

Step 1: Identification, sourcing and comparison of existing, reported catch times series

Implicit in this first step is that the spatial entity be identified and named that is to be reported on (e.g. EEZ or FAO area, Figure 1), something that is not always obvious, and which posed serious problems to some of our external collaborators.

For most countries, the baseline data are the statistics reported by member countries to FAO (and of whose existence a surprisingly large number of colleagues, especially in developing countries, are not aware). Whenever available, we also used data reported nationally for a first-order comparison with FAO data, which often assisted in identifying catches likely taken in areas beyond national jurisdiction, i.e., either in EEZs of other countries or in high seas waters. The reason for this is that many national datasets do not include catches by national distant-water fleets fishing and/or landing catches elsewhere. As FAO assembles and harmonizes data from various sources, this first-order comparison enabled catches 'taken elsewhere' to be identified and separated from truly domestic (national EEZ) fisheries (see Lam *et al.*, this volume, p. 39, for the spatial layering of reconstructed datasets).

For some countries, e.g., those resulting from the breakup of the USSR (Zeller and Rizzo, 2007), this involved using the post-breakup catch fractions to roughly split the pre-breakup catches reported for the former-USSR into the component new countries, as we treat all countries recognized in 2010 by the international community as having existed from 1950-2010. This was necessary, given our emphasis on 'places', i.e., on time-series of catches taken from specific ecosystems. Similar re-assignments of former-USSR catches was also undertaken for the global tuna and large pelagic dataset (see Le Manach *et al.*, this volume, see p. 25).

Step 2: Identification of missing sectors, taxa and gear

This step is one where the contribution of local co-authors and the input from local experts is crucial. Five sectors potentially exist in the marine fisheries of a given coastal country:

Industrial sector: this sector consists of relatively large motorized vessels, requiring large capital investments for their construction, maintenance and operation. These vessels operate either domestically, in the waters of other countries and/or the high seas, and land a catch that is overwhelmingly sold commercially (as opposed to being consumed and/or given away by the crew). The industrial sector can also be considered *large-scale and commercial* in nature;

Artisanal sector: this sector generally consists of smaller vessels requiring lower capital investments, and primarily utilizes small-scale (hand lines, gillnets etc.) or fixed gears (weirs, traps, etc.), and whose catch is predominantly sold commercially (notwithstanding a small fraction of this catch being consumed or given away by the crew). Note that, generally speaking, vessel size is not the sole classification category, as any fishing gear dragged through the water (e.g., mid-water trawl) or across the seafloor (e.g., bottom trawl, shrimp trawl), even if towed by a small, powered vessel, is deemed by us to be 'industrial' in nature (*sensu* Martín, 2012). Furthermore, our definition of artisanal fisheries relies also on adjacency: they are assumed to operate only in domestic waters (i.e., in their country's EEZ), and specifically are assumed to operate only within the Inshore Fishing Area (IFA), which is defined as the coastal waters up to either 50 km from shore or 200 m depth, whichever comes first (Chuenpagdee *et al.*, 2006).

Subsistence sector: consisting of fisheries that often are conducted by women (Harper *et al.*, 2013) for consumption by one's family. However, we also count as subsistence catch the fraction of the catch of mainly artisanal boats that is given away to the crew's family or the local community.

Recreational sector: consisting of fisheries conducted mainly for pleasure, although a fraction of the catch can be sold or consumed by the recreational fishers and their families and friends (Cisneros-Montemayor and Sumaila, 2010).

Finally, for all countries and territories, we account for *discards,* here treated as a 'catch type' (and is contrasted to 'retained landings'), and which overwhelmingly originate from industrial fisheries for the years 1950 to 2010. Discarded fish and invertebrates are generally assumed to be dead (since most discard estimates come from bottom and shrimp trawls with long shot times which reduce potential survival rates of subsequently discarded catch), except for the US fisheries where the fraction of fish and invertebrates reportedly surviving is often available on a per species basis (McCrea-Strub, in press).

For any country or territory we check whether catches (retained as well as discarded) originating from these four sectors are included in the reported baseline of catch data, notably by examining their taxonomic composition, and any metadata, which were particularly detailed in the early decades of the FAO yearbooks. Finally, if gears are identified in national data, but catch data from a gear known to exist in a given country are not included, then it can be assumed that its catch has been missed, as documented by Al-Abdulrazzak and Pauly (2013) for weirs in the Persian Gulf.

Step 3: Sourcing of available alternative information sources for missing data

The major initial source of information for catch reconstructions is governments' (and specifically their Department of Fisheries or equivalent agency) websites and publications, both online and in hard copies. Unlike what many fisheries scientists claim, there is no such thing as 'no data' for earlier periods, as publications (mainly governmemt reports or anthropological accounts) do exist, but often in less anticipated places. A good source of information for the earlier decades (especially the 1950s and 1960s) for countries that formerly were part of colonial empires (especially British or French) are the colonial archives in London and Paris (see Figure 2 for a visualization of the sources available over time). Contrary to what could be expected, it is often not the agency responsible for fisheries which supplies the catch statistics to FAO, but other agencies, e.g., the Trade Ministry, or some national statistical office, with the result that much of the granularity of the original data (i.e., catch by sector, by species or by gear) is lost even before it reaches Rome. Furthermore, the data request form sent by FAO each year to each country does not necessarily encourage improvements or changes in taxonomic composition, as the form contains the country's previous years' data in the same composition as submitted in earlier years, and requests the most recent year's data. This encourages the pooling of detailed data at the national level into the taxonomic categories inherited through earlier (often decades old) FAO reporting schemes (e.g., Bermuda, Luckhurst *et al.*, 2003). Thus, by getting back to the original data, much of the original granularity can be regained during reconstructions (e.g.,

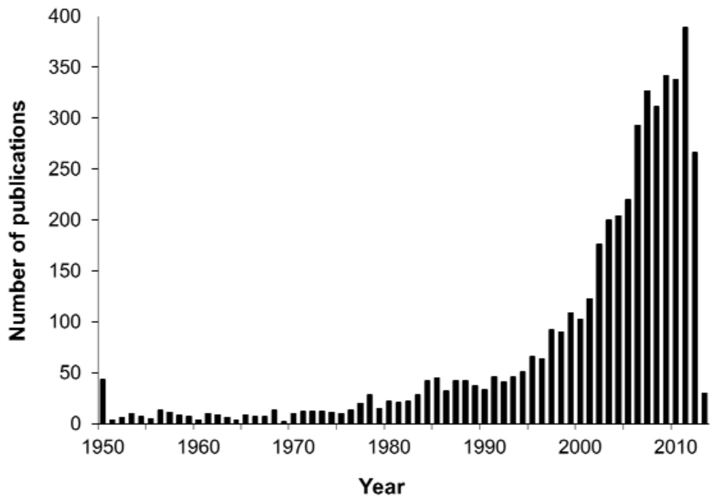

Figure 2. Number of publications (scientific and grey literature) and their publication date used for slightly over 110 country/territory catch reconstructions, i.e., less than half. A total of 4,000 publications (excluding personal communications and online sources) were consulted, resulting in an average of 35 publications being used per reconstruction. While more information sources are obviously available for recent time periods, this illustrates that publications are accessible for the entire time period. The slightly elevated number for 1950 is due to pooling of material dated pre-1950 (as far back as the early 20th century or even late 19th century) that was used conservatively to inform 1950 anchor point information.

Bermuda reconstruction, Teh *et al.*, 2014). A second major source of information on national catches are international research organizations such as FAO, ICES or SPC, or a Regional Fisheries Management Organization (RFMO) such as NAFO, or CCAMLR (Cullis-Suzuki and Pauly, 2010), or current or past regional fisheries development and/or management projects. Another source of information is obviously the academic literature, now widely accessible through Google Scholar.

Our global network of local collaborators was also crucial in this respect, as they had access to key data sets, publications and local knowledge not available elsewhere. Of particular note here is the relevance of searching for material and publications in local languages (e.g., Japanese, Korean, Russian, Arabic, Spanish, Portuguese etc.), either through us employing staff that speak these languages or engaging with local collaborators.

Step 4: Development and expansion of 'anchor points'.

'Anchor' points are catch estimates usually pertaining to a single year and sector, and often to an area not exactly matching the limits of the EEZ or IFA in question. Thus, an anchor point pertaining to a fraction of the coastline of a given country may need to be expanded to the country as a whole, using fisher or population density, or relative IFA or shelf area as raising factors, as appropriate given the local condition.

Step 5: Interpolation for time periods between anchor points.

Fishing, as a social activity involving multiple actors is very difficult to govern; particularly, fishing effort is difficult to reduce, at least in the short term. Thus, if anchor points are available for years separated by multi-year intervals, it will be usually more reasonable to assume that the underlying fishing activity went on in the intervening years with no data. Strangely enough, the 'continuity' assumption we take as default is something that many colleagues are reluctant to make, which is the reason why the catches of, e.g., small-scale fisheries monitored intermittingly often have jagged time-series of reported catches. Exceptions to such continuity assumptions are obvious major environmental impacts such as hurricanes or tsunamis, or major socio-political events that impact fishing opportunity, such as civil wars (e.g., Liberia; Belhabib *et al.*, 2013).

Step 6: Estimation of total catch times series by combining (1) and (5).

A reconstruction is completed when the estimated catch time-series derived through Steps 2-5 are combined and harmonized with the reported catch of Step 1.

Step 7: Quantifying the uncertainty in (6)

We apply to our recent reconstructions an approach inspired from the 'pedigrees' of Funtowicz and Ravetz (1990) and the approach used by the Intergovernmental Panel on Climate Change to quantify the uncertainty in its assessments (Mastrandrea *et al.,* 2010). This involves assigning a score to each catch estimate (by sector) expressing the quality of the time series, i.e., (1) 'very high', (2) 'high', (3) 'low' and (4) 'very low'. (Note the absence of a 'medium' score, to avoid the non-choice that this easy option would represent). The overall score for the reconstructed total catch of a period is then computed as the mean of the scores for each sectors, weighted by their catch, and confidence intervals assigned to each overall average score.

Foreign and illegal catches

Foreign catches are catches taken by *industrial* vessels (by definition all foreign fishing in the waters of another country is deemed to be industrial in nature) in the waters (or EEZ, or EEZ-equivalent waters) of another coastal state. As the high seas legally belong to no one (or to everybody, which is here equivalent), there can be no 'foreign' catches in the high sea. Prior to UNCLOS, and the declaration of EEZs by maritime countries, foreign catches were illegal only if conducted without permission within the *territorial* waters of such countries (generally, but not always 12 nm). After declarations of EEZs, foreign catches are considered illegal if conducted within the (usually 200 nm) EEZ and without access agreements, license or other permission (e.g., charter agreement) with the coastal state.

Such agreements can be tacit and based on historic rights, or more commonly explicit and involving compensatory payment for the coastal state. The *Sea Around Us* has created a database of such access and agreements, which is used to allocate the catches of distant-water fleets to the waters where they were taken (see Lam *et al.*, this volume, p. 39). Suffice here to say, therefore, that most catch reconstructions, in addition to identifying the catch of domestic fleets inside domestic waters, at least mention the foreign countries fishing in the waters of the country they cover (information we use in our access database), while other reconstructions explicitly quantify these catches (particularly in West Africa, see Belhabib *et al.,* 2012).

This information is then combined and harmonized with a) the catches deemed to have been taken outside a country's EEZ, as derived in Step 1 above and in Lam *et al.* (this volume, p. 39);

and b) the catches of countries reported by FAO as fishing outside the FAO areas in which they are located (e.g., Spain in FAO Area 27 reporting catches from Area 34), which always identifies this catch as distant-water catch, and thus allows estimation of the catch by foreign fisheries in a given area and even EEZ. Ultimately, the total catch thus extracted from a given area (i.e., a chunk of EEZ or EEZ-equivalent waters, or high seas waters within a given FAO area) is then computed as the sum of three data layers: (1) the reconstructed domestic catches (what the *Sea Around Us* calls “Layer 1” data); (2) the reconstructed catch by foreign fleets (“Layer 2” data); and (3) the tuna and other large pelagic fishes caught in the high sea and in EEZs, and here treated separately from all other catches, as “Layer 3” data (see Le Manach *et al.*, this volume, p. 25). Details of the harmonization and spatial allocation of these three data layers are presented in Lam *et al.* (this volume, p. 39).

Catch composition

The taxonomy of catches is what allows catches to be mapped in an ecosystem setting, as different taxa have different distribution ranges and habitat preferences (Close *et al.,* 2006; Palomares *et al.*, this volume, see p. 33). Also, temporal changes in the relative contribution of different taxa in the catch data also indicate changes in fishing operations and/or in dominance patterns in exploited ecosystems. Thus, various ecosystem state indicators can be derived from catch composition data, e.g., the ‘mean temperature of the catch’ which tracks global warming (Cheung *et al.,* 2013), the ‘stock-status plots’ which can provide a first-order assessment of the status of stocks (Kleisner *et al.,* 2013), and the marine trophic index which reveals instances of “fishing down marine food webs” (Pauly *et al.,* 1998; Pauly and Watson, 2005; Kleisner *et al.*, 2014).

Most statistical systems in the word manage to present at least some of their catch in taxonomically disaggregated form (i.e., by species), but many report a large fraction of their catch as over-aggregated, uninformative categories such as ‘other fish’ or ‘miscellaneous marine fishes’ (or ‘marine fishes nei’ [not elsewhere included] in FAO parlance). Interestingly, many official national datasets have better taxonomic resolution than the data reported to FAO by national authorities. It is highly likely that this is largely the result of the design of the data request form that FAO distributes to countries each year, which does not actively encourage (nor even suggest) that more detailed national taxonomic resolution data should be provided whenever possible. We have attempted to reduce the contribution of such over-aggregated groups to less than 15-20% of the total catch of a reconstruction, by using the approach outlined in Pauly (this volume, p. 9), suitably modified. The species and higher taxa in the catch of a given country or territory can thus belong to either one of three groups:

1. Species or higher taxa that were already included in the reported baseline data;
2. Species or higher taxa into which over-aggregated catches have been subdivided using three or more sets of catch composition data, such that the changing catch composition data reflect some observed changes of fishing operations and/or changes in the underlying ecosystem; or

3. Species or higher taxa into which over-aggregated catches have been subdivided using only one or two sets of catch composition data, and which therefore cannot be expected to reflect changes in on-the-ground catch compositions due to changes in fishing operations and/or changes in the underlying ecosystem. This score is also applied in cases where no local/national information on the taxonomic composition was available, and thus a taxonomic resolution from neighboring countries was applied.

We have labelled every taxon in the catch time-series of every country with (1), (2) or (3) such that (3) and perhaps also (2) are NOT used to compute catch-based indicators such as outlined above (they would falsely suggest an absence of change) – although we fear that this will still occur.

In summary, the approach we developed and utilized for undertaking the catch reconstructions for every maritime country/territory in the world consists of a structured system for utilizing all available data sources, and applying a conservative, but comprehensive integration approach. With the addition of the recently developed estimation approach for uncertainty around our reconstructed estimates (Step 7), the approach presented here can provide a more nuanced view of fisheries catches. Verifying and integrating these data into the global *Sea Around Us* database of fisheries catches, followed by spatial allocation of these catches in an ecosystem setting within given political constraints (i.e., EEZ access permissions) constitutes the next step in utilizing global reconstructed catch data. This process is described in Lam *et al.* (this volume, p. 39).

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Global catches of large pelagic fishes, with emphasis on tuna in the high seas[3]

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For centuries, tuna were targeted by small, localized fisheries using traps and other artisanal methods. Commercial tuna fishing by longline and pole-and-line began around the Pacific Islands in the 1910s and 1920s. However, it was not until after World War II that industrial effort began to intensify (Gillett, 2007). Initially, smaller species (e.g., skipjack tuna *Katsuwonus pelamis* and albacore tuna, *Thunnus alalunga*) were sought for canning purposes and dried export by foreign (but locally-based) fleets from the United States and Japan. However, improvements in fishing vessel technology and shipping methods - as well as the development of flash freezing capabilities - precipitated a rapid expansion in the industry, not only in terms of species targeted, but also with regard to the gears employed and the regions fished (Gillett, 2007).

The total global catch of tuna in 1950 was approximately 500,000 t (Miyake, 2005). By the late 1950s, Japanese and American longline fleets (targeting yellowfin tuna, *T. albacares* and albacore tuna for canning) had expanded from the Pacific into the Atlantic and Indian Oceans (Miyake, 2005; Evano and Bourjea, 2012). Shortly thereafter, locally-based pole-and-line vessels from Europe also began exploiting the waters off the west coast of Africa (Fonteneau *et al.,* 1993; Sahastume, 2002). Over the next decade, the Japanese longline fleet expanded its range globally, while both Korea and Taiwan began tuna fishing with longliners throughout the Pacific (Gillet, 2007). Until the 1970s, processing had been limited to canning and drying practices. However, the invention of flash freezing technology allowed fish to be transported across continents without spoiling, and Pacific bluefin tuna (*T. orientalis*) and bigeye tuna (*T. obesus*) quickly came into high demand for the sashimi market (Miyake, 2005).

Around the world, use of purse-seines quickly surpassed that of smaller pole-and-line operations, and a substantial shift in gear-preference occurred during the 1970s. Ultimately, by the early 1980s, the European purse-seine fleet had expanded into the Indian Ocean (Stequert and Marsac, 1989; Fonteneau, 1996; Bayliff *et al.,* 2005; Marsac *et al.,* 2014), while purse-seine fleets in the Western Pacific had simultaneously expanded outward from the Pacific Islands and South American countries began fishing in the Eastern Pacific (Gillet, 2007). The following decade saw an even greater increase in purse-seine effort and the use of drifting fish aggregating devices (d-FADs) by these fleets was occurring in all of the oceans by the end of the 1990s (Davies *et al.,* 2014). In tandem with increasing management measures, primarily with regard to the establishment of Regional Fisheries Management Organizations (RFMOs) (United Nations, 1995), the 1990s also saw an increased prevalence of illegal fishing (Gillet, 2007). It

[3] Abbreviated from: Le Manach, F, Cisneros-Montemayor, AM, Lindop, A, Padilla, A, Zeller, D, Schiller, L and Pauly, D. 2015. Global catches of large pelagic fishes, with emphasis on the High Seas, p. XX *In:* D. Pauly and D. Zeller (eds.) *Global Atlas of Marine Fisheries: Ecosystem Impacts and Analysis*. Island Press, Washington, D.C. [*in press*]

was during this time, that the use of Flags of Convenience (FOC) intensified, and many smaller coastal countries began chartering foreign longliners and purse-seiners (Gillet, 2007).

Despite tuna fisheries being among the most valuable in the world (FAO, 2012), as well as the considerable interest by civil society in the management of large pelagics, there are, to date, no global and comprehensive datasets presenting historical industrial catches of these species.

Methods

The aim of this chapter is to summarize the methods used to produce the first comprehensive[4], spatially harmonized global dataset of large pelagic fisheries catches. To produce this dataset, we assembled the various tuna datasets that already exist (Table 1), and harmonized them following a rule-based approach. Overall, the input data use can be separated into two components: nominal landings data (comprehensive data on catch tonnage by country, but with poor spatial information) and spatial catch data (good information on spatial location of catches by gear type, but not comprehensive in terms of taxa and fishing countries).

For each ocean, the nominal landings data were spatialized according to reported proportions in the spatial tuna cell data. Note that the initial spatial assignment for tuna data presented here differs from the standardized ½ x ½ degree resolution serving as final output of the *Sea Around Us*. Instead, spatial tuna input data are available in a variety of larger grid cells (i.e., 'tuna cells'), ranging from 1 to 20 degree cells (Table 1). Ultimately, after utilizing the spatial tuna grid cell data to spatially harmonize nominal landings data, the *Sea Around Us* will further spatialize the comprehensive tuna cell data generated here to the *Sea Around Us* ½ degree grid system (see Lam *et al.*, this volume, p. 39). For example, if France reported 100 t of yellowfin tuna in 1983 using longlines in the nominal landings dataset, but there were 85 t of yellowfin tuna reported for 1983 by French longlines in four separate tuna cells (potentially of varying spatial size), the nominal landings of 100 t for France were assigned to these four spatial tuna cells according to their reported proportion of total catch in the spatial tuna cell dataset.

This matching of the nominal landings and spatial tuna cell records was done over a series of successive refinements, with the first being the best-case scenario, in which there were matching records for year, country, gear and species. The last refinement was the worst-case scenario, in which there were no matching records except for the year of catch. For example, if country X reported 100 t of yellowfin tuna caught in 1983 using longlines, but there were no spatial tuna cell records for any country catching yellowfin tuna in 1983 in that ocean basin using any gear, the nominal 100 t for country X were split into spatial tuna cells according to the reported proportion of total catch of any species and gear in 1983. After each successive refinement, the

[4] The Food and Agriculture Organization of the United Nations (FAO) has published a global atlas, but it only includes the catch of 12 species of tuna and billfishes (i.e., albacore, Atlantic bluefin tuna, Atlantic white marlin, bigeye tuna, black marlin, blue marlin, Pacific bluefin tuna, skipjack tuna, southern bluefin tuna, striped marlin, swordfish, and yellowfin tuna). This atlas is available at: www.fao.org/figis/geoserver/tunaatlas. For reasons of confidentially, this dataset entirely lacks longline data for the eastern Pacific area after 1962, managed by the IATTC, although some data for the earlier time-period have already been published in highly aggregated form (Fonteneau, 1997). A recent resolution on confidentiality rules may, however, mean that these spatialized data will soon become publicly available (IATTC, 2013).

matched and non-matched records were stored separately, so that at each new refinement, only the previous step's non-matched records were used. The matched database was amended at the end of each refinement step. The final end result was a spatially harmonized tuna cell catch database containing all matched and spatialized catch records, which sum to the original nominal landings data.

Table 1. Summary of the various data sources used in the development of the spatially harmonized tuna cell input database.

Ocean		**Atlantic**	**Indian**	**Pacific**		**Southern**[a]
				Eastern	**Western**	
RFMO		**ICCAT**	**IOTC**	**IATTC**	**WCPFC**	**CCSBT**
Sources	Nominal data	website	website	Website	website	Provided by CCSBT statistical officer
	Spatialized data	website	website	www.fao.org/figis/geoserver/tunaatlas	website	website
Resolution		1°x1° 5°x5° 5°x10° 10°x10° 10°x20° 20°x20°	1°x1° 5°x5° 10°x10° 10°x20° 20°x20°	5°x5° [b]	5°x5°	5°x5°
Number of	Countries	114	57	28	41	11
	Gears	48	35	11	9	8
	Species	142	45	19	9	1

[a] This RFMO covers all three oceans, but only deals with southern bluefin tuna. Note that the other RFMOs also sometimes report this species (which we account for to avoid double-counting).

[b] A number of these cells were straddling the Pacific and Atlantic Oceans. Their total catch was split into these two ocean basins, proportionately to the surface of the cell included in each ocean. This step was then corrected for biological distributions (based on www.fishbase.org): catches of both Atlantic bluefin tuna and Atlantic white marlin that were obviously wrongly allocated to the Pacific Ocean were re-allocated back to the Atlantic Ocean.

Thereafter, a thorough review of the literature (peer-reviewed and grey) was performed for each ocean in order to collect estimates of discards. Due to the limited amount of specific discard data from approximately 20 sources, it was decided that discard percentages should be averaged across the entire time-period and applied to the region of origin of the fleet (e.g., East Asia or Western Europe), rather than the individual country of origin of the fleet. Similarly to the spatial harmonization refinement steps, successive refinements were done in order to add discards to all reported catch.

Once discards were added to the spatially harmonized tuna cell catch database, the total catch was spatialized to the ½ x ½ degree cells used by the *Sea Around Us*, and all artisanal catches (i.e., any gear other than longlines, purse-seines, and pole-and-lines) were re-assigned to the EEZ waters of each fleets flag-origin within each FAO area, because artisanal fleets are defined as operating within the IFA areas inside each country's EEZ only (see Zeller and Pauly, this volume, p. 15). Below are summary descriptions of the derived tuna data by major ocean basins.

Pacific Ocean

The main target species of the Pacific tuna fisheries are skipjack, yellowfin, bigeye, and albacore tuna, but two species - yellowfin and skipjack - have contributed over 79% of the catch by weight from 1950 to 2010 (Figure 1). However, at the individual level, the most valuable species caught in the Pacific are bigeye and Pacific bluefin (Majkowski, 2007).

Currently, the majority of the catch is taken by surface gears (e.g., purse seine accounts for 60% of Pacific tuna catch, Figure 1c) targeting yellowfin and skipjack for canning (Hall and Roman, 2013; Sumaila *et al.,* 2014). Historically, distant-water fleets from Japan, Korea, Taiwan and the USA were the main purse-seining operations in the Pacific, with fleets from China, Ecuador, El Salvador, New Zealand and Spain becoming more prevalent since the 2000s (Figure 1b). Since the late 1980s, Pacific Island-flagged purse-seine fleets have steadily increased in number, with 78 locally-flagged purse-seine vessels based in the western Pacific in 2010 (Williams and Terawasi 2011). However, a large proportion of this fleet is Pacific flagged in name only, with majority beneficial ownership residing in the major distant-water fleet countries.

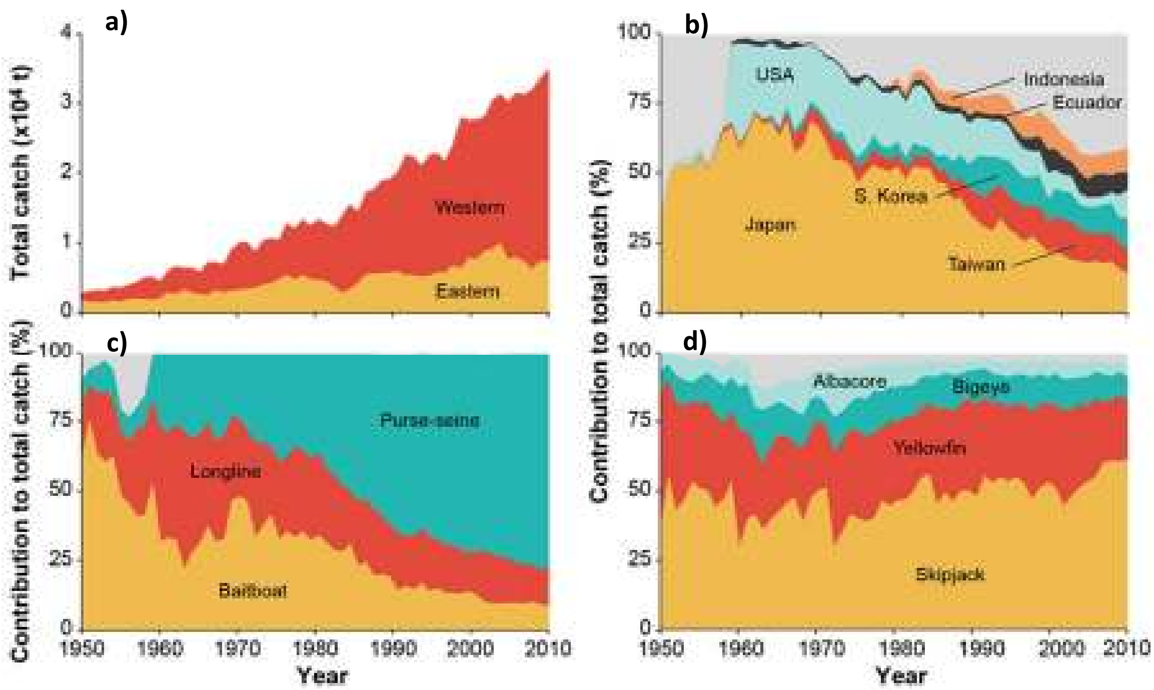

Figure 1. Industrial catch of large pelagic species in the Pacific Ocean, 1950-2010, showing a) the total annual reported catch by area; b) the percentage catch by country; c) the percentage catch by gear; and d) the percentage catch by target taxon. Note that these data are reported landings only; discards are not yet included.

Longlines are the second most common gear in the Pacific, contributing around 20% of the catch (Figure 1c). Longline fleets target primarily mature bigeye and yellowfin for the sashimi market, as well as some albacore and swordfish (WCPFC, 2011)

The use of pole-and-line (also called 'baitboats', Figure 1c) has decreased significantly over the last three decades (from 60% of the total catch in the early 1950s to around 10% in the 2000s). Nonetheless, this type of surface fishing targeting mainly skipjack (80% of catch) remains a seasonal venture for Australia, Fiji, Hawaii, as well as Japan, and a year-round fishery for domestic vessels from Indonesia, the Solomon Islands, and French Polynesia (Langley *et al.,* 2010; WPRFMC, 2013).

