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On the Multiple Uses of Forage Fish: From Ecosystems to Markets

A report to the Pew Institute for Ocean Science, University of Miami, Rosenstiel School of Marine & Atmospheric Science, Miami, FL

Fisheries Centre, University of British Columbia, Canada

On the Multiple Uses of Forage Fish: From Ecosystems to Markets

edited by
Jackie Alder and Daniel Pauly

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A Research Report from the Sea Around Us Project, Fisheries Centre, UBC, to the
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DIRECTOR'S FOREWORD

Many believe 'forage' fish to consist of species suitable only for reduction to fishmeal and fish oil, and that they are abundant, and therefore can be fished without regard to other components of the ecosystem, including marine mammals and seabirds. This document highlights that the term 'forage' fish is, mostly, another name for the sardine, anchovies, mackerels and other fish which are traditionally consumed in various cultures. Indeed, they represent a crucial component to the food security for many poor people, especially in developing countries. These fish, moreover, play an important role in marine ecosystems, as they mediate the transfers between plankton and the predators on top of marine food webs. In the process, they also transfer persistent organic pollutants (POP), for example dioxin, up the food web. Finally, they are key to the development of the aquaculture sector, and the continued affordability of intensively produced animal protein, whether tainted with POP or not.

This is the first study that integrates and presents a comprehensive, global view of forage fisheries, specifically: trends in catches since the 1950s; how landings are used by humans including consumption, trade and use in food production; the interaction between forage fisheries and humans and marine animals; and the link to a POP (dioxin) deposition into the ocean. This report is also the first to estimate the consumption of fishmeal in the aquaculture sectors of various countries since 1975, when global feed databases were initiated.

The catch trends of forage fisheries were analysed, highlighting that, in Europe, fish targeted for reduction change, with new species replacing those that are fished out, while in Latin America and the United States, the stocks are considered to be fully exploited. Also, there is little scope for finding new stocks to increase fishmeal and fish oil production. Moreover, while the industrial fleets of the world are 'fishing down marine food webs', the aquaculture sector, which increasingly feeds fishmeal even to herbivorous species, is 'farming up marine food webs', with the average trophic level of fish used in the feed itself increasing.

Small and medium pelagic fishes–i.e., forage fish–are a source of relatively cheap protein for several African nations, and thus, when the fisheries in question decline, the food security of people is at risk. Fishmeal and fish oil are key ingredients in feeds for aquaculture, and will in part determine the level of expansion of carnivorous aquaculture until alternatives are found. Expansion of aquaculture will be at the expense of intensively produced poultry, and will ultimately impact consumers in the market place.

Forage fish are also important as prey for marine mammals and seabirds. This document dispels the argument that marine mammals compete with commercial forage fisheries in more than a few areas of the world; also, the demonstration is made that there is virtually no overlap between seabirds and commercial fisheries. On the other hand, there are indications, albeit limited, that marine mammals and seabirds are impacted by commercial fisheries.

This is the first study to model the atmospheric deposition of dioxin in marine systems, and then simulate its uptake by various organisms. When combined with information on where forage fish are caught, processed and traded, this leads to a picture of the global human food systems in which POP can appear at any location and in any guise—a worrying thought.

This report is a joint effort by many members of the *Sea Around Us* Project, who have authored its various chapters. This report thus demonstrates the interdisciplinary nature of the *Sea Around Us* Project and its ability to synthesize information from a wide variety of sources into a coherent whole, the story of the world's largest fishery. I take this opportunity to thank Brooke Campbell for her assistance in producing the graphs and layout of this report.

Daniel Pauly,

August 2006

EXECUTIVE SUMMARY

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Forage fish are often described as the prey for other animals to forage on, and are composed primarily of small and some medium-sized pelagic fish. Forage fish are used directly for human food and reduced to fishmeal and fish oil for industrial purposes. These fishes tend to form large dense schools, which make them easy to catch using little fuel energy, especially in comparison with demersal fish, typically caught by bottom trawling. Small pelagics play a crucial role in most ecosystems because they are the group that transfers energy from the plankton to the larger fishes and marine mammals. The direct dependence of these relatively short-lived fishes on plankton, itself impacted by environmental fluctuations, often causes the biomass of these fishes to fluctuate.

This has led many fisheries scientists to conclude that fisheries have little impact on small pelagics, as their abundance seems determined mainly by environmental factors. Presently their catch is about 32 million tonnes per year, a staggering 37% of global marine landings.

Most of this catch is used to produce fishmeal and fish oil for use in both agriculture and aquaculture. The aquaculture industry is increasing, especially the farming of carnivorous species, and requires increasing supplies of fishmeal and fish oil, met in part by an increase of the fraction of global fishmeal supply being diverted away from agriculture and, for some species, the human food supply, and by increasing the pressure on small pelagics, including species that were previously unexploited.

The intense pressure on small pelagics has a number of consequences, notably a depletion of the food base of marine mammals and seabirds. Indeed this effect is so strong that it has become, in many parts of the world, a cause for the decline of seabird and marine mammal populations, e.g., as early as 1965 for seabirds in Peru, and currently for marine mammals in the Mediterranean.

The growing concern over seafood quality for wild capture fish and indirectly, as inputs into intensive animal production, has prompted policy makers to look at levels of pollutants (e.g., dioxins) in farmed fish such as salmon as well as the levels of pollutants in fishmeal and fish oil.

This report presents the results of various analyses of the uses of forage fishes by marine mammals and seabirds as well as direct and indirect use by humans. The analyses include investigation of long-term trends (1950 to the present), and include numerous displays based on the modeling and mapping approaches developed in the last five years by the *Sea Around Us* Project to illustrate where forage fish are subjected to pollutants, caught, traded, and used as human food or input to intensive animal production including aquaculture.

We hope that the reader will appreciate that the observed trends cannot be reliably projected into the future, which must forever remain uncertain. What we can do–and have done—is to propose alternative scenarios, and to make their assumptions explicit. This provides us with a tool to evaluate existing plans, e.g., for large-scale expansion of those forms of aquaculture requiring massive input of fishmeal and/or oil.

Forage fish are concentrated in a few regions of the world, but traded globally as human food as well as input into intensive animal production system. They are a factor in the current pricing of pork, poultry and farmed fish. Forage fish play an important role in the food security of many countries, but this role is under threat as the demand for forage fish plays an increasingly important role in feeding an expanding aquaculture sector. How forage fish will be affected by global climate change remains uncertain. However, with both El Niño events and ecosystem-wide collapses, such as occurred off Namibia, becoming more

frequent, declines in forage fish supplies are possible, with dire consequences for marine mammals and seabird populations, and economic effects that would reverberate across the globe.

The process of concentrating the protein and essential oils of forage fish into fishmeal and fish oils also concentrates pollutants such as dioxins in these products, which are also traded globally. These pollutants continue concentrating up the food chain as fishmeal and fish oil are used as feed in the production of animal protein. The importance of forage fish in the food security of many coastal communities, their global importance in sustaining the aquaculture sector, and their potential to transport pollutants are sufficient reason to further our understanding of these fisheries, as attempted in this report.

Jackie Alder, *Senior Research Fellow, Sea Around Us* Project, *UBC Fisheries Centre*
Daniel Pauly, *Principal Investigator*, *Sea Around Us* Project, *and Director, UBC Fisheries Centre*

CHAPTER 1

FISHERIES FOR FORAGE FISH, 1950 TO THE PRESENT [1]

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ABSTRACT

Following a brief historical review of the emergence of fisheries for forage fish that are primarily destined for reduction, and their competition with fisheries for human consumption, an account is given of landing trends in various parts of the world, and catch maps are provided for the 1970s and 2000s which allow spatial and temporal comparisons. A brief account is also given of the changing species composition of the landings, the exploitation status of the fisheries, the trophic levels trends of species destined for reduction, the fuel consumption of the global fleet exploiting forage fish which are primarily small pelagics, the fishing gear they use, and the ex-vessel prices they fetch.

The discussion, finally, attempts to amalgamate this material, which is further discussed in the other chapters in this report.

INTRODUCTION

Historically, all fish that could be caught, including small pelagic fish, were used as a source of food for humans (see Chapter 2), and the reduction of fish to fishmeal and fish oil for indirect use is relatively recent. Seasonally abundant catches of herring and sardines, which could not be absorbed by local markets, started the fish oil industry in northern Europe and North America at the beginning of the 19th century (Huntington *et al.*, 2004). The oil was used for lubrication of machinery and leather tanning, soap production and other non-food products, and the by-products of fish oil production were used as fertilizer.

In the early 20th century the production of fishmeal for animal feed began in Northern Europe, based on Herring (*Clupea harengus*), and in North America, based on Atlantic menhaden (*Brevoortia tyrannus*) in Chesapeake Bay and South American pilchard (*Sardinops sagax*) in California. Once the benefits of fishmeal as an inexpensive feed supplement for animal production were realized and demand increased, the fisheries began to deliberately target fish for reduction to fishmeal, with fish oil more as a by-product. In the early 1950s, a huge reduction fishery for Peruvian anchoveta (*Engraulis ringens*) developed in Peru, then in Chile, which at first complemented, then replaced, the earlier indirect exploitation of this fish, in the form of guano produced by fish-eating birds (Muck and Pauly, 1987; Muck, 1989).

In California, the benefits of fishmeal in the animal feed sector were quickly realized and demand for fishmeal with corresponding demands for increased landings raised concern over food supplies and sustainability of the industry (Radovich, 1981). The California legislature responded in the early 1920s with the introduction of legislation prohibiting the processing of fish for reduction if it was fit for human consumption. The controversial issue of competition between human and industrial consumption for raw material such as the 'California sardine' (which led to similar legislation in other areas and times) became moot when, due to excess fishing and the 'changes in environmental conditions' that are always evoked in such cases (see e.g., Radovich, 1981), catches peaked in the 1930s, and collapsed in the late 1940s, and the ghost of this fishery, through the works of John Steinbeck and Ed 'Doc' Ricketts, entered the realm of legend (Tamm, 2004).

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[1] Cite as: Watson, R., Alder, J., Pauly, D. 2006. Fisheries for forage fish, 1950 to the present, p. 1-20. *In*: Alder, J., Pauly, D. (eds.) On the multiple uses forage fish: from ecosystems to markets. Fisheries Centre Research Reports 14(3). Fisheries Centre, University of British Columbia [ISSN 1198-6727].

The same scenario was replayed a few decades later, off Peru, where the annual catch of Peruvian anchoveta grew to 17 million tonnes (t) in 1970 (Castillo and Mendo, 1987), about 6 million t higher than the official catch of 12 million t–itself higher than recommended by experts at the time (Gulland, 1968; Murphy, 1967; Schaefer, 1967). The fishery collapsed in 1972/73, following an El Niño event that was subsequently seen by many as solely responsible for the collapse.

As earlier in California, the Peruvian reduction fishery was seriously contested by those who felt that Peruvian anchoveta should somehow be processed for human food, e.g., in the form of fish protein concentrate (FPC) that could be used to fortify flours, an obvious product in a country with an animal protein deficiency in its highlands. Moreover, not only juvenile South American pilchard (*Sardinops sagax*) and Horse mackerel (*Trachurus murphyi*), which frequently occur in anchoveta schools (Bakun and Cury, 1999), were caught by the anchoveta reduction fishery, but also pure schools of full-sized *S. sagax* sardine and *T. murphyi*, adding to the controversy. Landing *S. sagax* for reduction has long been prohibited in Peru, and recently, regulations were announced which also limit the catch of *T. murphyi* to vessels fishing the stocks for human consumption, and not fishmeal (Fishing Information and Service, 2004).

This, however, is not the main research area for scientists working on the small pelagic fishes which support the most important reduction fisheries. Rather, it is their extraordinary responsiveness to environmental fluctuations, and their apparent resilience to fishing, notwithstanding collapses in South America, California, Southern Africa, and Europe. This research has yielded some powerful generalizations (Bakun, 1996), but still does not allow for prescription on how to ensure 'sustainable' catch levels in the face of environmental variability, growing industry demand and climate change.

In the following, we briefly review various aspects of the fisheries for 'forage' fish, based on geo-referenced catches, from 1950 to the present, and analyze some features of these catches and of the fleets that made them.

FORAGE FISH

Forage fish is a term used to describe schooling fish that are often the prey for larger fish, seabirds and marine mammals. These larger animals often 'forage' on smaller fish because they are found in large schools and are easy to capture. Small pelagic fish (< 30 cm in length) such as Peruvian anchoveta make up the bulk of forage fish, but some medium-sized fish (30-90 cm in length) such as mackerels are also considered forage fish. Many populations of forage fish, especially small pelagics, fluctuate in response to changing oceanographic conditions, which affect their planktonic food (Cury *et al.*, 2000). Other factors such as predation levels, current patterns for larval retention, food availability and water conditions such as temperature affect the annual abundance of these fish (Fréon *et al.,* 2005).

The schooling behaviour of forage fish allows them to be easily caught so that the fishing fleets do not require as much fuel as, for example, trawlers (Tyedmyers *et al.*, 2005; see also Chapter 2). This translates to lower operating costs and hence cheaper fish. Forage fish that are not consumed directly by humans are extremely inexpensive compared to other fish to the extent that they can be reduced to fishmeal and fish oil and still be price-competitive with soymeal. Some of these small and medium pelagic fish are also consumed by humans (Chapter 2), and caught using the same gear, often on the same fishing grounds.

GEO-REFERENCED CATCHES

Reported catch data from FAO, ICES, NAFO and other regional/national sources were allocated to a global system of 30-minute spatial cells using a rule-based approach that utilized databases of fish distributions and fishing access agreements as filters (Watson *et al.*, 2004b; see also www.seaaroundus.org). Emphasis was given to small pelagic fishes, and other species used in reduction fisheries, i.e., forage fish as defined in this report (Table 1). Also, the fuel consumed by the various gears used to catch forage fishes was estimated, based on the approach and data in Tyedmers *et al.* (2005).

RESULTS

Landing trends

Although forage fish have been used for industrial purposes for the last two centuries, it was not until after World War II that these fisheries became highly industrialized with landings of fish destined for reduction being a significant proportion of the fish landed globally (Figure 1). In 1948 only 7.7% of total landings was reduced to fishmeal and fish oil (Macer, 1974). In 2002, 29.2 million t or 36% of all marine fish landings were used for non-food purposes. Ninety percent of these landings (non-food), much of it consisting of forage fish, is reduced, with the remaining 10% used directly for feed in aquaculture and fur animals (FAO, 2005; Figure 1). The by-products of processing fish for human consumption are also destined for fishmeal and fish oil, but make up a small, albeit increasing, proportion of catches and are not included in this analysis (Kelleher, 2005).

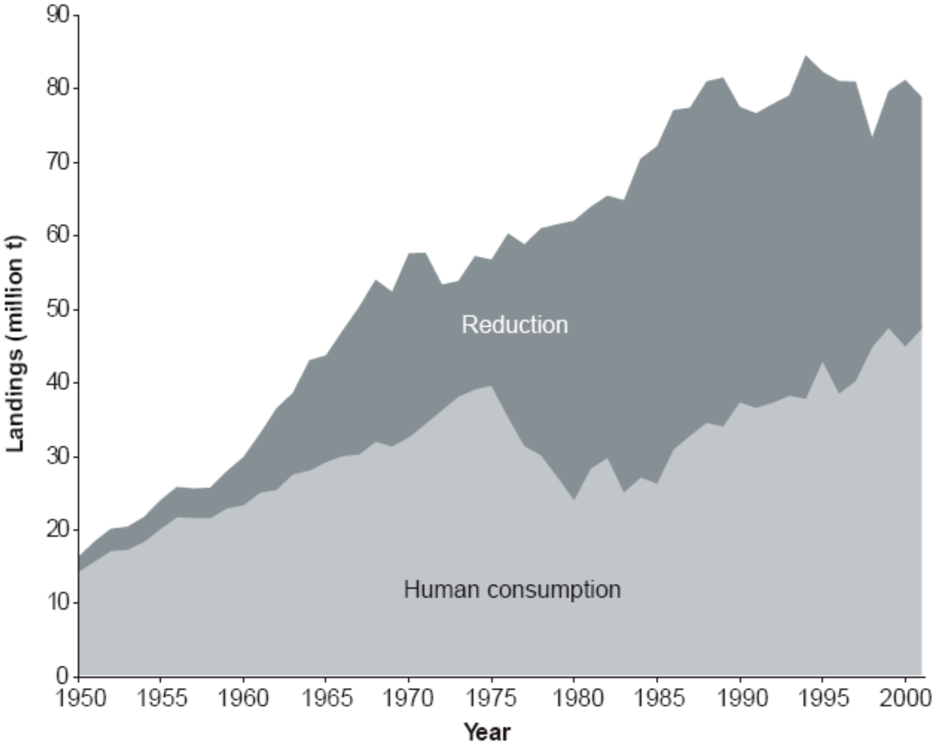


Figure 1. Trend in landings destined for reduction as proportion of total global landings 1950 to 2001. Data from 1950 to 1970 based on Macer (1974); 1971 to 1975 based on Grainger and Garcia (1996); data after 1975: *Sea Around Us* Project *(2006*).

Landings destined for reduction increased very slowly until 1958, when large industrial-scale fishing for Peruvian anchoveta (*Engraulis ringens*) began (Figure 2). This fishery is now the world's largest with recent landings of 2.5 million t in 2001, all of it destined for reduction. As indicated above, landings of fish destined for reduction are highly variable, as seen in total global landings of fish destined for reduction (Figure 2) and landings of forage fish by region (Figure 3). Declines in landings in the 1970s, 1980s and 1990s are due to the impact of the El Niño in those years.

The global pattern of fish catches destined for reduction has remained roughly similar since the 1970s (Figure 4). If anything, the role of the forage fish fisheries for economic well being and food security along the coasts of Chile and Peru has increased since the 1970s.

The bulk of the fish caught for reduction are caught in South America (Peru and Chile), Northern Europe (Denmark, Iceland and Norway) and on the east coast of the United States. In the early 1960s and 1970s, Japan and the USSR had sizeable fleets including factory ships fishing for fish for reduction. However, with the establishment of EEZs and consequent closing of fishing areas to these nations, they have scaled back their fishing operations, including processing plants.

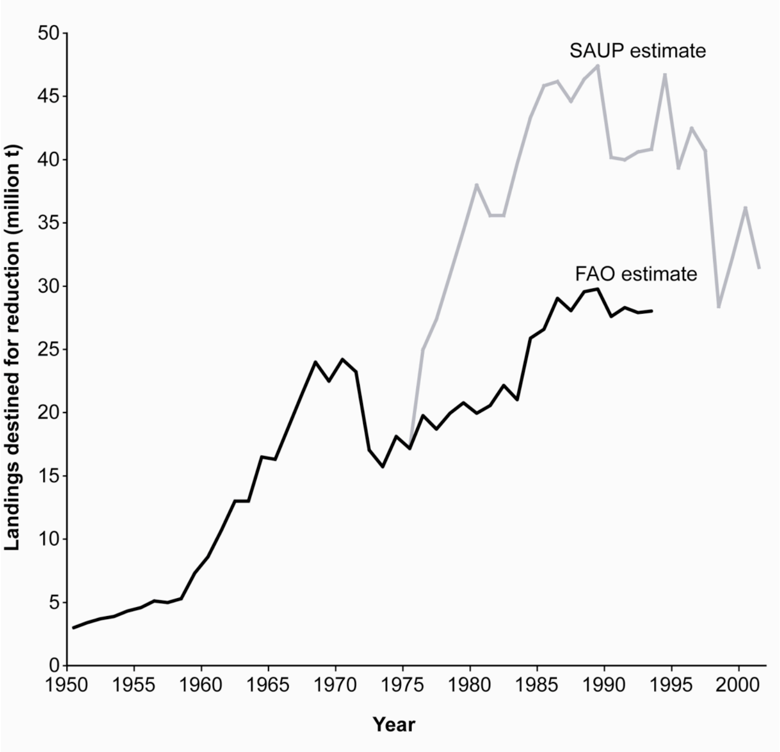


Figure 2. Trend in global landings for fish destined for reduction 1950 to 2001. Data prior to 1976 based on FAO total estimates (see Grainger and Garcia 1996), data from 1976 onwards based on *Sea Around Us* Project (2006) taxon- and country-based associations with fishmeal types reported in FAO production figures.

South America, in particular Peru and Chile, experienced considerable growth in landings from the late 1950s and peaked in the early 1990s; although catches are highly variable, they now fluctuate at around 20 million t annually (Figure 3). The fisheries in Northern Europe and on the east coast of the United States were established much earlier than those in South America. The landings in these two areas are less variable with much lower landings (Figure 3).

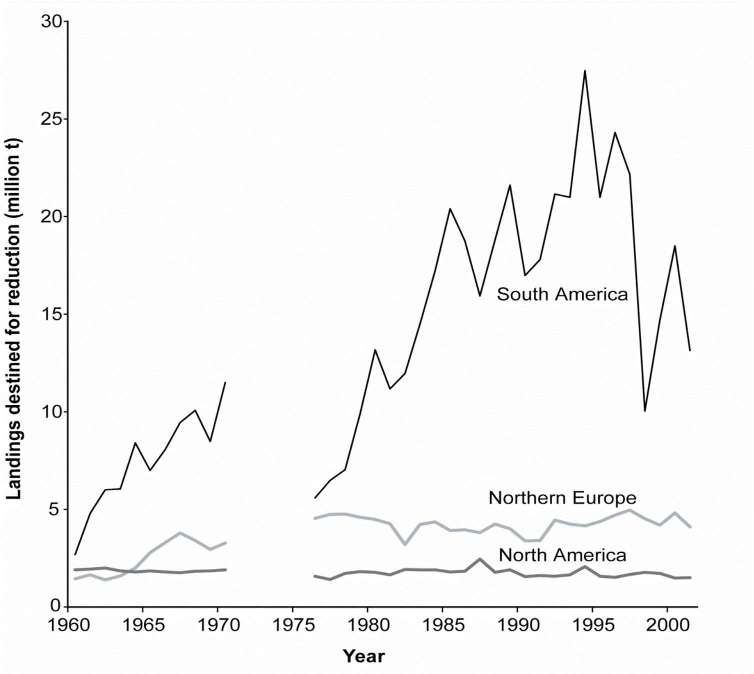


Figure 3. Trends in landings for fish destined for reduction 1960 to 2001, by major region. Region-specific information not available between 1970 and 1976 (Grainger and Garcia, 1996; *Sea Around Us* Project, 2006)

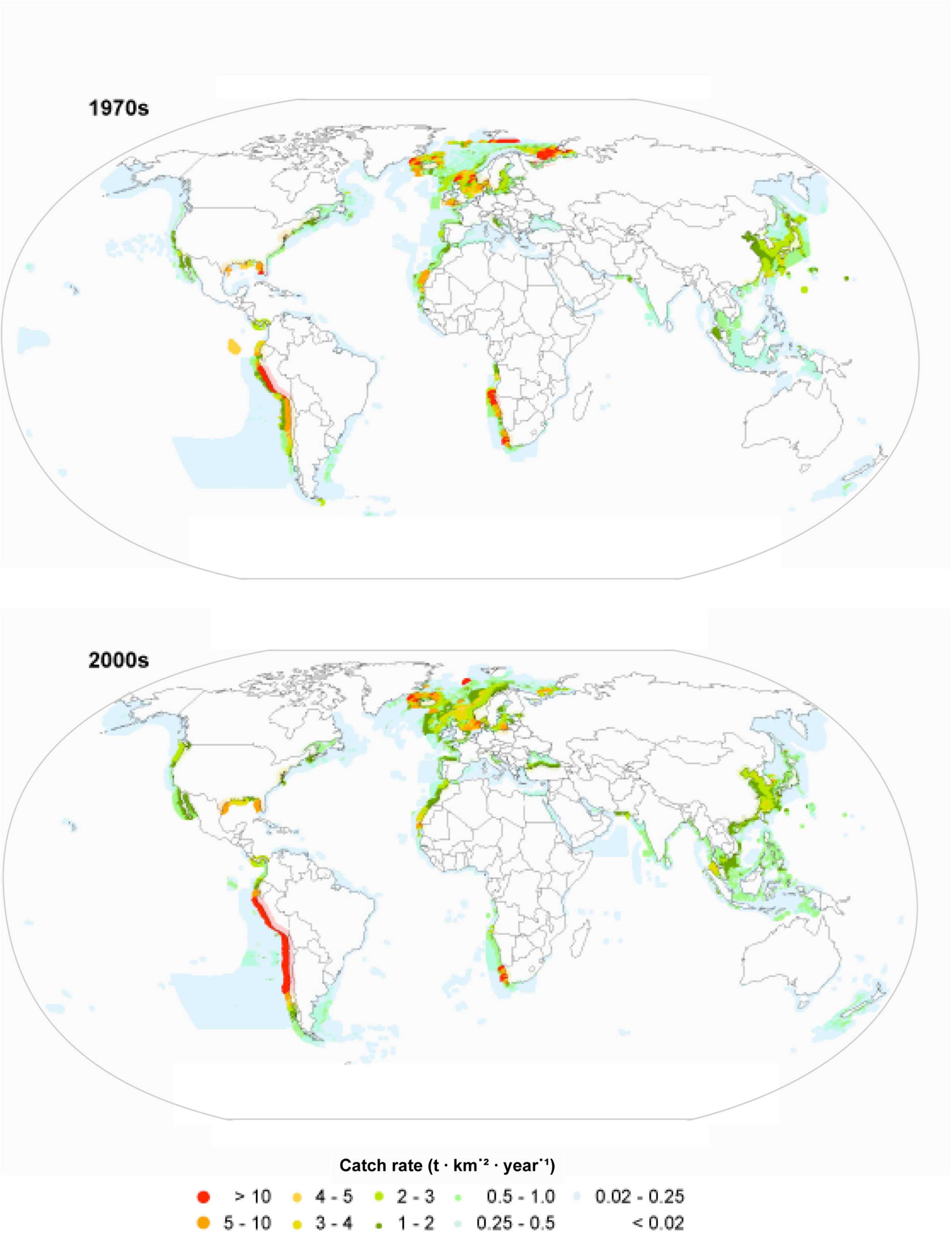


Figure 4. Catch rates for species associated with reported species destined for reduction for the 1970s and 2000s based on spatial cell-based global fisheries data (Watson *et al.*, 2004b) and a taxon- and country-based association with fishmeal types reported in FAO statistics.

Composition of landings

While herring, sardines and menhaden were the main species targeted early in the 20th century, the range of species targeted has expanded since then (Table 1). In the 1950s, the bulk of fishmeal produced originated from landings of fish from Northern Europe (e.g., herring), Japan (e.g., Pacific herring), Southwest Africa (e.g., Pilchard) and the United States (e.g., Menhaden). The development of the anchoveta fisheries off Peru and Chile soon overshadowed these countries with volumes of landings far exceeding that of the other countries.

Table 1: Species that made up 75% of the fishmeal produced globally in 1950, 1976 and 2001, starred species were present in only that year. Data from 1950 based on Grainger and Garcia (1996); other data: *Sea Around Us* database (2006).

1950	1976	2001
Atlantic herring	Peruvian anchoveta	Peruvian anchoveta
Atlantic menhaden*	Capelin	Inca scad*
Japanese pilchard*	South American pilchard	Capelin
Gulf menhaden	Chub mackerel	Blue whiting
Chub mackerel	Atlantic herring	Japanese anchovy*
European sprat	European pilchard*	Chub mackerel
Capelin	European sprat	South American pilchard
Blue whiting	Norway pout*	Atlantic herring
Pacific menhaden*	Atlantic mackerel	Threadfin breams*
Peruvian anchoveta	Gulf menhaden	Sandlances
Chilean jack mackerel	Sandlances	Gulf menhaden

The expansion of the reduction industry in South America from the late 1950s saw a dramatic change in the species composition of fishmeal globally (Table 1). In 1950 landings of anchoveta made up a very small proportion of the global production of fishmeal. However, by the late 1960s, it was the major species for fishmeal (Figure 5). It was estimated that, in 2003, Peruvian anchoveta landings made up 57% of global landings used for fishmeal production (IFFO, 2004). While Peruvian anchoveta was the major species used in Peru, Chile uses additional species such as Hake (*Merluccius gayi*) and Horse mackerel (*Trachurus murphyi*). In the past, Pilchard (*Sardinops sagax*) was also used. The estimated use of pilchards increased steadily until the early 1980s and then declined (Figure 6) while the estimated use of hake has fluctuated markedly throughout the 1990s (Figure 7).

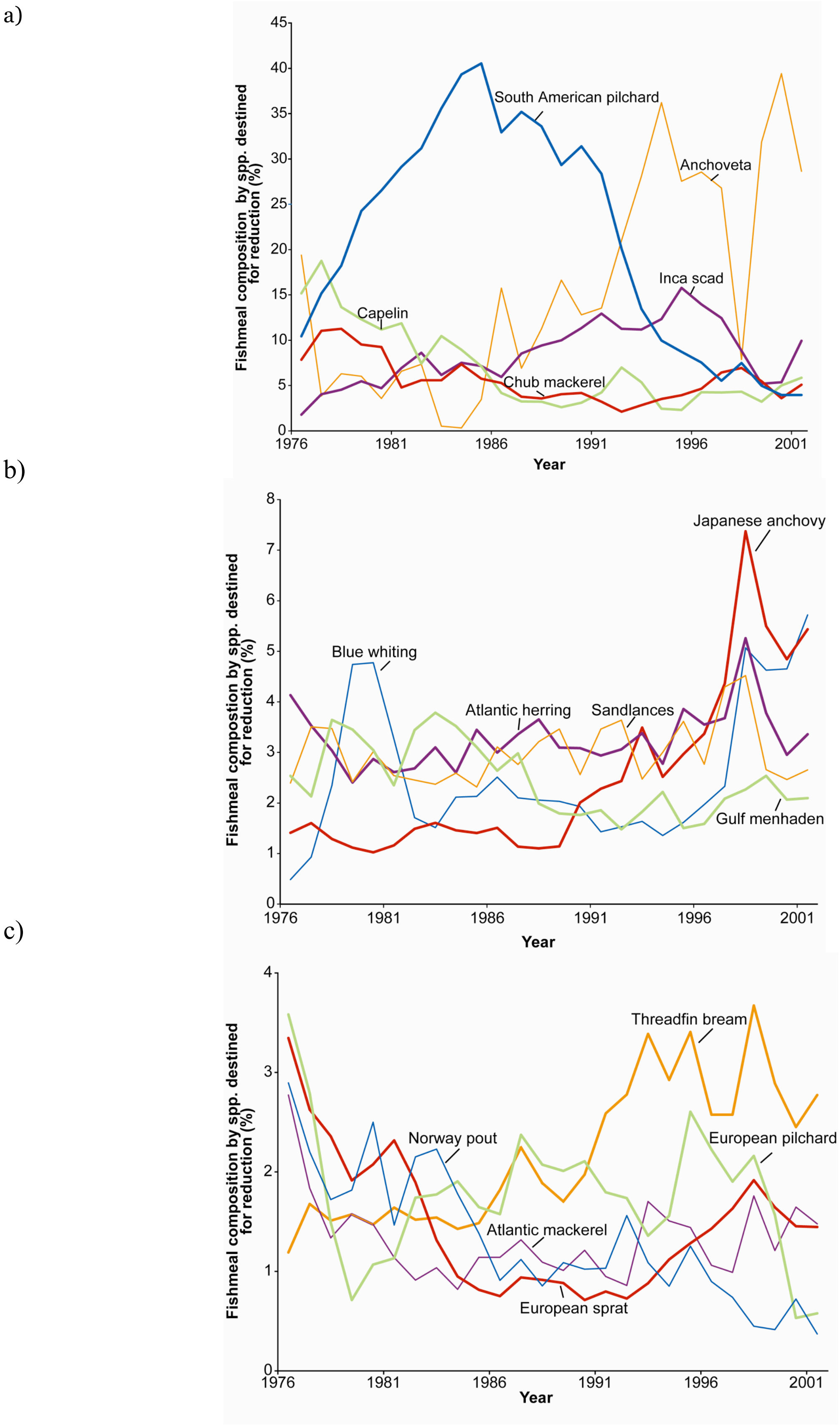


Figure 5a, b and c. Trends in the composition of fishmeal based on the top species destined for reduction that made up at least 75% of the fish used by volume for reduction in 1976 and 2001; 5a is top 5, 5b is middle five, 5c is bottom 5 (*Sea Around Us* Project, 2006).

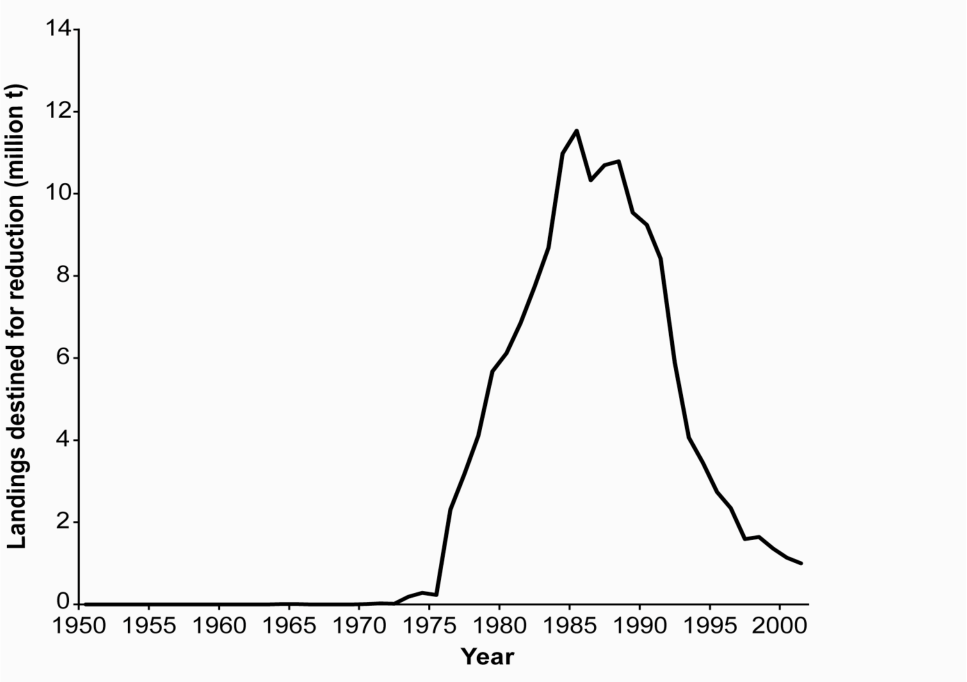


Figure 6. Landings of South American Pilchard destined for reduction 1950 to 2001. (*Sea Around Us* Project, 2006).

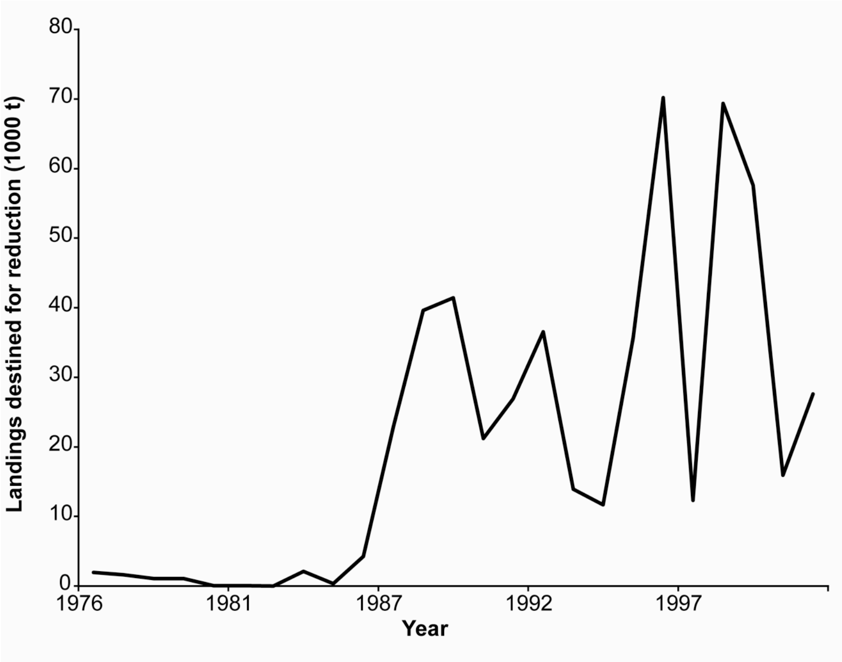


Figure 7. Chilean landings of Hake destined for reduction between 1976 and 2001. (*Sea Around Us* Project, 2006).

In Northern Europe, landings of Atlantic herring and Capelin have historically been used for fishmeal, and Capelin continues to be a major reduction fishery, making up 10% of fish landings used for fishmeal globally. Northern Europe has used a variety of species for fishmeal including the European sprat, Norway pout and Haddock (Macer, 1974). European sprat declined as a major component of fishmeal from the 1960s with current landings accounting for a small proportion of total fishmeal production (Figure 8). Norway pout followed a similar pattern. It was not used in fishmeal production until the early 1960s; landings peaked in the mid-1970s and then declined steadily, with current catches of this species estimated to be less than 100,000 t (Figure 9). Initially, Blue whiting was caught as bycatch from industrial fishing and used as fishmeal but as species of fish traditionally used for fishmeal were reduced a fishery targeting this species developed as early as the late 1960s (Macer, 1974). This species is now a major component of fishmeal globally and is especially important in Northern Europe where it makes up 2% (in 1975) to 35% (in 2001) of fish destined for reduction (Figure 9). While declines in the abundance of some target species can account for the change in composition of fishmeal in Europe historically the composition of herring in fishmeal has changed due to European policy. A recent EU directive prohibits the use of Atlantic herring being used for fishmeal and therefore may account for some of the decline in the use of these species (Huss *et al.*, 2003). However, the restriction does not apply to Norway and Iceland, two of the largest producers of fishmeal in Northern Europe.

In the United States, Menhaden has been used since the early 1900s for reduction and continues to be the major fishery for reduction in the United States; its catches are relatively constant (Figure 10). Japan has an even longer history of fishmeal production based on Pacific saury, Pacific herring and sardines. When the stocks of these fishes declined, Japan extended its fishing efforts further offshore using a distant water fleet and factory ships, especially for processing the by-products of the Alaska pollock fishery (Macer, 1974). Japan's landings of fish destined for reduction declined as did the volume of fishmeal and fish oil as countries established their EEZs and restricted access to their fish resources including fish used for reduction. In Southwest Africa, landings used for reduction have declined since the 1960s after peaking in the 1960s, with catches now fluctuating around 850,000 t annually (Figure 11).

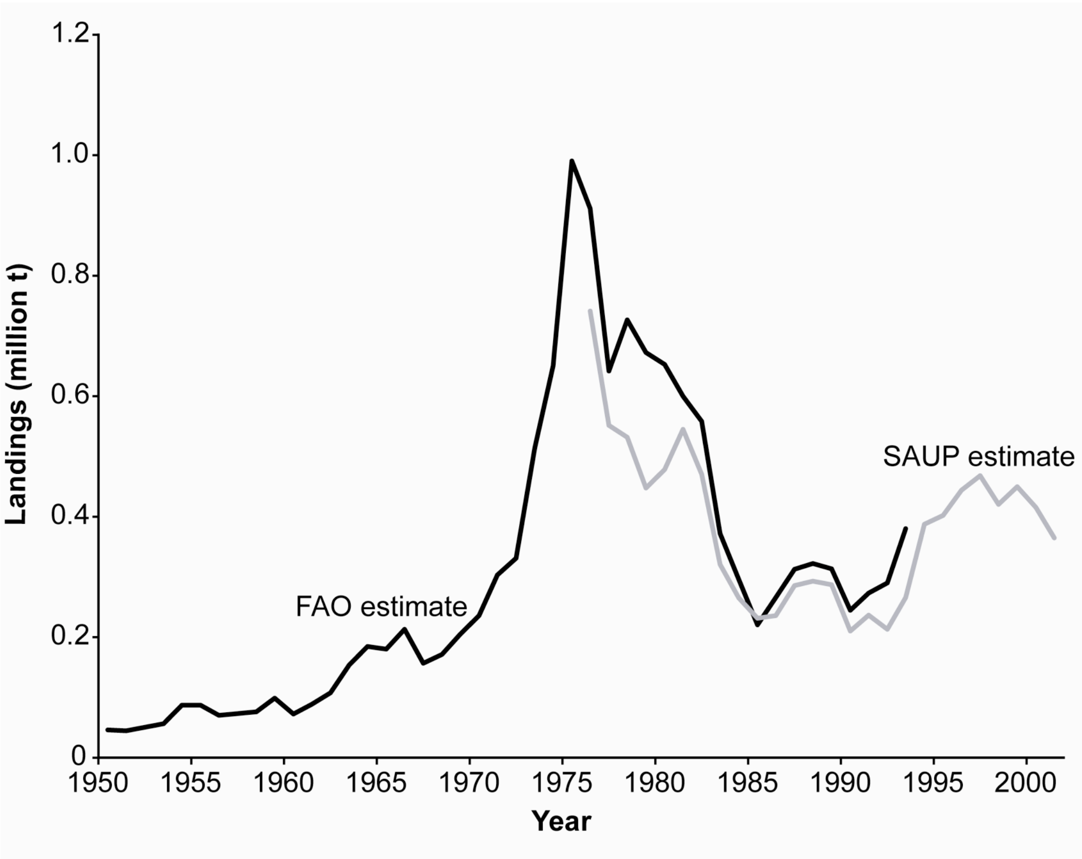


Figure 8. Northern European landings of European sprat from 1950 to 2001. Data from 1950 to 1992 from Grainger and Garcia (1996); after 1975, data are from the *Sea Around Us* Project (2006). The FAO estimates were derived from expert opinion, i.e., from the International Fishmeal and Fishoil Organisation (R.J.R. Grainger, FAO, *pers. comm.*). *Sea Around Us* Project data were derived as described in methods section.

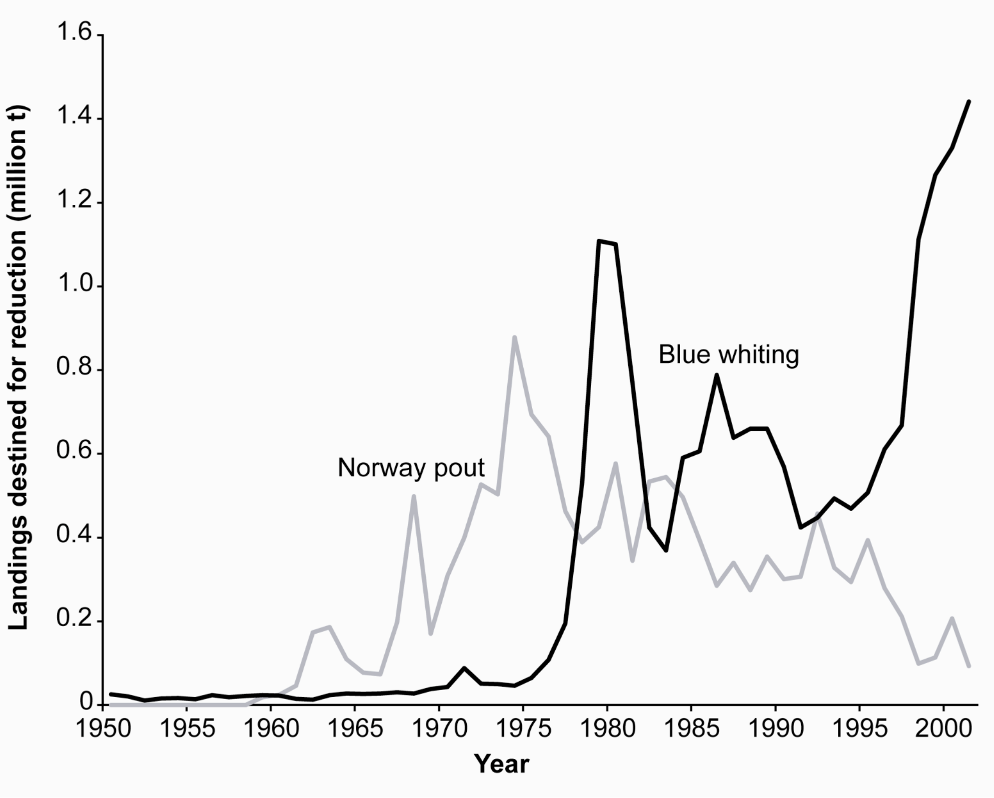


Figure 9. Northern European landings of Norway pout and Blue whiting destined for reduction from 1950 to 2001. Data from 1950 to 1975 are from Grainger and Garcia (1996); after 1975, data are from the *Sea Around Us* Project (2006).

