GREENPEACE

Report on the World's Oceans

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1. Introduction

1.1 The Oceans and Ecological Services

The worlds' oceans comprise the largest habitat on earth. 71% of the surface of the earth is covered by seawater to an average depth of 3.8km. The total volume of this water is around 1.3 billion cubic kilometres and comprises around 0.24% of the total mass of the earth. (Angel 1997). These statistics hide a considerable diversity of habitat. The abyssal depths of the ocean between 3 and 6km in depth cover 51% of the surface with depths over 6km accounting for less than 2%. The continental slopes between 200m and 3km depth cover 13% of the surface while the continental shelves underlying water up to 200m deep account for 5% of the earth's surface. This huge biological system is richer in major groupings of animals than the land. Of the thirty four major taxonomic groupings (phyla) of animals, twenty nine occur in the sea and fourteen are found exclusively there (Tickell 1997). Coral reefs have long been known to very species rich (Paulay 1997) but new research methods have shown the ocean floors also to be extremely rich in biodiversity (Scheltema 1996; Gage 1997). Quite apart from the commercially valuable species it is increasingly being recognised that this diversity of life has an intrinsic moral as well as monetary value (O'Niell 1997; Oksanen 1997; Moyle & Moyle 1995).

The support of human existence by the oceans goes far beyond simple exploitation of fisheries and other coastal resources. The oceans provide humanity with a diverse array of what may be termed ecosystem or ecological services. In most cases these services accrue directly to humans without passing through formal monetary economies and any assigned value has to be treated as at best a very approximate estimate. Based on the functions of the open ocean as a regulator of atmospheric composition, nutrient cycling, food production and biological control of natural systems it has been estimated that it contributes $8,000 billion to the world economy. When coastal waters are included in such estimates, the contribution increases to around $21,000 billion (Costanza *et al.* 1997), amounting to an estimated 63% of the total value of marine and terrestrial systems combined. These values comprise around 1.8 times the current world GNP. Such figures are only indicative and undoubtedly they underestimate the true contribution. For example, insufficient data exist to estimate a notional fiscal contribution of the oceans to climate regulation although oceanic processes drive the worlds' climate and weather systems (see: Bernes 1996; AMAP 1997). There are a number of serious problems with derivation and use of these valuations

1.2 Ecological Services and Nominal Values

Uncertainties attached to estimates of ecosystem service value arise from the fact that in many cases they are derived on the basis of the willingness of individuals to pay for the services or functions in question. It is obvious that a truly accurate set of values is contingent upon these individuals living in a sustainable manner, and recognising the full extent of their connection to, and their dependence upon, ecosystem services. In addition, the lack of information about many biological systems, distortion introduced into the notional prices by a variety of factors, assumptions made about supply and demand, differences in national income levels and interdependencies in the ecosystem functions themselves, all conspire to introduce inaccuracies. Most importantly, the analysis assumes that there are no sharp thresholds, discontinuities or irreversibilities in the responses of ecosystems to perturbations. This is acknowledged as a false premise. Indeed it may never be possible to have a very precise estimate of the value of such services. Nonetheless, by viewing these figures as basic minimum indicative values it is clear that ecosystem services provide an important contribution to human welfare over the planet as a whole. Further, the contribution of the oceans is very substantial (Costanza *et al.* 1997) with the most highly valued services provided by coastal waters, including the tidal wetlands.

Misconceptions, however, can arise from assigning fiscal values to ecosystem functions. The most serious is the misconception that should these ecosystems be utilised in a non-sustainable manner then the assigned value represents the cost of substituting them by technological means. In fact, these values represent an imprecise, and to an extent, subjective estimate of the benefits of ecosystem services. An allied misconception is that environmental damage can be paid for and that this is as good as, or even preferable, to avoiding the damage in the first place (Beder 1996). These views fail to recognise that without ecological services such as those provided by the oceans, sustainable use of the planet would simply not be possible. Some of these functions are, in actuality, irreplaceable (Cairns & Dickson 1995). Attempts to couch disbenefits of environmental degradation in financial terms have been largely restricted to analysis of specific costs associated with increased underwriting risks in the insurance market resulting from global climate change (Tucker 1997). The use of economic instruments for *inter alia* the control of pollution has proven, as yet, to be of little practical benefit (Beder 1996). Economists will doubtless continue to argue that the environment must be given a monetary value and that the resulting values should be incorporated into environmental decision making processes. These efforts, however, involve pricing and in turn these prices should always tell the truth about the values of the ecological services upon which humanity depends. (Tickell 1997; Arrow *et al.* 1995 Costanza *et al.*1997). The actual costs (expressed as value) of the loss of an irreplaceable ecosystem function could quickly become infinite.

TABLE 1

BIOME	**Area** (ha x 10^6)	**Value** (\$US ha^{-1} y^{-1})	**Total Global Value** (\$US x 10^9 y^{-1})
Marine	**36,302**	577	**20,949**
Open Ocean	**33,200**	252	**8,381**
Coastal	**3,102**	4,052	**12,568**
Estuaries	180	22,832	4,110
Seagrass/algae	200	19,004	3,801
Coral Reefs	62	6,075	375
Shelf	2,660	1,610	4,283
Terrestrial	**15,323**	804	**12,319**
			3,813
Wetlands	**330**	14,785	**4,879**
Tidal Marsh/Mangroves	165	9,990	1,648
Swamps/Floodplains	165	19,580	3,231
Forest	**4,855**	969	**4,706**
Tropical	1,990	2,007	3,813
Temperate/Boreal	2,955	302	894
Grass/Rangelands	**3,898**	232	**906**
Lakes/Rivers	**200**	8,498	**1,700**
Desert	**1,925**		
Tundra	**743**		
Ice/Rock	**1,640**		
Cropland	**1,400**	92	**128**
Urban	**332**		
TOTAL	**51,625**		**33,268**

Table 1: The global value of ecosystem services calculated for the year 1994. Missing values denote insufficient information to make a calculation. Values were derived on the basis of a limited set of ecosystem services by Costanza *et al.* 1997. For example, no value is included for the role of oceans in climate regulation. For further details see text. In the context of marine systems, the most highly valued ecological services accrue from coastal waters.

1.3 Sustainability and Precaution

Extending economic analogy leads to the basic truth that current world development is proceeding on the basis of utilising ecological "capital" rather than the "interest" accruing from sustainable utilisation of ecological services (Cairns 1996). The impacts of many human activities cannot ultimately be justified, even through the most severely reduced interpretation of ecological economics, namely the simple precedence of benefit over cost. Put another way, some ecological services are being overused at current assigned prices (Costanza *et al.*1997) though this view comes dangerously close to the misconceptions outlined above. The fact, moreover, that a sustainable system can generally only be identified as such after the fact of exploitation adds considerably to the difficulty. Accordingly, definitions of sustainability are usually only predictions of the sets of conditions that will actually lead to sustainable systems (Cairns 1996), rather than robust definitive criteria. Resolving the conundrum of ensuring that ecosystem services are provided at a sustainable rate which meets societal demand without compromising the service for future generations is undoubtedly extremely difficult (Cairns 1997). As provisional overarching principles of sustainability, however, the four listed below have some merit. Their use as predictive or definitional tools is inevitably somewhat limited by incomplete or uncertain data relating to any proposed human activity. Nonetheless, they are useful because all are requirements for sustainability and taken together are sufficient to ensure sustainability. Taken as a set of encompassing conditions for sustainability, they can be used conveniently as a checklist against which human activities can at least be retrospectively evaluated in a relatively simple way. Uses of ecosystem services on a sustainable basis should not violate any of these principles. It follows that environmental protection should set standards such that there is a very high degree of certainty that these principles will not be compromised.

1) Substances from the earth's crust must not systematically increase in the ecosphere
2) Substances produced by society must not systematically increase in the ecosphere
3) The physical basis for productivity and diversity of nature must not be systematically diminished
4) Fair and efficient use of resources with respect to meeting human needs

The four provisional principles must be viewed against a background of a continuing increase in the pressures upon marine environments and the ecological services which they provide. It has been estimated that already some 75% of the total habitable area of the planet has been disturbed by human activity (Hannah *et al.* 1994) and this activity is focused in coastal areas. The population of the world is expected to reach some 7 billion by the year 2010 and will, therefore, have effectively doubled in little under 50 years (UNEP 1989). The worlds' population is located predominantly near the coast. Globally, over 65% of cities with a population greater than 2.5 million are coastally located. As an example, more than half of the population of the United States lives within 80km of the coast (OECD 1991) and, moreover, coastal populations appear to be increasing at a higher relative rate than the population in general. By 2050, global population could well have increased to around 12 billion and it is estimated that 60% of these humans will live within 60km of the sea (Elder & Pernetta 1991). It is in the fertile coastal, continental shelf region, therefore, where many of the impacts of human activities are currently evident and where, unless measures are taken, they will continue to become manifest.

An in depth examination of the problems facing the world's oceans and of the means best employed to resolve them is warranted. As a starting point, an appreciation of the considerable uncertainties which exist in human understanding of the varied processes normally taking place in the ocean must be developed. The uncertainties attached to scientific ability to predict the responses of ecosystems to human interference also need to be strategically recognised. Given this, it follows that concrete strategic outcomes which significantly contribute to achieving greater sustainability and viability of ocean ecosystems will need to be formulated on a basis inclusive of the inherent uncertainties. Only then will policies be based upon realistic scientific evaluation and enjoy widespread public confidence.

The most widely accepted means of protecting the environment in the face of the multifarious uncertainties is through the adoption of a Precautionary Approach. The wisdom of applying a Precautionary Approach as a central paradigm of environmental protection can be illustrated in relation to some of the known current impacts upon ocean systems. Examples of such impacts can be drawn from a number of areas into a considerable list and include: Sea dumping; chemical, radioactive and sewage discharges; pollution from ships; fisheries exploitation; degradation and destruction of coastal ecosystems. By considering these in relation to a precautionary paradigm and by formulating regulatory strategy accordingly there is a much greater chance that real improvements will result. An important corollary is that the future appearance of unanticipated and unpleasant surprises is also likely to be greatly minimised.

This document details some of the current and potential future impacts of human activities upon ocean systems with regard to the four provisional principles of sustainability as a checklist. It considers some of the implications of these impacts for future environmental sustainability and provides suggestions for addressing these under a precautionary paradigm of environmental protection. Given the potential scope of the subject, it is by no means a comprehensive analysis of the state of the global oceans. Nonetheless, it details the sort of measurable outcomes against which the effectiveness of the "Year of the Oceans" initiative may be gauged

2. The Precautionary Approach

2.1 The Need for a Precautionary Approach

A great number of international fora have adopted a precautionary approach to environmental protection. It is affirmed as a general protective axiom in Principle 15 and Agenda 21 of the 1992 Rio Declaration on Environmental Development. Other agreements espousing a precautionary approach include the UN Agreement on the Conservation and Management of Straddling and Highly Migratory Fish Stocks (1995), The OSPAR Convention (1992) the 1996 Protocol to the London Convention. and the Barcelona Convention as amended in 1995. The approach evolved originally from efforts to regulate and control hazardous chemicals entering the sea (Stairs & Johnston 1990; Johnston & Simmonds 1991; Jackson & Taylor 1992) as exemplified by the 1987 Third Ministerial Declaration on the North Sea (MINDEC 1987). It resulted from increasing recognition that ecological systems cannot be comprehensively observed and that impacts cannot, therefore, be regulated and controlled. Broadly speaking, a precautionary approach recognises scientific and technical limitations and promotes regulatory action in the absence of full evidence of a cause effect relationship. In short, it allows incomplete data, uncertainty and indeterminacy to be taken into account in a meaningful way in the decision making process.

Increasingly, the limitations of conventional toxicity testing regimes and biological surveys are being recognised (see: Cairns 1989). Uncertainties in understanding ecosystems are conveniently illustrated in terms of Figure 1 below. As levels of biological organisation increase from the level of individual organisms to ecosystems, the knowledge concerning important functions declines. This observation is also true of impacts exerted upon ecosystems by activities other than the discharge of hazardous substances. For example, the impact upon an individual fish removed by fishing activity is fairly clear. The implications of the removal of a large number of individuals for whole populations and ecosystems, by contrast, cannot be easily determined. Paradoxically, while changes at the sub-organismal or organismal level are the current cornerstone of ecological assessments, the protection of populations, communities and ecosystems is of greatest concern to environmental managers. It follows that the choice of both test and assessment endpoints for any given management goal is of critical importance (Suter 1994). This, of course, assumes that endpoints actually exist in the form of conveniently measurable parameters, and this is not often the case.

Overall, the scientific and technical limitations act to completely undermine attempts at assessing the environmental consequences associated with any given human activity. Far from being an analysis in numerical terms, environmental risk assessment procedures are increasingly recognised as being based on various serious misconceptions. These misconceptions are best characterised for toxicological assessments (Power & McCarty 1997), perhaps explaining the early application of precautionary approaches towards chemical regulation. Hence, the term "risk assessment" when applied to ecological systems should not be confused with actuarial risk analysis based upon hard data as practiced in the fields of engineering or insurance underwriting. The interpretation of the relevance of, for example, toxicity tests or fish population data requires an insight into the functioning of the whole ecosystem in terms of multiple anthropogenic and natural stressors and the interactions between those stressors. Unfortunately, ecologists do not currently understand which factors are the most critical (Power & McCarty 1997). Ecological risk assessment can be best viewed, perhaps, as a developing arm of the "prediction industry" which is more traditionally associated with financial markets, meteorological forecasts and personal horoscopes. This industry, with considerable justification, is coming under increased critical scrutiny in relation to the accuracy of the results that it achieves (Sherden 1998). Considerable problems arise in attempting to define the primary aim of science in terms of prediction. The predictive prospects of science are limited by the role of chance and chaos which create uncertainties (Rescher 1998). Science undoubtedly has a role in description, classification, evaluation and control of problems. Unless uncertainties are explicitly recognised, the major constraint on scientific prediction is not incorporated into the predictive equation. The necessary alternative to the current permissive regimes of environmental regulation and control is found in a precautionary approach.

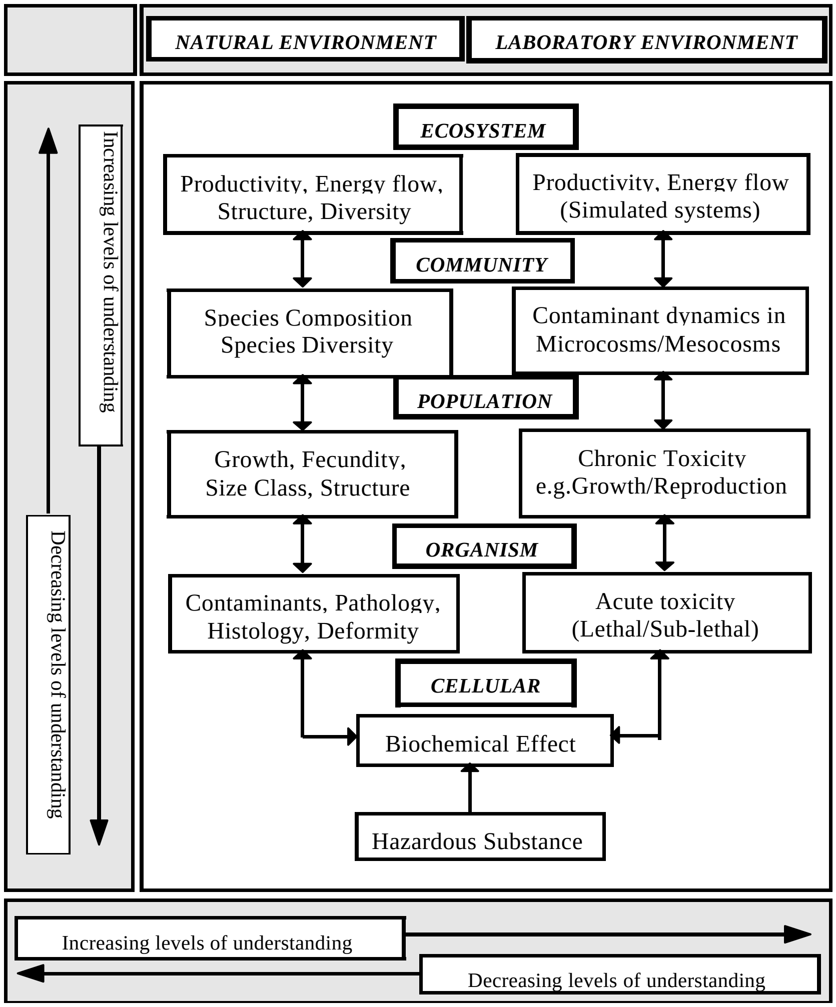


Figure 1: Diagrammatic representation of the relationship between level of biological organisation and degree of system understanding for both laboratory and natural environments. In general terms, as the level of organisation increases and control of conditions decreases so our understanding of system functions and interactions declines. The diagram illustrates these uncertainties in relation to toxic chemical inputs, but is equally valid in considering a range of environmental impacts.

2.2 A Precautionary Approach Defined

A precautionary approach to environmental protection can be defined as:

The emplacement of appropriate preventative measures when there is reason to believe that harm is likely to be caused by anthropogenic activities including the introduction of substances or energy into the environment and the extraction of marine species (including non-target species). Action should be taken even where there is not conclusive evidence to prove a causal relationship between the actions and their effects.

This precautionary approach to environmental protection contrasts with measures taken only after harm has been identified and allows consideration of all the information available, It effectively reverses the burden of proof and places it upon those seeking to exploit ocean resources at the potential expense of ecological services. Unresolved uncertainties compromise both ecological economics and environmental risk assessment. A precautionary approach allows these areas of uncertainty to be identified and considered explicitly in regulatory processes. Such an approach, however, has a wider application than the formulation of regulatory instruments for human activities and can be used more holistically. They can be applied to examination of the activities in relation to the four provisional principles of sustainability outlined above. If it cannot be proven that a given activity is not going to violate one or more of these principles, then this activity should be prohibited or more tightly controlled. In this way, a precautionary approach to environmental protection can be regarded as an instrument of both regulatory activity and sustainability.

Despite the noteworthy developments in accepting a precautionary approach in various international fora, it has not been universally applied. Effective implementation of the approach needs to be actively supported, promoted and implemented by national Governments. By promoting effective application of such an approach important linkages are more likely to be crystallised among the diverse, but interconnected initiatives which are intended to promote the sustainability and the viability of the marine realm.

3. Inputs of Polluting Substances

3.1 The Nature of Pollutants and their Sources

Much scientific confusion and difficulty has resulted from certain definitions of contamination and pollution. In particular, regarding a contaminant as a pollutant only when it causes demonstrable harmful effects calls into question the scientific techniques available to actually detect such changes. It also begs the question of what constitutes a background level for chemical substances foreign to living organisms. Taking these difficulties into account, more succinct, useful and mature definitions of pollution have evolved (King *et al.* 1984; Mason 1991). If a pollutant is defined as a potentially harmful agent that occurs in the environment as a result of human activity, pollution can, in turn be defined as the release to the environment of a chemical, physical or biological agent that has the potential to damage the health of humans or non-human organisms (Suter 1993). Importantly, the use of such a definition allows protective measures to be invoked on the basis that a potential for harm exists.

By far the greatest source of polluting substances into marine environments is from human land based activities. As shown in Figure 2, some 77% of pollutants enter the seas from this route, while shipping (12%) and sea dumping (10%) contribute around 22% in total (ICS 1997).

Hence, pollutants reach the marine environment by both direct and indirect means. King *et al.* (1984) provide a more comprehensive, but not complete, list of the potential sources.

a). Domestic Sewage
b). Industrial discharges
c). Leachate from waste tips
d). Atmospheric fall-out
e). Urban and industrial run-off
f). Accidents- Spillages and explosions
g). Oil production
h). Mining
i). Agriculture (nutrients and pesticides)
j). Chemical solubilisation due to e.g. acid rain
k). Power generation and other waste heat sources
l). Natural pollutant sources: e.g. Volcanoes and forest fires.

In this list, the last category does not strictly conform to the definition of pollution outlined above. There is also one important omission in addition to marine transport from the list. Radioactive pollutants must be added as a specific concern. Pollutants entering marine systems can be divided into two broad chemical categories. Inorganic substances include phosphates, nitrates, metals and radionuclides. Organic chemicals, based around carbon molecular structures include pesticides, PCBs, chlorinated dioxins, oil based hydrocarbons and combustion products such as the PAHs. In some cases, biological pollutants such as bacteria and viruses can have very important impacts upon water quality.

The plant nutrients, such as phosphates and nitrates, enter the sea through sewage discharges and as a result of the agricultural use of inorganic fertilisers. Impacts are most severe in nearshore waters. Metal inputs result from mining, smelting and other industrial operations. In addition to any local impacts, metals can be transported to distant areas through the atmosphere. Many organic chemicals, emitted from industrial processes and from their use in agriculture, have no natural counterparts. They too can be transported to remote areas by the atmosphere, and pollute marine environments far from their point of origin. Radioactive pollutants enter the sea from nuclear power plants, nuclear fuel reprocessing, weapons testing, oil extraction and phosphate rock processing. Ocean currents can transport these radioisotopes considerable distances. Figure 3 shows marine areas in which pollution risks of various kinds have been identified. Although the major areas affected appear to be coastal regions and enclosed seas this does not mean that open ocean areas are not threatened. As pointed out by Davis (1993), for many pollutants, open ocean concentrations may be equal to or exceed levels in coastal waters. Another factor which must be considered is that coastal waters are more amenable to study and therefore deleterious effects are more likely to be identified in these areas.