Atlantic Ocean

Industrial catches of large pelagics in the Atlantic Ocean steadily increased from very low levels in 1950 to a high of almost 600,000 t·year^{-1} in the mid-1990s (Figure 2a). They subsequently declined to around 400,000 t·year^{-1} by the mid-2000s.

Longline catches became prevalent in the 1960s with the arrival of the Japanese fleet in the region, but their contribution towards the total catch decreased over the years (Figure 2b,c). In the early 1980s, though, their contribution started to increase again to reach nearly 50% of the total catch by the mid-2000s, essentially due to the migration of European purse-seiners to the Indian Ocean (Stequert and Marsac, 1989; Fonteneau, 1996; Bayliff *et al.,* 2005; Marsac *et al.,* 2014), as well as the overall decline of the EU fleet and number of agreements (Anon., 2005).

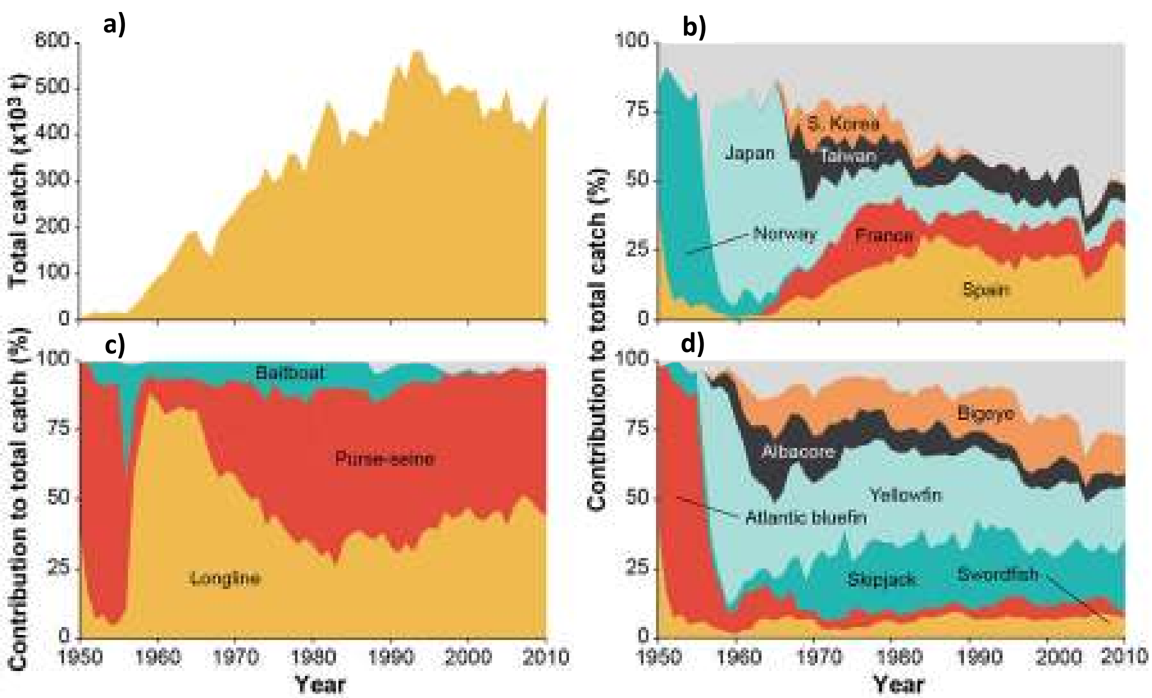

Figure 2. Industrial catch of large pelagic species in the Atlantic Ocean, 1950-2010, showing a) the total annual reported catch; b) the percentage catch by country; c) the percentage catch by gear; and d) the percentage catch by target taxon. Note that these data are reported landings only; discards are not yet included.

Purse-seiners (targeting skipjack and yellowfin tunas, Figure 2c,d) are the second major gear in terms of catch. Apart from a high contribution in the early 1950s (mostly from Norway, Figure 2b), the purse-seiner fleet only truly expanded in the 1960s and 1970s with the development of the French and Spanish fleet off the west coast of Africa (Figure 2b), following the decreasing catches of albacore tuna in the Bay of Biscay (Fonteneau *et al.,* 1993; Sahastume, 2002). The fleet of purse-seiners in the Atlantic Ocean has been slowly increasing again since the late 2000s, with some vessels coming back from the Indian Ocean to escape Somali piracy (see below).

A larger range of taxa are targeted in the Atlantic compared to the Pacific, with yellowfin dominating, followed by skipjack and bigeye tuna, but also including swordfish and, especially in the earlier years, Atlantic bluefin (Figure 2d).

Indian Ocean

Industrial fisheries only started in 1952 with the arrival of Japanese longliners in the Indian Ocean (Figure 3a,b). By the mid-1950s catches reached 100,000 t·year^{-1}. Until the arrival of the European purse-seiners in the early 1980s (Figure 3b,c), catches slightly fluctuated, but always remained below 200,000 t·year^{-1}. Catches increased to 600,000 t·year^{-1} by the mid-1990s, and then again to over 900,000 t·year^{-1} by the mid-2000s (mostly due to the expansion of the Iranian gillnet fleet, Figure 3b,c). Since then, industrial catches have steadily declined, arguably due to the negative effects of Somali piracy in the region (Chassot *et al.,* 2010; Martín, 2011; but see Waldo, 2009; Diaz and Dubner, 2010; Weldemichael, 2012 for a counter argument on the topic of 'piracy').

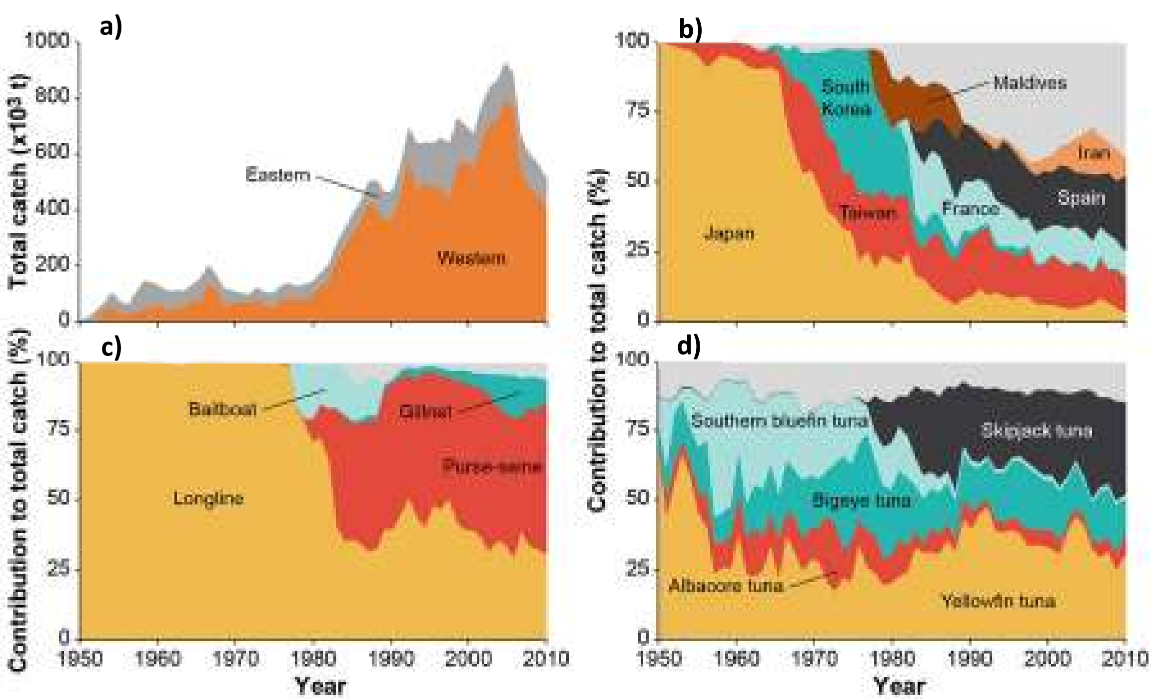

Figure 3. Industrial catch of large pelagic species in the Indian Ocean, 1950-2010, showing a) the total annual reported catch by area; b) the percentage catch by country; c) the percentage catch by gear; and d) the percentage catch by target taxon. Note that these data are reported landings only; discards are not yet included.

With the expansion of the Japanese fleet in the 1950s through the 1970s, followed by the arrival of European purse-seiners, the contribution of the western Indian Ocean to total Indian Ocean catches has consistently increased over time, from 0% in 1950, to a stable level of around 80% after 1990 (Figure 2a).

Yellowfin tuna dominate catches in the Indian Ocean, although southern bluefin and skipjack tuna contribute substantially in early and recent years, respectively (Figure 3d).

Conclusion

Overall, the global reconstruction of tuna and other large pelagic catches as described here, and as used for spatial allocation to the ½ x ½ degree grid cell system of the *Sea Around Us* (see Lam *et al.*, this volume, p. 39) provides for the first global harmonization of all readily available data on catches of large pelagics and associated discards. The present effort should be viewed as a first step in ultimately creating a refined and improved global, harmonized data layer of large pelagic catches.

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The distribution of exploited marine biodiversity [5]

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Ecosystem-based fisheries management (EBFM, Pikitch *et al.*, 2004) must include a sense of place, where fisheries interact with the animals and plants of specific ecosystems. To be useful to researchers, managers and policy makers attempting to implement EBFM schemes, the *Sea Around Us* presents biodiversity and fisheries data in spatial form onto a grid of 180,000 half degree latitude and longitude cells which can be regrouped into larger entities, e.g., the Exclusive Economic Zones (EEZs) of maritime countries (see, e.g., p. 113 ff), or the system of currently 66 Large Marine Ecosystems (LME) initiated by NOAA (Sherman *et al.*, 2007), and now used by practitioners throughout the world.

However, not all the marine biodiversity of the world can be mapped in this manner; thus, while FishBase (www.fishbase.org) includes all marine fishes described so far (more than 15,000 spp.), so little is known about the distribution of the majority of these species that they cannot be mapped in their entirety. The situation is even worse for marine invertebrates, despite huge efforts (see www.sealifebase.org).

We define as 'commercial' all marine fish or invertebrate species that are either reported in the catch statistics of at least one of the member countries of the Food and Agriculture Organization of the United Nations (FAO) or that are listed as part of commercial and non-commercial catches (retained as well as discarded) in country-specific catch reconstructions (see Zeller and Pauly, this volume, p. 15; Le Manach *et al.*, this volume, p. 25). For most species occurring in the landings statistics of FAO, there were enough data in FishBase for at least tentatively mapping their distribution ranges. Similarly, most species of commercial invertebrates had enough information in SeaLifeBase (www.sealifebase.org) for their approximate distribution range to be mapped. We discuss below the procedure we use for taxa that lacked sufficient data for mapping their distribution, which included few taxa in the FAO statistics, and many from reconstructed catches, including discards.

In the following, we document, in very abbreviated form how such mapping is done. Thus, it presents the methods (updated and improved from Close *et al.*, 2006) by which all commercial species distribution ranges (totaling over 1,500 for the 1950-2010 time period) were constructed and/or updated, and consisting of a set of rigorously applied filters that will markedly improve the accuracy of the *Sea Around Us* maps and other products.

The 'filters' used here are listed in the order that they are applied. Prior to the 'filter' approach presented below, the identity and nomenclature of each species is verified using FishBase (www.fishbase.org) or SeaLifeBase (www.sealifebase.org), two authoritative online encyclopedia covering the fishes of the world, and marine non-fish animals, respectively, and

[5] Abbreviated from : Palomares, M.L.D., W.W.L. Cheung, V. Lam and D. Pauly. Distribution of biodiversity in the seas around us, with emphasis on exploited fish and invertebrate species, p. XX *In:* D. Pauly and D. Zeller (eds.) *Global Atlas of Marine Fisheries: Ecosystem Impacts and Analysis*. Island Press, Washington, D.C. [*in press*]

their scientific and English common names corrected if necessary. This information is then standardized throughout all *Sea Around Us* databases (see Lam *et al.*, this volume, p. 39). Following the creation of all species-level distributions as described here, taxon distributions for higher taxonomic grouping, such as genus, family etc. are generated by combining each taxon-level's contributing components, e.g., for the genus *Gadus*, all distributions of species within this genus are combined.

Filter 1: FAO areas

The FAO has divided the world's oceans into 19 areas for statistical reporting purposes (Figure 1 in Zeller and Pauly, this volume, p. 15). Information on the occurrence of commercial species within these areas is available primarily through (a) FAO publications and the FAO website (www.fao.org); and (b) FishBase and SeaLifeBase. Figures 1A and 2A illustrate the occurrence by FAO area of Florida pompano (*Trachinotus carolinus*) and silver hake (*Merluccius bilinearis*), representing pelagic and demersal species, respectively.

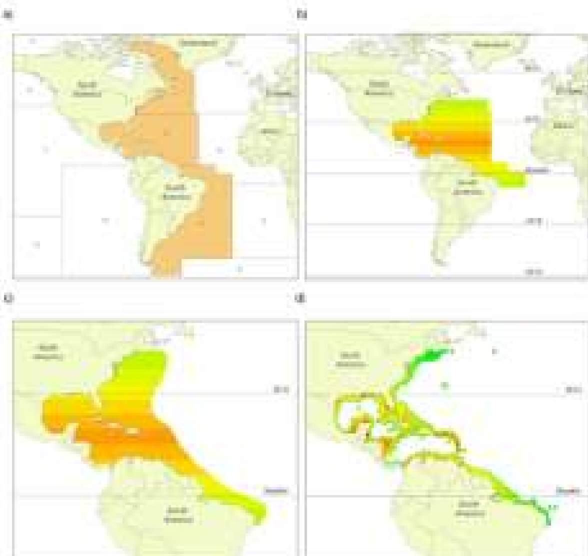

Figure 1. Partial results obtained following the application of the filters used for deriving a species distribution range map for the Florida pompano (*Trachinotus carolinus*): (**A**) illustrates the Florida pompano's presence in FAO areas 21, 31 and 41; (**B**) illustrates the result of overlaying the latitudinal range (43°N to 9°S) over the map in A; (**C**) shows the result of overlaying the (expert-reviewed) range-limiting polygon over B; and (**D**) illustrates the relative abundance of the Florida pompano resulting from the application of the depth range, habitat preference and equatorial submergence filters on the map in C.

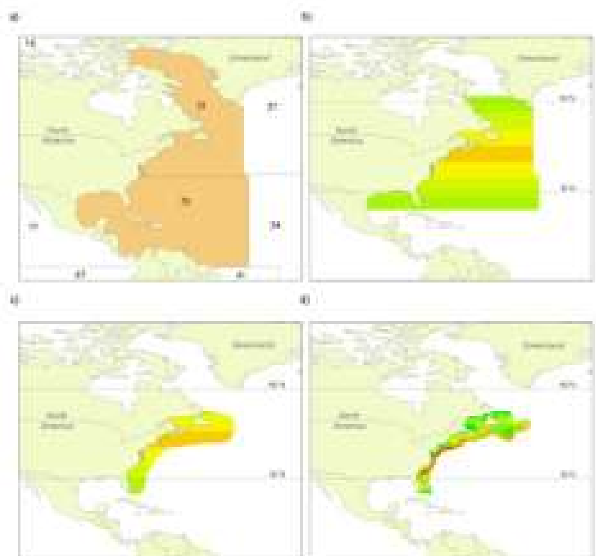

Figure 2. Partial results obtained following the application the filters used for deriving a species distribution range map for the silver hake (*Merluccius bilinearis*): (**A**) illustrates the silver hake's presence in FAO areas 21 and 31; (**B**) illustrates the result of applying the latitudinal range (55°N to 24°N); (**C**) shows the result of overlaying the (expert-reviewed) range-limiting polygon over B; and (**D**) illustrates the silver hake's relative abundance resulting from the application of the depth range, habitat preference and equatorial submergence filters on the map in C.

Filter 2: Latitudinal range

The second filter applied in this process is latitudinal ranges. The latitudinal range of a species is defined as the space between its northernmost and southernmost latitudes of occurrence. This range can be found in FishBase for most fishes and in SeaLifeBase for many invertebrates. For fishes and invertebrates for which this information was lacking, latitudes were inferred from the latitudinal range of the EEZs of countries where they are reported to occur as endemic or native species, and/or from occurrence records in the Ocean Biogeographic Information System (OBIS; www.iobis.org). Figures 1B and 2B illustrate the result of the FAO and latitudinal filters combined. Both the Florida pompano and the silver hake follow symmetrical triangular distributions whose base ranges between the northernmost and southernmost latitude, and whose (relative) midrange abundance is highest.

Filter 3: Range-limiting polygon

Range-limiting polygons help confine species in areas where they are known to occur, while preventing their occurrence in other areas where they could occur (because of environmental conditions), but do not. Distribution polygons for a vast number of species of commercial fish and invertebrates can be found in various publications, notably FAO's (species catalogues, species identification sheets, guides to the commercial species of various countries or regions), and in online resources, some of which were obtained from model predictions, e.g., Aquamaps (Kaschner *et al.*, 2007; see also www.aquamaps.org). Such polygons are mostly based on observed species occurrences, which may or may not be representative of the actual distribution range of the species.

All polygons, whether available from a publication or newly drawn, were digitized with ESRI's ArcGIS, and were later used for inferences on equatorial submergence (see Filter 6 below). Figures 1C and 2C illustrate the result of the combination of the first three filters, i.e., FAO, latitude and range-limiting polygons. These parameters and polygons will be revised periodically, as our knowledge of the species in question increases.

Filter 4: Depth range

Similar to the latitudinal range, the 'depth range', i.e., "[the] depth (in m) reported for juveniles and adults (but not larvae) from the most shallow to the deepest [waters]", is available from FishBase for most fish species and SeaLifeBase for many commercial invertebrates, along with their common depth, defined as the "[the] depth range (in m) where juveniles and adults are most often found. This range may be calculated as the depth range within which approximately 95 % of the biomass occurs" (Froese *et al.*, 2000). Given these data, and based on Alverson *et al.* (1964), Pauly and Chua (1988) and Zeller and Pauly (2001), among others, the abundance of a species within the water column is assumed to follow a scalene triangular distribution, where maximum abundance occurs in the top one-third of its depth range.

Filter 5: Habitat preference

Habitat preference is an important factor affecting the distribution of marine species. Thus, the aim of this filter is to enhance the prediction of the probability that a species occurs in an area, based on its association with different habitats. Two assumptions are made here:

1. That, other things being equal, the relative abundance of a species in a spatial ½ degree cell is determined by a fraction derived from the number of habitats that a species associates with in that same cell, and by how far the association effect will extend from that habitat; and
2. That the extent of this association is assumed to be a function of a species' maximum size (maximum length) and habitat 'versatility'. Thus, a large species that inhabits a wide range of habitats is more likely to occur far from the habitat(s) with which it is associated, while smaller species tend to have low habitat versatility (Kramer and Chapman 1999).

The procedure used by the *Sea Around Us* to consider these factors (which use 'fuzzy logic') is too complex to be discussed here (for details see Close *et al.*, 2006). Suffice to state here that while assumptions on the relationship between maximum length, habitat versatility and maximum distance from the habitat may render predicted distributions at a fine spatial scale uncertain, the routine, as implemented, provides an explicit and consistent way to incorporate habitat considerations into distribution ranges.

Filter 6: Equatorial submergence

Ekman (1967) gives the current definition of equatorial submergence: "*animals which in higher latitudes live in shallow water seek in more southern regions archibenthal or purely abyssal waters […]. This is a very common phenomenon and has been observed by several earlier investigators. We call it submergence after V. Haecker [1906-1908] who, in his studies on pelagic radiolarian, drew attention to it. In most cases, including those which interest us here, submergence increases towards the lower latitudes and therefore may be called equatorial submergence. Submergence is simply a consequence of the animal's reaction to temperature. Cold-water animals must seek colder, deeper water layers in regions with warm surface water if they are to inhabit such regions at all.*" Equatorial submergence, indeed, is caused by the same physiological constraints which also determine the 'normal' latitudinal range of species, as described above, and its shift due to global warming (Cheung and Pauly, this volume, p. 63), i.e., respiratory constraints fish and aquatic invertebrates experience at temperatures higher than that which they have evolved to prefer (Pauly, 1998, 2010).

Modifying the distribution ranges to account for equatorial submergence requires accounting for: (1) data scarcity; and (2) uneven distribution of environmental variables (temperature, light, food, etc.) with depth. FishBase and SeaLifeBase notwithstanding, there is little information on the depth distribution of most commercial species. However, in most cases, the following four data points are available for each species: the shallow or 'high' end of the depth range (D_{high}), its deep or 'low' end (D_{low}) of the depth range, the poleward limit of the latitudinal range (L_{high}), and its lower latitude limit (L_{low}). If it is assumed that equatorial submergence is to occur, then it is logical to also assume that D_{high} corresponds to L_{high}, and that D_{low} corresponds to L_{low}. Close *et al.* (2006) and Palomares *et al.* (in press), based on Pauly (2010) show how parabolic depth distributions can be generated which rely on these 4 data points to mimic the likely equatorial submergence of marine fishes and invertebrates.

Once these parabolic depth distributions are generated, they can be used to 'shave off' parts of the shallow-end of distributions at low latitudes, and similarly, shave off parts of the deep-end of distributions at high latitudes. Also, besides leading to narrower and more realistic distribution ranges, this leads to narrowing the temperature ranges inhabited by the species in question, which is important for the estimation of their preferred temperature, as used when modelling global warming effects on marine biodiversity and fisheries (Cheung and Pauly, this volume, p. 63).

The key outcome of the process described above consists of distribution ranges for over 1,500 taxa, such as for the Florida pompano and silver hake (Figures 1D and 2D, respectively). These distribution ranges serve as basis for the ½ degree spatial catch allocation done by the *Sea Around Us* (Lam *et al.*, this volume, p. 39).

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Reconstructed catches and their spatialization [6]

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While the fisheries catch reconstructions summarized in Pauly (this volume, p. 9), Zeller and Pauly (this volume, p. 15) and Le Manach *et al.* (this volume, p. 25) are all available online for checking (see publications at www.seaaroundus.org), the taxonomically disaggregated time series of catch data they contain, covering 61 years (1950-2010), 4 fishing sectors (industrial, artisanal, subsistence and recreational), 2 catch types (landed versus discarded catch) and 2 types of reporting status (reported versus unreported) for all maritime countries and territories of the world (n > 250), are too big to be presented as flat tables in papers, however detailed.

Thus, the catch data generated by the reconstruction project of the *Sea Around Us* are stored in a dedicated catch reconstruction database, which interacts with the other databases of the *Sea Around Us* to generate various products, foremost among them spatially allocated fisheries catches to the 180,000 half degree latitude and longitude cells covering the world ocean.

As global catch maps and related products that are meaningful in terms of ecology as well as policy are one of the major outputs of the *Sea Around Us*, and because the spatial allocation process is closely tied to the catch reconstruction database, this database and the spatial allocation process are described together in this section.

Catch reconstruction database

The catch reconstruction database comprises all of the catch reconstruction data for 1950 to 2010 by fishing country, taxon name, year of catch, catch amount, fishing sector, catch type, reporting status, input data source and spatial location of catch such as Exclusive Economic Zone (EEZ), FAO area, other area designation (if applicable). The database is further sub-divided into three different data layers, which include a layer with the catch taken by a fishing country in its own EEZ (called 'Layer 1'), the catch by each fishing country in other EEZs and/or the high seas ('Layer 2'), and the catch of all tuna and large pelagic species caught by each fishing country's industrial fleet ('Layer 3'). The basic structure of Layers 1 and 2 are identical, while Layer 3 differs slightly in structure due to the nature of the large pelagic input data sets (see Le Manach *et al.*, this volume, p. 25). The process of data integration into the catch reconstruction database includes a data verification process, which is the first integration step undertaken after receiving the original reconstruction dataset and associated reconstruction report after review of both by senior *Sea Around Us* staff. After completing the data verification process for each country dataset, each record is allocated to one of the layers based on the taxon, sector, and the area where the taxon was caught.

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[6] Abbreviated from: Lam, V.W.Y., A. Tavakolie, D. Knip, D. Zeller and D. Pauly. The Sea Around Us catch reconstruction database and its spatial expression, p. XX *In:* D. Pauly and D. Zeller (eds.) *Global Atlas of Marine Fisheries: Ecosystem Impacts and Analysis*. Island Press, Washington, D.C. [*in press*]

Data verification process

After initial, detailed review of each country/territory reconstruction dataset and associated technical report by senior *Sea Around Us* staff, the reconstruction dataset for each EEZ is further verified for accuracy and is formatted to fit the structure of the final database (see Figure 1 for overview). For example, the total *reported landings* presented in the reconstruction dataset of each country/territory (which represent the catches landed *and* deemed reported to national authorities from within the own EEZ of that country/territory) are compared with the reported data as present by FAO on behalf of the respective country/territory for each year. Any 'surplus' of FAO data are then considered to have been caught outside the EEZ of the given country/territory, and thus are treated as part of Layer 2 data. When any issue with the reconstructed catch data are identified, the issue is raised with the *Sea Around Us* catch reconstruction team and the original authors of the reconstruction for further checking and refining of the input data (Figure 1). Additional data verification steps include harmonization of scientific taxon names in the reconstruction data with the official, globally recognized and standardized taxon names via the global taxonomic authorities of FishBase (www.fishbase.org) and SeaLifeBase (www.sealifebase.org; see Palomares *et al*., this volume, p. 33). Fishing country names and EEZ names are also checked and standardized against the *Sea Around Us* spatial databases. The fishing country and EEZ names allow us to link the catch data to the foreign fishing access database, which contains the information on which fishing country can access the EEZs of another country (see '*foreign fishing access database*' section below).

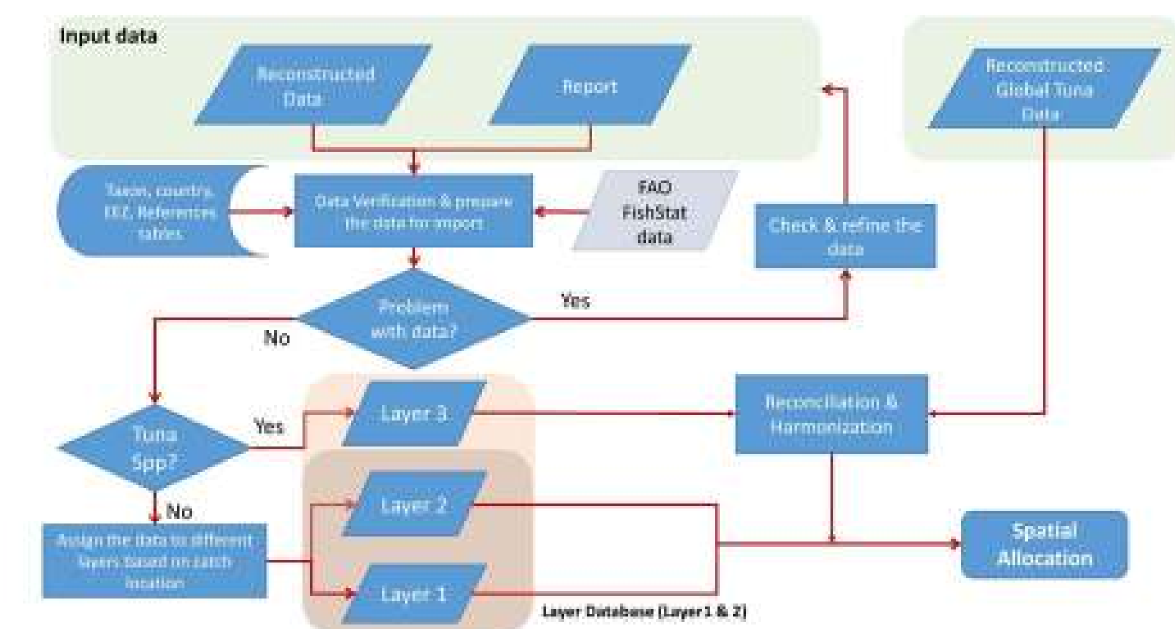

Figure 1. Data verification process for catch reconstruction data of the *Sea Around Us*. Details for the country/territory-specific '*Reconstructed Data*' and '*Report*' are provided in Zeller and Pauly (this volume, p. 15), while details for '*Reconstructed Global Tuna Data*' are described in Le Manach *et al.* (this volume, p. 25).