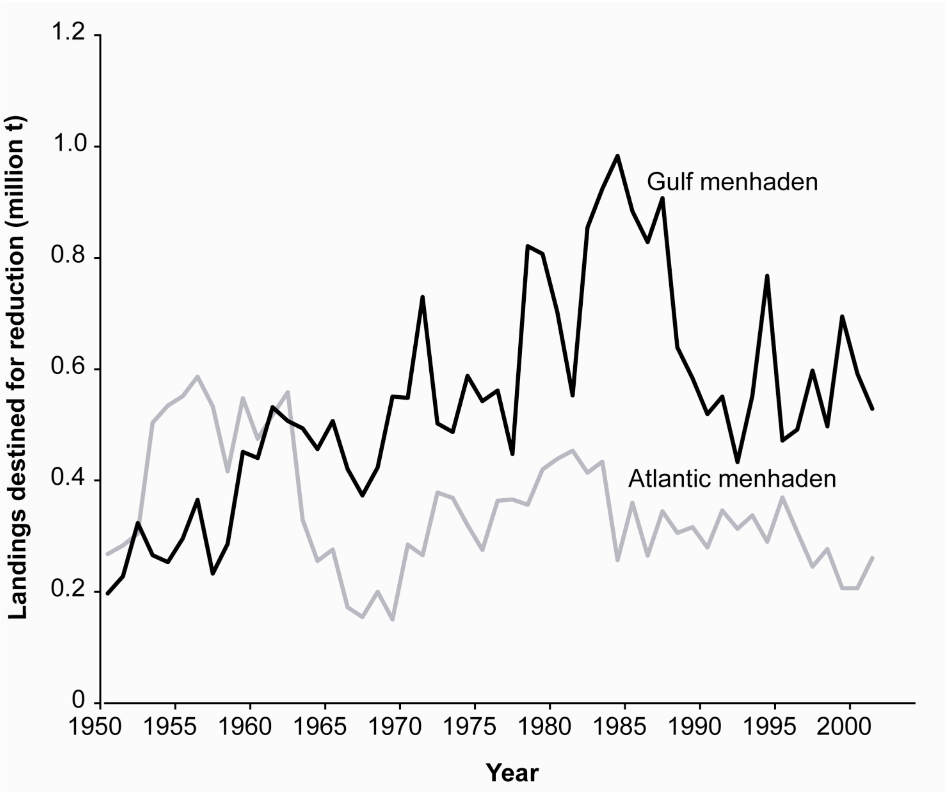


Figure 10. United States landings of menhaden from 1950 to 2001. Data from 1950 to 1975 are from Grainger and Grainger (1996); after 1975, data are from the *Sea Around Us* Project (2006).

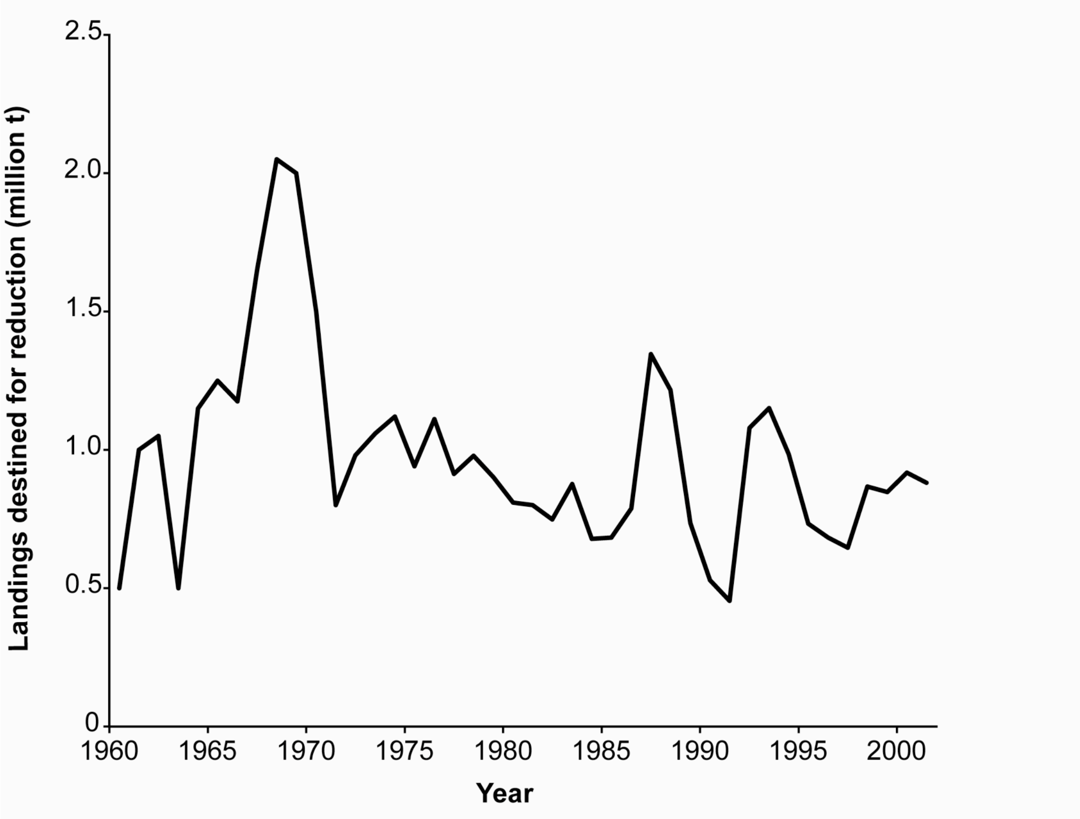


Figure 11. Trends in landings of fish destined for reduction in Southern Africa (Macer, 1974; Payne and Crawford, 1989 and *Sea Around Us* Project*,* 2006).

Globally, the species composition of fishmeal and fish oil is highly variable from year to year because the fisheries that are used as inputs into this sector are highly sensitive to oceanographic changes (Figure 4). For example, in strong El Niño years anchovy abundance declines, while sardine abundance increases (Bakun and Broad, 2003); this is also reflected in the catch and ultimately in fishmeal composition from Chile and Peru. The variability in the composition also affects the quality; fishmeal and fish oil high in Peruvian anchoveta are considered superior to fishmeal and fish oil with significant sardinella inputs since the fat content is different. Figure 4 illustrates the variability of landings of the majority of species that are used in the production of fishmeal. In 1976 the top five species (Peruvian anchoveta, Capelin, South American pilchard, Chub mackerel and Atlantic herring) accounted for 57% of the fish used for reduction, the same suite of species accounted for 47% of fish used in 2001.

The status of most species used for reduction is fully exploited or over-exploited, and in a few cases declining, indicating that overall there is little scope to increase landings of these species (Table 2). Technological advances have reduced the conversion ratio of live fish to fishmeal from 5 to 4, enabling production of fishmeal to increase slightly. However, fisheries based on other species will need to be developed if total fishmeal production is to increase from its current level.

Table 2. Stock status for fish destined for reduction in 2002 (based on FAO, 2005). Multiple values of exploitation are due to multiple stocks with the FAO area being in different states of exploitation.

Target Stock	FAO Area	State of exploitation in 2002 (FAO 2005)*
Atlantic menhaden	NW Atlantic (21)	F
	WC Atlantic (31)	F
Gulf menhaden	WC Atlantic (31)	F
Atlantic mackerel	NE Atlantic (27)	F
Blue whiting	NE Atlantic (27)	O
Norway pout	NE Atlantic (27)	F
Sandeels/Sandlances	NE Atlantic (27)	F
Atlantic herring	NW Atlantic (21)	U-F-R
	NE Atlantic (27)	F
European sprat	NE Atlantic (27)	F
	Mediterranean & Black Sea (37)	D
Capelin	NE Atlantic (27)	F
Chub mackerel	EC Atlantic (34)	F
South African anchovy	SE Atlantic (47)	F
Horse mackerel	SE Atlantic (47)	M/F
Pilchard	SE Atlantic (47)	F
Pacific herring	NW Pacific (61)	?
Pacific saury	NW Pacific (61)	F
Japanese sardine (anchovy)	NW Pacific (61)	F
Peruvian anchoveta	SE Pacific (87)	R-O
South American pilchard	SE Pacific (87)	F-O
Chilean jack mackerel	SE Pacific (87)	F-O
Hake	SE Pacific (87)	F-O-D

*F=fully exploited; O=overexploited; U=underexploited; R=recovering; M= moderately exploited (FAO, 2005)

Trophic level trends

The mean trophic level was calculated as described in Pauly and Watson (2005) for fish used for reduction from 1976 to 2001. Globally, the trophic level of fish destined for reduction has increased since the mid 1980s (Figure 12). Increasing average trophic levels has occurred in most areas of the world, the exception being Africa, where the average trophic level has remained relatively stable (Figure 13). The biggest increases have been in Europe where the trophic level increased from 3.17 to 3.4, and in Asia where the trophic level increased from 3.11 to 3.77. In North America and South America, the average trophic level has increased by 0.26 between 1986 and 2001.

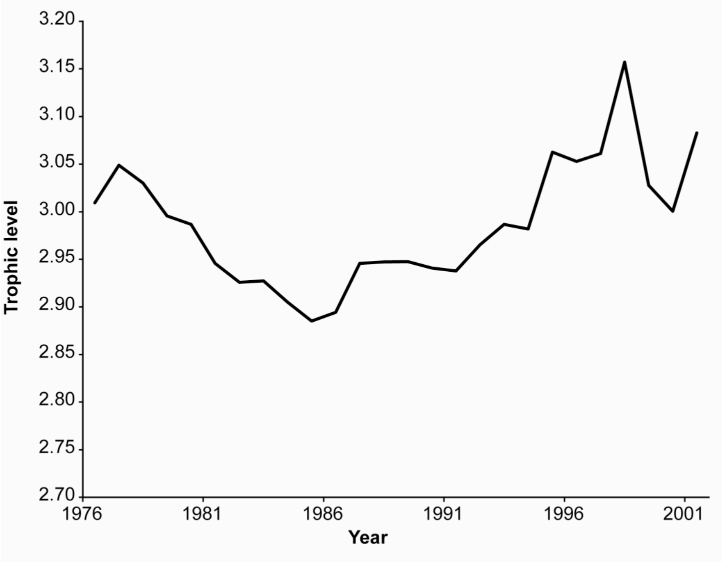


Figure 12. Trend in weighted mean trophic level of fish destined for reduction from 1976 to 2001 (*Sea Around Us* Project, 2006).

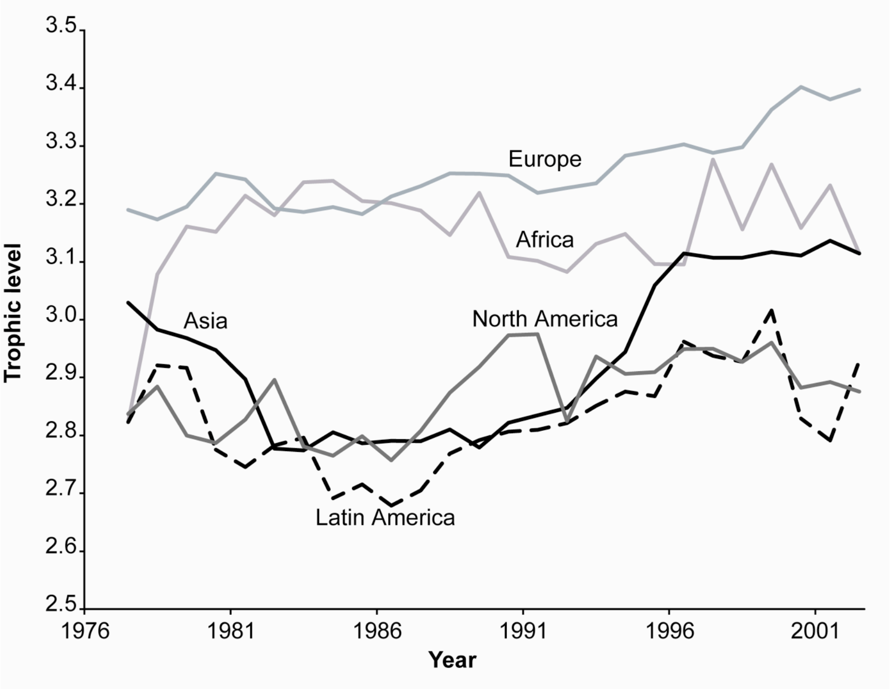


Figure 13. Trend in weighted mean trophic level of fish destined for reduction in the major continents from 1976 to 2001 (*Sea Around Us* Project, 2006).

Fuel consumption

Using global fuel intensities available for all reported catch (Tyedmers *et al*., 2005) it was possible to associate the landings of all forage fish with the fuel used to catch them. In Figure 14, we can see that a maximum of 10 billion litres of fuel was used in some years for the reported forage fish catch, which was nearly a fifth of all fuel expended in global fisheries. Forage fisheries use approximately 40% less fuel than the average fishery. Of all species used for reduction, more was expended on Atlantic horse mackerel then any other. This was followed closely by Chub mackerel, Japanese anchovy, South American pilchard, and Threadfin breams (*Nemipterus* spp.). There were many other species ('other') which, combined, accounted for more than two-thirds of fuel expended in forage fisheries. Because the value of fish used primarily for reduction is low compared to other species used as seafood, it is necessary to catch large quantities for the same amount of fuel as expended in other fisheries.

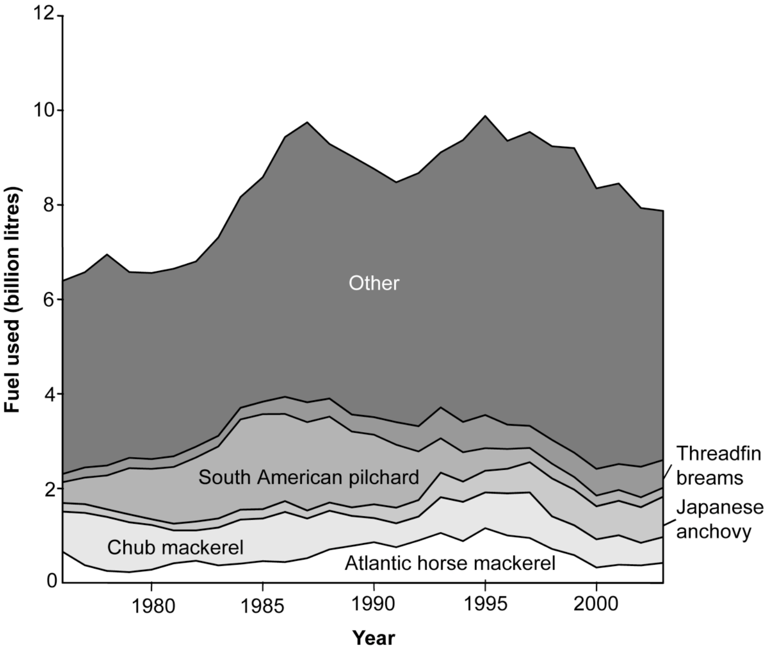


Figure 14. Fuel used to catch reported reduction fishery landings (*Sea Around Us* Project, 2006).

Fishing gear used

A database associating all global catch and the gear used to take it (Watson *et al*., 2004a, Watson *et al*., 2006) based on the gear descriptions of von Brandt (1984) was used to break down the landings of forage fish (Figure 15). This revealed that most were taken by seine net; however, this has declined as there is an increasing use of trawls to catch forage fishes. Other types of gears are not globally important (Figure 15).

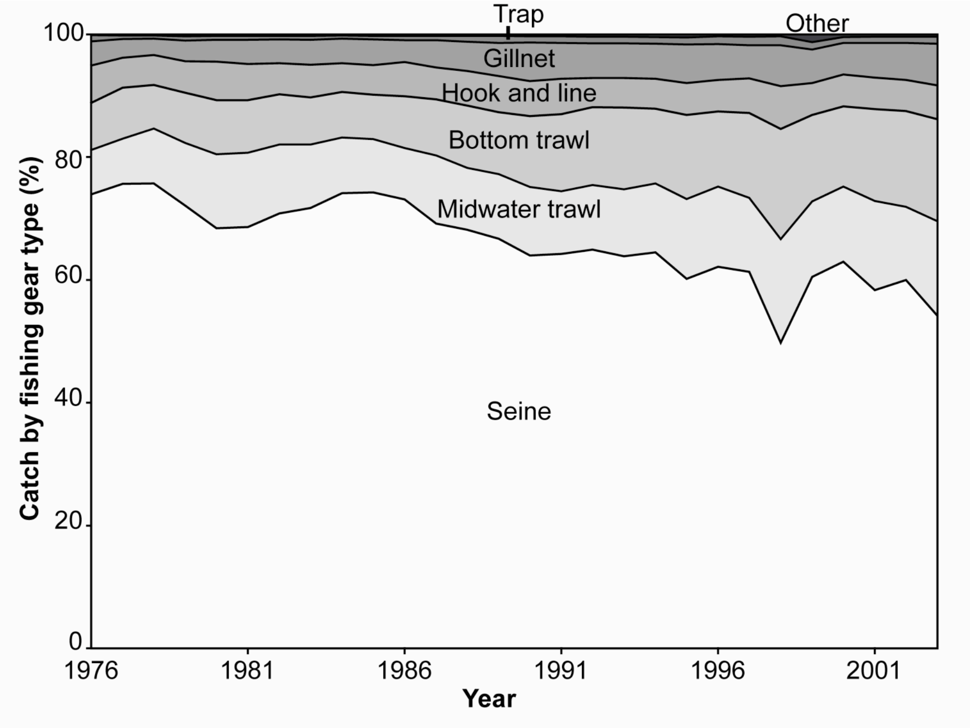


Figure 15. Gears used globally to catch forage fish. The dominance of seine nets is evident (*Sea Around Us* Project, 2006).

Prices

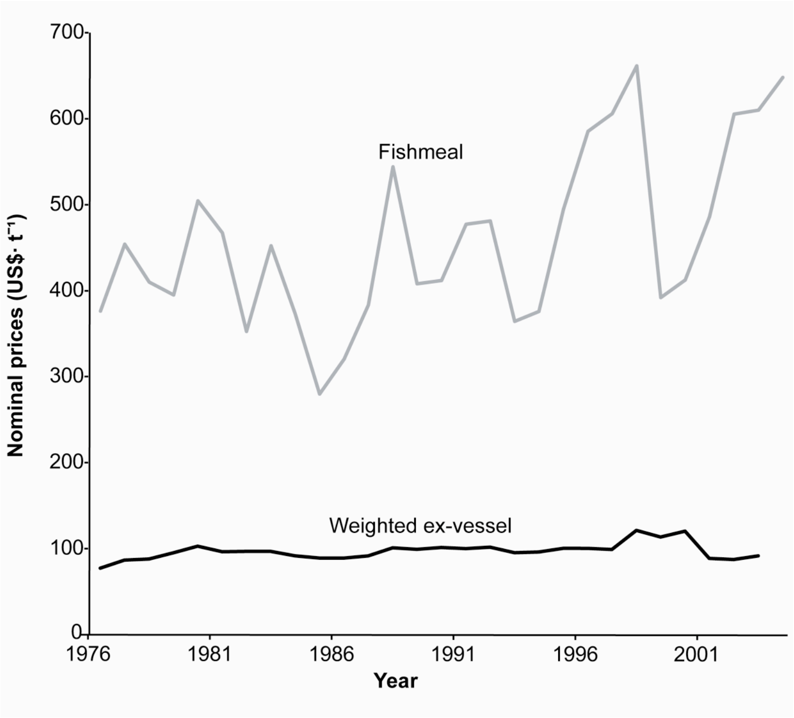
The *Sea Around Us* Project prices database (Sumaila et al. [*in press*], and www.seaaroundus.org) was used to estimate the average ex-vessel price of fish destined for reduction. Similarly, the UN Conference on Trade and Development (UN, 2006) database was used for fishmeal. Only prices that were observed or derived directly from observed ex-vessel prices were used (Table 3). A weighted average price was used since anchoveta dominates the species composition of fishmeal and has one of the lowest ex-vessel prices. The weighted prices are based on nine taxa that make up at least 60% of the landings used in fishmeal for the period 1976 to 2003. Both weighted nominal and real prices (adjusted to 2000) were calculated, and the nominal ex-vessel price compared to the nominal price of fishmeal (Figure 16 a and b).

Table 3. Countries and taxa used to estimate weighted ex-vessel prices for fish used for reduction.

Taxon	Countries				
Atlantic herring	Denmark	Germany	Iceland	Ireland	USA
Blue whiting	Denmark	Iceland	Ireland	Norway	UK
Capelin	Canada	Denmark	Iceland	Norway	☐
European sprat	Denmark	Germany	Norway	UK	☐
Sandlances	Denmark	Norway	UK	☐	☐
South American pilchard	Chile	Mexico	USA	☐	☐
Chub mackerel	Chile	USA	☐	☐	☐
Peruvian anchoveta	Chile	☐	☐	☐	☐
Gulf menhaden	USA	☐	☐	☐	☐

Since 1976, the nominal price of fishmeal has increased while the average nominal ex-vessel price has remained relatively constant (Figure 12). However, when real prices (adjusted to 2000) are used, the real price of fishmeal actually declines and the real ex-vessel price declines, albeit very slowly. The combined uncertainties associated with ex-vessel prices, of tonnage used and the Hamburg market prices (see Figure 16b) make the global value of fishmeal and the ex-vessel value of fish difficult to estimate reliably, and we abstain from presenting such a figure here.

a)



b)

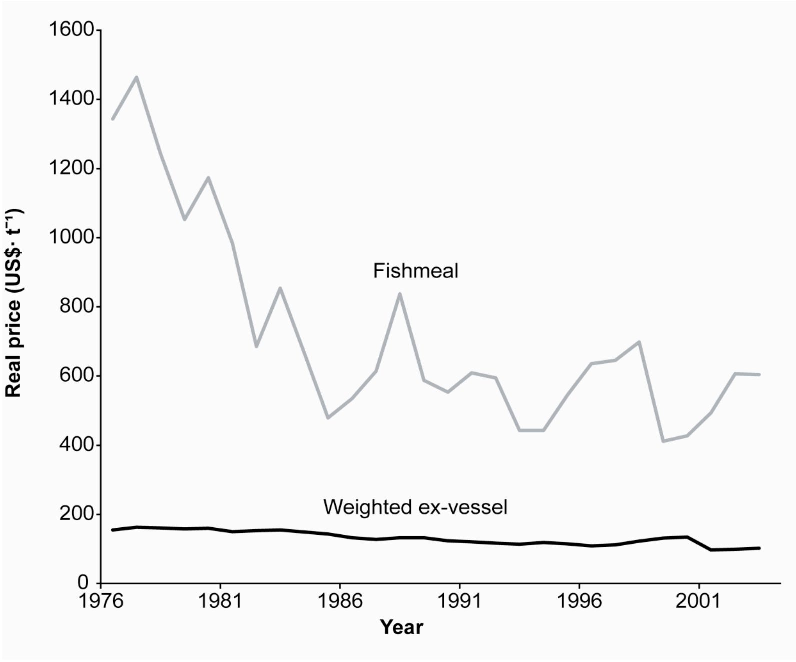


Figure 16a and b. Trends in weighted ex-vessel prices (nominal and real 2000) for fish used in reduction and prices for fishmeal sold at the Hamburg market (United Nations, 2006)

DISCUSSION

Landings of fish destined for reduction have increased since the 1950s and appear to have stabilized around 25 million t live weight. However, the species targeted for reduction have changed and, in most areas of the world, fish at higher trophic levels are being used. While some argue that most fish landed for reduction are not consumed by humans, the evidence is otherwise (Chapter 2).

The fluctuation in global landings and landings of particular species for reduction often raises the question of the impact of these catches on stocks and the ecosystem. Bakun and Broad (2003) noted that small pelagic fish, which make up much of the bulk of forage fish, are extremely important to the transfer of production 'upward' in the food webs of the ecosystems they are found in. They operate at the crucial 'wasp-waist' trophic level where one or more small plankton-consuming nektonic species tend to dominate the trophic transfers, whereas there are relatively more species involved in transfers at lower and higher trophic levels (Bakun and Broad, 2003).

For fisheries where assessments are available, the status of many of these fish stocks have been assessed as 'sustainable' and fished 'within safe biological limits' by various organizations (IFFO, 2004; Seafeeds, 2003). However, an analysis of the exploitation status of the major species used for reduction indicates that not all stocks are within safe limits (Table 2). Some stocks such as North Sea herring were used for reduction until 1997, when the stock collapsed and the EU banned the use of herring for reduction. Some herring that is caught as bycatch can be used for reduction (EU, 2004). The status of forage fish stocks are often reported on large geographic areas and are usually amalgamations of more than one stock within an FAO area, so the status of individual stocks in an FAO area can be masked if one or two stocks overwhelm the overall assessment.

Most fisheries are under some form of management and, given the variable nature of these fisheries and their sensitivity to changing oceanographic conditions, a precautionary approach to management including setting of quotas and effort limits is needed. In Europe, the quota for capelin was recently reduced from 700,000 t in 2005 to 200,000 t in 2006 (Carvajal, 2006). A notable exception is Blue whiting, one of the major inputs to fishmeal and fish oil in Europe. The stocks in the North Atlantic are exploited by a number of countries within and outside of the EU, which cannot agree on quotas for the fishery. In 2004 landings totalled over 2 million t, despite a recommended quota of less than a million t by ICES (EU, 2004). There is considerable concern over the long-term viability of this fishery, as there is for other forage fish fisheries such as Horse mackerel, European sprat, North Sea herring and other species throughout the world.

The impact of landing 40% to 60% of the Sandeel biomass consumed by many demersal fish in the North Sea and of landing 15% to 20% of the total biomass is poorly understood for this and other species (ICES, 2003). A study of the impact of reduction fisheries on EU marine systems was completed in 2003. The general conclusion was that *"the impact of industrial fisheries we were able to identify is relatively limited compared to the effects of fisheries for species destined for human consumption"* (Anon., 2004). ICES also looked at the particular questions of the impact of the fisheries on predators and bycatch. Information on predator interactions was limited to the North Sea and they concluded that the overall impact of industrial fisheries was relatively limited; however, interactions can be locally significant with certain populations of predators. Again, data were available from the North Sea to investigate only the impact of incidental bycatch and the conclusions were that most incidental catches are also species used for fishmeal (e.g., Blue whiting is a bycatch of the Norway pout fishery). Also, where edible species are caught incidentally, they generally make up a low proportion of the catch, which is considered acceptable by the EU (EU, 2004).

Forage fish are important prey for marine mammals and seabirds as seen in El Niño years in South America, where there are significant mortalities of these animals due to a reduction in prey abundance (Hays, 1986). Changes in the Benguela upwelling system also result in substantial mortalities of marine mammals and seabirds (Crawford *et al.*, 1992). The additional impact of commercial fishing of prey species on marine mammals and seabirds in El Niño years has yet to be assessed. In the US, there is growing concern over the landings of Menhaden impacting the catch of Striped bass in Chesapeake Bay (Uphoff, 2003). Similarly, concern in the UK over the interaction of Sandeels and seabirds resulted in the European Union Council of Fisheries Ministers banning catches of Sandeel in the North Sea, in an area of 20,000 km^2 bordering the east coast of Scotland and Northumberland (EU, 2004).

The weighted average nominal price of fish destined for reduction has increased from an estimated US$ 77 /t to $US 92 /t, an increase of 19% between 1976 and 2003, while the average Hamburg price for fishmeal has increased from $US 376 /t to $US 611 /t, an increase of 63% over the same time period. The weighted price of fish destined for fishmeal and fish oil has declined from $US 155 /t to $US 91 /t, a decline of 42% between 1976 and 2003, while the real Hamburg price of fishmeal declined from $US 1343 /t to $US 604 /t, a change of 55% over the same period. The price of fishmeal and in turn the ex-vessel price of fish destined for reduction is driven by:

- Supply of fish: when catches increase in South America, prices decline;
- Demand from Asia, which accounts for 60% of fishmeal consumption, much of it in China; and
- The price of soymeals, which can be used as a substitute for fishmeal in animal production (Asche and Bjorndal, 1999).

In the meal market, fishmeal accounts for 5% of total supply while plant meals, especially soy, account for almost 95% of supply. This high percentage of plant meal therefore influences the price of fishmeal and in turn, the ex-vessel price of fish destined for reduction. Although fishmeal is nutritionally superior to soy and other plant meals in intensive animal production systems, if fishmeal prices increase, consumers will substitute plant meals if quality is not compromised (Asche and Bjorndal, 1999), limiting the scope of fishmeal producers to increase price and in turn allowing fishers to increase ex-vessel prices. In Europe, it was noted that ex-vessel prices for fish used for reduction were generally 10% of the cost of fish used for human consumption (Nielsen, 2000).

The upward trends in the average trophic level of landings destined for reduction are in contrast to the downward trend in the average trophic level of fish for human consumption (Pauly and Watson, 2005). This indicates that fish potentially suitable for human consumption are being diverted to fishmeal (a topic discussed further in Chapter 3). This trend is not unexpected since the species composition of fishmeal has changed, including the growing use of Dorab wolf-herring in Asia, King mackerel in the United States, Japanese and Spanish mackerel in China and Japan, Southern hake in Chile and Blue whiting in Europe. Some of these species are also used for human consumption, or through processing technology directly consumed by humans, as is the case for Blue whiting (Chapter 2).

The fisheries for forage fish have changed significantly since 1950, especially with the development of the anchoveta fishery in Chile and Peru. Initially, the demand for fishmeal and fish oil was driven by intensive animal production systems seeking inexpensive, yet effective components to animal feeds. Although the growth of aquaculture over the last 20 years has changed the landscape of use of fishmeals and oils, it has not dramatically changed the nature of the fisheries in South America and the USA. However, the increasing demand for these products has expanded the species of fish targeted for reduction, which now include higher trophic level species. This expansion has also at times resulted in a competition with human needs.

In Europe there have been major shifts in the species exploited and in the sustainability status of some of those species. While the intensive animal production sector can substitute fish for soy, the aquaculture sector is currently limited and thus the pressure to exploit forage fisheries at their maximum will continue as new species become exploited with unknown impacts on marine mammals and seabirds.

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CHAPTER 2

HUMAN CONSUMPTION OF FORAGE FISH[2]

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ABSTRACT

A brief review of the human consumption of forage fishes is given, with emphasis on traditional use of fermented small pelagic fishes and the importance of these fish to food security. This contrasts with the notion, disseminated by some actors in fisheries and aquaculture, that the 'reduction' of such fishes to fishmeal and fish oil has no impact or only a positive impact on human food security. Current regional patterns of small pelagic consumption are reviewed, which suggest that agricultural uses of forage fish compete directly with human consumption in some areas of the world. This has resulted, overall, in a declining human consumption of relatively cheap pelagic fish and, in richer countries, in increased consumption of pricey seafood, some of it farmed and fed meals and oils derived from small fish, particularly small pelagics.

HISTORICAL INTRODUCTION

Many of the small, pelagic fishes described as 'forage fish' in other parts of this document have a long history as human food fishes. This is well documented; for example, over 2000 years ago, small pelagic fish were very popular around the Mediterranean basin (Gamer, 1987). Thus, many of the amphora recovered from sunken Roman vessels did not contain wine, as is generally assumed, but *garum*, a fish 'sauce' derived from fermented small pelagic fishes, similar to Vietnamese *noc mam*, or Thai *nam pla,* about which more is found below. *Garum* was a mainstay of the Roman diet, and the manufacture and consumption of *garum* has survived from the Roman Empire to the present, at least in a few villages of southern Italy.

In northwestern Europe, a transition from freshwater to marine fishes occurred at the end of the first millennium, possibly due to urban population growth and the concomitant overexploitation of freshwater species (Barrett *et al.*, 2004; Pauly, 2004). The ascent of Christianity intensified the demand for fish, the only permitted animal protein during an interminable Lent period, and on other religious occasions. Fish preservation required huge quantities of salt at specific times, and salt monopolies (and the blackmail of fishers that they allowed) were extremely profitable, as we also know from other parts of the world (Butcher, 1996, 2004). Such a salt monopoly was established in the high Middle Ages by an alliance of mainly German cities, the Hanseatic League, which derived its immense wealth – embodied in superb buildings we now can admire in Hamburg or Lübeck – ultimately from the Norwegian, North Sea and Baltic herring fisheries (Sarhage and Lundbeck, 1992).

The Hanseatic League monopoly was followed by Dutch supremacy, whose herring fisheries, in the 17th century, allowed the Low Countries, for a while, to delay the emergence of Britain's unchallenged naval power – all this thanks to the lowly herring. In fact, as documented by Cushing (1988), much of the northwestern European politics and warfare can be explained, at least partly, by their erstwhile dominance of fishing grounds for lucrative small pelagic fishes. This conflict over small pelagic fish was similar in other parts of the world, with colonial authorities deriving a sizeable part of their income from the salt monopolies they imposed on fishers in India (hence Ghandi's early activism on direct access to salt) or Indonesia (Butcher 1996, 2004).

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Indeed, it is in Southeast Asia that we currently find the tightest coupling of a strong dependence of both rural and urban population on small pelagic fishes and sophisticated method for their cost-effective preservation, i.e., fermentation (Ruddle and Ishige, 2005). This yields the highly nutritious *noc mam*, or *nam pla* alluded to above, along with their equivalent in other countries, for example *patis* in the Philippines (Ruddle, 1986, Pauly, 1996).

These products, along with their equivalents in other regions of the world, attest to the deep historical embeddedness of small pelagic fishes in the diet of people, and expose as convenient untruth the notion of some western fisheries managers and aquaculture advocates that the small fish used for 'reduction' to fishmeal and fish oil are not palatable to people.

FORAGE FISH CONSUMED BY HUMANS

The species of small to medium pelagic fish that are eaten vary between geographic regions and reflect historical and current taste preferences. While some medium pelagic fish such as mackerels are also consumed by humans, small pelagics such as sardines and herrings make up the bulk of fish consumed by humans and the focus of this chapter. Indeed some small species of fish that are only recently being exploited are fished both for human consumption and fishmeal (e.g. Blue whiting).

Small pelagic fish are relatively easy to preserve (see above). Also, these fish, due to their schooling habits, are easy to catch using small mesh nets with low operating costs. This is especially true for the fuel cost, which is much lower than for other fisheries such as line and trawl (Tyedmers *et al.,* 2005, and Chapter 1). Consequently, the fish are often much cheaper and more accessible than demersal fish. In countries where the demand for animal protein (or for cheap protein) is not met domestically, imports of small pelagics such as herrings, sardines and mackerel are often used to meet that demand, e.g., in Ghana, Togo or Nigeria (Alder and Sumaila, 2005). Similarly, in the Caribbean, Oceania and Latin America, imports of small pelagic fish as frozen or canned product are common (PFA, 2006).

Table 1 shows that almost all the small fish currently dominating reduction fisheries are (or were) eaten by people. This applies even to the Peruvian anchoveta and associated small fishes which were consumed by humans, until the development of the reduction fisheries in Peru in 1953 (Chapter 1). The increased demand for small pelagic fish for the fishmeal plants resulted in a decline in human consumption of these fish.

Table 1. Major forage fish species and human consumption

Fish	Human consumption	Reference
Peruvian anchoveta	Primarily canned for human consumption;	www.oceansalive.org
Capelin	Roe is used for human consumption; some is frozen for specific markets in Japan and Europe, market is increasing; 100,000 t of 633,000 t landed in 2004/05 was sold for human consumption;	Nortrade (2006)
Japanese pilchard	Approximately 33% of landings are destined for human consumption, primarily in Japan where it was consumed for centuries;	Chapter 1
Chilean jack mackerel	Historically used for human consumption as frozen or canned product sold in Latin America, Africa and Oceania but recently used in fishmeal;	Chapter 1
South American pilchard	Fresh, frozen and canned product is sold for human consumption in Latin America, North America and Europe;	FIS (2006)
Gulf and Atlantic menhaden	None is consumed directly; some is sold as a food oil/emulsifier;	Huss *et al.* (2003)
Sandeels	A small amount is sold for human consumption;	Nortrade, (2006)
European sprat	Small market for canning and smoking in Europe;	ICES (2005)
Norway pout	No human consumption;	Huss *et al.* (2003)
Northern blue whiting	Small market for human consumption due to processing problems not palatability; some of the landings are used in the production of surimi;	Min of Ag, Fish and Food (1980); Anon. (2004)
Round sardinella	Frozen exports to Africa, Asia & Eastern Europe for human consumption;	PFA (2005)
Atlantic horse mackerel	Historically targeted for reduction but in the 1970s some landings were used for human consumption and recently larger fish are targeted for consumption but much of it is exported to Eastern Europe and Asia;	ICES (2005)
Herring	Historically as fresh, salted or canned product for human consumption, but until recently increasing trend for its use in fishmeal, recently trend reversed with Nortrade processing 75% of its landings for human consumption in 2004/05 up from 32% in 2000/01;	Nortrade (2006)
Chub mackerel	Fresh or canned fish consumed in California and Mexico;	Anon. (2000)
Japanese anchovy	Consumed as fresh and salted products;	Zhang (2002)
European anchovy	Consumed as fresh, dried, smoked, canned or frozen product.	Eurostat (2005)

In Europe, prior to World War II, herring was also destined for human consumption (see above). However, now at least 50% of herring landings are destined for reduction (see Chapter 3), even though the European Union prohibits the processing of Atlantic herring for fishmeal by its member countries (Huss *et al.*, 2003). This is possible because Norway and Iceland are not part of the EU and not bound by the EU decision regarding herring for human consumption.

In the following, we present an analysis of the conflicting uses of small pelagic fishes, based on a database described below.

METHODS

The tonnage of small pelagic fish that are consumed by humans was estimated based on food consumption, catch and landing statistics, and the imports and exports of large pelagic fish such as tunas and billfish. Initially, two different FAO databases were used, the FAO 'food balance sheets' (FAO, 2006) and the Fisheries Commodities Production and Trade database, which covers the years 1976-2003 (FAO, 2003). The food balance sheet dataset provided information on the amount of 'pelagics' (including large fish such as tunas, billfish and swordfish) consumed annually by humans. To adjust consumption to only forage fish as described in Chapters 1 (i.e., to many of the species listed in Table 1), the net balance of large pelagic fish (i.e., catch + imports – exports – re-exports) was subtracted from total pelagic fish consumption.

Initial attempts were made to estimate consumption this way after converting the commodities to live weight equivalents on a country level, then region level. However, the estimates were not reliable, even including several cases with negative consumption values, due to discrepancies in the databases. Thus, the estimates of human consumption were based on the food balance sheets described above, adjusted for regional estimates of large pelagic from the *Sea Around Us* Project landings database (Watson *et al.*, 2004). For this, it was assumed that most landings of tuna were eventually consumed within the regions where they were caught, with the exception of Oceania where a considerable portion of the catch is caught by distant water fleets. Here, it was assumed that 75% of the landings were exported to the other regions (25% to North America, Europe and Asia respectively) since each region has a significant distant water fleet (e.g., US, France, Japan, Korea and Taiwan) in Oceania.

RESULTS

Global landing trends

Forage fish as a proportion of global marine landings and all pelagic landings fluctuated considerably between 1961 and 2002 (Figure 1). However, there was an increase in the late 1980s, which appears to be reversing since the late 1990s. The percentage of forage fish catch that is consumed as food directly has fluctuated between 10 and 20% of global landings since 1961. Much of the variation is a reflection of the environmentally induced variation in landings of small pelagic fish.

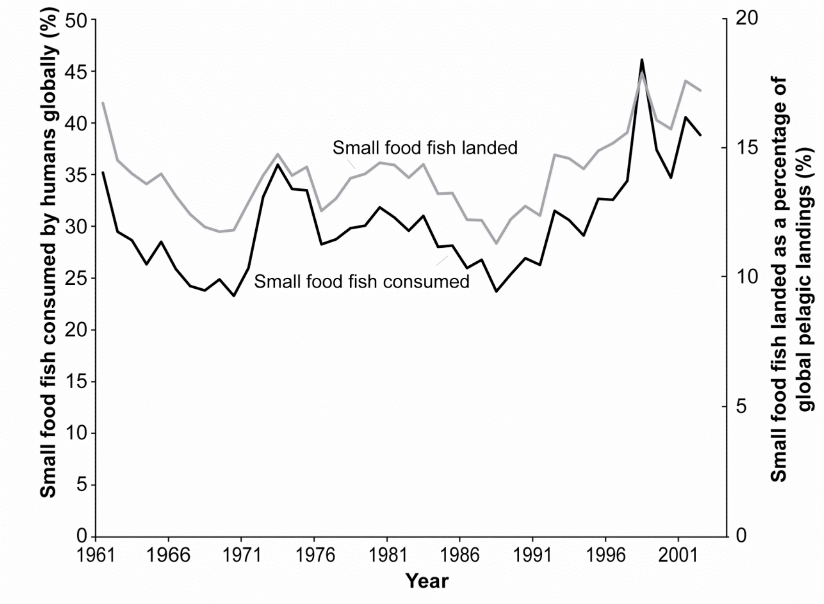
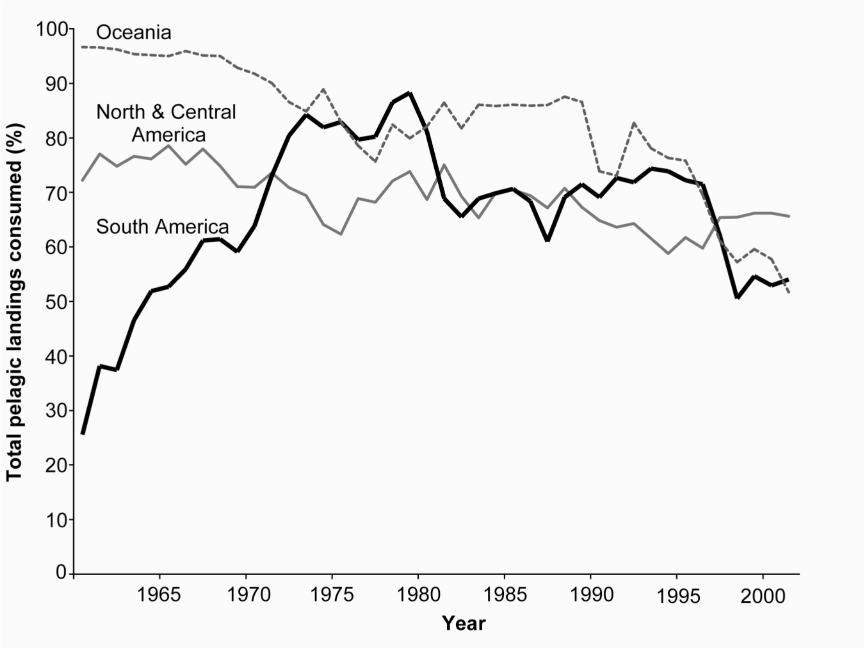


Figure 1. Forage fish consumed by humans as a percentage of global landings (left axis) and as a percentage of global pelagic landings (right axis). (FAO, 2006; *Sea Around Us* Project, 2006).