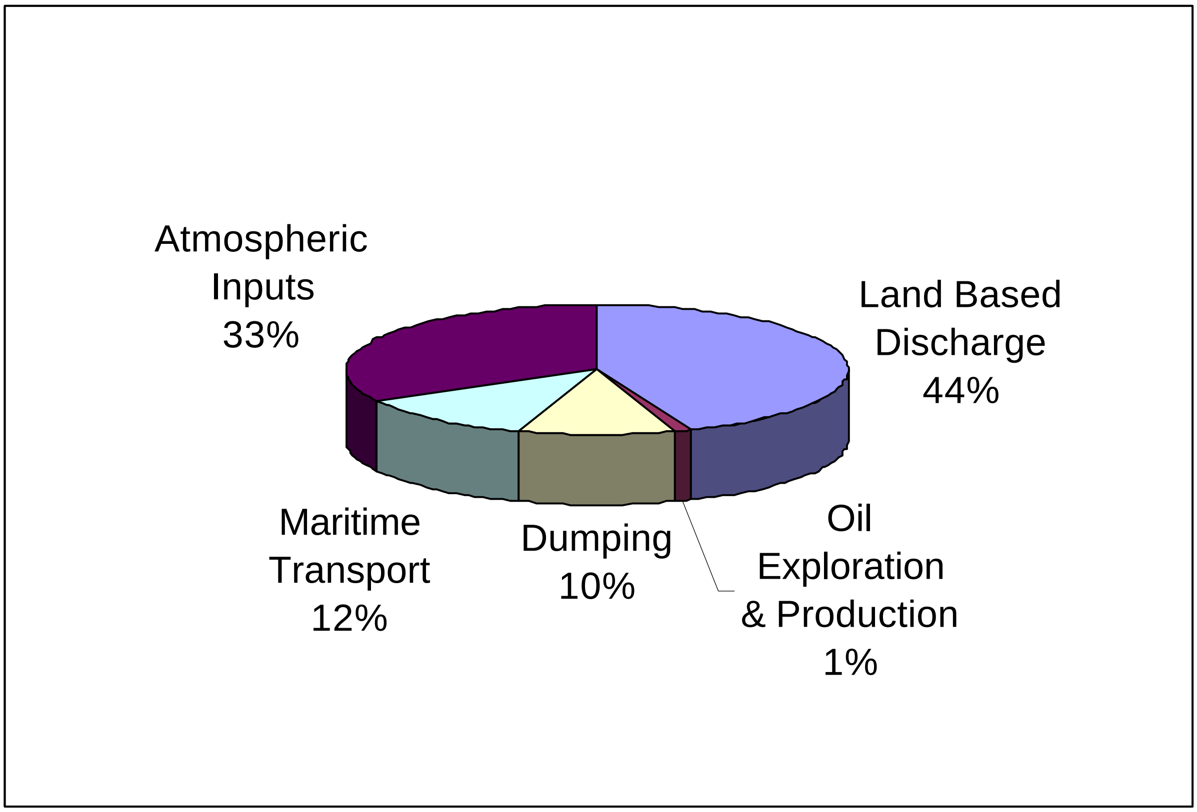


FIGURE 2: Proportions of pollutants introduced to the marine environment from various sources. Dumping of industrial waste at sea is now prohibited under the terms of the London Convention (1972) although sewage sludge and contaminated dredge spoil are still disposed of at sea. The major source of pollutants is human activity which also accounts for a significant proportion of pollutants entering the atmosphere.

On the basis of what is known about polluting substances in marine environments, it is clear that the provisional principles of sustainability are being violated. Naturally occurring and xenobiotic substances are systematically accumulating in the ecosphere. There is convincing evidence that these are causing widespread disturbance of biological systems.

3.2 Trace Metal Pollution

The discharge of metal pollutants and subsequent effects upon the environment and human health were first recognised many centuries ago. Nriagu (1992) notes literary references to metal pollution extending from classical Greek times, through the Middle Ages to the modern era. Iron and non-ferrous metals have long been central to human activity, but in early times the environmental impacts were localised in extent. The development of large smelters in Europe in the 16th century is reflected in increased levels of metals in dated peat and sediment cores. By the middle of the 17th century industrial activity in Britain and Central Europe was causing elevated levels of metals to build up in remote regions of Scandinavia (Nriagu 1992).

Overall, it has become clear that there has been a steady increase in the environmental burden of metals from human activity in historical times. Increases in the metal content of a series of agricultural soil samples collected since the mid-1800s in the UK and archived prior to analysis are described by Jones *et al.* (1987 a & b). Inputs of some metals into the atmosphere have trebled or quadrupled over the period between 1900 and 1990 (Nriagu 1992). Atmospheric deposition accounts for the increases in metal content observed in soils, but can also be a dominant component of the metals entering marine environments. For example, Chester *et al.* (1993) calculated that in excess of 50% of the metals entering the North Sea do so *via* the atmosphere.

Table 2 shows representative estimated values for global metal budgets from natural and anthropogenic sources (Nriagu 1989). These figures, while indicative, like all input estimates have relatively wide potential ranges in value. Nonetheless, considering inputs of metals to the environment from various sources (Nriagu 1990), it is clear that fluxes of metals from human activities now equal or exceed inputs from natural origin. In short, human activity is now the most important determinant of global biogeochemical cycling of the trace metals. The atmosphere, moreover is the single most important pathway for the wide environmental mobilisation and dissemination of these substances.

TABLE 2

	ELEMENT (x 1000 tonnes)							
	As	**Cd**	**Cr**	**Cu**	**Pb**	**Hg**	**Ni**	**Zn**
SOURCE								
Natural emissions to Atmosphere	12	1.4	43	28	12	2.5	29	45
Anthropogenic emissions to Atmosphere	19	7.6	31	35	332	3.6	52	132
Inputs to aquatic systems	42	9.1	143	112	138	6.5	114	237
Total flux due to weathering	90	4.5	810	375	180	0.9	255	540
River input to oceans (dissolved)	1.6	0.07	5.8	4.0	0.29	0.03	4.6	5.6
Atmospheric input to oceans	5.8	3.2	-	34	88	1.7	25	136

TABLE 2: Comparison of annual natural and anthropogenic inputs of toxic metals to the environment in thousands of tonnes. Aquatic inputs include inputs from atmosphere derived from anthropogenic sources. Weathering estimates calculated from average soil concentrations and riverine suspended particulate flux. Data from Nriagu (1990; 1992).

There are distinct local and regional variations in metal inputs to aquatic systems via the atmosphere and through direct discharges. Differences of over two orders of magnitude have been recorded between metal levels in urban rainfall as compared to those in precipitation in areas remote from industry and settlement. Estimated fluxes of metals to the oceans from the atmosphere show strong regional variations. Inputs follow the general pattern of N. Atlantic > N. Indian > N.Pacific > S. Atlantic > S. Indian > S. Pacific (Nriagu 1992). These inputs are closely related to the current intensities of metal emissions in the respective regions. Currently, the average lead concentration in the mixed surface layers of the North Atlantic is around 26 times higher than in equivalent waters from the South Pacific. Similar differences in the levels of metals found in Arctic as compared to Antarctic regions also reflect the greater degree of industrialisation in the Northern Hemisphere. In Antarctica itself, however, lead deposition appears to be dominated by aviation and automotive fuel exhaust, while other metal releases are detectable up to 10km downwind from point sources (Boutron & Wolff 1989; Suttie and Wolff 1993) Recently deposited coral skeletons in the Pacific contain 15 times more lead than skeletal material produced one hundred years ago. Unsurprisingly, atmospheric inputs of metals to regional seas such as the Mediterranean, North and Black Seas are between one and two orders of magnitude higher than those estimated for open ocean waters (Nriagu 1992). Figure 3 shows comparative values for cadmium and lead.

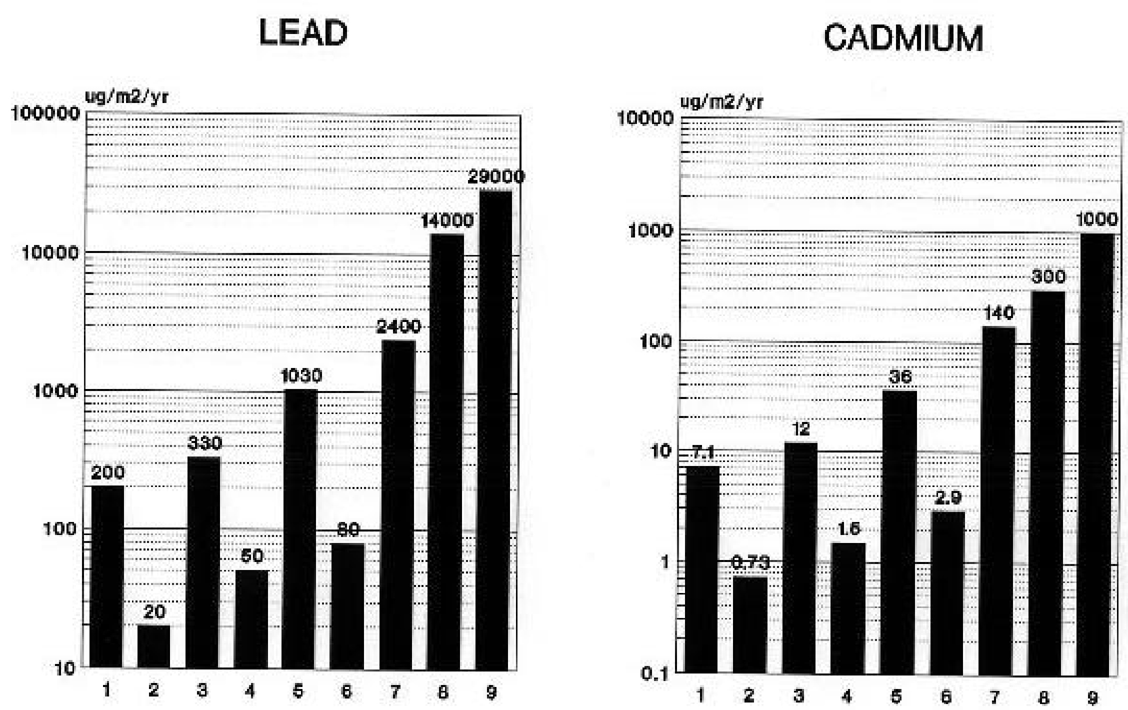


FIGURE 3: Fluxes of the trace metals lead and cadmium into seas and oceans calculated on a regional basis. Data taken from Nriagu (1992). Key: 1: North Pacific; 2: South Pacific; 3: North Indian; 4: South Indian; 5: North Atlantic; 6: South Atlantic; 7: Black Sea; 8: North Sea; 9: Northwest Mediterranean.

The processes governing the biogeochemical cycling of trace metals are highly complex. They include inorganic and organic complexation, particulate adsorption and redox processes in water and sediments. In turn, the biological availability and activity of metals in the environment is determined by such processes (Turner 1990; Sadiq 1992). In terrestrial environments, single point sources may exert substantial impacts. In the case of one primary lead smelter in the UK, severe contamination of woodland 3.2km downwind was correlated with disturbances of soil and woodland ecology, with elevated lead levels being found even at the control site 28km from the smelter (Martin *et al.* 1982; Hutton 1984). Estimated annual stack emissions of cadmium, lead and zinc were 3.5, 20-30 and 40-60 tonnes respectively (Hutton & Symon 1985). Recent research at the same site has shown that levels of zinc discharged by the smelter appear to be responsible for the absence of certain soil invertebrates in its vicinity (Hopkin & Hames 1994).

Severe disturbances of terrestrial and coastal marine ecosystems have also been recorded in the vicinity of mining operations where metal rich mine tailings are dumped, sometimes directly into the sea (see: Ellis 1988). In general, aquatic organisms, including fish eating birds and mammals at the apex of the food chain, are more sensitive to toxic metals in their habitat than terrestrial biota. Recently, the effects of one point source discharge of metals from a phosphate rock processing facility in the Irish Sea have been investigated. The diversity of shore dwelling organisms was found to be reduced several kilometres from the discharge and the overall detectable impact extended up to 40km from the source (Langston *et al* 1992).

Toxicological studies upon metals in marine environments are complicated by the fact that trace metals occur naturally and that, moreover, both vertebrate and invertebrate organisms have the ability to regulate tissue levels of these pollutants. Bryan & Hummerstone (1977) describe the tolerance of marine invertebrates to metals in an area of South West England exposed over a long period of time to metals from mining wastes. The mechanisms by which invertebrates accumulate, store and regulate their body metal burdens are discussed by Rainbow (1988; 1990), Rainbow *et al.* (1990), Thompson (1990) and George and Olsson (1994). Metals may be immobilised and stored in invertebrate tissues either temporarily or permanently. In some cases, biological transformations of metals may take place which alter their toxicity. Methylation of tin and mercury increases the toxicity compared to the original metal species. Methylation of arsenic decreases its toxicity (Hamasaki *et al.* 1995; Ford & Ryan 1995).

Since most marine invertebrates fail to match rates of metal excretion to rates of uptake they will tend to accumulate metals even when ambient levels are normal. These processes lead to age related metal accumulation and they also effectively mobilise metals into the food chain. The effect of mobilising metals into marine food chains is strikingly illustrated by historical changes in mercury concentrations in the feathers of adult common terns and herring gulls from the German North Sea coast. These have risen by 377% and 75% respectively as compared to levels present in pre-1940 museum specimens (Thompson *et al.* 1993). Adverse effects of increasing metal levels upon marine organisms have not been unequivocally demonstrated on a regional scale. This must, however, be placed within the context of a general lack of suitable evaluative protocols available to aquatic toxicologists to detect effects on such a scale (Cairns 1989; Nriagu 1990; Suter 1993). Nonetheless, such effects are possible. For example, Scheuhammer (1987) notes that the chronic exposure of birds to metals caused by feeding upon metal contaminated aquatic invertebrates may have important impacts upon reproduction. These could include decreased egg production, decreased hatchability and increased hatchling mortality. Young, growing birds are more sensitive to metal toxicity than adults.

A surprising number of metals have been implicated in interference with the reproductive processes of fish. In many cases, the juvenile and embryonic stages are the most sensitive life stages. Effects of metals range from degenerative lesions of male and female reproductive organs, to reduced production of eggs and sperm and reduction in fertilisation rates (Kime 1995).

Metal pollutants have been examined in the tissues of cetaceans from both coastal and open ocean environments (see: Thompson 1990). Coastal cetacean species tend to have higher levels of metals in their tissues than those frequenting deeper water. There also appears to be a relationship between metal levels and diet; cetaceans feeding upon cephalopods have higher levels of cadmium in their tissues than comparable fish eating species. There is a striking difference between the high mercury levels in tissue of the toothed whales and those found in the baleen whales (Dietz *et al.* 1990). The overall picture is complicated by the operation of numerous variables, hence intraspecific differences are more reliable indicators of the geographical variables. Hence, the presence of mercury in Beluga whales from the St. Lawrence Estuary at levels up to 15 times those found in the same species from 5 Arctic locations (Wagemann *et al.* 1990), is a reflection of industrial inputs to the system.

Evaluation of the toxicological significance of increased environmental metal pollution, as noted above, is made difficult by the existence of homeostatic mechanisms controlling organismal metal levels, mechanisms of biotransformation and equivocal toxicological indices. As pointed out by Nriagu (1990) and Depledge & Fossi (1994) changes in the vital signs of organisms affected by metals are likely to become obvious only after any (largely uncharacterised) homeostatic mechanisms are overwhelmed although various biomarkers may be used to chart the development of responses. Symptoms may well range from early lesions to overt disease, which may in turn be of a general non-specific nature. Some measure of the potential problem faced by human populations can be gained from estimates which suggest that anywhere between 250,000-500,000 humans may suffer from renal dysfunction as a result of cadmium exposure. 40,000-80,000 people globally may be suffering from mercury poisoning as a result of consumption of contaminated seafood. The figures for lead are even more alarming, with perhaps as many as 200 million people at risk of poisoning (Nriagu 1988) through a variety of pathways.

There is no doubt that metals usage and hence emissions of metals to the environment will continue to increase. Although initiatives in industrialised countries appear to be reducing metal emissions to a certain extent (Greiner *et al.* 1994), this must be offset against the potential effects industrial developments elsewhere, such as the African continent (Biney *et al.* 1994). Once dispersed into the biosphere metal pollution cannot be remediated and environmental effects are likely to be extremely long lasting. This problem is well illustrated by the formidable difficulties in remediating extensive areas of the Amazon basin where up to 1 million informal miners are engaged in gold extraction using the mercury amalgam process (Veiga & Meech 1995). During this process around 1kg of mercury is lost for every kilo of gold produced. On a wider basis, progressive increases in environmental metal levels are tied to market economies predicated upon stimulated demand, changing fashions and built-in obsolescence. This is coupled to a complete reliance on inefficient systems driven by fossil fuels which themselves contain metals. Unless this destructive spiral is broken, in both developed and developing economies, pollution of marine environments with metals will inevitably continue to increase.

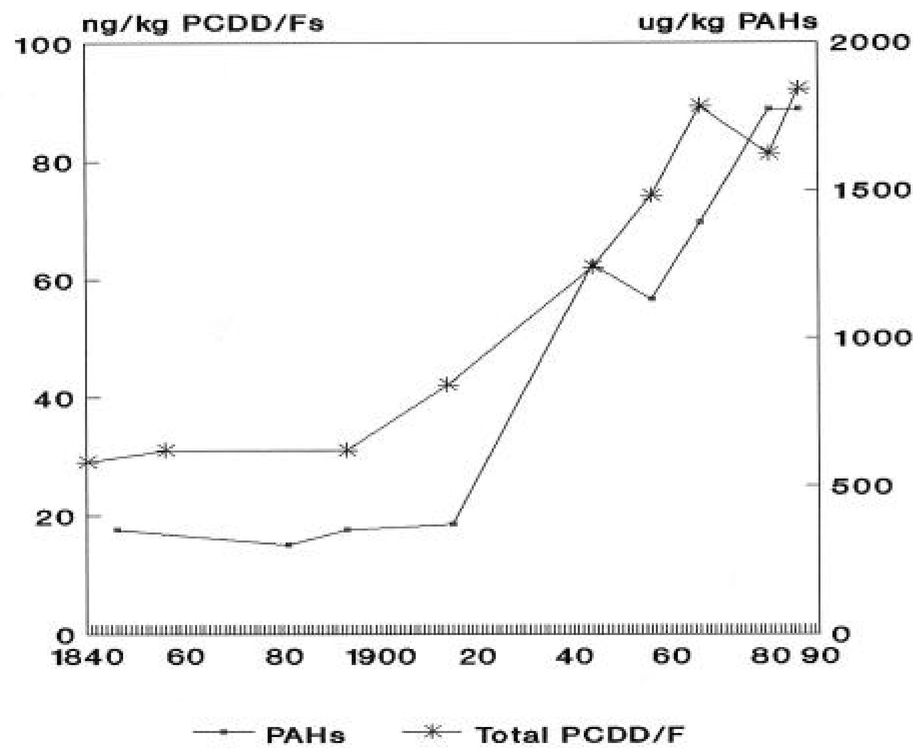
3.3 Organic Chemical Pollution

a) Sources of organic pollutants

Some 63,000 chemicals are estimated to be in common use world-wide and around 3,000 account for 90% of the total production. It is also estimated that anywhere between 200-1000 new synthetic chemicals enter the market each year (Shane 1994). Other figures suggest that in the EC alone, 50,000 substances are in use of which 4,500 are demonstrably toxic, persistent in the environment and may have the capacity to accumulate in the tissues of living organisms Edwards (1992). Emissions of organic chemicals began to increase with the advent of widespread industrialisation and associated fossil fuel consumption. This gained pace as industry specialising in petroleum processing developed, together with a market in petrochemicals. Figure 4 shows the way in which environmental levels of poly-nuclear aromatic hydrocarbons (PAHs), useful indicators of petroleum-based industrialisation, have increased with time together with concentrations of chlorinated dioxins and dibenzofurans.

A key industrial development was the emergence of a chemicals market based upon chlorine chemistry. This led to a range of synthesised chemicals being introduced in the late 1920's. The diversity of such chemicals and their market development and penetration increased rapidly in subsequent decades. Figure 4 shows the increase in environmental levels of chlorinated dioxins with time. These compounds, produced as unintentional by-products of chlorine chemistry, serve as a graphic index of the increase in the industrial production of chlorinated organics in the twentieth century. Both sets of data were obtained from the analysis of agricultural soil samples archived periodically from the mid-nineteenth century onwards (Kjeller *et al.* 1991; Jones *et al.* 1989).

Chemicals such as the organochlorine pesticides DDT, aldrin, dieldrin, toxaphene and chlordane first went into production this century. Polychlorinated biphenyls (PCBs) went into commercial production in the late 1920's, and as evidenced in Figure 4, the environmental levels of the chlorinated dioxins and dibenzofurans started to increase rapidly at this time. Public alarm was first sounded over DDT and other organochlorine pesticides in the landmark publication "Silent Spring" (Carson 1962). The presence of PCB residues in wildlife was reported by Jensen (1966) while the chlorinated dioxins and



dibenzofurans were recognised as a significant environmental health issue following the industrial accident at Seveso, Italy in 1976 (Marshall 1987). These identified persistent and bioaccumulative compounds are, however, a small subset of industrial chemicals discharged into marine ecosystems.

FIGURE 4: Increase with time of total chlorinated dioxins and dibenzofurans (PCDD/Fs) and PAHs with time in an archived series of agricultural soils from the UK. Data from Kjeller *et al.* (1991) and Jones *et al.* (1988).

b) Chemical complexity

The quantities of organic chemicals emitted to the environment are subject to considerable quantitative and qualitative uncertainty. Many more chemicals are present in wastes and discharges than are regulated. Moreover, many of the chemicals present cannot be readily identified using instrumental analytical techniques. The data in Table 3 are taken from discharges entering the marine environment in a number of countries. They demonstrate that chemicals which are not characterised, or only poorly so, are being routinely emitted to marine ecosystems (see: Johnston *et al.* 1991; 1994; Swindlehurst *et al.* 1995).

INDUSTRIAL SECTOR	PEAKS RESOLVED	MATCHED > 90% (% UNMATCHED)	MATCHED <90% >50% (% UNMATCHED)	LOCATION
A **Petrochemicals**	30	8 (73.3)	15 (23.3)	Italy
B **Mixed sewage/industrial**	18	4 (77.7)	10 (22.2)	Italy
C **Aluminium**	16	5 (68.8)	11 (0.0)	Italy

D **Mixed sewage/industrial**	132	23 (82.5)	50 (44.7)	UK
E **Industrial**	158	36 (77.3)	79 (13.5)	UK
F **Industrial**	16	9 (43.75)	4 (18.75)	Spain
G **Sewage**	26	2 (92.3)	12 (46.15)	France

TABLE 3: Number of peaks resolved under identical conditions of sample preparation and machine settings from simple hexane extracts of effluent samples. Samples were unmodified effluents as discharged. Peaks matched against the US NBS spectral library at >90% probability and <90% but >50% probability levels are recorded with the percentage remaining unmatched at each probability in brackets.