Based on the location where the each taxon was deemed to have been caught, each catch record is assigned to a different layer (see the section on '*Structure of the database*' below). This includes a cross-checking process between the various reconstruction input datasets. For example, if country A reported the landings of another country (Country B) in the EEZ of country A, this catch of country B is checked against the data in layer 2 of country B, as provided through country B reconstruction data. Emphasis is placed on avoiding double counting of catches.

Structure of the database

As outlined above, the catch reconstruction database contains the catch data assigned to one of three layers:

<u>Layer 1</u>

This layer retains all the catches taken by a country within that country's own EEZ. It contains industrial, artisanal, subsistence and recreational sector catches, sub-divided by catch type (retained and landed vs. discarded catch) and reporting status (reported vs. unreported). However, this layer excludes all industrial catches of large pelagics (see Table 1 for the list of reported taxa excluded here), which are moved to Layer 3 for later harmonization with the '*Reconstructed Global Tuan Data*' as derived by Le Manach *et al.* (this volume, p. 25 and Figure 1).

<u>Layer 2</u>

This layer contains data derived either directly from the reconstruction dataset and reconstruction technical report (i.e., catches listed as being taken outside the country's own EEZ), or indirectly by subtracting the reconstructed catch identified as reported landings in a country's own EEZ from the data reported by FAO on behalf of that country in the relevant (i.e., the 'home' FAO area of a given fishing country) FAO area (excluding the taxa listed in Table 1). Also, Layer 2 includes catches by a given fishing country in all non-home FAO areas (i.e., we refer to these catches as being taken by the given country's Distant-Water Fleets). This layer includes only industrial fishing sector catches, as we define all fleets or gears that can operate outside of a given country's own EEZ waters as industrial (i.e., large-scale) in nature. The few documented cases where locally so-called 'artisanal' fleets fish in neighboring EEZs, e.g., for Senegal (Belhabib *et al.*, 2014), we internally re-assign these catches to the industrial sector.

<u>Layer 3</u>

This layer initially included 29 specific large pelagic taxa (Table 1) whose reconstructed industrial catch data were moved to this layer to permit harmonization with the independently and globally reconstructed large pelagic dataset (Le Manach *et al.*, this volume, p. 25). The global tuna dataset combined taxonomically more diverse large pelagic catch datasets, and added bycatch and discards associated with the global industrial tuna and large pelagic fisheries.

Thus, the final harmonized large pelagic dataset (harmonized Layer 3, Figure 1) contains around 140 taxa and their associated catch.

Table 1: Tuna and large pelagic taxa (n = 29) initially moved from country reconstruction datasets to layer 3 for harmonization with the reconstructed global tuna data (Le Manach *et al.*, this volume, p. 25)..

Common name	TaxonName
Albacore	*Thunnus alalunga*
Atlantic bluefin tuna	*Thunnus thynnus*
Atlantic bonito	*Sarda sarda*
Atlantic sailfish	*Istiophorus albicans*
Atlantic white marlin	*Tetrapturus albidus*
Bigeye tuna	*Thunnus obesus*
Billfishes	Istiophoridae
Black marlin	*Makaira indica*
Black skipjack	*Euthynnus lineatus*
Blackfin tuna	*Thunnus atlanticus*
Blue marlin	*Makaira nigricans*
Bullet tuna	*Auxis rochei rochei*
Indo-Pacific blue marlin	*Makaira mazara*
Indo-Pacific sailfish	*Istiophorus platypterus*
Kawakawa	*Euthynnus affinis*
Little tunny	*Euthynnus alletteratus*
Longbill spearfish	*Tetrapturus pfluegeri*
Longtail tuna	*Thunnus tonggol*
Mediterranean spearfish	*Tetrapturus belone*
Pacific bluefin tuna	*Thunnus orientalis*
Shortbill spearfish	*Tetrapturus angustirostris*
Skipjack tuna	*Katsuwonus pelamis*
Slender tuna	*Allothunnus fallai*
Southern bluefin tuna	*Thunnus maccoyii*
Striped marlin	*Kajikia audax*
Swordfish	*Xiphias gladius*
Tuna	Thunnus
Wahoo	*Acanthocybium solandri*
Yellowfin tuna	*Thunnus albacares*

Foreign fishing access database

The foreign fishing access database, which initially built on a fishing agreements database by FAO (1999), contains observed foreign fishing records, and fishing agreements and treaties that were signed by fishing countries and the host countries in whose EEZs the foreign fleets were permitted to fish. In addition, the database also has start and end year of agreements and/or the observed access. The type of access is also specified, as ‘assumed unilateral’, ‘assumed reciprocal’, ‘unilateral’ or ‘reciprocal’. Also, the type of agreement is recorded in the database and the agreement can be classified into bilateral agreements such as partnership, multilateral agreements such as international conventions or agreements with regional fisheries organizations, private, licensing or exploratory agreements. Additional information contained in this database relates to the type of taxa likely targeted by foreign fleets (e.g., tuna vs. demersal taxa), as well as any available data on fees paid or quotas included in the agreements.

This database is used in conjunction with the catch reconstruction database and the taxon distribution database (see Palomares *et al.*, this volume, p. 33) in the spatial allocation process that assigns catches to the global *Sea Around Us* ½ x ½ degree cell system.

Spatial allocation procedure

The spatial allocation procedure - although it relies on the same global *Sea Around Us* grid of ½ x ½ degree cells that was used previously - is different from the approach used in the early phase of the *Sea Around Us* (until 2006) and described in Watson *et al.* (2004). In the earlier allocations, catches pertaining to large reporting areas (mainly FAO Areas, see Figure 1 in Zeller and Pauly, this volume, p. 15) were allocated directly to the half-degree cells, subject only to constraints provided by initially derived taxon distributions for the various taxa (Close *et al.*, 2006), and an earlier and more limited version of the fishing access database granting foreign fleets differential access to the EEZs of various countries (Watson *et al.*, 2004). Thereafter, the catch by a given fishing country in a given EEZ was obtained by summing the catch that had been allocated to the cells making up the EEZ of that country (Watson *et al.*, 2004). This process made the spatial allocation overly sensitive to the precise shape and cell-probabilities of the taxon distribution maps, and the precision of very problematic EEZ access rules for different countries. It regularly resulted in sudden and unrealistic shifts of allocated catches into and out of given EEZs purely due to the lifting or imposing of EEZ access constraints. Attempts to improve the allocation procedure with more internal rules made it unwieldy, fragile and extremely time consuming, and thus the *Sea Around Us* abandoned this approach in the mid 2000s.

The more structured allocation procedure that was devised as a replacement, and is described here (Figure 2), relies on catch data that are spatially pre-assigned (through in-depth catch reconstructions, see Zeller and Pauly, this volume, p. 15) to the EEZ or EEZ-equivalent waters (for years pre-dating the declaration of individual EEZs) of a given maritime country or territory, and, in the case of small-scale fisheries (i.e., the artisanal, subsistence and recreational sectors), to the Inshore Fishing Areas (Chuenpagdee *et al.*, 2006). This radically reduces the number of access rules and constraints that the allocation procedure must consider, avoids fish catches showing up in the EEZs of the wrong country, and dramatically reduces the processing times of the allocation procedure.

Watson *et al.* (2004) also used the spatial allocation process to simultaneously disaggregate (i.e., taxonomically improves) uninformative taxonomic groups such as 'miscellaneous marine fishes' (FAO parlance 'marine fishes nei') by relying on taxonomic information in neighboring ½ degree cells. This further added to the complexity of the allocation procedure and increased the difficulty of tracing actual country/taxon/catch entities through the process. Instead, our 'new and improved' allocation procedure disaggregates the input catch data as part of the country-by-country catch reconstruction process (Zeller and Pauly, this volume, p. 15), which therefore clearly documents the taxonomic changes in the associated technical report for each reconstruction. Within the catch reconstruction database, we keep track of the quality of the taxonomic disaggregation, such that indicators sensitive to the quality of the disaggregation are not computed from inappropriate data (see 'catch composition' in Zeller and Pauly, this volume, p. 15).

These pre-allocation data processing modifications allow focusing on the truly spatial elements of the allocation, which are handled through a series of conceptual algorithmic steps. The general algorithm of spatial allocation of catches is harmonized for Layers 1, 2, and 3 (Table 2), which means better software flow, while maintaining the conceptual differences in data layers. We will first start with an overview of the new allocation process (Figure 2), followed by how each data layer is conceptually unique and how it is handled, and end with an overview of the general algorithm of the spatial allocation.

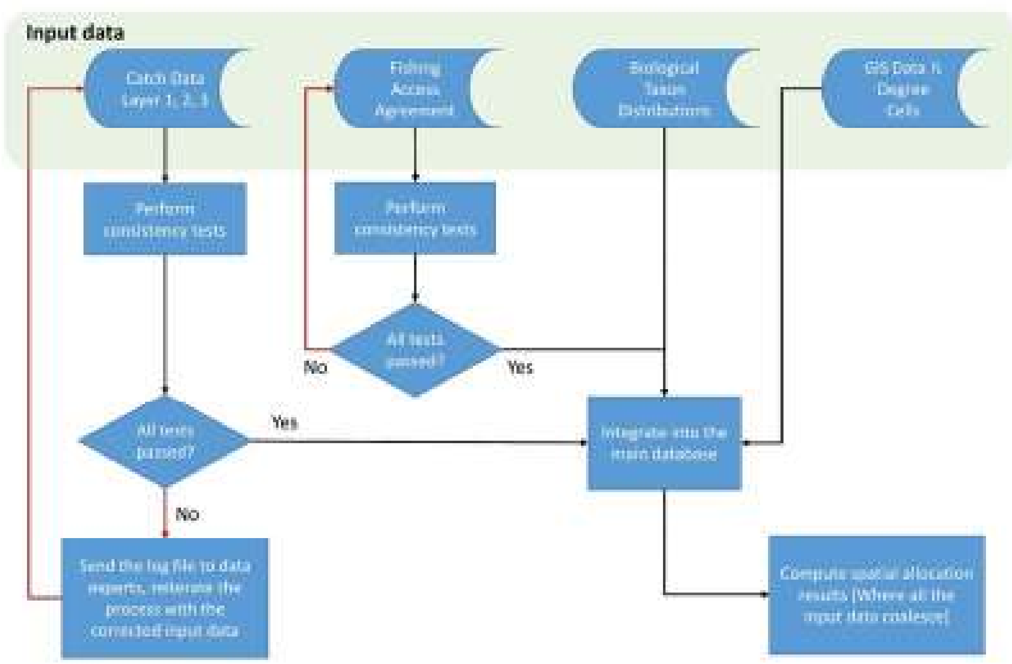

Figure 2. Spatial allocation procedure for catch reconstruction data of the *Sea Around Us*, resulting in the final ½ x ½ degree allocated cell data. Details for the Biological Taxon Distributions are provided in Palomares *et al.* (this volume, p. 33).

The spatial allocation of the catch is the process of computing the catch that can be allocated to each ½ degree cell based on the overlap of three main components: 1) the catch data, 2) the fishing access observations/agreements, and 3) the biological taxon distributions (Figure 2). The relationship/overlap amongst these components is facilitated by a series of Geographic Information System (GIS) layers which essentially bind them together.

Table 2: Parameters of the three spatial catch data input layers as used in the spatial allocation to ½ x ½ degree cells of the *Sea Around Us*.

Data layer	**1**	**2**	**3**
Taxa included	All except industrial large pelagics	All except large pelagics	Large pelagics (n =140+)
Spatial scope	Country's own EEZ	Other EEZs and high seas	Global tuna cells
Sectors	Industrial, artisanal, subsistence, recreational	Industrial	Industrial
Distributions	Biological	Biological	Biological
Fishing access	Automatically granted	Used	used
Granularity of data	EEZ, IFA[1]	EEZ, high seas, ICES, CCAMLAR, NAFO, FAO and other areas	Six types of tuna cells: 1x1, 5x5, 5x10, 10x10, 10x20, 20x20 (degrees)

[1] Inshore Fishing Area (IFA), defined as the area up to 50 km from shore or 200 m depth, whichever comes first (Chuenpagdee *et al.*, 2006). Note that IFAs occur only along inhabited coastlines.

How each data layer is conceptually unique and how it is handled

In Layer 1, the data come spatially organized by each fishing entity's EEZ(s). The allocation algorithm allocates the small-scale catches (i.e., artisanal, subsistence, and recreational) only to the cells associated with the Inshore Fishing Area (IFA, Chuenpagdee *et al.*, 2006) of that fishing entity's EEZ, while industrial catches can be allocated anywhere within that fishing entity's EEZ(s), as long as they remain compatible with the biological taxon distributions. Fishing access agreements are not applicable to this data layer, as each fishing entity (i.e., country) is always allowed to fish in its own EEZ waters. To represent the historical expansion of industrial fishing since the 1950s, from more easily accessible areas closer to shore to the full extent of each country's EEZ, we use a depth adjustment function for domestic industrial catches, as described in Watson and Morato (2013). This function takes into account that, as domestic industrial catches increase over time, an increasing fraction are being taken progressively further offshore (but within EEZ boundaries).

In Layer 2, the spatial granularity of the catch data can be by EEZ, high seas, or any other form of regional reporting areas, i.e., ICES, CCAMLAR, NAFO, or FAO statistical areas. However, in all cases it excludes the fishing entity (fishing country's) own EEZ waters (which are treated in Layer 1, Table 2). In Layer 2, the fishing access observations/agreements are used to compute the areas which allow fishing for a particular fishing entity, year, and taxon. Once this area is computed, it is superimposed on the biological taxon distributions to derive the final Layer 2 catch allocation.

In Layer 3, which only covers industrial large pelagics and their associated bycatch and discards, the input catch data are spatially organized by larger tuna cells which range from 1 x 1 to 20 x 20 degrees (Table 2, see also Le Manach *et al.*, this volume, p. 25). Similar to the region specific areas in Layer 2, these larger cells are intersected with all the EEZ boundaries to create a GIS layer which is suitable for use in the algorithm. Thereafter, the fishing access observations/agreements and taxon distributions are applied to calculate the final layer 3 catch allocation.

An overview of the algorithm

The spatial allocation algorithm has 4 main processes:

1. Validating and importing the fishing access observations/agreements database;
2. Validating and importing the catch reconstruction database;
3. Importing the biological taxon distributions; and
4. Computing the catch that can be allocated to each ½ degree cell for each catch data layer in an iterative process (allowing for verifications and corrections to any of the input parameters).

1. Validating and importing the fishing access observations/agreements database

The fishing access observations/agreements are first verified using several consistency and 'matching' tests (Figure 2) and upon passing they are imported into the main allocation database. This fishing access information is subsequently used in two different processes: (a) the verification process of the catch data (Layers 1, 2, and 3); and (b) the computing of the areas where a given fishing entity (i.e., country) is allow to fish for a specific year and taxon.

2. Validating and importing the catch reconstruction database

The validating and importing of the catch data is a more complex process than the validating and importing process for the fishing access database. This process involves about 22 different pre-allocation data tests (Figure 2). These tests are designed to make sure that the data are coherent from the stand point of database logic, and do not contain any accidental errors. These tests range from simple tests like "*is the TaxonKey valid?*" to more complex tests like "*validate if the given fishing entity has the required fishing access observations/agreements to fish in the given marine area*". Every single row of catch data is examined via these tests, and if it passes all tests the data row in question is added to the main allocation database. If it fails *any* of the tests it is returned to the relevant *Sea Around Us* data experts for review (Figure 2). This is an iterative process and is repeated until all the data rows pass all the pre-allocation tests.

The process of importing the catch reconstruction database includes an important sub-module which is tasked with harmonizing the marine areas. This module is crucial, as the catch data come in a variety of different spatial reporting areas that are not globally homogenous in GIS definitions (e.g., EEZ of Albania is one entity, while the EEZ of India and Brazil are subdivided into states/provinces; the north-east Atlantic uses ICES statistical areas, etc.). To harmonize these marine areas and make them accessible to the core allocation process, any given ½ degree marine area is split into its constituent countries EEZs and high sea components, then the fishing access observations/agreements are applied to this layer to determine which of these 'shards' of ½ degree cells are allowing access to a given fishing entity. Once this is determined, these collection of ‘shards’ are assigned to the given row of catch data, the result is a harmonized view of all the different marine areas. Presently, we have assigned over 12,000 marine areas into their constituent ‘shards’ of ½ degree cells, these marine areas range from EEZs and LMEs, to ICES, CCAMLAR, NAFO, and FAO statistical areas. The procedure allows future marine areas to be readily assigned.

3. Importing biological taxon distributions

Importing the biological taxon distributions is a fairly straightforward process. The over 1,600 individual taxon distributions (see Palomares *et al.*, this volume, p. 33) are generated as individual text files (csv) containing for each ½ x ½ degree cell the specific taxon’s probability of occurrence. These individual taxon distribution files are compiled into a database table for a more centralized and database centric use.

4. Computing/allocating the catch to ½ degree cells

Once the Steps 1, 2, and 3 are completed, we perform the computations which yields the final spatial ½ x ½ degree allocation results. The catch of a given data row, *TotalCatch*, of taxon *T* is distributed amongst eligible ½ degree cells, *Cells 1...n*, using the following weighted average formula:

$$Cell_{i_{AllocatedCatch}} = TotalCatch \times \frac{Cell_{i_{SurfaceArea}} \times Cell_{i_{RelativeAbundance\ of\ Taxon\ T}}}{\sum_1^n Cell_{i_{SurfaceArea}} \times Cell_{i_{RelativeAbundance\ of\ Taxon\ T}}}$$

Throughout the allocation process, catch reconstruction parameters in addition to *year* and *taxon*, such as *fishing sector*, *catch type*, *reporting status* etc. are preserved and carried over into the final ½ x ½ degree allocated database.

Final output

The final results of the intense and detailed database preparation and spatial allocation are time series of catches by ½ degree cells that are ecologically reliable (i.e., taxa are caught where they occur, and in relation to their relative abundance) and politically likely (e.g., by fishing country and within EEZ waters to which they have access to, see country summaries, p. 113 onwards).

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The economics of global marine fisheries [7]

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Fishing is an economic activity, with global connections and linkages, including to other sectors of the global economy. Thus, there is a need to study the economics of fisheries on a decidedly global basis, and not generate a pseudo-global coverage through 'case studies' of dubious representativeness. However, this perspective is relatively new to the study of fisheries, and initially, important data were not available on a global basis to support such work. Hence, the starting point for global fisheries economics work by the *Sea Around Us* (later in close collaboration with the Fisheries Economics Research Unit) was the creation, documentation and preliminary analyses of several global economics databases. Over the last 15 years, several global databases were created, updated and improved on:

1) Ex-vessel fish prices (Sumaila *et al.*, 2007; Swartz *et al.*, 2013);
2) Cost of fishing (Lam *et al.*, 2011);
3) Fisheries employment (Teh and Sumaila, 2011);
4) Fisheries subsidies (Sumaila and Pauly, 2006; Sumaila *et al.*, 2010, 2014);
5) Ecosystem-based marine recreational values (Cisneros-Montemayor and Sumaila, 2010); and
6) Economic multipliers (Dyck and Sumaila, 2010).

These studies were all conducted based on global marine fisheries catch data reported by FAO on behalf of its member countries. As these official catch data are often considerably lower than the reconstructed catches assembled by the *Sea Around Us* over the last decade, it can be expected that the economic impact of fisheries is underestimated in several of the studies presented below. Thus, they will be gradually updated in the coming years. In the meantime, the economic databases upon which these studies were based will be made available through the website of the *Sea Around Us*; they can be used to undertake large-scale bioeconomic analyses, as illustrated by Srinivasan *et al.* (2010, 2012) and Sumaila *et al.* (2012).

Ex-vessel fish price database

The global Ex-vessel fish price database described in Sumaila *et al.* (2007) was the first comprehensive database that presents average annual ex-vessel prices for all commercially

[7] Abbreviated from: Sumaila, U.R., A. Cisneros, A. Dyck, A. Khan, V. Lam, W. Swartz and L. Teh 2015. The economics of global marine fisheries, p. XX *In:* D. Pauly and D. Zeller (eds.) *Global Atlas of Marine Fisheries: Ecosystem Impacts and Analysis*. Island Press, Washington, D.C. [*in press*]

exploited marine fish species and higher taxa, and does so by country. It contained over 30,000 reported price items, covering the period from 1950 to the present, and supplemented missing prices with estimates based on prices from a different year, species (group) or fleet nationality. The initial database version was updated and expanded by Swartz *et al.* (2013), who also revised the method for estimating missing prices. Key advantages of the new estimation approach are that it allows a larger number of observed prices to be used in the estimation of missing prices, and better accounts for the relative price differences that exist between countries. This database is linked to the *Sea Around Us* catch database, and thus allows estimation of landed value for any spatial area in the world.

These prices suggest that the worldwide marine fisheries catches - as reported by FAO (i.e., not yet incorporating the reconstructed catch data) - had an ex-vessel value of US $100 billion in 2005, which is higher than the value of US $80-85 billion used in the studies documented further below.

Cost of fishing

The database of fishing costs presented in Lam *et al.* (2013) was the first, and so far is the only global cost of fishing database to be documented in the primary literature. It provides crucial economic information that is required for assessing the economics of global fisheries and should be useful for developing sustainable management scenarios. The database, which covers the 114 countries which landed approximately 98% of global marine fish catch in 2003, deals with two broad cost categories, i.e., variable and fixed costs. Variable costs include fuel cost, salaries for crew members, and repair and maintenance cost, while fixed costs include interest and depreciation cost of the invested capital (i.e., boat).

Costs varied between fishing gear types, with dredge and hook/line having the highest variable fishing costs, and North America had the highest unit variable cost among regions. The global average variable cost per tonne of catch in 2003 was estimated to be US $1,125 and the corresponding global variable fishing cost was US$ 86 billion. Given a global landed value of marine fisheries catch of about US$80-85 billion per year (see above), this implies that the global fishing fleet is running at an annual operating loss of about US$ 1-6 billion per year without subsidies (Lam *et al.*, 2011).

Fisheries employment

Marine fisheries contribute to the global economy in various ways, from the catching of fish through to the provision of support services for the fishing industry. A general lack of detailed data and uncertainty about the level of employment in marine fisheries can lead to under-estimation of fishing effort and hence overexploited fisheries, or result in inaccurate projections of economic and societal costs and benefits. To address this gap, a database of marine fisheries employment for 144 coastal countries was compiled. Gaps in employment data that emerged were filled using a Monte Carlo approach to estimate the number of direct and indirect fisheries jobs (Teh and Sumaila, 2011).

This study focused on the small-scale sector, and more precisely on artisanal and subsistence fisheries, as this globally provides the most job (for detailed definitions, see Zeller and Pauly, this volume, p. 15). Around 260 ± 6 million people were found to be employed in global marine fisheries, encompassing full-time and part-time jobs in catching, processing and marketing and otherwise handling of fish, corresponding to 203 ± 5 million full-time equivalent jobs. Of these, 22 ± 0.45 million would be small-scale fishers, a figure similar to the estimate previously published by Chuenpagdee *et al.* (2006).

The results of this study can be used to improve management decision making, and highlight the need to improve monitoring and reporting of the number of people employed in marine fisheries globally.

Fisheries subsidies

Building on the publication of Munro and Sumaila (2002), a global database of subsidies provided to marine fisheries was developed and documented in Sumaila and Pauly (2006), and updated in Sumaila *et al.* (2010, 2014). Therein, subsidies are grouped into three categories: 'beneficial' ('good'), 'capacity-enhancing' ('bad') and 'ambiguous' ('ugly'). The basis for this classification is the potential impact of given subsidy types on the sustainability of the fishery resource. 'Beneficial' subsidies enhance the conservation of fish stocks over time; this includes subsidies that fund fisheries management, and funds dispensed to establish and operate marine protected areas (Cullis-Suzuki and Pauly, 2008). In contrast, 'capacity-enhancing' subsidies such as, e.g., fuel subsidies, lead to overcapacity and overexploitation. 'Ambiguous' subsidies can lead to either the conservation or overfishing of a given fish stock, e.g., buyback subsidies, which if not properly designed, can lead to overcapacity (e.g., when a fisher can use the funds obtained from such schemes for the down payment on a new, more powerful vessel).

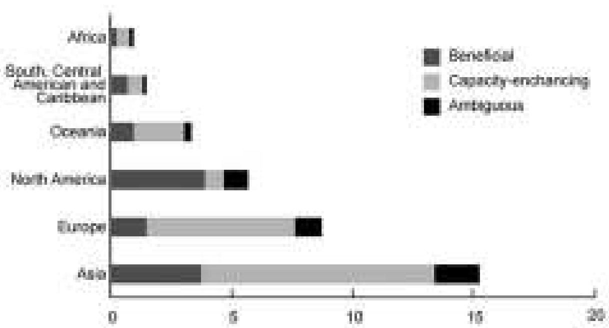

Figure 1. Government subsidies to marine fisheries by type and region. This shows that Asia is by far the greatest subsidizing region (43% of total), followed by Europe (25% of total) and North America (16% of total). For all regions, the amount of capacity-enhancing subsidies is higher than other categories, except in both North and South America, which have higher beneficial subsidies. Adapted from FAO (1992), Milazzo (1998), Sumaila and Pauly (2006), and Sumaila *et al.* (2010).

These estimates of global fisheries subsidies were presented to the World Trade Organization in Geneva, and shaped numerous debates. The most recent update of these subsidies estimates (Sumaila *et al.*, 2013), commissioned by and presented in October 2013 to the Fisheries Committee of the European Parliament in Brussels are summarized in Figure 1 by type and region. Figure 2 shows the similarity of the global estimates back to 2006 (once they are adjusted for inflation), and their differences to estimates published earlier by FAO (too high) and the World Bank (too low).

Ecosystem-based marine recreational values

Participation in ecosystem-based marine recreational activities (MRAs) has increased around the world, adding a new dimension to human uses of marine ecosystems, and is another good reason to create effective management measures. A first step in studying the effects of MRAs at a global scale is to estimate their socioeconomic benefits, which are captured here by three indicators: the amount of participation, employment and direct expenditure by users (Cisneros-Montemayor and Sumaila, 2010). A database of reported expenditure on MRAs was compiled for 144 coastal countries, and a meta-analysis performed to calculate the yearly global benefits of MRAs in terms of expenditure, participation and employment. It was estimated that over 120 million people a year participate in MRAs, generating 47 billion USD (in 2003 USD) in expenditures and supporting one million jobs. The results of this study have several implications for resource managers and for the tourism industry. Aside from offering the first estimation of the global socioeconomic benefits of MRAs, this work provides insights into the drivers of participation and possible ecological impacts of these activities. Our results could also help direct efforts to promote adequate implementation of MRAs.

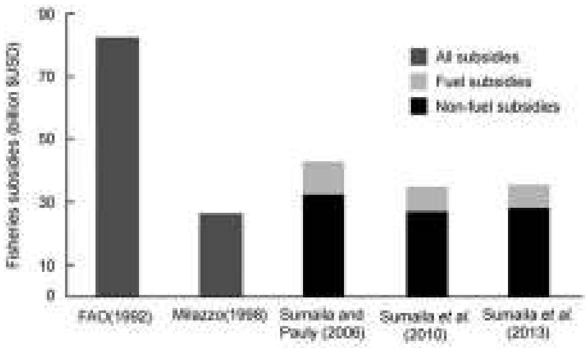

Figure 2. Successive global estimates of government subsidies to fisheries. Adapted from FAO (1992), Milazzo (1998), Sumaila and Pauly (2006), and Sumaila *et al.* (2010, 2013). Note that the numbers in this figure were all converted to real 2009 USD, in order to make them comparable.