Regional landing trends

There is considerable variation between continents on consumption trends of forage fish consumed by humans as a percentage of all pelagic fish consumed as food (Figure 2). In Africa, the percentage of small pelagic fish that are consumed by humans has remained constant since the early 1960s, when many African countries became independent. In Europe, there was a decline during the same period; this lasted until the early 1990s, increasing since. For Asia, North and Central America, and particularly in South America and Oceania, the percentage of forage fish consumption has declined. In Latin America, the decline commenced in the mid-1970s, coinciding with a major decline in the small pelagic fisheries along the west coast of South America.

a)



b)

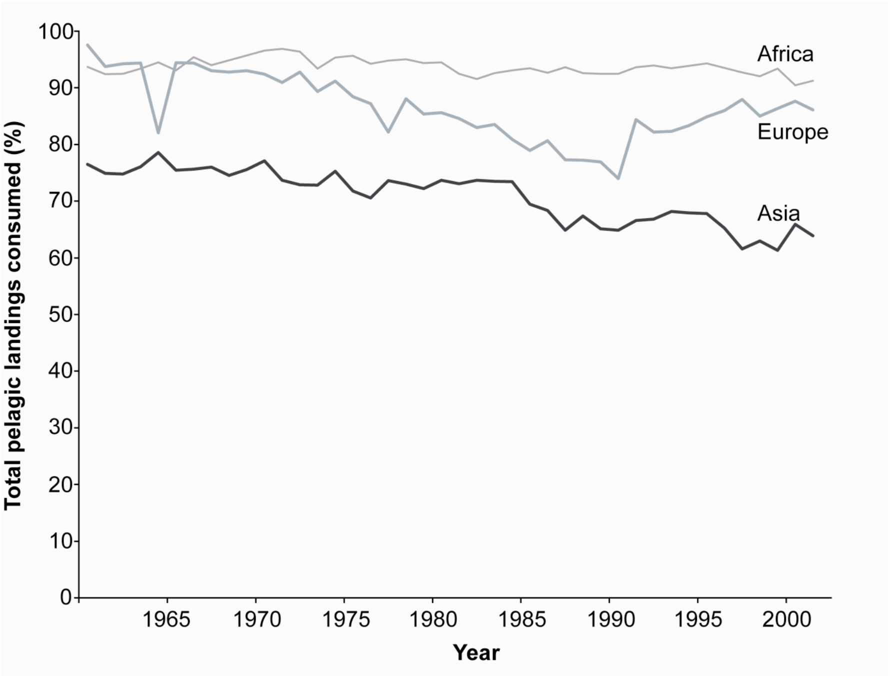


Figure 2a and b. Trends in forage fish as a percentage of total pelagic consumption on a regional basis, adjusted for the high volume of large pelagic fish caught by distant water fleets from North America, Europe and Asia. (FAOSTAT, 2005; *Sea Around Us* Project, 2006).

In Oceania, the tonnage of forage fish consumed by humans is small relative to the commercial landing, and also in comparison to fish consumed in Asia and Africa (Figure 3). Also, Africa and Europe increased their share of forage fish consumption, while the consumption in Asia, North America and South America declined since the 1990s.

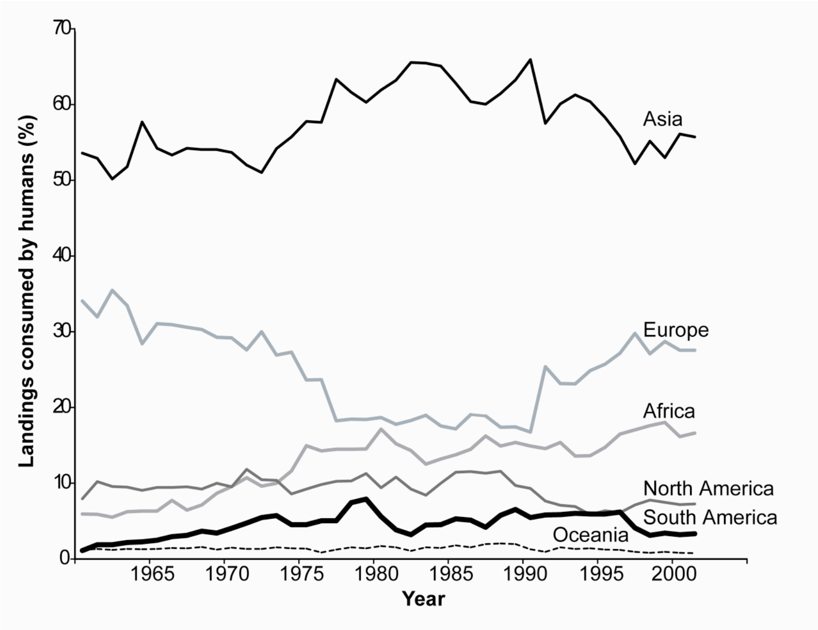


Figure 3. Trends in regional forage fish consumption as a percentage of global forage fish consumption from 1961 to 2002. (FAO, 2006; Sea *Around Us* Project, 2006).

Consumption of forage fish as a proportion of small and medium pelagic landings varies across continents (Figures 4 and 5). Africa's consumption of forage fish increased from 1960 to the early 1990s and now fluctuates around 80% of total small and medium pelagic landings in the area. In Europe and Asia, consumption has been increasing since the mid 1960s, while in Latin America this declined since the mid 1970s.

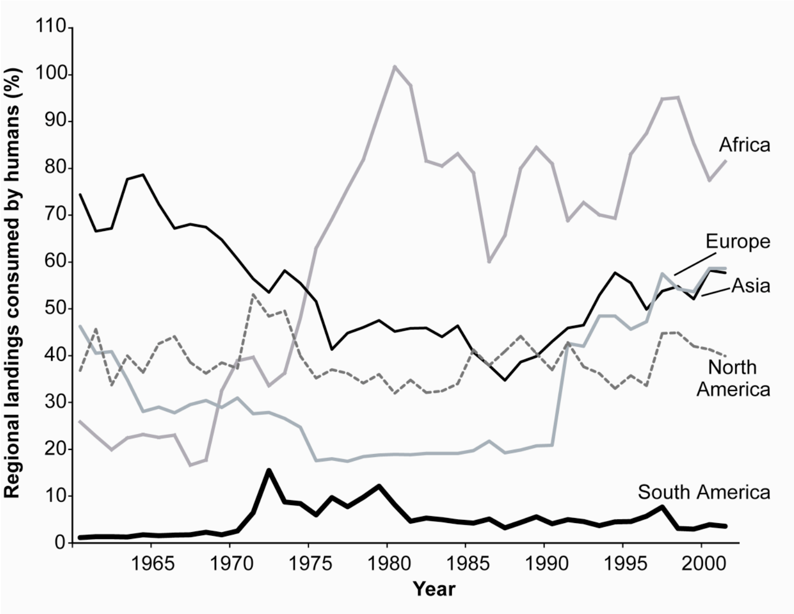


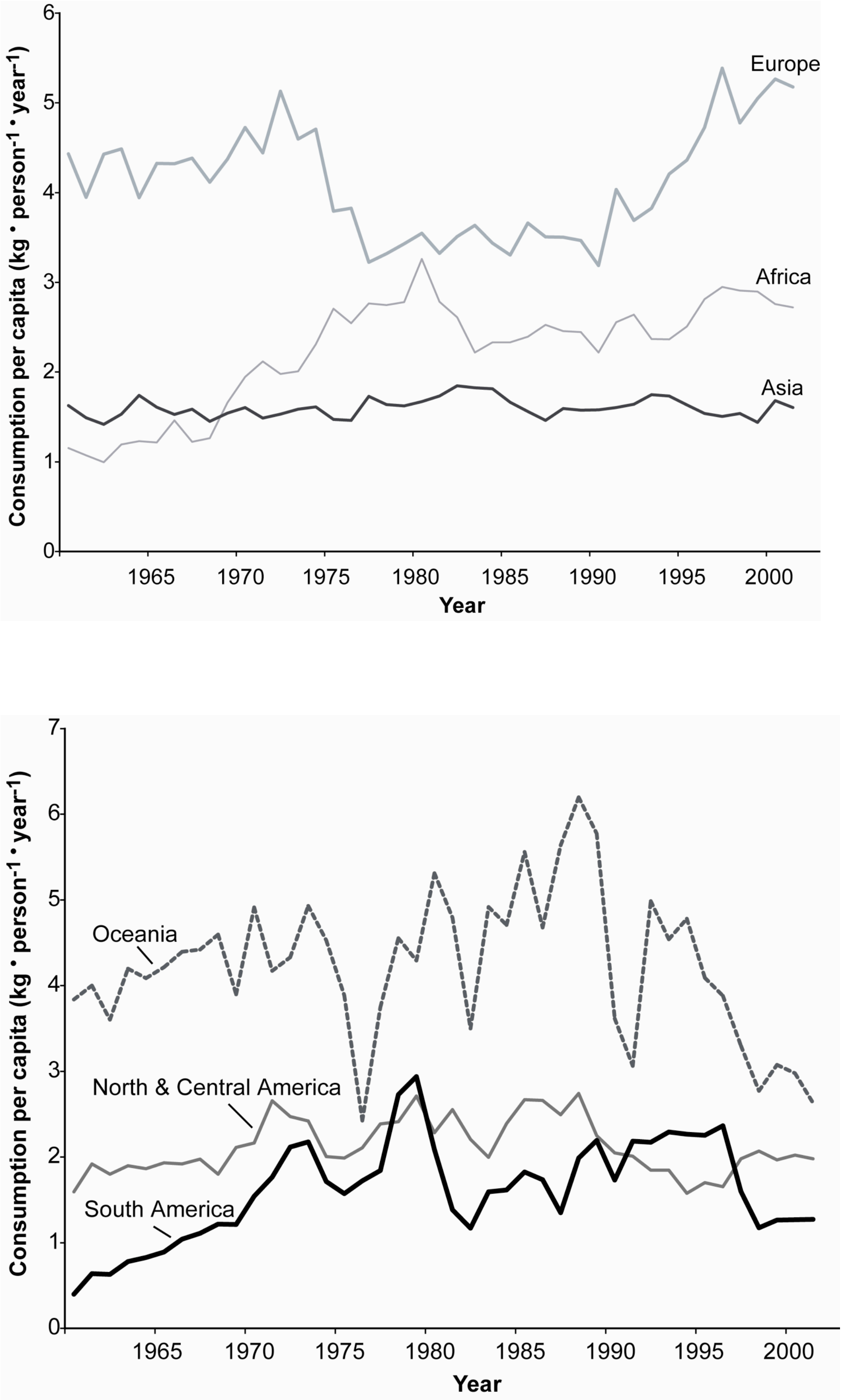
Figure 4. Landings of forage fish as a percentage of regional forage fish landings (adjusted for distant water fleets in Oceania) from 1961 to 2002. Consumption of forage fish in Oceania was combined with Asia due to the relatively small amounts that are consumed in the region. Values greater than 100% indicate that landings were supplemented with imports to meet the demand for low-value food fish. (FAO, 2006; *Sea Around Us* Project, 2006).

Figure 5. Landings of forage fish as a percentage of total small pelagic fish landings in Oceania from 1961 to 2002. Note that percentage can be greater than 100 due to extremely low landings of small fish and large imports of processed small fish, such as tinned mackerel (FAO, 2006; *Sea Around Us* Project, 2006).

Per capita consumption trends

The trend in per capita consumption of forage fish since 1961 varies with each continent (Figure 6). In Africa and Oceania, where these fish play an important role in food security, per capita consumption declined since the mid-1980s in Oceania, and late 1990s in Africa. There has been an overall decline in per capita consumption in South America since the late 1970s. In Asia, where these fish are also important for food security, per capita consumption has remained steady. In Europe, per capita consumption increased since the late 1980s (Figure 7), while in North and Central America, it has declined.

a)



b)

Figure 6a and b. Per capita consumption of small fish 1961 to 2002 (FAO, 2005).

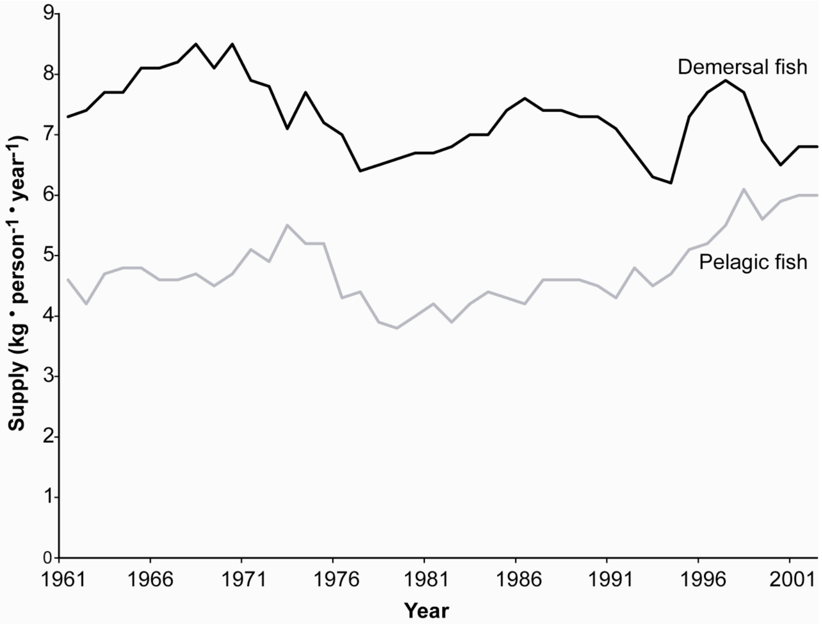


Figure 7. Annual supply of demersal and pelagic fish in Europe since 1961 (FAOSTAT, 2005).

DISCUSSION

A recent study by Delgado *et al.* (2003) examined trends in "low value food fish", defined as a mixture of unspecified marine fish, freshwater fish and pelagic marine fish. Although this definition makes direct comparisons with this study difficult, some of their results are worth noting. Low-value food fish as a proportion of total fish consumed by humans dropped from 76% in 1973 to 65% in 1997. However, if China is excluded (which is justified by the unreliability of their statistics; see Watson and Pauly, 2001), the decline is much less: 77% in 1973 to 72% in 1997. Globally low-valued food fish increased from 41% to 47% of food fish consumed for the same time period (Delgado *et al.,* 2003). The study, however, noted that the rise in consumption was due in part to the poor in Asia (especially China) increasing their consumption of farmed freshwater fish (Delgado *et al.,* 2003). These authors also concluded that net imports of low-value food fish to the developing world totalled 2.3 million t in 1997.

Nevertheless, more recently, there has been a steady increase in the percentage of marine landings (total and as a percentage of small pelagic) that are consumed by people (Figure 1), although this differs between continents, both for volume and per capita consumption (Figures 2, 3, 4, 5 and 6).

Small pelagic fish populations are highly sensitive to oceanographic variability (Bakun, 1996) and, therefore, landings can vary significantly from one year to the next. However, this has limited impact on worldwide consumption patterns, except when the events are extreme, as in the case of the El Niño events of 1972-1973, 1983, and in the 1990s.

Indeed, Figure 2 in Chapter 1 shows the decline of small pelagic landings in Peru in the early 1970s. As a consequence of a strong El Niño, consumption changed, but not in proportion to the decline (Figure 1, this chapter). There are a number of reasons for this, beyond prices, which account for the spatial differences and fluctuations in human consumption of small pelagic food fish. They are:

- *Increasing wealth* in some countries, resulting in a switch to higher valued fish such as cods and haddock, and large pelagics such as tuna and billfish;
- *Substitution* of small pelagic fish when there is limited availability of demersal and large pelagic fish;
- *Increasing competition* for small pelagic fish for fishmeal and for human consumption, driving the price of these pelagic fish up and making it difficult for poorer countries to purchase the fish; and
- Soybean price fluctuations, due to soymeal being used as a substitute for fishmeal in some industries.

We address in the following each of these four factors.

Increasing wealth: It has been noted that for countries such as China, where incomes are rising in major urban areas, there has been an increase in the demand for higher valued fish, including demersal species such as 'red fish' as well as for the larger, higher valued tunas and swordfish (FAO, 2005). The increased consumption of large pelagic fish in China may in part account for the decline in the Chinese share of small pelagic fish consumption in Asia, especially since China has recently expanded its distant water fleet with tuna as one of its target species (Pang and Pauly, 2001). A recent study of fish supply and demand to 2020 noted that developing countries still have great latitude for increasing demand for low-value food fish, but suggested that 'upgrading' is occurring in the wealthier segments of the developing world. Some of this will be realized via high valued farmed fish, further fuelling the demand for fish for feed/reduction to fishmeal (Delgado *et al.,* 2003).

In North America, consumption of low-value food fish is low compared to Asia and Africa (Figures 1, 3 and 5). Increasing wealth among North Americans may account for declining consumption of these low value food fish. Paradoxically, the increasing awareness of the health benefits of eating fish has not increased the consumption of healthy small pelagics, but of high-end products, notably farmed salmon, which are fed fishmeal and oil derived from small pelagics. As a result, overall seafood consumption rates in North America have shown a slight increase since the mid-1980s (Delgado *et al.,* 2003).

Substitution: Replacement of small pelagic fish for high-valued demersal fish such as cod and haddock occurs in Europe, where consumption of small pelagic fish declined from 1961 to the late 1980s, with a subsequent increase since then (Figures 1, 3, 4, 5, and 6). This may be explained by the supply of higher-valued demersal fish declining, and small pelagics meeting the overall increasing demand for fish (Figure 7). The expansion of fish species such as salmon, once considered high-value, has resulted in a lowering of market prices, thus enabling consumers to substitute affordably priced farmed salmon for cheap, small fish (Asche and Tveterås, 2005).

Increasing competition and price: Asche and Bjorndal (1999) suggested that demand for food fish is price-elastic, and that real price rises will cut into fish consumption on average in developing countries. Also, they suggest that the cut into consumption would be stronger for lower-income groups. Delgado *et al.* (2003), in their modeling of fish supply and demand to 2020, noted that a cross-price elasticity of 0.3 implies a 1% rise in poultry prices. This, other things being equal, implies a 0.3 % rise in fish demand. This appears to hold for small pelagic food fish especially in Oceania, Asia, Africa and South America.

In Asia, more than 50% of small pelagic low-value food fish landed are consumed, but the percentage of these fish that are consumed has declined recently (Figures 1 and 3). While the amount of fish consumed is high, on a per capita basis, it is less than for many other areas such as Oceania and Africa (Figure 6). The overall trend is a decline in the consumption of these fish (Figures 1 and 3), despite an overall increasing consumption of the percentage of landings (Figure 4) and a constant per capita consumption rate (Figure 6). Some of this decline in consumption may be due to a shift to higher value species, as a result of increasing wealth in some segments of the population, as discussed above. Some of this decline may also be due to diverting some fish such as bycatch and fish suitable for human consumption to the aquaculture sector (Kelleher, 2005).

The impact of changes in demand for small pelagic fish for reduction or human consumption is well-illustrated by Africa. There, incomes have not increased, and small pelagic fish are an important food source, highly sensitive to price changes. The consumption of small pelagic fish as a percentage of the total pelagics landed in Africa has fluctuated little since the 1990s (Figure 1), but is increasing as a percentage of the global human consumption of small pelagic fish (Figure 3). More recently, Africa has increased its global share of these low-value food fish from less than 10% in 1961 to almost 20% in 2000. The percentage of small pelagics landings in Africa that are consumed increased from 1961 (25%) to the early 1980s, then declined until the late 1980s. It now fluctuates around 80% of total African landings. However, the overall trend in per capita consumption of small pelagics, while fluctuating, is positive over time (Figure 6). These fluctuations in consumption are, in part, a reflection of changes in price and supply on the world market, with the increases or decreases of the landings of Peru and Chile having the strongest impact on world supplies and prices.

In South America, consumption of small pelagic food fish as a percentage of total pelagic landings increased rapidly until the early 1970s, coinciding with the increased landings of small pelagic fish in the region (Figures 1 and 4). It then declined significantly in the 1970s, in the aftermath of the first collapse of the anchoveta fishery. Since then, consumption has fluctuated with an overall declining trend. Similarly, since the late 1970s, there has been an overall decline in South America's consumption of global landings of small pelagic food fish (Figure 3). Per capita consumption of these low-value food fish in South America follows the same increasing trend until the 1970s and early 1980s, with a subsequent decline, accelerating in the last five years (Figure 6).

In South America, especially Chile, salmon aquaculture is expanding. As a result, and given that catches of small pelagics and hence fishmeal/oil production in Chile and Peru have not increased, exports from the region have remained roughly constant, while 'internal' consumption increased (in the aquaculture sector). Concern was raised in the region over food security issues since many of the species, such as mackerels, used for fishmeal are also consumed by local residents (Table 1). In Peru, the government has restricted industrial fisheries (most of it destined for fishmeal) from fishing for some small pelagic species and in certain areas to ensure there is an affordable supply of fish for human consumption (FIS, 2003).

Historically, the supply and demand of small fish for human consumption was a simple model of fishers catching the fish, processing and preserving the products and selling the fish to local consumers or buyers for export to other countries. However, with the expansion of reduction plants in the 1960s, this model became more complex, with fish destined for human consumption competing with several other consumers of fishmeal and fish oil, such as intensive animal producers, especially for pigs and poultry, aquaculture producers, especially for fish oil, the pharmaceutical industry and, more recently, the pet food industry. There is general agreement that there is no scope for increasing the exploitation rates of these small pelagic stocks. Rather, there are limits to the potential expansion of intensive animal production, including aquaculture, which compete with human consumption.

Soymeal can be used as a substitute for fishmeal in intensive animal production, especially in the pig and poultry sectors (Chapter 4). Indeed, in 1996/97, fishmeal accounted for 4% of the world's protein meals supply, while soymeal accounted for 53% (Asche and Tveterås, 2005). Soymeal can also substitute for fishmeal in aquaculture, although the fish cultured do not achieve the same high growth rates (Durand, 1998). Fishmeal is also a small part of a large protein meal market and a close substitute for soymeal. Therefore, it is the total supply and demand for protein meals that determine price (Asche and Tveterås, 2005). If the total demand for protein meals increases, the price for protein meals will increase, including the price of fishmeal. This will also increase the likelihood of small pelagic fish destined for human consumption being diverted to the reduction sector. This was the case in Peru in 2003, as described previously.

CONCLUSION

Forage fish have been and still are consumed by humans as part of a larger suite of fish that were historically consumed, and humans are finding new ways to process and make palatable small fish that were previously shunned or undiscovered by consumers. There are some areas in the world where consumption of these fish is increasing, especially in the developing world, where traditional stocks are depleted as well as other sources of protein being too expensive or difficult to buy. In other areas of the world, especially in developed countries, consumption is declining as consumers substitute these fish for more affordable farmed fish. This is only possible because the small pelagics that are used as feed are extremely cheap on the world market. Changes in consumption patterns vary, but the underlying main driver of change in the last 20 years or more has been the price of forage fish, which risks the food security of people relying on cheap fish protein. The future of balancing food security and the development of an aquaculture sector relies on a number of factors, all controlled by global events such as the price of soy beans, technological developments in the aquafeed industry, climate change and world oil prices.

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CHAPTER 3

FORAGE FISH CONSUMPTION BY MARINE MAMMALS AND SEABIRDS[3]

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ABSTRACT

This contribution presents estimates of the amounts and distributions of small pelagic fishes (i.e., 'forage fish') consumed by the 115 species of marine mammals and the 351 species of seabirds known to feed in the world's marine waters. The methods used for mapping the distribution of these predators and of their prey, detailed elsewhere, are briefly recalled. Marine mammals, in the 1990s, annually consumed about 20.2 million t of small pelagics, or roughly 12 % of their food consumption; for seabirds, the corresponding figures, also applying to the 1990s, are 12.1 million t of small pelagics and 12.5 % of their food consumption. Overall, we find that marine mammals and seabirds do not compromise human exploitation of small pelagics. On the other hand, reduced biomasses of small pelagics are a challenge for numerous species of marine mammals and seabirds.

INTRODUCTION

The marine mammals of the world, belonging to 115 species, jointly consume a large amount of food, about four times as much as humans catch (above 100 million t if illegal and misreported catches are included), while the 351 species of seabirds consume about as much as we catch (Kaschner and Pauly, 2005; Brooke, 2004; Karpouzi, 2005).

This study is devoted to investigating how much of this consumption (which, due to the nature of our data, represents an average year of the 1990s) consist(ed) of small pelagic fishes (i.e. 'forage fish'), and thus to what extent humans may compete with marine mammals and seabirds for small pelagic fishes.

METHODS

Marine mammals

Our model encompassed 115 species of marine mammals that live predominantly in the marine environment (Kaschner, 2004), but did not include sirenians, sea otters, the polar bear nor any of the exclusively freshwater cetacean or pinniped species. We used a relatively simple generic model (Trites *et al*., 1997) to generate estimates of forage fish consumption of marine mammal species:

(1) $$Q_i = 365 * \sum_s N_{i,s} W_{i,s} R_{i,s}$$

where the annual food consumption *Q* of forage fish of species *i* was estimated from the pelagic prey consumed daily by each individual of species *i* and sex *s* is calculated based on the number of individuals *N* of the sex *s* of a species *i*, and a weight- and sex-specific daily ration *R*.

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We obtained global abundance of marine mammal species during the 1990s by extracting and combining available regional abundance estimates and information about associated uncertainties from more than 1,000 published primary (e.g., Branch and Butterworth, 2001, Whitehead, 2002, Mullin and Fulling, 2003) and secondary sources (e.g., Reijnders *et al.*, 1993, Ridgeway and Harrison, 1999, Perrin *et al.*, 2002) Mean abundance estimates that were used in this study are available in Appendix 3 (Kaschner, 2004). We assumed sex ratios were balanced for most species, except for those for which available published information explicitly indicated otherwise (e.g., Wickens and York, 1997) or if unequal sex ratios seemed highly likely based on information about closely-related species with similar life history traits. We used the sex-specific mean body mass estimates for each species generated by Trites and Pauly (1998), who estimated female and male body weights averaged across all age classes for marine mammal species, based on the strong relationship between more readily-available maximum length information and species-specific growth rates, survival and longevity.

We calculated daily food ration of small pelagic fish consumed by each species based on the weight-specific energy requirements and information about diet composition. Food intake of specific species was calculated using an empirical model developed by Innes *et al.* (1987) and later modified by Trites *et al.* (1997):

$$(2) \qquad R_{i,s} = 0.1 * W_{i.s}^{0.8} * pSP_{i,s}$$

where R is the daily food intake of forage fish of an individual of sex s belonging to species i, W is the mean body weight of that individual in kilograms, and pSP is the proportion of small pelagic fishes in the species diet as determined by Pauly *et al.* (1998). Annual spatially-explicit food-consumption densities were generated by linking food intake to global distributions of species predicted using a large-scale Relative Environmental Suitability (RES) model (Kaschner *et al.*, 2006). Within a global grid of 0.5 degree latitude x 0.5 degree longitude cells, we then compared predicted marine mammal food-consumption densities with average fisheries catch densities during the 1990s as generated by the *Sea Around Us* Project (Watson *et al.*, 2004) using a modified version of an ecological niche overlap index. This index is derived from or related to the 'competition coefficients' of the Lotka-Volterra equations by Morisita (1959; see also Horn, 1966) and originally only considered the qualitative overlap of resource utilization of two players exploiting the same resources (i.e., the similarity of marine mammal diet and fisheries catch composition), but ignored the absolute amounts of the resource that is being used or consumed. We therefore further modified this index by introducing a weighting factor to provide a measure of the importance of each cell for either fisheries or marine mammals based on overall quantity of catch or food taken by either consumer in this cell, leading to:

$$(3) \qquad \alpha_{j,l} = \left(\frac{2\sum_{k} p_{l,k} p_{j,k}}{\sum_{k} p_{l,k}^2 + p_{j,k}^2} \right) * \left(pQ_l * pC_j \right)$$

where $\alpha_{j,l}$ describes the quantitative overlap between a fishery j and marine mammals l in each cell, and the first term expresses the qualitative similarity in diet/catch composition between marine mammals and fisheries sharing the resource or food type k (in this case: small pelagics) as the ratio of 'niche proximity' to 'niche breadth' (MacArthur and Levins, 1967), with $p_{l,k}$ and $p_{j,k}$ representing the proportions of forage fish in the diet or catch. This term is multiplied by the product of the proportion of global food consumption of small pelagic prey taken by marine mammals (Q) and the total fisheries (C) within this cell. The continuous resource overlap values thus generated were subsequently sub-divided to create a categorical index ranging from low to high.

Seabirds

We compiled information for 351 marine bird species (belonging to four orders and 14 families) in a Microsoft Access database. Of these species, 334 are traditionally considered to be seabirds. The remaining 17 were species of sea ducks, which consist of birds that breed inland but which winter at sea close to shore, and prey upon small fish and invertebrates that occur along the coast. Information was gathered using the following databases: (a) Aquatic Sciences and Fisheries Abstracts (ASFA); (b) Web of

Science, Institute for Scientific Information (ISI); and (c) BioSciences Information Service (BIOSIS) of Biological Abstracts. These cover peer-reviewed journals and grey literature sources. The following online databases were also consulted: (a) Avibase – the world bird database (Anon 2006); (b) The World Conservation Monitoring Centre Species Database (WCMC, 2006); (c) BirdLife International (www.birdlife.net); (d) the National Audubon Society (www.audubon.org) Christmas Bird Count (http://www.audubon.org/bird/cbc/index.html); (e) the Birds of North America Online (http://bna.birds.cornell.edu/BNA/); and (f) Wetlands International (www.wetlands.org).

Information required to estimate the seabirds' daily food intake (DFI), and hence their annual food consumption, included: (a) body mass (m; in g) of seabird species taken from Dunning (1993) and Schreiber and Burger (2002); (b) Basal and Field Metabolic Rates [BMR and FMR respectively, estimated using order-specific allometric equations from Ellis & Gabrielsen (2002); in kJ/day]. BMR and FMR were used to estimate energy requirements (ER) of seabirds in the non-breeding and breeding season respectively; (c) A matrix of standardized diet composition and; (d) population sizes of breeding seabirds.

Daily food intake (DFI) was estimated using a bioenergetic model created by the ICES Working Group on Seabird Ecology (ICES, 2000):

$$\text{(4)} \qquad \mathrm{DFI_i} = \frac{\mathrm{ER_i}}{\sum_{j=1}^{G} \mathrm{DC_{ij}} * \mathrm{ED_j}} * \frac{1}{\mathrm{AE_i}}$$

where DFI_i denotes daily food intake for each seabird species i, ER_i is the energy requirements for each i, DC_{ij} is the fraction of food item j in the diet of each i, and ED_j is the mean energy density of each prey j. ED_j values were available either at the species or the genus level for prey items. AE_i is the mean food assimilation efficiency for each i, and G the total number of food groups encountered in the diet of each i. AE_i was assumed equal to 75% (Gabrielsen, 1994; ICES, 2000; Barrett *et al.*, 2002), unless species-specific information was found in the literature (see Karpouzi, 2005).

In order to estimate seabird species' annual food consumption, DFI_i was estimated separately for the breeding and non-breeding season, by considering ER_i for the breeding season equal to *FMR* (ICES, 2000), and for the non-breeding season equal to $2.5 \cdot BMR$ (ICES,2000). The length of the breeding season was assumed equal to incubation period + time from hatching to leaving the nest or burrow + 20 days (Cramp 1985).

Consumption by seabirds was specified using 25 food groups. Six out of 25 food groups represented the forage fish groups considered here (i.e., Capelin, *Mallotus villosus*; Exocoetidae; Engraulidae; Clupeidae; Osmeridae; Sandlance, *Ammodytes* spp.). Food groups were compiled based on the taxonomic groups represented in the *Sea Around Us* Project database. In order to map total food consumption by seabirds, we used a GIS-based modeling approach and the same spatial grid of 30-min cells developed by the *Sea Around Us* Project (Watson *et al.*, 2004; www.seaaroundus.org). Total food consumption by seabirds was estimated per spatial cell, based on the seabird density of each cell (see details in Karpouzi, 2005).

RESULTS

Spatially explicit consumption of forage fish by marine mammals

We estimated that food intake of forage fish by all marine mammal species combined was about two-thirds of the amount of small pelagics taken by fisheries in the 1990s (Figure 1). While small pelagics represent the single most important prey type targeted by fisheries, contributing over 50 % of the total catch, this food type makes up – at the most – 20% of the diet of any marine mammal species group. Baleen whales and pinnipeds consume the bulk of small pelagics consumed by marine mammals, although which of the two species groups takes the majority of this prey type may vary depending on the type of feeding rate model used to estimate food intake. Toothed whales, in contrast, are much less dependent on forage fish, and this prey type makes up less than 10% of the total amount consumed by both small and large odontocetes (Figure 2).

Spatial disaggregation of fisheries catches in the 1990s showed that the vast majority of reported catches was taken on the continental shelves of Europe, North America, Southeast Asia and the west coast of South America (Figure 3). Fisheries were concentrated in relatively small areas and fishing rates can be extremely high, amounting to more than 1,000 t per km^2 per year in some of the dark red areas shown in Figure 3. Highest catches occurred in areas where continental shelves are wide, such as the East China or North Seas, or in productive upwelling systems, such as those that can be found along the west coasts of South America and South Africa. However, despite the many distant water fleets and the development of deep-sea fisheries operating far offshore, major fishing grounds generally lay in close proximity to areas with high human populations along the coasts of major industrial fishing nations.

We predicted most of the forage fish that marine mammals consume to be taken in polar waters, with particularly high concentrations along the continental shelves in the North Atlantic (Figure 4). Due to the sheer size of the distributional ranges of many of the baleen and larger toothed whale species, consumption densities (annual food intake per km^2) are comparatively low and homogenous across large areas (Figure 4). Areas of highest forage fish consumption are closely linked with pinniped occurrence, since the generally more restricted, coastal ranges of this species group – combined with high abundances and a preference for small pelagic prey types – resulted in locally concentrated feeding densities. However, predicted maximum food consumption densities did not exceed 0.75 t per km^2 per year anywhere in the world, i.e., maximum food intake of small pelagic prey by marine mammals is several orders of magnitude lower than the highest fisheries catch rates.

Overall, the model predicted low overlap in resource exploitation between all marine mammals and fisheries in the 1990s (Figure 5). High overlap appeared to be restricted to small geographical regions and was mostly concentrated in temperate continental shelf areas of the northern hemisphere and the highly productive upwelling systems in the southern hemisphere. Highest overlap was predicted in areas where high fishing effort coincided with high densities of seals such as the North Atlantic shelves (harp, hooded, harbour and grey seals) or the Benguela system (South African fur seals) or along the coast of western South America where productive upwelling systems support a wide range of marine mammal species and fisheries. Although only a few pinniped species occur in the waters around Japan, high overlap in this region can be attributed to the large number of dolphins and some baleen whale species feeding on small pelagics, combined with very high fishing rates in these waters.

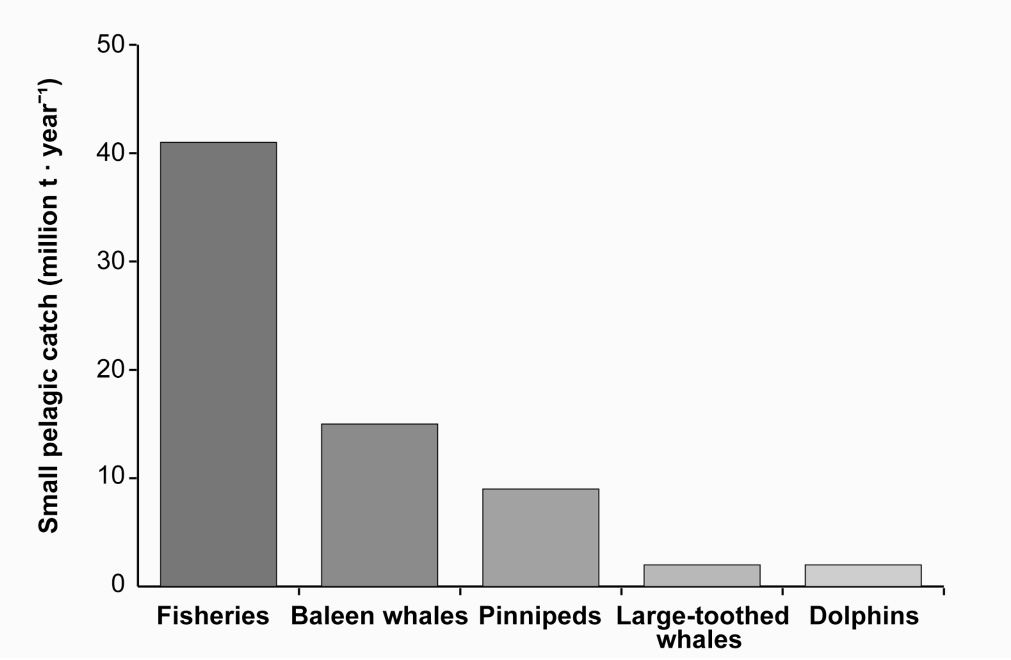

Figure 1. Mean estimated annual consumption of small pelagics by marine mammals in comparison to fisheries catches in the 1990s.

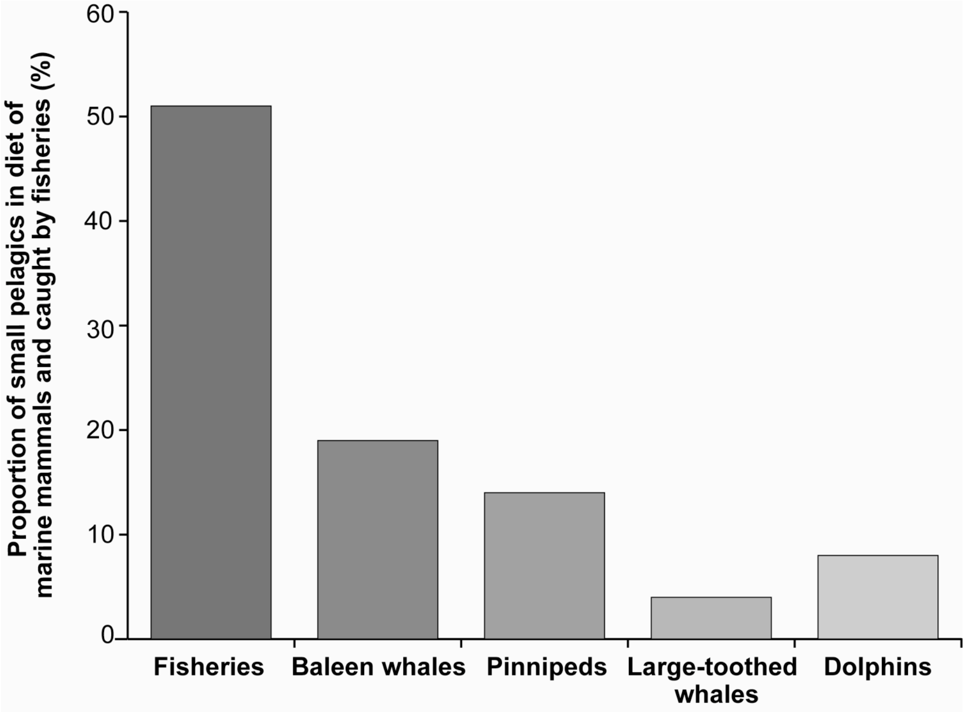

Figure 2. Mean estimated annual proportion of small pelagics in the diet of marine mammals and in fisheries catches during the 1990s.

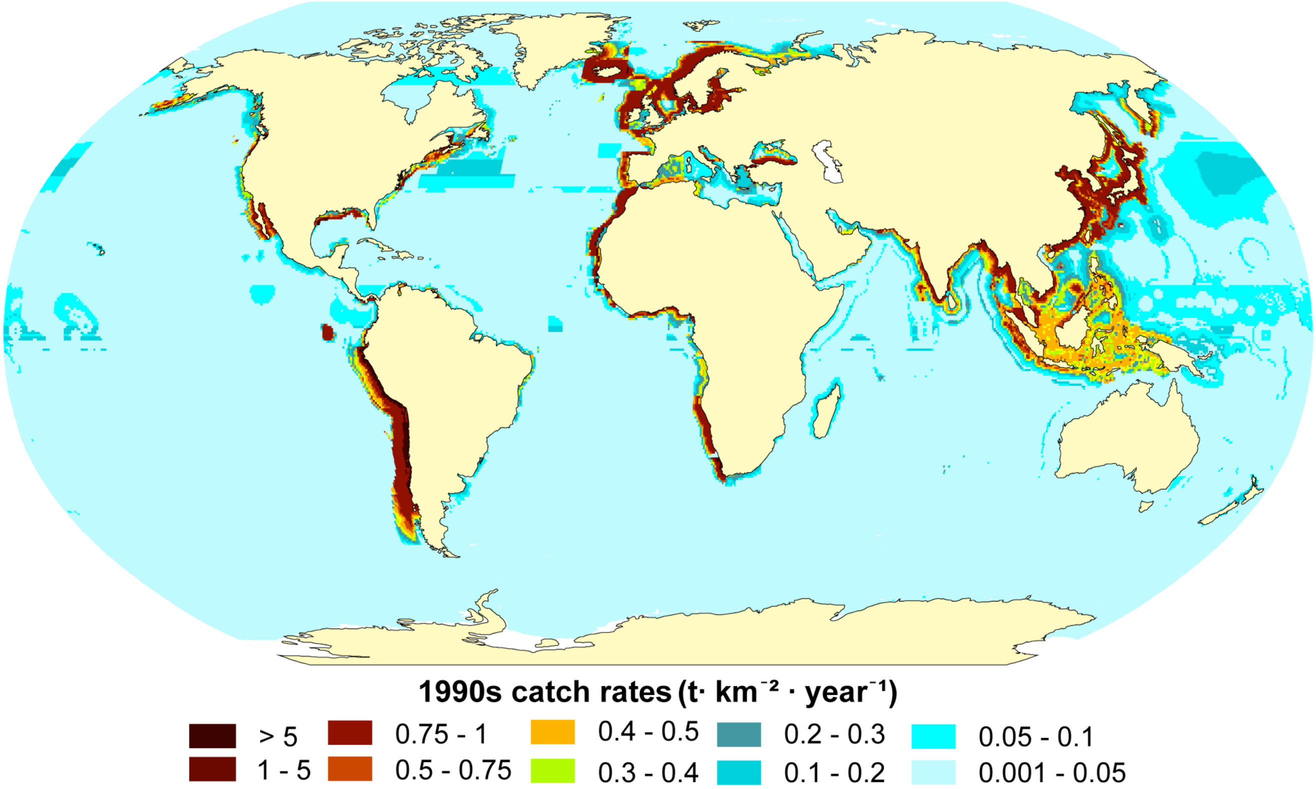

Figure 3. Distribution of fisheries catches of small pelagics for an average year in the 1990s. Note open-ended scale of legend, which indicates that fisheries catch rates can reach up to 150 000 $t\cdot km^{-2}\cdot year^{-1}$ in some areas.

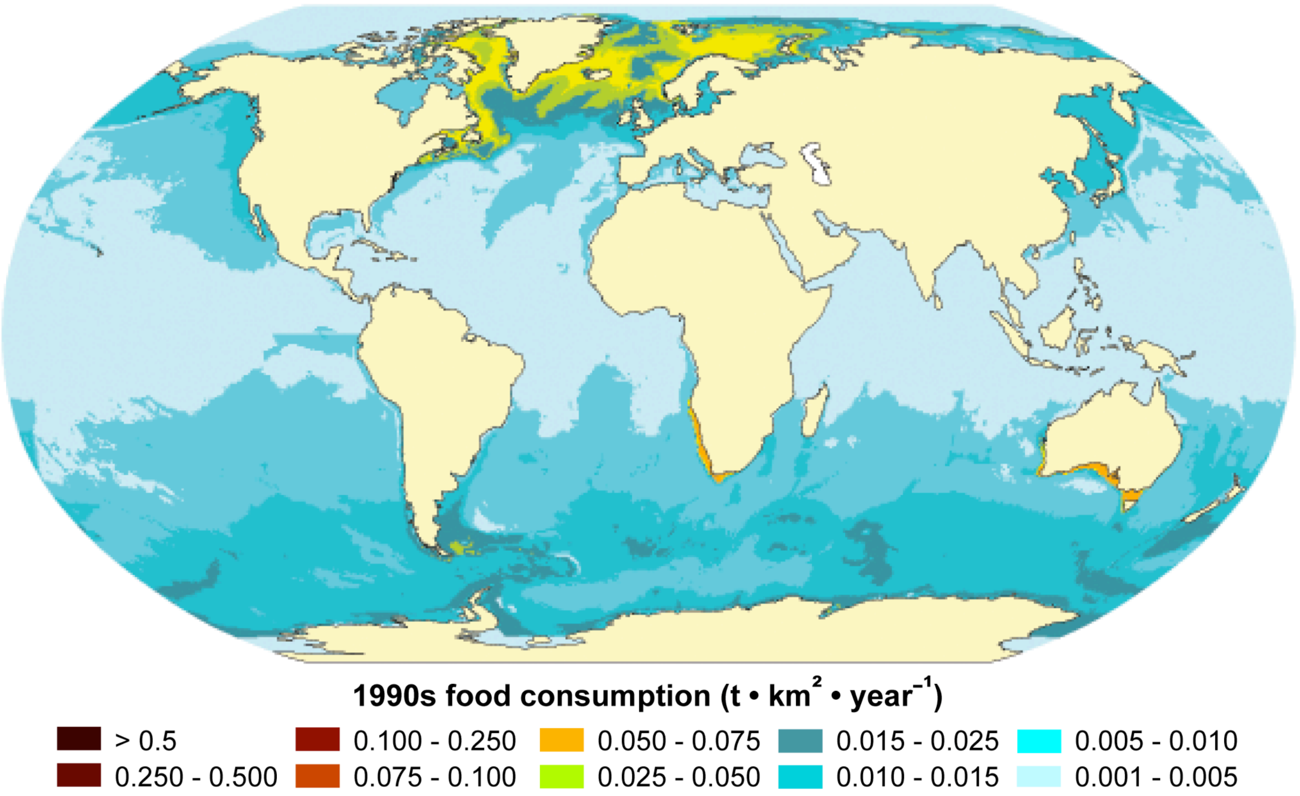

Figure 4. Distribution of estimated marine mammal food consumption rates (t·km^{-2}·year^{-1}) of small pelagics for an average year in the 1990s.