It follows that if it is not possible to identify a chemical then it is not possible to establish the threat that it poses to the integrity of the receiving system. The data for sample E in Table 3 are of particular interest. Chemicals present in this discharge have been detected a considerable distance offshore in the North Sea (Law *et al.* 1991). Additionally, evaluation of the identified chemicals present in the estuarial waters in close proximity to the discharge revealed that ecotoxicological data were not available for 75% of the chemicals isolated. The researchers made the observation that no reliable method exists for assessing the toxicity of chemical mixtures (Matthiesen *et al.* 1993). The problem of complex chemical mixtures is one of global dimension.

c) Organochlorine pollutants

i) PCBs

There is no doubt that the major focus of research and of subsequent regulation has been the organochlorine pollutants. The pesticide DDT was the first of these to be identified as present in the tissues of marine organisms. The polychlorinated biphenyls (PCBs) were isolated in marine mammal tissues shortly thereafter. The group of chlorinated pesticides, PCBs and chlorinated dioxins have been identified as Persistent Organic Pollutants (POPs). Because of their environmental significance they have been described as "one of the greatest environmental challenges the world faces" by the US State Department (Wania & Mackay 1996). The PCBs give a useful insight into problems associated with other organochlorine compounds. It is often assumed that because regulations have been emplaced for PCBs and many of the chlorinated pesticides by developed nations, the environmental problems have now been solved. This is far from the truth.

A voluntary ban on PCB production was emplaced by both US and UK manufacturers in 1977. This followed the identification of PCBs as contaminants of marine ecosystems by Jensen (1966), following on from realisation of the environmental problems associated with DDT. Production continued in other countries until 1985, with France and Spain being the last known producers (IEM 1995). The ban on production in the US and Europe came at a point when cumulative production from 1929 onwards was estimated at 1.5 million tonnes. Production peaked at the end of the 1960's (De Voogt & Brinkman 1989). It has been estimated that around 31% of the total tonnage of PCBs produced have entered the environment (Tanabe 1988a). Around 4% were accounted for by degradation and incineration, leaving the balance of 65% in dumps, landfills and current electrical applications. Over the next 10-15 years many of these electrical capacitors and transformers will come to the end of their useful lives (OECD 1987). Hence, in the absence of a rigorous plan to address the problem, future losses of PCBs to the environment could be double the estimated releases to date.

Undoubtedly, in those areas where local inputs have been brought under tight control PCB levels in the environment have generally fallen. This is true of the Baltic region (Ahlborg *et al.* 1992), the Great Lakes (Swackhamer & Eisenreich 1991), the Rhine Delta (RIVO 1993). There is an important *caveat* in each of these cases. In the Baltic region human exposure is currently of the same order of magnitude as that associated with subtle human health effects. In the Great Lakes, contaminated sediments are degassing PCBs and acting as a source into the global environment. In the Rhine Delta at some sampling sites levels of PCBs in eels have actually been found to have increased and elsewhere levels have fallen to levels only marginally lower in the 1990s than at the end of the 1970's when controls first came into force.

In general, regulatory measures appear to have done little more than halt the upward trend of PCBs in wider marine systems. In the Baltic Sea, with the exception of DDT, levels of organochlorines in cod liver oil including PCBs are declining only at a very slow rate (Kannan *et al.* 1992). Similarly, Loganathan *et al.* (1990) noted that while in most cases local declines resulted from regulation of point sources of PCBs, only slow downward temporal trends were observed in marine mammals from the Western North Pacific. Similar findings have been made for PCBs in animals in the Canadian Arctic (Muir *et al.* 1992). Initial sharp falls in PCB levels have not been maintained and levels have generally only stabilised. Similarly, no clear trend in organochlorine levels could be found in the fat of Arctic fox at Svalbard (Wang-Andersen *et al.* 1993). No downward trend was observed either in PCB levels in the fat of polar bear from Svalbard between 1978 and 1989 (Norheim *et al.* 1992), a phenomenon observed also in Canadian polar bear populations sampled between 1969 and 1984 (Norstrom *et al.* 1988).

Although a spatial, declining, gradient in PCB levels in cod liver oil has been demonstrated for samples originating from the North Atlantic, Norwegian Sea, North Sea and Baltic Sea respectively, temporal trends have been flat or show only a slow decline (Falandysz 1994, Falandysz *et al.* 1994a, b). Even in open ocean cetacean species, PCB levels can be substantial (Simmonds *et al.* 1994). Unsurprisingly, concerns have now been raised about the possible intake of PCBs and other organochlorines by people living in the high Arctic who are dependent upon marine resources.(Kinloch *et al.* 1991; Dewailly *et al.* 1994). This contrasts with marked falls in levels of PCB inputs to terrestrial and freshwater aquatic systems (Sanders *et al.* 1994).

The apparent paradox, that regulation has resulted in very little change in wider marine environmental levels of PCBs is attributable to two factors identified by Ballschmiter *et al.* (1989). Firstly, PCB inputs have not yet halted. Secondly, global equilibrium of these chemicals has not yet been established. Many examples exist of current PCB inputs. A facility on the Ebro River in Spain was recently identified as a point source of these chemicals (Swindlehurst *et al.* 1995) for example. Global equilibrium seems to comprise two major components. The first is release of PCB pollutants form soils and sediments. An important quantitative estimate of PCB losses from UK soils by volatilisation was recently made by Alcock *et al.* (1993). These workers determined levels of PCBs in archived agricultural soil samples and showed that soil PCB levels increased sharply between 1940 and the early 1960s and reached a maximum in the early 1970's. They concluded that if these soils were representative of UK soils in general, then the overall UK soil burden of these pollutants had fallen from c. 26,600 tonnes in 1970 to a current burden of 1500 tonnes. Hence, contaminated soils and sediments can act as a highly important source for the remobilisation of PCBs (see also Panshin & Hites 1994b). Significantly, atmospheric concentrations of PCBs measured at Bermuda in the North Atlantic appear to have remained constant since 1973 (Panshin & Hites 1994a).

The second component involved in global equilibrium of PCBs appears to be a general movement from low to high latitude. A net deposition of tonnes of PCBs per year to Swedish soils has been estimated by Larsson & Okla (1989). In simple terms, low volatile and semi-volatile compounds transported in the atmosphere "condense" out at high latitudes where ambient temperatures are lower. This "global condensation" mechanism has been advanced by Wania & Mackay (1993a) to account for high levels of PCBs found in Arctic regions. With such a mechanism in play the global equilibration process for PCBs has no predictable endpoint. Indeed global cycling was a completely unanticipated aspect of the environmental fate of these chemicals, for which it had previously been assumed that the bulk would eventually adsorb to marine particulates and fall to the deep ocean floor.

ii) The chlorinated dioxins and dibenzofurans

The chlorinated dioxins and dibenzofurans are a group of chemicals which have also attracted an immense amount of scrutiny. Known colloquially and collectively as "the dioxins", they present regulatory problems of a rather different nature to the PCBs and pesticides. The group totals 210 structurally related chemicals which have no application in industry. Despite numerous attempts to find natural sources for these chemicals, they are formed predominantly as unwanted by-products of industry using chlorine chemistry in their processes (Fiedler *et al.* 1990). Table 4 shows identified sources of these compounds, but this is likely to be incomplete.

The work of Czuczwa & Hites (1983) on dated sediment cores from Lake Huron demonstrated that levels of chlorinated dioxins began to increase markedly with the genesis of the chlorine chemical industry in the late 1920s. Contributions from other sources such as coal utilisation were considered very minor: coal consumption had been increasing since 1870 in the region, peaking in around 1920. A similar historical trend is discernible in data from the UK (Kjeller *et al.* 1991) (see: Figure 4) and the Baltic region (Kjeller & Rappe 1995). In the latter case, the rise in background dioxin levels is attributed to the use of dioxin contaminated chlorophenol products.

In many respects the progressive identification of the environmental threat posed by the chlorinated dioxins parallels the evaluation of the PCBs. The identified toxicological effects of dioxin, however, are due to very much smaller quantities of chemical. Hence, in the case of PCBs global budgets are measured in multiples of hundreds of tonnes. In the case of the dioxins, kilogram quantities are highly significant. The entire soil dioxin inventory of the UK is estimated at 5.7 tonnes (Harrad & Jones 1992) accumulated over many decades. This contrasts with the historical figure of 26,000 tonnes for PCBs in UK soils noted above. Control and regulation of emissions and exposures of these chemicals is, therefore, necessary at the nanogram and picogram level.

TABLE 4:

Chlorine gas Production	Brine electrolysis, carbon & titanium electrodes
Use of Chlorine gas	Chlorinated aromatic chemical manufacture Dyes Chlorinated solvents manufacture PVC plastic manufacture Manufacture of epichlorohydrin, chlorobutadiene Manufacture of ferric and copper chlorides Hypochlorite production
Other Uses	Pulp and paper bleaching Water and wastewater disinfection Metal refining and recycling (Cu; Al)
Uses of Organochlorines	Manufacturer of nitrophenols, parathion Degreasing in alkaline or reactive environments Oil refining with organochlorine catalysts Use of pesticides in wood treatment Ferrous sintering with cutting oils, solvents Chlorinated petrol additives Chlorinated domestic and industrial bleaches
Combustion Sources	Medical waste incineration Municipal waste incinerators Hazardous waste incinerators Cement kilns burning hazardous waste

	Fires involving PVC
	Burning of treated wood

Table 4: Known or suspected sources of chlorinated dioxins and related chemicals from industrial and other processes. Based upon information given by Fiedler *et al.* (1990), Cleverly (1993) and Schaum *et al.* (1993).

Despite the extreme toxicological potency of the chlorinated dioxins and dibenzofurans, some workers have considered them to be less of a threat to marine ecosystems than the PCBs and pesticides. This view is predicated upon the assumption that the physico-chemical properties of these chemicals, in particular their low volatility, will tend to stop them entering global atmospheric transport pathways (Tanabe 1988b). Primary evidence that this view may be over-optimistic was provided by the work of Freisen *et al.* (1990) who studied a radio-labelled dioxin congener applied to soil in Canada. After two years, 70% could not be accounted for. The radio-label was not found in degradation products, suggesting that volatilisation of the parent compound was responsible for the observed losses. Model predictions also suggest that dioxins will volatilise appreciably from soils (Eduljee 1987).

The dioxins can also be also be transported in association with airborne particulates. Of all the organochlorines routinely monitored in the Arctic they are the most non-uniform. Highest levels are found in marine mammals from the Canadian Arctic Archipelago (Norstrom & Muir 1994). This is plausibly explained (Norstrom *et al.* 1990) as resulting from deposition from the Arctic haze. This is a particulate aerosol which can become as thick as a city smog. The haze is transported over the Arctic in winter from polluting industrial sources in Europe and Asia. The current distribution of these chemicals implies a greater environmental burden in the Northern hemisphere, that they are ubiquitous environmental pollutants and that they are also in the process of equilibration on a global basis through atmospheric transport mechanisms. Hence, the relative importance of the chlorinated dioxins and dibenzofurans in marine systems could increase markedly in the future.

<u>iii) Other Organochlorine Chemicals</u>

The PCBs and chlorinated dioxins exemplify concerns attached to many organochlorine compounds. They are not the only compounds which now appear to be cycling on a global basis. Table 5 shows estimated world production of various of the persistent organochlorines currently attracting concern for this reason. Toxaphene, for example, is a complex mixture of 600-700 chlorinated bicyclic terpenes (mainly camphenes). It is not used in Western Europe, is banned in thirty seven countries and severely restricted in eleven others. Despite this, levels of 120µg/kg fat weight have been found in mackerel caught off the Shetland Islands (RIVO 1993). 580µg/kg of toxaphene were found in the liver of cod caught in the northern North Sea while hake liver from fish taken from the West of Ireland contained 1300µg.kg on a fat weight basis. A sample of Baltic herring oil contained 7 mg/kg. Unsurprisingly, porpoises sampled from the North Sea were found to contain 6.8 mg/kg in blubber while white beaked dolphin blubber contained 19 mg/kg. (de Boer & Wester (1993).

TABLE 5:

Compound(s)	Date Production Started	Cumulative World Production/use (tonnes)	Current Use (tonnes)
1) DDT	1942	2.8-3 million	50,000
2) Chlordane	1947	70,000	200
3) Heptachlor	1952	no data	no data
4) Toxaphene	1948	1.33 million	none known
5) Technical HCH	1942	550,000	not known
6) Lindane	1942	720,000	10,000
7) Aldrin/Dieldrin	1950	240,000	none known

8) Mirex	1959	not known	none known
9) Hexachlorobenzene	1945	highly uncertain	200,000
10) PCBs	1929	1-2 million tonnes	none known

TABLE 5: Historical and current estimates of production of organochlorine pesticides and chemicals. All figures are subject to wide confidence limits since no production figures are known in many cases for former Eastern Bloc countries. Sources: Voldner & Li (1995); IEM (1995); Smith (1991); Tanabe (1988a).

To date, the best explanation offered in the literature to explain toxaphene distribution is atmospheric transport from Caribbean areas where it is used in cotton growing. It is regarded as more toxic than technical mixtures of PCBs and is extremely difficult to analyse in environmental matrices (Bidleman & Muir 1993). High levels of the pesticide lindane (þ-HCH) are found in the European Arctic and again, long range transport is thought to be responsible (Oehme 1991 a & b).

With other organochlorine pesticides, the global picture is made more complex by the fact that many are being increasingly produced and used in developing countries. In general, atmospheric and seawater concentrations of organochlorines are higher in the Northern than the Southern Hemisphere although the distribution pattern suggests a shift or expansion of major sources towards the South (Iwata *et al.* 1993). Underlying discernible regional distribution patterns, a positive correlation between the ratio of sediment concentrations and those in the overlying water can be found with latitude, indicating the global, northwards transport of these compounds. (Iwata *et al.* 1994). Similarly, levels of organochlorine pesticides found in tree bark have been found to show a positive correlation with latitude in some cases, but not in others (Simonich & Hites 1995). The various differences that exist in the physico-chemical properties of the various chemicals of concern suggest that elucidating global fluxes and transport is likely to prove time consuming and extremely difficult (Wania & Mackay 1993b; Pacyna 1995). Even so, long term programmes set up to investigate this phenomenon are now providing positive evidence of the poleward transport of persistent organic pollutants (Gregor *et al.* 1996; Stern *et al.* 1997). Figure 5 shows a diagrammatic representation of global transport and distribution mechanisms for organic pollutants.

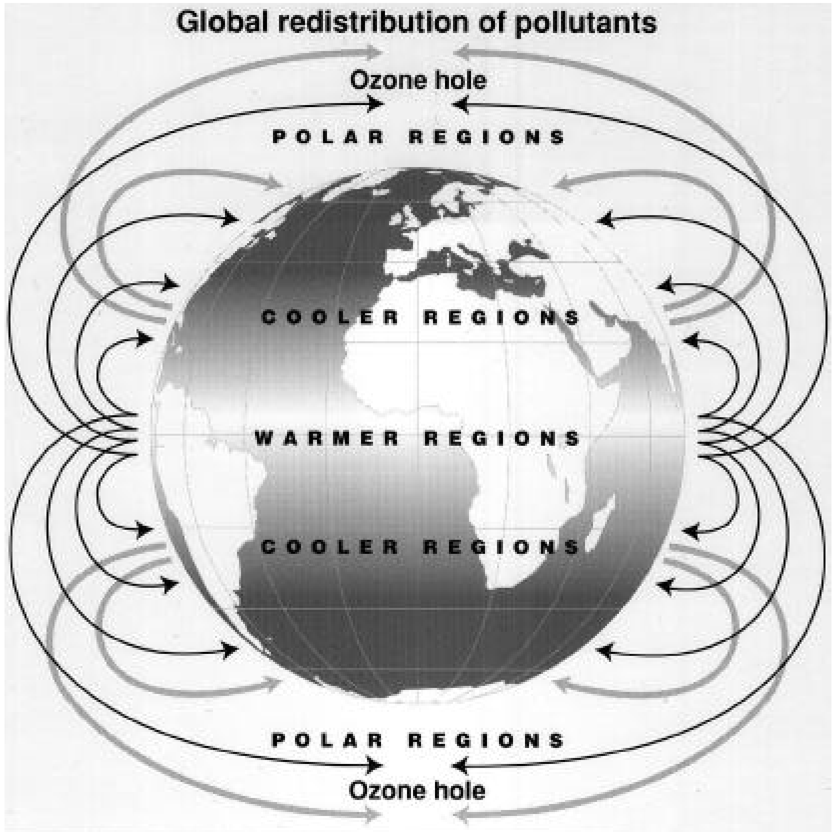


FIGURE 5: Diagrammatic representation of global atmospheric circulation patterns involved in the transport of persistent organic pollutants to high latitudes by means of a "global condensation" mechanism. Chemicals volatilised in areas with high ambient temperatures are transported to cooler regions where they condense and remain in a series of steps (after Wania and MacKay 1993a).

Such problems are likely to grow in scope. As Forget (1991) points out, many developing countries are importing industrial processes that make use of toxic chemicals. Pesticides which are highly restricted in developed countries are being increasingly used in agriculture and in public health programmes to control pests and vector borne diseases. Indeed, this premise is borne out by the results of an extensive global sampling programme. This showed that high residue levels of HCH and DDT were present in the atmosphere and waters of tropical Asia. The residue distribution also suggested that chlordane and PCB emissions are increasing in tropical countries (Iwata *et al.*1993; Tanabe et al. 1994). Falandysz *et al.* (1994b) note, probably correctly, that pollution by organochlorines in remote areas can be controlled only by a complete ban on their use throughout the world. Transport from areas of use to remote regions appears to be inevitable and that high levels of some of these pollutants will persist for many decades in high latitude regions even when regulatory measures are taken (Harner 1997). This point of view is becoming widely accepted (Wania & Mackay 1996) and the properties of chemicals which make them liable to long range transport are being progressively characterised. It is important that these substances are not simply replaced with other chemicals exhibiting the same undesirable properties.

Problems are not confined to the "conventional" organochlorines listed in Table 5. Thirty one halogenated substances (30 of them chlorinated) have been designated as monitoring target compounds in Sweden alone (Jansson *et al.* 1993). Moreover, new classes of organochlorine contaminants has been discovered some of which appear to be globally distributed (Jarman *et al.* 1992). Tris(4-chlorophenyl) methanol (TCP-methanol) and tris(chlorophenyl) methane (TCP-methane) have been found in a range of biological samples from widely separated geographical regions. 41 µg/kg of TCP-methanol was isolated from the blubber of St Lawrence Estuary beluga whales, for example (Walker *et al.* 1989).

Overall, substantial global releases are implied by isolation of these compounds from Antarctic biota yet the source(s) remain unidentified. Source reconciliation has also proven problematic for another persistent pollutant recently isolated from Baltic ecosystems. Bis (chlorophenyl) sulfone (BCPS) has been found in the tissues of fish, marine mammals and birds (Olsson & Bergman 1995). The chemical is used in the manufacture of high temperature polymers, as a component in reactive dyes and is a contaminant in some pesticides. In addition, brominated chemicals are causing increasing concerns as they are isolated from marine organisms (see: Haglund *et al.* 1997).

In the UK, a previously undescribed group of methylated hexachlorocyclohexane (HCH) isomers were detected in mussels from the Mersey estuary (McNeish *et al.* 1994). The biological and toxicological significance of these chemicals is unknown but of concern due to their bioaccumulative potential and structural resemblance to the pesticide lindane. Other pollutants of concern include the chlorinated paraffins which are neurotoxic in fish and also bioaccumulate (Svanberg *et al.* 1978). These compounds are retained in test organisms in an altered chemical form (Madely & Birtley 1980) making analysis very difficult. Similarly, the metabolism of PCBs to PCB methyl-sulfones, which are more persistent than the parent chemicals, also results in considerable analytical and toxicological difficulties (Bergman *et al.* 1992). The ultimate analytical difficulties occur in samples which have received complex effluents over a long period. In the Baltic, where pulp and paper bleachery effluents have been discharged, considerable numbers of exotic organochlorine chemicals are found. Of the total organically bound chlorine which can be extracted from fish, however, only 10-15% can be attributed to known, identified pollutants (Wesen *et al.* 1990).

d) Polycyclic aromatic hydrocarbons

Polycyclic aromatic hydrocarbons (PAHs) are chemicals which are produced by combustion processes. The quantities released to the environment have increased enormously in the environment with the development of industry and concomitant fossil fuel consumption (see: Figure 4). These sources have overwhelmed natural inputs such as bush and forest fires and have also changed the relative proportions of these chemicals entering the environment (Jones *et al.* 1989). PAHs or their nitrogen derivatives are commonly found downwind of stationary combustion sources such as power stations and are also emitted in substantial amounts from coke ovens and gasification plants (Sloss & Smith 1993). In addition, the burning of petroleum products, including automotive fuels also emit PAHs, largely associated with particulates. These are of increasing concern in public health terms (see Reichardt 1995) but the contribution of such particulates to marine fluxes of toxic chemicals is largely unknown.

Early research (Laflamme & Hites 1978) established that PAHs and their alkyl homologues are distributed in sediments throughout the world. In qualitative terms, the consistency of the homologue pattern and the fact that total PAH sediment levels tend to increase with increasing proximity to urban centres indicates that anthropogenic combustion processes are the dominant source. Benzo[a]pyrene (B[a]P) is an important PAH since it can be metabolised to form reactive intermediates through several biochemical pathways. One of these intermediates, the metabolite 7,8-*trans*-dihydrodiol-9,10,-epoxide, is known to be capable of binding directly with the molecules of the genetic material DNA to form a DNA adduct an extremely potent carcinogen (Hansen & Shane 1994). Some of the nitrogen substituted PAHs appear to exhibit direct mutagenic or carcinogenic activity, without the need to undergo metabolic changes (Josephson 1984). A variety of PAHs have been shown to effect fish, particularly those fish in contact with contaminated sediments. Effects range from the induction of liver enzymes and carcinogenic liver disease to interference with the reproductive system.

e) Mechanisms of Action of Organic Pollutants

The variety of mechanisms of the biochemical pathways potentially involved in metabolising organic chemical pollutants are described by Hansen & Shane (1994). These include a wide array of enzyme mediated reactions. Phase I reactions introduce a reactive function to the molecule in question through hydrolysis or oxidation. This makes it more polar and more water soluble. Phase II reactions involve the Phase I product in conjugation with an endogenous substance which then renders the xenobiotic less bioactive and more readily excreted. In some cases, however, the Phase II reaction actually results in a product with increased toxicity. This is particularly true of reactions with the alkyl or aryl halides and it is becoming clear that such reactions are not as uncommon as was once supposed.