Marine fish populations' contribution to the world economy

While the estimates of gross revenue from marine capture fisheries range from US $80 to 85 billion annually, there are a vast number of secondary economic activities - from boat building to running fish restaurants - that are supported by the world's marine fisheries. Yet these related activities are rarely considered when evaluating the economic impact of fisheries. A study applying an input-output methodology was conducted to estimate the total direct, indirect, and induced impact of marine capture fisheries on the world economy. Specifically, the goal was to estimate the total output in an economy that is dependent (at least partially) on the output of marine fisheries. Herein, Leontief's 'technological coefficients' at the catch levels reported by FAO for the early 2000s were used to estimate total output supported by marine fisheries throughout the economy (Dyck and Sumaila, 2010). While results suggest that there is a great deal of variation in fishing output multipliers between regions and countries, the output multipliers suggest, at the global level, that the direct and indirect impacts of the marine fisheries sector are about three times the value of the landings at first sales, i.e., between US $225 and $235 billion per year.

Conclusion

The databases described herein, and accessible via the *Sea Around Us*, should contribute to the gradual broadening of marine resource economics from the analysis of case studies, and of regional and/or national fisheries, to the study and understanding of global marine populations and their utilization. One important advantage of global-scale studies is that they may reveal patterns, problems and solutions which cannot be readily distinguished at smaller scales (Rosenthal and DiMatteo, 2001). More importantly, however, we live in an increasingly globalized world, where many challenges raised by our use of marine resource are global in nature. To deal with these challenges, we need global studies, and the contributions described here are important initial steps towards a truly global fisheries economics.

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Global high seas management [8]

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Fishing is no longer a coastal phenomenon (O'Leary *et al.*, 2012; Ban et al., 2014a). Over the last half century, advances in new fishing technology coupled with coastal stock declines have prompted fishers to expand beyond coastal waters, and out into the high seas (Swartz *et al.*, 2010). These previously difficult to access 'areas beyond national jurisdiction', i.e., beyond the 200 mile Exclusive Economic Zones (EEZs) of maritime countries, offered access to previously unexploited and extremely valuable fish stocks, especially of tuna, and global fish catch from the high seas thus increased tremendously (see Le Manach *et al.*, this volume, p. 25). However, limited regulations in these remote areas of ocean and inadequate management quickly led to severe stock declines (FAO, 2012).

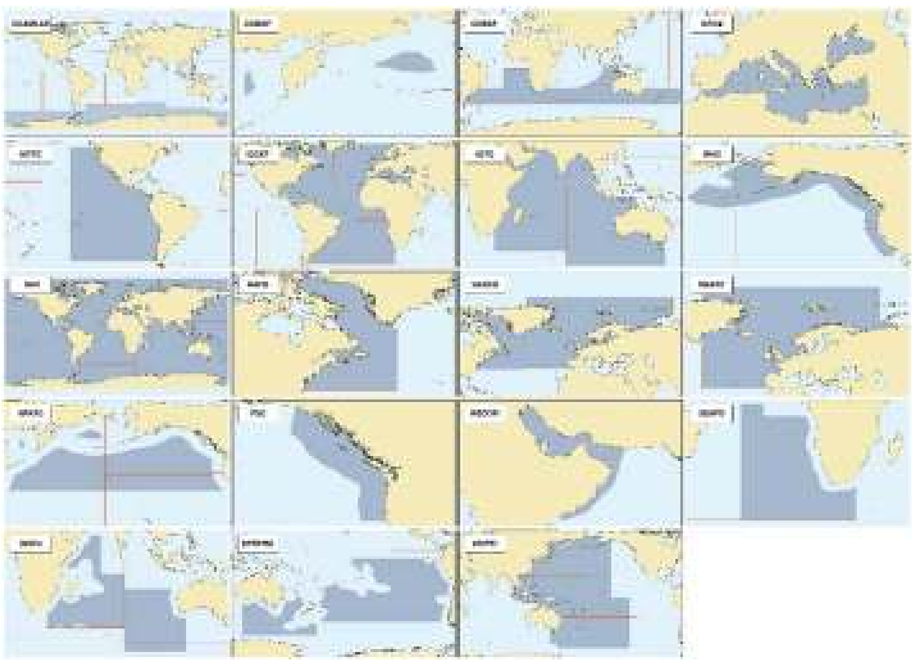

Figure 1. The current 19 RFMOs cover the entire World Ocean; note their boundaries frequently overlap (see Table 1 for full names and characteristics of each RFMO). Three RFMOs cover only the EEZ of their member states (i.e., exclude the High Seas): IPHC, PSC and RECOFI.

Regional Fisheries Management Organizations (RFMOs) are intergovernmental bodies tasked with managing fish stocks found mostly in the high sea areas of the world ocean (Figure 1). Established by and comprised of 'member countries', often maritime countries located in that part of the world ocean covered by the RFMO, but also including any nation with a "real interest" in the fishery - these members must manage, conserve, and ensure the long-term sustainability of the fisheries resources in their remit (UN, 1982, 1995). This has proved to be a difficult task, and RFMOs face many challenges, from structural difficulties (e.g., new member allocation; Munro, 2007) to internal problems (e.g., data

[8] Abbreviated from: Cullis-Suzuki, S. and D. Pauly. Global evaluation of High Seas fishery management, p. XX *In:* D. Pauly and D. Zeller (eds.) *Global Atlas of Marine Fisheries: Ecosystem Impacts and Analysis*. Island Press, Washington, D.C. [*in press*]

deficiencies; Colette *et al.*, 2011; O'Leary *et al.*, 2012), regional issues (including illegal fishing, corruption, and lack of enforcement; Sumaila *et al.*, 2007; Pintassiglo *et al.*, 2010) and broader problems associated with non-compliance with international treaties (Bjorndal and Munro, 2003). These issues are not new, and have been discussed for many years, but such complexities have inhibited RFMO progress.

In 2010, in response to declining high seas stock trends and the observation that "RFMO performance has not lived up to expectation" (Lodge *et al.*, 2007), a first global evaluation of the effectiveness of RFMOs was conducted (Cullis-Suzuli and Pauly, 2010a, 2010b). Here, the key results of this analysis are summarized and updated, based on feedback from RFMO representatives, input from colleagues, and, where available, current data from recent stock assessments.

There are currently 19 marine RFMOs with management capacity (Table 1). Over the last decade, international calls for increasing RFMO coverage have been met (FAO, 2012): today, the entire global ocean is covered by at least one RFMO (Figure 1).

2010 study: Failing the high seas

In 2010, a study was published which assessed the global effectiveness of the, at the time, 18 existing RFMOs (Table 1). This study, entitled 'Failing the High Seas: a global evaluation of regional fisheries management organizations' (Cullis-Suzuki and Pauly, 2010a) assessed the overall performance of RFMOs as determined by how well they achieved management and conservation objectives mandated by international treaties (UN, 1982, 1995). This was based on a two-tiered approach: assessing the effectiveness of RFMOs 'on paper' and 'on the ground'.

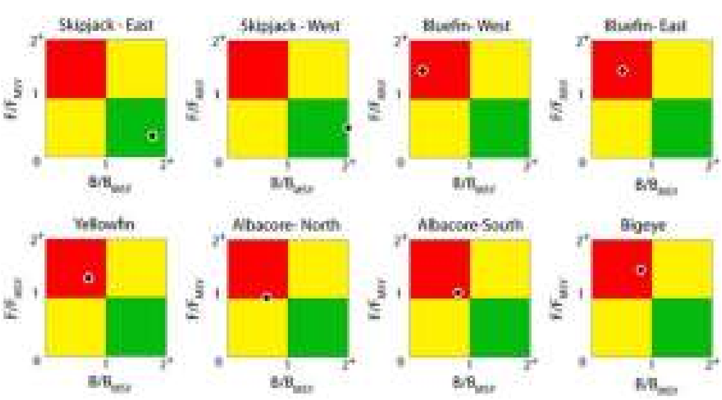

Figure 2. Phase plots of eight principal tuna species under the management of ICCAT (similar phase plots for the other RFMOs with stock assessment results may be found at www.seaaroundus.org).

To assess RFMO effectiveness 'on paper', each RFMO was scored against a set of 26 best-practices criteria developed from Lodge *et al.* (2007), where each criterion had 10 possible scores, ranging from 1-10 (see also Alder *et al.*, 2001). In addition to the 18 RFMOs, two 'outgroups' were also scored to test the criteria's discriminating ability: the World Wildlife Fund (an environmental NGO) and the U.S. National Marine Fisheries Service (a national fisheries management agency); a cluster analysis clearly identified the two non-RFMOs as outgroups, thus demonstrating that the criteria used in the study could distinguish between non-RFMOs and RFMOs (Cullis-Suzuki and Pauly, 2010b). Across RFMOs, results revealed an average score of 57%, with a range of 43% to 74%. Out of five overarching categories, the highest scoring was that of 'General information and organization', while the worst was 'Allocation' (Cullis-Suzuki and Pauly, 2010b).

To calculate RFMO effectiveness 'on the ground', we depended on stock assessments and scientific data to determine the state of stocks. By plotting relative fishing mortality and biomass data points, we obtained a score that reflected whether the stock was overfished, and/or depleted (Figure 2). Results showed that two-thirds of fish stocks on the high seas and under RFMO management were either overfished and/or depleted, matching estimates presented by FAO. The average score across RFMOs was 49%, ranging from 0% to 100% (Table 1). There was no correlation between scores 'on paper' and 'on the ground', suggesting the existence of a disconnect between RFMO intentions and actions.

Current updated evaluation

For this update, the focus is on recalculating RFMO effectiveness 'on the ground'. Setbacks in determining RFMO effectiveness 'on paper' centered mostly on data attributes, which without standardization, can be difficult to score (Kjartan Hoydal, NEAFC, pers. comm.). Also, publicly accessible information can be limited or complicated to locate, or RFMOs can outright fail to provide information (even if it exists), resulting in a low score. Finally, high compliance does not always correlate with healthy fisheries, as suggested above and also shown in Alder *et al.* (2001). Thus by focusing on a quantitative and internationally recognized description of stock status, we obtain a framework that is more easily standardized (Froese and Proelss, 2012).

To compute scores 'on the ground', here we evaluated forty-six fish stocks under current management across the fourteen different RFMOs with sufficient information for assessment (Table 1). Of the 48 stocks assessed in 2010, three were since removed from further assessment following comments from RFMO managers (see Cullis-Suzuki and Pauly, 2010b); we also substituted three stocks with different stocks of the same species in response to data constraints and availability, and added one new stock to the current study. Scores were calculated by plotting B/B_{MSY} by F/F_{MSY}, where B is the current stock biomass, F the current fishing mortality rate, and B_{MSY} and F_{MSY} generally accepted limits (for scoring details, see 'Q' scores in Cullis-Suzuki and Pauly, 2010a). Each plot had four quadrants: depending into which quadrant the data fell, the stock was given a score of 0 (red quadrant: overfished and depleted, i.e., 'threatened'), 1 (yellow quadrant: overfished or depleted, i.e., 'at risk'), or 3 (green quadrant: not overfished or depleted; i.e., 'stable').

Since the 2010 evaluation, ten stocks have changed score: five have gone up, and five went down, while a further two have moved from an 'overfished' to a 'depleted' state with no overall change to their score. The updated results reveal that currently, nearly three-quarters of stocks examined are in poor condition, with 20% being threatened (i.e., overfished and depleted) and 52% being at risk (i.e., overfished or depleted). However, there has been a slight improvement in overall average stock scores across RFMOs, from 48% in 2010 to 50% (Table 1).

Table 1. Average scores across RFMOs in 2010 and 2013 (note: five RFMOs- PSC, RECOFI, SEAFO, SIOFA and SPRFMO- lacked sufficient data to be assessed). For supplementary information including score calculations and stock-specific data, see www.seaaroundus.org.

			Avg. score (%)	
	Acronym	**Species assessed**	**2010**	**2013**
CCAMLR	Commission for the Conservation of Antarctic Marine Living Resources	Patagonian toothfish	100.0	100.0
CCBSP	Convention on the Conservation and Management of the Pollock Resources in the Central Bering Sea	Alaska Pollock	33.3	33.3
CCSBT	Commission for the Conservation of Southern Bluefin Tuna	Southern bluefin tuna	0.0	33.3
GFCM	General Fisheries Commission for the Mediterranean	Sardine, anchovy	33.3	33.3
IATTC	Inter-American Tropical Tuna Commission	Yellowfin, bigeye and skipjack tuna	33.3	77.8
ICCAT	International Commission for the Conservation of Atlantic Tunas	Bluefin tuna (West & East), yellowfin and skipjack tuna (West & East); bigeye and albacore tuna (North & South)	37.5	25.0
IOTC	Indian Ocean Tuna Commission	Yellowfin, albacore tuna and bigeye tuna	77.8	77.8
IPHC	International Pacific Halibut Commission	Pacific halibut	33.3	33.3
IWC	International Whaling Commission	Fin, blue, sperm, right, sei, Bryde's, humpback and minke whales (2 stocks)	33.3	33.3
NAFO	Northwest Atlantic Fisheries Organization	Redfish, cod (2 stocks), American plaice, Greenland halibut	41.7	20.0
NASCO	North Atlantic Salmon Conservation Organization	Atlantic salmon	33.3	33.3
NEAFC	North East Atlantic Fisheries Commission	Blue whiting, mackerel, golden redfish, herring	75.0	41.7
NPAFC	North Pacific Anadromous Fish Commission	Sockeye, chum and pink salmon	77.8	77.8
PSC	Pacific Salmon Commission	--		--
RECOFI*	Regional Commission for Fisheries	--	N/A*	--
SEAFO	South East Atlantic Fisheries Organization	--		--
SIOFA	South Indian Ocean Fisheries Agreement	--		--
SPRFMO	South Pacific Regional Fisheries Management Organization	--		--
WCPFC	Western and Central Pacific Fisheries Commission	Yellowfin, albacore, bigeye and skipjack tuna	66.7	83.3
Total		**46 stocks**	**48.3**	**50.2**

* The Regional Commission for Fisheries (RECOFI) was not assessed in 2010; while RECOFI entered into force in 2001, it still does not provide enough information in its reports to assess the current state of stocks in its remit. RECOFI covers all marine organisms in waters of its member states, i.e., Bahrain, Iraq, Iran, Kuwait, Oman, Qatar, Saudi Arabia, and the United Arab Emirates.

Recent developments

Even though some RFMOs are doing better than others, and despite some steps towards progress (de Bruyn *et al.*, 2013), it remains overwhelmingly clear that RFMOs are in need of improvement. This is emphasized here through three important international events that have transpired over the last few years and which reflect various aspects of the underwhelming performance of RFMOs.

The road ahead for RFMOs

The results from Cullis-Suzuki and Pauly (2010a) and the present evaluation suggest that RFMOs are not effective management and conservation bodies on the high seas. Further, they have not substantially improved over the last few years, as determined by the state of the stocks in their remit. This is further supported by the recent rejection by RFMOs of conservation-based recommendations from the international community (UN, 2012; Cressy, 2013). Additionally, one of the biggest impediments to conducting studies such as Cullis-Suzuki and Pauly (2010a) and here is the dependence on available stock assessments (not to mention, relevant reference points): these data are either lacking in RFMOs or not made publicly available, and seriously impede stock evaluation (Froese and Proelss, 2012; Powers and Medley, 2013).

High seas management appears to be in a state of uncertainty: recommended best practices have yet to be seriously implemented by RFMOs (Lodge *et al.*, 2007), and strengthened international commitments under United Nations treaties still await consideration (Druel *et al.*, 2012; UN, 2012). While many documents outline possible avenues for improvements in high seas fisheries management (Veitch *et al.*, 2012; Ban *et al.*, 2014a; Druel and Gjerde, 2014; Englender *et al.*, 2014), the high seas remain among the least understood and least protected ecosystems in the world (Ban *et al.*, 2014b).

In May 2010, shortly after the contribution by Cullis-Suzuki and Pauly (2010a) was accepted for publication, the authors were invited to present their findings at the United Nations headquarters in New York during the United Nations Fish Stocks Agreement Review Conference[9], and the first author attended (Cullis-Suzuki, 2010). The turnout for this panel - which was organized by staff of the Pew Charitable Trusts and included two other marine scientists and a lawyer - was unexpectedly large and comprised mainly of RFMO delegates, many of whom expressed strong reservations and criticisms about the presentation on the results of the RFMO evaluation. Indeed, not only did they overwhelmingly reject its results, but many disagreed with the underlying data, although they originated for the most part from the stock assessments the RFMOs themselves, conducted and made available on their websites. An hour and a half of denunciations during the post-talk Q&A led to the extraordinary consensus among the delegates that RFMOs could not be the source of these unfavourable data, and that any critique was unwarranted. Later on, as a follow up to this presentation and to the supporting publication, we received emails from RFMO managers with detailed criticisms of our research; as a result of one such commentary, we eliminated three stocks from our initial Q score assessment and decreased the total number of stocks assessed from 48 to 45; this did not alter the results of Cullis-Suzuki and Pauly (2010b).

This, and a similar, though less intensive experience by the second author at an event in early 2014 in Stockholm, Sweden, where the updated RFMO evaluation results were presented, exemplify what is perhaps the greatest setback to all RFMOs in achieving sustainably managed fisheries: RFMOs were created to allocate catch between competing fleets. RFMOs are fisheries-orientated bodies first, and conservation bodies second, if at all. RFMO delegates therefore represent fishing industry interests only (Gjerde *et al.*, 2013), and thus RFMO operations reflect their primary objective, which is to catch as much fish as possible.

[9] www.un.org/depts/los/convention_agreements/review_conf_fish_stocks.htm

This basic orientation lies at the heart of failed management on the high seas. Moving away from allocation-only based objectives will require recasting RFMOs as conservation bodies, which could in turn change the tide and begin the fundamental reform so urgently needed on the high seas (Gjerde *et al.*, 2013; Gilman *et al.*, 2014). Actually, we consider, along with White and Costello (2014), that the more equitable policy, in the long run, whould be to close the high seas to fishing entirely, and to let maritime countries throughout the world benefit from the resulting resource recovery in their Exclusive Economic Zones.

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Marine exploited species and climate change [10]

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Of the various ways in which humans impact marine ecosystems, climate change (CC) may be the most insidious and unrecognized. In fact, even if people 'believe' that it is occurring, most think climate change is going to affect us 'later', and thus there is no need for real urgency. However, as we will show here, climate change has in fact already begun to affect us in multiple ways, though mostly indirectly, including through its effect on the oceans and marine fisheries. This chapter is thus devoted to documenting some of the work through which scattered observations on the effects of climate change on marine organisms was generalized, and in the process, the first global maps of observed and predicted climate change impacts on marine biodiversity and fisheries were produced. Thus, the work conducted by the *Changing Climate Unit*, in collaboration with the *Sea Around Us*, complements work performed in the terrestrial realm.

Climate change affects ocean properties including water temperature, oxygen levels and acidity. According to the 5th Assessment Report (AR5) of the Intergovernmental Panel on Climate Change (IPCC), there is compelling evidence that the heat content and stratification of the ocean have been increasing in the 20th century, while sea-ice and pH have been decreasing, and that these trends can be expected to continue in the next century under the climate change scenarios considered by the IPCC (IPCC, 2014). Furthermore, available evidence indicates that climate change is expected to result in an expansion of oxygen minimum zones, changes in primary productivity, changes in ocean circulation patterns, sea level rises and increases in extreme weather events.

Marine fisheries catches consist largely of fishes and invertebrates that are biologically sensitive to changes in temperature, oxygen level and other ocean conditions; thus we expect that fisheries are being affected by climate change and ocean acidification (OA). In the ocean, physiological performance of aquatic and marine water-breathing organisms is strongly dependent on temperature and oxygen (Pauly, 2010; Pörtner, 2010). Thus, changes in oxygen supply is expected to have large implications for respiration and body functions of fishes and invertebrates. When temperature becomes either too high or too low, oxygen supply capacity decreases relative to oxygen demand and thus limits animals' metabolism. Our understanding of the physiological sensitivity and responses to ocean temperature, oxygen, acidity and other water properties allows us to develop hypotheses about how climate change and ocean acidification are affecting exploited fish stocks and fisheries.

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Theses hypotheses, all tested at a global scale, include:

1. Given ocean warming, fishes and invertebrates will be shifting their distributions, mainly to higher latitude and deeper waters to maintain their thermal niche;
2. In non-tropical systems, warmer-water species will increase their contribution to local catches;
3. Maximum body size of fishes decreases as the oceans become warmer and less oxygenated;
4. Global marine catches will decline, particularly in the tropics.

Here, in this abbreviated contribution, we outline only how (1) and (2) were addressed (see Cheung *et al.,* 2013; and Cheung *et al.*, 2010, respectively, for points 3 and 4).

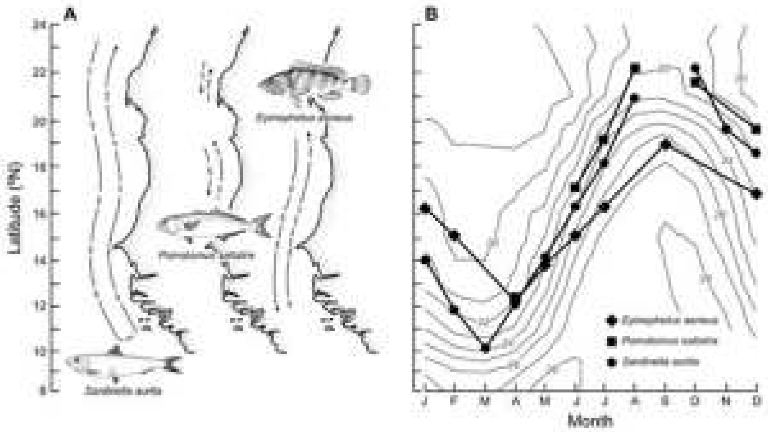

Figure 1. Seasonal latitudinal migrations of some Northwest African fishes; A) summary of information on the occurrence in space (latitude) and time (month) of three species, *Sardinella aurita*, *Pomatomus saltator* and *Epinephelus aeneus,* from Boëly (1979); Boëly *et al.* (1978); Champagnat and Domain (1978) and Barry-Gérard (1994); B) Same as in A, but plotted against mean monthly temperature (data from COADS). The seasonal migrations result in the three species remaining in approximately the same temperature (and hence oxygen regime) throughout the year (adapted from Pauly 1994).

As a conceptual first step (though one taken last in the sequence of studies described below), we studied the signature of the effects of ocean temperature changes on species composition of fisheries catches, using a newly developed metric called 'mean temperature of the catch' (MTC). The second step was to develop a species distribution model, called Dynamic Bioclimate Envelope Model (DBEM) that predicts changes in the distribution ranges of exploited marine species, and the patterns of species richness in response to changing ocean conditions. Once developed, the DBEM was modified to progressively account for an increasing number of features, i.e., the population dynamics of the species included their dispersal modes, interactions with other species, association with different habitats, oxygen requirements, resistance to low pH, etc. The third step was to use macroecological theory to derive the theoretical relationship between net primary production (NPP), biogeography and fisheries catch potential, and to express that relationship in a single empirical equation. Then, by combining projected changes in net primary production and species distributions (Palomares *et al.*, this volume, p. 33), future changes in distribution of catch potential and the maximum body size of exploited species could be projected. Finally, using the DBEM and basic principles of geometry and physiology, the effects of ocean warming and deoxygenation on the maximum body size of exploited fishes could be projected.

Mean temperature of catch

Marine fishes and invertebrates exhibit physiological thermal tolerances that constrain them to live within a certain range of water temperatures (Pauly, 2010). Thus, for example, the seasonal migration of fishes north and south along the coast of Northwest Africa tracks the seasonal temperature oscillations along that coast. (Figure 1). Similalry, as the oceans warm up, fish and invertebrates have to shift their distribution in order to maintain themselves in habitats with their preferred temperature. This results (at locations outside of the tropics), in changes in species composition, as those taxa increase in abundance that are adapted to warmer waters.

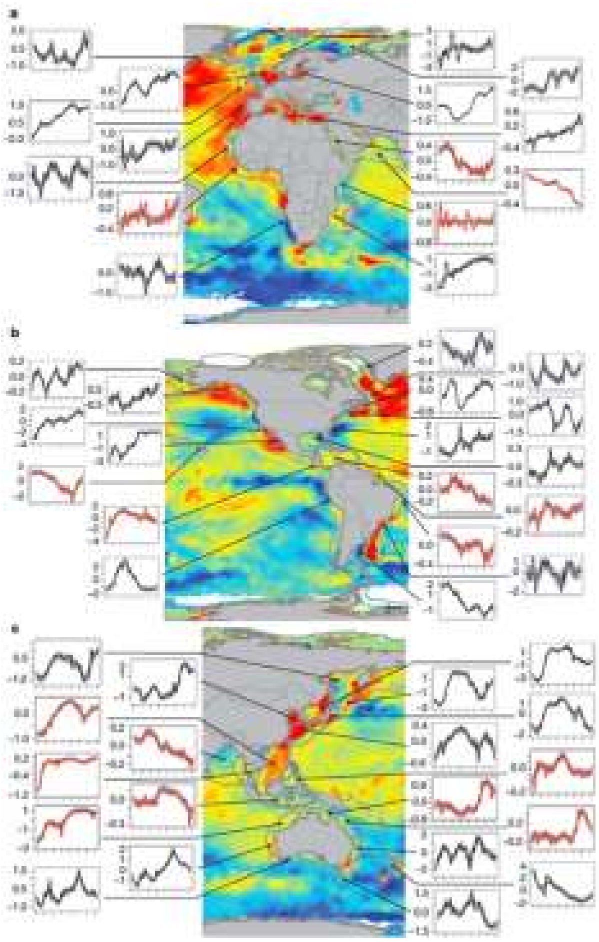

Figure 2. Estimated mean temperature of catch (MTC) anomalies from 1970 to 2006 in the large marine ecosystems in (a) Europe and Africa; (b) North and South America; and (c) Asia and Oceania. The red lines represent LMEs in the tropics, while the grey areas represent 95% confidence intervals (from Cheung *et al*., 2013a).

A newly developed index, i.e., the *Mean Temperature of Catch* (MTC) shows that global catches are increasing dominated by warmer water species (Cheung *et al.,* 2013a). The MTC is the weighted average of the preferred temperatures of the various fish and invertebrate species in the catch. The preferred temperature of each species (which is expected to be fairly stable in evolutionary time) was predicted from overlaying the current distribution of the species (as predicted using the method described in Close *et al.,* 2006, and being udpated using Palomares *et al.*, this volume, p. 33) and sea surface temperature (SST). Therein, species that are distributed in warmer waters will have a higher preferred mean temperature and *vice versa*. Thus, if the catch, e.g., of a small country in the temperate zone is increasingly dominated by warmer-water species, its MTC would increase.

Using the *Sea Around Us* catch data up to 2006, the MTC was calculated for all the large marine ecosystems (LMEs) of the world from 1970 to 2006. After accounting for the effects of fishing and large-scale oceanographic variability, global MTC increased at a rate of 0.19° Celsius per decade between 1970 and 2006, and non-tropical MTC increased at a rate of 0.23° Celsius per decade (Figure 2). In tropical areas, the MTC increased initially because of the reduction in the proportion of subtropical species catches, but subsequently stabilized as the scope for further tropicalization of communities became limited (Figure 3). By showing that changes in the MTC are significantly related to changes in SST across large marine ecosystems, this study showed conclusively that ocean warming has already affected global fisheries catch composition in the past four decades.

Projecting distribution shifts of exploited species

Given that changes in the composition of fisheries catches are likely to be driven by warming-induced biogeographic shifts, the next step was to investigate whether exploited species would continue to shift their biogeography in the future under climate change conditions, and how it would continue to affect the composition of exploited species.

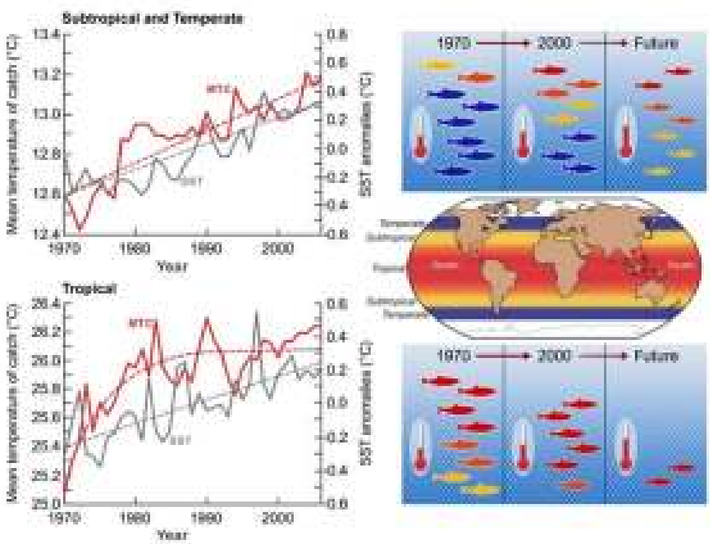

Figure 3. Average mean temperature of catch (MTC) and sea surface temperature (SST) from non-tropical (upper panels) and tropical Large Marine Ecosystems (lower panels); MTC tracks SST in temperate and subtropical LME, but not in tropical LMEs (Cheung *et al.*, 2013a). The right panels combine these results with future projections, where fishes with cold-water (dark blue) and subtropical affinities (yellow) are gradually replaced by tropical fishes (red) (Cheung *et al.*, 2010), while the size of all species generally declines (Cheung *et al.*, 2013b).