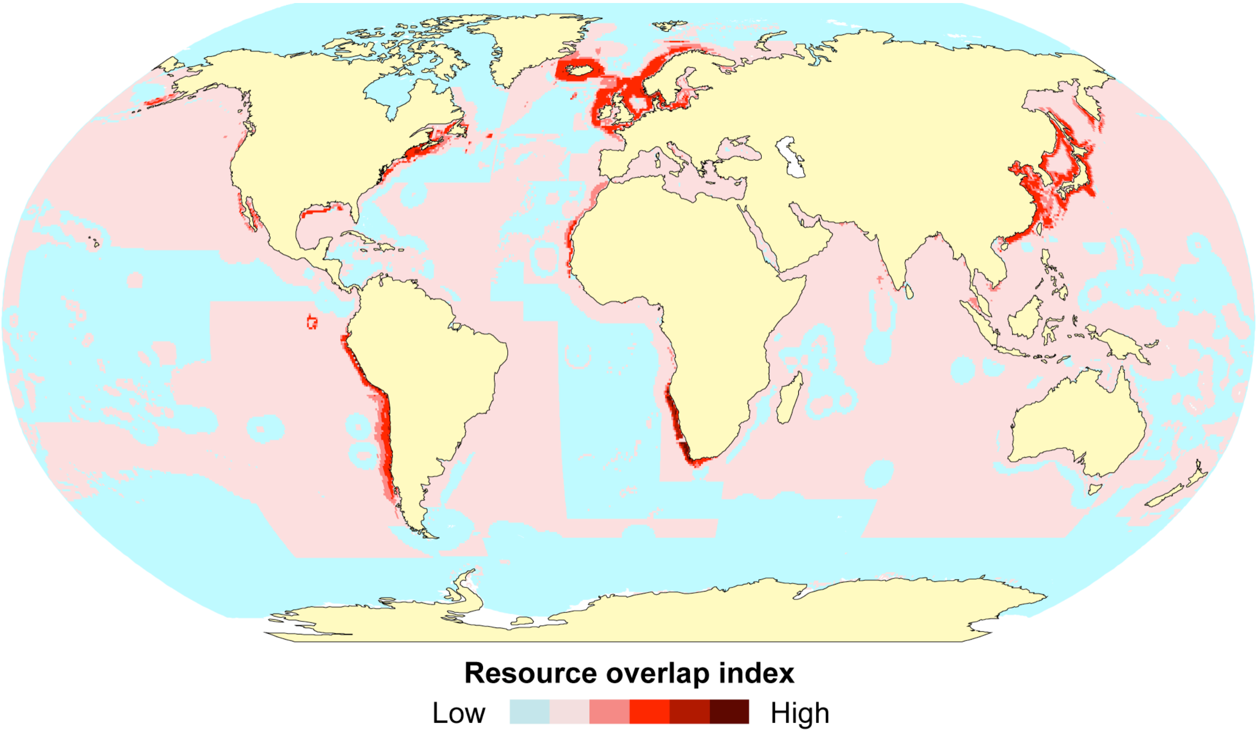

Figure 5: Map of estimated overlap in resource exploitation of small pelagics by marine mammals and fisheries for an average year in the 1990s.

Spatially explicit consumption of forage fish by seabirds

We estimated that food intake of forage fish by all seabird species combined was at least eight times lower than what is taken by fisheries in the 1990s (Table 1). Small pelagics comprised about 12.5% of the overall food consumed annually by the world's seabirds. Alcids (puffins and murres) and larids (gulls) are responsible for about 75% of small pelagic fish consumption by all seabird species combined.

Table 1. Small pelagic fish (in t) taken annually by seabirds from different FAO statistical areas.

FAO Area	Capelin	Exocoetidae	Engraulidae	Clupeidae	Osmeridae	Ammodytes	All
18	31,289	-	-	1,716	-	62,621	95,625
21	207,258	-	144	22,722	86	119,055	349,265
27	2,556,958	-	42	657,668	22	3,071,392	6,286,082
31	3,457	-	5,282	2,439	31	5,809	17,018
34	-	-	2,241	812	-	-	3,053
37	-	-	234	24,843	-	-	25,077
41	-	-	20,888	17	-	-	20,905
47	-	1	7,018	5,927	-	-	12,946
48	-	-	-	-	-	-	-
51	-	13,828	15,322	4,431	-	-	33,582
57	-	4,569	11,288	650	-	-	16,506
58	-	-	-	-	-	-	-
61	5,854	-	43,726	34,554	2,278	172,607	259,019
67	111,008	-	12,044	48,321	28,785	405,430	605,588
71	-	410,265	26,193	3,252	-	2,337	442,048
77	-	2,310,191	174,629	27,784	-	7,328	2,519,931
81	-	40,054	809,532	207,668	-	174	1,057,427
87	-	56,489	249,765	13,059	-	-	319,312
88	-	-	-	-	-	-	-
Total	2,915,824	2,835,396	1,378,348	1,055,864	31,202	3,846,752	12,063,385

We predicted that more than 52% of the forage fish that seabirds consume is extracted over the continental shelves of the Northeast Atlantic Ocean (Figure 6; Table 1). In this area, sandlance and capelin accounted for >54% of the food taken annually by seabirds. The Eastern Central Pacific Ocean (Figure 6; Table 1) was the second most important area, where small pelagics taken by seabirds comprised 21% of the overall forage fish consumption. In this area, forage fish groups in the diet of seabirds were dominated (up to 92%) by fish species of the family Exocoetidae. Areas of highest forage fish consumption were closely linked with the distribution of those seabird species that are limited to waters above continental shelves when foraging (Karpouzi, 2005).

Furthermore, predicted maximum food consumption rates exceeded 10 t · km^2 · year^1 along the continental shelves of the Northeast Atlantic Ocean and around islands of the Western Central Pacific Ocean. This figure is more than ten times larger than what marine mammals were predicted to consume. However, it still remains several orders of magnitude lower than the highest fisheries catch rates.

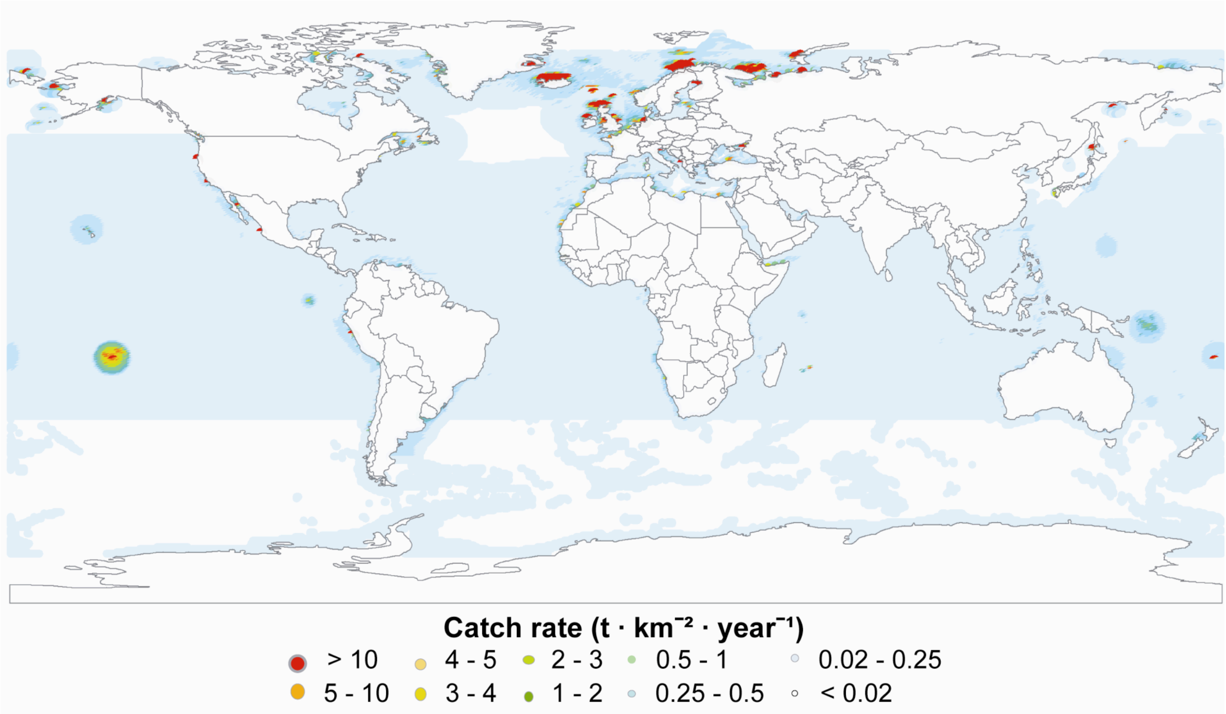

Figure 6. Map of predicted global small pelagic fish consumption rate by all seabirds combined for an average year in the 1990s.

DISCUSSION

Spatially explicit food consumption and overlap with fisheries

Based on the small size of predicted 'hotspots' of potential conflict, in combination with highly concentrated fishing operations and the mobility of many species of marine mammal and seabirds, we suggest that it is unlikely for direct competition to represent a severe threat to species with large foraging ranges. In contrast, our findings support a previously proposed hypothesis that the most common type of harmful competitive interaction will be one in which fisheries adversely impact species with restricted distributional ranges (DeMaster *et al*., 2001; Frederiksen *et al*., 2004), indicating that local depletions of food resources through intensive fisheries may pose serious threats to species such as the Vaquita in the Gulf of California, South Africa's Heaviside's dolphins, the North Sea populations of Black-legged kittiwakes, and also to localized populations of other species.

Quantitative validation of our resource overlap analysis will be difficult to achieve. However, even though resource overlap does not automatically imply competition and *vice versa*, it is reassuring that the 'hotspots' of potential conflict highlighted by our approach coincide with many areas that have been the focal points of much previous debate about marine mammal-fisheries and seabird-fisheries interactions. This indicates that the models capture at least some important aspects of the processes that drive these interactions. Prominent hotspots for marine mammals in Figure 5 include the east coast of North America, where the largest annual cull worldwide is – in part – being justified based on the perception that the growing harp seal population impedes the recovery of the northwest Atlantic cod stocks (e.g., Yodzis, 2001). In addition, the model identifies areas of potential conflict in the Benguela system off southwest Africa, where the potential impacts of the increasing population of South African fur seals on the hake stocks has been an issue of much debate (Wickens *et al*., 1992, Punt and Butterworth, 1995), or in the waters surrounding Japan where the perception of marine mammals as competitors appears to be particularly prevalent (Anon., 2001b, a). Furthermore for seabirds, the model identified several areas of potential conflict, for instance in the Norwegian and Barents Seas, where rapid decline in the numbers of Common murres has been attributed to the development of industrial fisheries, which target mainly Sandeel for industrial raw materials, fish oil and fishmeal (Anker-Nilssen *et al*., 1997). Looking at our maps, the skewed perception of this issue by nations in close vicinity to these hotspots of interaction becomes an understandable, if somewhat myopic, viewpoint when extrapolated to the global scale.

On the other hand, there is a growing literature on marine mammals and seabirds being starved by depletion of small pelagics by fishing. We shall not pursue this theme here, except for mentioning the Mediterranean, where the fattening of Bluefin tuna (*Thunnus thynnus)* has generated a huge demand for small pelagics. It has been proposed that this demand, which is added to that of the seafood markets of Mediterranean countries, has led to a radical depletion of the stocks of small pelagics in certain areas (e.g. the Ionian Sea) and, therefore, to a depletion of previously abundant Common dolphins (*Delphinus delphi*) (CIESM 2004; Bearzi *et al*., 2004,2005).

BIASES AND LIMITATIONS OF FOOD CONSUMPTION MODELS

All input parameters of the basic food consumption models are affected by a number of conceptual and/or methodological biases.

Abundance

Estimating abundance of any marine mammal and seabird species is challenging due to the vast distributional ranges of most species and, for marine mammals, the fact that they spend the majority of their time underwater (Buckland *et al*., 1993). Moreover, dedicated surveys are labour- and cost-intensive, are generally conducted at irregular intervals and cover only a small proportion of a species' total range. A lack of standard surveying techniques and coverages, and seasonal and inter-annual variation in species occurrence patterns, hamper direct comparison and summation of available regional areas. For all of these reasons, the global estimates used here should be regarded with caution. Nevertheless, we arrive at abundance estimates that are largely comparable to those previously used in similar studies assessing food consumption of marine mammals and seabirds on very large scales (Trites *et al*., 1997; Young, 2000; Tamura, 2003; Brooke, 2004).

Feeding rates

The selection of feeding rates used in any food consumption model strongly affects estimates of total consumption. Feeding rates have been estimated based on a variety of different methods ranging from direct measurements of food intake or maximum stomach contents (Innes *et al*., 1987) to bioenergtic models (Lockyer, 1981b, a; Nagy *et al*., 1999; Winship *et al*., 2002). All models are based on certain assumptions about physiological parameters and the feeding requirements of a specific individual (e.g., Klumov, 1963; Innes *et al*., 1986) or standard metabolic rates of species (Lilliendahl and Solmundsson, 1997; Sigurjónsson and Víkingsson, 1997). Models are thus associated with high uncertainties, particularly for baleen whales owing, e.g., to the difficulties associated with studying metabolic rates of large animals and the non-linear relationship between body mass and consumption (Leaper and Lavigne, 2002).

In the case of marine mammals, our model ignored seasonal difference in food intake. The annual life cycle of many marine mammal species includes extensive fasting periods, often coinciding with reproductive activities (Brown and Lockyer, 1984). Some of the methods used to estimate daily rations implicitly account for the seasonal differences in food intake through the adjustment of the feeding rate exponent, but the effects of such feeding patterns are mostly ignored in these simple models. Unfortunately, evaluating the impact of the lack of seasonal feeding patterns on our estimates is difficult as direct comparisons with other studies that have considered such seasonal variation (Kenney *et al*., 1997; Shelton *et al*., 1997; Sigurjónsson and Víkingsson, 1997; Boyd, 2002) are hampered by the differences in modeling approaches and parameterization.

In the case of seabirds, energy requirements also fluctuate seasonally, because the energetic costs of various stages in the life cycle of mature seabirds differ (see e.g., Ellis and Gabrielsen, 2002). This is why energy demands of seabird species were estimated here using BMR and FMR for the breeding and the non-breeding season respectively. However, BMR and FMR values were estimated using order-specific allometric equations, which may generate bias in the predictions of the model.

Diet composition

The composition of the diet is affected by various uncertainties, including the difficulties associated with obtaining diet information from sufficient sample sizes in the wild (Duffy and Jackson, 1986; Barros and Clarke, 2002). Diet composition estimates based on stomach content or scat analyses tend to be biased with respect to cephalopods, as their hard parts are less readily digested than those of other prey groups and accumulate in the stomach (Furness *et al.*, 1984; Zeppelin *et al.*, 2004). Such biases may, however, be addressed by applying correction factors that compensate for differential effects of digestion on different prey types (Tollit *et al.*, 1997, 2003). More serious biases are introduced, for marine mammals, by the predominance of stranded animals in the overall sample. Such animals may not be representative of the rest of the population, as they are often sick and/or their stomach contents over-represent the coastal components of their diet (Barros and Clarke, 2002).

Overall, stomach and scat samples only represent brief snapshots of what is often a highly variable, geographically and inter- and intra-annually changing diet spectrum of a given species (Haug *et al.*, 1995; Nilssen, 1995; Velando and Freire, 1999; Tamura 2001). More recently developed molecular methods, including stable isotope (Hobson *et al.*, 1994; Best and Schell, 1996; Hooker *et al.*, 2001; Das *et al.*, 2003) and fatty acid (Iverson 1993, Hooker *et al.*, 2001; Lea *et al.*, 2002; Grahl-Nielsen *et al.*, 2003) analyses allow the investigation of diets over longer time periods, but results are often difficult to interpret and come with their own set of uncertainties (Smith *et al.*, 1997).

Spatially explicit food consumption model

There are a number of discrepancies between the species distributions that underlie our food consumption maps and the currently-documented occurrence of the species. This is not surprising given the broad approach we took.

By their nature, predictions are often closer to likely historical distributions of species than their currently utilized range extent (Kaschner, 2004). The predictions likely overestimate food intake in the lower latitudes by not considering seasonal differences in species occurrence and associated feeding patterns. Many marine mammal species undertake extensive annual or semi-annual migrations that cover large distances between areas used primarily for foraging and reproductive purposes (Stern, 2002; Stevick *et al.*, 2002). Migrations and feeding patterns have been incorporated into the model for a subset of species (i.e. the Great whales) in the context of another project (Kaschner *et al.*, 2006) and results indicated that the consideration of these factors only exaggerated the patterns shown in Figure 5. Nevertheless, there are on-going efforts to generate seasonal RES predictions for all marine mammal species in the near future which will be incorporated in future versions of the analysis conducted here.

Similarly to marine mammals, the current lack of consideration of seasonal difference in distributions will have impacted the seabird results presented here. For instance, some species of the genus *Puffinus* (shearwaters) breed in areas of the southern hemisphere. At the end of the breeding season they migrate to feed and winter in the temperate and polar waters of the North Pacific and the North Atlantic Oceans (e.g., Spear and Ainley, 1999; Ito, 2002).

Spatially-explicit resource overlap and sensitivity analysis

A multitude of different conceptual approaches have been developed to investigate different aspects of ecological niche overlap between species and communities (Hanski, 1978; Hurlbert, 1978; Chase and Leibold, 2003). The index developed by Horn (1966) and Morisita (1959) that formed the basis for our resource overlap index (Equation 3) has been deemed inappropriate to measure ecological niche overlap by some (e.g., Hurlbert, 1978). Hurlbert's (1978) main criticism concerns an implicit assumption of this approach that the overlap index is partially determined by the niche width outside the overlap zone (i.e., the extent of utilization of non-shared resources by either player and the overall availability of the resource used is not taken into account). However, in the context of investigating marine mammal-fisheries and seabird-fisheries interactions, we regarded the extent to which either relied on resources not consumed/targeted by the others as an important factor. The abundance or availability of resources would be an important consideration that will partially determine the extent of actual competition between two players (i.e., if the resource is limited and available amounts cannot sustain existing demands of all present consumers). Given the index used here, our model would, for instance, predict low overlap in areas where both marine mammals (or seabirds) and fisheries take relatively small amounts. However, if the abundance of the targeted food type is very low, competition may still conceivably be high in areas of predicted low resource overlap.

Efforts are underway to develop models to generate large-scale biomass estimates of fish (Christensen *et al.*, 2003) that could be incorporated into the analysis in the future. However, for the most part, global estimates for most prey types are currently unavailable, making the consideration of prey abundance in overlap equations difficult.

Our analysis of resource overlap was affected by the biases of all input parameters as discussed above. Yet, the nature of the models and the type of data used make it difficult to attach a quantitative estimate of uncertainty to our predictions. Nevertheless, a basic sensitivity analysis was conducted by running the model with global marine mammal food consumption estimates that varied by an order of magnitude. Results showed that such variation had little to no effect on the spatial extent of areas of predicted high resource overlap. This indicates that areas of high overlap are largely caused by extremely high catch rates, due to highly concentrated fishing effort.

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CHAPTER 4

FISHMEAL AND FISH OIL: PRODUCTION, TRADE AND CONSUMPTION[4]

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ABSTRACT

This chapter documents the scale of production, global trade and consumption of fishmeals and oils and subsequent implications for food security and safety, and aquaculture. A brief overview of global production, trade, and consumption of fishmeals and fish oil from 1961 to present is presented. Trade data for fishmeal and fish oil commodities were reviewed and analyses indicate a significant global increase in the production, trade, and consumption of these forage fish products since 1961. Expansion of intensive food production, particularly in aquaculture, and the low relative pricing of fishmeals and oils to other meal products has made this growth possible. These increases in agricultural consumption are reflected in a change of global producers and consumers over the past 43 years. The scope for further increasing production in fishmeals and oils is limited, since most fisheries exploited for reduction into meals are fished to their biological limits. While alternatives are now being sought in some industries, consumption continues to increase as fishmeals and oils remain the preferred and superior feed inputs.

INTRODUCTION

The reduction of landed forage fish to fishmeal and fish oil was historically focused on where the fish were caught (see Chapter 1). However, with the cost of shipping being low relative to other costs associated with production, such as labour, as well as the environmental requirements for fishmeal factories, centres of production are changing with China emerging as a major importer of raw fish and a producer of fishmeal.

Seafood products are widely traded in the international market place with approximately 40% of fish landed traded in one form or another (Delgado *et al.*, 2003) and fishmeal and fish oil are no exceptions. Fishmeal and fish oil are beneficial in the intensive production of poultry, pigs and ruminants as well as farmed fish. While soymeal can be substituted for fishmeal, the essential fatty acids in fishmeal and fish oil are superior to other meals, with several benefits such as increased disease resistance. These benefits generate a demand for fishmeal and fish oil production throughout the world, but with landings of small fish for reduction focused in a few areas of the world, it is a highly traded commodity. However, the demand for fishmeal in the feeds of these animals is determined by the least cost of meals, especially soy, with the upper limit set by the taste imparted into the meat (Robinson and Crispoldi, 1971).

Fishmeal and fish oil have been and continue to be a significant portion of these traded products. Prior to the 1950s, much of the fishmeal and fish oil was produced and consumed in Northern Europe, North America and Japan, with little international trade of either product. In the late 1950s, international trade in these products expanded as anchoveta fisheries and associated fishmeal and fish oil plants developed along the Peruvian and Chilean coasts. By the late 1960s, fishmeal and fish oil, much of it from Latin America, were traded globally. Japan declined as a producer of fishmeal and fish oil as its fishing fleets were excluded from coastal waters when other nations declared their EEZs, leaving Latin and North America and Northern Europe as the major producers and traders. Until recently, much of the fishmeal produced in Latin America was traded internationally, while most of the US production was used

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domestically with imports supplementing domestic production, and Europe traded within the continent as well as overseas depending on the level of production and product quality. Over the last 10 years the pattern of production and trade trends have changed with the growth of aquaculture, and the emergence of China as a net seafood consumer.

Historically, the bulk of fishmeal and fish oil produced and traded were consumed primarily in the intensive animal production sector: poultry, pigs and cattle and primarily in countries with significant animal production sectors. Until recently, the poultry and pig production sectors consumed most of the fishmeal production, with pig or poultry production dominating at the country level, depending on number of factors including culture. Today nearly 50% of fishmeal production is consumed in the aquaculture sector (Malherbe, 2005).

The trends in production, trade and consumption are changing rapidly especially in Latin America where aquaculture is expanding, and in China which is developing its seafood sector. These and other changes are described in the following sections.

METHODS

Production and supply

Annual fishmeal (all processed products combined, including feed) and fish oil production, import and export data from 1961 to 2001 were obtained from the FAO's Processed Products database (FAO, 2006a) globally and for the top producers in 2001. The countries selected represent at least 80% of the annual production in 2001. Adding domestic production and imports, and subtracting exports allowed estimation of domestic supply.

Trade

The fishmeal and oil export trade matrices were compiled for the top exporters of fishmeal (Table 1) and fish oil (Table 2). Twenty-four countries were identified as representing the principal exporters by quantity of fishmeal and oil products between 1961 and 2004 in six regions. These countries, accounting for 90% of global production, were placed in a trade matrix by exporter quantity in metric tonnes (t) to track global trade routes and trends over a 43-year period. These exporters represent the top 5+ countries in the specified region most consistently over the 43-year period, accounting for at least 80% of the fishmeal and fish oil exports from that region.

Table 1. Top fishmeal exporting and importing countries from 1961, 1980, 2003; countries are ranked in order of their tonnage of fishmeal (UN, 2006; FAO, 2006a, 2006c and 2006g)

Exporting			Importing		
1961	**1980**	**2003**	**1961**	**1980**	**2003**
Peru	Chile	Peru	Germany	Germany	China
South Africa	Peru	Chile	UK	UK	Japan
Norway	Denmark	Iceland	USA	Japan	Germany
Iceland	Norway	Denmark	Netherlands	Sweden	UK
Angola	Iceland	Germany	France	Switzerland	Denmark
Denmark	Germany	USA	Italy	Finland	Norway
Chile	Ecuador	Norway	Denmark	Cuba	Russian Fed.
Canada	USA	Ecuador	Switzerland	Poland	Spain
Morocco	Canada	South Africa	Austria	Italy	Canada
UK/Netherlands	Panama	Morocco	Singapore	Romania	Italy

Fishmeal and fish oil exporters used in the trade matrix were divided into four global regions for a more accurate analysis. These regions represent much of the trade in both commodities: Northern Europe, Latin America, USA and Canada, Africa, and 'Other'. The 'Other' category includes all exporters in the specific commodity for a given year not in the list of top exporters by region. In this way, 100% of global trade in fish and oils is accounted for. Trade in the 'Other' category often represented a small proportion of the traded products and was not included in these analyses.

Table 2. Top fish oil exporting and importing countries from 1961, 1980, 2003; countries are ranked in order of the tonnage of fishmeal (UN, 2006; FAO, 2006a, 2006c and 2006g)

Exporting			Importing		
1961	**1980**	**2003**	**1961**	**1980**	**2003**
Peru	USA	Peru	UK	UK	Norway
Norway	Denmark	Iceland	Netherlands	Germany	Netherlands
USA	Norway	Denmark	Germany	Netherlands	Chile
Iceland	Chile	USA	Norway	Peru	Germany
Germany	Iceland	Norway	Sweden	Norway	UK
Netherlands	Panama	Chile	USSR	Colombia	Canada
Denmark	Netherlands	Netherlands	France	Belgium	France
Chile	Canada	Germany	USA	Poland	Japan
UK	Faeroe Isl.	Canada	Denmark	France	Denmark
Canada	Sweden	Ecuador	Canada	USA	Spain

The trade matrices were derived primarily from three main sources: the FAO World Agricultural Trade Matrix (WATM), the FAOSTAT Fisheries: processed products database and commodity balance database (FAO 2006 a, b and c), and the UN commodity trade statistics database or UN Comtrade (UN, 2006). 'Trade Analyser' provided additional data for select years and countries missing or having incomplete data (CHASS, 2006). These databases were combined for a more complete and accurate data set than either group could provide individually. The data are presented based on export quantities by net dry weight (t or 1000 t) from the source country to the trade partner. UN Comtrade values were converted from kilograms and FAO values were rounded up to the nearest whole tonne.

Commodity categories were chosen based on their relevance, completeness, and time span covered by the dataset. As most of the trade data were derived from Comtrade, the trade matrix used in this analysis was configured to the same 'exporter country-to-export partner' format, as well as most of the available data from other sources (Trade Analyser, FAO etc.) which were also focused on exports. The availability of data from the combined databases covers a 43-year period, from 1961 to 2004. Data for exporters but not trade partners were available through FAOSTAT for 1961; trade matrix data exists from 1962 onward. The bulk of the data from 1997-2003 originated from WATM, while the remainder of the data originated in the UN Comtrade database. A matrix composed of 24 trading countries was analysed in this chapter (Appendix 1).

While every effort was made to use trade categories where the use was explicitly stated not for human consumption, this was not the case for the categorization scheme of the UN Comtrade and FAO datasets. A detailed description of the terms and rules in estimating trade flows is given in Appendix 2.

The trade matrix data for fishmeal was derived from FAOSTAT Code 1174 'Fish Meal' and UN Comtrade commodity category Standard International Trade Classification (SITC Rev.1 using data from 1962) Code 0814, 'Meat & fish meal, unfit for human consumption'. SITC Rev. 1 was used as this category was the most comprehensive and complete. FAOSTAT defines fishmeal as "flours, meals, and pellets from fish, crustaceans, mollusks, or other aquatic invertebrates used for feed" (FAO, 2006c). UN Comtrade's defines fishmeal as "Animal Feed Made from Meat, Meat Offal, Fish, Crustaceans, Mollusks, or Aquatic Invertebrates" (UN, 2006).

The meat meal content contained within the 'trade-matrixed' Comtrade fishmeal trade category was adjusted in the compiled dataset to reflect only fishmeal quantities to more accurately represent the flow of fishmeal from exporter to trade partner. The annual proportion of meat meal to the combined meat and fishmeal category in the FAO Commodities Balance database (containing only total export quantities) (FAO, 2000c) was used to estimate the fishmeal proportion in UN Comtrade database, which did not contain data for meal meals.

It was assumed that all databases used in analysis were consistent in their quantities of fishmeals reported. Based on this assumption, trade categories where the meat and fishmeal commodities were combined were treated as having the same proportions of meat meals and fishmeals as in the two categories where meat and fish meals were separate. Only countries with proportions of meat meal >5% had their totals adjusted; some country totals may therefore still contain negligible amounts of meat meal.

For the year 2004, where there were no FAO statistics available for meat meals, the proportions from 2003 were used. This method for adjusting fishmeal volumes was validated using independent data from Macer (1974) and described in Appendix 3.

The fish oil trade matrix data was also derived from both the UN Comtrade and FAOSTAT databases. FAOSTAT Code 1223 was used and is defined as "Fats and oils from fish and marine mammals extracted from the body or liver, whether or not refined, but not chemically modified" (FAO, 2006d). Code 4111 "(Fats and Oils of Fish and Marine Mammals" which the UN Comtrade defines as "Fats and oils and their fractions, of fish or marine mammals, whether or not refined, but not chemically modified" was also used (UN, 2006). There was no explicitly stated category defining fish oils unfit for human consumption in either database. It is possible that a human consumption component may be present, but it is assumed to contribute negligible amounts.

The marine mammal oil content contained within the oils trade category was removed where possible, using the available Comtrade data from Code 41113 "Oils and Fats of Marine Mammals" to more accurately represent the true flow of fish oils from exporter to trade partner. Data on marine mammal oils in UN Comtrade were available and were subtracted from the combined category. Prior to 1976, however, no marine mammal oil category data were listed in Comtrade and no data for these oils exist in FAOSTAT; marine mammal oil proportions from 1976 were therefore used to remove estimated marine mammal oil components from the years prior to 1976.

DATA LIMITATIONS

While every effort was made to ensure data accuracy and consistency, there are some limitations to the trade matrix. Trade commodity categories, for the most part, do not include exclusively 'fishmeal' or 'fish oil' unfit for human consumption categories, with the exception of some later years, or data are not in a matrix format. Although categories were adjusted where possible to better reflect only these commodities, trade quantities may still be over-estimated, especially for certain countries where meat meals or marine mammal oils may exist in the commodity category.

Data availability for some countries and some years is poor or inconsistent. This absence of data varies in duration depending on the country in question, and was either apparently random or systematic (e.g., a country may only report every 2 years). In many cases it was unclear if there was no trade from those countries, or just no data available. These inconsistencies may under-estimate total exports from the region and country in question. This was particularly the case in earlier years (e.g. 1962. and latter years (e.g., 2004) of the 43-year time period; therefore, 1962 and 2004 were excluded from the analysis.

There were data inconsistencies within and between databases. Commodity categories from both the FAO and UN Comtrade were a compilation of a number of different international commodity classification schemes, each with their own limitations. The FAO and Comtrade list in their disclaimers that the degree of comparability between classification systems is plagued with uncertainty (FAO, 2006a). While human error may contribute a minimal amount, errors may also be attributed to these differences in reporting and the classification systems used by the UN and the FAO (See Appendix 2).

Meal quantities reported by FAOSTAT and WATM differ from each other either minimally (≤10%) or significantly (>10%) depending on the year and country involved. FAO commodity balance numbers are significantly lower than WATM and the processed products database. WATM has the greatest quantities listed while FAO processed products are in between the two. All sources indicate that their unit of measure is exports in 'metric tonnes' (ie., t) and is the net/product weight for processed fish commodities (FAO 2006a). The UN Comtrade database also differs in quantity reported by the FAO's databases, and depending on the year or country reporting this quantity may be higher or lower from the FAO quantities reported.

Specific countries are also highly variable in reported quantities between trade databases, particularly with respect to fishmeal. The USA, Germany, Denmark, Belgium, Argentina, Ecuador, and Panama were notable in this respect. The collection and processing of the data collected in these two systems may be one explanation for these differences. Changes in political boundaries (e.g. the re-unification of Germany) and consequential changes in departments responsible for fisheries statistics may also account for differences in reporting.

Aquaculture and Fishmeal Consumption

The amount of fishmeal (as feed) consumed in the aquaculture sector was estimated for countries with major fishmeal supplies in 2001 (Table 3) for 1980, 1985, 1990 and 1995-2001. This analysis starts in 1980 since that is the earliest year that written information on feed ratios, fishmeal content and usage starts. The countries listed in Table 2 represent over 80% of the global fishmeal supply in 2001. Like many countries in Asia, China, India, Indonesia, The Philippines, Thailand and Vietnam have large supplies of fishmeal and they also use bycatch as direct feed in aquaculture, which is often not recorded in official production statistics. This makes it difficult to reliably estimate the use of fishmeal in the aquaculture sector for these countries (FAO, 2005). The following conventions are used:

Fishmeal Consumption (i) = aquaculture production (i)*FCR (i) * USE (i) * CONTENT (i) for species (i)

Where FCR = Food Conversion Ratio;

USE = proportion of fish raised on fishmeal; and

CONTENT = the proportion of aquafeed that contains fishmeal.

Table 3. Comparison of countries producing and consuming fishmeal. Highlighted in bold are major producers but not consumers; countries in italics are where there are major supplies (after adjusting for imports and exports) but not production in 2001 (FAO, 2006a).

Top 20 Countries for Production	Top 20 Countries for Consumption
Peru	China
Chile	Japan
China	Denmark
Denmark	Thailand
Thailand	Norway
United States of America	United Kingdom
Norway	United States of America
Iceland	USSR, Former Area of
Japan	Spain
Morocco	Chile
Mexico	*The Philippines*
Spain	Mexico
USSR, Former Area of	*Canada*
South Africa	*Indonesia*
Ecuador	South Africa
India	*Italy*
United Kingdom	*Netherlands*
Faeroe Islands	India
Sweden	Morocco
Pakistan	*Greece*

The consumption estimate is based on FAO's aquaculture production database (FAO, 2006e). Feed conversion ratios, the proportion of production using fishmeal, and the proportion of fishmeal in aquafeeds was used to 'back calculate' the volume of fishmeal needed for producing the recorded volume of cultured species that consume fishmeal.

The conversion factors used are based on the work of Tacon (1997) and FAO (New and Wijkstrom, 2002). The species used varied between countries, and included freshwater production where appropriate. The use of fishmeal in China's aquaculture was adjusted further based on Yang (2003), who reported that 40% of imported fishmeal is used for aquaculture. The fishmeal consumed is expressed as a percentage of total fishmeal feed supply for countries making up more than 80% of the total fishmeal feed supply globally. In estimating the percentage of the supply consumed in the aquaculture sector average values were used for the conversion ratios, resulting in percentages over 100% for countries with better than average feed systems. For these countries it was assumed that the total national fishmeal supply was consumed by the aquaculture sector. The method was validated using known percentages for China (Jiwu and Rortveit, 2004; Yang, 2003) and Europe (FIN, 2003) in early 2000.

RESULTS

Production and Supply

Globally fishmeal production and supply has increased steadily since 1961, when data first became available (Figure 1). The variation in production reflects the variations in landings due to climatic factors such as El Niño. Fish oil production, however, appears to be steady at around 1 million t annually (Figure 1). A comparison between production and supply highlights discrepancies in fishmeal data from the mid 1990s, where the difference between supply and production is positive and large with differences of 3% to 11% of annual production. This may be due to re-exports of fishmeal. However, re-export data starting in 1976 in the FAO Fisheries Trade database (FAO, 2006f) does not suggest an increasing trend in re-exports of fishmeals for the same time period (Figure 2).

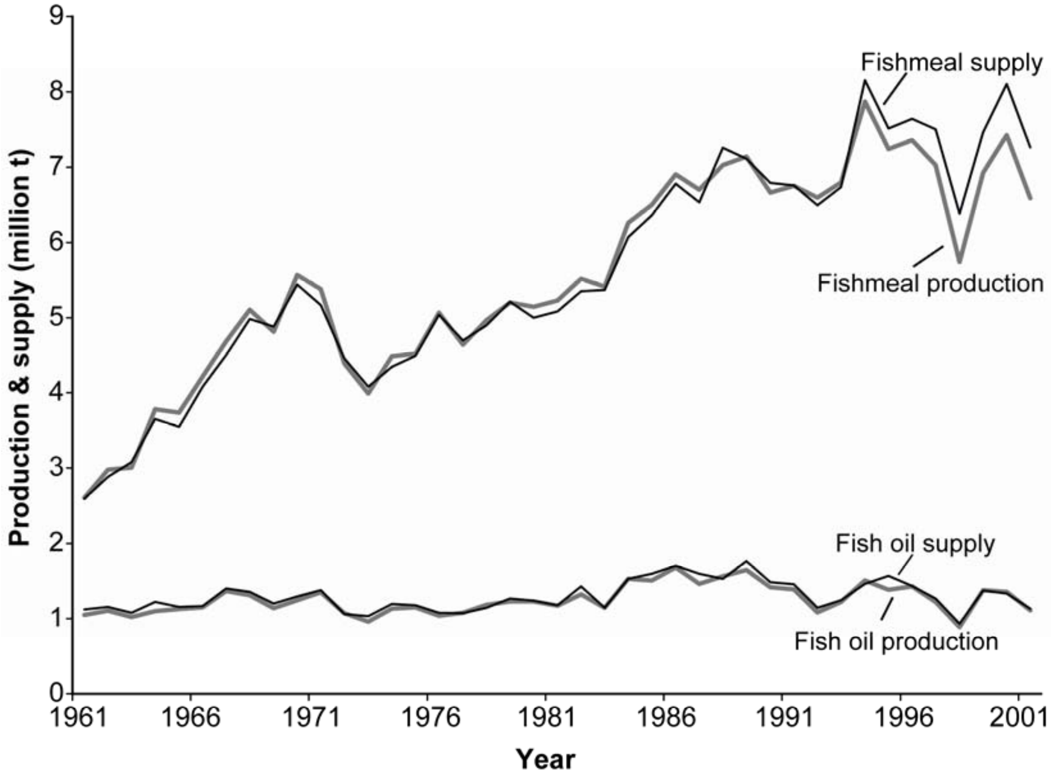

Figure 1. Global trends in the production and supply of fishmeal and fish oil from 1961 to 2001 (FAO, 2006a).

Chile and Peru are the world's top fishmeal producers, but the geographic focus of where fishmeal and fish oil is produced has diversified from countries which land the fish used in reduction to other countries where investment, labour and environment requirements are more favourable for the industry, such as China (Figure 3). Countries that are major fishmeal producers are also not necessarily major consumers of fishmeal, as seen in Table 2. The consumption of fishmeal (based on supply) has changed from being dominated by Europe, the former USSR and the United States in the 1960s to China, Japan, Thailand and Denmark in 2001 (Figure 4).

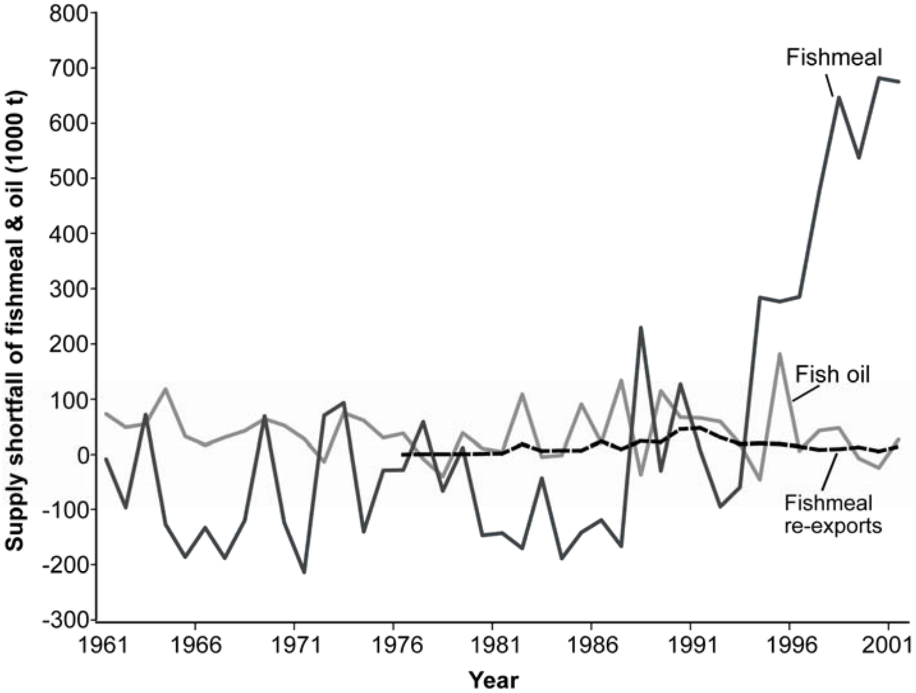

Figure 2. Documenting the recent uncoupling of fishmeal from fish oil production. Fishmeal and fish oil 'net' (i.e., export – local production), and re-exports (dotted line) are not the reason for the increase of fishmeal relative to fish oil (see text).

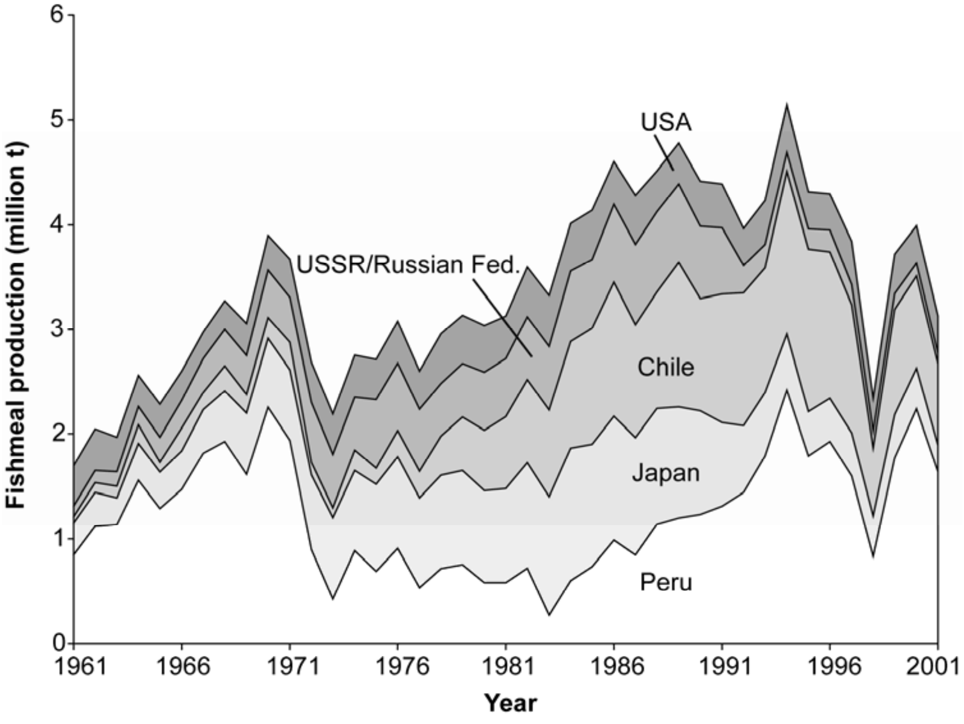

Figure 3. Top 5 fishmeal producing countries from 1961 to 2001 (FAO, 2006a).