Many of the pollutants of concern have the ability to bind to the cytosolic arylhydrocarbon (Ah) receptor thereby inducing the increased activity of oxidising enzymes. This receptor has been identified in a number of animal groups, including the cetaceans. and underpins a number of critical biochemical pathways. In particular, it is involved in hormone metabolism (see: Goldstein & Safe 1989; Seegal *et al.* 1991). The Ah receptor is a common factor governing the toxicological behaviour of a number of aromatic compounds including the PAHs, certain PCBs and chlorinated dioxins. On the basis of chemical structure, Giesy *et al.* (1994) list 22 groups of chemicals (mostly halogenated) whose toxicological action is possibly mediated through the Ah receptor system and which are widely distributed in the environment (Table 6).

TABLE 6:

Polycyclic aromatic hydrocarbons (PAH)	Polychlorinated fluorenes
Polychlorinated biphenyls (PCBs)	Polychlorinated dihydroanthracenes
Polychlorinated dibenzo-p-dioxins (PCDDs)	Polychlorinated diphenylmethanes
Polychlorinated dibenzofurans (PCDFS)	Polychlorinated phenylxylyl ethanes
Polychlorinated naphthalenes	Polychlorinated dibenzothiophenes
Polychlorinated diphenyltoluenes	Polychlorinated quaterphenyls
Polychlorinated diphenylethers	Polychlorinated quaterphenyl ethers
Polychlorinated anisoles	Polychlorinated biphenylenes
Polychlorinated phenoxy anisoles	Polychlorinated thioanthrenes
Polychlorinated xanthenes	Polybrominated diphenylethers
Polychlorinated xanthones	Polychlorinated anthracenes
Polychlorinated azoanthracenes	*Polybrominated biphenyls
*Polybrominated dibenzo-p-dioxins	
*Polybrominated dibenzofurans	

Table 6: List of compounds that may have the potential to cause adverse effects through interaction with the Ah receptor inferred from experimental evidence or the molecular structure. From: Geisy *et al.* (1994). * denotes widespread group of chemicals with appropriate structure not included in the original list. The status of fluorinated analogues is largely unclear.

In this sense, not only are the PCBs, pesticides and dioxins acting as toxicants in their own right, but also as an index of environmental pollution by similar compounds which may be less readily amenable to instrumental analysis or source reconciliation. Many of the chemicals shown in Table 6 are in fact chemical groups comprised of large numbers of closely related compounds, echoing the concerns raised above in relation to the complexity of chemical discharges. A key concern relating to chemicals whose effects are mediated through the Ah receptor is their ability to interfere with hormonal pathways. The degree to which the significance of this toxicological aspect may have been underestimated is indicated by Colborn & Clement (1992) and Colborn (1994). Hormonal systems are at the centre of the biochemical processes underpinning the functionality and maintenance of organisms in general and, more specifically, their reproductive processes.

3.4 NUTRIENT POLLUTION

a) Introduction

Nutrients, including nitrogen, phosphorus and a range of inorganic and organic micronutrients, are essential for primary production. In any one region of the marine environment, nutrients may be supplied to the phytoplankton and/or phytobenthos through tight coupling of production, consumption and decay processes or by influx of "new" nutrients through upwelling, advection from other areas, inputs from rivers and other land-based surface sources and, for some nutrients at least, through atmospheric deposition. Surface primary production in all but the most turbid or nutrient-rich areas is ultimately controlled, or limited, by the availability of one or more nutrients. Primary productivity in coastal and offshore marine environments is generally assumed to be nitrogen-limited (because of the high denitrification capacity of estuaries and shallow coastal waters) or, more rarely, phosphorus-limited (Rhyther and Dunstan 1971, Billen and Garnier 1997). Nevertheless, other less abundant ions and substances may play a significant role in some circumstances, either as micronutrients in themselves or by controlling the availability of other essential growth factors (Stal *et al.* 1996).

Given their proximity to land-based nutrient sources, coastal waters tend to be richer in nutrients, and support higher levels of production at all levels of the community, than waters further offshore (with the exception of those offshore areas fed by upwelling systems). Indeed, although coastal waters account for only 10-15% of total sea area, they support approximately 50% of global marine primary production (Paerl 1997). Fluctuations in nutrient supply, resulting from upwelling events, variations in river discharge, run-off and precipitation, coupled with the complex physical variability of marine systems, can result in strong fluctuations in primary productivity on a short-term, seasonal and interannual basis. Furthermore, consumption by grazers and the overall complexity and plasticity of food web dynamics, can further complicate natural cycles of phytoplankton biomass and production by exerting “top-down” control (Fogg and Thake 1987).

Superimposed on this highly complex and unpredictable system are the contributions of human activities to the cycling of nutrients and, therefore, to their relative availability in the marine environment (Howarth *et al.* 1996). As noted by Ministers at the Fourth International Conference on the Protection of the North Sea (MINDEC 1995), while nutrients are essential for marine ecosystem function, problems arise “when inputs from land become excessive and/or the ratio between nutrients is substantially changed”. Whereas exploitation (fisheries) and environmental degradation (pollution, ecosystem damage) may impact the plankton primarily through changes in “top-down” pressure, enhancement of nutrient supply can force “bottom-up” changes by stimulating primary production and/or modifying plankton community structure. Due to their proximity to principal sources of anthropogenic nutrient loading, both point and diffuse, coastal waters are generally more severely impacted by enhanced nutrient inputs than offshore waters (McClelland *et al.* 1997).

Nevertheless, as discussed below, the effects of such nutrient enhancement rarely result in simple, proportional increases in system productivity and biomass. Instead, changes in nutrient balance and overall nutrient supply can result in changes in community structure which are frequently complex and generally poorly understood. Such changes may or may not result in nutrient enrichment being translated to higher productivity at higher trophic levels.

b) Contribution of anthropogenic activities to marine nutrient budgets

There is little doubt that anthropogenic nutrient inputs now contribute substantially to total nutrient fluxes. GESAMP (1990) concluded that "globally, present inputs of nutrients from rivers due to mans activities are at least as great as those from natural processes" and have led to "clearly detectable and sustained increases in nutrient concentrations in the [receiving] water". Global estimates of anthropogenic contributions to river-borne nutrients vary by almost an order of magnitude, from 7 to 35 x 109 kg/y for nitrogen and from 0.6 to 3.75 x 109 kg/y for phosphorus, but there is little disagreement that these inputs represent significant enhancement of natural loadings. As part of a review of nutrient inputs from 14 key basins discharging to the North Atlantic, Howarth *et al.* (1996) estimated that riverine nitrogen (N) flux has increased by 2-20 fold above background in urban, agricultural and industrialised catchments. The North Sea was recognised as one of the most disturbed sea areas in relation to nutrient enrichment. Inputs of nitrogen are generally strongly correlated with river flow (Provini *et al.* 1992), while phosphate inputs increase with sediment load (Barbanti et al. 1992).

Nutrient inputs and budgets are notoriously difficult to calculate, given the diversity of sources and pathways and the fact that natural inputs are substantial (GESAMP 1990). Gerlach (1988) estimated that anthropogenic inputs accounted for roughly 32% of N and P loads in riverine and direct discharges to the southern North Sea, with the Rivers Rhine and Meuse representing the greatest point sources.

An earlier budget presented by Lee (1980), suggesting that riverine and direct discharge inputs of nitrogen and phosphorus were small compared to loads entering the Northern North Sea from the North Atlantic, failed to account for the fact that approximately 80% of the land-based inputs discharge within an area south of 54.5 N. On a more local scale anthropogenic influences may be even greater than suggested by Gerlach's (1988) calculations, accounting for 70% and 80% respectively of the N and P loads to the inner German Bight and the Jutland Current (Nelissen and Steffels 1988).

Estimating anthropogenic influences on offshore nutrient budgets is much more difficult as there is a lack of reliable data on which to base nutrient budgets, particularly in relation to atmospheric processes (Stromberg 1997). As 80-90% of the nutrient loads from land-based sources are assimilated in estuaries and near-shore systems (GESAMP 1990), the major anthropogenic influences will be exerted in these regions. For example, only 10% of the nutrient load to the Baltic from riverine and direct discharges leaves the basin unassimilated (Nehring 1992).

c) Sources

Demographic changes, particularly the spread of coastal urbanisation, increases in emissions from transport and power generation and changes in agricultural and aquacultural practices are primarily responsible for increases in direct, diffuse and atmospheric inputs. Agricultural sources (artificial fertiliser application, intensive animal hubandry, etc.) dominate enhanced nitrogen inputs in many areas, particularly in the Northern Gulf of Mexico and North Sea (Lancelot 1995, Howarth *et al.* 1996), although municipal wastewater discharges clerly also account for substantial inputs in these regions. Forestry may be significant in some catchments, particularly in Northern USA and in Scandinavian waters (Rosenberg *et al.* 1990, Howarth *et al.* 1996). For nitrogen, deposition of atmospheric nitrogen oxides, derived primarily from industrial and transport sources, can also be a major contributor; stationary sources of nitrogen oxides probably dominate elevated nitrogen inputs to the coastal waters of the North Eastern USA, for example (Jaworski *et al.* 1997). Inputs of other nitrogen compounds from manufacturing and processing industries may have local significance but probably do not contribute greatly to the overall nutrient load.

The growth of human coastal populations has generally overtaken the development and installation of adequate sewage treatment systems. Of 367.2 million population equivalents of sewage discharged to the North Sea in the period 1990-92, 12% received no treatment and a further 6% only primary treatment (OSPAR 1995a). Thus the equivalent of sewage from 67.5 million people was discharged without nutrient reduction to the North Sea. Lidgate (1988) estimated that 25% of the P load carried by the Rhine originated from domestic sewage.

In a survey of 42 river basins worldwide, Peierls *et al.* (1991) noted that variations in human population density accounted for an average of 53% of observed variations in total nitrate exported to coastal waters. Similarly, Lancelot (1995) highlights the primary importance of discharges from the heavily populated, cultivated and industrialised catchments of the major rivers of continental Europe in determining nutrient budgets of the Southern North Sea.

The introduction of tertiary treatment in the Baltic catchment reduced phosphorus loading by 50% between 1969 and 1977, but nitrogen loading continued to increase over this time (Pawlak 1980). Indeed, nitrogen loading of the southern Baltic Sea from sewage treatment plants continued to increase throughout the 1980s (Rosenberg *et al.* 1990). Furthermore, in some North Sea areas (eg. Rhine discharge), the introduction of sewage treatment has exacerbated the problem of nitrogen enrichment; Billen (1990) suggested that this was due to a reduction in potential for denitrification as organic matter was more effectively removed. The capacity for denitrification in surface coastal waters is naturally fairly low (Rosenberg *et al.* 1990) and may well have already been exceeded over substantial areas (Rosenberg *et al.* 1990).

Although the greatest mass inputs of nutrients associated with sewage and municipal waste water discharges undoubtedly arise from large urban centres, inputs from small communities can be very significant on a local scale, especially in isolated or otherwise sensitive coastal areas. In developed countries, the level of sewage treatment and nutrient removal is often poorer for the large number of small communities than for larger centres of population. Small point sources of nutrient discharge (e.g. septic tank overflows and seepage) can be locally significant, as demonstrated by Lapointe and Matzie (1996) in shallow waters adjacent to small communities on the Florida Keys.

It is estimated that 60% of the anthropogenic N inputs to the North Sea, and 25% of P, are of agricultural origin (de Jong and de Oude 1988) . Figures of a similar order have been suggested for coastal waters in the Mediterranean and USA. The move towards widespread application of inorganic fertilizers, often poorly targeted, has greatly increased the mobility of both N and P through soils and in surface and groundwaters (OSPAR 1995a). Nevertheless, the importance of specific sources vary with land use. In the Po basin, for example, discharging to the Northern Adriatic, agricultural loadings are dominated by direct inputs from intensive animal units or as run-off from slurry spreading grounds, with a relatively small contribution from arable land (Rossi *et al.* 1992). Intensive animal rearing has led to problems of disposal of vast quantities of nutrient-rich slurry waste, especially as the numbers of pigs, poultry and other animals intensively farmed worldwide have increased so dramatically over the last 20-30 years (Nixon 1995).

In some regions, the explosive growth of intensive aquaculture has accounted for substantial increases in the nutrient loading of coastal systems. In Thailand, for example, it has been estimated that nitrogen and phosphorus discharges from intensive shrimp aquaculture alone may be equivalent to an increase in coastal human population by between 50 and 100% with no provision for sewage treatment (Dierberg and Kiattisimkul 1996). More subtle, but no less significant, impacts on nitrogen supply have been reported in the vicinity of shellfish farms in coastal lagoons (Gilbert *et al.* 1997) and may be associated with cage culture of salmonids in temperate waters (Folke *et al.* 1994). The significance of aquaculture operations as point sources of nutrient pollution is addressed in more detail in Section 5.3).

The contribution of human activities to atmospheric sources may be significant for nitrogen (in the form of nitrogen oxides from vehicles and power generation plants (Howarth *et al.* 1996, Paerl 1997) but of lesser importance for phosphorus. Atmospheric inputs of nitrogen to the North Sea as a whole may be of the same order as riverine inputs (OSPAR 1993). Although atmospheric deposition accounts for more widely spread inputs to coastal than river discharges themselves, Paerl (1997) estimated that, for nitrogen, such deposition coupled with diffuse inputs via groundwater, may account for between 20-50% of enhancement of loading to the coastal fringe. North Sea Ministers also recognised the significance of such diffuse inputs, while recognising the substantial difficulties inherent in estimating loadings and controlling source emissions (MINDEC 1995).

Precise source reconciliation for defined marine areas is likely to remain extremely difficult, not least because of the problems in obtaining reliable estimates of diffuse inputs (GESAMP 1990, McClelland *et al.* 1997). For this reason, legislation designed to control anthropogenic enhancement of nutrient loads has tended towards rather arbitrary goals for input reduction, while recognising a number of key sectors which need to be targeted for reduction of source emissions. The five sectors prioritised by OSPAR, namely agriculture, horticulture and forestry, municipal wastewater, industry and fossil fuel combustion (OSPAR 1995b) contribute the bulk of enhanced nutrient inputs to the North Sea and, undoubtedly, other sea areas; these measures are discussed further later. New techniques for identification of sources, employing, for example, stable isotope ratios (McClelland *et al.* 1997) may ultimately assist source reconciliation but, at present, such techniques remain under development.

d) Consequences of enhanced nutrient loading

i) Changes in ambient nutrient concentrations and availability

Increased inputs of nitrogen and phosphorus from rivers and direct sewage and industrial discharges may be negligible in terms of global oceanic nutrient budgets but, when entrained in estuaries, bays and shallow seas, can represent substantial additions to natural budgets (van Bennekom *et al.* 1975; Justic et al. 1995; Taylor 1987; Tett and Mills 1991). Impact is enhanced further by the poor mixing of nutrient-rich coastal waters with offshore waters, and accompanying hydrological stability, which generally results from saline stratification. Van Bennekom *et al.* (1975) calculated that the estimated 450,000 tonnes of nitrogen y^{-1} introduced via the Rhine, Meuse and Scheldt rivers could raise the winter nitrate concentrations of the Southern Bight of the North Sea from 7 to 14 uM, assuming effective mixing. In practice, concentrations of up to 60 uM have been recorded (Tett and Mills 1991). Although inputs from these sources have increased (estimated at 700000 tonnes y^{-1} in 1991), the complexity of mixing regimes has undoubtedly contributed.). Substantial increases in winter nitrate concentrations have also been reported for the Fladen Ground (North Sea) and the Baltic between the 1970s and late 1980s (Rosenberg et al. 1990). Similarly, although for the Kattegat as a whole land-based inputs supply only 30% and 10% of the total N and P budget respectively, in coastal regions such as Laholm Bay on the West Coast of Sweden as much as 90% may be of riverine origin (Rosenberg *et al.* 1990).

The influence of major rivers is also clear in other sea areas. The River Po delivers roughly 50% of the annual N and P loads to the Northern Adriatic (Vukadin 1992), although the importance of a large number of smaller rivers must not be overlooked (Marchetti and Verna 1992). Inputs of nitrogen from the River Po have more than doubled over the period 1968-1988 (Marchetti *et al.* 1989). Inputs of P showed similar, if not greater, increases over the same period (Justic *et al.* 1995), although restrictions under Italian law of the P content of detergents slowed the trend in the early 80s (Provini *et al.* 1992).

Justic *et al.* (1995) highlighted the importance of the Po discharge to the overall nutrient budget of the Northern Adriatic, and similarly of the Mississippi in determining the budget of the Northern Gulf of Mexico. Mean surface nutrient concentrations in the Northern Adriatic increased from 0.95 to 2.42 uM for N and 0.018 to 0.070 uM for P between the late 60s and early 80s. In the Gulf of Mexico, surface nitrogen increased from 2.23 to 8.13 uM and phosphorus from 0.140 to 0.340 uM over roughly the same period (Justic *et al.* 1995).

In addition to increasing the overall nutrient load to marine systems, enhanced inputs of specific nutrients can effectively shift the ratios of nutrients available for primary production (Sommer 1994, Justic *et al.* 1995, Nelson and Dortch 1996). Such shifts in ratio may be as, or even more, significant than the absolute concentrations of individual nutrients as factors controlling the biological response to enhanced nutrient inputs (Lancelot 1995, Hodgkiss and Ho 1997). Nelson and Dortch (1996) note that silicon limitation of phytoplankton growth, observed in summer in the Northern Gulf of Mexico, may be a relatively new phenomenon, resulting, in large part, to progressive increases in the load of nitrate delivered by the Mississippi. The significance of changes to nutrient availability ratios was also noted in the Declaration of the Fourth North Sea Ministerial Conference (MINDEC 1995).

Although the immediate and most visible impacts of increased nutrient loading may be confined to coastal waters, significant increases in winter nutrient concentrations may be detectable further offshore if enhanced riverine inputs are maintained over time (Hickel et al. 1993) or in regions where atmospheric sources are locally substantial (Jaworski *et al.* 1997). Long-term increases in nitrogen availability offshore have generally been observed to be more rapid than for phosphorus, probably in part because phosphorus is generally more tightly bound to sediment particles (Barbanti *et al.* 1992). The binding and release of phosphorus from sediments, a complex redox driven process (Golterman 1995), undoubtedly also accounts for the slower decline in enhanced phosphorus concentrations in coastal and offshore marine systems following the introduction of source control measures (OSPAR 1993). Such mechanisms could help explain anomalies such as the continued high levels of algal growth recorded in the Western Dutch Wadden Sea (de Jonge 1997), despite the introduction of source controls for phosphorus early in the 1980s.

ii) "Cultural eutrophication"

The enhanced addition of plant nutrients to the marine environment from anthropogenic sources frequently has the effect of increasing primary production (e.g. increased algal growth), a process commonly termed "cultural eutrophication". In very shallow waters, this may be manifest as an increase in the biomass of bottom dwelling seaweeds. Further offshore, however, the elevated supply of nutrients may be assimilated primarily by phytoplankton leading to significant increases in growth rate and abundance in algal populations. Where hydrographic conditions allow, such increases in algal biomass can be very rapid, greatly exceeding grazing rates by zooplankton and resulting in a visible “algal bloom”.

Nevertheless, as discussed below, eutrophication may not always be manifest as an increase in biomass of primary producers, especially if increased production is passed on effectively to higher trophic levels. Eutrophication may, therefore, be better defined in more general terms, such as “an increase in the rate of supply of organic matter to a system”, as suggested by Nixon (1995). Indeed, as noted by Nixon (1995), although widely used, the term eutrophication remains poorly defined, with a number of different interpretations in common parlance. That which is currently used within OSPAR (1995c), refers specifically to increased production of phytoplankton, with general mention on nitrogen and phosphorus, but should not be misinterpreted as relating only to situations in which an increase in algal standing stock biomass is recorded.

Notwithstanding this, the most visible and, consequently, most frequently documented impacts of eutrophication are substantial increases in the biomass of benthic or planktonic algae. These are often accompanied by shifts in the structure of the planktonic community, an increase in the quantity of biomass settling out of the water column, elevation of heterotrophic activity and a consequent decline in oxygen saturation of the water column, particularly near the sediment-water interface.

Algal blooms can occur as a natural phenomenon in both coastal and offshore waters. Nevertheless, numerous studies and reviews have noted an increase in both the frequency and persistence of algal blooms in coastal waters and enclosed sea areas over the past 20-30 years, including the Baltic, Southern North Sea, Black Sea, Adriatic, Mediterranean and the coastal waters of North and South America and Japan (Kerr and Ryder 1993; Sarokin and Schulkin 1992; Vollenweider 1993, McClelland *et al.* 1997).

While some of the most acute effects may be apparent in enclosed harbours and embayments downstream from large urban centres (Abdelmoati 1996, Diaz Zaballa and Gaudy 1996), increases in algal growth are by no means restricted to such localised systems. Eutrophication of substantial areas of the Northern Adriatic, supported largely by inputs from the River Po, has been relatively well documented (Barbanti *et al.* 1993, Rinaldi *et al.* 1995, Solic *et al.* 1997). Lancelot (1995) reviewed the evidence for eutrophication as a factor supporting the more frequent and persistent appearance of substantial blooms of the mucilaginous flagellate *Phaeocystis* spp. in the coastal regions of the Southern North Sea. Rosenberg *et al.* (1990) stress the significance of land-based sources of nutrients in the enrichment of the Baltic and Skagerrak, while Justic *et al.* (1995) and Nelson and Dortch (1996) highlight similar concerns in the Gulf of Mexico. Norkko and Bonsdorff (1996) describe a marked increase in the incidence of dense mats of drifting benthic algae in the Baltic, correlating with the enhancement of nutrient loading over the past few decades. Mean and peak phytoplankton biomasses have also increased substantially over a similar period (Bonsdorff *et al.* 1997), greatly reducing water transparency in some areas. Increases in the abundance of attached benthic marine macroalgae (seaweeds) have also been recorded in some shallow waters receiving enhanced nutrient inputs (Morand and Briand 1996, McClelland *et al.* 1997, Stromberg 1997).