A Dynamic Bioclimate Envelope Model (DBEM) was developed to project future distributions of over 1,000 exploited fishes and invertebrates. The DBEM, described in Cheung *et al.* (2008a, 2009) and later in Cheung *et al.* (2011) and Fernandes *et al.* (2013), predicts a species' range (on the global *Sea Around Us* ½ x ½ degree cell grid) based on the association between the modelled distributions and environmental variables. The original distributions modelled using the method described by Close *et al.* (2006) are being updated and improved as described in Palomares *et al.* (this volume, p. 33).

We applied the DBEM to project likely future distributions of the over 1,000 species of exploited fishes and invertebrates under climate change scenarios developed by the IPCC. These species (Figure 1) include the overwhelming majority of the taxa whose population is large enough to generate catches that are reported at the species level in the global fisheries statistics of the Food and Agriculture Organization of the United Nations (FAO), and thus represent a very large sample of marine macrofauna. The rate of range shift and the intensity of species invasion and local extinction in the global ocean by 2050 relative to the 2000 period were then calculated.

The resulting projections show that climate change leads, overall, to range shifts to higher latitude and deeper waters (Figure 4), although some species display range shifts in the opposite direction, as they follow local, rather than large-scale climate change gradients (Cheung *et al.,* 2009). Thus, numerous local extinctions (exterpations) in sub-polar regions, the tropics and semi-enclosed seas can be expected. Simultaneously, species invasions are projected to be most frequent in the Arctic and the Southern Ocean. Moreover, these results support the hypothesis that the observed pattern of changes in species composition of catches, as indicated by the MTC introduced above, will continue in the future.

Conclusions

We have detected a signature of ocean warming on the global fisheries in the last four decades, and have also projected that such changes would continue in the next decades. This will lead to strong species turnover (Cheung *et al.*, 2009), redistribution of fisheries catch potential (Cheung *et al.*, 2010) and decreases in the maximum body size of exploited species of fish and invertebrates (Cheung *et al.*, 2013b). Results from these global scale analyses highlighted the natural inequality in climate change impacts to different regions of the world. Specifically, the tropics will be impacted by high rates of local species extinctions, decreases in catch potential, and a relatively larger decrease in mean body size of fishes. While many tropical communities are dependent on fisheries resources for food and livelihood (Zeller *et al.*, 2014), their economic and societal capacity to adapt to climate change impacts on fisheries is often low. Thus, tropical fisheries are highly vulnerable to climate change, although tropical countries contribute little to the greenhouse gas emissions that cause climate change.

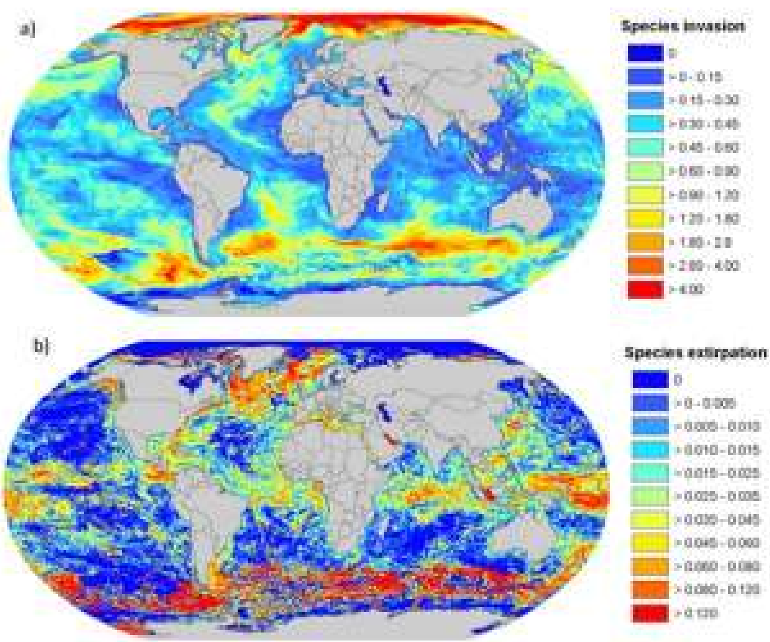

Figure 4. Projected intensity of species invasion (a) and local extinction (b) by 2050 relative to 2000 (10-year average) under the SRES A1B scenario. (Redrawn from Cheung *et al.*, 2009).

Future studies should address additional challenges to detecting, attributing and projecting climate change and ocean acidification impacts on marine fisheries. First, the adaptive scope of exploited marine species and their fisheries to impacts from climate change and ocean acidification will need to be evaluated. Second, different modeling approaches for projecting future seafood production under climate change and ocean acidification will need to be tested, to assess the skills of these approaches and quantify the level of uncertainties associated with the model projections. Third, more regional studies to down-scale the global analyses will need to be conducted, through which the weaknesses associated with the coarse projections of ocean properties from global circulation models can be better addressed. Also, these regional-scale analyses are more useful to informing national fisheries and coastal management agencies, which will both be challenged by global warming in coming years.

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Modelling the ecosystems of the global oceans [11]

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The Life Sciences have reached a new era, that of the 'Big New Biology' (Thessen and Patterson, 2011). Ecology is following a similar path, and has turned into a 'data-intensive science' (Michener and Jones, 2012). This is also the case for marine biology and fisheries science. Indeed, our work is increasingly relying on large pre-existing datasets, allowing for new insights on phenomena visible mainly or only at very large or even global scales (e.g., Pauly, 2007; Christensen *et al.*, 2009; see also Pauly and Zeller, this volume, p. 109).

However, open and reliable data sharing in marine biology and fisheries science is still not as extensive as in the historical 'big' sciences, such as oceanography, meteorology or astronomy, where massive data-sharing is the norm (Pauly, 1995; Edwards, 2010; Thessen and Patterson, 2011). The open-access principle of sharing information online for free has been increasingly applied to publications, but much less to unpublished data, mainly due to issues with recognition and sense of data ownership (Vision, 2010; Thessen and Patterson, 2011). Although incentives for digitization of non-digital materials have been growing, existing repositories likely represent less than 1% of the data in ecology (Reichman *et al.*, 2011; Thessen and Patterson, 2011).

Gathering information for and from Ecopath with Ecosim models

In aquatic ecology, the Ecopath with Ecosim (EwE) modelling approach has been widely applied to inform ecosystem-based management (e.g., Jarre-Teichmann, 1998; Christensen and Walters, 2011; Coll and Libralato, 2012), since its original development in the early 1980s (Polovina, 1984) and its relaunch in 1992 (Christensen and Pauly, 1992). The EwE modeling approach was primarily developed as a tool-box to help fisheries managers and answer '*what if*' questions about policy that could not be addressed with single-species stock assessment models (Christensen and Walters, 2011; Pauly *et al.*, 2000). The EwE software is user-friendly, free (under the terms of the GNU General Public License) and downloadable online (www.ecopath.org). Thus, hundreds of EwE models representing aquatic (but also some terrestrial) ecosystems have been developed and published worldwide. The foundation of the EwE modelling approach is an Ecopath model, which creates a static mass-balanced snapshot

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of the resources in an ecosystem and their interactions, represented by trophically linked biomass 'pools'.

By formalizing available knowledge on a given ecosystem, EwE helps in understanding its scructure and functioning, and thus may be seen as an important source of mutually compatible data. Indeed, building an EwE model requires the collection, compilation and harmonization of various types of information: descriptive data on species abundance, diet composition and catch; computed data on species production and consumption, and the biomass trends resulting from various exploitation scenarios. Several meta-analyses based on smaller sets of EwE models have been performed, focusing either on theoretical ecology and ecological concepts (e.g., Arreguín-Sánchez, 2011; Gascuel *et al.*, 2008), or on ecosystems and species of particular interest (e.g., Christensen *et al.*, 2003; Pauly *et al.*, 2009). However, only few meta-analyses based on a large collection of EwE models have been published (e.g., Christensen, 1995; Coll *et al.*, 2012; Pikitch *et al.*, 2012; Heymans *et al.*, 2014).

Global overview of EwE applications and presentation of a meta-analysis case study

EcoBase is an online information repository of EwE models published in the scientific literature, developed with the intention of making the models discoverable, accessible, and reusable to the scientific community (sirs.agrocampus-ouest.fr/EcoBase). Details on the structure, usage and capabilities of *EcoBase* can be found in the report introducing *EcoBase* (Colléter *et al.*, 2013), which is available online. Colléter *et al.* (2013) first gave a global overview of the applications of the EwE modeling approach in the scientific literature, using metadata gathered on the 435 EwE models registered in *EcoBase* to-date. We focused on the objectives of the EwE-based studies, the complexity and scope of the models, and the general characteristics of the modeled ecosystems and noted the complementary use of EcoTroph in EwE models. Based on the year of publication of the models, we also analyzed the evolution of the EwE applications over the past thirty years.

We present an application example detailed in Christensen *et al.* (2014), based on 200 models and a methodology that has been previously applied to the North Atlantic, South East Asia, and West Africa (e.g., Christensen *et al.*, 2003). Therein, the 200 EwE models were used to provide snapshots of how much life there was in the ocean at given points in time and space. Christensen *et al.* (2014) then evaluated how the environmental conditions at each point relate to environmental parameters, from which they developed a regression model to predict biomass trends. Finally, they used global environmental databases to predict the spatial distribution of fish biomass. This allowed Christensen *et al.* (2014) to predict the biomass trends for higher-trophic level predatory fish, i.e. the larger predatory ‘table fish’, as well as for the lower-trophic level prey fish, such as small pelagics (sardines, anchovies, capelins, etc.), which are used mainly for fishmeal and oil. Given the recent controversy over whether ‘fishing down the food web’ is a phenomenon actually occurring in nature (Pauly *et al.*, 1998) or a sampling artifact with no or little relation to the underlying ecosystem structure. This study contributes to the the discussion by evaluating how the biomass of high-trophic level species has changed relative to the biomass of low-trophic level species. For a rapidly growing list of examples and published case studies on ‘fishing down’, see also www.fishingdown.org.

Global overview of the applications of the EwE modelling approach

The 435 Ecopath and/or EwE models, covering the entire world ocean were documented in *EcoBase*. These models were used to tackle a wide range of ecological issues; notably, 87% of the models were developed to answer questions regarding the functioning of the ecosystem, 64% to analyze fisheries, 34% to focus on particular species of interest, and 11% to consider environmental variability (the percentages add to >100 because models may have more than one purpose). Less than 10% of the models focused on MPAs, pollution or aquaculture. The module which identifies the ‘keystone’ species (or groups) in ecosystems, based on Libralato *et al.* (2006), was used in 11% of the models, whereas the Ecotracer plug-in for tracking pollutants has been applied in less than 1% of the models (but see Booth *et al.*, this volume, p. 99).

Figure 1. Distribution of EwE models illustrating the wide global coverage. Color density indicates the number of distinct models in each region.

The best represented ecosystem types are continental shelves (32% of the models), bays/fjords (14%), open oceans (13%) and freshwater lakes (8%); 49% of the models are located in the tropics, 44% in temperate areas, and only 7% in high latitudes (see Figure 1). EwE models have been developed to study aquatic ecosystems worldwide, with some regions better covered than other. Overall, the Northern and Central Atlantic Ocean is the region with the highest proportion of EwE models. All FAO areas (see Figure 1 in Zeller and Pauly, this volume, p. 15) have at least one model, but five areas have about 40 models each: the Northeast Atlantic and the Eastern Central Atlantic comprise 10% of the models each; and the Western Central Atlantic, the Northwest Atlantic, and the Mediterranean and Black Sea comprise 9% of the models each. The Humboldt Current, the Gulf of Alaska, the Mediterranean and the Guinea Current are the Large Matine Ecosystems (LMEs; see http://www.seaaroundus.org/lme/) with the highest number of models (at least 5% each).

Recently developed models tend to be less aggregated and thus more complex, although highly aggregated models are still being published. During the first decade of the development of the EwE modeling approach, the total number of groups defined in the models ranged from 7 to 27. Over time, the range of the number of groups has expanded toward more groups, up to 67 groups in the past decade. The median number if groups was around 15 groups between 1984 and 1993, while it wa around 30 groups between 2004 and 2014. In contrast, the time period represented by the models tended to decrease over time; thus the median number of years represented by the models ranged from 3 years in 1984-1993 to 1 year in 2004-2014. The areas covered by the models has expanded towards very large areas, and the median area has shifted accordingly, from about 1,000 km^2 in 1984-1993 to about 100,000 km^2 in 1994-2014.

Fish biomass in the world ocean: a century of decline

Using 200 EwE models, each providing a snapshot of how much life there was in the ocean at given points in time and space (Figure 1), Christensen *et al.* (2014) evaluated trends in biomass of fish separately for higher-trophic level predatory fish ('table fish') and for the lower-trophic level prey fish. Their results suggested that the biomass of predatory fish has declined strongly (and significantly) over the last hundred years (Figure 2). For the 200 models, covering the period from 1880-2010, they evaluated how the conditions at each point relate to environmental parameters and other variables, and obtained a multiple regression whose coefficient of determination (R^2) is 0.70, indicating that it explains 70% of the variation in the data set. The predictor variables are all highly significant apart from the factorial variable for FAO areas 18 and 31 (representing the Amerasian Arctic and the Caribbean). The signs of the predictor variable coefficients all are as expected, negative for biomass, distance, and temperature, and positive for primary production and the upwelling index. The model suggested that we have lost 1.5% of the biomass of higher trophic level fish per year, suggesting that higher trophic level biomass may have declined by as much as 60-70% over the last 100 years. Over a one hundred year time period, this implies also that there are now more than twice as much low-trophic level ('prey') fish in the global ocean than they were were a century ago.

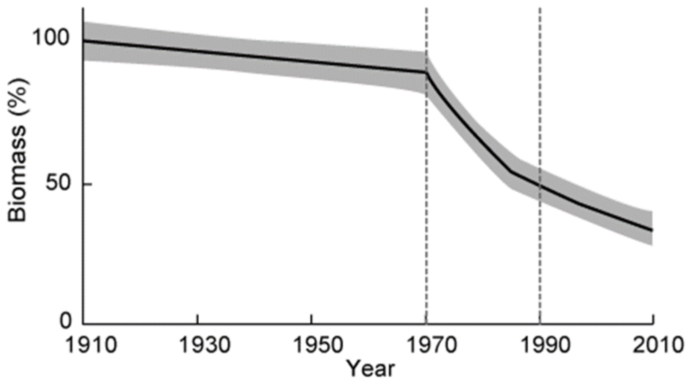

Figure 2. Global biomass trends for predatory fish during 1910-2010 as predicted based on 200 ecosystem models and 1000 times random resampling of 30% of data points. The lines indicate median values and 95% confidence intervals.

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Jellyfish fisheries – a global assessment [12]

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Jellyfish are considered traditional cuisine in China, where they have been eaten for more than 1,700 years (Omori and Nakano, 2001; Li and Hsieh, 2004). Other countries in Southeast Asia such as Thailand, Indonesia, and Malaysia have been catching jellyfish for decades, primarily for export to China and Japan. Despite this history, information on jellyfish fisheries is sparse and disaggregated. There are currently at least 18 countries catching jellyfish for food, and a dozen or more are either exploring new fisheries or have been involved in jellyfish fisheries in the past. Many countries do not report their catches of jellyfish explicitly to the Food and Agriculture Organization of the United Nations (FAO), including them either as 'miscellaneous marine invertebrates' or not at all. As a result, the current average annual catch of jellyfish reported by FAO is approximately 350,000 t, while the present global analysis reveals the average annual catch during the period from 2000 to 2013 (Figure 1) is at least 892,000 t, more than 2.5 times the official estimates.

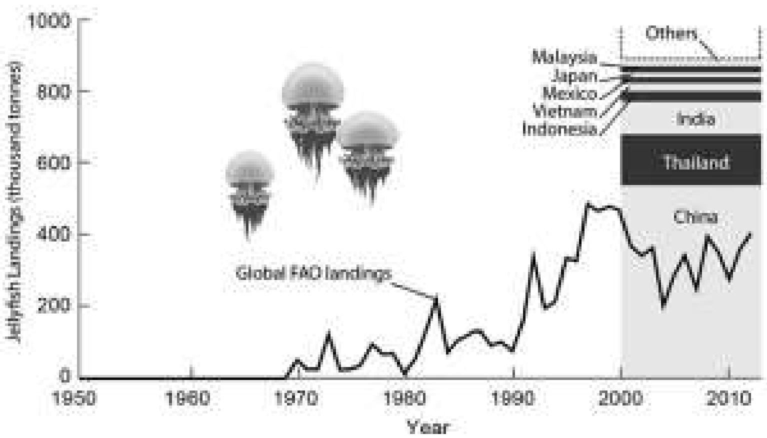

Figure 1. Chart showing jellyfish landings according to FAO (line) and estimates from this study (bar).

Fisheries for jellyfish are usually characterized by short fishing seasons of a few months as well as dramatic inter-annual variations in catch (Omori, 1978; Omori and Nakano, 2001). In fact, rapid changes in exploitable biomass of jellyfish are probably more of a concern than for any other fishery (Kingsford *et al.,* 2000). Combined with pollution from processing plants, as well as a conspicuous lack of research and regulation, this has led to conflict and instability in jellyfish fisheries in many regions. While the asexual reproductive phases of edible jellyfish are likely a buffer against overfishing, they do not appear to be a reliable safeguard, and overfishing of jellyfish stocks appears to have occurred in some locations, e.g., in China (Dong *et al.,* 2014) and the Salish Sea in the Pacific Nortwest (Mills, 2001).

More recently, fisheries for jellyfish have expanded around the globe (Figure 2), often driven by a combination of factors that may include a collapse of local fish stocks and increased interest from East Asian buyers. Established fishers, to whom jellyfish are often a costly nuisance, typically welcome these test fisheries enthusiastically. However, such exploratory fisheries are

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often unsuccessful, potentially due to scant research, e.g., Canada (Sloan and Gunn, 1985) or onerous regulations, e.g., Australia (Kingsford *et al.,* 2000).

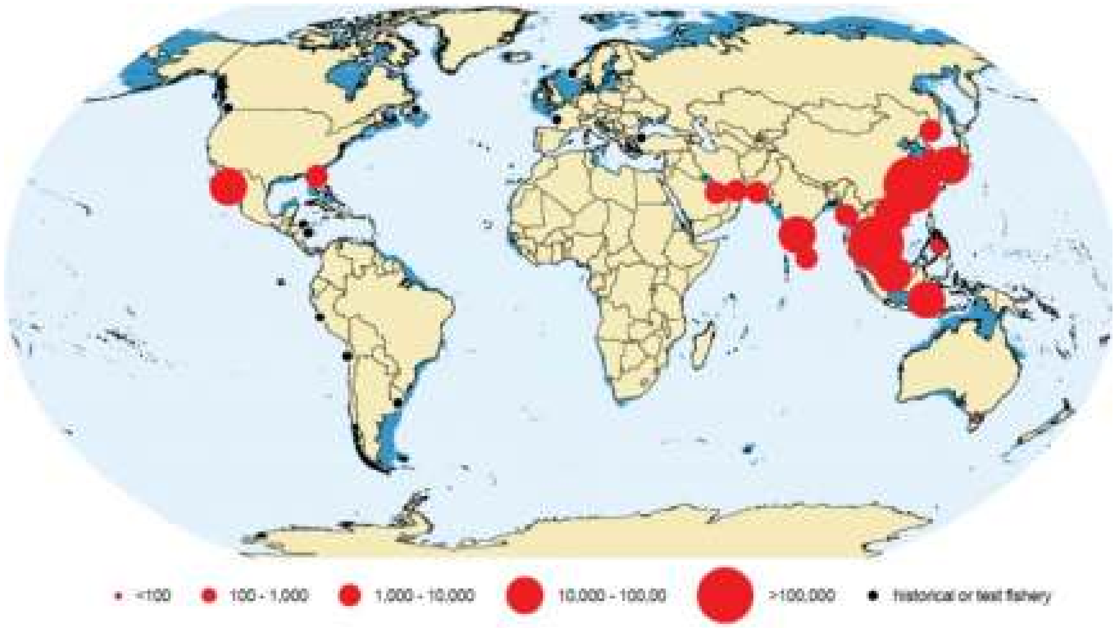

Figure 2. Jellyfish fisheries around the world. Red circles indicate magnitude category of catch in tonnes (see legend). Black circles indicate either historical fisheries (which no longer operate) or test fisheries (which are not yet established). Circles are not representative of precise catch locations.

Target species

With the exception of Mexico, currently all catches of jellyfish reported by FAO are classified as '*Rhizostoma* spp.', which is incorrect in many cases. The number of identified species of edible jellyfish worldwide is unclear, and is typically underestimated (e.g., Omori, 1981; Sloan, 1986; Hsieh and Rudloe, 1994; Omori and Nakano, 2001; Armani *et al.,* 2013), due also to the taxonomy of edible jellyfish species being confused (Omori and Kitamura, 2004). However, at least 20 different species of jellyfish have been identified as being consumed by humans, with up to an additional 15 edible species either unconfirmed or under evaluation. Most edible species of jellyfish belong to the scyphozoan Order Rhizostomeae. These jellyfish are typically large, with relatively tough and rigid tissues. *Rhopilema esculentum* is the most valuable species and is currently the choice for hatchery and aquaculture operations in China (You *et al.,* 2007; Dong *et al.,* 2009). The giant jellyfish, *Nemopilema nomurai*, is also widely exploited in East Asia, in much larger quantities than have been reported until recently (Li *et al.,* 2014). There are reports that scyphomedusae from the Order Semaeostomeae may also be consumed, such as *Aurelia* spp., *Chrysaora* spp., and *Cyanea* spp.; however, it does not appear that any operations are currently targeting these less desirable species at commercial scales. There is also limited information to suggest cubozoans are consumed in some regions. Shih (1977) reported that the people of the Pacific atoll of Tawara, in Kiribati, consume freshly caught or sun-dried *Tamoya* sp. after boiling them. Purcell *et al.* (2007) reported that aboriginal peoples in Taitung, Taiwan eat cubomedusae. A number of jellyfish species have also been targeted for other reasons (e.g., nuisance, research, pharmaceuticals, etc.), which means that the total number of exploited jellyfish species is even higher (Kingsford *et al.,* 2000).

Rhizostome jellyfish, which constitute the bulk of the edible species, have several life history characteristics that may help to mitigate overfishing. These jellyfish have a bipartite life cycle, consisting of a pelagic medusoid phase and a sessile polypoid phase. Female medusae are typically highly fecund, producing millions of eggs (e.g., Huang *et al.,* 1985; Kikinger, 1992). Fertilized planulae attach to hard substrates, which may be decreasing, as is the case with mangroves (Valiela *et al.,* 2001), or increasing, as with anthropogenic substrates (Duarte *et al.,* 2013). Polyps of many species may asexually bud additional polyps (Lucas *et al.,* 2012) or transform into cysts capable of resisting harsh environmental conditions (Arai, 2009). When conditions become favourable, polyps begin to segment and asexually release ephyrae through the process of strobilation. Each polyp may release numerous ephyrae and will often strobilate more than once within the same season. Ephyrae join the plankton and grow rapidly into medusae (Palomares and Pauly, 2009), at which point they may be targeted for fisheries. This bipartite life cycle may provide a buffer against overfishing, as subsequent recruitment is possible even without spawning adults. Nonetheless, overfishing of jellyfish stocks is possible as several case studies attest, and therefore management strategies for sustainable fisheries should be employed.

Estimating the Current Global Catch

A global estimate of current jellyfish landings was calculated by estimating the mean annual catch by country since the year 2000. Where possible, FAO catch statistics for jellyfish were verified using additional sources of data at the country or regional level. Some countries may report bycatch of jellyfish from other fisheries to FAO, regardless of whether or not it is landed. On one hand, it is positive that FAO reports these values, as they are part of the total catch. However, FAO makes no distinction between bycatch and targeted landings, which is problematic when interpreting the data. In this case, fishing entities such as Namibia, the United Kingdom, and the Falkland Islands appear to have fisheries for jellyfish, when in fact these statistics are likely to indicate discarded bycatch. More detailed reporting by FAO and individual countries would be beneficial. In select cases (e.g., India), only ‘production’ statistics were available, and a scaling factor of 4 was used to convert from semi-dried processed product weight back to wet weight. Processed jellyfish product can range anywhere from 7% to nearly 30% of the original wet weight, depending on the species and processing formula used. As reported values are typically much less than 25% of the original wet weight (e.g., Omori, 1981; Morikawa, 1984; Huang, 1986; Jones and Rudloe, 1995; Fisheries Victoria and MAFRI, 2002; Li *et al.,* 2014), a scaling factor of 4 is conservative. In some cases, illegal, unreported, or unregulated (IUU) landing estimates were added to reported FAO statistics. IUU catches of jellyfish were based on estimates from catch reconstructions performed as part of the *Sea Around Us* (Zeller and Pauly, this volume, p. 15).

The global estimate of nearly 900,000 t is approximately 2.5 times larger than previous estimates (e.g., Omori and Nakano, 2001) and that derived using FAO catch statistics. Despite this difference, 900,000 t is likely an underestimate of the true global catch due to the conservative assumptions used and the fact that reporting of jellyfish catches is poor. For example, the estimate for India was calculated using mean catch between 2000 and 2003 (Anonymous, 2005), which were reported by only 1-3 states depending on the year in question. However, at least 6 states in India are known to catch jellyfish. As such, the world catch of jellyfish for food likely exceeds 1 million t annually. In addition, the estimate does not include any bycatch or discards of jellyfish, which can be huge, often resulting in losses to fishers of tens or hundreds of millions of dollars annually (Purcell *et al.,* 2007; Uye, 2008; Kim *et al.,* 2012). In fact, the amount of discarded jellyfish bycatch is likely to exceed by far the landings of edible jellyfish, i.e., to add millions of tonnes to the world's marine catches.

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Global seabird populations and their food consumption [13]

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Seabirds share the oceans with us, and they are strongly impacted by fisheries either directly, e.g., as bycatch of longline fisheries, and indirectly, because fisheries reduce the abundance of their fish prey below crucial thresholds (Cury *et al.*, 2011). Because they globally consume millions of tonnes of fish and marine invertebrates per year, seabirds play an integral role in the structure, function and resilience of marine ecosystems. In this, they are similar to marine mammals, though nobody has so far dared propose to 'cull' seabirds so we would have more fish to catch, a claim some people routinely make with regards to marine mammals (see Gerber *et al.,* 2009; Morissette *et al.*, 2012; Pannozzo, 2013).

Because of seabirds' role in the functioning of marine ecosystems, the *Sea Around Us* has taken an early interest in mapping their worldwide distributions, such that they could be considered in global modelling efforts of the sort discussed in Christensen *et al.* (2009) and Colléter *et al.* (this volume, p. 71). The first such product was the preliminary maps of Karpouzi *et al.* (2007), which came along with a database of the estimates of abundance through time and the body sizes and diet composition of 351 species of seabirds (Karpouzi, 2005), now incorporated in SeaLifeBase (see www.sealifebase.org).

This database, which was used for a number of contributions on seabirds and their roles in marine ecosystems (Kaschner et al. 2006; Karpouzi and Pauly 2008), was extended by Paleczny (2012), while the number of species covered was reduced to true seabirds (benthic feeding ducks scoters, eiders and merganser with little potential for overlap with fisheries were not considered further). Moreover, two distribution range maps (breeding and non-breeding, i.e., foraging ranges) were generated for each of these 324 remaining species, and were then used to generate the global maps in this chapter, and other products (Paleczny and Pauly 2011; Cheung *et al.*, 2012; Coll *et al.*, 2012).