The same countries that dominate fishmeal production also dominate fish oil production with Japan declining in importance since the early 1990s (Figure 5). Countries with major supplies of fish oil are not the same as for fishmeal with Norway having the largest supply in 2001, followed by Chile, while China is well below other countries (Figure 6). This difference is due to Chile and Norway focusing on salmon aquaculture, while China's supplies of fishmeal are used for a range of intensive animal food production systems and not just aquaculture.

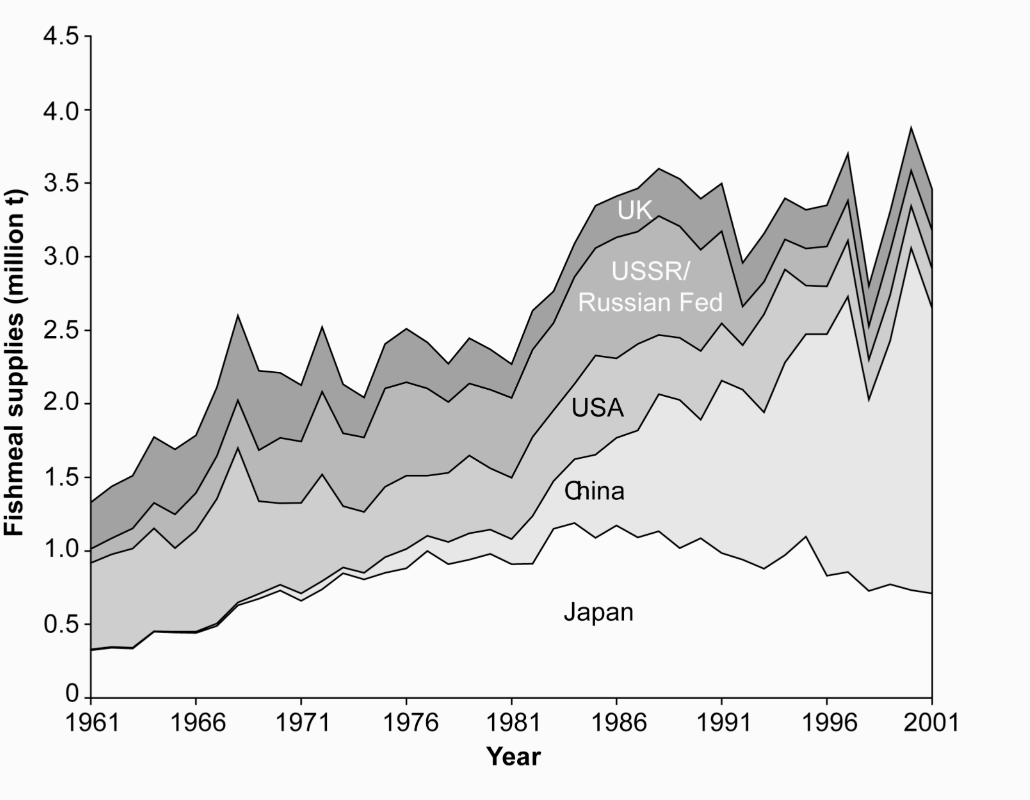

Figure 4. Top 5 consumers of fishmeal from 1961 to 2001 (FAO, 2006a).

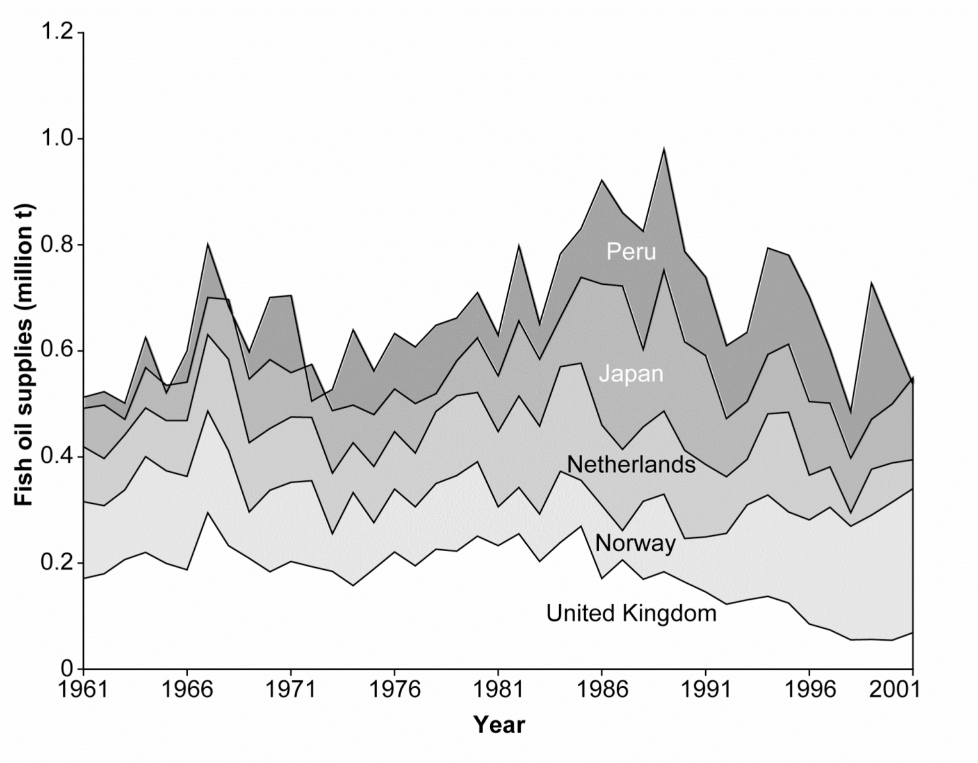

Figure 5. Top 5 producing countries of fish oil from 1961 to 2001 (FAO, 2006a).

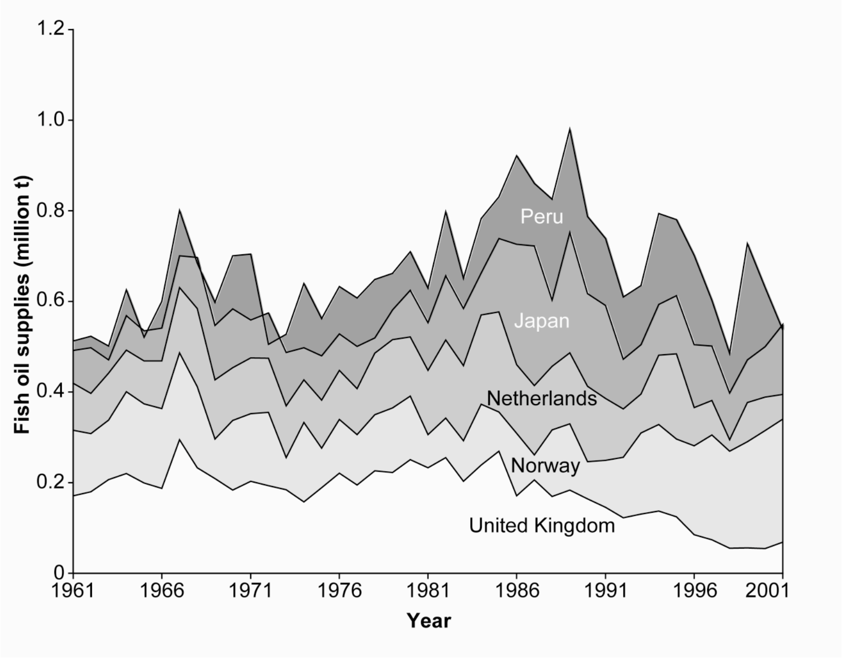

Figure 6.Top 5 consumers of fish oil from 1961 to 2001 (FAO, 2006a).

Fishmeal Trade

Fishmeal is traded globally with much of the raw material originating in three regions: Latin America (Pacific coast), Northern Europe and North America (North Atlantic and Alaska). Fishmeal and fish oil produced in these regions are exported to Europe, Asia, North America and Latin America. There is trade within the regions, especially within Europe and North America, but less so within Latin America.

Fishmeal exports have been steadily increasing since 1961, in the three main producing regions, despite the high variability in global export volumes (Figure 7). Between 1961 and 2004, approximately 129 million metric t (product weight) of fishmeal was produced globally (Thomson, 1990; Barlow, 2002; Zaldivar, 2004). The 24 countries in the trade matrix represented approximately 90 % of the global trade in fishmeal over the 43 years. Five of these countries (Peru, Chile, Denmark, Norway, and Iceland) account for approximately 76% of the total world exports. China is the largest importer of fishmeal, and is sourcing the product from Latin America, North America and Northern Europe. Over the same time period Peru alone accounts for over 39% of global exports. Very little trade currently occurs with Africa and Oceania.

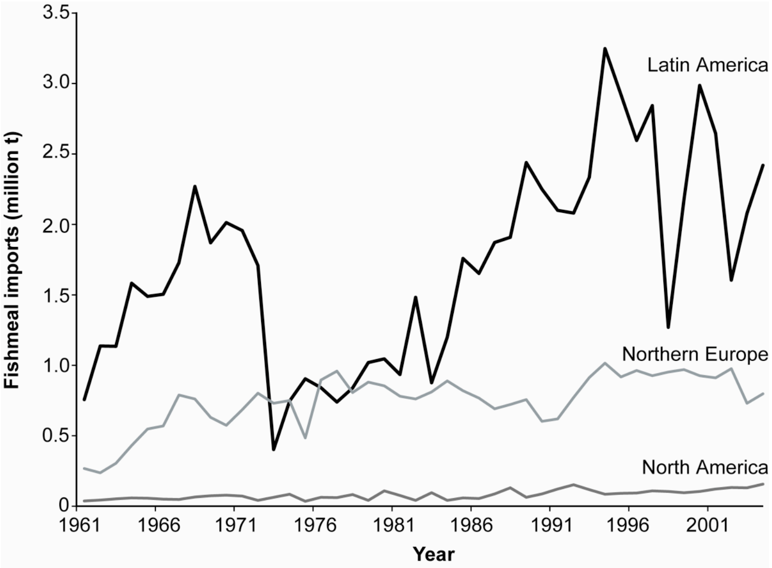

Figure 7. Regional fishmeal exports 1961 to 2001 (FAO, 2006g).

Latin America, primarily Peru and Chile are the biggest exporters of fishmeal globally with fishmeal shipped throughout the world (Figure 7 and Figure 8). Significant declines in exports from Latin America correspond to strong El Niño years in 1972/73 and 1997/98 (Figure 7). Peru and Chile accounted for over 56% of the global exports. Most of the product was shipped to Europe prior to the 1990s; since then Asia is now the largest market for Latin American fishmeal with Japan, China and Taiwan as the largest importers (Figure 8). Prior to the 1980s, the major European importers were Germany, Italy, Poland and Spain; in 2000 the major importers were Spain, Italy, Germany, Norway and the United Kingdom. The USA has historically been the main North American importer of Latin American fishmeal, but total imports have not changed significantly relative to Asia and Europe, especially since the 1990s.

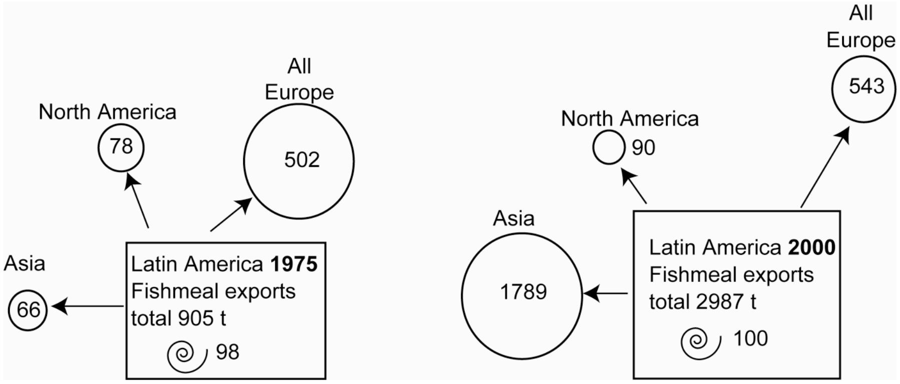

Figure 8. Amount (in 1000 t) of fishmeal exported from Latin America to other regions and within the region in 1975 (left) and 2000 (right); numbers in bold are the year of export. The circles' radii are proportional to total export.

Northern European fishmeal production is traded extensively and contributes more than 25% of the global fishmeal export quantities. Exports from Northern Europe have generally ranged between 600 and 800 thousand t since the mid-1960s. Most fishmeal was historically and is currently traded within Europe (Figure 9). Four countries, Denmark, Germany, Iceland, and Norway account for 93.5% of regional fishmeal exports and 24% of total global exports. Denmark alone accounts for 8% of the global exports over the 43-year time period. The main importing countries prior to the 1980s were the United Kingdom, Sweden and Germany; in 2000, the main importing countries were the United Kingdom, Netherlands and Norway. There have been modest increases in exports to Asia since 1975, primarily to Japan, Taiwan, Thailand, and China, with South Korea emerging as a player in the late 1990s. Exports to North America have increased since 1975 but not to the levels seen in the early 1970s.

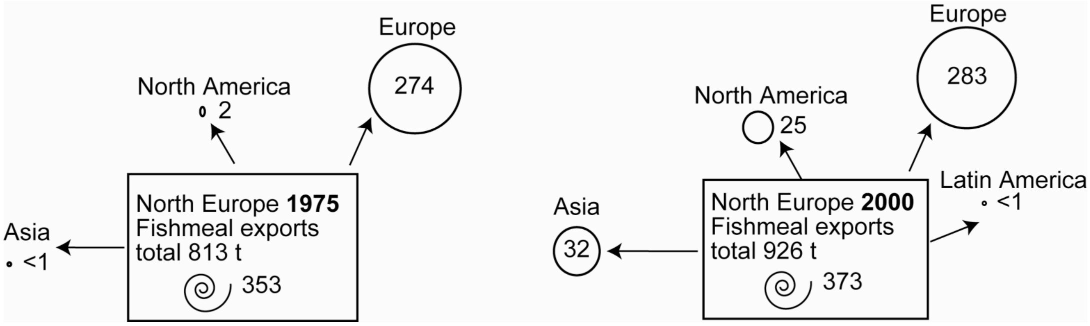

Figure 9. Amount (in 1000 t) of fishmeal exported from Northern Europe to other regions and within the region in 1975 (left) and 2000 (right), numbers in bold are the year of export. The circles' radii are proportional to total export.

The USA and Canada have been almost exclusively net importers of fishmeal since 1961, and they are a minor contributor with 3% of the global export of fishmeal. Most North American fishmeal production is traded between the United States and Canada (Figure 10). Prior to the 1980s Europe, in particular the United Kingdom, was a significant importer of North American fishmeal. However, by 2000 imports to Europe were insignificant. Exports to Latin America have increased since 1975 but have decreased from levels seen during the period of peak imports during the late 1980s to the late 1990s. Exports to Asia have also increased, particularly since the 1990s. These two regions imported virtually no fishmeal from North America in the 1970s, but by 2001 imports had increased significantly.

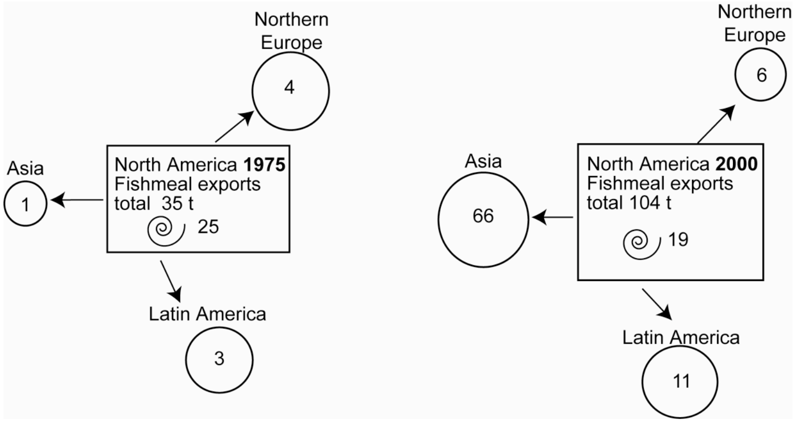

Figure 10. Amount (in 1000 t) of fishmeal exported from North America to other regions and within the region in 1975 (left) and 2000 (right), numbers in bold are the year of export. The circles' radii are proportional to total export.

Africa is the only region to decrease its quantities of fishmeal exported over time. Export quantities from the six major African countries reached nearly 400,000 t in 1968, and averaged approximately 270,000 t of fishmeal exports a year from 1961 to 1975/76 when exports abruptly decreased to an average of 32,000 t a year until the mid-1980s. South Africa and Angola are responsible for this large decrease in exports: From 1975 to 1976 South Africa's exports of fishmeal dropped from 147,000 t to 45,000 t and the country remained variable in its reporting until the late 1990s. After 1976, Angola only reported exports for three more years over the entire time period; this is most likely a reflection of the political instability of the country.

Fish Oil Trade

Global exports of fish oils have remained relatively steady since 1961 with only a minor gradual growth in exports starting in the late 1970s (Figure 11). Quantities exported remained for the most part between 600 and 950 thousand t over the 43-year period. These steady trends are reflected in Northern Europe and North America. However, Latin America is highly variable, though increasing since 2000.

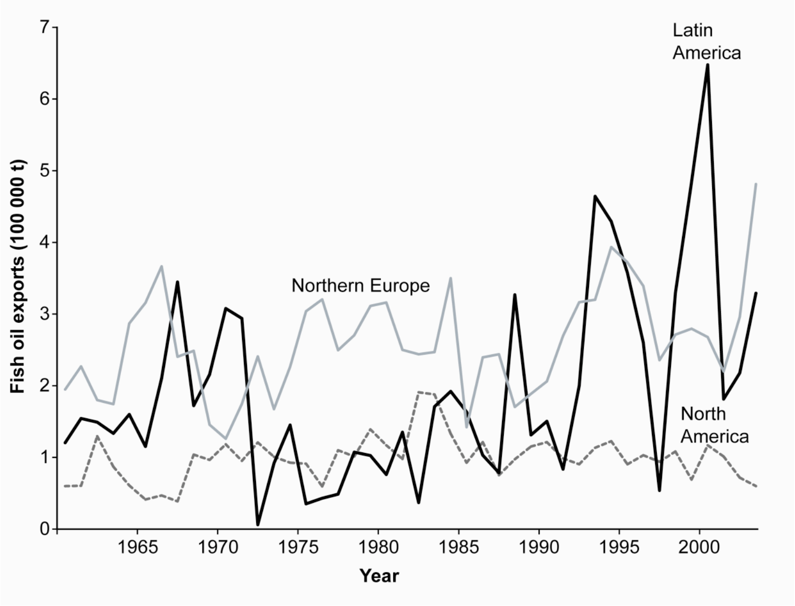

Figure 11. Regional fish oil exports 1961 to 2001 (FAO, 2006g; UN, 2006).

Between 1961 and 2003, approximately 35 million t of fish oil were exported globally (FAO, 2006b), averaging around 1 million t a year (Figure 1). Twenty exporting countries within the fish oil trade matrix represent over 68% of global fish oil exports from 1961-2004, with the remaining 32% exported by the rest of the world. Six countries, Norway, Denmark, Iceland, Peru, Chile, and the USA, represent over 62% of global fish oil exports. Peru is the largest exporter of fish oils over time with 17% of the global trade, followed by the USA and Norway.

Fish oil is also traded globally, and much of the trade is sourced in the same three regions as for fishmeal: Latin America (Pacific coast), Northern Europe and North America (North Atlantic and Alaska). Fish oil produced in these regions is exported to Europe, Asia, North America and Latin America. There is some trade within the regions, especially within Europe and North America, but comparatively little within Latin America, at least until recently, with Chile importing oil from Peru. Norway is the largest world importer of fish oil, followed by Chile, the UK, and Denmark; all have significant aquaculture sectors.

Latin America, primarily Peru and Chile, is also a significant exporter of fishmeal globally, with fish oil shipped throughout the world, especially in the last 20 years (Figure 11). Europe continues to be the major importer of fish oil from Latin America, with Germany and the Netherlands as the largest importers. Asian imports of fish oil have increased since the 1970s, and North American imports have declined (Figure 12). Prior to the 1980s, the major European importers were the Netherlands, Germany and the United Kingdom; in 2000 the major importers were Norway and the Netherlands. In the 1970s the North American oil imports were minimal, but in 2000 oil imports to Canada increased.

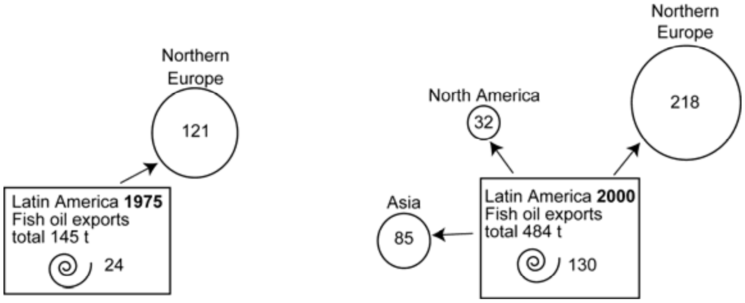

Figure 12. Amount (in 1000 t) of fish oil exported from Latin America to other regions and within the region in 1975 (left) and 2000 (right), numbers in bold are the year of export. The circles' radii are proportional to total export.

Northern Europe's fish oil production contributes 33% of the global fish oil exported. Exports from Northern Europe have generally ranged between 150 and 300 thousand t since the mid-1960s. Most fish oil was historically and is currently traded within Europe (Figure 13). Five countries (Denmark, Iceland, Norway, Germany and the Netherlands) account for 94% of regional fish oil exports and 31% of total global exports. Norway alone accounted for 11% of the global exports. The main importing countries prior to the 1980s were the United Kingdom Germany, and Sweden; in 2000, the main importing countries were Norway, the Netherlands, the UK, France and Denmark. There have been modest increases in exports to Asia, primarily Japan. Exports to North America have increased slightly since the 1970s.

Figure 13. Amount (in 1000 t) of fish oil exported from Northern Europe to other regions and within the region in 1975 (left) and 2000 (right), numbers in bold are the year of export. The circles' radii are proportional to total export.

Exports of fish oil production out of North America since 1961 have not been high relative to exports of oil out of Latin America and Northern Europe. North America is a minor contributor with just over 12% to the export of global fish oils. Most North American fish oil production is traded between the United States and

Europe (Figure 14). Prior to the 1980s Europe, in particular the United Kingdom and the Netherlands were significant importers of North American fish oil. However, in 2000, European imports were declining while exports to Asia, in particular Japan were increasing. Exports to Latin America, Mexico in particular, increased from the 1970s. These two regions imported virtually no fish oil from North America in the 1970s, but by 2000 Mexico's imports had increased to 25 t.

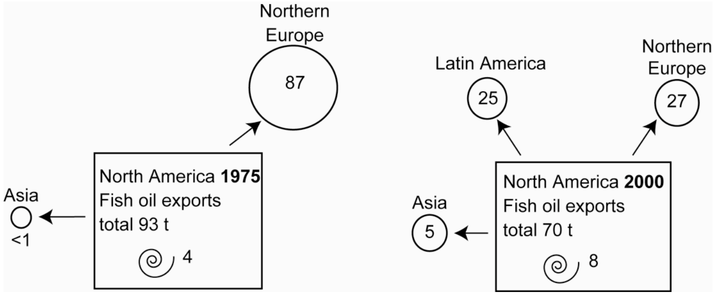

Figure 14. Volume (in 1000 t) of fish oil exports from North America to other regions and within the region in 1975 (left) and 2000 (right), numbers in bold are the year of export. The circles' radii are proportional to total export.

CONSUMPTION

Fishmeal and fish oil are primarily used for intensive food production with 24 % for pigs, 22% for poultry and 46% for aquaculture. A small proportion (8%) is used for ruminants, pet food and the pharmaceutical industries (Malherbe, 2005). Until recently, most fishmeal and fish oil was used in the production of poultry and pigs; this has changed with the development of aquaculture, in particular the farming of carnivorous species as well as changes in feeding practices for omnivorous species such as shrimp and tilapia. Since 1981, the proportion of fishmeal used in aquaculture has been increasing (Table 4). Where the aquaculture sector for high value species is well developed there has been an increase in the proportion of fishmeal consumed in the aquaculture sector (Figure 15).

Table 4. Percentage of fishmeal supply used for aquaculture in countries with significant fishmeal as feed supplies 1980, 1985, 1990, 1995-2001 (FAO, 2006a and e; Tacon, 1997; New and Wijkstrom, 2002).

	Year									
Country	**1980**	**1985**	**1990**	**1995**	**1996**	**1997**	**1998**	**1999**	**2000**	**2001**
Chile	0	1	23	37	46	59	72	35	57	100
Norway	21	36	78	62	80	75	100	100	79	92
Greece	0	4	22	74	96	100	100	100	100	88
Canada	1	3	30	32	41	45	54	66	49	60
Italy	13	21	22	41	37	40	58	52	48	46
Korea, Republic of	6	5	5	10	11	42	48	51	38	35
United Kingdom	1	4	9	21	23	23	30	34	28	32
China	8	7	19	18	20	21	34	30	24	32
Japan	18	21	27	21	27	26	31	30	29	29
Spain	9	8	11	15	20	13	30	37	17	20
Egypt	15	12	71	73	64	100	92	49	51	17
FMR USSR	1	1	2	7	8	8	18	18	16	16
USA	4	5	8	14	14	13	17	16	15	16
Mexico	1	1	2	5	5	6	10	11	9	15
Iran	1	2	3	1	1	3	3	5	6	13
Netherlands	0	0	1	1	4	4	7	13	10	5
Denmark	12	10	14	5	6	4	4	4	4	3
Morocco	0	0	1	1	2	2	2	2	1	1
South Africa	0	0	0	0	0	0	1	1	1	1

*Although India, Indonesia, Philippines, Thailand, and Vietnam have high domestic fishmeal supplies, the proportion used in aquaculture is difficult to estimate since there is direct use of bycatch.

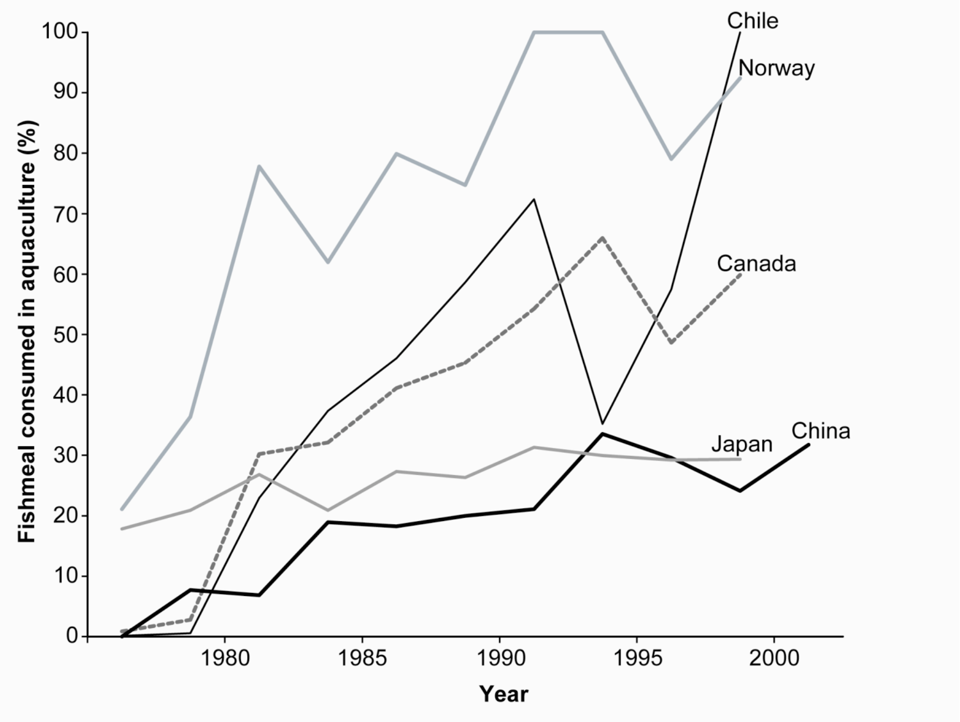

Figure 15. Proportion of fishmeal consumed in aquaculture for major aquaculture producing countries in 1980, 1985, 1990, 1995-2001 (FAO 2006a and e; Tacon 1997; New and Wijkstrom 2002).

In 1984 only 8 % of global fishmeal production was used in aquaculture; by 1988 use had grown to 10% for fishmeal and 16% for fish oil and by 2000 aquaculture was consuming 35% of the fishmeal and 54% of fish oil (Delgado *et al.*, 2003). The use of fishmeal and fish oil by the aquaculture sector in 2002 was estimated to be 46% with projections to 2012 of 50% (Malherbe, 2005), with significant declines in the use of fishmeal in the poultry sector. In 2002, 81% of fish oil production was used in aquaculture, with projections of 88% in use by 2010 (FIN, 2006).

These are global estimates based on reports of global production as well as industry reports on fishmeal and fish oil use, and composition in industrial feeds (Tacon 1997; New and Wijkstrom 2002). While there are estimates for global fishmeal and fish oil consumption for the major farmed species, there are no estimates of the proportion of fishmeal consumed by the aquaculture sector on a national basis.

DISCUSSION

The production of fishmeal has increased since 1961, while fish oil has remained relatively stable over the same time frame. Some of this increase is likely due to the inclusion of other species such as Blue whiting (see Chapter 1), which have some of the elements needed in fishmeal, but they may not have the same oil content as the Peruvian anchoveta and may account for the lack of growth in the oil sector. The increase in fishmeal could also be attributed to improved processing, which is reported to be as efficient as 4:1 (Khemakorn *et al.*, 2005) and an increased use of discards (Kelleher, 2005). However, a 20% improvement in processing does not account for all of this increase, and there is no available information on discards that are diverted to fishmeal.

The scope for increasing fishmeal and fish oil production largely depends on changes in capture fisheries for both reduction and human consumption. There is a general consensus that current fisheries are fished at or beyond their maximum levels (see Chapter 1) and there is little scope for developing new fisheries for reduction. Some fisheries that are used for reduction and human consumption offer some scope, but this option risks the food security of many coastal populations (Chapter 3). This has resulted in calls for the aquafeed industry to find alternative feed materials (Sorgeloos, 2000).

Peru and Chile produce much of the world's fishmeal. However, the expansion of the fisheries and seafood sector in China has also changed the nature of production (Figure 3). Historically, fishmeal factories were located close to where the forage fish were landed. However, over the last 5 years China has become a major fishmeal processing country, importing raw material and selling the processed product on the domestic market (Madsen, 2004). Similar changes have happened in the supply of fishmeal, with China being the world's largest consumer of fishmeal, and Denmark and Thailand also increasing their supplies of fishmeal, while in Japan supplies are declining (Figure 4).

Peru and Chile have emerged as major fish oil producers, while Japan and Norway have seen declines in their production. Norway is no longer a major producer, and it imports significant amounts. Indeed, it is the largest consumer of fish oil, followed by Chile. These trends reflect the demand in Chile and Norway for fish oil for their aquaculture sectors that are focused on salmon.

Fishmeal and fish oil are highly traded products, with much of the material exported from Latin America. Exports of fishmeal have increased much faster than fish oil exports, especially in Latin America. There is little variation in the trade of fishmeal and fish oil from North America despite the increase in aquaculture in Canada (salmon) and the United States (catfish).

Exports of fishmeal and fish oil from Latin America make a significant contribution to the national economies of Peru and Chile. The impact these exports have on coastal fishing communities is mixed. The factories processing the fish are major sources of water and air pollution, affecting the living conditions of coastal communities (Cerda and Aliaga, 2000). These same factories provide alternative employment for fishers; factory workers may have adequate incomes in countries where poverty is prevalent. Improving the lives of fishers and the environment in which they live can be improved by upgrading the factories to pollute less. However, it is more challenging to improve the incomes of the people involved in fishing and processing, because the price of fishmeal is determined by global factors including the price of soymeal. Increasing the price of fishmeal too high above soymeal can reduce demand, since animal producers can substitute soy for fishmeal and fish oil in animal feeds.

The aquaculture sector has increased its use of fishmeal in feeds since the early 1990s (Table 5), but there is still considerable scope for the sector to increase its use of fishmeal, since only half of global production is used in aquaculture, with the other half primarily used by the pig and poultry sector. These sectors can substitute other meals such as soy for fishmeal and do so depending on the price of these commodities. How much more fishmeal can be diverted from pig and poultry will in part depend on the price of fishmeal and soy meal as well as on the increase in price consumers are willing to pay for poultry and pigs, which in terms of real costs are much lower than in the past, in part due to cheap feeds. It has been predicted that there is considerable room for diverting fishmeal from poultry to aquaculture, which currently accounts for approximately 23%. Clearly, as fishmeal demand increases for the aquaculture sector, the price will increase and pig and poultry producers will consider how the unit cost of production will change given the increased risk of disease and lower meat quality when substituting soymeal for fishmeal.

There is a trade-off in replacing fishmeal and oil with plant-derived products; studies have shown that for young poultry and piglets, using fishmeal and fish oil in the diet increases disease resistance, decreases the impact of the disease if contracted, decreases the severity of inflammatory diseases, improves the nutritional status of animals leading to better quality and leaner meat, and improved overall production through better growth and food conversion rates making the unit cost of production lower compared to diets of exclusively plant-based meals (IFFO, 2006).

The situation for fish oil, however, is not the same. Recent 2003 estimates allocate 87% of global fish oil production currently consumed to the aquaculture sector (Tacon, 2005), with much of it used for carnivorous fish (Table 1), and predictions indicate an increase to 88% by 2010 (FIN, 2006). There is also limited scope for increasing fish oil production; in fishmeal higher trophic level fish and fish that would otherwise be discarded can be used to produce fishmeal, but the recovery of fish oil from these fish is much lower than for targeted species such as Peruvian anchoveta, Capelin and herring. There is concern over the reliance on fish oil within the industry, especially in terms of expanding production of carnivorous species as well as introducing new species. The industry is continually developing new feeds and improving feed conversion ratios to reduce the amount of oil required, as well as searching for alternatives derived from cultured algae. Tacon (2004) indicated that by 2010, feed conversion efficiency should decrease so that the use of fish oil in feeds should be reduced by 8% for salmon. However, considering how fast aquaculture has expanded over the last 15 years, a saving of 8% will not be sufficient to allow for much more expansion of salmon and other carnivorous species unless conversion ratios are improved significantly and affordable alternative feeds and oils are developed.

Table 5. Use of fishmeal and fish oil in culturing selected finfish and crustaceans (thousand t, as fed dry weight). Based on Tacon (2005), who used various sources of information for each year so that comparison of individual years should be conducted with caution.

	Year								
	1992	**1994**	**1995**	**1998**	**1999**	**2000**	**2001**	**2002**	**2003**
Total fishmeal	963	1084	1728	2256	2091	2365	2585	2591	2963
Total fish oil	234	380	494	649	662	635	669	742	802
Salmon fish oil	60.4	169	176	265	273	285	282	530	409

The use of fishmeal and fish oil in aquaculture has increased in most of the countries with a significant aquaculture industry. Asia is the exception, where bycatch are often fed directly to high valued species as noted in China (Grainger *et al.*, 2005), Japan (Satoh, 2004), Thailand and Vietnam (FAO, 2005). The benefits of decreasing the amount of bycatch disposed at sea are debatable. Some argue that it is a better to use the fish than to return them to the ecosystem, others argue that the biomass returned to the sea is beneficial as it will be recycled by other organisms (Cushing, 1984). In some countries, fish bycatch is also a cheap source of food as seen in Ghana and therefore diverting bycatch can threaten local food security (Atta-Mills *et al.*, 2004).

The recent warnings of toxins in farmed salmon (Hites *et al.,* 2004) are not without merit, in light of the considerable trade that occurs in fishmeal and fish oil. The process of reducing small fish to fishmeal and fish oil concentrates toxins such as dioxins in these products which when fed to animals. Therefore in areas where there is a high concentration of dioxins and large catches of forage fish as shown in Figure 9 of Chapter 6, fishmeal and oil from these areas are likely to have high levels of dioxins (see also Chapter 5; Christensen and Booth, 2006). The fish, and consequently fishmeal and fish oil from Northern Europe contain the higher levels of dioxins compared to Latin America and North America and therefore, when these processed products are exported to other areas of the world, such as North America and Asia, dioxins and other toxins are also transported and enter the intensive animal food production system and ultimately the human food system (Figure 9 and 12).

In countries with high aquaculture production, some farmed species that are carnivorous or of high value are often fed bycatch directly. However, there has been a trend in some countries to feed omnivorous and herbivorous fish aquafeeds containing fishmeal and fish oil to promote faster growth for a better return on investment as seen in China (Sorgeloos, 2000). This is only possible if fishmeal and fish oil prices are low.

CONCLUSION

The expansion of intensive food production, especially aquaculture over the last 25 years has driven increases in the production, trade and consumption of the products derived from forage fish. The low price of fishmeal and fish oil relative to other meals such as soy has made this growth possible. However, there is little scope for increasing fishmeal and fish oil production, since most of the fisheries that are exploited for the reduction are fished at their biological limits.

The destination of fishmeal and fish oil is a function of the ability to meet the international price of fishmeal and oil as seen in China, whose its booming economy has made it possible for the country to become the largest importer of fishmeal (over 10% of world production). Chile has been able to capitalize on being one of the world's major suppliers of fishmeal and fish oil, with low transportation costs helping to turn the country into a major producer and exporter of farmed salmon as well as a major exporter of fishmeal. Its position as a significant processor of fishmeal is being challenged by China, which is developing its fishmeal sector by importing fish, building factories close to shipping ports with few environmental regulations, and providing a large cheap labour force. The future of this development relies on being able to source cheap raw materials as well as being able to keep transportation costs low.

The limited supply of fishmeal has seen a shift from it being used for intensive production of poultry and pig to aquaculture, and it is predicted that the use of fishmeal will continue to increase at the expense of the poultry sector. Aquaculture is the main consumer of fish oil and its use in aquaculture is predicted to continue to increase since there are fewer alternatives compared to fishmeal. This has prompted the aquafeed industry to search for alternatives to fish oil, however, when these alternatives will be commercially available is unknown.

The expansion of aquaculture will continue to influence the production, trade and consumption trends. How these trends will change depends on a number of international factors including the international price of soymeal and fuel, and food quality and safety standards which are all currently in a state of flux.

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APPENDICES

Appendix 1

Countries analyzed in trade matrices

Export Countries		Import Countries (Principal)	
Fishmeal	**Fish Oil**	**Fishmeal**	**Fish Oil**
North America	**North America**	**North America**	**North America**
Canada	Canada	Canada	Canada
USA	USA	USA	USA
Northern Europe	**Northern Europe**	**Europe**	**Europe**
Norway	Norway	Norway	Norway
Denmark	Denmark	Denmark	Denmark
Iceland	Iceland	Germany	Germany
Netherlands	Netherlands	United Kingdom	United Kingdom
Germany	Germany	Netherlands	Netherlands
United Kingdom	Sweden	Belgium	Belgium
Ireland	United Kingdom	Sweden	Sweden
Belgium	Faroe Isl.	South Europe & Mediterranean	Southern Europe
	Belgium	Switzerland	
		Finland	**Latin America**
Latin America	**Latin America**	Fmr Yugoslavia	Mexico
Peru	Peru	Fmr Czechoslovakia	Venezuela
Chile	Chile	Fmr USSR	Colombia
Panama	Panama	Eastern Europe	Brazil
Mexico	Mexico		El Salvador
Ecuador	Ecuador	**Latin America**	Guatemala
Argentina	Argentina	Mexico	Ecuador
Uruguay	Uruguay	Venezuela	Chile
Brazil	Brazil	Colombia	Peru
Paraguay	Venezuela	Brazil	Argentina
		El Salvador	Dominican Rep
Africa		Guatemala	Honduras
So. Africa		Ecuador	Belize
Angola		Cuba	Uruguay
Morocco		Suriname	
Mauritania			**Asia**
Namibia		**Asia**	China
Algeria		China	China, H K SAR
		China, H K SAR	Japan
		Japan	Rep. of Korea
		Rep. of Korea	Indonesia
		Indonesia	Philippines
		Philippines	Singapore
		Singapore	India
		Thailand	Thailand
		Malaysia	Malaysia
		Taiwan	

Appendix 2

Fishmeal Trade Data Rules and Definitions

'Significant' was defined as a >10% variation in quantity values and >5% in ratio values.

The FAO WATM data was only available configured in a trade matrix format for the years 1997-2003; before then data was provided as country total exports or imports for a given year (t). The data prior to 1997 were used to compare the accuracy of reported quantities between databases, and for obtaining proportional ratios for removal of the meat meal component from the Comtrade data.

In FAO, trade quantities are rounded to the nearest whole unit (FAO, 2006a).

The UN Comtrade database also differs in quantity reported from the FAO's databases, and depending on year or country reporting, this quantity may be higher or lower form the FAO quantity reported. This database lists its most complete data set in a Standard International Trade Classification category that includes both fish and meat meals. SITC classifies commodities based on their stage of production, and reports quantities in kg net weight (UN, 2006). The quantities are derived using either direct or standard unit conversion depending on the format of the data sent by the reporting country.

The UN Comtrade data was used as it was configured in a trade matrix format based on country reporting, and included all trade partners globally with data from 1962-2004. The 0814 category (meals unfit for human consumption) was adjusted in the compiled dataset used in this study to reflect only fishmeal quantities using the proportion of meat meal to the combined meat and fishmeal total export quantities in the FAO Agriculture database (FAO 2006c) and subtracting this from Comtrade data.

While it was assumed that all databases were consistent in their quantities reported (and therefore in their proportions of meat and fishmeals in the categories that combined the two as compared to the categories that had these commodities separate), this was not always, the case. This may be due to differences in data collection techniques and categorizations between the different departments.

Appendix 3

Ground Truthing Fishmeal Data

Data from 1964-1968 for a sample of exporters and their partners (Macer 1974) were plotted against the quantities derived from the compiled, adjusted dataset, for the purposes of determining data accuracy. Macer's 1974 study obtained its totals directly from the FAO. Peru and Chile were found to have virtually the same (99.9 and 98.1% respectively) total export quantities. The percent of exports sent to the UK and the Netherlands were also accurate to within a similar percentage. Trade to the USA and Germany was accurate between datasets for Chile, but these countries differed between datasets for Peru by 12% and 17% respectively. Norwegian exports were similar to within 95%. Comparisons using Iceland and Denmark yielded dissimilar results, with total exports from both datasets differing by 24 and 27% respectively. Imports to Denmark differed by between 12 and 23% originating from Iceland, and 27 and 29% from the UK and the Netherlands. The USA and Germany yielded a similar percentage for Denmark. All countries studied displayed the same trend of slightly higher export totals in the adjusted dataset compared to the FAO-derived quantities, and slightly lower import percentages. The exception to this trend was Germany trading from Norway, Iceland, and Denmark.

CHAPTER 5

GLOBAL DISPERSION OF DIOXIN: A SPATIAL DYNAMIC MODEL, WITH EMPHASIS ON OCEAN DEPOSITION[5]

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ABSTRACT

The present study models the dispersion and deposition of airborne dioxin, a known carcinogen, on a global scale using atmospheric transport, with emphasis on ocean deposition. Human exposure is primarily through diet, and recent reporting of high levels in farmed seafood and fishmeal and fish oil products has increased concerns. A previously-published estimate of global dioxin production (13,100 kg·year^{-1}) was allocated as airborne dioxin to terrestrial ½ x ½ degree GIS cells based on a linear relationship between dioxin production and GDP. Dioxin was then dispersed and deposited globally to both land—and water—GIS cells using two seasonal, 10-year time-averaged wind patterns. Our findings suggest that approximately one-third (4,214 kg·year^{-1}) of globally-produced, and atmospherically-transported, total dioxins are annually deposited directly into marine environments. Based on published Characteristic Travel Distances and decay functions for dioxins, it would take over 270 days for the estimated total annual dioxin production to be deposited onto the Earth's surface, and our wind dispersion patterns supported previous empirical observations of regional hot spots of marine dioxin loads (e.g., NE Atlantic). Comparison to field observations taken from terrestrial samples of deposited dioxins suggested relatively good fit, with measured concentrations on average being approximately 20% higher than modeled data suggested. It has previously been held that the majority of dioxin emissions is deposited near the source of emission, and only some dioxins are known to distribute globally. Our study suggested that the combination of average wind direction and predominantly coastal or near-costal location of high GDP areas facilitate the oceanic dispersion of over 30% of the estimated total dioxin production, which thus becomes directly available to the marine food webs.