The lack of baseline data for algal productivity and biomass in all but a few marine areas presents substantial limitations for the quantitative study of long-term trends. Although some authors have used models validated on current nutrient : productivity relationships to hindcast historic trends in production (e.g. Peeters *et al.* 1995 for the Oyster Ground, Southern North Sea), output is limited in utility by the levels of uncertainties inherent in such models. The data set presented by Harding and Perry (1997) for Chesapeake Bay represents one of very few long-term studies of phytoplankton community structure and dynamics(1950-1994), but provides clear evidence for a substantial and consistent increase in algal biomass over time, possibly as a result of enhanced nutrient loading.

iii) Impacts of eutrophication

The combination of factors which trigger algal blooms is complex and remains poorly understood (Rinaldi *et al.* 1995), but an elevated supply of nutrients appears to be a common contributing factor, as incidences of high phytoplankton productivity in coastal waters generally correlate well with changes in river discharge and nutrient loading (Cadee 1992 a and b; Marchetti 1993). In contrast, the biological progression of algal blooms and their subsequent impacts on the marine environment have been relatively well documented. One of the most severe effects following the peak and subsequent decay of dense algal growth is a decrease in oxygen concentration in the water column leading, in many incidences, to localised or more widespread anoxia. This in turn can have devastating impacts on the benthic community.

a) Anoxia

Anoxia can result both as a result of smothering of the benthos with dense algal mats or decaying material (Rinaldi *et al.* 1995, Norkko and Bonsdorff 1996) and as consequence of an increase in the activity of heterotrophic bacterial communities which are associated with the decaying bloom (Sorokin *et al.* 1996, Dubinsky and Stambler 1996). Where enrichment is moderate, an increase in the productivity of benthic communities may be observed (e.g. Skold and Gunnarson 1996), but massive blooms followed by severe anoxia appear to be increasingly common in many coastal systems.

Sorokin *et al.* (1996) describe a severe anoxic event in Ca Pisani (Veneto, North Italy) which followed an extensive blooms of the dinoflagellate *Alexandrium tamarense* in the vicinity of intensive fish farming operations. Similarly, Abdelmoati (1996) reported a massive fish kill and die-off of benthic organisms resulting from decay of a bloom in Alexandria harbour. Such impacts are not limited to blooms and decay of plankton organisms; to some degree the greatest problems experienced in the coastal lagoons and shallow waters of the Northern Adriatic result from massive production of benthic macroalgae (Bartoli *et al.* 1996).

Moreover, eutrophication-enhanced anoxia is not restricted to small shallow embayments, but can also impact very large sea areas; as much as one third of the sea floor of the Baltic is now subject to permanent hypoxia, with additional areas suffering low oxygen tension on a seasonal basis (Rosenberg *et al.* 1990). The Northern Gulf of Mexico, already the site of the largest hypoxic zone in the Western Atlantic (Justic *et al.* 1996), could face anoxia over an increasingly wide area if current nutrient inputs are maintained (Lohrenz *et al.* 1997).

Johannessen and Dahl (1996) present data on long-term trends in the extent and severity of hypoxia along the Norwegian coast of the Skagerrak. Seasonally adjusted oxygen concentrations have declined at all depths over the past 30-40 years, with declines in surface waters beginning in the 1960s. Development of chronic hypoxia in deeper waters appeared to be more delayed, dependent on longer-term increases in inputs from surface production, but has subsequently been more severe. Indeed, Aure *et al.* (1996) notes that the naturally lower oxygen levels in the sill basins along the Skagerrak coast, coupled with their limited water exchange, renders these systems especially sensitive to the consequences of eutrophication.

As Rosenberg *et al.* (1990) stress, such hypoxic or anoxic events can have severe impacts on benthic community structure. Norkko and Bonsdorff (1996) reported rapid and widespread loss of key benthic species, including the bivalve *Macoma balthica* as a result of smothering and anoxia caused by drifting algal mats in the northern Baltic. Oxygen depletion may also initiate complex positive feedback mechanisms which could further exacerbate the problem. Anoxia at the sediment - water interface can enhance the rate at which phosphorus is reinjected to the water column from the sediment. In addition, a decrease in the surface area of bioturbated sediment, resulting from die-off of benthic fauna, can limit rates of both nitrification and denitrification, potentially increasing availability of nitrogen for further algal growth over the long-term (Rosenberg *et al.* 1990).

b) Shifts in phytoplankton composition

In addition to overall increases in biomass, enhancement of nutrient loading can also lead to an shift in relative abundance of taxa at all levels of the food web, although effects are best described for the phytoplankton and benthic communities. Much evidence for the impact of enhanced nutrient loading and/or shifts in relative nutrient abundance on primary production and phytoplankton relative abundance is correlative in nature, supported to some extent by manipulative studies in mesocosms. Nevertheless, in many instances the evidence suggests consistent trends.

For example, progressive enrichment of nitrogen and phosphorus over silicon (Smayda 1990, Justic *et al.* 1995) generally forces a shift from diatom to flagellate dominance in enriched coastal waters (Lancelot *et al.* 1987; Officer and Ryther 1980; OSPAR 1993; Tett and Mills 1991). Flagellates can reach very high abundances and may dominate the phytoplankton numerically by several orders of magnitude. The general trend away from diatoms towards flagellates in coastal waters (Smayda 1990) has been linked to the increasing incidence of silicon-limitation resulting from anthropogenic inputs of nitrogen and phosphorus. According to Justic *et al.* (1995), silicon-limitation may already prevail in the Northern Adriatic and in the Northern Gulf of Mexico as a result of enhanced loading N and P from anthropogenic sources.

Blooms of the prymnesiophyte flagellate *Phaeocystis pouchettii* have undoubtedly occurred periodically in coastal waters of the southern North Sea over many hundreds of years. Nevertheless, substantial evidence exists that both the duration and density of such blooms have increased in recent years, possibly in response to the increased loading of nutrients, primarily nitrate, via the major European rivers (Billen 1990, Lancelot 1995). This is supported by the ability of *Phaeocystis* sp. to outcompete other phytoplanktonic taxa for nitrate in mesocosm experiments (Lancelot 1995), and the increased prominence of this organism in other coastal waters undergoing similar eutrophication processes (Al-Hasan *et al.* 1990). Other organisms may be outcompeted for essential nutrients. For example, Herman *et al.*(1996) suggest that the decline in sea grass communities in the Wadden Sea may, to some extent, result from decreased availability of silicon due to increased scavenging by diatoms responding to enhanced loading of nitrogen and phosphorus to nearshore waters.

A shift in the relative abundances of key diatom species appears to be accompanying the progressive shift towards annual silicon-limitation of production in the Gulf of Mexico (Nelson and Dortch 1996). Evidence that nutrient enrichment and changes in nutrient ratios may also trigger blooms of other phytoplanktonic organisms is more circumstantial, although Hodgkiss and Ho (1997) report the appearance of consistent relationships between relative nutrient availability and the abundance of red-tide dinoflagellates in Japanese, Chinese and some North European waters.

Changes in the phytoplankton community structure can have impacts, directly or indirectly, on the consumers (e.g. the grazing zooplankton) and higher trophic levels, although it is important to note that increased primary productivity is not necessarily passed to higher components of the food web. A shift in species composition towards small taxa capable of high growth rates, for example, may result in much of the primary production biomass becoming unavailable for grazing by copepods and other crustacean zooplankton. In turn, this may lead to changes in availability of food for fish, marine mammals and other higher organisms. Thus, an increase in primary productivity does not necessarily result in the transfer of increased productivity to higher trophic levels, at least not in the pelagic phase (Ingrid *et al.* 1996). Indeed, much of the biomass fixed in large algal blooms may be lost to the benthos or be degraded via the planktonic microbial loop.

This is especially true in the cases of the mucilaginous algae, including *Phaeocystis* sp. and the mucilage producing diatoms *Chaetoceros* spp. and *Skeletonema costatum*, the latter being particularly prominent in the Northern Adriatic. The mucilaginous colonies produced under certain physiological conditions (still incompletely understood) by *Phaeocystis pouchettii* are too large to be available to most of the grazing zooplankton. In addition, the mucilage is highly resistant to bacterial decay in the pelagic zone (Lancelot 1995), appearing to exert an antimicrobial effect. As a result, particularly during spells of bad weather, vast quantities of mucilage can accumulate as a foam on the beaches of continental Europe. Since the mucilage can represent up to 90% of the total carbon biomass of the colony at the height of a bloom, this represents a substantial loss of production which is not available to higher organisms. Whether the colonies remain and eventually undergo microbial decay in the water column or at the benthos, or accumulate on the beaches, little of the fixed carbon is translated to increased pelagic production at higher trophic levels. Mass settling of decaying matter represents a “pump” delivering carbon and nutrients acquired at the surface to the benthic zone, with the potential to over-enrich the system with organic matter (Ingrid *et al.* 1996). The food web is, essentially, disrupted as a result of the dominance of the phytoplankton by *Phaeocystis* sp. and its physiological shift from motile single cells to non-motile, mucilaginous colonies (Lancelot 1995). Although eutrophication probably contributes to this process, the underlying physiological mechanisms remain largely unexplained.

The production of mucilaginous blooms has also been documented in the Northern Adriatic Sea, although again the mechanisms are unclear. While total phytoplankton production in this region is strongly correlated to enhanced nutrient loading, the development of mucilaginous events does not appear to be a direct response to eutrophication (Rinaldi *et al.* 1995). Again, however, such events can prevent a large proportion of primary production from being available to support increased production at higher trophic levels, at least in the pelagic zone. The long-term consequences of such restrictions are poorly described but, undoubtedly, would be significant. In addition, mucilaginous events in coastal regions can have a substantial negative impact on amenity and tourism, a particular problem on the Adriatic coast.

c) Toxic algal blooms

Some species of phytoplankton, most notably the red-tide dinoflagellates but also some prymnesiophytes and diatoms, are capable of producing potent toxins. Again, as is the case for mucilage production, the physiological trigger for toxin production in any one species remains largely unknown. Enhanced nutrient availability and shifts in nutrient ratio may stimulate increases in biomass of toxic forms (Olsgard 1993, Hodgkiss and Ho 1997, Nakamura *et al.* 1997), but controls over toxin production appear more complex. Nutrient availability may, however, be a contributing factor. For example, the flagellate *Chrysochromulina polylepis* had been present in the plankton of the Skagerrak and Kattegat prior to 1988 with no evidence of toxin production. Conditions in the summer of 1988 triggered the production of a lethal toxin by this organism which devastated fish populations over a wide area. The aquaculture sector was particularly badly effected. Studies of benthic organisms revealed that benthic communities down to 180m depth, which were normally remarkably stable in composition, showed significant declines in both species diversity and abundances corresponding with the *Chrysochromulina* sp. bloom which recovered only two years after the event (Olsgard 1993). In this study, as in most others on the impacts of toxic algal blooms in the field, was correlative only; the precise mechanisms by which the benthic organisms were impacted could not be determined.

The toxic phytoplankter *Pfiesteria piscicida*, and the *Pfiesteria*-like dinoflagellates (Burkholder and Glasgow 1997), represent a significant emerging problem and a potentially devastating threat to wild and farmed fish stocks. *Pfiesteria piscicida* appears to be widely distributed and produces toxins under certain, poorly defined physiological conditions which are lethal to all fish species tested to date, even at very low cell densities. At sub-lethal cell densities, the toxins induce severe skin ulceration, which can also lead to death through secondary infection. *Pfiesteria piscicida* has a complex and poorly understood lifecycle. Although its physiology and, to some extent, development appear to depend in part on inorganic phosphorus and nitrogen supply, the precise triggering factors and mechanisms involved are not known (Burkholder and Glasgow 1997); there is some evidence that outbreaks may coincide with localised nutrient pulses. Further research is urgently required to determine the relationship, if any, between recent outbreaks of this organism and coastal eutrophication.

d) Potential impacts on higher trophic levels

Clearly, however structural changes in the plankton community can lead to changes in both pelagic and benthic community at all trophic levels (Johansessen and Dahl 1996, Nakamura *et al.* 1997). Such changes could be long-lasting, fundamentally altering the pathways of carbon and energy transfer over wide areas of the coastal marine environment (Heiskanen *et al.* 1996, Ingrid *et al.* 1996, Bonsdorff *et al.* 1997).

Tett and Mills (1991) suggested that a shift from diatom to flagellate dominance in phytoplankton communities could lead to the replacement of crustacean zooplankton and fish by cnidarians (jellyfish and ctenophores) as primary predators, or their inclusion as an additional trophic level. The authors further speculate that increased productivity could allow for growth of cnidarians beyond the size range in which they normally fall prey to zooplankton predators. As few predators eat cnidarians, the latter would increase in number and the principal fate of their carbon would be export to the benthos rather than transfer to higher consumers. This would have obvious consequences in the form of increased production and/or hypoxia of the sediment and deep water communities. Such cnidarian-dominated communities already exist in some enriched Norwegian and Swedish fjords, suggesting that such fundamental changes in community structure may be feasible consequences of continued enrichment over wider areas (Tett and Mills 1991). Clearly such changes would have profound impacts on higher trophic levels and could lead to irreversible breakdown of current ecosystem function.

Given the complexity of interactions between physical, chemical and biological processes in marine systems, it is practically impossible to determine the relative importance of eutrophication, as compared to other factors, in determining fish stocks. Direct evidence for effects of increased primary production on fisheries is, understandably, scarce. Most data are derived from the catch per unit effort statistics of commercial fisheries which themselves, have an enormous impact on fish stocks. Rosenberg *et al.* (1990) speculated that the increases in catches in the Baltic during this century could result in part from eutrophication (see also: Hansson and Rudstam 1990), although increases in catching efficiency and the decline in seal populations were undoubtedly also important contributors to this trend. Rijnsdorp and van Leeuwen (1996) reported that, in shallow waters, observed long-term increases in populations of North Sea plaice might reflect impacts of eutrophication on higher trophic levels. Further offshore the primary trends appeared to relate more closely to trawling effort. Bonsdorff et al (1997) noted similar trends for some coastal fish species in parts of the Baltic, and suggested that eutrophication may have contributed in some measure. There is some correlative evidence that fisheries yield in the Kattegat may have increased in line with changes in primary production over the last few decades. Nevertheless, Nielsen and Richardson (1996) warn that this may not be a cause effect relationship and certainly should not be used as a justification for initiation of marine fertilisation programmes in an attempt to enhance primary production and, hence, fish biomass.

e) Extent of the impacts

That eutrophication is a significant concern in many coastal marine areas is no longer in dispute. Nevertheless, the debate as to how widespread the existing or potential impacts of enhanced nutrient loading are is ongoing. Certainly impacts of eutrophication are generally most evident in shallow waters, particularly in embayments, harbours and lagoons with limited water exchange and/or which receive substantial inputs of nutrients from anthropogenic sources (e.g. Abdulmoati 1996, Diaz Zaballa and Gaudy 1996, Webber and Roff 1996). It is also true that impacts further offshore will generally be less marked and, consequently, more difficult to detect and quantify in terms of trends (Stromberg 1997). Nevertheless, the significance of an impact, irrespective of its magnitude, will depend to a large extent on the sensitivity of any one region to disturbance of the food web, and this may vary greatly over small geographical scales (Mackas and Harrison 1997). Moreover, although the most acute effects may be restricted to areas immediately adjacent to substantial point sources of nutrients, some impacts have been documented over wider sea areas, including the Gulf of Mexico, Baltic and Southern North Sea (Lancelot 1995, Aure *et al.* 1996, Ingrid *et al.* 1996, Lohrenz *et al.* 1997).

Despite the acknowledged importance of the problem, and its severity in some locations, documentation of the impacts of eutrophication remains patchy and understanding of underlying mechanisms, particularly those triggering algal blooms and changes in physiology, is still limited. Impacts will always be difficult to predict because of the complexity of physical, chemical and biological interactions which occur in marine systems, particularly in high energy coastal environments (Lapape *et al.* 1996). Such interactions will ensure that increased inputs of nutrients will not always lead to an increase in system production, or even production of one component species (Billen and Garnier 1997). Simple assumptions that a quantity of nutrients “in” will translate to a level of production “out” cannot be expected to hold within marine systems.

f) Existing Measures

Despite the complexity and uncertainty surrounding the impacts of anthropogenic elevations of nutrient inputs, there is a wide recognition that action is required in order to reverse environmental degradation which has already taken place and to avoid further eutrophication related impacts. Within the North East Atlantic region, PARCOM Recommendation 88/2 set the target of a 50% reduction in anthropogenic inputs of nitrogen and phosphorus to the marine environment to be achieved between 1985 and 1995. This approach was further supported by the recognition of the need for a coordinated approach under PARCOM Recommendation 89/4 and a later agreement to address inputs as far as possible through source control measures under PARCOM Recommendation 92/7. Similar provisions, including the same 50% targets, were subsequently adopted by other regional seas programmes, including the Helsinki Commission (HELCOM) , the Rhine Commission and the North Sea Ministerial Conference (NIVA 1997).

Although initially a politically-based target, rather arbitrary in relation to the goal of protection of the marine environment, the proposal for 50% reduction as an effective measure subsequently received some scientific support, notably from the studies of Rosenberg *et al.* (1990) in the Skagerrak and the modeling work or Peeters *et al.* (1995) in the Southern North Sea. Since the original target was set, substantial work has been undertaken to achieve these aims, particularly within North Sea and Baltic States. Nevertheless, the Final Declaration of the Fourth International Conference on the Protection of the North Sea (MINDEC 1995) noted that, while most North Sea countries were likely to meet the 50% target reduction for phosphorus, most would fall short of the target for nitrogen emissions. The wisdom of the introduction of expensive measures further to reduce emissions of nitrogen without a complete understanding of the nutrient budget and dynamics of the system has been criticised (Soderstrom 1996). Much debate remains focussed on the most appropriate ways in which to calculate reliable input figures. Within HELCOM, the target date has recently been extended to 2005 (NIVA 1997).

Quantification and control of diffuse inputs remains one of the most significant challenges of the nutrient input reduction strategy for the North Sea and other regional programmes. OSPAR (1994) recognised that more specific would be required to tackle the complexity of sources and difficulties in assessing their relative contributions. Currently the revised OSPAR strategy remains in draft form. Furthermore, North Sea Ministers (MINDEC 1995) recognised the need to strengthen existing commitments in order to achieve the objectives, recommending that the entire North Sea should be treated as nitrate-sensitive area according to the EU Urban Wastewater Directive. In addition, renewed commitment was given to address diffuse agricultural and atmospheric inputs at source. How effective such measures will be in achieving the stated goals and, moreover, how effective these goals will be in protecting the marine environment from the threats of eutrophication, remain to be seen.

It is likely that many developing countries will continue to have a growing impact on their coastal ecosystems, particularly as large cities and industrial complexes expand and as more intensive agricultural and aquacultural practices develop (Nixon 1995, Howarth *et al.* 1996). With the benefit of hindsight and the experience gained in the developed world, it should be possible to avoid the potential for cultural eutrophication and resulting environmental degradation in developing nations as far as possible, rather than trying to solve the problem later.

g) Overview

In summary, although, by definition, eutrophication involves an increase in primary productivity, an increase in productivity at higher trophic levels does not necessarily follow. The relationships between enhanced nutrient loading and ecosystem function and productivity are complicated by secondary effects of species shifts, changes in the number and nature of trophic interactions, production of toxins and deoxygenation of bottom waters and sediments. Because of the complexity of effects of eutrophication it is difficult to predict the total ecosystem response to an increase in nutrient loading in any particular case. The extent to which changes in phytoplankton community structure resulting from coastal eutrophication have an impact on higher trophic levels is largely unknown.

3.5 Biological Pollution

a) Sewage disposal into the marine environment

Sewage is composed of human excreta and domestic waste, with or without industrial additions. It has been estimated that 100g of raw sewage is produced per person per day and that in sewerage systems this translates into a flow of 180 litres per day. At present, the EC legislation that is designed to protect the quality of coastal waters focuses on two aspects; the bacteriological profile of the water itself and the levels of contaminants found in shellfish. These are commonly thought to be the two main pathways for human infection. Disposal of sewage into marine waters jeopardizes both these conditions. Sewage management policies are constructed around this legislation. It will be seen that entirely beneficial outcomes are not necessarily obtained through attempts to adhere with these standards. This is as a consequence of the limitations that are inherent in such narrowly defined legislation. For this reason management plans for sewage disposal should be designed with due diligence; preferably involving public debate, independent scientific assessment and open communication between all the concerned parties (McIntyre 1990, Giroult 1995, Rogers 1995).

In the UK, fifty out of the 447 designated bathing waters failed to comply with the *mandatory* EC Bathing Water Directive (76/160/EEC) standards in 1997. These standards require that 95% of the samples collected during the bathing season (15 May- 30 September) contain less than 10,000 total coliforms per 100ml and no more than 2,000 faecal coliforms per 100ml. Thus 88.8% of our beaches attained a reasonable level of bathing water quality according to current legislation. 78% of the beaches have complied with these levels for the last three years while 19% have only passed once or twice. This high level of variability is partly due to the variability of the dataset and also the arbitrary nature of present monitoring techniques. Twenty single samples are taken from each site, during the bathing season, and chance plays an important and undefined role in determining the likelihood of any particular beach passing. Weather is a particularly significant contributor to the variability that bacteriological levels are subject to. The *guideline* coliform and faecal streptococci standards that are also outlined in the EC Directive are as follows; 80% of samples must not contain more than 500 total coliforms or 100 faecal coliforms per 100ml and 90% of samples must not contain more than 100 faecal streptococci per 100ml. Only 167 out of 447 or 37% of the UK bathing waters achieved this blue flag grading despite huge expenditures last year on sewage treatment and outfall improvements (Wyer et al. 1997, E.A. 1996).