Seabird Biodiversity and Ecology

Seabirds are birds that have evolved to forage in the ocean, but nest in colonies on islands and coastal cliffs. While Brooke (2004) estimated 309 species of seabirds to have a cumulative population of 0.7 billion, Paleczny (2012) estimated a global seabird population of 0.77 billion, belonging to approximately 324 species (approximate because of recent taxonomic revisions, e.g., Rains *et al.,* 2011), and four orders, Procellariiformes (i.e., petrels, diving petrels, storm-petrels, and albatrosses), Charadriiformes (i.e., auks, terns, gulls, and skuas), Sphenisciformes (penguins), and Pelecaniformes (boobies, cormorants, frigatebirds, pelicans, and tropicbirds).

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Seabirds are unique among avian taxa for their relatively *K*-selected life-history strategy (i.e., large body size, low population growth rate, and long lifespan) and ability to travel long distances to forage for prey (up to thousands of kilometres per foraging trip in some species). Jointly, the distribution of these 324 species covers the world's oceans, with species richness being highest in productive regions, particularly in the southern hemisphere (Figure 1). This greater species endemism in the southern hemisphere may be a consequence of spatial isolation between breeding populations, as the distances between islands and continents supporting seabird colonies are greater than in the northern hemisphere.

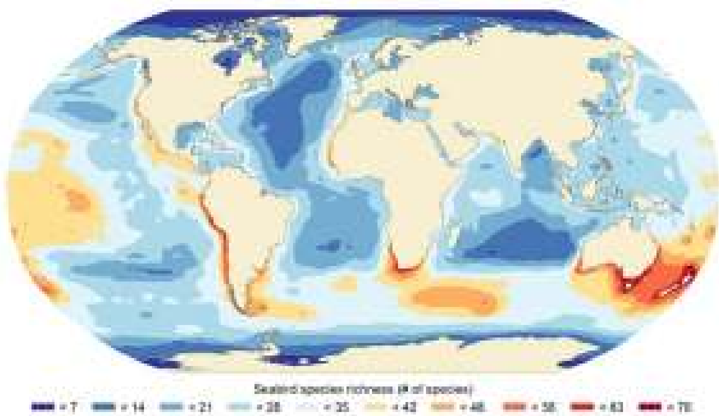

Figure 1. Seabird species richness, as was constructed using maps of total range for each of the world's 324 seabird species.

The main prey of seabirds are krill, fish, and squid, and less commonly benthic crustaceans, other seabirds, marine mammal carrion, and jellyfish (see Brotz, this volume, p. 77). The relative importance of these diet items varies between seabird taxa, as well as regionally and seasonally. For example, seabirds may switch diets between breeding and non-breeding season, with adults commonly provisioning high energy density prey (e.g., forage fish) to their chicks. Seabirds are also prey in marine and coastal ecosystems, consumed by a variety of marine mammals (e.g., seals, sea lions, walrus, sea otters, killer whales, polar bears), sharks, coastal birds of prey (e.g., hawks, eagles), and other seabirds (see Hipfner *et al.,* 2012). Seabirds share symbiotic foraging interactions with other marine fauna, for example, temperate foraging auks have mutualistic relationships with marine mammals (e.g., Anderwald *et al.,* 2011), and most tropical seabirds forage commensally with dolphins and tunas (Ballance and Pitman, 1999). Seabirds are also important cross-ecosystem nutrient subsidizers, transporting nutrients via their guano to their breeding colonies, where they play a major role in enriching the productivity and biodiversity of the terrestrial and marine ecosystems surrounding their colonies (e.g., Croll *et al.,* 2005). Due to their charismatic nature and accessibility at terrestrial breeding colonies, seabirds provide additional ecosystem services such as opportunities for wildlife interactions and ecotourism (Lewis *et al.,* 2012) and opportunities to monitor change in marine ecosystems (Piatt *et al.,* 2007), including fisheries-induced changes (Einoder, 2009).

Seabird populations are, however, threatened by humans. Throughout history, we have depleted seabird populations by hunting seabirds for their feathers, meat and oil, and introducing previously absent predators to colonies (Croxall *et al.,* 1984; Roberts, 2007). More recently, within the modern industrial era, humans additionally threaten seabirds through coastal development, pollution, climate change, and fisheries (Croxall *et al.,* 2012), and even through renewed targeted commercial harvesting[14]. Seabird populations are particularly vulnerable to these threats because they have inherently low reproductive output and therefore slow population recovery rates (Russell, 1999). Also, they range over large areas, which increase their probability of exposure to spatially heterogeneous anthropogenic threats (Jodice and Suryan, 2010).

Global prey consumption by seabirds

The global consumption of prey by seabirds is estimated to have declined from approximately 104 million t·year^{-1} in the 1970s/1980s (Figure 2) to 75 million t·year^{-1} in 1990s/2000s. Our modern estimate of global prey consumption is comparable to a previous, but less detailed estimate of 70 million t·year^{-1} (Brooke, 2004), which it thus validates. For comparison, it is estimated that the global marine mammal population consumes 168 million t·year^{-1} (Kaschner *et al.,* 2006), while the global fisheries catch was about 70-80 million t·year^{-1} In the 1990s/2000s (Pauly *et al.,* 2005).

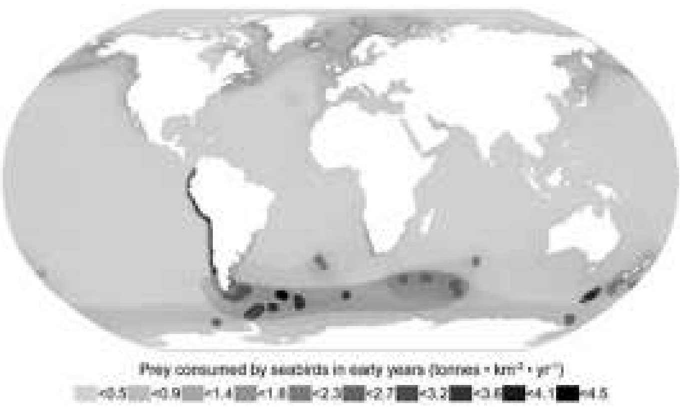

Figure 2. Annual food consumption by the global seabird population in the late 1970s/early 1980s. Based on per species estimates of breeding and non-breeding season density, and the daily food intake, estimated separately for the breeding and non-breeding seasons.

The order of importance of prey types was consistent between early and recent years. Ordered in declining contribution to overall biomass, prey consumed were krill, fish, squid, and other diet items. Forage fish, an important commercial fish group, comprised 15-16% (by mass) of all food consumed by seabirds, and 31-34% (by mass) of fish consumed by seabirds. Thus, while forage fish constitute a relatively small percentage of the global consumption by seabirds, they are of particular importance to the productivity of seabirds in upwelling ecosystems around the world (Cury *et al.,* 2011).

Seabird prey consumption was historically highest in the temperate and upwelling regions, especially off Peru, where it declined most severely, mirroring the spatial distribution of global seabird population changes.

It is important when interpreting these global estimates of prey consumption to be aware that food composition data are often biased towards the breeding season diet of seabirds, which may cause overestimation of the importance of fish in the diets of seabirds. On the other hand, by calculating the relative contribution of diet items using fixed diet compositions, we do not account for the long-term change in seabird diets that has been observed in some seabirds as a

[14] As evidenced by the Chinese fishing vessel that was caught in Mauritanian waters with its hull packed with 'dressed' seabirds (http://seabirds.net/posts/2013/02/13/evidence-for-massive-bycatch-in-chinese-fisheries/).

result of long-term ecosystem changes, e.g., a decline in the trophic level of some species (e.g., Becker and Beissinger, 2006).

The primary production required (PPR) to support seabirds declined from $0.79 \cdot 10^9$ t·year^{-1} in the 1970s/1980s to $0.63 \cdot 10^9$ t·year^{-1} in the 1990s/2000s, both estimates corresponding to approximately 1% of annual marine primary production. This is far less than the annual marine PPR to support marine fisheries, estimated at 8% by Pauly and Christensen (1995). Moreover, this estimate being based only on official reported catch data (and not reconstructed total catches), probably underestimated the PPR of fisheries. This could make the actual PPR of seabirds' food consumption one order of magnitude smaller than that of fisheries catches. This result was to be expected: while seabirds and marine fisheries take roughly similar amounts of biomass from the world's oceans, seabirds target much lower trophic levels than most fisheries.

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Global trend in mariculture production, 1950-2030 [15]

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The biodiversity of the ecosystems in which our economy and culture are embedded provide us with food. However, we often act as though these environmental food services are somehow free and infinite. In reality, the scope and scale of our current human activities, and our tendency to rely on a short-term mindset, are damaging to our environment, and threatening this provisioning role of natural systems (Sumaila and Walters, 2005). This threat is evident in the progression of many wild-capture fisheries to their present parlous state. Indeed, capture fisheries alone are no longer expected to be capable of supplying the projected increases in the demand for food 'fish', a term used to collectively refer to finfish, mollusks, crustaceans, and other aquatic animals that are caught or farmed. In the face of this expectation, aquaculture is anticipated to both fill the supply gap and to meet the growing worldwide consumption demand for fish (Ye, 1999; FAO, 2014).

Aquaculture is *"[t]he farming of aquatic organisms including fish, molluscs, crustaceans and aquatic plants, with some sort of intervention in the rearing process to enhance production, such as regular stocking, feeding, protection from predators, etc."* (Crespi and Coche, 2008a), and its ability to provide fish for human consumption has changed dramatically since the first documented production of herbivorous pond fish in China over 3000 years ago (Ling, 1977). Historically, aquaculture began as a low-intensity farming practice that applied basic rearing techniques to naturalized or native fish, primarily in freshwater pond environments. Today, global-scale commercial aquaculture production across freshwater, brackish, and marine environments provides a sizeable fraction of the fish consumed worldwide. Aquaculture, therefore, can also be expected to play a pivotal role in our attempt to meet the projected increases in global seafood demand.

While the freshwater sector continues to be a very important contributor to global supplies of food fish, there has been, since 1970, a reported threefold increase in the production and economic value of industrial-scale and intensively-reared marine and brackish, or 'mariculture', species (www.fao.org/fishery/statistics/global-aquaculture-production/en), a trend which appears to be holding in the early 2010s (FAO, 2014). These species fetch a high price in international markets, but the effects of their rearing practices can be detrimental to the health of coastal ecosystems and their people (Trujillo, 2007), as well as to fisheries (Naylor *et al*., 1998, 2000; Primavera 2006; Pullin *et al*., 2007; Goldburg, 2008). As mariculture production continues to increase, expand, and intensify worldwide, these negative trends may increasingly overshadow positive benefits and be further exacerbated in the future, particularly if policy

[15] Abbreviated from: Campbell, B., J. Alder, P. Trujillo and D. Pauly. A global analysis of mariculture production and its sustainability, 1950-2030, p. XX *In:* D. Pauly and D. Zeller (eds.) *Global Atlas of Marine Fisheries: Ecosystem Impacts and Analysis*. Island Press, Washington, D.C. [*in press*]

measures that promote and support sustainable mariculture development are not more widely implemented.

As part of its goal of improving understanding of the impact of fisheries on the world's marine ecosystems, the *Sea Around Us* supported mariculture-focused research intended to improve understanding of global mariculture sector trends, linkages and processes, and their relationship to global fisheries, to people, and to the environment through time. This work led to a spatially and taxonomically disaggregated database of mariculture production from 1950 to 2010; an index of mariculture sustainability; and scenario-based simulations exploring how sustainable mariculture development policies might affect the long-term health and well-being of people and their environment vis-à-vis meeting the future demand for food fish in 2030. The following highlights aspects of this work (see also Campbell and Pauly, 2013).

Assessing the accuracy of global mariculture production data

To assess the accuracy of currently reported global mariculture production trends, provincial-scale *Sea Around Us* mariculture production data (see www.searoundus.org/mariculture) were aggregated nationally and then globally. The resulting trend was then compared to the equivalent FAO FishStat Plus (*v*.2.31) Global Aquaculture Production Database trend, whose content for 1950 to 1984 was derived by FAO through a *post hoc* disaggregation of their combined fisheries and aquaculture database, a necessary step, albeit fraught with uncertainties. Both datasets indicate a tripling in production between 1950 and 1970, as well as an overall similarity in total annual global production growth from 1970 to 2010 (i.e., just under three quarters of the compared annual production in these years is similar to within 10%). The similarity between the datasets increases to 80% when China is excluded from the analysis (Figure 1).

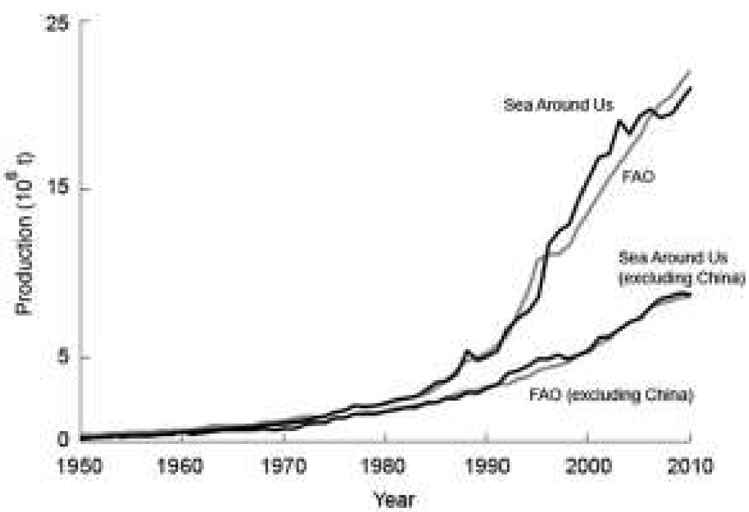

Figure 1. Comparison of global mariculture dataset trends between the FAO and the *Sea Around Us'* GMD datasets, with and without China. Log-linear regressions of both datasets for the years 1970 to 2010 yielded R^2 values of 0.993 with China and 0.990 without China, indicating a strong match between datasets. The similarity in slopes during this same time period suggests a mean global rate of production increase of 7.7% per year with China and 5.5% per year without China.

The general resemblance between the datasets applies to all regions of the world except Africa, whose mariculture production continues to be negligible (see FAO, 2014). The relatively largest discrepancies between global datasets were found in the data-poor era prior to 1970, more than a decade prior to the establishment of the FAO's aquaculture data repository. In these years, the *Sea Around Us* data provide the more conservative production estimate.

The overall match between the two databases also applies at the level of major taxonomic groups. Mollusks, a grouping composed primarily of bivalves (this analysis excludes a negligible production of cephalopods and gastropods) account for more than 70% of all mariculture production of 'fish' by weight worldwide since 1970. The time-series production trends for mollusks therefore strongly resemble the trends of total global mariculture production. Global marine and brackishwater finfish trends in both datasets increase more than tenfold between 1980 and 2010 and have comparable annual rates of growth. Reported crustacean production grew about fifty-fold between 1980 and 2010 in each dataset.

Note that we cannot exclude that the similarity between the FAO-reported mariculture statistic and the database presented here is due (at least in part) to the same bias, e.g., due to over-reporting of provincial mariculture production from China. This possibility was hinted at in a previous issue of SOFIA (FAO, 2012), but while over-reporting of fisheries catches was alluded to (see also Pauly and Zeller, this volume, p. 109), the potential over-reporting of mariculture production was not touched on in FAO (2014).

Geography of global mariculture

The mariculture data in the *Sea Around Us* global mariculture database were attributed to more than 600 different 'provinces' (i.e., subnational entities) in 112 coastal countries and territories between 1950 and 2004, with an additional half-dozen countries initiating commercial production between 2005 and 2010 (Figure 2). By comparison, the FAO distributes this historical production across a total of 21 FAO areas (www.fao.org/fishery/statistics/).

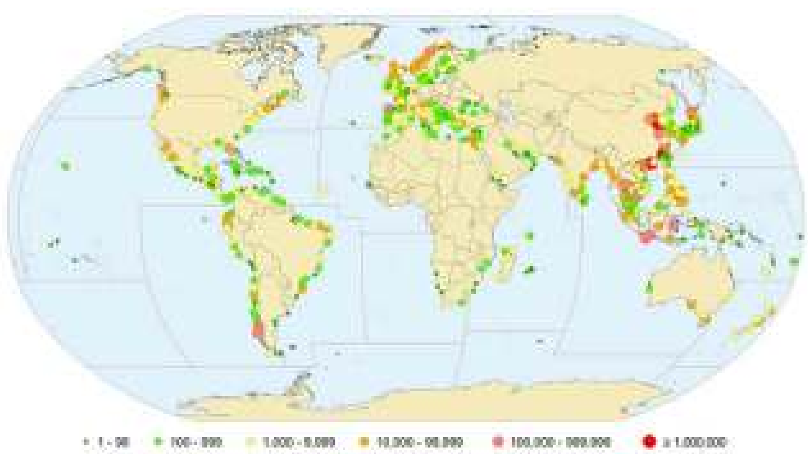

Figure 2. Average annual global mariculture production (t; all species combined) for 2000 to 2010, by coastal 'province' within countries.

Asia, both including and excluding China, has consistently produced the largest quantity of farmed marine and brackish species worldwide since 1950. Since 2000, China's top four mariculture coastal provinces (Liaoning, Shandong, Fujian, and Guangdong) each produced more than any other maritime country, i.e., an annual average of well in excess of one million t. Since 1980, three of these provinces experienced reported production increases of between one-and-a-half and three million t, primarily bivalves such as Pacific cup oyster (*Crassostrea gigas*) and Manila clam (*Ruditapes philippinarum*). Note that while finfish and crustacean production is substantial in Asia, regional mariculture production is consistently dominated by bivalves. It remains to be determined if China's mariculture data suffer from the same over-reporting issues previously identified for China's wild capture fisheries (Watson and Pauly, 2001).

Farming up the marine food web confirmed

As the total number of farmed taxa has increased over time, so too has the (production weighted) mean trophic level (TL) of the species produced (Figure 3A), or put differently, relatively greater quantities of predator species are being farmed around the world. This phenomenon, previously observed in studies of FAO data which analyzed total global aquaculture (Pauly *et al*., 2001), as well as mariculture production in the Mediterranean (Stergiou *et al*., 2008; Tsirlikas *et al.,* 2014), has been described as "farming up the food web" (Tacon *et al*., 2010). Farming up the food web is also apparent regionally (Figure 3A and B). However, since the 1990s, some regions and countries experienced either a leveling off or a decrease in their weighted mean TL. A decline in the mean TL of mariculture production occurred between 1980 and 2010 in Brazil, Denmark, Finland, Germany, Hong Kong, Nigeria, Norway, Peru, Singapore, and the United Kingdom, i.e., these countries are currently producing greater quantities of lower TL (herbivorous and omnivorous) species than they were in 1980. In contrast, China's weighted mean TL, with the majority of its production attributable to bivalves with a TL of 2.0, has remained relatively stable since the mid-1980s. The significant quantity of low-TL bivalves, brackish finfish, and crustaceans produced in China, and in Asia more broadly, are responsible for the low overall weighted mean global TL for mariculture.

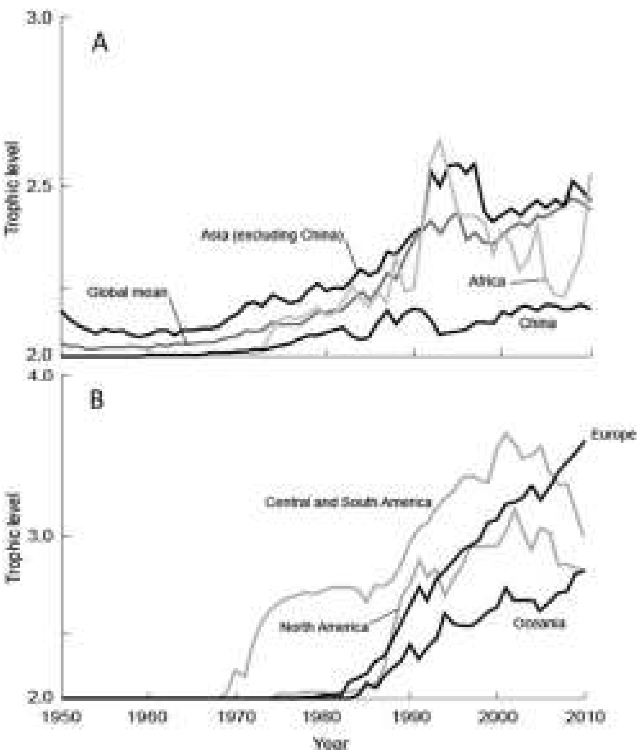

Figure 3. Change in the weighted mean trophic level (TL) of mariculture production in the GMD, demonstrating 'farming up the food web'; A) for the world on average, and for regions where the weighted mean TL is relatively stable or has decreased since the 1990s; and B) for regions with a marked increase in weighted mean TL since 1990.

Mariculture development scenarios for the next decades

Aquaculture, in particular the mariculture subsector, is a growing contributor to global fish supply worldwide. This trend is anticipated to continue in the future as fish demand increases (FAO, 2010). This potential increase in global mariculture production has led to concerns over the sustainability of the sector (Naylor *et al.,* 1998, 2000; Pauly *et al.,* 2002; Naylor and Burke, 2005; Primavera, 2006; Goldburg, 2008). However, few forecasts and scenario exercises exist that explicitly examine the future of global aquaculture (Delgado *et al.,* 2003; Brugère and Ridler, 2004). Also, the models that do exist do not explicitly consider the changing role and influence of mariculture in their price-based market drivers of supply and demand, nor do they consider potential supply-side production issues that may arise from social, economic, and environmental drivers of change.

The United Nations Global Environmental Outlook (GEO) "story and simulation" scenarios assessment methodology represents a departure from more traditional predictive models, which contain almost exclusively quantitative and price-mediated drivers of change. This is accomplished by providing both quantitative and replicable assessments of possible futures as well as a range of well-reasoned qualitative storylines (UNEP, 2002; Pauly *et al.*, 2003; Perterson *et al.*, 2003; Raskin, 2005). The most comprehensive UN report on the environment and development to date, the GEO-4 "environment for developement" assessment is primarily a capacity-building process (UNEP, 2007). The GEO's four overarching global development themes: *Markets First*, *Policy First*, *Security First*, and *Sustainability First*, and their underlying drivers, uncertainties and critical assumptions, were conceptualized and developed using a comprehensive process. The original GEO-4 analysis ideas and principles are assumed to apply for mariculture in the new analysis and are summarized below.

Markets first

In a *Markets First* world in 2030, key private sector actors, with active government support, are focused on improving the well-being of people and the environment through maximized economic growth and efficiency in the mariculture sector (UNEP, 2007). This emphasis on economic drivers of sustainable development has led to an increased liberalization, strengthening, expansion, and creation of international and regional trade agreements, particularly within Asia but also between Asia and the rest of the world.

By 2030, the growing Indian and Chinese middle classes are a driving force behind increases in both total and *per caput* global demand for diversified and high-value marine seafood (Delgado *et al.,* 2003; FAO, 2009). The widespread removal of trade barriers and technological constraints to increased production increases overall mariculture production more than the other global development scenarios. However, the overarching social priority of this scenario is to sustain profit rather than to sustain and improve the availability and accessibility of seafood for people (UNEP, 2007). Therefore, seafood markets remain dictated by traditional supply and demand economics with few government controls and the bulk of economic and social benefits derived from production still flows predominantly from poorer to richer countries and private entities (Kent, 1997; Delgado *et al.,* 2003).

Policy first

Under a *Policy First* scenario in 2030, government institutions worldwide, with active private and civil support, make efforts to resolve many of the issues facing humanity and the environment through top-down, policy-based reforms (UNEP, 2007). While economic growth remains a focal point for global mariculture development, it is acknowledged that such growth cannot be sustained without a stronger consideration of the negative social and environmental impacts that can accompany development. However, in practice, most reform initiatives focus first and foremost on social considerations such as jobs and total production.

Policy reforms for mariculture are led by national governments and international institutions, including the FAO. These lead to improved resource sharing, a better alignment among social and political institutions, and greater political cohesion with international agreements such as FAO's *Code of Conduct for Responsible Fishing*. However, the slow pace of institutional reform and the inflexibility of a more centralized approach to implementing change means that few major reforms to the mariculture industry are widely implemented by 2030 (Lake, 1994; UNEP, 2007).

Security first

In a world where security comes first, the benefits of mariculture production and development are available only to a privileged few (UNEP, 2007). By 2030, to better control and monitor the movement of people, goods, and services within and across their respective borders, governments around the world, with support from powerful private actors, have implemented stronger restrictions on migration and trade. Often these actions are influenced by ongoing political and physical conflicts fed by the socio-political interests of governments and private entities, as well as from the struggle to control increasingly scarce natural resources. As countries around the world adopt increasingly protectionist measures, the human population continues to grow within the confines of national borders.

The internal security focus of many government policies has lead to a reduction in international cooperation and trade by 2030. Both Official Development Assistance for aquaculture extension activities and international trade in seafood are reduced and what remains is strongly conditional on the interests of powerful governments, multinational corporations, and other powerful private interests (UNEP, 2007). There is a growing distrust in the role and effectiveness of the United Nations and their specialized organizations such as the FAO and these institutions are increasingly marginalized. The World Trade Organization (WTO) becomes a leverage tool to gain more political and economic control (Smith, 2006). As has occurred in capture fisheries (Alder and Watson, 2007), countries unable to gain sufficient political and economic autonomy are strong-armed into expanding and intensifying mariculture production for export to economically-developed foreign countries. The revenue from exported sales is brokered by, and primarily returned to, the wallets of government and private actors; the social benefits of mariculture production for poor and rural communities are marginalized.

Sustainability first

In a *Sustainability First* world in 2030, all government, private, and civil sector actors across all institutional levels are following through on their individual and collaborative commitments to address the most pressing social and environmental sustainability issues (UNEP, 2007). In response to a growing social movement over the past 20 years which advocates for a more equitable treatment of social, economic, and environmental issues in development policies, both national and international institutions have collaboratively begun to rework their institutional and trade governance mandates to incorporate more than drivers of economic growth and efficiency. Globally, an increase in jobs and total production are socially valued in the fisheries and aquaculture sectors, but only if the underlying marine ecosystem is maintained and/or restored. This new approach to governance increases the global focus on ecosystem restoration, includes a stronger emphasis on decision-making inputs from the private sector and civil society, and results in significant improvements to general cooperation and compliance in resource use issues worldwide.

Among wealthier major seafood consumers in the USA, Canada, and the EU, there is an increasing growth and diversification in the demand for more responsible, ecologically sustainable, and ethically-produced seafood products. This is a trend carried over and strengthened from previous decades (Lebel *et al.,* 2002; Jansen and Vellema, 2004).

Projected mariculture production under different scenarios

As a complement to the qualitative narrative storylines of possible production and sector futures, quantitative simulations of potential future production were generated to 2030, using past trends in mariculture production extrapolated forward using segmented linear regression. As with all quantitative models developed in the GEO-4 assessment, this new analysis uses historical time series data standardized up to a common base year of 2000 (UNEP, 2007).

If business-as-usual rates of mariculture production continue onwards from the 2004 baseline year (holding all else constant), by 2030 the quantity of farmed marine and brackish water products worldwide could reach 67 million t, of which China might contribute nearly 70%. When this is compared to production simulations under the lowest growth rate scenario (*Sustainability First*) and the highest growth rate scenario (*Markets First*), the total difference in global mariculture tonnage is ± 4.3 million t from the business-as-usual baseline in 2030, with China contributing slightly less in the lowest growth scenario. This implies an annual average increase in production of over 1.5 million t.

Conclusion

As in the original GEO-4 assessment (UNEP, 2007), a number of overarching policy messages can be summarized for mariculture from the exploratory scenario outcomes. Notably, even under the overarching thematic influence of "environment for development", all but one of the development scenarios (*Sustainability First*) continues to prioritize a worldwide expansion, production increase, and intensification of high-value, high-environmental input, carnivorous marine finfish and crustacean species. While market-driven choices are likely to increase total global mariculture production over the next two decades (as well as profits and some jobs), longer-term production growth may ultimately decrease in countries around the world due to rising environmental constraints. With many of the most serious negative ecological and social effects likely to be experienced by developing countries, the perceived benefits of market-driven pathways of action risk translating to only a privileged few people over a short time horizon.