INTRODUCTION

Persistent Organic Pollutants (POP), such as DDT, dioxins and PCBs, are chemicals which generally pose health risks to humans and animals (Foran *et al*., 2005). Typically, POP are of environmental concern because they are persistent, biomagnify and bioaccumulate in food webs, and have adverse effects on human health. Dioxins and furans are a class of POPs that are known carcinogens and have other sub-lethal effects in animals and humans (Anon., 2000). The general public first became aware of and concerned about these chemicals as a result of the use of Agent Orange as a dioxin containing defoliant used in Vietnam, and later due to industrial accidents such as in that Seveso, Italy (Baker and Hites, 2000).

The chemical structures of dioxins and furans are similar and consist of 75 polychlorinated dibenzo-*para*-dioxins (PCDD) and 135 polychlorinated dibenzofurans (PCDF). These two classes of POPs are usually combined in analysis when measurements are made on contaminated animal tissues, although in some

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cases dioxin-like PCBs are also included. In the context of the present chapter, 'dioxin' is meant to include only the 75 congeners of dioxins and the 135 congeners of furans.

The congeners of dioxins are all lipophilic to varying degrees, and thus they bioaccumulate particularly well in oily tissues. However, they have different toxicology. Therefore, in order to assess the detrimental health effects brought about by exposure to the different forms, a relative ranking system is used in toxicological studies. This assigns toxic equivalency factors (TEFs) to the different forms relative to the most toxic congener, 2, 3, 7, 8-tetrachlordibenzo-*p*-dioxin (TCDD), which is given a TEF of one (Van den Berg *et al.*, 1998). In this way, a total relative concentration in tissues can be assessed in terms of toxic equivalencies (TEQs). However, TEFs have been determined for only the 17 congeners of dioxin which are at least chlorinated in the 2,3,7,8 positions and of most toxicological concern (Buckly-Golder, 1999).

Dioxins are not deliberately produced for any specific human use, but rather are formed as incidental by-products during industrial processes, including the production of chlorinated chemicals, and as the result of combustion. Formation of dioxin during combustion is important globally, since some dioxins are known to disperse over long distances, primarily via atmospheric transport, although riverine inputs can be important in localized areas (Anon., 2004). Many dioxins are thought to be deposited relatively close to the source of emission (Brzuzy and Hites, 1996). While chemical sources of dioxin pollution (such as wood pulp bleaching) have resulted in localized environmental contamination, it is primarily through combustion that wide-spread contamination occurs due to atmospheric transport (Brzuzy and Hites, 1996).

For humans, approximately 90% of exposure is from dietary items, with fish, beef and dairy products being of prime concern (WHO, 1999). The World Health Organization (WHO) has set a Tolerable Daily Intake (TDI) of 1-4 picograms per kilogram bodyweight (Anon., 2000). The European Union's limit for dioxin in fish is 4 $pg \cdot g^{-1}$ wet weight, while livestock and dairy products have a limit of 1-3 $pg \cdot g^{-1}$ lipid base (Lind, 2004).

A global model of general emissions and depositions of dioxin homologues noted that these two processes were not mass balanced (Brzuzy and Hites, 1996). Global estimates of annual dioxin depositions were thought to be approximately four times greater than estimated emissions (13,100 kg vs. 3,000 kg, Brzuzy and Hites, 1996). However, it has since been shown that atmospheric photochemical reactions of anthropogenic-produced pentachlorophenol (PCP), a common wood preservative, can produce dioxin, and this atmospheric formation of dioxin from PCP is thought to account for the imbalance (Baker and Hites, 2000).

The focus of our study is the presence of POPs, specifically dioxins, in small pelagic or 'forage' fish (herrings, sardines, anchovies, mackerels, etc; see Chapter 1), which are the main source for fishmeal and fish oil production for intensive aquaculture and poultry and swine production. Farmed salmon have been found to contain significantly elevated levels of dioxin compared to wild salmon, with European farm-raised salmon showing significantly higher levels of contamination than their North and South American counterparts (Hites *et al.*, 2004). In order to account for the observed geographic differences in dioxin loads in farmed and wild-caught salmon (and other aquatic and agricultural products), we developed a GIS-based modeling approach to map the estimated global dispersion of dioxins via atmospheric transport. Given the lack of empirical measurements of ocean deposits of dioxins (Brzuzy and Hites, 1996), and the paucity of global dispersal model for POPs, we developed a global, spatially dynamic model for predicting global atmospheric dispersal and deposition of dioxins[6]. The aim was to predict the global dispersal of land-generated dioxin, and resultant deposits into the oceans, using global wind patterns and published travel-distance and deposition rates.

We utilized the global spatial resolution model of ½ x ½ degree spatial cells defined and used by the *Sea Around Us* project (SAUP, 2006). The basic methodology developed in this study rested upon several fundamental assumptions:

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[6] Note that, for the present, we did not include the effects of river run-off, which would increase coastal concentrations near major estuaries. Given the fast biogenic uptake of dioxins from water, this was assumed to have impacts primarily on ecosystems close to river deltas only.

- All dioxin was produced over land, and the amount based on published information;
- Spatial distribution of global dioxin production was directly related to level of economic activity of any given area, i.e., spatial dioxin production was assumed to be proportional to GDP;
- Dioxin dispersal was by atmospheric transport, driven by and influenced by average wind patterns in the lower atmosphere; freshwater run-off was not explicitly considered; and
- Rate of deposition of dioxin to the earth surface (land or water) followed published relationships for characteristic travel distances for dispersion, but included simulation of diffusion. We did not differentiate between wet and dry deposition.

The resultant estimate of global annual concentration of dioxin entering the world's marine systems at the ½ x ½ degree resolution via atmospheric dispersion was used as input parameter into marine systems in the trophic modeling undertaken in other sections of this report.

METHODS

Global dioxin emission distribution

As starting position for our modeling simulations, we took the estimated mean total global deposition of dioxin (chlorinated dioxins and dibenzofurans) from the atmosphere of 13,100 ± 2,000 kg·year^{-1} as derived by Brzuzy and Hites (1996), and assumed this amount corresponded to the annual global production of dioxins by all sources. Hence, this was thought to also include those forms of dioxin that are known to form in the atmosphere via photochemical synthesis from pentachlorophenol (Baker and Hites, 2000). We distributed this total estimated annual dioxin load to all global land cells (½ x ½ degree) using the assumption of linear relationship between GDP and dioxin production (Baker and Hites, 2000).[7] Global, spatialized GDP data were based on Dilley *et al.* (2005). It was assumed that the spatially distributed dioxin amounts were fully airborne above their assigned ½ x ½ degree land cells at the start of the dynamic dispersal phase. For technical details of the global dispersion model, see Appendix 1.

Simulated dispersion and deposition of airborne dioxin

Dioxin was dispersed from its starting distribution in the atmosphere above land cells by modeling atmospheric transport driven by global lower atmosphere wind patterns. Two global seasonal wind patterns, representing the two main global seasonal and monsoonal periods (April-September and October-March), were obtained from the European Centre for Medium-Range Weather Forecasts (Anon., 2006). We averaged these data for the 1990-2000 time period, and recomputed the original vector-based data for present use into direction and velocity components by season (Figures 1, 2). The two seasonal global distribution models (April-September and October-March) were run separately, and thus half of the global estimate of dioxin production of 13,100 kg·year^{-1} (Brzuzy and Hites, 1996) was dispersed in each seasonal run, and the final deposited amounts from each run were combined to derive annual total deposits. Hence, we assumed equal dioxin production potential for each season. The model of dioxin dispersal included the use of a wind-speed dependent characteristic travel distance for dioxin to derive an effective decay rate and resultant deposition (Bennett *et al.*, 1998; Beyer *et al.*, 2000). We also incorporated diffusion, using published diffusion coefficient in air for dioxins (Chiao *et al.*, 1994).

Each seasonal model was run for 200,000 time steps (each time step representing 60 s in real time, see Appendix 1), which is equivalent to approximately 139 days or 4½ months in real-time, after which over 99% of the seasonal atmospheric dioxin had been deposited into either land or water cells. Subsequently, the outputs from the two seasonal models were combined to derive the total annual dispersion and deposition of dioxin to both land and ocean cells.

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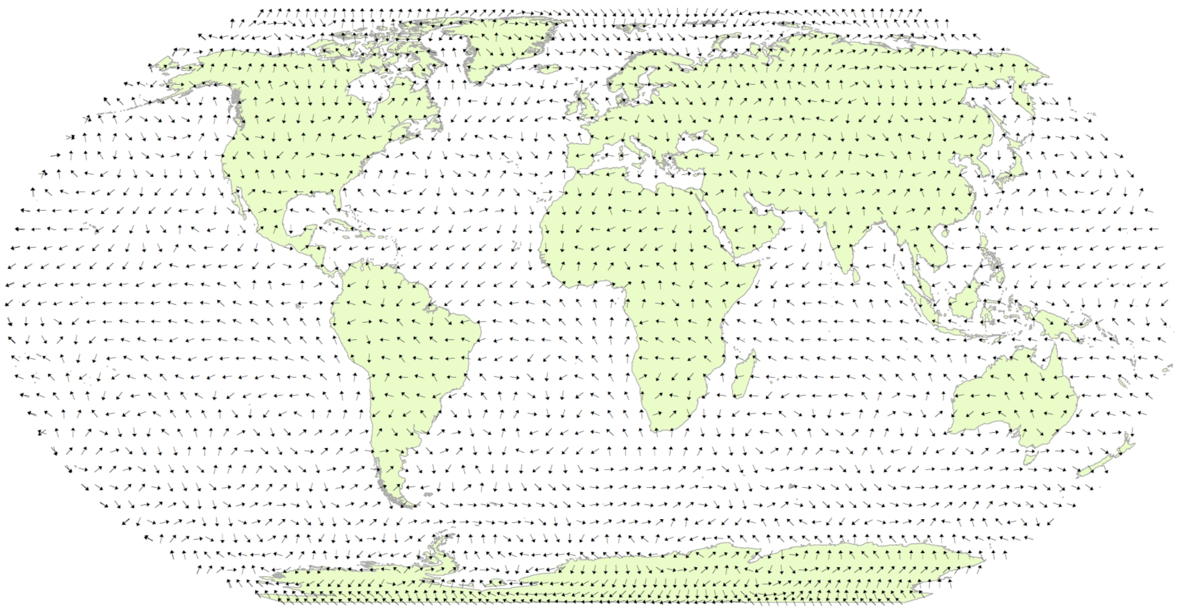
[7] Baker and Hites (2000) found a close correlation (r^2 = 0.8) between the logarithm of GDP and the logarithm of annual dioxin production per country examined. As their log-log relationship suggested a slope of close to unity, we assumed a linear relationship between GDP and dioxin production.

a)



b)

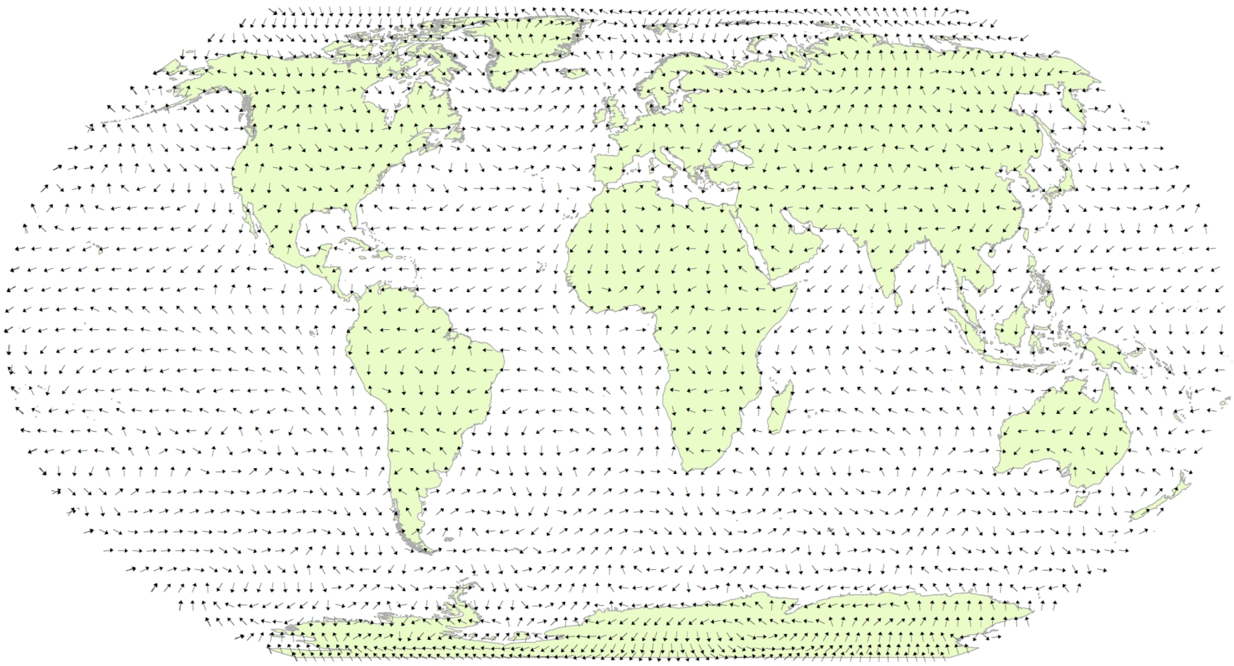


Figure 1. Global, lower atmospheric wind patterns, separated into two seasons: a) April-September; and b) October-March, reflecting the two predominant global seasonal and monsoonal periods. Shown are wind direction only, aggregated to 5 degree scale. Note that model runs were undertaken with wind resolution at ½ x ½ degree cells. Source: European Centre for Medium-Range Weather Forecasts (Anon., 2006).

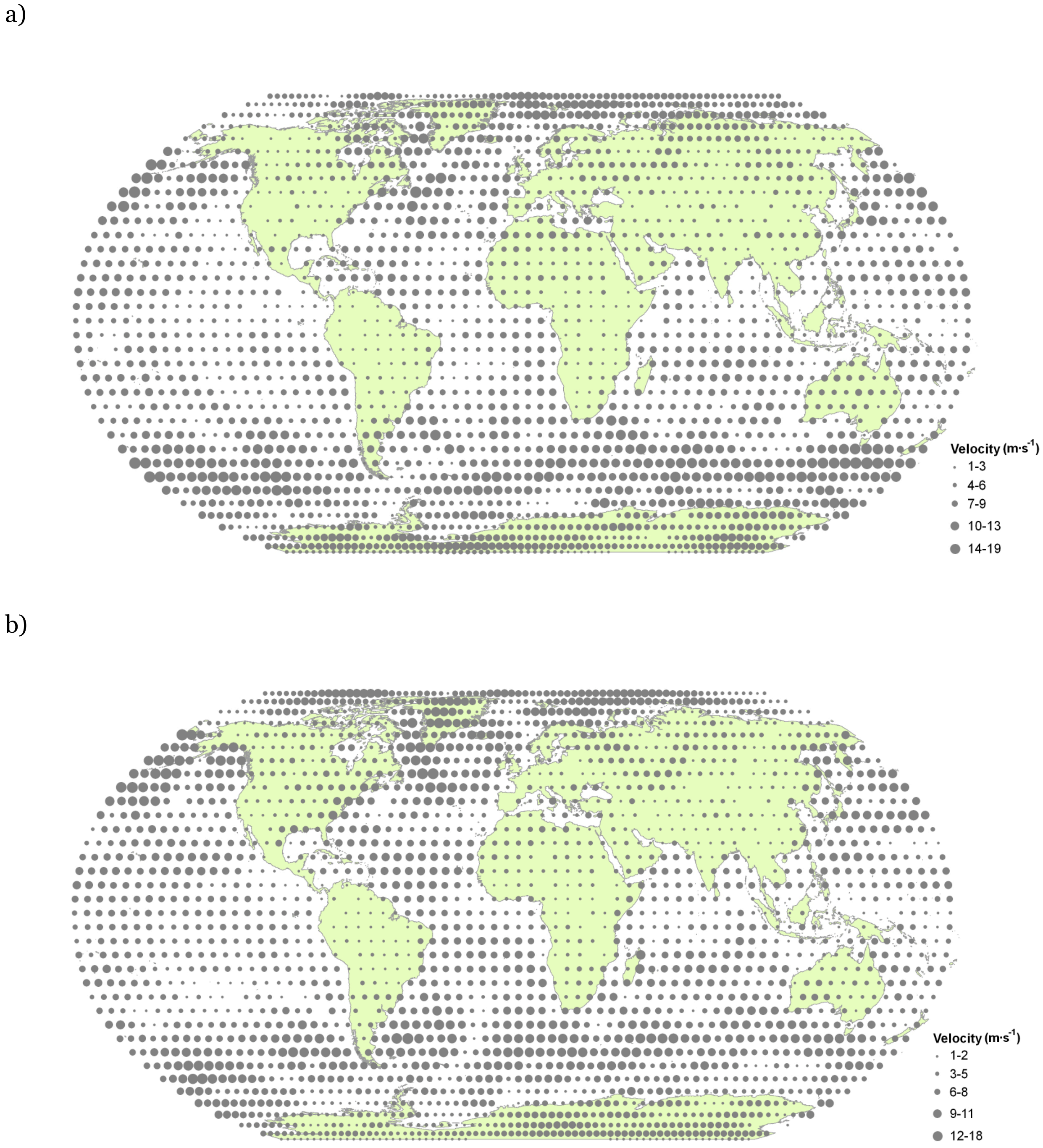


Figure 2. Global, lower atmospheric wind patterns, separated into two seasons: a) April-September; and b) October-March, reflecting the two predominant global seasonal and monsoonal periods. Shown are wind strengths only, aggregated to 5 degree scale. Note that model runs were undertaken with wind resolution at ½ x ½ degree cells. Source: European Centre for Medium-Range Weather Forecasts (Anon., 2006).

RESULTS

Global dioxin emission distribution

The approach used for initial distribution of airborne dioxin (starting position for dispersion modeling) was based on the observation that dioxin production is generally proportional to GDP of any given area (Baker and Hites, 2000). Applying this relationship to the spatial distribution of the estimated global atmospheric dioxin load of 13,100 kg·year^{-1} to the ½ x ½ degree land cells, suggested several areas of likely high atmospheric dioxin concentration due to high local production of dioxins (Figure 3). These were dominated by eastern North America, Europe, South Asia (particularly the Indian subcontinent), and East Asia (China, Japan and South Korea).

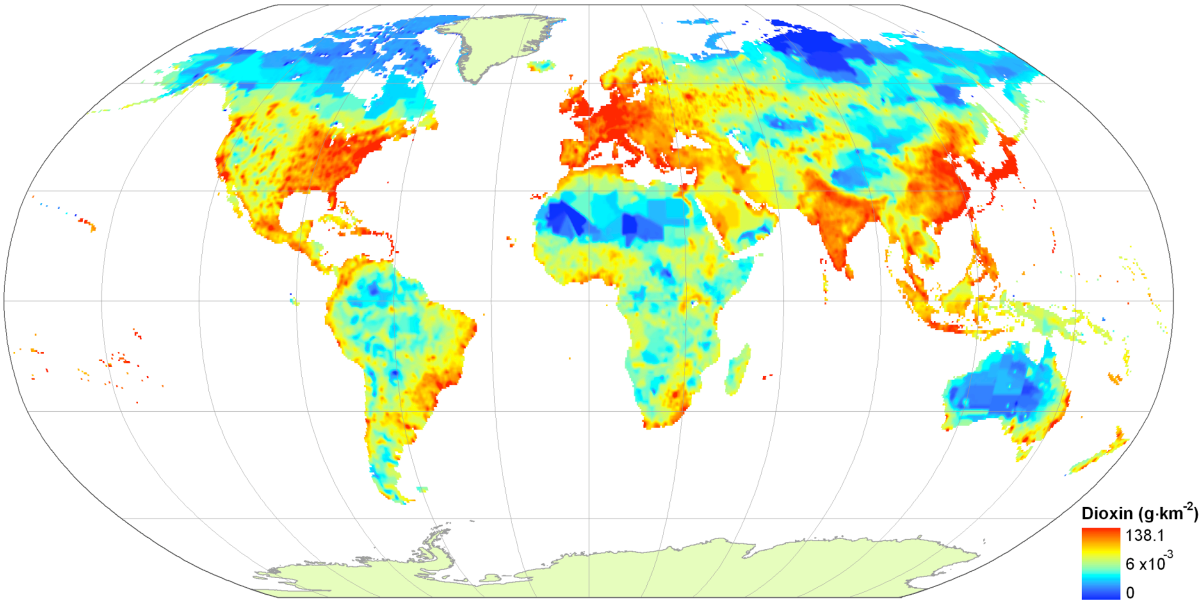


Figure 3. Estimated spatial distribution pattern of global dioxin production, based on the published linear relationship between dioxin production and GDP (Baker and Hites, 2000).

Dispersion and deposition

The dispersion, diffusion and decay model using published characteristic travel distance and diffusion coefficient for dioxins (Chiao *et al.*, 1994; Bennett *et al.*, 1998; Beyer *et al.*, 2000), resulted in 99.14% of airborne dioxin being deposited during the two seasonal runs of 200,000 time steps each (~139 days real time each (Table 1)). Thus, the annually-produced 13,100 kg of dioxin would take approximately 278 days (2 • 139 days) for over 99% to be deposited onto the earth's surface. However, atmospheric deposition is not a linear process; the majority of dioxin is deposited within a much shorter time period, with approximately 50% and 75% of airborne dioxin deposited within 5 and 13 days of emission, respectively (Figure 4). Given the terrestrial sources and the non-linear deposition function, the distribution between land and ocean deposition was uneven, with 67% and 32% of deposited dioxin, respectively (Table 1). Thus, our model suggests that, on average, approximately one third of globally-produced dioxins are deposited directly in the ocean due to atmospheric transport.

Table 1: Summary of seasonal and total dispersion and deposition of dioxins based on global atmospheric dispersal and reported atmospheric decay (deposition) functions.

Time period	**Initial dioxin in air (kg)**	**Amount of dioxin after simulation runs[a] (kg)**			**Percentage of airborne dioxin deposited**
		Air	**Land**	**Ocean**	
April-September	6,550	62.2	4,572.3	1,915.5	99.05
October-March	6,550	49.8	4,201.3	2,298.9	99.24
Annual combined	13,100	112.0	8,773.6	4,214.4	99.14
Percentage of total dioxin deposit			66.97	32.17	

[a] Seasonal runs were for 200,000 time steps, being approximately 139 days real-time each, while annual totals equate to 278 days (2 • 200,000 time steps).

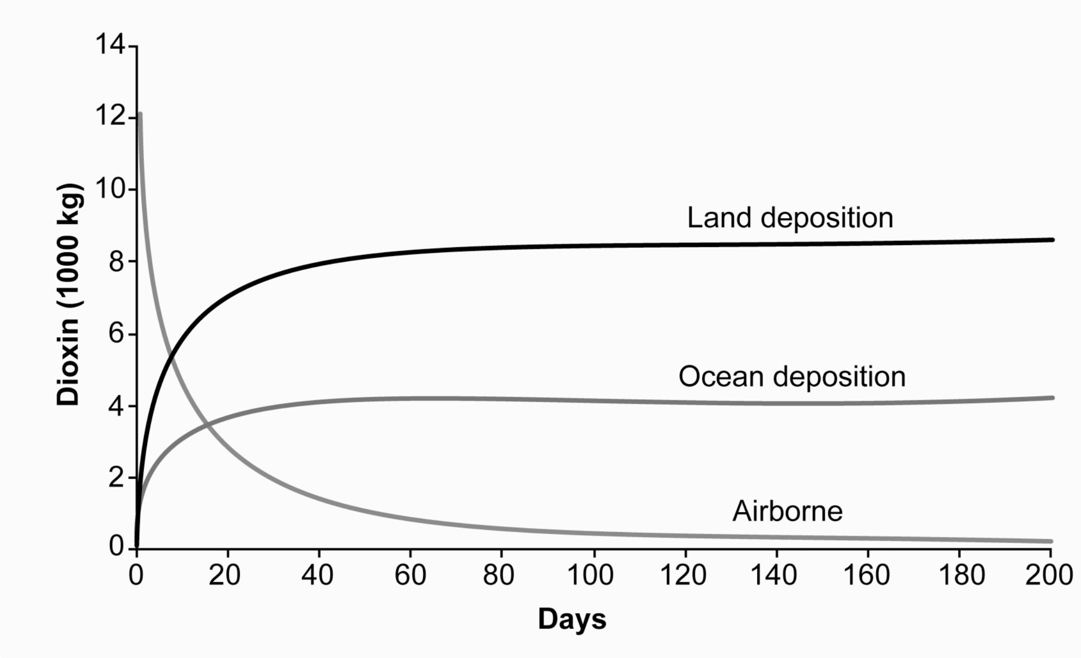


Figure 4. Amount of dioxin in the air, and deposited into the ocean and land cells over time, based on the dispersion and diffusion model with the two seasonal models pooled.

Spatial patterns of dispersion

Based on the two seasonal, 10-year average wind field patterns, the global distribution of modeled dioxin deposition suggested several terrestrial and oceanic hot spots (Figure 5). Specifically, 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 5a). Under the assumption that the published characteristic travel distance and effective decay rate of dioxins as measured under terrestrial conditions also applied to oceanic conditions, it suggested that many ocean areas around the world may also have relatively high dioxin loads. These include the northeast and west Atlantic, Caribbean, Mediterranean, northern Indian Ocean, and large parts of the north-western Pacific and South China Seas (Figure 5b). However, several areas of relatively low concentration of dioxins also emerged, specifically parts of the west coast of South America and northern parts of the west coast of North America (Figure 5b). Our model suggested that final concentrations of deposited dioxins in our ½ x ½ degree model cells ranged from as low as $4.5\cdot10^{-40}$ g·km^{-2} to as high as 7.5 g·km^{-2} for marine environments, with the April-September seasonal run showing a larger range of concentrations than the October-March season (Table 2).

Table 2: Minimum, maximum and mean deposited dioxin concentrations per ½ x ½ degree cells, separated by season and land-versus-ocean cells.

	Deposited dioxin concentration per cell (g·km^{2})					
	Water		Annual	Land		Annual
	April-September	October-March		April-September	October-March	
Minimum	$4.46\cdot10^{40}$	$2.02\cdot10^{39}$	$8.40\cdot10^{31}$	$7.11\cdot10^{61}$	$1.51\cdot10^{33}$	$1.51\cdot10^{33}$
Maximum	7.46	3.21	7.54	55.39	6.77	55.80
Mean	$5.65\cdot10^{3}$	$6.81\cdot10^{3}$	$1.25\cdot10^{2}$	$2.26\cdot10^{2}$	$2.14\cdot10^{2}$	$4.40\cdot10^{2}$

Terrestrial environments had concentrations from $7.11\cdot10^{-61}$ g·km^{-2} to 55.39 g·km^{-2}, with the larger range of concentrations also observed during the April-September period (Table 2). Global annual average concentrations were $1.25\cdot10^{-2}$ and $4.4\cdot10^{-2}$ g·km^{-2} for marine and terrestrial areas, respectively (Table 2). Note, however, the difference in scale between terrestrial and oceanic deposition (Figure 5a,b), confirming that, globally, most dioxins were deposited on land, close to their area of production (Brzuzy and Hites, 1996).

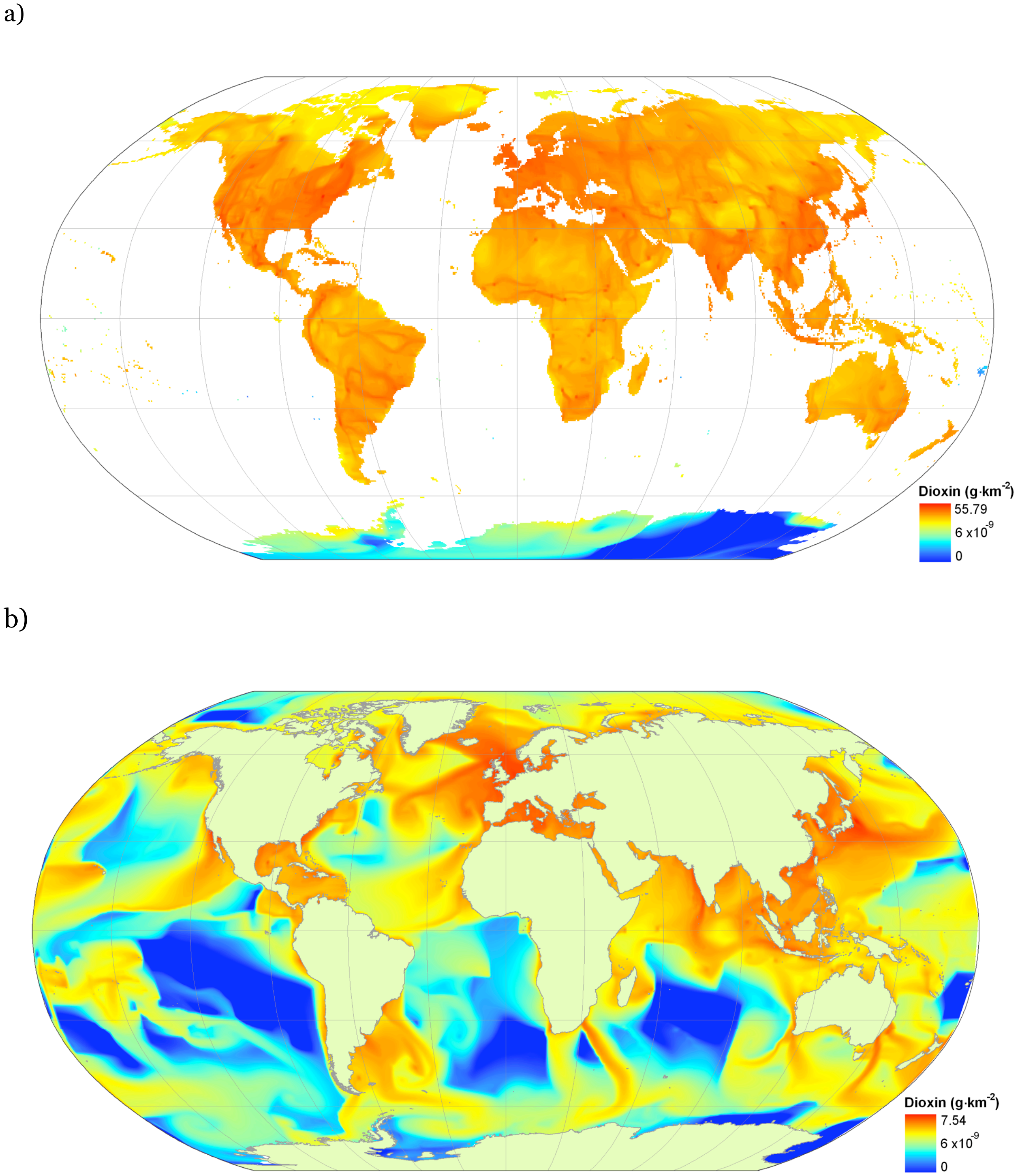


Figure 5. Global distribution of modeled dioxin deposition, based on the two seasonal, 10 year average wind field patterns combined, showing spatial distribution (a) on land, and (b) in the ocean (note difference in scales).

Validation of spatial dispersion

In order to determine how well our dispersal and deposition algorithm replicated real-world distribution and deposition of airborne dioxins, we compared 107 empirical field measurements taken from soil samples reported by Brzuzy and Hites (1996) with the values predicted from our model runs for the same spatial coordinates. As no empirical dioxin measures exist for seawater samples, we had to rely on terrestrial validations only. Due to our spatial resolution of ½ x ½ degree cells, 11 model cells were coastal in location, i.e., did not correspond to land locations. These samples were excluded from the present comparison, resulting in 96 actual measures being compared between modeled results and the corresponding empirical terrestrial field measurements. This comparison (Figure 6a) showed a significant positive relationship between empirical and model results with a moderate fit ($p<<0.001$, $R^2=0.203$). On average, empirically-measured dioxin concentrations in soil were 20.6% higher than our modeled results suggested. The observed relationship could likely be improved by weighing the initial GDP-related distribution of dioxin production by an indicator of environmental and pollution standards by country, as our results suggest possible differences related to these standards. For example, European countries, which have relatively well established pollution mitigation standards, had measured values generally lower than suggested by our model, while North American countries (Mexico, Canada and USA), with their more diverse range of environmental standards, showed more variable patterns. The few available empirical measurements from African countries suggested our model slightly underestimated their empirical dioxin loading (Figure 6b), possibly because our assumptions about dioxin being linked to GDP did not account for the dioxins produced by extensive burning of vegetation in many parts of Africa (e.g., Cahoon *et al.*, 1992). Furthermore, it has been suggested that by replacing the seasonally-averaged wind patterns by monthly patterns and simulations, as is commonly done in atmospheric chemistry, a better fit between empirical deposition and modeled results might be obtainable (D. Mauzerall, University of Princeton, *pers. comm.*).

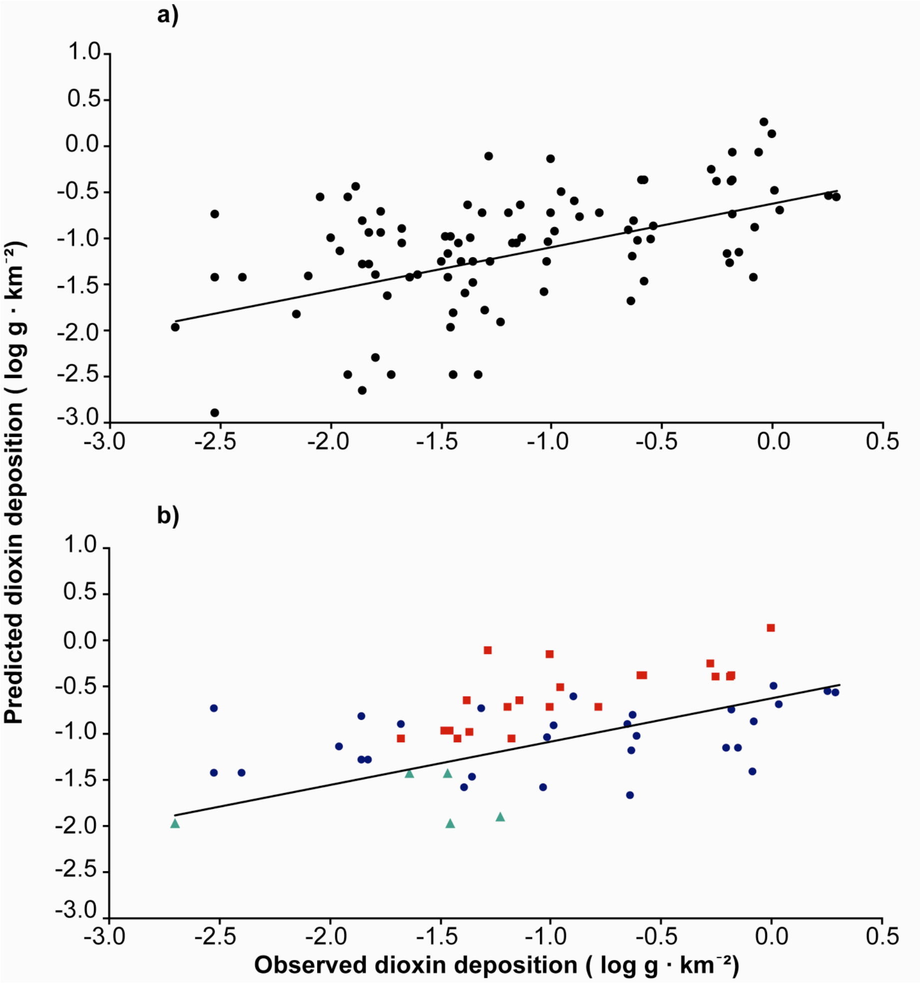


Figure 6. Model validation comparison: a) between empirically-measured terrestrial dioxin depositions based on Brzuzy and Hites (1996) and the geographic equivalent model results from the present dispersal and deposition model; b) a subset of (a) showing data from European (🟥), North American (🟦) and African (▲) countries only. The significant regression for the whole data set with moderate fit ($p<<0.001$, $R^2=0.203$, n= 96), is also shown. Data were log-converted to account for large number of small values. Note that field measurements exist only for terrestrial locations.

CONCLUSION

To our knowledge, this is the first attempt to model the dispersion and deposition of dioxins on a global scale using atmospheric transport. Our findings suggest that:

- Approximately one-third (4,214 kg) of globally-produced, and atmospherically-transported total dioxins are annually deposited directly into marine environments;
- Based on published Characteristic Travel Distances and decay functions for dioxins, it would take over 270 days for the estimated total annual dioxin production of 13,100 kg to be deposited onto the earth surface;
- Our wind dispersion patterns supported previous empirical observations of regional hot spots of marine dioxin loads (e.g. NE Atlantic); and
- Comparison to field observations taken from terrestrial samples of deposited dioxins suggested relatively good fit, with measured concentrations being on average approximately 20% higher than modeled data suggested.

It has previously been held that the majority of dioxin emissions appear to be deposited in climate zones near to the source of emission, and only some dioxins are known to be distributed globally, due to long-range atmospheric transport (Brzuzy and Hites, 1996). While fundamentally concurring with this observation, our study suggested that the combination of average wind direction and predominantly coastal or near-costal location of high GDP areas facilitates the oceanic dispersion of dioxins. This caused over 30% of the estimated total dioxin production to be deposited directly into the marine environment during our simulations, thus becoming directly available to the marine food web.

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APPENDIX

Technical details of the spatially dynamic model for predicting wind-driven global distribution of dioxin

Aims

To predict the global wind-driven dispersion of airborne dioxin and to determine the amount of dioxin deposited into the ocean and on land.

Methodology

To estimate the global dispersion and deposition of dioxin in the ocean and on land, we developed a spatially dynamic model with two driving parameters: a) anthropogenic dioxin production set proportional to GDP; and b) global wind patterns as the primary means of long-distance dispersal of dioxin. Details of each parameter and methodology are explained in the later sections of this document.

We initiated the simulations with terrestrially-produced, airborne dioxin distributed proportional to GDP to all land cells in a global, spatial grid system with ½ x ½ degree resolution, based on the global ½ degree cell system of the *Sea Around Us* project (given the spherical nature of the globe, this implied that cell surface areas and east-west dimensions varied with latitude).

Since wind direction can vary globally between seasons (e.g., tropical monsoons), we separated the wind data into two time periods: April-September and October-March. We assumed equal production of dioxin throughout the year, thus the initial starting amount of airborne dioxin (13,100 kg) was separated into two halves (6,550 kg) and assigned to each seasonal time period.

A spatially dynamic model was developed using Visual Basic (Microsoft vb.net) to predict the amount of dioxin transferred to neighbouring cells. We iterated the initial state of dioxin through all the cells in the world, applying two driving mechanisms: diffusion (independent of wind) and dispersion (wind-dependent). Within the dispersion module, a decay function was applied to account for deposition onto the earth surface (land or water). These mechanisms distributed dioxin during each simulation iteration (time-step) to a new model state, which has some dioxin in each cell remaining in the air, and some having been deposited (decayed) onto the ground or water. Once dioxin was deposited, it no longer was exposed to the distribution mechanisms. This new state was then reapplied to the distribution mechanisms, until ≥ 99% of the initial airborne dioxin had been deposited onto land/water.

Data

Below is a table that outlines the basic input data.

Table A1. Basic input data for spatial model

Data Sets	Type	Purpose	Source	Data Format (Resolution)
Average N-S wind component at 10m (April–Sept, 1990-2000)	Data Grid	Wind acts as dioxin dispersal force. Data were decomposed into wind direction and speed.	ECMWP[1]	NetCDF (2.5 degree)
Average E-W wind component at 10m (April–Sept, 1990-2000)	Data Grid	-	ECMWP[1]	NetCDF (2.5 degree)
Average N-S wind component at 10m (Oct–March, 1990-2000)	Data Grid	-	ECMWP[1]	NetCDF (2.5 degree)
Average E-W wind component at 10m (Oct–March, 1990-2000)	Data Grid	-	ECMWP[1]	NetCDF (2.5 degree)
Initial concentration of dioxin in the atmosphere	Interpolated from GDP	Initial amount of dioxin in the air. Global total based on literature (Brzuzy & Hites, 1996)	SAUP[2]	Raster (2.5 arc minutes)
World Table	Access	Percent water/land in each ½ degree cell	SAUP	Access table (0.5 degree)

[1] European Center for Medium Range Weather Forecasting. [2] GDP data based on Dilley *et al.* (2005)

Dioxin

The amount of dioxin generated globally was set as 13,100,000 g (Brzuzy and Hites, 1996). We assigned the total amount of dioxin to the air above all land cells throughout the world in proportion to the Gross Domestic Product (GDP) of each cell, based on the documented correlation between dioxin production and GDP (Baker and Hites, 2000). The GDP data had a resolution of 2.5 arc minutes (Dilley *et al.*, 2005) and was adjusted to 0.5 degree cells by interpolation**.** The interpolation was carried out using the Nearest Neighbour algorithm in ArcGIS.

Dioxins, like all airborne chemicals, are subject to simple diffusion. We simulated diffusion of dioxin based on the published diffusion rate for 2,3,7,8-TCDD dioxin (Chiao *et al.*, 1994),. Diffusion was the first operation applied in each computational time step to distribute dioxin between cells.

The modeling of dioxin dispersion based on wind speed, wind direction and characteristic travel distance was the second operation undertaken in each time step. Deposition of dioxin from air to land/ocean cells was accounted for with an exponential decay model (Bennett *et al.*, 1998). We applied the proportion of marine versus terrestrial environment per cell to the allocated dioxin deposition per cell to derive land and ocean deposition amounts.

Wind

The data consisted of global monthly daily means of east-west and north-south wind component data ($m \cdot s^{-1}$) from the '40 years Re-analysis Database' of the European Center for Medium Range Weather Forecasting (ECMWP)[1]. These data were further averaged over the 1990 to 2000 time period into two sets of values one set representing April to September and one set representing October to March. These values were applied to each 2.5-degree cell for each time period. The resulting NetCDF files were then converted into raster files and decomposed into wind magnitude and wind direction using the DECOMP module in IDRISI. The 2.5-degree resolution of the original wind data was converted to 0.5-degree cell resolution via the interpolation method of Nearest Neighbour.

In this document, μ is the wind speed in $m \cdot s^{-1}$ and θ is the wind angle (0 degrees being due east).

We are aware that the temporal resolution provided by 6-month wind fields is low, and will be re-doing these analyses using monthly wind grids, as is the standard in atmospheric chemistry.

Parameter Estimation

Latitude

Latitude (*lat*) is defined as the current latitude in the center of a given cell. The most northerly and southerly row of cells have the latitude of 89.75 and -89.75 degrees, respectively. The cell latitude for the two rows at the equator is 0.25 and -0.25 degrees, for north and south respectively.

Time step

The computational time step (*ts*) was defined as the amount of time allowed for dioxin to travel per iteration. We chose a time step that was less than the time taken for the highest wind speed (μ_{max}) to cross the smallest cell dimension (cd_{min}):

$$ts \leq \frac{cd_{\min}}{\mu_{\max}}$$

This was done to ensure that *ts* was not large enough for dioxin to move past any neighbouring cells in any one iteration. Given the latitudinal differences in east-west cell dimensions of the ½ x ½ degree cells used,

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[1] ECMWP website: http://data.ecmwf.int/data/index.html

the smallest dimension was taken as the east-west dimension for cells at 89.75 degree latitude (1320 m), with fastest global wind speed from both seasonal wind fields (21.3 $m \cdot s^{-1}$). Thus, minimum time needed at 89 degree was 61.97 s (i.e., 1320 m / 21.3 $m \cdot s^{-1}$). We reduced this to a standardized *ts* of 60 s for our runs. Thus, the number of time steps represents minutes in real time.