Elsewhere in the world, water quality standards in the range of the European Community *guideline* levels are regularly used. They are even becoming considered as too lenient. The US National Technical Advisory Committee set bacteriological standards for bathing waters in 1968 at an upper limit of 400 faecal coliforms per 100ml with a geometric mean of 200 per 100ml. In 1986 the US Environmental Protection Agency amended this standard to include maximum allowable counts for single samples. Australia and Canada implemented similar standards in the early 80s (Dadswell 1993).

TABLE 7:.

Criticism	**Effect on predictive powers of coliform counts as a sewage indicator.**	**Reference**
Comparatively low survival rate of faecal coliforms in comparison to other microorganisms found in sewage; including pathogenic viruses and faecal streptococci.	Could lead to false negatives and underestimate the level of sewage contamination. The relationship between survival rates of pathogenic organisms and any indicator organism has not been established. It is also known to vary from one type of ecosystem to another.	Bruni et al. (1997), Dadswell (1993), Springthorpe et al. (1997)
Conventional testing techniques do not	Could lead to an underestimate of as	Johnston et al.

include gram-negative bacteria that can survive in a viable but non-culturable form.	much as an order of magnitude.	(1993)
A lack of reproducibility has been observed between different laboratory techniques used for enumerating faecal coliforms.	Unreliability and low comparability of results.	Dadswell (1993)
Coliform bacteria can develop antibiotic and/or stress-mediated resistence via "survival genes". This leads to greater viability or persistence of populations of coliforms that have been placed under duress at a previous point in time (e.g. in STPs).	Results that vary depending on previous treatment and environmental conditions.	Munro et al. (1994), Johnston et al. (1993)
The use of an organism that does not cause any of the reported symptoms to assess human health risk is accused of being an unscientifically grounded choice of indicator organism.	Little correlation between the detected levels of total coliforms and reported health problems.	Dadswell (1993)
The legislation only applies to designated parts of our coastline and for less than half of the year.	Produces incomplete datasets and encourages the use of part-time control strategies.	Dadswell (1993)
Faecal coliforms are capable of regrowth in river water.	Questions their validity as an indicator organism.	Springthorpe et al. (1997)
Significant amounts of coliform bacteria have been derived from environmental/non-point sources that are not sewage based.	Natural background levels have to be estimated at each site. They will vary with environmental conditions such as weather.	Roll & Fujioka (1997), Smith (1994), Wyer et al. (1997), McIntyre (1990)

STPs = sewage treatment plants

TABLE 7: Some limitations of present standards set by the EC Directive 76/160/EEC using faecal coliforms as the primary environmental indicator of sewage contamination

Faecal coliforms have been used traditionally as the universal sewage indicator organism. The criticisms leveled at the EC Directive and the use of faecal coliform counts are outlined in Table 7. Several authors have identified faecal streptococci as the most viable alternative indicator organism to coliforms, primarily as they persist longer in the environment, are excreted by mammals but can not multiply in sewage effluents (Anderson et al. 1997, Dadswell 1993, Johnston et al. 1993). However, Anderson et al. isolated enterococci bacteria in such significant amounts from sources other than sewage that this could not be used as an indicator for sewage contamination in New Zealand. A recent evaluation of *Eschericha coli*, a single species of coliform bacteria looks more promising and was chosen in 1993 by the World Health Organisation as the most suitable indicator organism of faecal pollution (Baudiࡊov↔ 1997). However no single organism will be universally applicable to all ecosystems. The behaviour and significance of contaminants and microorganisms that are derived form sewage systems and pollute our marine ecosystems will only be thoroughly understood through the use of environmental monitoring programs that utilize several analytical techniques that are specifically chosen to suit the specific ecosystem that is being evaluated (Bernstein & Dorsey 1991).

3.6 Radioactive Pollution

Radioactive isotopes of at least 16 of the 91 naturally occurring chemical elements are present in the oceans and possibly there are traces of a further 16 (GESAMP 1990). Natural radionuclides can be mobilised in large quantities into the marine environment by industrial processes. For example, discharges of natural uranium result from phosphate rock processing. (McCartney *et al.* 1990). In both the inshore and offshore oil industry, the scale which builds up on the inside of pipes carrying a mixture of oil and formation water has long been recognised as a radiation hazard (Wilson & Scott 1992).

Emissions of artificial radionuclides, however, have attracted the greatest attention. Subsequent to the use of atomic bombs in 1945, the atmospheric testing of weapons began. Some 400 devices have been exploded, predominantly between 1954 and 1962. Each nuclear explosion releases over 400 radioactive isotopes into the biosphere of which around 40 are considered particularly hazardous (Cockerham & Cockerham 1994). Without doubt weapons testing is the largest single source of artificial radionuclides to the oceans through fallout. Low level concentrations are measurable in the oceans world-wide (GESAMP 1990). Radiation in the environment can act as a potent carcinogen and mutagen (Cockerham & Cockerham 1994).

Radionuclides have also been introduced to marine systems through the dumping of radioactive wastes, operational discharges from nuclear power facilities and research reactors, together with the reprocessing of spent nuclear fuel elements. In 1987 around 580 reactors existed for electricity production, although not all of these were completed or operational. Together with other reactors used for research, isotope production and military purposes (Hewitt 1990), the land based total is around 1000 units (Cockerham & Cockerham 1994). The two major commercial reprocessing operations are located in Europe but others are known to exist in Japan, India and Russia. Inevitably, there have been a number of accidents involving reactors (see Table 8). Marine accidents, losses and deliberate dumping now mean that some 50 warheads, 23 reactors and an unknown number of other devices rest on the sea bed (Broadus & Vartanov 1994). This includes power units from satellites which have re-entered the Earth's atmosphere.

Although 98% of radioactivity in the sea is attributed to natural sources, this gives a misleading perspective on the significance of artificial radionuclides in marine systems. The biological behaviour of many of artificial nuclides is extremely poorly understood. Many of the dose models (see e.g. GESAMP 1990) do not adequately take account of the chemical and biochemical properties of radioisotopes in addition to their radiological properties. Further, inputs of artificial radioisotopes from nuclear industry or the military are often highly localised, providing a point source input rather than the diffuse inputs associated with atmospheric weapons testing or natural radionuclides. The same is true of dumped wastes, lost weapons and sunken vessels.

SOURCE	**QUANTITY**
COSMOGENIC/TERRESTRIAL (Natural)	$1.6\text{-}2.0 \times 10^{7}$ PBq
ATMOSPHERIC TESTING	$1.5\text{-}2.0 \times 10^{5}$ PBq
FUEL PROCESSING*	4.8 PBq
FUEL REPROCESSING	Not known
Sellafield (1992) (Excl. Tritium)	76TBq
After THORP Commissioning (Excl. Tritium)	520TBq
Tritium discharged (1992)	3,500TBq
ACCIDENTAL RELEASES/DEBRIS	Total unknown

Windscale UK, 1957	768 TBq
Idaho Falls USA, 1961	4 TBq
Three Mile Island USA, 1979	$1x10^2$ PBq
Chernobyl USSR, 1986	$1.85x10^3$ PBq
DISPOSAL/DUMPING (Total)	>$1.5x10^2$ PBq
North Atlantic dumpsites (26)	45 PBq
Pacific dumpsites (21)	0.57 PBq
Arctic dumpsites (USSR) (>20)	90 PBq

TABLE 8: Cumulative inputs of natural and artificial radionuclides into the oceans by source. 1 PBq = $1Bqx10^{15}$, TBq = $1Bqx10^{12}$. Releases from Three Mile Island largely radioactive noble gases with short half life. Numbers of dumpsites shown in parentheses. Information from Broadus & Vartanov 1994; Hewitt 1990. *The figure given for fuel processing does not appear to include emissions from fuel reprocessing. Illustrative figures for Sellafield for 1992 and after the new THORP plant is commissioned (OSPAR 1992; NRPB 1993) are given exclusive of values for tritium. Tritium figures for 1992 are given in parentheses.

a) Illegal Dumping and Accidental losses

The scale of illegal dumping was not known until recently, but such operations took place over a period of 30 years in the Barents and Kara Seas. Greenpeace Inetrnational was instrumental in drawing attention to this issue through a number of submissions made to the London Convention (see e.g. Greenpeace 1993 a & b). The total radioactive inventory consisted of around 90PBq. There is evidence that some containers of low-level waste dumped in relatively shallow water have leaked (IAEA 1997; Salbu *et al.* 1997) although dumped reactors do not appear so far to have leaked. Perhaps the best studied example of an accidental loss is that of the Soviet submarine *Komsomolets* which sank complete with its pressurised water reactor and two nuclear tipped torpedoes in international waters in the Norwegian sea. The total radioactive inventory aboard is estimated at 8PBq. Of this it is estimated that 1TBq year may be being released into the surrounding environment of which 500GBq is caesium 137 (Hoibraten *et al.* 1997). Although in both the cases of the dumping and the loss of the submarine the radioactivity is not thought to pose any long term global hazard (Nielsen *et al.* 1997), the potential for severe local impacts means that these sites will require careful monitoring into the future. Eventually, the recovery of these materials and removal to secure land based storage will be necessary, before containment is breached by corrosion or physical damage.

b) Discharges from Reprocessing

Emissions of artificial radionuclides from point sources are dominated by those from the Cap-la-Hague plant in France and from the Sellafield and Dounreay sites in the UK. Although a wide variety of other potential sources exist including nuclear power stations, other industries and historical dumpsites in the UK, these accounted for around 3% of emissions. By contrast reprocessing accounts for around 97% of the aggregate emissions from all sites (MAFF/SEPA 1997). In the North Sea, the different origins of the water can be differentiated on the basis of the concentrations and composition of radionuclides which they contain (Du Bois *et al.* 1993). These radionuclides arise principally from reprocessing of used nuclear reactor fuel rods. The transuranic elements are solely a product of the nuclear industry and have been discharged in large amounts by the nuclear reprocessing industry. One ton of spent uranium fuel from a typical reactor after ten years cooling time is estimated to contain 10kg of plutonium, 0.48kg of neptunium and 0.16kg of americium. Curium is the only other transuranic element produced in quantities greater than 0.01kg. Together with a variety of other radioactive activation and fission products, these can be released when these fuel rods are reprocessed to produce oxides of plutonium and uranium. At each stage of the reprocessing operation liquid effluents are treated. The efficiency of treatment procedures is inevitably less that 100% and significant quantities of these materials are, therefore, discharged to sea. The quantities and relative proportions have varied considerably with time and depend upon the type of fuel reprocessed, storage time of the fuel and the existence and performance of effluent treatment plants. In addition, significant releases of radionuclides take place to atmosphere.

Discharges of the actinide elements totalled some 5.21 x 10^{15} Bq, of which $5x10^{15}$ Bq was plutonium-241, from the Sellafield site alone between 1978 and 1982 (Morse & Choppin 1991). By 1991 it was estimated that the total environmental inventory of Cs-137 was $2.9x10^{4}$ TBq, Pu-238 was 100TBq, Pu-239/240 was 610TBq and Am-241, 945TBq (Cook *et al.* 1997). The mid to late 1970s represented a peak of discharges for both caesium-137 and actinides elements from the Sellafield site. At that point an overall decrease occurred, although some very high discharges were recorded in the 1980s. This downward trend was reflected in downward trends in caesium and strontium activities in regional marine waters (Nies 1990). It was widely predicted that commissioning the Thermal Oxide Reprocessing Plant at Sellafield would reverse this trend very significantly. On the basis of published figures (BNFL 1993) excluding tritium, around 76TBq of radioactivity were discharged annually in the early 1990s. Projections were made that this would rise to around 520TBq annually, falling to 393 TBq after commissioning of the Enhanced Actinide Removal Plant (EARP) (NRPB 1993). This includes increased levels of caesium-137 and some transuranic elements. Much of the current increase is due to treating of concentrated effluents and wastes (Medium Active Wastes) which had previously been allowed to accumulate on the site (ENDS 1996; Leonard *et al.* 1997). These wastes were routinely discharged to sea until 1980 when concerns about plutonium led to them being stockpiled until completion of the EARP plant. Future increases arising from the THORP plant are expected for some isotopes as the plant becomes fully operational, including carbon-14, tritium and iodine-129.

The EARP plant is designed to remove transuranic radionuclides. Following commissioning of the plant it has been found that significantly raised levels of the isotope technetium-99, with a half-life of 210,000 years, are now being discharged since the flocculation and ultrafiltration process removes plutonium and americium but is ineffective for technetium. This has led to very significantly elevated concentrations of Tc-99 being accumulated by marine organisms. Concentrations in lobster measured in 1996 were found to be 17,000 Bq/kg. This was twice the figure recorded in 1995 and 44 times the concentration recorded in 1993 prior to the EARP plant going into operation. Overall, Tc-99 levels have increased by a factor of 20-40 between 1993 and 1995 in marine species found off the coast (ENDS 1996), with elevated levels being found in Scottish and Irish marine organisms.

The findings followed from the raising of permitted discharges of this isotope from 10Tbq to 200TBq. Actual discharges rose from around to 6TBq to 72TBq in 1994 and to 190TBq in 1995. The elevation of Tc-99 concentrations further afield from the plant comes as no surprise. From the late 1970s onwards, discharges of 40TBq per year were estimated based upon the accumulation of Tc-99 in seaweeds from the West Greenland Coast (Kershaw & Baxter 1995) and recent work has shown that Tc-99 moves out of the Irish Sea via the North Channel in 3 to 6 months (Leonard *et al.* 1997). The bioaccumulation of Tc-99 should not have come as a surprise. A concentration factor of over 2000 has been reported previously for lobster (Swift 1992) for Tc-99. Significantly, this situation is not being dealt with in a precautionary manner. It appears that the backlog of waste may simply be discharged at lower levels over a longer time period. The operator, BNFL has applied to reduce permitted discharges from 200TBq to 150TBq. Some 700TBq of Tc-99 remain in storage with further arisings of 50-60TBq per annum implying that the backlog of wastes will be cleared six years later than originally planned but without reducing the total amount discharged.

Sellafield began discharging radioactive isotopes in 1952. La Hague in France went into operation in 1966. Over this period, expressed as a percentage of the Sellafield releases, the French site contributed around 2.3% of the Cs-137, 12.2% of the Sr-90, 12.6% of the Tc-99 and 0.4% of the alpha emitting plutonium (Kershaw & Baxter 1995). In general, discharges of these from La Hague radionuclides are now of the same order of magnitude as those recently reported from Sellafield, increasing the relative importance of this site (see: Guegueniat *et al.* 1997). Discharges originally peaked during the late 1970s and early 1980s at which time they were much lower than those from Sellafield. After a period when discharges were reduced (Betis 1993), since the early 1990s they have been rising again due to increased fuel throughput. There are some further differences reported in the literature.

Tritium releases have been at consistently higher levels from La Hague since 1984. In the case of releases of iodine-129, both Sellafield and La Hague emitted an estimated 15-40kg year but in 1989 discharges began to increase at La Hague and reached around 80 kg/year. in 1992. This is due to up to 50% of the I-129 trapped in the fuel being released to the environment. (Raisbeck *et al.* 1995). A further difference between the two discharges is the greater quantity of antimony-125 released from operations at La Hague between 1982 and 1992. Overall, these have been on average a factor of 5 higher than from Sellafield when comparing decay corrected environmental inventories. Total discharge of this radioisotope is estimated at 335TBq in total. These discharges began to decline after a maximum reached in January 1987 to 22,000 GBq/month falling to around 6000GBq/month in April 1988, finally falling to 450Gb/month in September 1991 (Guegueniat *et al.* 1994). In the case of many of the radionuclides discharged, data from La Hague has not been made publicly available to anything like the same extent as that from Sellafield, resulting in substantial gaps in the data.

The third European reprocessing facility is the UK Atomic Energy Authority facility at Dounreay. This carries out commercial reprocessing of research reactor fuel rods made from highly enriched uranium. Existing discharges increased between 1992 and 1995 for alpha and beta emitters. At the moment, discharges are around an order of magnitude lower than those from Sellafield but these are likely to be raised to between 3 and 22 times the existing values as a result of applications for new discharge limits. This will raise total beta discharges, for example from 6.6TBq per annum to 49Tbq per annum (AEA 1995).

Overall, the pattern is one of rising discharges from reprocessing of nuclear materials. Many of these radionuclides have extremely long half lives and are unknown in nature. Continuing discharges effectively cause a build up of artificial radionuclides in the ecosphere in clear violation of the principles of sustainable development. Study of the complex behaviour of these radionuclides makes it clear that many of the predictions made about the impacts of the discharges proved to be false, particularly in regard to their environmental behaviour and dispersion.

c) Environmental Behaviour of Artificial Radionuclides

The behaviour of radionuclides from reprocessing has been particularly intensively studied in relation to the discharges from Sellafield. In the vicinity of this plant very high levels of radioactivity are found. It is only relatively recently that the complex environmental behaviour of these artificial elements has been appreciated . Some of the dispersion pathways which have been identified were not originally anticipated by the mathematical models used to predict environmental fate of radioisotopes. Their environmental distribution results from both physical and biological processes.

There is no doubt that emissions from Sellafield in conjunction with La Hague and Dounreay have significantly elevated radiocaesium levels over the whole of the North Atlantic (MAFF 1991). In addition, the northern seas areas are also receiving radioactive contamination from military nuclear sites in the former Soviet Union, via rivers (Aarkrog 1995; Christensen *et al.*1997).

Radioisotopes dissolved in seawater are transported to remote regions as evidenced by the behaviour of Tc-99 described above. Cs-137 is also distributed very widely in Northern waters as a result of reprocessing. Together with other soluble radionuclides, some Cs-137 leaves the Irish Sea and following a well defined path around the West and North Coast of Scotland enters the North Sea. This initial transport is followed by transport north with the Norwegian Coastal Current. Radionuclides are carried from Northern Norway into the Barents Sea and also via the West Spitsbergen Current passing through the Fram Strait (Kershaw and Baxter 1995). Some of the soluble radionuclides also become associated with sediments. In Arctic regions themselves, current patterns will transport dissolved contaminants out of the Arctic over a time frame of several years to decades. Waters below the surface layer will retain contaminants on a time frame up to several centuries, before leaving the Arctic basin (Schlosser *et al.* 1995). There is increasing evidence that sea-ice can play an important role in the transport of radionuclides into biologically productive areas. This phenomenon has only recently been identified (Meese *et al.* 1997; Rigor & Colony 1997). Discharges of isotopes from La Hague are carried northwards through the North Sea and also reach Norwegian waters (Guegueniat *et al.* 1997).

The non-soluble nuclides such as plutonium and americium are quickly removed from the water column close to the point of discharge by precipitation or adsorption to particulates. The distribution between water and sediment is determined by the physical and chemical properties of the isotope, the water and the sediment itself (Oughton *et al.* 1997; Nash *et al* 1988; Mudge *et al.* 1987; Penrose *et al.* 1987). These interactions are complex and not fully understood. While there is now a large body of information on the current spatial distributions of the sediment conservative isotopes discharged from Sellafield, no confident predictions can be made on the effect of future redistribution of sediments or exactly what the fate of these nuclides is likely to be (Kershaw *et al.* 1992).

On the basis of core samples taken in the Irish Sea in 1977-1978 it has been estimated that at least 80% of the plutonium discharged up to that point was localised within a 30km strip of coastal sediment between Scotland and North East England. A small dissolved and particulate fraction being remobilised and removed from the Irish Sea via the North Channel (Mitchell *et al.* 1986). There is now evidence that these sediments are now acting as sources of soluble plutonium in seawater through sediment disturbance (McKay & Pattenden 1989) Although discharges of Pu from Sellafield fell from 37TBq in 1979 to 1TBq in 1988, concentrations of Pu in seawater close to Sellafield have risen from 0.43Bq/litre to 2.7 Bq/litre normalised to discharge levels (Swift 1992). This was not predicted by models and evaluations used to gauge likely impacts of these discharges.

The sediments have also been biologically disturbed. Near the discharge point the isotopes are concentrated in the top 2-5cm of sediment, in muddy offshore areas they are found some 20cm deep as a result of the activities of burrowing organisms (see: Kershaw *et al.* 1992) . This may help the Pu to dissolve as conditions in the sediment become more oxidising (Hamilton-Taylor *et al.* 1987). Micro-organisms may also play a role in the process (Gadd 1996). Currently, it is estimated that around 1.2TBq of Pu-239/240 are remobilised and lost from the Irish Sea (Cook *et al.* 1997). Remobilisation of sediments and dissolution of isotopes means that in turn, the estuaries of rivers local to the plant are accumulating isotopes (Kershaw *et al.*1992; Curtis *et al.* 1991). Estimates for the plutonium budget of the Esk estuary over a single spring tide cycle showed the input to be 285MBq of which 89% was in the particulate phase and 11% in solution. Of this about two thirds was returned to the sea and one third retained in the inner estuary. (Kelly *et al.* 1991).

The radionuclides associated with particulates can be carried into low energy saltmarsh and tidal pasture areas used for grazing animals. This appears to be seasonally variable. In the Esk estuary, highest levels of Pu239/240, Pu-238; Am-241; Cs-137; Ru-106 and Nb-95 were found to be present in sheep grazing the marshes in spring and autumn (Howard 1985; Howard & Lindley 1985). Further studies have shown that similar deposition takes place in the low energy areas of other local estuaries (Howard *et al.* 1996) including the Mersey (Jones *et al.* 1994) and Ribble (Assinder *et al.* 1997).

The return to land of radioisotopes discharged from reprocessing operations has been documented for all three European sites. At Dounreay the principle mechanism appears to be via nuclide enriched stable sea foam being carried ashore by the wind (Toole *et al.* 1990) whereas at Sellafield and La Hague sea to land transfer appears to involve air transport of marine aerosols containing particulate associated Pu (McKay & Pattenden 1990). The phenomenon appears to have been only poorly studied in the case of the French site. At Sellafield Pu has been detected up to 60Km inland from the plant while at Dounreay, contamination is restricted to a smaller geographical area. Overall, this pathway is estimated, based on soil samples taken between 1977 and 1979, to have deposited up to 80GBq of Plutonium in the land area within a 50km radius. The longer term dimensions of this phenomenon remain unknown. Other figures suggest that a coastal strip 5km wide by 40km long had been contaminated by the year 1980 in England. Similar findings have been made in Northern Ireland implying that this may be a widespread problem.

d) Accumulation and Persistence of Radionuclides

An evaluation of the environmental significance of a particular radionuclide needs to consider not only the half-life of the substance but also its biological half life within an organism and its ecological half life within the receiving system. In general the physical half life is longer than the other parameters, but the latter two functions can vary considerably. For example, US research has shown that Cs-137 with a physical half life of 30 years has a biological half life ranging from 5.6 days in ducks to 902 days in snakes. Ecosystem half-life was found to range between 1.9 and more than 20 years (Brisbin 1990). Possible food web interactions also need to be considered. A wide range of prey animals accumulate transuranic elements. These include molluscs and other food resources (Swift & Pentreath 1988; Miramand *et al.* 1987; Lowe 1991).