However, if the global human population surpasses a projected 8.3 billion people by 2030, an increase in the pressure on the world's oceans and marine and coastal resources under any scenario is inevitable. Furthermore, a *Sustainability First* approach to mariculture development does not overcome global inequities in the distribution of production and profit (UNEP, 2007), nor will it eliminate the demand for high value carnivorous species for consumption. Global trends are likely to mask more significant changes at the country level, and countries will be favored differently within each scenario. In this regard, a difference in production of a few hundred thousand tonnes and in the availability of certain fish and fish protein could have significant social and environmental ramifications for a given country. For example, while a *Sustainability First* future may increase the total global production by farming more bivalve, which have a small environmental footprint, and contribute to an increase in total global seafood tonnage, the actual availability of meat for consumption could be dramatically reduced because bivalve production is typically reported in shell weight, which may differ from meat weight by a factor of six for some species (Ye, 1999; Wijkstrom, 2003). In addition, the lower comparative economic value of bivalves to finfish and crustaceans could mean that the overall profits derived from mariculture may decline in some countries even though production is increasing. Ultimately, this simulated variation highlights the uncertainty in dealing with the future, as well as the range of effects that individual and collective decisions can have on future global mariculture development.

One need not agree with all the assumptions or elements used in this analysis to derive value from it. The primary focus of such an undertaking is to enable more tangible ways in which to ask 'what if' and highlight the current role and impact of assumptions and choices made by individuals and groups on the future direction of aquaculture development. Under any scenario, and regardless of the balance of social, economic, and environmental considerations addressed, the global increases in both people and in food fish demand stands to further intensify the pressure placed on already heavily exploited coastal and marine resources, as mariculture continues to generate products in response to market demand. The existing social, environmental and regulatory issues of current mariculture production and development, widely discussed in the scientific literature, are currently at odds with existing international policy commitments (FAO, 1995). What is needed to move future mariculture development in a more responsible direction is a clearer vision of the potential options for action before us, as well as their potential consequences. The rewards of doing so are a global seafood industry, countries, people, and an environment with a better resilience and capacity to adapt to the uncertainties of the future.

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Dioxin in the Seas Around Us [16]

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The *Sea Around Us* was named after Rachel Carson's book of the same title, and thus it is fitting that we undertook studies on the effects that pollutants have had on marine ecosystems, in part inspired by Carson's *Silent Spring.* This chapter documents the results of two global models, one on the bioaccumulation and concentration of dioxin in marine organisms as a result of atmospheric deposition, the other on the production, atmospheric transport, and deposition of dioxin.

For a long time it seemed true, especially for marine systems, that, as the dictum goes, "the solution to pollution is dilution". However, biomagnification up the food web and bioaccumulation in long-lived organisms can effectively reverse the effect of dilution. Thus, even the open oceans have now reached a stage where pollutants originating from various societal activities, ranging from mining, manufacturing and agriculture to consuming their products are reaching worrisome levels. Foremost is the thermal pollution and increased acidification of the oceans, both the results of carbon dioxide emission also responsible for global warming, and which have profound effect on ocean life (Cheung and Pauly, this volume, p. 63). Another kind of marine pollution is due to plastic debris, some very small, which now contribute an increasing fraction of what fish (Moore, 2008), marine turtles (Bugoni *et al.*, 2001) and seabirds (Paleczny *et al.*, this volume, p. 83) mistakenly ingest instead of their natural prey.

The *Sea Around Us* used, and further developed, ecosystem modeling tools, notably Ecopath with Ecosim (Colléter *et al.*, this volume, p. 63; Pauly *et al.*, 2000), which allow tracking nutrients as well as pollutants through and up a food web. This chapter briefly reviews the work of the *Sea Around Us* on pollutant tracking, including global models to simulate the oceanic dispersion and uptake of dioxin, whose global scale corresponds to the other products of the *Sea Around Us*. We recall that the word 'dioxin' refers to a group of chemicals that have 17 forms with varying toxicological effects, with the most toxic form 2,3,7,8-tetrachlorodibenzo-*p*-dioxin being classified as a human carcinogen (IARC, 1997).

Bioaccumulation of dioxin

A global model involving spatial data was developed to describe the movement of dioxin up through the marine food web of the global oceans using the Ecospace and Ecotracer routines of the EwE modeling routine was presented by Christensen and Booth (2006). It uses input data

[16] Abbreviated from: Booth, S. W.W.L. Cheung, A.P.C. Wallace, V. Lam[a], Dirk Zeller, V. Christensen and D. Pauly. Nutrients and Pollutants in the Seas Around Us, p. XX *In:* D. Pauly and D. Zeller (eds.) *Global Atlas of Marine Fisheries: Ecosystem Impacts and Analysis*. Island Press, Washington, D.C. [*in press*]

from a preliminary spatial dioxin loading model to oceans as a result of atmospheric depositions (Zeller *et al.*, 2006).

The underlying EwE model includes 42 functional groups and is based on a modified model ('Generic 37') developed for database-driven model construction distributed with the EwE software. The ecosystem is represented by a grid of 2 degree latitude x 2 degree longitude cells and extends from the equator to 70^{o} latitude north and south. The oceans are divided into two depth zones (<200 m, >200 m), and most functional groups were assigned to both zones, except for small demersal fishes, reef fishes, seals, corals and benthic plants, which were assigned to the shallower depth zone. Functional groups were assigned to all 19 FAO statistical areas, and the groups were assigned primarily on estimates of primary production.

The modeling approach involved using predation, catches and an assumed ecotrophic efficiency to estimate the biomass amount of each functional group. Default values for the Generic 37 model were maintained, but density levels were assigned for large sharks (0.1 $t \cdot km^{-2}$), jellyfish (0.1 $t \cdot km^{-2}$), seals/pinnipeds (0.003 $t \cdot km^{-2}$), toothed whales (0.002 $t \cdot km^{-2}$), baleen whales (0.001 $t \cdot km^{-2}$), seabirds (0.001 $t \cdot km^{-2}$), macro- and meiobenthos (1.5 and 2 $t \cdot km^{-2}$, respectively) , corals (1 $t \cdot km^{-2}$), soft corals/sponges (2 $t \cdot km^{-2}$), and benthic plants (10 $t \cdot km^{-2}$). Catch data for each functional group were taken from the *Sea Around Us* database and represent the catch taken in 2000. Spatially explicit data used in the Ecospace routine includes primary production, biomass estimates for zooplankton, macro- and meiobenthos, small and large mesopelagic fishes, and depth information for each cell.

Concentrations of dioxin in marine organisms were primarily taken from the primary literature and represent reported values since 1990 in toxic equivalencies (TEQs). Seventeen congeners of dioxin have been reported as being toxic and data reported as individual congeners were transformed into TEQs using the appropriate toxic equivalency factors (TEFs; Van den Berg *et al.*, 1998). Concentrations in marine mammals were standardized to $ng \cdot kg^{-1}$ lipid weight, and those for other organisms were standardized to $ng \cdot kg^{-1}$ wet weight. Species with reported values were sorted into their respective functional groups, and were placed within their representative region.

Within the oceans, direct uptake rates of dioxin by primary producers and invertebrates are an important pathway for the transfer of pollutants up the food web. However, because of the lack of data concerning uptake rates, we assume that the dioxin is taken up only by phytoplankton once deposited to the ocean and, to prevent the accumulation of dioxin in the oceans, a set decay rate was used for the environment. Thus, concentrations in the biota are a result of uptake of dioxin by phytoplankton and trophic transfer through the food web with no decay in biota. Under these initial conditions, the simulation was run for 22 years, and the results from the model were compared to the reported concentration values.

After running the simulation, most groups reached equilibrium dioxin concentrations excluding the whales (baleen and toothed), seals, and bird groups, which are long-lived. Excluding four outliers, the regression between predicted versus observed dioxin concentrations explains 25% of the variation in the sample values ($p << 0.001$), with a slope value of 0.84. Of the four outliers, two were associated with polar regions, and the other two were associated with coastal areas in Asia. Predicted values for polar regions may be affected by both the duration of the atmospheric model (one year) and non-consideration of re-emission of dioxin from land back to the atmosphere due to the 'grasshopper effect' (Wania and Mackay, 1996). Values for the coastal regions in Asia may also be affected by the input from coastal run-off and riverine outflow. The grasshopper effect and non-atmospheric inputs are not included in the preliminary atmospheric transport and deposition model of dioxin (but they were considered in the deposition models further below).

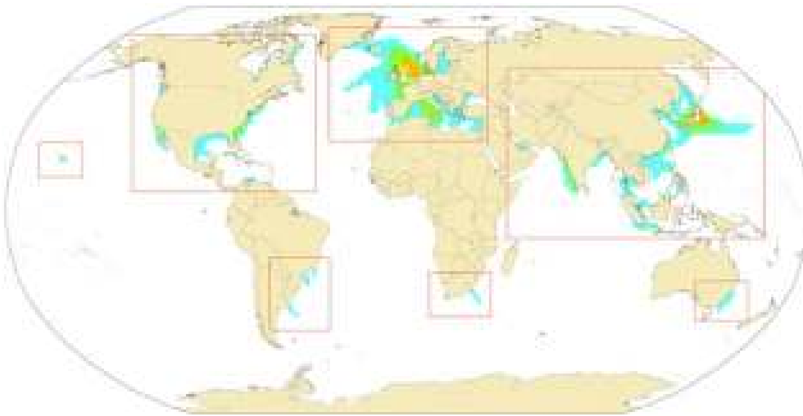

Figure 1. Predicted relative dioxin concentration in small pelagic fishes. Scale is from white to light blue to green, yellow and orange (adapted from Figure 9 in Christensen and Booth, 2006). Note high concentrations in the waters around Asia, Europe and North America, versus lower concentrations in South America, Africa and Australia.

An important outcome of using a global EwE model with the Ecospace routine is that Ecotracer can predict the observed values of dioxin within two orders of magnitude. The predictive power could be improved by having larger sample sizes for under-represented areas. As most concentration values are occurring in coastal areas in developed countries (Figure 1), samples from depths greater than 200 m and from developing areas could lead to a better fit. Improved fits may also be achieved by improving the atmospheric transport and deposition model to include the grasshopper effect, coastal run-off and riverine inputs of dioxin, and these effects were considered in the updated atmospheric model for dioxin (see below).

Global deposition of atmospherically released dioxin

As a follow up to the modeling work reported above, a global model of dioxin was developed that includes production, atmospheric diffusion and dispersion, transport from land to coastal waters, and depositions to land and oceans (Booth *et al.*, 2013). Its purpose was to highlight the deposition of dioxin to marine areas including countries' EEZs and the high seas to improve models such as presented previously on the bioaccumulation of dioxin in marine ecosystems Dioxin concentrations measured in marine organisms result from the input of dioxin to the oceans, but dioxin is not measured in the water column and thus organism concentrations and sediment concentrations have served as a proxy to indicate areas that are more impacted by dioxin than others. Monitoring programs of organism concentrations of dioxin are sparse, and developing countries lack the resources to properly monitor the impacts. Thus, this model, of which only an outline is presented here (but see Booth *et al.*, 2013) identifies areas of potential concern, i.e., where airborne dioxin are likely to be deposited.

A previous global mass balance of dioxin estimated global annual emissions of 13,100 kg +/- 200 kg (Brzuzy and Hites, 1996), which assumed annual depositions to oceans contributed 5%

to the global mass balance. In a preliminary run of our model we found that ocean depositions were approximately 38%, and therefore increase global emissions to 17,226 kg. Mass balance studies have shown that depositions of dioxin are about 10 times greater than reported emissions due to the formation of dioxin from pentachlorophenol, a common wood preservative, by photochemical transformation in the atmosphere (Baker and Hites, 2000). Thus, each emission value was multiplied by 9.7 to account for this discrepancy and the estimated annual emissions were considered to be released in weekly increments.

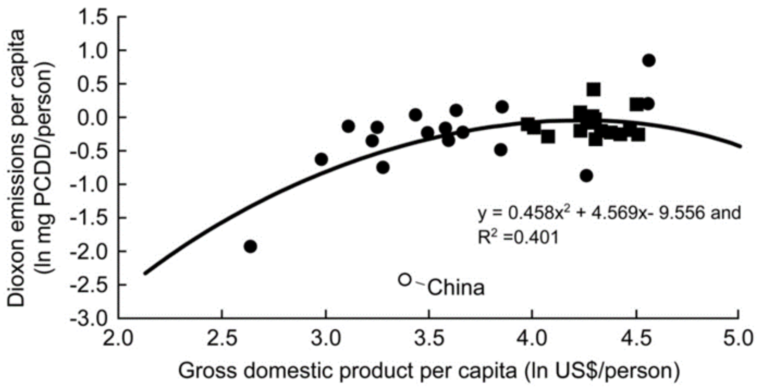

Figure 2. Environmental Kuznets curve used to estimate countries' *per capita* dioxin emissions. Original data used in Baker and Hites (2000) are shown in square symbols (■), new data are shown in circles (●) with China (○) omitted from analysis.

The production of dioxin for 35 countries was based on their reported inventories of annual atmospheric releases with most of these countries having a single estimate between 1995 and 2002 (see Booth *et al.*, 2013). 1998 was chosen as the representative year for the emission inventories, corresponding gross domestic product (GDP) data, and population data. Population data for each of the 35 countries were used to transform GDP and atmospheric releases of dioxin into *per capita* rates. The line of best fit through these points, representing an environmental Kuznets curve, was used to generate the atmospheric dioxin emissions for countries that have not completed a dioxin inventory (Figure 2).

Within each country, we assign dioxin emissions using spatial estimates of GDP. Global spatialized estimates of GDP (Dilley *et al.*, 2005) were mapped to the *Sea Around Us* grid of ½ x ½ degree cells, and dioxin emissions were then made directly proportional to the fraction that each land cell contributed to the country's total GDP *per* land area (i.e., GDP·km^{-2}).

Dioxin dispersion in the atmosphere involved diffusion and the transport of dioxin with wind. Weekly releases of dioxin to the atmosphere were subjected to diffusion, using the diffusion constant for 2,3,7,8-TCDD dioxin of 4.86×10^{-6} m^2·s^{-1} (Chiao *et al.*, 1994), and to global wind patterns. Wind data consisted of global daily means of east-west and north-south wind components from the '40 years Re-analysis Database' of the European Center for Medium Range Weather Forecasting (Anon., 2006). These data were further averaged over the 1991 to 2000 time period into weekly values.

The deposition of dioxin was simulated and the characteristic travel distance approach (CTD), which describes the distance an airborne semivolatile organic pollutant travels before reaching 1/e (i.e., ~37 %) of its initial value was applied (Bennett *et al.*, 1998). The temperature dependent characteristic travel distance, which accounts for the 'grasshopper effect', for 2,3,7,8-TCDD dioxin was used to estimate the amount of dioxin deposited from the atmosphere to land and water (Beyer *et al.*, 2000). Since the distance travelled is dependent on temperature and wind speeds, we use the temperature dependent CTD for 2,3,7,8-TCDD dioxin at 5°, 15°, and 25° Celsius (Klasmeier *et al.*, 2004) to derive a temperature dependent CTD for temperatures greater than 0° C.

The CTD is described by wind speed and the effective decay rate, i.e.,

$$CTD = \mu/k_{eff}$$

where μ is the wind speed ($m \cdot s^{-1}$) and k_{eff} (s^{-1}) is the effective decay rate. The effective decay rate accounts for the transfer of dioxin from air to land and water surfaces, and to biological components (e.g., plants). Since each GIS cell has the wind speed and temperature as an attribute, a temperature dependent CTD is defined, and therefore we re-arrange the equation to solve for the effective decay rate for each GIS cell, i.e.,

$$k_{eff} = CTD/\mu$$

Once deposited to land, water basin transport of dioxin from land to coastal marine areas was also simulated using water run-off amounts as the driver. Data kindly supplied by Dr. C.J. Vörösmarty of the Water Systems Analysis Group at the University of New Hampshire (www.wsag.unh.edu) describe a global total of 6,031 basins with 5,865 basins identified as ultimately exiting to coastal marine waters and 166 were identified as landlocked. To identify coastal cells that receive dioxin via water basin transport, salinity gradient plots were used to determine if a basin's outflow created a freshwater plume in marine waters. For discharge areas that had plumes, dioxin was deposited into the cells that created the plume; for river discharge areas that did not have identifiable plumes, we specified a single central coastal cell to receive the dioxin. For regions between 0° and 65° latitude N or S, we used a salinity threshold of 30 psu to identify freshwater plumes, whereas for regions located above 65° latitude N or S, we used a salinity threshold of 25 psu due to the large inputs of freshwater that remain in surface waters in polar areas. Salinity data were taken from Antonov *et al.* (2006).

Water run-off was shown to be the dominant pathway of dioxin transport in a Japanese watershed (Kanematsu *et al.*, 2009) accounting for over 98% of total dioxin transport. We use a proportionality constant of $0.0004 \cdot year^{-1}$, derived from two studies examining dioxin transport, (Vasquez *et al.*, 2004; Kanematsu *et al.*, 2009) in combination with water run-off data for each water basin (Fekete et al. 2000) to transport dioxin from water basins to coastal marine cells.

We simulated the dispersion and deposition of airborne dioxin by a two dimensional diffusion-advection differential equation:

$$\frac{\partial A}{\partial t} = \frac{\partial}{\partial x}\left(D\frac{\partial A}{\partial x}\right) + \frac{\partial}{\partial y}\left(D\frac{\partial A}{\partial y}\right) - \frac{\partial}{\partial x}(\mu * A) - \frac{\partial}{\partial y}(\nu * A) - \lambda * A$$

where A is the amount of dioxin in a cell, D is the diffusion coefficient (4.86 x 10^{-6} $m^{-2} \cdot s^{-1}$), μ and v are the wind velocity components (in the N-S direction and E-W direction, respectively) and lambda (λ) is the decay rate (i.e., k_{eff}).

To numerically estimate the spatial and temporal amount of airborne dioxin above each cell, a finite difference technique using the alternating direction approach was employed (Sibert and Fournier, 1991). Briefly stated, the alternating direction approach requires that the amount of dioxin above each cell is determined by splitting each time step in half, and determining the amount of dioxin at the end of the first half time step in a grid row by grid row fashion, and then determining the amount of dioxin at the end of the each second half time step by column. Thus, we divided our 30 second time steps into two 15 second time steps and solved for the E-W and then the N-S movement. These two half time step processes were repeated for each time step in a week. In order to maintain mass balance, any losses in the cumulative amount of dioxin in the system at the end of each time step were redistributed based on each cell's proportion of the overall total. The entire computation was then repeated for each week of the year.

At the beginning of each week, each grid cell received its GDP-based share of global dioxin production which was added to the amount that remained airborne at the end of the previous week. A circular boundary condition was also applied at all four edges of the global cell grid. This, in effect, re-connected the cells in the right most column (i.e., 180° E Longitude) to the cells of the same latitudes in the left most column (i.e., 180° W Longitude). The application of circular boundary condition also meant that the cells at the top row (i.e., 90° N Latitude) were re-connected to those on the same row with a longitude difference of 180 degrees. Similar re-connection of cells at the bottom row (i.e., 90° S Latitude) also occurred.

For each cell, the amount of dioxin deposited to the earth's surface within each time step was determined as:

$$\text{Dioxin deposited} = A*(1-\exp[-(\omega/CTD)(ts)])$$

where A is the amount of airborne dioxin above each cell, ω is the wind speed, CTD is the temperature dependent characteristic travel distance, and ts is the time step (30 sec).

The computer simulation of dioxin production suggested several areas of high local production of dioxin due to higher levels of economic activity (Figure 3). These were dominated by eastern North America, Europe, South Asia (particularly the Indian subcontinent), and East Asia (China, Japan and South Korea). Countries belonging to the G20 account for over 80 % of the estimated annual emissions with Japan, the USA, and China accounting for 30% of the annual global emissions. However, it is smaller states such as Singapore and Malta that have the highest emissions on a *per* area basis.

After we ran the model to simulate one year's production, dispersion, deposition and transport of dioxin, approximately 9 kg-TEQ (3%) of the annual dioxin production remained in the atmosphere. The model predicted that most of the annual production of dioxin, 163 kg-TEQ (57%), was deposited to land areas, while ocean waters received approximately 115 kg-TEQ

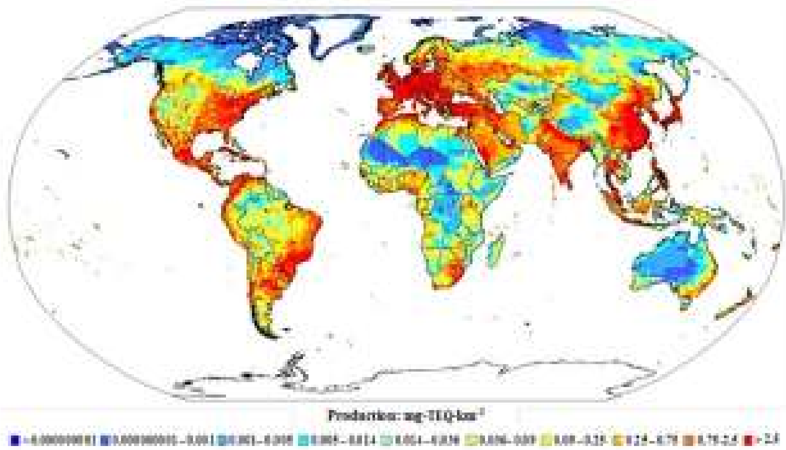

Figure 3. Global production of dioxin as toxic equivalents of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin spatialized over the earth's surface with emissions based on an environmental Kuznets curve.

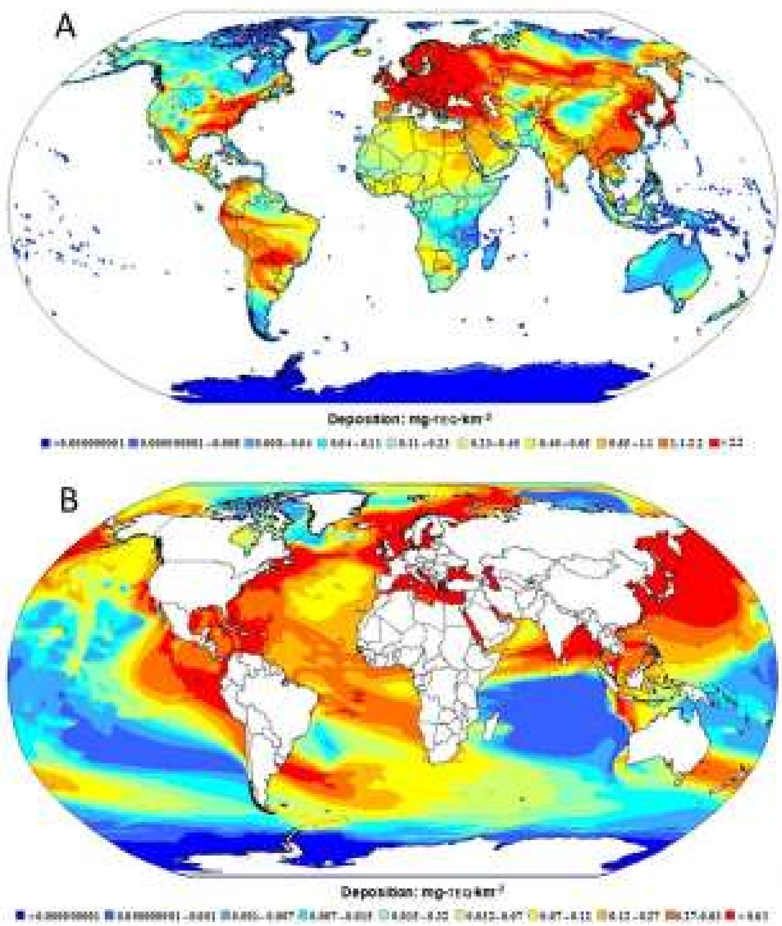

Figure 4. Deposits of dioxin presented as toxic equivalents of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin after simulating one year of transport processes of global atmospheric emissions, to A) land; and B) oceans.

(40%). Large parts of North America, most of central, northern and Eastern Europe, as well as much of the Indian sub-continent and East Asia have high terrestrial depositions of dioxins (Figure 3). Dioxin depositions to land range from $1x10^{-8}$ to 146 mg-TEQ·km^{-2} with the lower values in the Antarctic, and the highest values found in Europe and South Korea.

The model also suggested that ocean areas near the source emission areas also received relatively high dioxin loads. These include the northeast and northwest Atlantic, Caribbean, Mediterranean, northern Indian Ocean, and large parts of the north-western Pacific and South China Seas (Figure 4). However, several areas of relatively low concentration of dioxins were also identified, specifically parts of the west coast of South America and northern parts of the west coast of North America. Marine deposited dioxin ranged from $1x10^{-8}$ mg-TEQ·km^{-2} to 33.5 mg-TEQ·km^{-2} and were similar to terrestrial deposits in that lower values were associated with the Antarctic, but the highest values were found in waters off Japan and South Korea. High dioxin depositions were also found in the marine waters of countries around Baltic and Mediterranean Seas.

Dioxin deposited to the oceans results from the production of dioxin on land, and thus the oceans can act as a sink for dioxin. The High Seas receive the largest amount of dioxin (~36 kg-TEQ) as modeled here and, once standardized by area, depositions to the High Seas are approximately 0.16 mg-TEQ·km^{-2}. The most impacted countries when comparing the ratio of deposits to emissions were found in Africa and Asia. Of the top 20 impacted countries, 11 are located in Africa and 6 in Asia. The 11 African countries' *per capita* GDP average less than US$250·person^{-1}·year^{-1}, and the Asian countries average less than US$450·person^{-1}·year^{-1}.

This work provides the opportunity to examine the impacts that dioxin has on marine ecosystems. Coastal shelves provide most of the fish destined for human consumption and some coastal ecosystems (e.g., eastern North America, China, and Europe) receive much larger dioxin loads than other marine areas (e.g., most of South America and Australia). Past research has shown that dioxin levels in fish oils derived from forage fish around Europe and eastern North America have higher concentrations than those sourced from Peru (New and Wijkström, 2002; Hites *et al.,* 2004). We would expect a similar relationship for all ecosystem components, with higher concentrations found in places that receive higher inputs of dioxin.

Our model suggests that the oceans are more impacted by dioxin than previously thought. Previously, it was assumed that the ocean only received approximately 5% of the global annual production of dioxin (Baker and Hites, 2000). Here, we have shown that, using spatial and temporal distributions of dioxin emissions in a kinematic model, the oceans receive approximately 40% of the annual deposits. Although much of this is confined to the coastal areas, the impacts on the high seas are not negligible and have consequences for food security. One concern is that dioxin is more likely to partition to plastic particles that are eaten by marine plankton and fishes, an entry point for accumulation (Rios *et al.,* 2010). Thus, human populations with high seafood consumption levels may be exposed to higher levels of dioxin than previously thought.

This model does not account for direct (i.e., non-atmospheric) releases of dioxin to land or water, and this contributes to the lack of mass balance between emissions and depositions for dioxin.. Polar regions receive little dioxin over the model simulation time of one year, but these areas may be impacted by the accumulation of this toxin over longer time periods. Simulations that were run for a longer duration would show higher levels of dioxin in polar regions as dioxin migrated polewards as a result of the grasshopper effect.

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The view past peak catches

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Following WWII, the United Nations (UN) and their technical organizations, notably the Food and Agriculture Organization (FAO), began a major project of ”quantifying the world” (Ward, 2004) to give decision makers in national and international agencies the data upon which they could base their policies and compare their outcomes. This project was very successful and the UN system is now the largest provider of data on the world’s economy, before the WTO, OECD or EU. Unfortunately, some of these quantitative data are wrong (Jerven, 2013), particularly when assembled from national reports. Thus, for example, reports of member countries to FAO about the state of their forests, when aggregated at global level, suggested that the annual rate of forest loss was nearly halved between 1990/2000 and 2000/2005, while the actual loss rate doubled, as estimate by remote sensing and rigorous sampling (Lindquist *et al.*, 2012, p. 23). Here we show, similarly, that the main trend of the world marine fisheries catches is not one of “stability” as suggested by FAO (2014), but one of decline. Moreover, this decline, which began in the early 1980s and is now accelerating, started from a higher peak catch than suggested by the aggregate statistics supplied by FAO members, implying that we have more to lose if we let this decline continue, but also that there is more to be gained by rebuilding stocks.