Characteristic travel distance

The characteristic travel distance (*CTD*) for a chemical is defined as the distance required for its concentration to be reduced to 1/e (~37%) of its initial value via deposition (Bennett *et al.*, 1998; Beyer *et al.*, 2000). The amount of dioxin in the air of a particular cell was reduced by both deposition and transport (diffusion and dispersion) to neighbouring cells. The *CTD* of dioxin for the most toxic congener 2,3,7,8-tetrachlorodibenzo-*p*-dioxin in air is 810 km at wind velocity of 4 $m \cdot s^{-1}$ (Beyer *et al.*, 2000), and was chosen as the dispersion rate for this model.

Cell distance

The cell distance (*d*) is the coordinal distance of one of the 720 equatorial cells of 0.5 degree, and is also related to the circumference of the earth (40,075.16 km/720 cells).

Wind distance

Wind distance is the distance the wind in a given cell travels during one time step. Wind distance was defined as:

$$\mu_{ts} = \mu \times ts$$

where μ represents the wind speed and *ts* represent the time step.

Diffusion

As noted above, diffusion is first applied to every cell. The amount of dioxin diffused in each time step was calculated as:

$$D_{diffused} = N_o \times D_{diffusedRatio}$$

Where N_o is the amount of dioxin in the air in a cell at the beginning of each time step, and $D_{Diffused\ Ratio}$ was calculated as:

$$D_{diffusedRatio} = \frac{\sqrt{2 \times DiffCoef \times ts}}{d \times \cos(lat)}$$

Where *lat* is the latitude of the current cell at the center of that cell, *ts* is the time step, and *DiffCoef* is the diffusion coefficient of dioxin equal to 0.42 $m^2 \cdot day^{-1}$ or $4.86 \cdot 10^{-6}$ $m^2 \cdot s^{-1}$ (Chiao *et al.*, 1994)**.**

$D_{diffused}$ was then divided into 4 equal values (i.e., 25% each), and was allocated to the air of the neighbouring cells in the cardinal directions (north, east, south and west) of the originating cell only. This diffused dioxin was then available for the next iteration in the receiving cells.

Decay (deposition)

The dioxin remaining in the air of a given cell after diffusion was exposed to deposition via decay. The amount of dioxin deposited to land and/or ocean cells from the air was calculated as:

$$D_{decay} = \left(N_o - D_{diffused}\right) \times D_{decayRatio}$$

Where:

$$D_{decayRatio} = 1 - e^{\mu \times ts / CTD}$$

Where *CTD* is the characteristic travel distance of 810 km at a wind speed of 4 m·s^{-1}. The resulting D_{decay} was then assigned to the ground/water of the current cell location and was not further exposed to any distribution.

Dispersion

The dioxin remaining in the air after diffusion and decay was then subjected to transport to neighbouring cells by wind-driven dispersion. The amount left for dispersion was calculated by:

$$D_{dispersion} = N_0 - D_{diffused} - D_{decay}$$

The amount of dioxin was then transported to neighbouring cells by a 'four cell method' (Figure A1). This method takes the entire rectangle that is above the cell and moves it in the direction of the wind at the reported wind speed, but limited by the time step duration. The proportions of the cell that had left the originating cell after *ts* was then calculated and moved to the corresponding neighbouring cells.

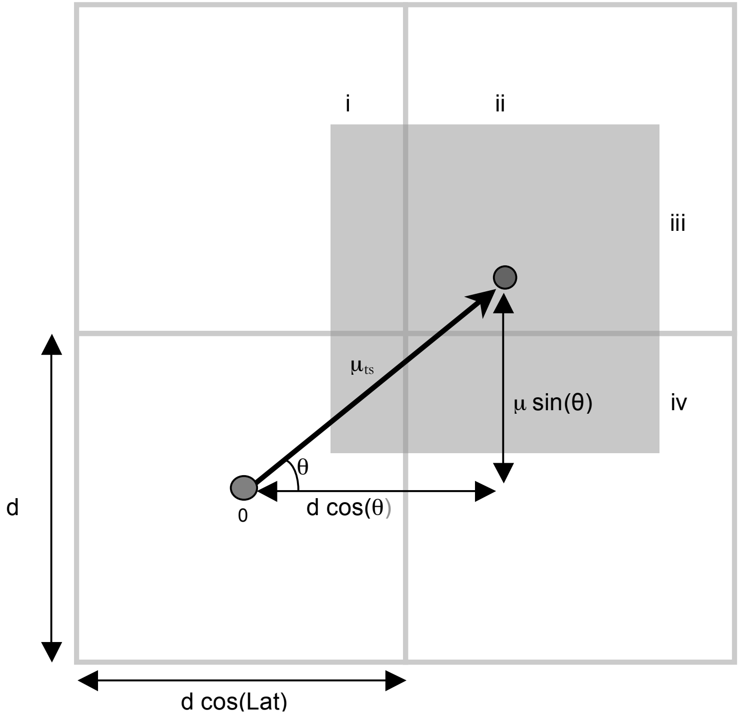

Figure A1. Visualization of method used for calculating the distribution of dioxin using dispersion. The original cell is denoted by 0 (bottom left). The gray square is the end state of the dioxin from cell 0 after a single dispersion iteration ts. i = d cos(θ) - μts cos(θ), ii = μts cos(θ), iii = μ ts sin(θ), iv = d – μ ts sin(θ), where μ ts is the wind magnitude μ·ts.

In Figure A1, the cell boundary grid lines are shown in light gray with the current cell (originating cell) on the bottom left. The final cell content after applying the given wind-direction and –speed for duration *ts* is shown as the light gray region. The area of each of the four sections of the gray square could be calculated as a fraction of the entire amount in the original cell, and allocated to the air of the receiving cells for the next iteration.

For example, the top left region could be calculated as i • iii and the top right could be ii • iii. Below are formulae to compute fractions of each region.

Top left region:

$$\frac{i \times iii}{entireArea} = \frac{\left(d \times \cos(\theta) - \mu_{ts}\cos(\theta)\right) \times \left(\mu_{ts}\sin(\theta)\right)}{d \times d \times \cos(Lat)}$$

Top right region:

$$\frac{ii \times iii}{entireArea} = \frac{\left(\mu_{ts}\cos(\theta)\right) \times \left(\mu_{ts}\sin(\theta)\right)}{d \times d \times \cos(Lat)}$$

Bottom left region:

$$\frac{i \times iv}{entireArea} = \frac{\left(d \times \cos(\theta) - \mu_{ts}\cos(\theta)\right) \times \left(d - \mu_{ts}\sin(\theta)\right)}{d \times d \times \cos(Lat)}$$

Bottom right region:

$$\frac{ii \times iv}{entireArea} = \frac{\left(\mu_{ts}\cos(\theta)\right) \times \left(d - \mu_{ts}\sin(\theta)\right)}{d \times d \times \cos(Lat)}$$

These fractions are then added to the airborne dioxin of the respective cell for the next iteration.

Output

We kept track of the amount of dioxin remaining in the air of each cell after diffusion and dispersion, the amount of dioxin deposited either to land or ocean in each cell at each time step, the total sum of dioxin deposited to land or ocean in each cell, and the amount of dioxin arriving at the cell by dispersion or diffusion. The model was run for 200,000 time steps (approximately 138.9 days real-time), until the total global amount of dioxin in the air had diminished to less than 99% of starting amount (Figure 4 main text).

CHAPTER 6

ECOSYSTEM MODELING OF DIOXIN DISTRIBUTION PATTERNS IN THE MARINE ENVIRONMENT[8]

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ABSTRACT

We developed a spatial ecosystem model of the world's marine environment, and used it to simulate the uptake of dioxin through the food web. Dioxin is assumed to be taken up by phytoplankton only when it enters the ocean (i.e., 'the environment'), while uptake for other organisms is assumed to take place through food intake. The model was developed using the Ecopath with Ecosim modeling approach, specifically the Ecotracer and Ecospace models, while adding a capability to use spatial reference data to the modeling approach. We use global dioxin loading patterns aggregated to a spatial scale of 2° latitude by 2° longitude to drive the accumulation of dioxin in the Ecotracer model. In order to tune the model, we made an extensive literature search for dioxin-level measurements of marine organisms. The spatial model was able to predict loading factors and there was a strong correlation between predicted and observed estimates of dioxin loading based on approximately 200 samples, distributed through the food web (from phytoplankton to whales) and from pole to pole. The model is rather preliminary, however, and we recommend: (1) that studies of dioxin (and other persistent pollutants) be carried out also from low-impact areas; (2) that modeling studies of a highly-impacted and of a low- to medium-impacted area be carried out to further tune the dioxin-food web model; (3) that the loading model underlying this work be further developed to include monthly wind patterns as well as coastal runoff impact; and (4) that the initial global model we developed be further developed based on the above suggested studies. The global model thus revised will be able to predict spatial dioxin-loading factors, e.g., in forage fishes from throughout the world.

INTRODUCTION

Concern over the freshness and quality of seafood has increased over the last decade, and most recently concern over the levels of pollutants including mercury, polychlorinated biphenyls (PCBs) and dioxin. The recent study of pollutants in salmon by Hites *et al.* (2004) and the continuing controversy about mercury in tuna highlights the need to better understand how pollutants move through marine food webs. This contribution builds on the dioxin deposition maps of Zeller *et al.* (Chapter 5) to present a model of uptake and transfer of dioxin up through the food web.

The model we present here is preliminary, but it shows great promise in placing scattered observations of dioxin loading in marine organisms throughout the world in an ecosystem context, i.e., considering explicitly their food intake and their loss due to predation. Studies of dioxin in marine organisms and their transfer between various components of the ecosystem are lacking, which in turns makes it difficult to demonstrate to policy-makers the importance of understanding the role of forage fish in relation to other organisms, especially those that are important to humans for food, and to also illustrate the dynamics of how some pollutants are transferred from one organism to another.

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[8] Cite as: Christensen, V., Booth, S.. 2006. Ecosystem modeling of dioxin distribution patterns in the marine environment, p. 83-102. *In*: Alder, J. Pauly, D (eds.) On the multiple uses of forage fish: from ecosystems to markets. Fisheries Centre Research Reports 14(3). Fisheries Centre, University of British Columbia [ISSN 1198-6727].

The model covers the entire world, and hence includes areas of high dioxin loading (and high numbers of dioxin loading samples in marine organisms) and areas of low dioxin loading (with fewer samples). The model is shown to be capable of explaining much of the variance in those samples, but the fit between observed and predicted loadings could be improved through a number of straightforward steps detailed in the discussion.

METHODOLOGY

The Ecopath model

The Ecopath approach and model are described in detail by Christensen and Walters (2004), while a User's Guide is available as Christensen *et al.* (2005). In excess of a hundred ecosystem models based on the approach have been published (see www.ecopath.org) while there are only few published applications of the Ecotracer module of EwE (Watkinson, 2001; Coombs, 2004; Booth and Zeller, 2005).

We developed a global ecosystem model with a spatial resolution of 2 degree latitude by 2 degree longitude. The model includes 42 functional groups (Table 1). The model is a 'generic' model developed by us for database-driven model construction, and is a slightly modified version of the one we distribute with the EwE software ('Generic37').

Table 1. Overview of functional groups as represented in the spatial ecosystem model. 'Small' includes species with asymptotic length, $L_\infty < 30$ cm; 'medium' have L_∞ in the range 30-89 cm; and 'large, $L_\infty > 90$ cm.

No.	Group name	No.	Group name
1	Pelagics, small	22	Rays, large
2	Pelagics, medium	23	Flatfish, small medium
3	Pelagics, large	24	Flatfish, large
4	Demersals, small	25	Cephalopods
5	Demersals, medium	26	Shrimps
6	Demersals, large	27	Lobsters, crabs
7	Bathypelagics, small	28	Jellyfish
8	Bathypelagics, medium	29	Molluscs
9	Bathypelagics, large	30	Krill
10	Bathydemersals, small	31	Baleen whales
11	Bathydemersals, medium	32	Toothed whales
12	Bathydemersals, large	33	Seals
13	Benthopelagics, small	34	Birds
14	Benthopelagics, medium	35	Macrobenthos
15	Benthopelagics, large	36	Meiobenthos
16	Reef fish, small	37	Corals
17	Reef fish, medium	38	Soft corals, sponges, etc.
18	Reef fish, large	39	Zooplankton, other
19	Sharks, small medium	40	Phytoplankton
20	Sharks, large	41	Benthic plants
21	Rays, small medium	42	Detritus

Ecopath model parameters

The dioxin modeling is rather insensitive to initial model parameters as it seeks to estimate long-term equilibrium concentrations. For most groups the initial biomasses were estimated by the Ecopath model based on predation, catches and assumed ecotrophic efficiencies. For large sharks the initial biomass was set to 0.1 $t \cdot km^{-2} \cdot year^{-1}$ as it was for jellyfish. For baleens and birds we used a biomass of 0.001 $t \cdot km^{-2} \cdot year^{-1}$, while twice this value was used for toothed whales, and three times for seals/pinnipeds. Macro- and meiobenthos biomasses were set to 1.5 and 2 $t \cdot km^{-2} \cdot year^{-1}$, respectively, corals to 1 $t \cdot km^{-2} \cdot year^{-1}$, soft-corals/sponges to 2 $t \cdot km^{-2} \cdot year^{-1}$ and benthic plants to 10 $t \cdot km^{-2} \cdot year^{-1}$. Other basic input were kept at the default values they have in the generic model.

Catches were extracted from the *Sea Around Us* Project database (www.seaaroundus.org), averaged globally, and summed up to a global total for each functional group. The values used were representative for year 2000.

Ecosim model parameters

We used default values throughout for the Ecosim model parameters – which also are used for the Ecospace and Ecotracer simulations. The use of default values mainly has implications for how Ecosim will react to perturbations. Given that we here are aiming for long-term equilibrium solutions, this should not have implications for the simulations conducted as part of the present study.

Ecospace model parameters

We extracted spatial parameters from the *Sea Around Us* Project databases, which includes a large number of parameters all at a half-degree latitude by half-degree longitude scale:

- Primary production estimates averages for 1998 were obtained from the GoMor database (Marine Environment Unit, Joint Research Centre, European Commission, Hoepffner *et al.* (2001) as modified by Lai (2004) and as described in the *Sea Around Us* Project website Methods Manual (2006));
- Biomass abundance for zooplankton, macrobenthos and meiobenthos was from the database described by Vasconcellos (2003);
- A combined spatial biomass of small and large mesopelagic fishes was obtained from the database described by Vasconcellos (2003), based on the analysis of Gjøsaeter and Kawaguchi (1980);
- Depth information was obtained from the ETOPO5 dataset available on the U.S. National Geophysical Data Center's Global Relief Data CD (www.ngdc.noaa.gov/products/ngdc_products.html);
- Catches were obtained from the rule-based spatial allocation distribution described by Watson and Pauly (2001).

We averaged (area-weighted) the half-degree estimates to the two degree latitude by two degree longitude spatial scale, giving us a total of 70 by 180 spatial cells, as we excluded areas above 70° latitude (north or south) from the analysis.

We designed the spatial map to include two depth strata (<200 m, >200 m) within each of the 19 FAO statistical areas, and allocated functional groups to strata allowing nearly all groups to occur at all depths. Exceptions were the following groups, assigned to the shallower depth strata: small demersals, reef fishes, seals, corals and benthic plants. This means that available primary production is the main determinant of species distributions.

DIOXIN MODELING

We are using the time-dynamic Ecosim and the spatial-dynamic Ecospace modules of the Ecopath with Ecosim software system (www.ecopath.org) to predict flow patterns of biomass among species or functional groups in ecosystems using nonlinear functional relationships between flow rates and abundances of the interacting groups/species. These flow rates (along with information about dioxin loading) can be used to predict changes in concentrations (and per-biomass burdens) of dioxin (or other organic contaminants as well as for isotope tracers) that 'flow' passively along with the biomass flows (Christensen *et al.*, 2005). The dynamic equations for such passive flow (and accumulation) are generally linear dynamical equations with time-varying rate coefficients that depend on the biomass flow rates; these linear equations are solved in parallel with the Ecosim biomass-dynamics equations. For methodological details see, Appendix 1.

The dioxin modeling is conducted using the Ecotracer module of EwE. This module has only been used for a few previous studies (Watkinson, 2001; Coombs, 2004; Booth and Zeller, 2005). Here, we developed the Ecotracer module further to handle spatial tracer data to evaluate goodness-of-fit of the simulations. In previous applications the fitting procedure has not included a goodness-of-fit estimation, let alone one considering spatial differences.

Dioxin loading

We used the global dioxin loading model derived by Zeller *et al.* (2006) to provide spatial loading factors. The loading factors were available at the half degree by half degree scale, and we averaged (area-weighted) to derive loading estimates for each 2 by 2 degree cell in the global model (Figure 1). The loading was assumed to be continuous throughout the model period, with rapid breakdown of the dioxin not taken up by phytoplankton. This was done to avoid a massive build-up of dioxin over time and to approach an equilibrium loading situation.

Figure 1. Dioxin loading as predicted from the model of Zeller *et al.* (2006). The loading is expressed with a linear scale ranging from low (white or light) to high (red or dark).

Dioxin estimates

Dioxin concentrations for marine organisms were primarily taken from literature sources, and were restricted to measurements taken since 1990. Sources reporting the 17 dioxin congeners of concern individually were standardized to toxic equivalencies (TEQs) by applying the appropriate toxic equivalency factor (TEF) to each of the 17 congeners. These were then summed to get the TEQ and then standardized to ng·kg^{-1} (lipid weight) for marine mammals or ng·kg^{-1} (wet weight) for all other organisms.

For the model runs we assigned the samples in Appendix 2, Table A1 to spatial cells as shown in Figure 2. The values were read into the Ecospace module as time series, with the loading assumed to represent year 2000 in all cases. For each simulation we ran the spatial model for the period 1980-2001, and extracted spatial values for year 2000 for each of the cells. Subsequently, we compared observed and predicted spatial dioxin loading factors.

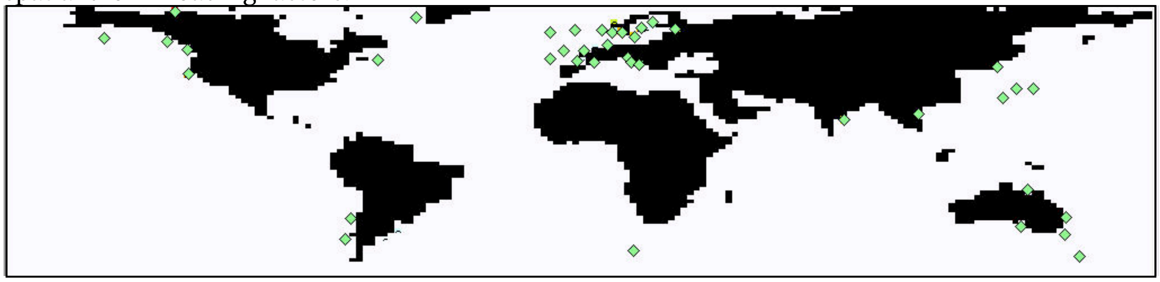

Figure 2. Locations ('regions', marked with diamonds) used for comparing predicted and observed dioxin concentrations.

Parameter settings for tracer model

We assume that dioxin enters the ocean from the air, and that it, once deposited, will be taken up by phytoplankton very rapidly. We set the initial concentration of dioxin in the 'environment' (the water) to 0.1 t · km^{-2}, and the decay rate for the environment to 1 year^{-1} to minimize long-term build-up of dioxin in the environment. We assume that the base inflow rate of dioxin is 1 t · km^{-2} · year^{-1}, and that the direct absorption rate of dioxin by phytoplankton is at 0.5 t dioxin · t^{-1} in the environment per t of phytoplankton · year^{-1}.

For fish, the main route of exposure is through trophic transfers, whereas for lower trophic organisms (e.g., phytoplankton, algae and invertebrates) direct uptake from the water column is the main route of exposure (Anon., 1998; Commoner *et al.*, 2000). Thus, all other parameters are set to default values (0), so that it is assumed that dioxin passes through the food web from phytoplankton only, and that there is no decay of dioxin.

RESULTS AND DISCUSSION

We ran the Ecospace model (and hence the dioxin uptake simulations) for 22 years. This was not long enough for very long-lived groups to reach equilibrium with regard to dioxin loading. However, such groups would probably not have reached equilibrium in real life either. We thus consider the run length an acceptable compromise.

An overview of predicted biomass distributions for all functional groups in the spatial model is presented in Figure 3.

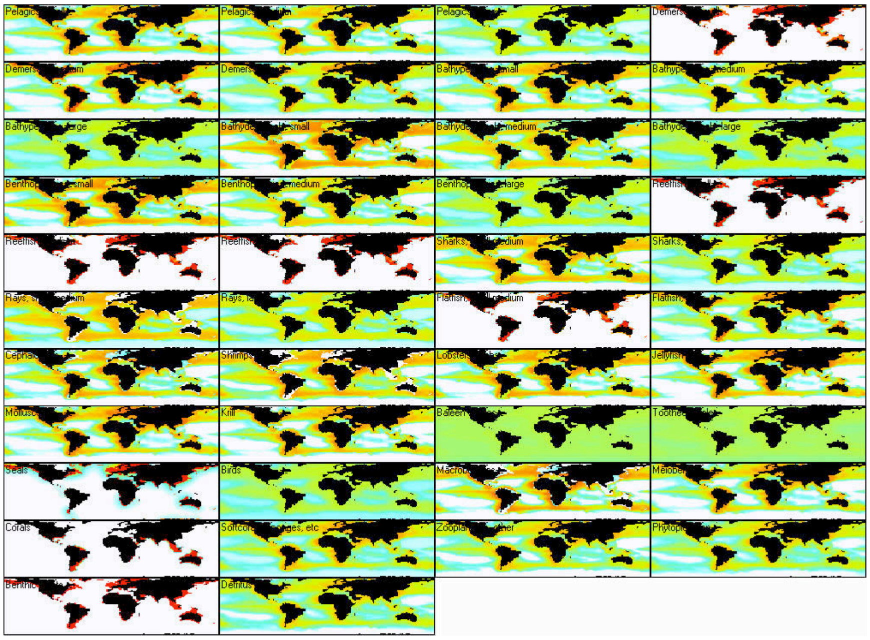

Figure 3. Predicted biomass distributions for all 42 groups in the global ecosystem model. Scales from low (white or light) to high (red or dark). Groups are arranged with 1 in the upper left corner, 2 to its right, etc.

The model produces what seem to be reasonable biomass distributions (Figure 4) even if it ignores advection and dispersion. We conclude though, that the actual biomass distributions are of minor importance for the present study, as our focus here is on how dioxin is added to the marine environment, and on how it is subsequently taken up by living groups through the food web.

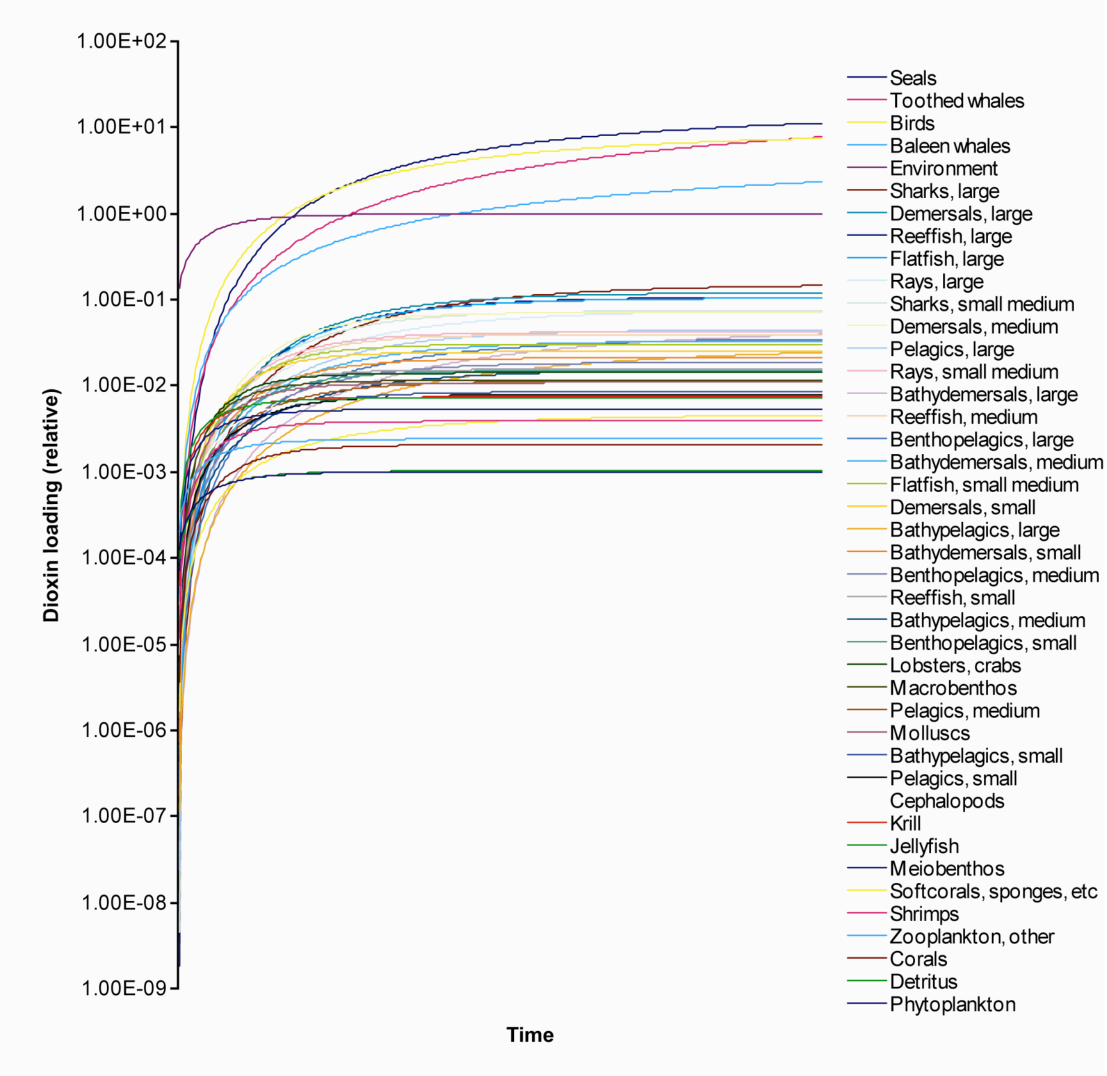

Figure 4. Dioxin loading (logarithmic scale) for the ecosystem groupings in the global ecosystem model as predicted based on the Ecotracer module of EwE. The functional groups are arranged in legend with the group with highest loading at the end of the simulation at the top (seals), and the group with lowest loading at the bottom (phytoplankton). Notice that the loading for the top four groups (seals, toothed whales, birds and baleen whales) was still increasing at the end of the simulation period of 22 years.

Fitting the dioxin model to observations

We limited the fitting to an absolute minimum, given that the data material had very few constraints to limit the fitting procedure. As an important measure of fit we compared spatially-explicit estimates of dioxin loading by functional groups as observed (Table A1 and A2) and as predicted based on the Ecotracer model. The results are presented as a plot of predicted versus observed values in Figure 5, while Figure 6 presents the ratio between predicted and observed values by sample number, and Figure 7 shows the plot of residuals versus fitted observations using R (Anon., 2006).

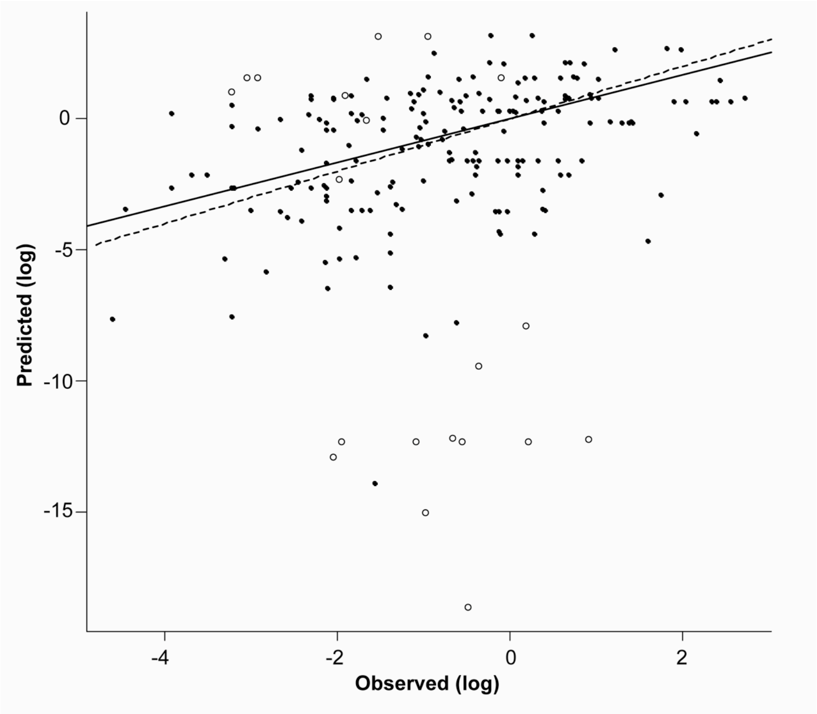

Figure 5. Predicted vs. observed dioxin concentrations. The dotted line indicates the expected 1:1 ratio, while the solid line is the regression line (r2=0.19, slope = 0.78, P<10-9 ***). The open symbols are from 4 sample areas considered to be outliers (see Figure 6) and thus not included in the regression.

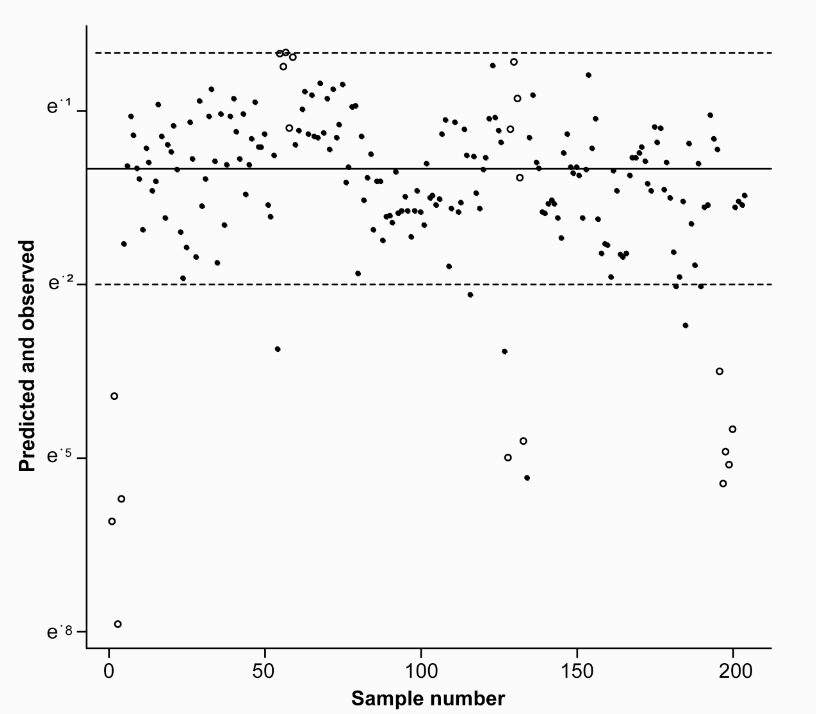

Figure 6. Ratio between predicted and observed spatial dioxin loading factors for a variety of functional groups or species. The vast majority (93%) of the predicted values fall within two orders of magnitude of the observed values, and 69% within one order of magnitude. Solid line indicates predicted = observed values, while the two stippled lines indicate ±2 orders of magnitude deviations. Outliers are shown with open symbols.

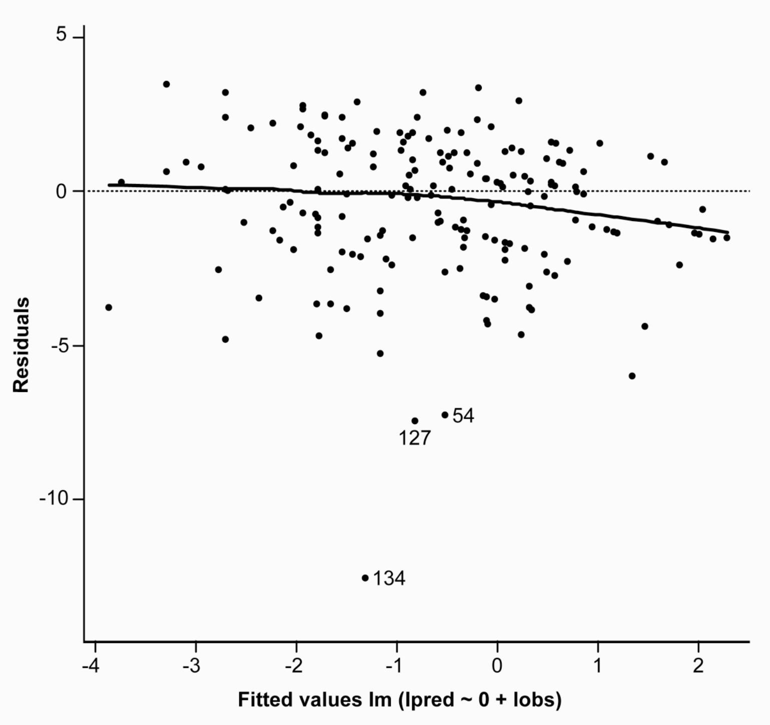

Figure 7. Residuals vs. fitted values for the spatial dioxin loading factors. Residual plot indicates no clear systematic bias over the range of fitted values, i.e., they are well-behaved. Outliers are marked with sample numbers.

The regression between predicted versus observed dioxin values in Figure 5 is very clear, with a correlation coefficient (r^2) of 0.25, indicating that the model is able to explain 25% of the variation in the dioxin samples (see Table 2 for a regression summary). The slope of the regression (0.84) is less than expected (unity) but not significantly so, given the large variation in the dataset. The slope is, however, significantly different from zero. Thus it is unlikely that the correlation between predicted and observed values is occurring by chance alone. We also note that the regression is well-behaved, in the sense that the plot of residuals versus fitted values indicates no trend (Figure 7).

Table 2. Regression summary for observed vs. predicted dioxin concentrations

Call	lm(formula = log(pred) ~ 0 + log(obs), weights = wt)				
Residuals	Min -12.5782	1Q -1.5285	Median 0.0000	3Q 0.9298	Max 3.4664
Coefficients lobs	Estimate 0.8393	Std. Error 0.1054	t-value 7.964	Pr (>|t|) 1.71e-13 ***	
Signif. Codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1 **Residual standard error**: 2.336 on 183 degrees of freedom **Multiple R-squared:** 0.2574, Adjusted R-squared: 0.2533 **F-statistic:** 63.42 on 1 and 183 DF, p-value 1.71e-13					

Some of the predicted values were clearly outliers and not included in the regression analysis (see Figures 5-7). We especially note four sample areas, two from polar regions and two from coastal regions in Asia (Table 3). For the polar regions, this may indicate a problem with the dioxin loading factors. It is well known that persistent organic pollutants (POP) tend to evaporate in warmer climates and to transport with wind and dust to eventually settle in polar regions through what has been called the 'grasshopper effect' (Wania *et al.*, 1999), which is not captured in the dioxin loading model we are using for this study (Zeller *et al.*, 2006).

Table 3. Summary of areas with dioxin samples that were excluded from the regression analysis (see text).

No	Area
1	Antarctic, Ross Sea
21	Japan, coastal
28	Sri Lanka, Kelani River mouth
44	Arctic, Holman, NWT

Further, we find it likely that the two coastal areas highlighted as outliers in Figures 5-7 may be from areas that are impacted by coastal runoff (Table 3). Again, such processes are not captured in the dioxin loading model we use. We expect similar problems will occur for other sampling areas, and indeed noted problems in allocating the samples to the 2 by 2 degree map (see below). Overall, we conclude that the predictive model is doing a very good job as indicated by the significant slope of the predicted versus observed regression being close to unity. We are certain that further work with this model could lead to improvements in its capability to predict dioxin loading factors spatially and from throughout the food web. We emphasize that the development should include a refinement of the model for dioxin loading factors (Zeller *et al.*, 2006).

Dioxin loading

The comparison between observed and predicted dioxin levels (Figure 5) indicates that our global Ecotracer model as currently implemented can predict dioxin levels within 2 orders of magnitude of loading. This precision can undoubtedly be improved through better tuning of model parameters. Thus, we find that we can use the model to predict spatial patterns of dioxin loading in the marine environment.

We give an example of the model prediction in Figure 8 where the biomass of small pelagics is predicted, and in Figure 9 where the relative dioxin level, also in small pelagics, is presented at the global level.

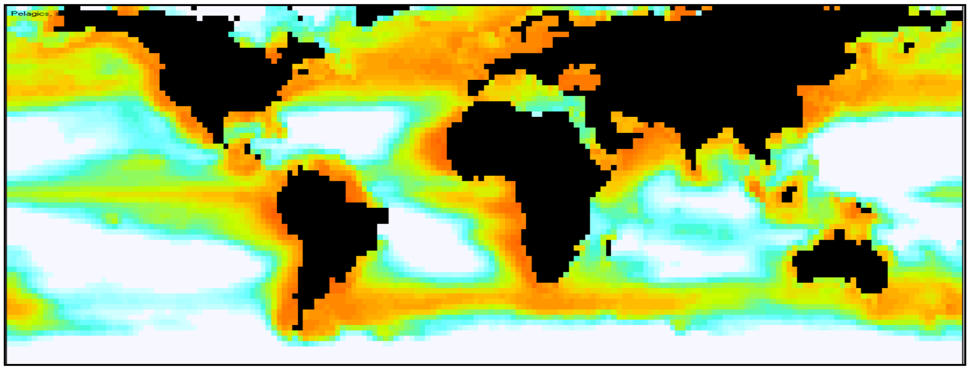

Figure 8. Predicted relative biomass distribution for small pelagics. Scale is from low to high, (white or light) to (red or dark).

It is clear that the predicted dioxin levels bear a very strong resemblance to the environmental loading factors – as they should. We have extracted the loading factors for small pelagics by FAO areas and by depth strata (<200 m, > 200 m) and present the results as relative loading factors in Table 4.

Table 4. Predicted dioxin loading for small pelagics by FAO area and depth strata. The loading factors are expressed as relative values.

FAO area		<200m	>200m
18	Arctic	$5 \cdot 10^5$	$3 \cdot 10^5$
21	Atlantic, NW	$4 \cdot 10^6$	$5 \cdot 10^5$
27	Atlantic, NE	$1 \cdot 10^7$	$8 \cdot 10^6$
31	Atlantic, WC	$8 \cdot 10^6$	$4 \cdot 10^5$
34	Atlantic, EC	$4 \cdot 10^5$	$2 \cdot 10^5$
37	Mediterranean	$5 \cdot 10^6$	$1 \cdot 10^7$
41	Atlantic SW	$1 \cdot 10^6$	$2 \cdot 10^5$
47	Atlantic SE	$1 \cdot 10^6$	$1 \cdot 10^4$
48	Atlantic Antarctic	$1 \cdot 10^0$	$1 \cdot 10^3$
51	Indian Ocean, W	$3 \cdot 10^6$	$7 \cdot 10^5$
57	Indian Ocean, E	$8 \cdot 10^5$	$3 \cdot 10^5$
58	Indian Ocean, Antarctic	$1 \cdot 10^0$	$2 \cdot 10^3$
61	Pacific, NW	$6 \cdot 10^6$	$5 \cdot 10^6$
67	Pacific, NE	$4 \cdot 10^5$	$8 \cdot 10^4$
71	Pacific, WC	$2 \cdot 10^6$	$2 \cdot 10^5$
77	Pacific, EC	$3 \cdot 10^7$	$5 \cdot 10^5$
81	Pacific SW	$1 \cdot 10^6$	$2 \cdot 10^5$
87	Pacific SE	$3 \cdot 10^6$	$5 \cdot 10^4$
88	Pacific Antarctic	--	$4 \cdot 10^1$

The loading factors vary more than 7 orders of magnitude between strata and FAO areas, which is more than they are likely to actually do. This is a consequence of the loading as predicted by the loading model not being dispersed widely enough, and, notably, not reaching the polar regions to any significant degree. We therefore consider the results expressed in Figure 8 and 9 as illustrating only the results that *may* be obtained from modeling activities of the sort carried out here.

This form for modeling allows for predictions of what level of dioxin loading one can expect for a given type of marine organism in a given location as well for evaluating if the measured level for a given organism indicates that a given site is more or less impacted that should be expected. We consider that such predictive food web modeling has potential as part of monitoring programs for persistent organic pollutants.

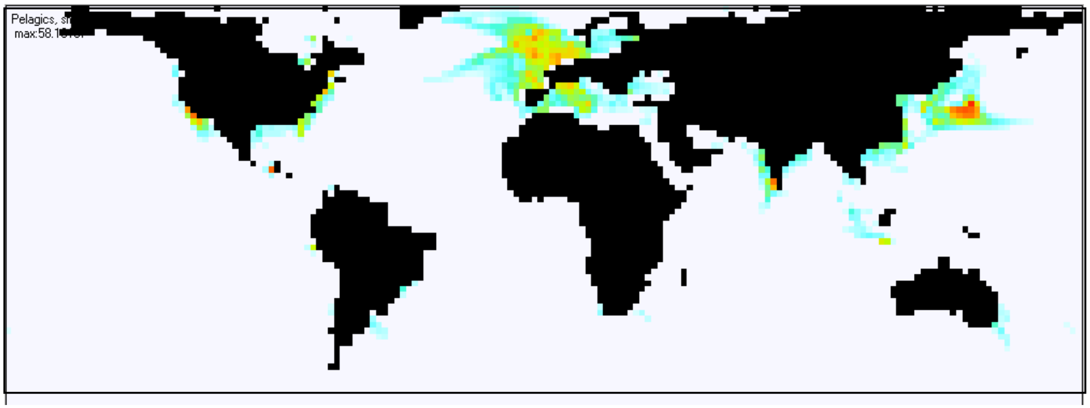

Figure 9. Predicted relative dioxin concentrations for small pelagics. Scale is from low to high, (white or light) to (red or dark).

RECOMMENDATIONS

To improve the modeling presented in this chapter we recommend that the following studies be carried out:

1. Detailed studies of dioxin loading in a highly-impacted and a low- to- moderate impacted ecosystem along the same lines as the study of Booth and Zeller (2005). We do not have group-specific estimates for dioxin decay rates, nor specific information about possible uptake routes for dioxin from the environment. Studies of this sort would help tune the global model;
2. The samples we have obtained are largely from coastal areas, many of which are likely to be from areas which are more impacted than the loading model we use predicted. Samples from more geographic areas, notably from less impacted oceanic waters would be useful for tuning the model;
3. The loading model of Zeller *et al.* (2006) should be improved to include monthly wind patterns and coastal runoff, and if possible to consider the 'grasshopper effect'; and
4. The present model should be updated based on the above studies.