Protection of natural ecosystems from the damaging effects of radiation is predicated upon the questionable assumption that if humans are adequately protected from radioactivity, then natural ecosystems will also be protected (ICRP 1977). It seems reasonable to assume that increase in discharges will be accompanied by increased risks to humans, although these are not clear. The impacts of radioactive pollutants on species other than man are, however, not known well enough for this assumption to be validated (Thompson 1988; McKee *et al.* 1989). Quite apart from methodological problems which have been identified (see: IAEA 1979), areas of uncertainty include taxonomic variation in genetic repair capacity and cell biochemistry. Within animal populations, genetically determined susceptibility and sensitivity to radiation have not been considered adequately.

In consequence, there has been extensive routine evaluation of radionuclides in organisms which could constitute a significant radiation pathway to humans (see: e.g. MAFF 1993a). Very little work has been done to assess the potential hazards to these organisms in their own right. The limited studies which do exist indicate that concern is warranted. One calculation estimated a potential dose to grey seals (*Halichoerus grypus*) of 36 mSv per year, considering only caesium-137, in the vicinity of Sellafield, as against a recommended principal dose limit to the public of 1mSv annually (Pentreath & Woodhead 1988). Similar findings were made for black headed gulls which spend around 120 days per year on the Ravenglass estuary near Sellafield. The whole body dose was estimated at 2.8mSv (Woodhead 1986) while levels of 200-600Bq/kg of Cs-137 were found in skeletal muscle of other birds as against a level of around 3Bq/kg muscle in humans (Hunt *et al.* 1989; Lowe, 1991). Clearly, there are critical groups of marine wildlife which require realistic evaluation in relation to radiation hazards which they face (see: Johnston *et al.* 1996 McKee *et al.*1989).

The regulation of radioactive discharges is clearly not being conducted on a precautionary basis with respect to the poorly understood potential impacts upon natural systems. It appears that to an extent focus in radioecological studies has shifted more towards terrestrial systems following the Chernobyl accident (Aarkrog 1994). Given that the major point source emissions are to marine environments, this shift in emphasis is, at best, unfortunate.

e) Natural Radionuclides

Natural radionuclides have received relatively little attention despite the fact that they may be mobilised in considerable quantities by the bulk processing of minerals. Scales formed from production waters in oil production build up in pipes which have long been recognised as a radiation hazard (Wilson & Scott 1992) in both onshore and offshore operations.

Both potash and phosphate fertilisers are manufactured from ores high in natural radionuclides (Gray 1993). Radioactive elements are present in the finished products and in wastes discharged to sea. In phosphate ores isotopes arising from the natural uranium decay series are discharged as a result of this ore being naturally rich in uranium (Cowart & Burnett 1994). These include thorium-230, bismuth-214, lead-210 and polonium-210. Isotopes present in the fertiliser product are introduced into the environment as soluble phosphate complexes. In commercial fertilisers prepared from Moroccan and Senegalese phosphate rock each percentage of phosphate by weight results in a uranium-238 activity of 53Bq/keg. Up to 20% of this may be lost in the drainage from agricultural land (Barisic *et al.* 1992).

These natural radionuclides can be bioconcentrated. Concentration factors range between 0.5 for fish muscle to 60 in soft mollusc tissues for uranium. Thorium can be concentrated by a factor of 10 in fish muscle and 1200 in crustacean species. (Szefer *et al.* 1990). Polonium-210 is accumulated particularly strongly in fish. Cod, herring and flounder show accumulation factors of 36,000, 13,00 and 7,000 respectively. Zooplankton also accumulate this element with a concentration factor between 5,000 and

42,000 relative to concentrations in seawater and it can also be accumulated by benthic invertebrates (Skwarzec & Bojanowski 1988; Skwarzec & Falkowski 1988). Fish provide an important source of polonium -210 to humans (Skwarzec 1997).

The impacts of bulk processing of phosphate rock for fertiliser and detergent manufacture have been studied in the UK (Pentreath *et al.* 1989). From the 1960s until 1992, this plant alone discharged around 30 tonnes of natural uranium annually amounting to 0.4TBq of uranium-238 (Camplin *et al.* 1996) although other estimates have placed this at around 40-60 tons (McCartney *et al.* 1990) Edible molluscs sampled from waters in the vicinity were found to contain over 250 Bq/kg of polonium-210 as opposed to around 7Bq/kg from control samples. At the peak of production at the plant, doses to sea food consumers in the locale were estimated to have peaked at 6mSv/y harbour waters close to the factory. Elevated levels of radon-226 have been found in harbour waters close to the plant and significantly elevated concentrations of lead-210 have been found in sea sediments in the vicinity. Recently, discharges have been much reduced and doses from this source have fallen to 0.04mSv per year.

Overall, in the North-East Atlantic region, the phosphate industry provides the main input of natural radionuclides, amounting to around 10TBq/year of uranium 238 and daughter isotopes. Many countries in this area have substantially reduced discharges from these sources (Camplin *et al.*1996). Nonetheless, the means whereby radionuclide releases have been reduced in the UK plants give cause for concerns arising in other areas. Although a specific plant was emplaced at the UK site to remove contaminating metals from the phosphoric acid used in the process. At this stage uranium discharges were reduced by a factor of four, and of daughter products by an order of magnitude (Camplin *et al.* 1996). The bulk of the improvement was achieved, however, by ceasing the production of phosphoric acid on the site, and importing acid from the ore producing countries. This is an increasing trend, implying that there is a potential for radiation hazards arising in areas where these operations are relocated

3.7 Pollution, Regulations, Controls and Progress

There is no doubt that chemicals and radionuclides are being discharged as complex, variable mixtures. The quantities being emitted are not accurately known in the case of chemicals although the data for radioisotopes is comprehensive (see e.g. RCEP 1992; DoE 1987). Monitoring of point sources is generally carried out on a less intensive basis than the sixty times annually required to reliably detect compliance with regulations (Ellis 1986). This problem translates regionally where regulation is formulated on the basis of progressive emission reductions (MINDEC 1990). The chemicals identified as priority hazards to be reduced by between 50% and 70% are only a very small subset of those entering the environment. Selection of further priorities taking place on a substance by substance basis, implies that full evaluation of the hazards will take very many years (see: Willes *et al.* 1993). This in turn implies that progressive reduction programmes are likely to prove unworkable. A further difficulty is that percentage reductions tend to be based on field analytical data, rather than measurements from the source (Johnston *et al.* 1994) and different authorities pursue different monitoring strategies (Jarvie *et al.* 1997) Consequently, input estimates for PCBs may be in error by between 30% and 50% for each monitored riverine source and for other substances by an order of magnitude (Klamer *et al.* 1991; Hupkes 1991; DoE 1987). This is also true of the Mediterranean Sea but here the body of data is much less well developed (Jeftic 1993).

Problems are compounded by the lack of good intercalibrated laboratory data. Even when intercalibration exercises have been carried out, these will address relatively few pollutants (Wells 1993; Wells *et al.* 1993; Wells & Balls 1994). In addition sharp regional differences in data collection further confuse the picture. In the Mediterranean by September 1988, approximately 14,600 data items concerning monitoring of heavy metals had been reported to the MED POL Co-ordinating Unit, which controls the long term programme for pollution monitoring and research in the region. Twelve countries submitted data; 87% of the data came, however, from only four countries (Jeftic 1993). Accordingly, the feasibility of providing the data necessary to effectively police chemical inputs and reductions on a region wide basis is highly questionable. In the face of these difficulties identified in monitoring and surveillance protocols, much could be gained from focusing increased attention on point sources of chemical discharges (see: Johnston *et al.* 1994) as a complement to the existing more widely based programmes.

From this far from exhaustive examination of pollution problems in the marine environment several points are clear. Since the Industrial Revolution human activity has not only significantly increased the cycling of natural substances in the environment but has also introduced organic chemicals which are entirely foreign to nature and a range of artificial radioactive isotopes. Moreover, while some of the effects of these changes are quite obvious there are many areas of uncertainty and doubt. There is no doubt, however, that the provisional principles of sustainability are bring comprehensively violated. Holistic protection of the environment cannot be reliably achieved on the basis of piecemeal legislation, nor on the basis of regulating single chemicals. Given this, the following quote provides some indications of the way forward and is worth reproducing in full:

The objective of regulations for controlling industrial water pollution is to protect natural communities of organisms. If done, that will include protection of humans as part of the general ecosystem. The ideal and ultimate goal must be no discharge of effluent, since plants and animals in the natural communities are adapted through millennia to conditions without human industrial input. On the other hand, humans are part of the system and cannot live without producing some sort of waste. Hence, any regulation of industrial discharges will always be a compromise between the ideal and whatever is possible at the moment. Regulations should be considered as temporary resting-places on the road to a goal of zero discharge. (Sprague 1991)

Far from being a utopian ideal, the zero emissions approach is gaining political momentum as well as being recognised in the scientific community. In 1995 at the North Sea Ministerial Meeting in Esbjerg (MINDEC 1995) signatory Governments noted the obligation accepted by all North Sea States to adopt the precautionary principle and the "polluter pays" principle in their work to protect the marine environment as a result of signing the 1992 Convention for the Protection of the Marine Environment of the North East Atlantic (OSPAR 1992). The following text was agreed on the regulation of hazardous substances:

17. The Ministers AGREE that the objective is to ensure a sustainable, sound and healthy North Sea ecosystem. The guiding principle for achieving this objective is the precautionary principle.

This implies the prevention of the pollution of the North Sea by continuously reducing discharges, emissions and losses of hazardous substances thereby moving towards the target of their cessation within one generation (25 years) with the ultimate aim of concentrations in the environment near background values for natural occurring substances and close to zero concentrations for man made synthetic substances.

This broad approach has also been adopted by the Helsinki Convention, the OSPAR Convention and is incorporated into the 1996 Barcelona Convention Land Based Activities Protocol. In addition, in relation to radioactive waste, the International Atomic Energy Authority in Principle 2 of Radioactive Waste Management has made it clear that containment and concentration of radioactive wastes is a preferred option to dilution and dispersion. Paragraph 17 of the Esbjerg Declaration makes it explicit that a precautionary approach to the discharge of hazardous substances is to eliminate them. New regulations need to be formulated with this end in sight and also to take account of the intrinsic uncertainties which exist in any evaluation of potential and actual environmental impact. As noted earlier, a precautionary approach to environmental protection has been adopted in principle by various national and international regulatory fora. Such an approach implies taking action in anticipation of deleterious effects. In order to protect wider marine ecosystems, these declared intentions urgently need to be translated into practical regulatory instruments which endorse zero-emissions for hazardous and radioactive substances.

4. Marine Capture Fisheries

4.1 Current Fishing Activity

Marine fishing is a highly significant human activity. Currently, marine fisheries generate around 1% of the visible global economy. Fishing and related industries support the livelihoods of around 200 million people globally. Although marine fish account for around 16% of the animal protein consumed by humans, dependence upon this food resource is greater in developing countries. (Everett 1996; Botsford *et al.* 1997).

From a base of 21 million tons in 1950, the world fish harvest including farmed fish, reached 116 million tons in 1996 (McGinn 1998). Fish caught in marine and inland waters have remained steady at around 90-91 million tonnes and although the growth in world catch has slowed, it is not yet declining. In 1995, the total catch figure for marine waters was 84 million tons of a total of 112.3 million tons. The balance was provided by aquaculture and capture fisheries in inland waters (14.6 Mt & 7Mt respectively) and aquaculture in marine waters (6.7Mt) (FAO 1997). Aquaculture production has accounted for a considerable proportion of the growth in fish production, rising from 7Mt 1984 to 23 Mt 1996 (McGinn 1998). Over the period 1990 to 1995, marine capture fisheries rose from 79 Mt to 84 Mt (FAO 1997). These figures do not include catches in artisanal fisheries which are not recorded but have been estimated at around 24 Mt *per annum* (Lean & Hinrichsen 1992). A further uncertainty is the take of fish on non-commercial species, those caught in lost and discarded fishing gear and those taken by "poachers" (Dayton 1998).

On the face of it, the above statistics could be interpreted as applying to a mature industry which is sustainably exploiting the world’s marine resources. This interpretation is very far from the truth. More careful analysis of what underlies these figures indicates that global fisheries are systematically violating principles of sustainability and that this situation has reached a crisis point. Simply, fish stocks around the world are being too intensively exploited. Open access to fisheries, overdevelopment of fishing capability and inadequate predictive tools have all played a part (Kesteven 1996). Over-intensive exploitation arises simply from institutional failures to adjust fishing pressures to finite and varying fish yields (Everett 1996). The conventional management systems were formulated at a time when fish stocks were more plentiful. Uncertainty and indeterminacy were not recognised as significant factors in fisheries management. Consequently, management strategies did not successfully evolve to be capable of managing intensively and fully exploited fishery resources. Developments in fishery activity can be categorised within time frames defined by international events (Garcia & Newton 1994) from whose work, the following account is summarised.

The period 1945-1958 was characterised by intense fisheries development during which marine capture fisheries increased production from 17.7 to 28.4 million tonnes. As early as 1945 the problems of resource depletion and discards had been recognised and important fisheries such as the North Sea and Atlanto-Scandian herring fishery together with the Hokkaido sardine fishery became depleted or collapsed. A consistent theme throughout this period was the emphasis upon a need for improved data and fisheries management. By drawing attention to underexploited resources in the Southern Hemisphere, the first FAO Technical Committee prompted the declaration of a 200 mile territorial sea jurisdiction in 1947 by Chile and Peru, an idea carried over to the Santiago Declaration which was signed in 1952. At the end of this period, in 1958 the UN Convention on Fishing and Conservation of the Living Resources of the High Seas was adopted in Geneva (UNCLOS I), but was not ratified by some of the most important fishing nations.

Between 1959 and 1972 fisheries rapidly expanded geographically and world catches rose from 30 to 60 million tonnes. At this time surveys were carried out in relation to undeveloped resources and long distance fishing fleets expanded operations off northwest and south west Africa and in the tropical oceans. These operations were supported by governmental subsidies. Many developing countries developed an industrial fishing sector and severe overfishing followed in coastal areas of these countries. In turn this was followed by resource collapses. The North Sea remained severely overfished. UNCLOS I was implemented in 1966, but together with the UNCLOS II Conference in 1960 failed to agree on national marine jurisdictional limits. At the beginning of the 1970s FAO predicted that many fisheries were no longer sustainable and that better management and greater exploitation of under exploited resources were needed. With the jurisdictional issue unsolved, fishing nations refused to reduce fishing effort directly and instead opted for indirect controls *via* quotas or Total Allowable Catches. In any case, the efficacy of 200 mile limits as a means to deny access and thereby control fishing effort was somewhat contradicted by the collapse of the Peruvian anchoveta in 1972.

1973-1982 saw fisheries production increase from 60 to 68 million tonnes and the state of stocks almost universally continued to deteriorate. The expansion of fisheries was constrained to some extent by fuel prices, but nonetheless there was a last expansion into the Indian, South Pacific and South West Atlantic in pursuit of high value species. The favourable natural stock fluctuations of low value small pelagic species allowed increased catches in the early 1980s which concealed the overfishing of high value demersal species. "Fishing licences" became widely used to constrain fleet sizes, particularly of foreign fleets, and a number of countries extended their territorial claims to 200 miles in anticipation of the UNCLOS III process which began in 1973. As a result, continental shelf areas which remained accessible were widely fished by foreign fleets and straddling stocks were put under extreme pressure. High seas fishing intensified. By 1982, free access to fishing grounds had virtually disappeared and nations had the opportunity to improve management of fisheries. At this stage it was becoming recognised that MSY and TACs derived from MSY had problems and some states started to consider adopting lower fishing mortality targets. During this period the bycatch from intensive shrimp fishing became an issue of significance. In 1982, UNCLOS was adopted.

Between 1983 and 1992 marine capture fisheries rose from 68 to 85 million tonnes. Global environmental conservation and sustainability became increasingly important issues. Several developed countries attempted to limit effort and improve economic efficiency through the use of individual transferable quotas (ITQs). Developing countries began to focus on controlling their national operators. The 1982 Convention was finally ratified in 1993 and entered into force in 1994. Previous to its entry into force, many of its provisions had been considered as customary law, reflected in national policies. Some nations began to work to protect straddling stocks in zones adjacent to their 200 mile Economic Exclusion Zones (EEZs). The negative impacts of subsidies began to be recognised as did the need to protect non-target sensitive species. The world competition for access to markets began as long range fleets sought to regain access rights. The 1992 UN Conference on Economics and Development agreed a set of principles for sustainable development under Agenda 21, including Chapter 17, of direct relevance to fisheries. This was followed in 1995 by the UN Fish Stocks Agreement.

Fisheries rely on relatively few species. Of the total of 20,000 known species of fish, around 9,000 are routinely fished. Only 22 are taken in amounts over 100,000 tons while five groups make up 50% of global fisheries. These are the herring, cod, jack, redfish and mackerel species (Lean & Hinrichsen 1992). Currently, (Botsford *et al.* 1997) 44% of fisheries are fully to heavily exploited, 16% are overexploited and 6% are depleted. Only 9% are regarded as under exploited while moderate exploitation of 23% of stocks is suggested. A further 3% are recovering from previous overexploitation. Most recently, using an alternative analysis procedure (FAO 1997) trends in 200 major fish resources indicate that 35% of these resources are overfished, 25% are fully exploited and 40% are still in the development stage. Taken together, the figures imply that now over 60% of world stocks urgently require appropriate management measures to be taken. Even so, around 8 million tonnes of fish were landed from all of the resources defined as depleted or over-exploited in 1994, falling from 14 million tonnes in 1985. The economic basis for current world fisheries is also highly suspect.

Various analyses of the fishing industry have identified that overcapacity in the global fishing fleet is a central issue to the profitability and sustainability of the industry. Between 1970 and 1989, the gross tonnage of the global fishing fleet increased by 4.6% a year. In 1992 the FAO estimated that the global fishing fleet had 30% more capacity than needed to harvest the worlds stocks (McGinn 1998) although estimates also suggest that it is as high as 50%. This has been driven in part by open access fisheries but, paradoxically also by the extension of jurisdictional limits followed by expansion of domestic fleets. Technological change has also played a part. The key to the overcapacity problem is continuing government subsidies. As a result of overcapacity in world fishing fleets, economic losses in 1989 were estimated at around $US54 billion. About half of this was covered by government subsidies. For every single dollar earned, the costs were $US1.77 (McGinn 1998). Other figures suggest that the estimated costs of fishing exceed revenues by $US 16 billion annually (Sissenwine & Rosenberg 1993). Subsidisation of the industry continues. While this may help to mitigate economic impacts of fisheries decline it does so by continued over-exploitation of resources.

With time there has been a distinct change in the pattern of the species exploited (FAO 1996). Until the

1970s the commercial landings of demersal fish increased and then became stable. Pelagic fish have been increasingly exploited and landings have continued to rise since 1950. These fish are generally of lower value than demersal species. The trend of exploiting lower value fish after depletion of preferred target species has been observed in the Canadian sector where species regarded as trash prior to the collapse of groundfish stocks began to be commercially exploited (McGinn 1998). This brings with it the possibility that slowly reproducing species like sharks may become extinct as fisheries develop to target them (Manire & Gruber 1990). The increasing exploitation of pelagic species has had a further effect. A reanalysis of landings data has shown that over the past 45 years, global fisheries have shifted perceptibly away from the large predatory fish toward smaller fish which feed lower in the food chain. This is particularly true of the Northern Hemisphere (Pauly *et al.* 1998). In fact this confirms a regional trend which was first observed in 1973-1982, although obscured by landings figures, where large long lived species were superseded by small short lived ones. The North Sea, the Gulf of Thailand and the waters off West Africa were affected in this way (Garcia & Newton 1994).

In the Southern Hemisphere, the wider trend of fishing down trophic levels may be partially obscured for the time being by development of new fisheries (e.g horse mackerel) coupled with the greater historical dependence on pelagic fish such as the Peruvian anchoveta. In Antarctica, however, removal of bony fishes followed by exploitation of krill has produced a significant lowering of trophic level of the exploited resources. Changing composition of catch in fisheries can also be traced through global catch per capita figures over the period 1950-1990. In 1950 fishing activity produced 8.4kg per person annually and 86% of this was used for food with the remainder being converted to fish meal. In 1990, catch had risen to 18.4 kg per person while only 71% was used directly for food. This figure fell to 61% in 1970 (Schnute & Richards 1994), perhaps reflecting the high quantities of low value pelagic fish taken in South Pacific fisheries where the fishery ultimately collapsed (FAO 1996).

The statistics (FAO 1996) also mask the significant problem of bycatch. Overall, it has been estimated that 28.7 million metric tonnes of fish are taken together with commercially exploited species. Of this, an estimated 27 million tonnes are discarded globally. This total does not include discards from many invertebrate, recreational and subsistence figures. A significant proportion (37.2%) of discards arise from shrimp trawl fisheries but cod, hake, flounder and crab fisheries are also important contributors and may contain very large numbers of juvenile fish. Survival of discards is generally recognised to be low (Alverson *et al.* 1994). This amounts to over 30 % of the landings from marine capture fisheries. Great uncertainty attaches to these estimates and they are likely to be underestimated. The magnitude of the problem for different type of fishing gear in different regions is made clear, but reliable data are absent or scarce for most fisheries with problems most clearly identified for those that are closely monitored (Hall 1996). Fish species are not the only victims of bycatch. Seabirds are caught in large numbers by several types of fishing gear (Lanza & Griffin 1996). Turtles also become entangled in fishing gear Brady & Boreman (1996). Dolphins are killed in large numbers by purse seine fishing operations for tuna (Hall 1996). Overfishing and bycatch combined have raised concerns about the possibility of extinctions in marine fish species (Vincent & Hall 1996) and highlighted the difficulties in evaluating the scope of the problem.