‘Wrong’ statistics do not refer to the imprecision that mars our ability to measure or estimate certain quantities and which may variably result in under- or overestimates. This sort of uncertainty, while regrettable, can be remedied straightforwardly at the local level, and is largely overcome when aggregating national statistics at the regional or global level. Rather, the wrong data that we refer to are based on a systematic bias, such that they grossly misrepresent trends occurring on the ground.

In the case of marine fisheries catch data, the most common source of bias in the official statistics of coastal countries is to simply ignore in data collection those fisheries perceived *a priori* to be unimportant or trivial in catch volumes (e.g., subsistence and recreational fisheries), or whose catch is difficult to estimate (illegal fisheries, artisanal fisheries in remote areas). This result, *ipso facto* in potentially available uncertain estimates for these fisheries being effectively replaced by statistically very precise (but incorrect) values of zero. These default procedures, mostly applied without reflections on their implications, are sometimes justified by statements to the effect that “there are no data”. However, fishing is a social activity embedded in the local, regional and global economy, whose pursuit will invariably ‘throw a shadow’ on the surrounding economy, i.e., fishing impacts its other sectors through the input it requires (fuel, net material, space on beaches, etc.) and the output it generates (seafood that is locally consumed or exported, jobs, conflicts etc.).

For such cases of poorly monitored or unmonitored fisheries, we suggest the statistic be ‘reconstructed’ (see Zeller and Pauly, this volume, p. 15) to provide the best estimates of catches for these fisheies. Our insights were gained by adding up the estimates of ‘reconstructions’, all covering the years 1950 to 2010 and conducted over the span of the last 10 years, of the marine catches made by all fisheries in 267 Exclusive Economic Zones (EEZ) and sub-zones pertaining to all maritime countries and their overseas territories (see pp. 113, for examples), plus the fisheries catches from the global fisheries for large pelagics (see Le Manach *et al.*, this volume, p. 25).

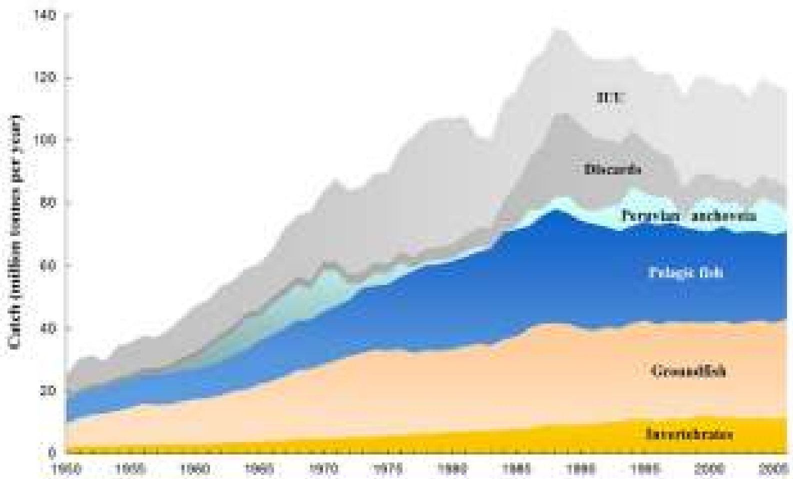

Figure 1. Global fish landings 1950–2006 as approximated by Pauly *et al.* (2002) and updated to 2006 based on Zeller and Pauly (2005). Figures for invertebrates, groundfish, pelagic fish and Peruvian anchoveta are from FAO catch statistics, with adjustment for over-reporting from China (Watson and Pauly, 2001), i.e., are not yet based on reconstructed catches. The resulting estimates for unreported (i.e., 'IUU') catches were tentative in Pauly *et al.* (2002), while the preliminary estimates of discards were based on Zeller and Pauly (2005). Pauly *et al.* (2002) noted that "complementing landings statistics with more reliable estimates of discards and IUU is crucial for a transition to ecosystem-based management". The global catch reconstructions undertaken over the last decade by the *Sea Around Us* provide such estimates.

The total catch that will result from the combined catch reconstructions will likely resemble a pattern similar to that presented in Figure 1, which was inspired by Pauly *et al.* (2002; Figure 1 therein) and modified in Zeller and Pauly (2005; Figure 2 therein), but will be based on the completed catch reconstructions of 267 EEZs and a global large pelagics dataset. The final global dataset will, overall, be considerably higher than the data reported by the FAO on behalf of its member countries and the Regional Fisheries Management Organizations (RFMOs) in charge of fisheries in the high seas (Cullis-Suzuki and Pauly, 2010).

The discrepancy between officially reported data and reconstructed total catches will be mainly due to:

- The non-inclusion of discarded fish in the fisheries statistics of national fisheries agencies and RFMO (except CCAMLR), despite assertions by most countries in the world for ecosystem-based fisheries management (Pikitch *et al.*, 2004);
- The non-inclusion of the catch of subsistence and recreational fishing by nearly all fisheries agencies (Zeller *et al.*, 2011a, 2011b, 2014); and
- The non-inclusion of estimates of illegal catches (e.g., Belhabib *et al.*, 2014), although methods exist for including such estimates in fisheries stock assessments (Zeller *et al.*, 2011b).

The reconstructed catch data also will differ from the total catches presented by FAO in the Fishstat database and in successive *State of the World Fisheries and Aquaculture* reports (e.g.,

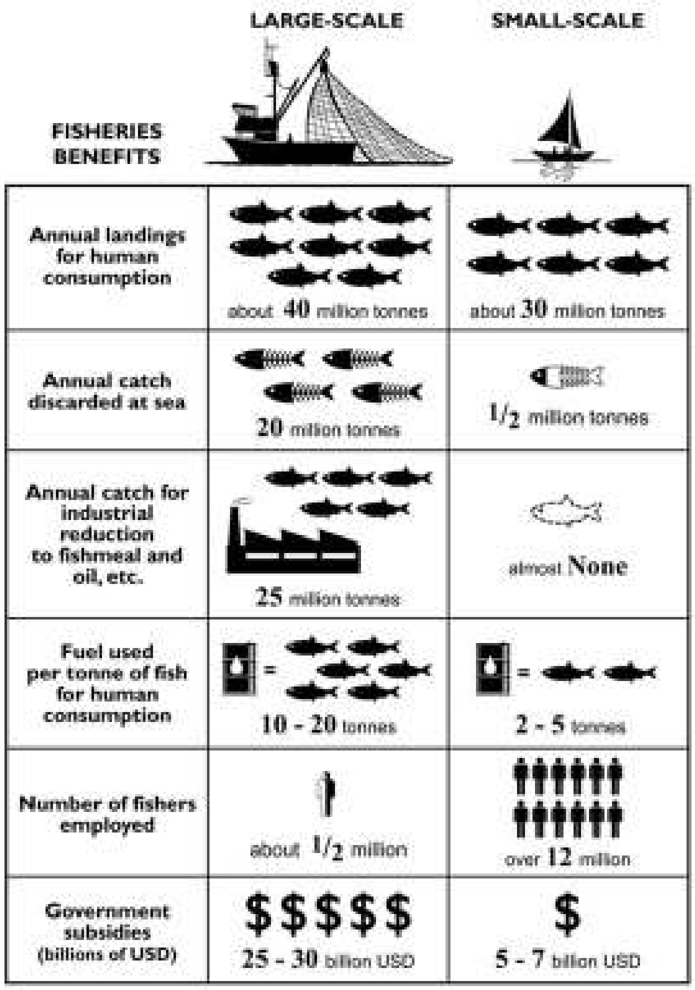

Figure 2. Updated Thompson graph (Thompson, 1988) for the comparison of the world's small-scale (i.e., artisanal and subsistence) and large-scale (industrial) fisheries from 2000 to 2010. Note that the small-scale fisheries perform better on social (jobs), sustainability (fuel consumption) and food security (ratio of landings for human consumption to catches including discards and fish used as animal feeds) criteria, but receive far less government subsidies.

FAO, 2014) in that the *Sea Around Us* catch data are disaggregated by fisheries sectors (i.e., industrial, artisanal, subsistence and recreational). This will also allow comparisons via a Thomspon graph (Thompson, 1988; see Figure 2), which will update and correct earlier comparisons of this sort (Pauly, 2006), and make even clearer the stark mismatch between the economic and social benefits of small-scale fisheries and the government support (i.e., subsidies) they receive (Jacquet and Pauly, 2008).

Indeed, this mismatch may be one of the clearest documentation of policy failures that bad data can lead to. Creating a database presenting corrected datasets is a lot of work, but it is possible. We must do this work if we are to rebuild our fisheries (Sumaila *et al.*, 2012) and, in the process, direct more support to the small-scale sector, which although neglected (Pauly, 2006), contribute directly to the food security in rural areas (Zeller *et al.*, 2014), provide employment to millions throughout the world (Teh and Sumaila, 2013), and may reduce the climate change burden per tonne of fish caught compared to large-scale industrial fisheries, as shown in Figure 2.

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COUNTRY/TERRITORY SUMMARIES

Below, we present 5 examples of EEZ-specific catch reconstruction summaries, in the form of the micro-chapters presented for all 267 EEZs of countries and territories in the '*Global Atlas of Marine Fisheries: Ecosystem Impacts and Analysis*' being published by Island Press.

These micro-chapters present a map of the Exclusive Economic Zone (EEZ) for each country or other territorial entity covered, a brief vignette about the development of its fisheries, and bivariate graphs with time series (1950-2010) of (i) reconstructed domestic catches by sector and where appropriate, foreign catch by country, as well as (ii) the catch by species and species groups (i.e., taxonomically disaggregrated).

The graphs presenting the domestic reconstructions quantify uncertainty for three periods (1950-1969; 1970-1989 and 1990-2010) according to the methods described in Zeller and Pauly (this volume, p. 15), and where available, as a red line, the official catch, either national, or as the statisics compiled from member country reports by the Food and Agriculture Organization of the UN (FAO). Foreign catches are shown as 'negatives' because in most EEZs, they do not complement domestic catches, but largely compete with them, as can be seen in Subarctic Alaska (p. 114), and especially in West Africa, here represented by Mauritania (p. 116).

USA (Alaska, Subarctic) [17]

Beau Doherty, Darah Gibson and Yunlei Zhai

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Alaska, the largest U.S. state, also has an immense EEZ (Figure 1). Doherty *et al.* (2014) reconstructed the U.S. catch for Alaska within its EEZ or equivalent waters from 1950 to 2010 using commercial landings data from the National Marine Fisheries Service (NMFS) and Queirolo *et al.* (1995) as a reporting baseline. Additional sources of catch in the form of recreational, subsistence, discards and joint venture catches were compiled from historical data from the National Oceanic and Atmospheric Administration, the Alaska Department of Fish and Game, and the International Pacific Halibut Commission. Domestic catches from 1950-1975 averaged around 200,000 t·year^{-1} (Figure 2A), over half of which were Pacific salmon (Figure 2B).

Figure 1 Subarctic Alaska (USA) has an EEZ of about 3.2 million km^2, encompassing a shelf of 1 million km^2, within FAO Area 67 and 61.

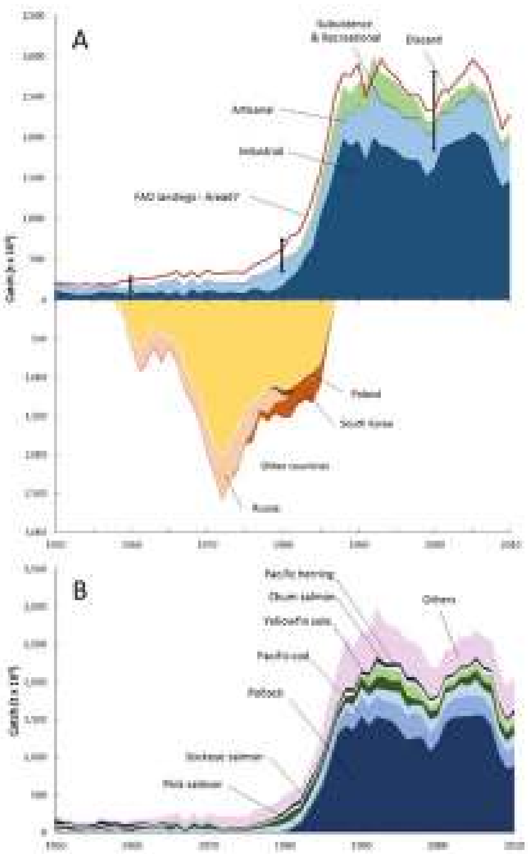

Figure 2. Total reconstructed catch within the Alaskan EEZ from 1950-2010. A) top showing domestic catches by sector and its uncertainty (the invisible subsistence and recreational catches make up 0.7 % of the total) and bottom showing foreign catches by country; and B) domestic catches by species and species groups.

Catches increased sharply in the late 1970s and 1980s, coinciding with the establishment of joint venture fisheries for groundfish between foreign processing ships and domestic fishing vessels (Queirolo *et al.* 1995). Since 1985, catches average 2.4 million t·year^{-1} (Figure 2A), peaking at nearly 3 million tonnes in 1992, with Alaska pollock (*Theragra chalcogramma*) accounting for 43-63% of annual catch (Figure 2B).

Overall, reconstructed catches are 1.1 times the commercial landings baseline, a discrepancy mostly caused by joint venture catch and discards from trawlers. Catch and related data were available, transparent and detailed, providing a sound basis for this reconstruction and, more importantly, for the prudent management of Alaska's fisheries, which should serve as a model elsewhere.

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Bahamas [18]

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[b] Earth to Ocean Research Group, Department of Biological Sciences, Simon Fraser University, Burnaby, B.C., Canada
[c] *Sea Around Us*, Fisheries Centre, University of British Columbia, Vancouver, Canada

The Commonwealth of the Bahamas is an archipelago of more than 3,000 low-lying islands, cays and rocks located east of Florida, USA (Figure 1). Tourism is the primary industry, and since the 1970s, the total number of visitor arrivals per year has outnumbered the resident population by an order of magnitude. Both tourists and residents expect to catch and eat local fish. This account, based on a reconstruction by Smith and Zeller (2013), presents a comprehensive accounting of Bahamian fisheries catches, which ranged from about 2,000 t·year^{-1} in the 1950s to a peak of over 20,000 t in 1985, and 8,000 to 10,000 t·year^{-1} in the late 2000s. Over the entire 1950 to 2010 period, the reconstructed catches were 2.6 times the landings reported by FAO on behalf of The Bahamas, mainly because of failure of the latter to include the catches from the recreational and subsistence fisheries (Figure 2A). In particular, the former sector, also described in Deleveaux and Higgs (1995), contributed 55% of total reconstructed catch - yet recreational catches remain unreported. Also, tourists consume much larger quantities of local fish than the reported catch data would allow, and substantial amounts of unreported catches end up in hotel restaurants. These results provide a novel baseline for historic fisheries catches and their composition (Figure 2B), which should be revised as better data become available.

Figure 1 The Bahamas has a land area of just under 14,000 km^2 and an EEZ of over 629,000 km^2.

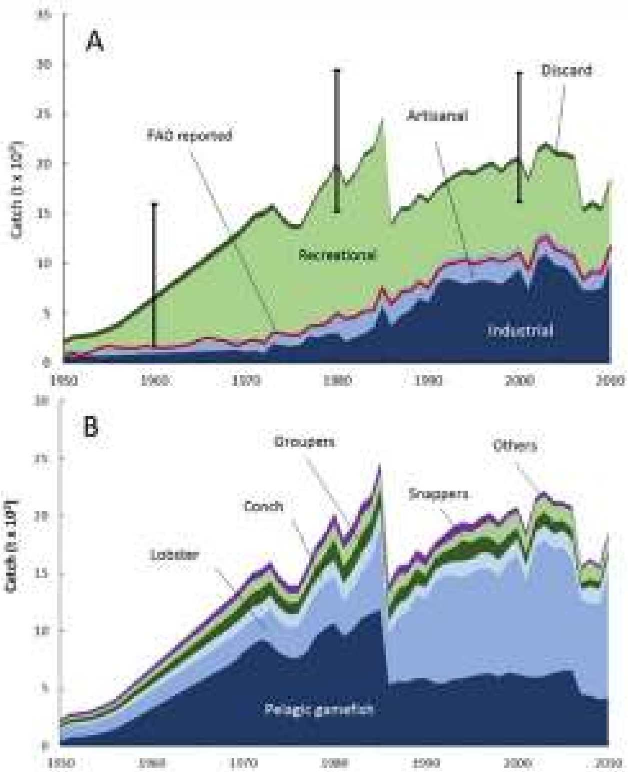

Figure 2. Total reconstructed catch within The Bahamas EEZ from 1950-2010. A) by sector (with uncertainty); and B) by species and species groups.

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Mauritania [19]

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Mauritania, which claims the largest marine protected area in West Africa, the Parc National du Banc d'Arguin (PNBA; Figure 1), has very productive fisheries due to a large continental shelf and a strong seasonal upwelling. The domestic catch was around 20,000 t·year^{-1} in the 1950s, increased rapidly to 280,000 t·year^{-1} in the late 1990s, and was around 480,000 t·year^{-1} in the 2000s, despite increasing fishing effort (Belhabib *et al.*, 2012; Figure 2A). Overall, this is about twice the landings reported by FAO on behalf of Mauritania. About 60% of this domestic catch (and 40% of the discards) is generated by Mauritanian flagged Chinese industrial vessels. Foreign fishing was responsible for the bulk of total catches, with over 60 million t of withdrawals over the 1950-2010 period, of which 38% were extracted by Eastern European pelagic trawlers, 27% by Chinese and 20% by European Union vessels, with China increasing its take in recent years (Figure 2A).

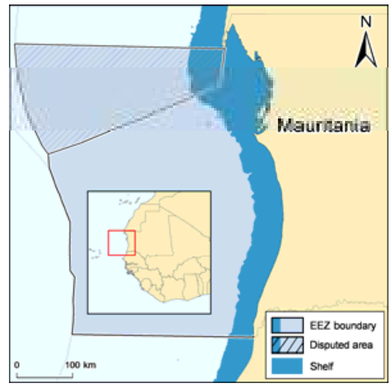

Figure 1 The Mauritanisn EEZ of around 155,000 km^2 and shelf area of 28,000 km^2.

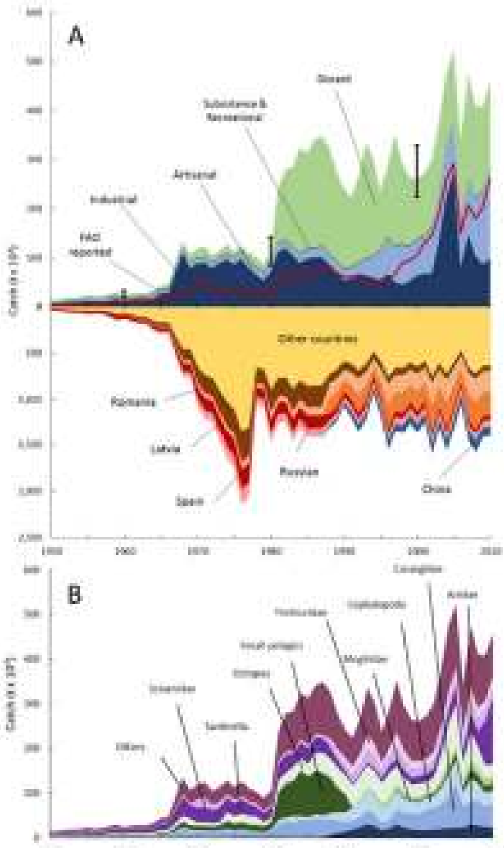

Figure 2. Total reconstructed catch within the Mauritanian EEZ from 1950-2010. A) top showing domestic catches by sector (and its uncertainty) and bottom showing foreign catches by country (note different scales); and B) domestic catches by species and species groups.

Part of the foreign catch was taken without explicit authorization from the Mauritanian government, i.e., illegally. Catches are dominated by pelagic species (Figure 2B), notably sardinella (*Sardinella aurita*). The only legal fishery inside the PNBA is conducted by the Imraguen people, whose traditional ways and small sailing craft (Picon, 2002) stand in huge contrast to the massive industrial vessels operating outside.

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México (Pacific) [20]

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The Pacific coast of México (Figure 1) supports most of the country's fishing activity, including the largest small pelagic fish stocks and most the valuable shrimp and tuna fisheries. Despite the political clout of these mainly industrial fisheries, the largest contributors to total catches are by far small-scale artisanal fishers, who catch any available species given seasonal and market conditions. Particularly since the 1970s, subsidized fishery expansion and operations have resulted in a large fleet, *de facto* open-access conditions, and poor industry oversight (Cisneros-Montemayor *et al.*, 2013). For the Mexican Pacific, reconstructed catches totaled 400,000 t·year^{-1} in the early 1950s, and ranged between 1.5 and 2 million t·year^{-1} in the late 2000s, which is 1.8 times the landings reported by the FAO on behalf of Mexico (Figure 2A; see also Cisneros-Montemayor *et al.*, 2015). Given the high rates of malnutrition in México, the currently large discards from industrial trawlers and artisanal gillnets are particularly troubling. Though total catch would appear to be increasing, current large industrial catches of small pelagic fishes (50% of total), mainly Pacific sardine (*Sardinops sagax*), have masked the concurrent declines of many species (e.g., benthopelagic fishes) that are likely much more important for fishing communities (Figure 2B). In addition to improving knowledge of human impacts such as presented here, future actions should aim to increase compliance with existing policies, while curtailing capacity expansion.

Figure 1 The EEZ of México in the Pacific covers 2.4 million km^2, of which 174,000 km^2 is shelf.

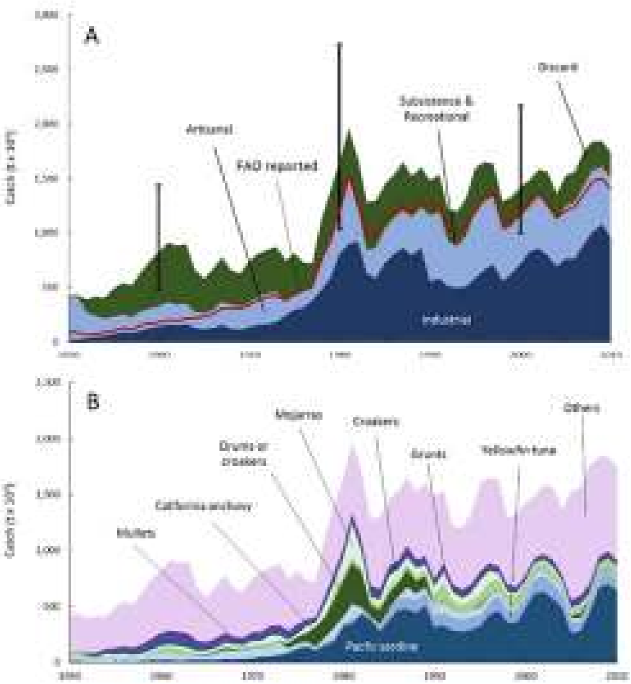

Figure 2 Marine fisheries catches of Mexico in its EEZ (Pacific coast). A): by sector (with uncertainty); and B): by species and species groups.

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Rapa Nui (Easter Island, Chile) [21]

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Easter Island, or Rapa Nui in the Polynesian language of its first inhabitants, is located 3,760 kilometers southwest of mainland Chile and forms, with the tiny, uninhabited island of Salas y Gómez, Chile's 'Easter Island Province' (Figure 1). This reconstruction focuses on Rapa Nui, as Salas y Gómez was never subjected to a sustained fishery, and most of its surrounding area became a marine reserve in 2010. Using fishers' interviews, various Chilean reports (e.g., Yáñez *et al.*, 2007) and limited official data, Zylich *et al.* (2014) reconstructed fisheries catches around Rapa Nui of about 40 t·year^{-1} in the 1950s, which increased toward values fluctuating around 185 t·year^{-1} in the 2000s (Figure 2A). Major targets are Pacific chub, or *Nānue* (*Kyphosus sandwicensis*) and yellowfin tuna, *Kahi ave ave* (*Thunnus albacares*), with spiny lobster *Ura* (*Panulirus pascuensis*) being the most important invertebrate species caught (Figure 2B). Satellite observation and radio communications also suggest that there is an illegal tuna fishery in the EEZ of Easter Island Province, with catches very tentatively estimated at 630 t·year^{-1}, and which may be the reason behind the declining artisanal catch per unit of effort for pelagics experienced by Rapa Nui fishers (pers. comm. to R.V. and D.P). The marine resources of the relatively unproductive waters around Rapa Nui are fragile, including those popular amongst tourists, and accounting for their catches is an important first step toward ensuring their sustainable exploitation.

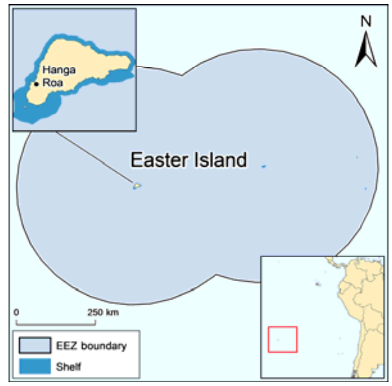

Figure 1 Easter Island Province, Chile (land area 164 km^2) and its EEZ (720,000 km^2).

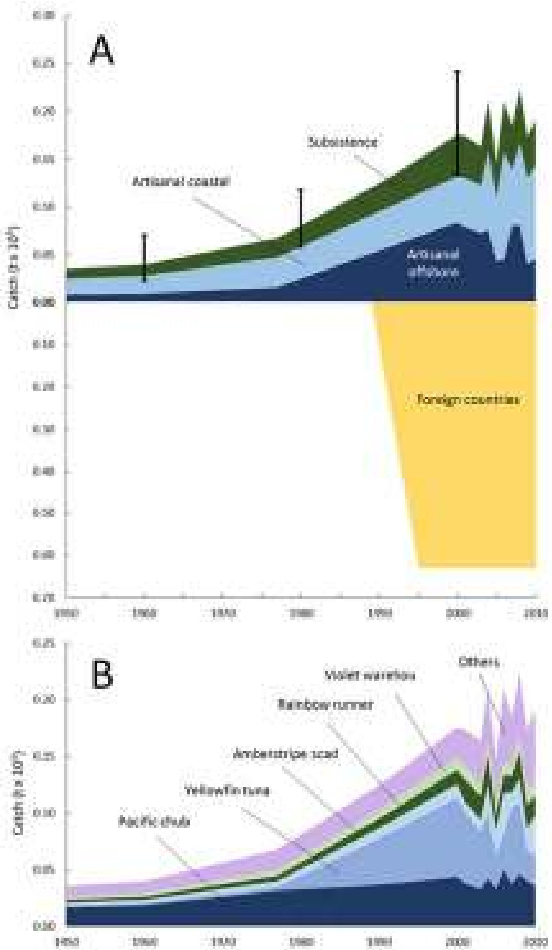

Figure 2. Estimated catch of Easter Island Province, Chile. A) top showing domestic catches by sector (and its uncertainty) and bottom showing catches by foreign country; and B) domestic catches by species and species groups.

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LIST OF *SEA AROUND US* PUBLICATIONS (1999-2014)

Below we list the over 700 scientific and other publications by members of the *Sea Around Us* over the 15 year time span covered here (1999-2014). While primary focus here is on our approximately 300 articles in peer-reviewed scientific journals (see p. 121 below), over 250 chapters in books and reports (see ‘Chapters’ p. 135), over 120 books and reports (see ‘Books/reports’ p. 151) and the over 50 contributions in the more recently created Fisheries Centre Working Papers Series (see ‘Working papers’ p. 159), we also present selected publication opportunities in other outlets in this list (see ‘Others’ p. 163).

Furthermore, we have regularly published the *Sea Around Us Newsletter*, of which one issue was published in 1999, followed by 81 issues in the following years (i.e., 5–6 issues per year). The newsletters were not only distributed through our electronic mailing list, but are also available (and searchable) on our website (www.seaaroundus.org).

Last, but definitely not least, we have also published numerous distinct *Sea Around Us* reports, including, but not limited to 5- and 10-year retrospectives of the *Sea Around Us*, as well as several directed contributions to activies of The Pew Charitable Trusts, such as reports on the biodiversity and fisheries catches for several Global Ocean Legacy sites.

Articles in peer-reviewed journals

2014

Al-Abdulrazzak D and Pauly D (2014) Ground-truthing the ground-truth: reply to Garibaldi et al.'s comment on "Managing fisheries from space: Google Earth improves estimates of distant fish catches". ICES Journal of Marine Science 71(4): 5.

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'***So long, and thanks for all the fish***'

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