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APPENDIX 1[9]

The Ecotracer module of EwE

Ecosim allows parallel simulation of one tracer or contaminant type while the biomass dynamics equations are being solved and we here use this approach to simulate the spatial distribution of dioxin based on the loading factors developed from the model of Zeller *et al.* (2006). Dioxin molecules are assumed to be either in the 'environment' or in the biota (in biomass and detritus functional groups) at any moment. Molecules are assumed to flow between groups at instantaneous rates (i.e., to be sampled along with biomass during biomass flows at rates) equal to the probabilities of being 'sampled' as part of the biomass flow: instantaneous rate = (flow) / (biomass in prey pool). We allow for direct flows from the environment into functional groups, representing direct uptake or absorption of the tracer material, and for differential decomposition/decay/export rates by pool and from the environmental pool. Schematically, the flow of tracer molecules through any biomass group is represented by the components shown below. In the rate equation for time changes in contaminant concentration in group *i*, these components are represented as follows:

1) Uptake from food: $C_j \cdot GC_i \cdot Q_{ji} / B_j$ where C_j = conc in food *j*, GC_i = proportion of food assimilated by type *i* organisms; Q_{ji} = biomass flow rate from *j* to *i* (estimated in Ecopath as $B_i \cdot (Q/B)_I \cdot DC_{ij}$) *i*, B_j=food *j* biomass);
2) Direct uptake from environment: $u_i \cdot B_i \cdot C_o$, where u_i=parameter representing uptake per biomass per time, per unit environmental concentration, B_i=biomass, C_o=environmental concentration;
3) Concentration in immigrating organisms: $c_i \cdot I_i$, where c_i = parameter (tracer per unit biomass in immigrating biomass), I_i = biomass of group *i* immigrants per time;
4) Predation: $C_i \cdot Q_{ij} / B_i$, where C_i=concentration in group *i*, Q_{ij} = consumption rate of type *i* organisms by predator type *j*, B_i = biomass in group *i*;
5) Detritus: $(C_i \cdot MO_i + (1\text{-}GC_i) \cdot S_j \cdot C_j \cdot Q_{ji}\ ) B_j$, where MO_i = non-predation death rate of type *i* (year^{1}), GC_i = fraction of food intake assimilated, Q_{ji} = intake rate of type *j* biomass by type *i*;
6) Emigration: $e_i \cdot C_i$, where e_i = emigration rate (per year);
7) Metabolism: $d_i \cdot C_i$, where d_i = metabolism + decay rate for the material while in group *i*.

Hence, users must specify some parameters (using the Ecotracer input form) besides those needed for Ecopath/Ecosim mass balance and biomass dynamics calculations:

a. initial group concentrations C_i, including environmental concentration C_o;
b. direct uptake rate parameters u_i as rates per time per biomass per unit C_o;
c. concentrations per biomass c_i in immigrating organisms;
d. metabolism/decay rates d_i.

The other parameters (GC_i, MO_i) are already entered or calculated in Ecopath, and the Q_{ij} flows are calculated dynamically in Ecosim. Ecosim integrates the rate components listed above to generate time concentration patterns C_i (i = 0 to number of biomass group), and time patterns of concentration per biomass C_i/B_i for all Ecopath/Ecosim biomass group. These results can be compared to data on bioaccumulation or tracer movement. Discrepancies between model and data may help identify weaknesses in the Ecopath/Ecosim trophic flow rate (Q_{ij}) estimates, and/or in the Ecotracer rate parameters (u_i, e_i, d_i).

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[9] This method description is based on Christensen *et al.* (2005)

APPENDIX 2

Dioxin samples

Table A 1. Concentrations of dioxins in marine organisms reported in wet weight.

Year	Common Name	Species Name	Area	Mean Concentration (ng I-TEQ/kg)	% Lipid	Tissue Type	Source
1995	Cod	*Gadus morhua*	UK	0.040	-	muscle	1
1993	Cod	*Gadus morhua*	Norway	0.080	-	muscle	1
1993	Plaice	*Pleuronectes platessa*	Norway	0.180	-	muscle	1
1995	Herring	*Clupea harengus*	UK	2.100	-	muscle	1
1993	Herring	*Clupea harengus*	Norway	1.100	-	muscle	1
1995	Mackerel	*Scomber scombrus*	UK	0.660	-	muscle	1
1993	Mackerel	*Scomber scombrus*	Norway	0.610	-	muscle	1
1991	Yellow eel	*Anguilla anguilla*	Netherlands	1.800	-	whole	2
1991	Herring	*Clupea harengus*	Netherlands	1.900	-	whole	2
1991	Mackerel	*Scomber scombrus*	Netherlands	0.800	-	whole	2
1991	Sole	*Solea solea*	Netherlands	0.800	-	whole	2
1991	Cod	*Gadus morhua*	Netherlands	0.100	-	whole	2
1991	Shrimp		Netherlands	1.200	-	whole	2
1991	Mussels	*Mytilus*	Netherlands	1.300	-	whole	2
1991	Pike perch	*Sander lucioperca*	Netherlands	0.700	-	whole	2
1994	Shrimp		Netherlands	1.340	-	whole	2
1994	Mussel	*Mytilus*	Netherlands	0.810	-	whole	2
1994	Herring	*Clupea harengus*	Netherlands	2.010	-	whole	2
1994	Cod	*Gadus morhua*	Netherlands	0.130	-	whole	2
1994	Eel	*Anguilla anguilla*	Netherlands	2.190	-	whole	2
1994	Sole	*Solea solea*	Netherlands	0.190	-	muscle	2
1996	Cod	*Gadus morhua*	Sweden	0.360	0.64	muscle	2
1996	Cod	*Gadus morhua*	Sweden	0.460	-	muscle	2
1996	Herring	*Clupea*	Sweden	10.400	7.70	muscle	2
1996	Herring	*Clupea*	Sweden	11.000	-	muscle	2
1988	Herring	*Clupea*	Sweden	7.700	-	muscle	2
1987	Herring	*Clupea*	Sweden	6.700	-	muscle	2
1989	Herring	*Clupea*	Sweden	7.700	-	muscle	2
1996	Herring	*Clupea*	Sweden	13.000	-	muscle	2
1996	Herring	*Clupea*	Sweden	2.000	18.20	muscle	2
1988	Herring	*Clupea*	Sweden	2.600	-	muscle	2
1989	Herring	*Clupea*	Sweden	2.000	-	muscle	2
1996	Mackerel	*Scomber scombrus*	Sweden	2.800	15.90	muscle	2
1996	Norway lobster	*Nephrops norvegicus*	Sweden	0.270	0.31	muscle	2
1996	Plaice	*Pleuronectes platessa*	Sweden	0.350	0.75	muscle	2
1996	Sea trout	*Salmo trutta trutta*	Sweden	8.800	6.70	muscle	2
1996	Whitefish	*Coregonus lavaretus lavaretus*	Sweden	7.300	4.40	muscle	2
1996	Whitefish	*Coregonus lavaretus lavaretus*	Sweden	3.400	-	muscle	2
2000	Phytoplankton		Japan	0.380	-	whole	3
2000	Seaweed	*Porphyra yezoensis*	Japan	0.103	-	whole	3
2000	Seaweed	*Eisenia bicyclis*	Japan	0.103	-	whole	3
2000	Zooplankton		Japan	0.155	-	whole	3

2000	Mysid	*Acanthomysis mituskurii*	Japan	0.514	-	whole	3
2000	Shrimp	*Trachypenaeus*	Japan	0.688	-	whole	3
2000	Shrimp	*Crangon*	Japan	1.099	-	whole	3
2000	Sandworms	*(Cirratulidae,Nereidae Eunicidae)*	Japan	1.211	-	whole	3
2000	Sandlance	*Ammodytes personatus*	Japan	0.541	-	whole	3
1994	White croaker	*Genyonemus lineatus*	USA	1.300	-	muscle	4
1994	White croaker	*Genyonemus lineatus*	USA	1.460	-	muscle	4
1994	Striped bass	*Morone saxatilis*	USA	0.670	-	muscle	4
1994	Shiner surf perch	*Cymatogaster aggregata*	USA	0.890	-	muscle	4
1994	Shiner surf perch	*Cymatogaster aggregata*	USA	0.970	-	muscle	4
1994	Shiner surf perch	*Cymatogaster aggregata*	USA	0.850	-	muscle	4
1994	White croaker	*Genyonemus lineatus*	USA	1.750	-	muscle	4
1994	White croaker	*Genyonemus lineatus*	USA	0.890	-	muscle	4
1994	White croaker	*Genyonemus lineatus*	USA	0.880	-	muscle	4
1994	White croaker	*Genyonemus lineatus*	USA	0.730	-	muscle	4
1994	White croaker	*Genyonemus lineatus*	USA	0.570	-	muscle	4
1994	White croaker	*Genyonemus lineatus*	USA	1.000	-	muscle	4
1994	Striped bass	*Morone saxatilis*	USA	0.500	-	muscle	4
1994	Sturgeon	*Acipenser transmontanus*	USA	0.510	-	muscle	4
1994	Leopard sharks	*Triakis semifasciata*	USA	0.120	-	muscle	4
1994	Leopard sharks	*Triakis semifasciata*	USA	0.230	-	muscle	4
1994	Leopard sharks	*Triakis semifasciata*	USA	0.130	-	muscle	4
1994	Halibut	*Paralichthys californicus*	USA	0.120	-	muscle	4
1994	White croaker	*Genyonemus lineatus*	USA	1.040	-	muscle	4
1994	Sprat	*Sprattus sprattus*	Finland	15.400	-	whole	5
1997	Anchovy	*Engraulis encrasicolis*	Italy	0.470	3.30	muscle	6
1997	Anchovy	*Engraulis encrasicolis*	Italy	0.340	3.20	muscle	6
1997	Anchovy	*Engraulis encrasicolis*	Italy	0.230	1.80	muscle	6
1997	Squid	*Loligo vulgaris*	Italy	0.250	1.90	mantle	6
1997	Squid	*Loligo vulgaris*	Italy	0.170	1.80	mantle	6
1997	Squid	*Loligo vulgaris*	Italy	0.120	1.20	mantle	6
1997	Mussel	*Mytilus galloprovincialis*	Italy	0.240	1.60	meat	6
1997	Mussel	*Mytilus galloprovincialis*	Italy	0.160	1.40	meat	6
1997	Mussel	*Mytilus galloprovincialis*	Italy	0.110	1.40	meat	6
1997	Norway lobster	*Nephrops norvegicus*	Italy	0.140	0.60	meat	6
1997	Norway lobster	*Nephrops norvegicus*	Italy	0.090	0.50	meat	6
1997	Norway lobster	*Nephrops norvegicus*	Italy	0.120	0.70	meat	6
1997	Mackerel	*Scomber scombrus*	Italy	0.590	8.70	muscle	6
1997	Mackerel	*Scomber scombrus*	Italy	0.940	7.30	muscle	6
1997	Mackerel	*Scomber scombrus*	Italy	1.070	5.60	muscle	6
1997	Red mullet	*Mullus barbatus barbatus*	Italy	0.560	4.90	muscle	6
1997	Red mullet	*Mullus barbatus barbatus*	Italy	0.370	4.40	muscle	6
1997	Red mullet	*Mullus barbatus barbatus*	Italy	0.370	4.30	muscle	6

1997	Clam	*Chamelea gallina*	Italy	0.130	1.00	meat	6
1997	Clam	*Chamelea gallina*	Italy	0.100	1.10	meat	6
1997	Clam	*Chamelea gallina*	Italy	0.070	0.90	meat	6
2000	Prawns		Belgium	0.650	-	-	7
2000	Mackerel	*Scomber scombrus*	Belgium	6.200	-	-	7
1996	Mullet	*Liza subviridis*	Sri Lanka	0.520	-	-	8
1996	Cat fish	*Mystus gulio*	Sri Lanka	0.190	-	-	8
1996	Jack (Juvenile)	*Carangidae*	Sri Lanka	0.040	-	-	8
1996	Green cromide	*Etroplus suratensis*	Sri Lanka	0.150	-	-	8
1996	Mud crab	*Scyllar serrataa*	Sri Lanka	0.140	-	-	8
1996	Rabbit fish	*Siganus vermiculatus*	Sri Lanka	0.130	-	-	8
1997	Snapper	*Lutjanus decussatus*	Sri Lanka	0.210	-	-	8
1997	Cat fish	*Mystus gulio*	Sri Lanka	0.009	-	-	8
1997	Green cromide	*Etroplus suratensis*	Sri Lanka	0.004	-	-	8
1993	Sea slag	*Aplysia (Varria) kurodai*	Japan	0.048	-	meat	9
1996	Sea urchin	*Hemicentrotus pulcherrimus*	Japan	0.390	-	meat	9
1996	Starfish	*Asterina pectinifera*	Japan	0.220	-	meat	9
1995	Mussel	*Septifer (Mytilisepta) virgatus*	Japan	0.910	-	meat	9
1995	Snail	*Thais claviger*	Japan	0.054	-	meat	9
1993	Mussel	*Septifer virgatus*	South Korea	0.020	-	meat	10
1993	Clam	*Anomalocardia*	South Korea	0.160	-	meat	10
1991	Mullet	*Liza macrolepis*	South Korea	0.320	-	muscle	10
1993	Gizzard	*Konosirus punctatus*	South Korea	0.530	-	muscle	10
1993	Flounder	*Cleisthenes pinetorum herzensteini*	South Korea	0.380	-	muscle	10
2005	Eel	*Anguilla anguilla*	Norway	1.100	-	muscle	11
2004	Mackerel	*Scomber scombrus*	Norway	0.170	-	muscle	11
2004	North Sea herring	*Clupea harengus*	Norway	0.660	-	muscle	11
2004	Norway pout	*Trisopterus esmarkii*	Norway	0.260	-	whole	11
2003	Norwegian Spring herring	*Clupea harengus*	Norway	0.870	-	muscle	11
2004	Redfish	*Sebastes marinus*	Norway	0.250	-	muscle	11
2004	Saithe	*Pollachius virens*	Norway	0.030	-	muscle	11
2004	Sand eel	*Ammodytes tobianus*	Norway	0.350	-	muscle	11
2004	Halibut	*Hippoglossus hippoglossus*	Norway	1.510	-	muscle	11
1995	Krill	*Euphausia superba*	Antarctic	0.379	1.50	whole	12
1995	Crocodile icefish	*Chionodraco hamatus*	Antarctic	0.702	2.65	whole	12
1995	Sharp-spined notothen	*Trematomus pennellii*	Antarctic	0.622	1.90	whole	12
1995	Antarctic silverfish	*Pleuragramma antarcticum*	Antarctic	2.501	9.40	muscle	12
2000	Shellfish	*(Mytilus + Crassostrea + Dressinia)*	France	0.800	-	-	13
2000	Sprat	*Sprattus sprattus balticus*	Russia	2.810	-	-	14
2000	Herring	*Clupea harengus membras*	Russia	11.460	-	-	14
2000	Salmon	*Salmo salar*	Russia	4.170	-	-	14
2000	flounder	*Platichthys flesus*	Russia	2.560	-	-	14
2000	Eel	*Anguilla anguilla*	Russia	1.400	-	-	14

2000	Cod	*Gadus morhua*	Russia	0.120	-	-	14
2002	Chum salmon	*Oncorhynchus keta*	US	0.037	4.81	muscle	15
2002	Chinook salmon	*Oncorhynchus tshawytscha*	US	0.139	8.60	muscle	15
2002	Sockeye	*Oncorhynchus nerka*	US	0.253	8.22	muscle	15
2002	Halibut	*Hippoglossus stenolepis*	US	0.001	0.46	muscle	15
2002	Sablefish	*Anoplopoma fimbria*	US	0.004	3.17	muscle	15
1990	Herring	*Clupea harengus*	Norway	2.300	-	muscle	16
1990	Herring	*Clupea harengus*	Norway	1.750	-	muscle	16
1993	Herring	*Clupea harengus*	Norway	1.160	-	muscle	16
1993	Herring	*Clupea harengus*	Norway	1.100	-	muscle	16
1994	Herring	*Clupea harengus*	Norway	1.400	-	muscle	16
1989	Mackerel	*Scomber scombrus*	Norway	0.700	-	muscle	16
1993	Mackerel	*Scomber scombrus*	Norway	0.500	-	muscle	16
1997	Mackerel	*Scomber scombrus*	Norway	0.560	-	muscle	16
1990	Cod	*Gadus morhua*	Norway	0.100	-	muscle	16
1995	Cod	*Gadus morhua*	Norway	0.040	-	muscle	16
1995	Cod	*Gadus morhua*	Norway	0.020	-	muscle	16
1996	Cod	*Gadus morhua*	Norway	0.120	-	muscle	16
1992	Plaice	*Pleuronectes platessa*	Norway	0.160	-	muscle	16
1994	Plaice	*Pleuronectes platessa*	Norway	0.200	-	muscle	16
1996	Flounder	*Platichthys flesus*	Norway	0.160	-	muscle	16
1991	Eel	*Anguilla anguilla*	Norway	2.000	-	muscle	16
1991	Eel	*Anguilla anguilla*	Norway	1.800	-	muscle	16
1996	Herring	*Clupea harengus*	North Sea	0.567	-	muscle	17
1996	Herring	*Clupea harengus*	Baltic Sea	1.905	-	muscle	17
1995	Herring	*Clupea harengus*	Ireland	0.449	-	muscle	17
1998	Herring	*Clupea harengus*	Norway	0.976	-	muscle	17
1996	Mackerel	*Scomber scombrus*	Bay of Biscay	0.317	-	muscle	17
1997	Mackerel	*Scomber scombrus*	North Sea	0.330	-	muscle	17
1995	Ocean perch	*Sebastes*	Greenland	0.171	-	muscle	17
1997	Ocean perch	*Sebastes*	Norway	0.182	-	muscle	17
1997	Ocean perch	*Sebastes*	North Sea	0.420	-	muscle	17
1995	Sardine	*Sardina pilchardus*	Bay of Biscay	0.603	-	muscle	17
1995	Catfish	*Anarhichas*	Greenland	0.388	-	muscle	17
1997	Catfish	*Anarhichas*	Norway	0.671	-	muscle	17
1996	Greenland halibut	*Reinhardtius hippoglossoides*	Greenland	0.541	-	muscle	17
1997	Greenland halibut	*Reinhardtius hippoglossoides*	North Sea	1.494	-	muscle	17
1997	Greenland halibut	*Reinhardtius hippoglossoides*	Faroe Islands	1.105	-	muscle	17
1997	Greenland halibut	*Reinhardtius hippoglossoides*	Iceland	1.469	-	muscle	17
1997	Greenland halibut	*Reinhardtius hippoglossoides*	Norway	0.370	-	muscle	17
1995	Hake	*Merluccius*	Bay of Biscay	0.086	-	muscle	17
1998	Hake	*Merluccius*	Pacific	0.030	-	muscle	17
1998	Hake	*Merluccius*	Chile	0.005	-	muscle	17
1998	Hake	*Merluccius*	Argentina	0.006	-	muscle	17
1996	Plaice	*Pleuronectes platessa*	North Sea	0.288	-	muscle	17
1997	Plaice	*Pleuronectes platessa*	German	0.355	-	muscle	17

			Bight				
1996	Saithe	*Pollachius virens*	North Sea	0.098	-	muscle	17
1997	Saithe	*Pollachius virens*	Norway	0.025	-	muscle	17
1995	Cod	*Gadus morhua*	North Sea	0.054	-	muscle	17
1997	Cod	*Gadus morhua*	Norway	0.041	-	muscle	17
1998	Alaska pollock	*Theragra chalcogramma*	Canada	0.007	-	muscle	17
1995	Shrimps	*Pandalus borealis*	Greenland	0.121	-	muscle	17
1997	Shrimps	*Pandalus borealis*	Norway	0.251	-	muscle	17
1998	Brown shrimp	*Crangon crangon*	North Sea	0.882	-	muscle	17
1998	Blue mussel	*Mytilus edulis*	Denmark	0.390	-	muscle	17
1995	Squid	*Loligo*	Bay of Biscay	0.117	-	muscle	17
2002	Blue crab	*Caliinectes sapidus*	Brazil	1.500	-	-	18
1999	North Pacific beaked whale	*Mesoplodon stejnegeri*	Japan	12.41379	2.90	muscle	28
1999	Finless black porpoise	*Neophocaena phocaenoides*	Japan	11.00000	5.40	muscle	28

Table A 2. Concentrations of dioxin in marine mammals reported in lipid weight.

Year	Common name	Scientific name	Area	Mean concentration (ng TEQ·kg^{1} of fat)	% lipid	Tissue type	Source
1990s	Hector's dolphin	*Cephalorhynchus hectori*	New Zealand	27.27181	-	blubber	19
1990s	Common dolphin	*Delphinus delphis*	New Zealand	0.22950	-	blubber	19
1990s	Gray's beaked whale	*Mesoplodon grayi*	New Zealand	0.34580	-	blubber	19
1990s	Minke whale	*Balaenoptera acutorostrata*	New Zealand	0.35355	-	blubber	19
2002	Sperm whale	*Physeter catodon*	Australia	4.00000	-	blubber	20
2002	Gray’s beaked whale	*Mesoplodon grayi*	Australia	0.57000	-	blubber	20
2002	Sperm whale	*Physeter catodon*	Australia	2.68000	-	blubber	20
2002	Long-finned pilot whale	*Globicephala mela*	Australia	0.65000	-	blubber	20
-	Humpback dolphin	*Sousa chinensis*	Australia	14.92000	-	blubber	20
2000	Bottlenose dolphin	*Tursiops truncatus*	Australia	0.00000	55.00	blubber	21
2000	Bottlenose dolphin	*Tursiops truncatus*	Australia	0.00035	56.00	blubber	21
2000	Bottlenose dolphin	*Tursiops truncatus*	Australia	0.86400	75.00	blubber	21
2000	Bottlenose dolphin	*Tursiops truncatus*	Australia	4.39000	54.00	blubber	21
1990s	Harp seal	*Phoca groenlandica*	Greenland	5.75000	-	blubber	22
1996	Dugong	*Dugong dugon*	Australia	17.30000	-	blubber	23
1999	Dugong	*Dugong dugon*	Thailand	12.00000	2.45	muscle	24
2001	Dugong	*Dugong dugon*	Thailand	11.00000	2.41	muscle	24
1991	Harbour seals	*Phoca vitulina*	Vancouver Island	2.83000	96.50	blubber	25
1991	Harbour seals	*Phoca vitulina*	Vancouver Island	7.15000	96.40	blubber	25
1995	Grey seals	*Halichoerus grypus*	Nova Scotia	3.23000	79.50	blubber	26
1995	Weddell seal	*Leptonichotes weddelli*	Antarctic	3.62600	100.00	blubber	12
1991	Ringed seals	*Phoca hispida*	Arctic	3.82000	89.10	blubber	27
1996	Ringed seals	*Phoca hispida*	Arctic	3.73000	96.30	blubber	27
2000	Ringed seals	*Phoca hispida*	Arctic	1.02000	91.40	blubber	27
1996	Ringed seals	*Phoca hispida*	Arctic	1.74000	93.60	blubber	27
2000	Ringed seals	*Phoca hispida*	Arctic	0.43000	91.50	blubber	27
1999	North Pacific beaked whale	*Mesoplodon stejnegeri*	Japan	9.65854	82.00	blubber	28
1999	Finless black porpoise	*Neophocaena phocaenoides*	Japan	7.59740	77.00	blubber	28

Table A 3. List of references for sources of dioxin concentrations in fish and other marine organisms.

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CHAPTER 7

SYNTHESIS: ON THE MULTIPLE USES OF FORAGE FISHES[10]

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ABSTRACT

Within the preceding chapters in this report it has been shown that 'forage' fish fisheries impact marine ecosystems, as well as human food security from a nutritional and food safety perspective. Also they are closely related to the development of aquaculture and other intensive food production systems. In part, these impacts and uses are misunderstood, notably due to the myth that forage fish are unsuitable for human consumption. In addition, the role of these fisheries in trophic transfers and in intensive food production systems is generally not well understood or appreciated. Nevertheless, there is considerable scope for policy-makers to change the current directions in managing these fisheries and their role in food security as well as economic development. There is also a greater role for industry to play in finding the balance between these fisheries contributing to poverty alleviation and sustaining intensive animal food production systems, especially aquaculture. This report highlights these respective roles.

INTRODUCTION

The 'Forage Fish Study', conducted in 2005-2006 by the *Sea Around Us* Project, and documented in this report, aimed to improve our understanding of the role of forage fish in marine ecosystems, highlight their importance to human well-being and their role in intensive food production including aquaculture, and make management recommendations for their sustainable use. Forage fisheries target small schooling pelagic fish that are easy to catch in large numbers and are inexpensive (Chapter 1). The fisheries which target forage fish are now concentrated in three areas of the world, the west coast of South America, which sustains the world's largest fishery for small pelagics, Northern Europe and the United States (Western Atlantic and Alaska). These fisheries are not only important to human well-being (Chapter 2), they are also important food to marine mammals and seabirds (Figure 1 and Chapter 3). Our studies on the overlap of forage fish and marine mammals and seabirds show, however, that we impact those animals far more than they impact us. Modeling of atmospheric dioxin (Chapter 5) and subsequent uptake in marine ecosystems (Chapter 6) in this study suggests that dioxin levels in forage fish caught in Europe are higher than in other major fishing areas. Processing these fish into fishmeal and fish oil further concentrates these pollutants. The trade of contaminated fish products (Chapter 4) distributes the dioxins further around the world and introduces them into the human food system.

Throughout the centuries, humans in all areas of the world consumed small pelagic fish, and in many places in the world they are still very popular (Chapter 2). The combination of low fishing costs, especially for fuel (Chapter 1), cheap market prices and the superior nutritional content of these fish also make them an important input of animal feeds for poultry and pigs, and more recently aquafeeds. Small pelagic fish also play an important role in the future of high-value aquaculture (Chapter 4). This competition is an emerging issue, as the results of this study show the increasing use of higher trophic level fish, which are consumed by humans in fishmeal (Chapter 4). The concentration of fishing in these three areas of the world also means that these fish and the derived fishmeal and fish oil are widely traded on the

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international market. The economics of these fisheries are subject to international factors such as the price of soymeal and are well beyond the control of fishers and producers (Chapter 4). Who benefits from this trade, however, is not well known.

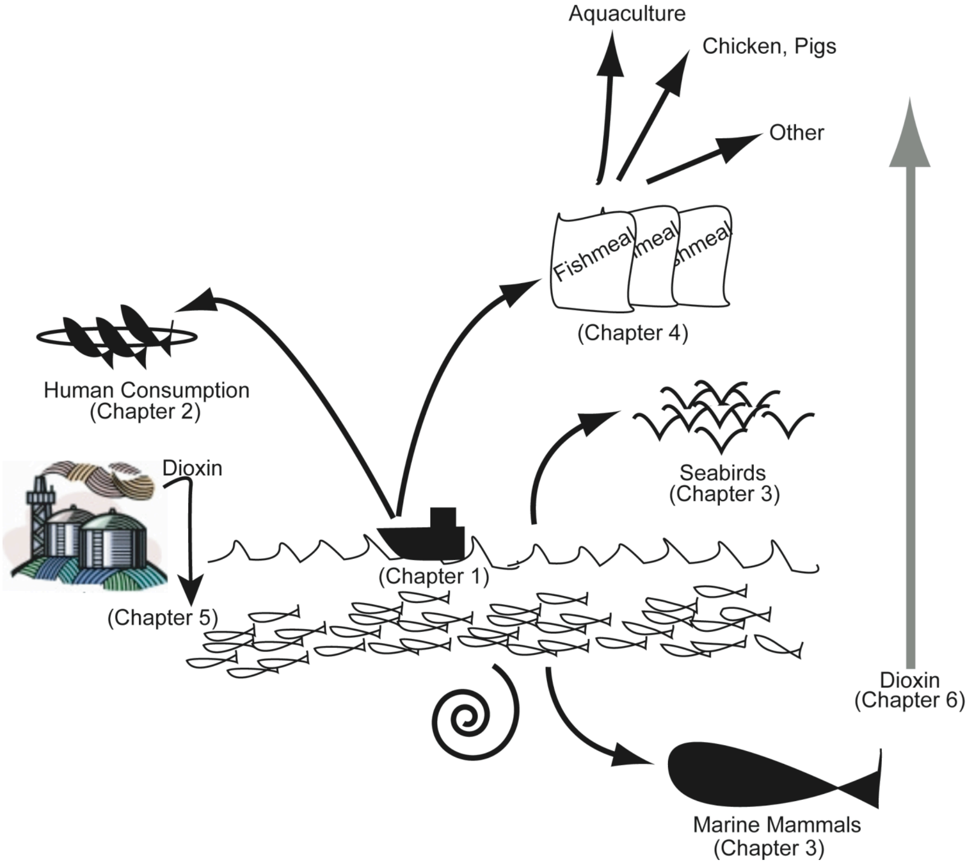

Figure 1. Fate of forage fish with the arrows representing flows described in the different chapters of this report

Fish products from the North Sea are particularly high in dioxin (Chapters 5 and 6) and other pollutants, and therefore trade on the global market results in the transportation of dioxins to other areas of the world (Chapter 4). When feeds and oils with high concentrations of these pollutants are fed to farmed animals, pollutants enter the human food chain and represent a risk for consumers, as seen in the controversy surrounding dioxin in farmed salmon. Several countries, developing and developed, are farming fish, especially high-value fish that are often carnivorous and require large inputs of fishmeal and fish oil. The aquafeed and aquaculture industries have made considerable improvements in the efficient use of fishmeal and fish oil and are working towards alternatives as well as ways to extract dioxin out of fish oil. Until alternatives that are as inexpensive as or cheaper than fishmeal and oil are found, the need to balance food security and economic development will remain. In the meantime, policy-makers need to consider the implications of current and future management of these fisheries. Below is a discussion of the implications of the findings of this study, with recommendations for management.

IMPLICATIONS

Fisheries management

Forage fisheries are currently concentrated in three areas of the world and there is a general consensus that most stocks are fished at their biological limits, with limited scope for expansion. One species in particular, Blue whiting, is fished outside of an agreed management plan and there is concern about its sustainability. The forage fisheries have also exhibited highly variable catches in part due to their sensitivity to changing oceanographic conditions, but also in part to depleting species and finding replacement species, as seen in Northern Europe (Chapter 1 and Figure 2).

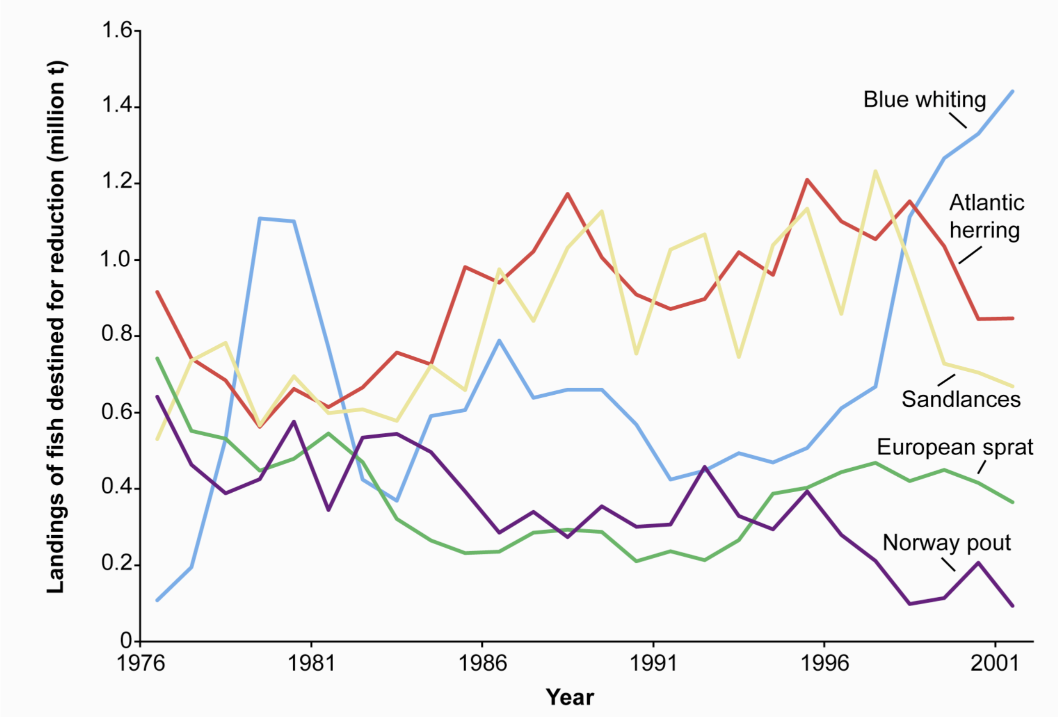

Figure 2. Trends in landing of fish destined for reduction 1976 to 2001 (Watson *et al.*, 2006; Chapter 1).

Management needs to consider the impacts of fishing on marine ecosystems and one mechanism is through certification programs such as that managed by the Marine Stewardship Council (MSC). The International Fishmeal and Fish oil Organization has already initiated contact to assess the feasibility of certification. Similarly purveyors of farmed fish are beginning to demand sustainably farmed fish and they are concerned with the issue of forage fish sustainability.

The industry will face significant challenges in meeting MSC certification since information on the effects of fishing is limited for most fisheries. This is seen in FAO's State of Fisheries and Aquaculture reports where the exploitation status of forage fish species is often reported on large geographic scales, masking the status of small and locally important stocks (Chapter 1).

The aquafeed sector still has scope to reduce pressure on landings of forage fish by continuing to improve the efficiency of feed and feeding regimes as well as developing alternatives to fishmeal and fish. The aquaculture sector can also continue to improve feeding efficiencies which ultimately use less fishmeal.

Managers and policy- makers should take a precautionary approach to managing forage fish fisheries because of the influence of oceanographic conditions and the unknown consequences of climate change and associated warming sea temperatures. Previous oceanographic events such as El Niño can have a significant impact on landings as well as impacting the prey populations for many marine mammals and seabirds (Chapter 1).

Marine ecosystems

Small pelagic fish, which make up the bulk of forage fisheries, have a very important role in trophic transfer and yet the impact of fishing forage fish to their maximum biological limits is not well understood (Chapter 1). The corollary of how fisheries impact seabird and marine mammals is also poorly understood. A recent study of the North Sea highlighted the lack of information to assess the effects of industrial fisheries on predators, seabirds and marine mammals (Chapter 1). Studies of the overlap of fisheries with marine mammals and seabirds (Chapter 3) have dispelled the myth that these animals consume fish that would be eaten by humans or compete with industrial users.

An improved understanding of the role of forage fish in marine ecosystems will enable policy makers to use an ecosystem-based approach to managing these fisheries ensuring that there is sufficient prey for other components of the ecosystem. This is especially important if sea temperatures rise as predicted since a well-managed ecosystem has more potential to buffer the predicted impacts of climate change and continue to contribute to human well being.

Recent concern over the level of pollutants such as dioxins and mercury in wild capture and farmed fish has also raised concerns about the quality of the marine ecosystems. This concern extends to marine systems that support forage fish because they are a significant component of aquafeeds. Atmospheric modeling of dioxin identified marine areas where the toxin has the highest concentrations globally and is available to marine food webs (Chapter 5). Overlaying these results with areas of major forage fisheries identified the northeast Atlantic as a potential hot spot for fish with high dioxin concentrations (Figure 3 and Chapters 5, 6).

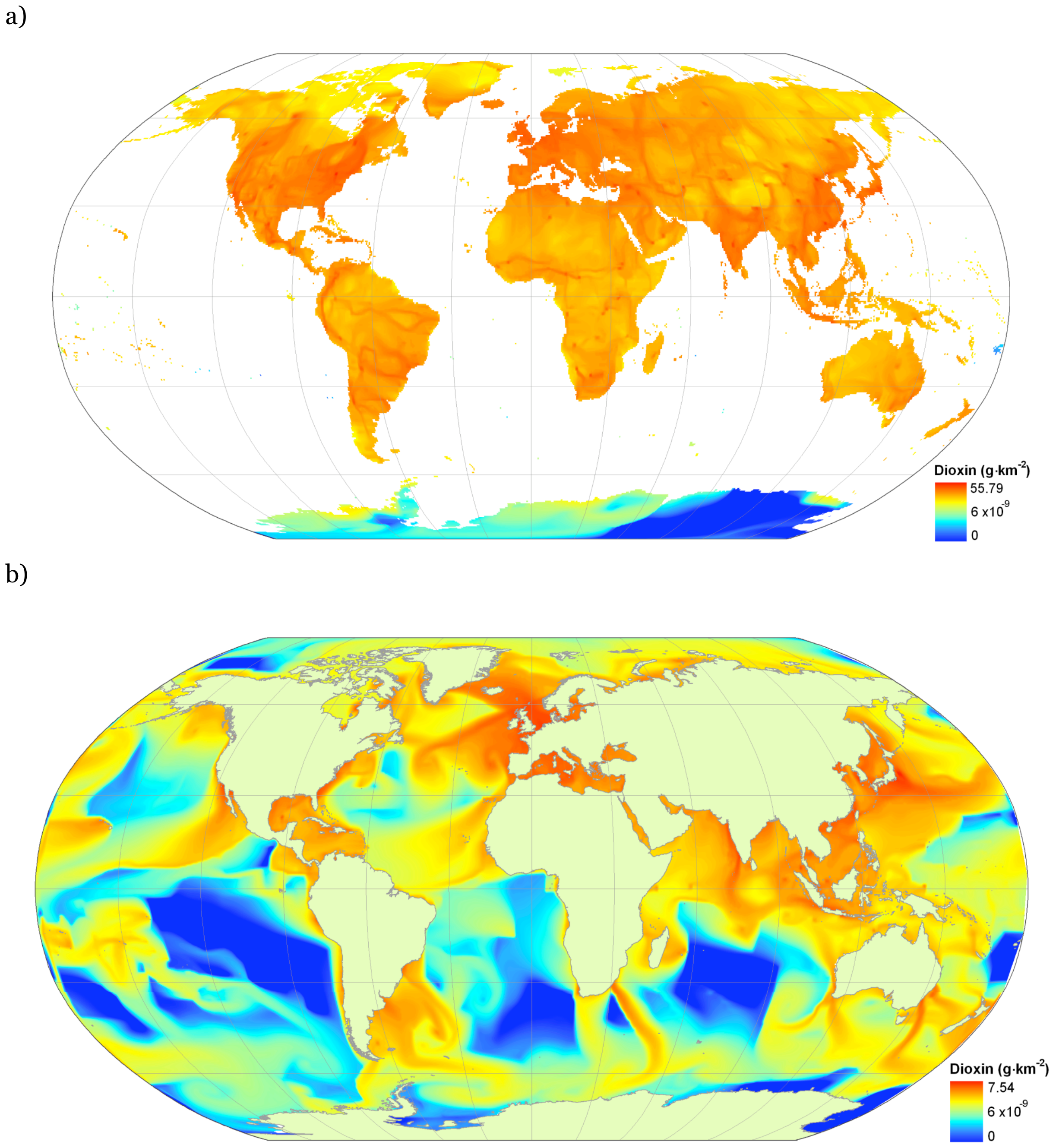

Figure 3. Global distribution of modeled dioxin deposition, based on the two seasonal, 10-year average wind field patterns combined, showing spatial distribution (a) on land, and (b) in the ocean. Note the difference in scale (Zeller *et al.*, 2006; Chapter 5).

Climate Change

The impact of climate change on forage fish stock is highly uncertain given the uncertainty associated with changes in ocean currents, upwellings, mixing of water masses, water salinity, oxygen levels, currents, ice formation and melting, etc. (Arnason, 2006). These hydrographic changes in turn affect habitats and ultimately the distribution and abundance of fish stocks. The intensity of these changes is unknown but there is consensus that it will vary from area to area depending on a multitude of factors including latitude and depth. The Arctic Climate Assessment Study (ACIA, 2003) found that the predicted global warming would expand the habitat range and thus benefit Atlanto-Scandian herring and Blue whiting but have the opposite effect for Capelin.

The certainty of climate change impacting on the world's oceans combined with the uncertainty in understanding its impacts on fish populations highlights the need to better understand the role of forage fish in marine systems. An ecosystem-based approach to management and the exploration of policy options that incorporate adaptation to climate in managing these fish should be taken now rather than waiting until the impacts of climate change are more evident. The kind of management initiatives that should be included to facilitate adaptation to climate are just beginning to be explored (Hannesson *et al.*, 2006).

Intensive food production

Fishmeal and fish oil, which are derived from forage fish, are important inputs into the intensive production of animals, primarily chickens, pigs and fish. There is a consistent and strong demand for fishmeal and fish oil because of their superior performance and low cost, resulting in the products being traded globally (Chapter 4). Disruptions to forage fish landings will ultimately affect the production of a range of food products. Although soymeal can substitute for fishmeal, the risk of diseases and lower growth rates is higher. Animal production is more price-sensitive than aquaculture and thus when prices increase producers will consume more soymeal. The aquaculture sector can substitute soymeal for fishmeal but no substitute for fish oil is yet available. As intensive animal production expands the demand for fishmeal and fish oil will also increase (Chapter 4) and can only be met by exploitation of stocks that are destined for human consumption, better utilization of bycatch and development of alternatives for fishmeal and fish oil. Otherwise these industries, especially aquaculture, will be constrained.

Food security and safety

The sensitivity of forage fish to oceanographic changes also highlights the urgency to better understand how their fisheries will respond to climate change. In terms of food security, forage fish are used to meet shortfalls in demand for cheap protein, as seen for example in Ghana. Disruptions to landings of small pelagics consumed by humans would directly impact on the food security of many developing countries (Chapters 1 and 2). All findings on the use of fishmeal and fish oil note that both commodities are important inputs into intensive animal production especially chickens and pigs, and more recently aquaculture. Thus declines in landings would impact the price of inputs into aquaculture and ultimately the affordability of protein in many areas of the world (Chapter 1 and 4).

The demand for fishmeal and fish oil to sustain growth in the intensive food production sector can impact food security. An analysis of changes in the trophic level of fish destined for reduction revealed the index has been increasing over the last 20 years; this suggests that fish earlier destined for human consumption are being diverted to producing fishmeal.

Food safety may be compromised when fishmeal and fish oil originate from areas where there are high dioxin levels. The North Atlantic marine waters are subjected to high deposition rates of dioxins (Chapter 5) and numerous studies have highlighted the high concentrations of dioxins in fishmeal and fish oil from this region (Figure 4 and Chapter 6). When the fishmeal and fish oil are used in aquafeeds, dioxins are introduced in the human food system and can pose a risk especially to children and pregnant women. This risk is global since fishmeal is traded globally.

Figure 4. Predicted relative dioxin concentrations for small pelagics. Scale is from low to high, (white or light blue) to (red/dark). (From Christensen and Booth, 2006; Chapter 6)

Plausible Futures

Forage fisheries products, fishmeal and fish oil, are highly traded globally and their future will depend on a number of factors including the development of aquaculture, the trends in soy production, the development of alternatives that are just as effective as fish oil, and climate change. The future of forage fish in the face of climate change is filled with uncertainty since it is unclear how oceans will respond to changes in upwellings, currents and sea temperatures. There is no doubt that forage fish will be impacted. However, the nature and severity of these impacts is unclear and needs further research.

In a future where aquaculture, especially for carnivorous fish, continues to expand without finding alternative sources of feeds, the price of fishmeal and fish oil will likely rise and the meals and oils will be increasingly consumed in the aquaculture sector at the expense of first the poultry sector and second the pig sector. The increasing price of fishmeal and fish oil will ultimately increase production costs with consumers paying more for farmed fish, and will affect the food security of developing countries by pricing forage fish and farmed fish out of their price range.

In a world of expanding aquaculture, price-competitive alternatives to fishmeal and fish oil will be found, with the likely result that the forage fish fisheries will continue at current levels, but the price of fishmeal and fish oil may decrease since demand will be lower. However, if fuel costs continue to rise, some fleets may fold so that only the most economically efficient vessels will continue to operate. In such a scenario, it is possible that increased use will be made of fishmeal and fish oil in higher-value products destined for human consumption.

However, if production of soymeal slows down or stalls, the price of fishmeal and fish oil will rise. There will be considerable pressure to find new sources of fishmeal as well as initiatives to improve the food conversion ratios where it is used. Such a scenario has significant implications for food security since high prices could see low-value fish normally destined for developing countries being diverted to reduction plants instead. This scenario also has implications for the use of bycatch, with an even greater incentive to use it in the aquaculture sector either as direct feed or as inputs to fishmeal and fish oil.

Conclusions

Forage fisheries are globally the largest fisheries, and they are critical to human well-being through direct and indirect uses of fish caught. Although these fisheries are found in three major areas of the world's oceans, their importance is global since they are a significant source of cheap protein for developing countries, an important input into animal food production and a key prey species for marine mammals and seabirds. Management of these fisheries is still single-species-based for many of the stock assessments; the highly variable annual catches and sensitivity of forage fisheries suggest that a precautionary approach to management using ecosystem-based management principles must be employed if a balance between meeting human food security, food production systems and meeting the ecological needs of the other animals in marine ecosystems is to be achieved.

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