As a consequence of poor management and regulation of fish stocks, many are in decline. In the great majority of cases this is simply due to overfishing (Myers *et al.* 1995). There is some evidence that at low population levels some fish species may prove unable to reproduce but, in theory, for most stocks the impacts of overfishing should be reversible at the present point. The adequate management and protection of stocks is obviously the key to restoring these fisheries. In turn, the formulation of adequate descriptive models which reflect all the operational factors needs to be achieved. The current models used for fisheries management suffer from considerable inadequacies.

4.2 Fisheries Modelling

In general, fishery management is based upon mathematical modelling and analysis procedures. Fisheries scientists are now numbered in the tens of thousands world-wide and have approximately doubled in number every decade between 1940 and 1980 in the North American sector alone (Stephenson & Lane

1995) generating a prodigious scientific literature. Scientific effort has been focused upon determining the direct interactions between fishing activities and individual fish stocks. This is designed to optimise yields (Daan 1989) in specific fisheries. Control of the yield follows the theoretically simple strategy of modulating fishing effort and other technological factors, for example, net mesh sizes. These single species models depends upon one key assumption: That exploitation is the forcing function in the dynamics of the population with all other factors of secondary importance. It is reasoned that other factors should impose only background "noise" on observed trends in the population. This critical assumption is supported, however, only by the fact that heavily fished stocks are generally detectably depleted (Wilson *et al.* 1994). An associated assumption has been that control of fishing mortality can lead to an increase in numerical abundance of a species. Essentially, fisheries management attempts to protect against two forms of overfishing. Growth overfishing takes place when fishing activity is so high that fish are caught before they have achieved their full growth potential. This is regarded as being economically inefficient. Recruitment overfishing occurs when fishing activity reduces the stock to the point where it no longer produces enough eggs to regenerate itself each year (OSPAR 1993). This can be extremely difficult to diagnose due to natural variability. If not identified, however, stock collapse may result.

Fisheries scientists rely on time series of data which provide information on annual catch, fishing mortality rate, recruitment and biomass (Armstrong *et al.* 1989; Wilson *et al.*1994) and measures of fishing effort. These data are usually derived from the fishery landings themselves and then run through models from which the status of the fishery and future yields are estimated. The flawed rationale behind sustainable fishing on single stocks is well known. A stock reduced by fishing from a natural unexploited level will respond by improving growth, survival and reproduction as density dependent functions. This supposedly allows a sustainable yield to be taken from the stock as the natural increase in population. Where these density dependent functions reach their limit, stock collapse is likely to result. For many years, yield estimation has been based upon the "surplus yield" model to derive a "Maximum Sustainable Yield" (MSY) (Roberts 1997).This approach is still widely used and indeed is a term used in the most recent global assessment of fisheries (FAO 1996) in the terminology "maximum production". Calculation of this assumes that an equilibrium can be established between fishing effort and stock size. Its continued use as a concept has attracted much criticism (see: Kesteven 1996) since it was discredited over 20 years ago (Larkin 1977).

Management of single species of fish is, therefore, predicated upon detection of changes which are imposed by the fishing industry itself and subsequent modeled predictions rather than the natural population dynamics of the target species. The management of fisheries in a sustainable manner requires that information is available concerning the target population. Currently, the commercial fishery itself is generally the source of much of the data used in scientific studies. Gulland (1977) provides a useful review of the information used, and the procedures available, for processing such data. Virtual Population Analysis (VPA) began to be used extensively with the development of adequate computing facilities. It has evolved to include simplistic multi-species models (Serchuk *et al.* 1996; Rice & Gislason 1996). Some methods can theoretically be "tuned" to accommodate some uncertainties in the data. The output from these models is used in the assessment of the target stocks. The simple models based on MSY continue to a very large extent to be used in providing management advice. The basic assumptions behind the so called "surplus yield" models used have not altered despite the use of virtual population analysis (VPA) techniques (Gulland 1989) and the development of "age structured models" (Hilborn & Walters 1992). Indeed the various models suffer from similar basic limitations (Laloe 1995). Even though MSY has been discredited for some years and measures such as controlling fishing mortality rates have also met with very limited success, the single species approach remains intuitively attractive to decision makers. It suffers from a number of other key deficiencies (Daan 1989). These include the need for a great deal of biological information which means that in many cases the data are deficient. Despite this, it is probable that such models will continue to be used, although it is being recognised that their use may be restricted simply to descriptive exploration and simulation tools (Laloe 1995).

The predictive utility of a model can also be fatally affected by variability due to both biotic and abiotic factors. Stocks are highly variable. High levels of stock variation led eventually to a further high-risk assumption for modeling purposes that recruitment to stock was independent of stock size at least at low spawning stock biomass. Various failures in management on this basis have led to attempts to define thresholds to protect spawning stock (Myers *et al.* 1994). This is seen in an often applied "rule of thumb" that egg output should be between 20% and 30% of an unexploited stock by preserving a minimum spawning stock biomass to keep populations above Minimum Biologically Acceptable Levels (MBAL). This is regarded as a threshold below which decreased numbers of spawning fish can be expected to impair recruitment. In practice the relationship between spawning stock biomass (the spawning adults in the population) and recruitment remains elusive and this contributes markedly to difficulties in detecting recruitment overfishing.

The stock recruitment (SR) relationship is central to most models used to determine optimal fishing strategies and is widely assumed both in the scientific literature and in fisheries management policies (see Maury & Gascuel 1996). The stock recruitment paradigm involves the idea that the annual number of recruits into the population, at least at low spawning stock biomass (SSB), is positively related to the SSB. While analysis has shown that this may be true for salmonid populations (Gilbert 1997), in marine bony fishes it does not appear to be correct. Periods of low recruitment in marine fish may be environmentally induced and unavoidable, implying that current management strategies will need to change. Estimates of larval production and subsequent recruitment from larval abundance surveys are subject to considerable potential error (Koslow 1992). Attempts to link recruitment to environmental conditions over large areas have been unsuccessful (Myers *et al.* 1995a &b) and it was suggested that the original evidence for this was in fact fishery induced reduction in SSB in a number of populations around 1950. The processes regulating population size and its variability remain unknown for most fishes. This confounds attempts to predict recruitment and indicates the need to re-examine sustainability of fisheries when this is expressed partly in simple relationship to stock size. In short the SR may not be definable for many marine populations, or may have such wide confidence limits attached so as to be of little value for management purposes. The consequences of rejecting the SR paradigm have been characterised as an increased potential for allowing SSB to fall below a critical level and hence encourage recruitment overfishing. (Francis 1997). The corollary is unstated, but it follows that if there is no definable relationship then allowing management decisions based upon it is associated with similar risks. This is likely to become evident as an undue commercial weighting in favour of maintained intensive exploitation of a given stock in the absence of evidence indicating regulation.

In practice, there are a considerable number of other variables which theory requires to be taken into account in stock estimates and predictions (Ulltang 1996). Past failures in stock predictions in the mid 1980s, using survey data for north eastern arctic cod, can be attributed to a number of factors. These failures incidentally highlighted long term systematic errors in calculating SSBs. Food shortages, possibly partially provoked by changes in the Barents Sea ecosystem, but certainly due also to industrial fishing on capelin, increased the level of cannibalism amongst young cod. Capelin and shrimp stocks, the normal cod prey, were severely reduced at the time. Coupled with this, food shortages also decreased growth. Hence both recruitment and growth as model input parameter were overestimated, while increasing numbers of small cod were also discarded by the fishery but not accountedfor in the models. Although some indications of this problem with recruitment were available at the time, the survey data were used as predictors in the normal way, the shortfall being identified later by VPA. Growth in cod from this stock shows marked short term variation over 1-5 years. Growth is closely linked to maturation was severely reduced by reduced availability of capelin, the main prey of this stock. In addition, catchability of the fish increased above what would be expected from increased fishing effort alone, despite decreased stock sizes. Some evidence also exists that catchability of immature cod in the Icelandic stock is greater than mature fish of the same age (Thorarinsson & Johannesson 1997). This example illustrates an important wider truth, that without a sound understanding of the biological processes taking place within the population, all predictions are subject to substantial potential error.

Among the conclusions reached on the north eastern Arctic cod stock, for example, was that in order to understand fish population dynamics it is necessary to look persistently for the underlying mechanisms behind observations (Ulltang 1996). Other scientists have put it more bluntly and have noted that in the past decades fisheries science has lost touch with the realities of fisheries, management and ocean ecosystem dynamics. Unwarranted faith in population and bio-economic models on the part of a cadre of "keyboard ecologists" has placed fisheries science in the position where, lacking familiarity with nature, difficulties follow with the realistic interpretation of data (Rose 1997). Highlighting the fruitless debate between proponents of fisheries as the cause of the greatest change in abundance of fish (Hutchings & Myers 1994) and those highlighting environmental factors (Mann 1993), it is pointed out that combined factors can only be realistically investigated by excluding fishery effects at some sites. Models are criticised, too, for being unable to incorporate the complexity of ocean systems.

Single species fishery models have developed and been evolved to address populations with a relatively well understood population structure and whose reproductive biology is also relatively well understood. The uncertainties in fundamental population variables continue to severely compromise the predictive value of these models. In practice, attempts to control fishing mortality are likely to be confounded by inaccuracies and imprecision in the short term catch forecast which is a critical component in designating target fishing levels (Cook *et al.* 1991) . The mortality in catch discards may also affect fish populations as a whole (Van Beek *et al* 1990). If such figures are in error or subject to wide uncertainty then it is simply not possible to enforce a downward trend in fishing mortality in a reliable and effective way.

Other major limitations also remain unresolved. These include the response of other species in the ecosystem to fishing, natural levels of variation and thresholds at which changes become irreversible (see: Anon 1996). The models cannot take account of intra and interspecific interactions among target species during the recruitment phase. There are very few data on production of other components of the food web and the changes resulting from exploitation. Finally, the models do not account for direct side effects of fishing activities on benthic communities, the effects of other human interferences on the ecosystem or the effect of changing physico-chemical variables on fish recruitment and development (Daan 1989). In many ways, the failings can be traced back to derivation of the original models in the late 1950s. These early mathematical models were grounded in biology only to the extent that various terms in them were given biologically evocative names. The modern models which have developed from these early models have discarded the limited scientific principles originally incorporated into them. Ultimately this lead to an almost entirely mathematical treatment producing a well crafted statistical time series (Gauldie 1995). Regulations were emplaced on the assumption that the models would eventually work and when it was shown that they did not, the regulations were already in place. This damning treatment of fisheries models leads to a simple conclusion. In short, management systems need to be refitted around realistic science and around methods that are robust to scientific error and to the industrial and political mood.

4.3 Fisheries Management

Fisheries modeling and resultant biological advice on stock status have undoubtedly been the cornerstone of fisheries management. Indeed, biological advice is often the sole basis for management in the absence of input from other domains (Stephenson & Lane 1995). It is tempting, therefore, to point to the inadequacies in these models and the failure to adequately communicate uncertainty and errors in the data as being solely responsible for failures in management regimes. While part of the problem with over exploitation has been the models themselves, and in particular the general failure to attach confidence limits to their predictions, the management strategies and their implementation are also to blame (Kesteven 1996). The concept of the MSY coupled with the twin concepts of common property and open access were sufficient to cause problems even in the absence of the "hordes of modelers gifted in extrapolation in untested assumption". The collapse of the California sardine and the Peruvian anchoveta in the Pacific, together with the failure of the North Sea and Georges Bank herring stocks and the recent closure of Canadian demersal fisheries, serve as oft quoted examples of the wholesale failure of the current management paradigm. In fact, the failure arises as a result not only of failing to address issues of conservation in the form of overfishing, but also issues of economics in the form of overcapacity.

The progressive application of management instruments in the form of total allowable catches (TACs), individual transferable quotas (ITQs), extended economic exclusion zones (EEZs), property rights and limited fisheries access marked a move away from fisheries development *per se* in an attempt to address conservation and economics issues but did not prevent fisheries failures. These instruments have considerable limitations not least of which is the fact that they were largely predicated on the basis that the stock recruitment paradigm holds true and that models supply accurate predictions. Accordingly, fisheries management and regulation has proceeded on the basis of modifying fishing effort in relation to the findings of surveys and the prediction of stocks. The theory is simply that if a depleted or overfished stock is less heavily fished it will recover. In practice, fisheries management has been more preoccupied with tactics rather than holistic strategies. Underlying the regulatory instruments have been objectives which have been described as broad, ill-defined and not operationally feasible (Stephenson & Lane 1995). Where strategies have existed, they have been ignored at critical times. Overall, there has been an inability to implement social and economic objectives through the definition of goals and targets.

Both TACs and the more recent co-dependent instrument of ITQs both suffer from a number of problems. In the North Sea where TACs are used for regulation of fisheries it has become quite clear that these have failed since they were introduced in 1983 (IMM 1997a). An extensive evaluation of TAC management in North Sea flatfish fisheries has indicated that there have been problems at all levels ranging from the consistency of the scientific advice to the translation of this advice into congruent management decisions (Daan 1997). In addition, there have been problems with policing and enforcement of the catch limits. It was concluded that a TAC system cannot control fishing mortality because it controls landings not catches and does not address discards. TACs also require the co-operation of the fishing industry to avoid regulations being flouted. When regulations are flouted, this leads in turn to a deterioration of catch statistics used in the scientific assessments. A science based TAC system can, therefore, tend to undermine itself. In the absence of defined management objectives, it is difficult for scientists to produce convincing advice. This problem is compounded by final decisions being taken on political grounds rather than in deference to well defined objectives (Daan 1997).

The shortcomings of TACs as a management tool can be broadly translated to all fisheries managed on this basis. Recent developments have served to further undermine the concept. Currently, in many fisheries, including those in the North Sea advice given for the purposes of managing the fishery has changed to emphasise the Minimum Biologically Acceptable Level (MBAL) of spawning stock, moving away from the MSY concept. In fact this shift is ecologically even less defensible than MSY and appears in many cases to now be the *de facto* management objective. TACs remain the primary regulatory instrument and are translated into national catch quotas. The intention is to control the fishing mortality. It is generally recognised that TACs have failed in the North Sea. Simply, without direct controls on fishing mortality, fish can be caught in excess of quota and discarded or landed illegally. One important consequence of this is that the quality and reliability of catch statistics has deteriorated. Fundamentally (IMM 1997a), the failure of the TAC system has been caused by an imbalance between the exploitable resources and current fishing capacity. This has resulted in considerable socio-economic pressure, leading to the widespread misreporting and discarding of fish. These failures have also been a feature of fisheries elsewhere. The Canadian cod stocks were also regulated by a TAC system prior to their collapse in 1992, and similar deterioration of catch statistics preceded this collapse (Maguire 1997). A common problem has been a consistent tendency on the part of modelers to underestimate fishing mortalities. A key problem is the failure to attach confidence limits to the stock predictions and the general low reliability of these forecasts which determine the TACs which are set (Biais 1995).

Individual transferable quotas ITQs have also attracted criticism as a means of limiting fishing mortality despite being widely championed (Davis 1996). ITQs are associated with achievement of long sought management goals such as resource conservation, economic efficiency and hence fisheries sustainability. Essentially ITQs privatise the right to fish and depend for effectiveness upon several assumptions regarding human economic behaviour. By allocating individual resource use rights and explicit quotas (expressed as TACs) in theory discipline engendered by self interest will prevent over-exploitation. This, however fails to recognise the considerable disparities between the structure of industrial fishing activities and the "livelihood " sector. In particular, the conformation of the industrial sector to predictions of operational characteristics driven by notions of open access as opposed to the more co-operative and participative characteristics of small enterprises has been highlighted. The industrial sector is the most highly capitalised of the two, more corporately driven and measures its results predominantly in terms relevant to financial markets. By contrast, livelihood fisheries support far larger numbers of people and sustains the majority of fishing communities. Hence to equate these two disparate sectors is likely to result in inequitable impacts on the greater number of people involved in livelihood sector.

Some evidence of inequity is furnished by experiences in the New Zealand sector where an ITQ approach was adopted in 1986 in 32 commercial fisheries. A year afterwards the three largest fishing companies held title to 43% of the ITQs. Within five years the same companies held title to 50% of the ITQs in the system (McGinn 1998). Many small vessel based operations either subcontracted to larger companies or were eliminated from the industry. Concentration of the rights to exploit fish populations results in any case when ITQs are allocated on the basis of the proportion of the resource extracted by operators prior to adoption of the system. Similar criticisms have been leveled at the Icelandic ITQ system which was established from 1975 onwards (Arnason 1996). While the impacts of ITQ systems on the structure of the industry have been clear in terms of increased efficiency in some areas (Annala 1996; Arnason 1996) by tending to drive capacity downwards, the benefits to the stocks have been variable. Herring and capelin fisheries in Iceland improved but demersal fisheries have not been rebuilt under ITQ arrangements. In Icelandic demersal fisheries, however, the concept has been constantly revised since its inception making evaluation difficult.. Evidence for improvement in stocks in New Zealand, however, has also been equivocal. Other data obtained in a survey by FAO suggests that of 31 ITQ managed fisheries, 24 kept catches below the TAC but for several of these the TAC was set too high to allow sustainable exploitation (McGinn 1994). The utility and efficacy of ITQs relies ultimately on the precision with which TACs can be set relative to stock levels, and hence upon the accuracy of these predictions. While an ITQ system may alleviate some pressure upon fish stocks, the evidence that they allow resource rebuilding is not convincing from the current data. If they address economic considerations in a naive way, they can exert significant negative effects in socio-political terms.

Failures of the TAC/ITQ approach stem from the thresholds used to determine them. The greatest concerns stem from the use of MBAL as a management goal. The logic is simply that if a stock is deemed to be safe from a failure of recruitment, then the available stock can be managed with socio-economic considerations factored in. The use of MBAL is compromised the fact that the values are set at generally very low levels relative to an unexploited stock and cannot confer highly certain protection. If the MBAL is to be used as an action threshold and catches regulated by TACs, even if these are updated annually, it is necessary to manage stocks so that they remain well above the MBAL (Marchal & Horwood 1995). This may be some seven times the MBAL value to avoid the possibility that this critical threshold is actually reached, using the example of Celtic Sea cod. As such, adopting (MBAL) as a management objective, which is what seems to have happened in many cases, represents a worsening of situations where MSY was the goal. The effect in the North Sea has been to allow stocks to fall towards the MBAL reference point rather than management reference points based upon fishing mortality (Marchal & Horwood 1995). Rather than a limit to be targeted as a management goal, many scientists have pointed out that this is a critical point at which serious remedial action needs to be taken to achieve more robust limit targets and allow the stocks to rebuild as quickly as possible.

The inadequacies of current management systems, derived from a variety of interacting factors can be illustrated by fisheries which have collapsed even where a good body of data exists as well as by populations that have been overexploited in the absence of good fisheries data. Even where supposedly good levels of understanding exist, supported by extensive data, this has not prevented the collapse of stocks such as the cod stock in Canada in 1991-1992 (Hutchings & Myers 1994) or the current poor status of herring stocks in the North Sea (Toresen 1997). In addition, North Sea cod for which a relatively good data set also exists is also thought to be in potential danger of collapse at current exploitation levels (Cook *et al*. 1997). In the Canadian example, notwithstanding the relatively good data set, erroneous stock growth predictions were made while the actual stock size was overestimated. In the case of North Sea stocks the decline seems to be attributable to failure to implement robust management practices in the face of consistent scientific advice that fishing mortality should be substantially reduced. The example of Southern Bluefin Tuna in the Southern Oceans is an example of a highly migratory stock, fished by a number of nations, poorly regulated and in a critical state.

a) The North Sea Fish Stocks

Despite the identified shortcomings of the current models, the major management tool in the North East Atlantic fisheries generally, remains the Total Allowable Catch (TAC) (ICES 1993). In recent years, a TAC has been set for North Sea stocks which, in theory, represents a modest reduction in fishing mortality rate for the year. The North Sea is regarded as one of the world's richest fishing grounds. It is a relatively shallow temperate shelf area with a diversity of habitats and substrates. The generally shallow, well mixed, water is flushed with nutrient rich oceanic waters supplemented with riverine inputs. It therefore supports substantial fishery resources (McIntyre 1988) and one of the most intensively studied areas of sea in the world (Kinne 1995). Fisheries data have been collected in a relatively systematic way since the beginning of the century. At present up to 50% of the biomass of commercially exploited fish species are taken each year. In the case of some species such as cod up to 70% of the two year old fish are removed in any given year, 20% of these by natural mortality. The mortality figures do not reflect discards except in the case of haddock and whiting. Total catches are approximately 3 million tonnes per year of a total fish biomass (including non-commercial species) estimated to be around 12 million tonnes (OSPAR 1993). It has been estimated, however, that unreported landings may range from 10% of the reported catch for cod rising to 40% for haddock and 60% for sole (Cook 1997) potentially compromising catch estimates and the reliability of modeled predictions.

Historically, fish population crashes have been reported from the North Sea for important pelagic fish species. The collapse of herring and mackerel stocks in the North Sea both resulted from overfishing. International catches of herring peaked at around 1.2 million tonnes in 1965 when fishing intensity was such that about 50% of the adult stock was taken. Fishing intensity increased to 60% over subsequent years, spawning biomass and recruitment declined steadily. Spawning biomass fell from 2 million tonnes in the early 1960's to around 0.13 million tonnes in 1975 and the fishery was closed in 1977 (Armstrong *et al.* 1989) This followed redeployment of the powerful Norwegian purse-seining fleet after collapse of the Atlanto-Scandian herring fishery in 1964 (McIntyre 1988). The problem of protecting these stocks is complicated by the existence of several component stocks which have not all recovered at the same rate (see: Cushing 1992; Toresen 1997). The stocks recovered through the 1980s and the fishery was reopened in 1982 but subsequently collapsed to current low levels with a spawning stock biomass of 500000 tonnes following failure to heed scientific advice between 1991 and 1995 declining to below safe biological limits after 1992. Mackerel, the other major pelagic species in the North Sea suffered a similar fate to the herring. They became the major target of the purse seine fishery after the collapse of the herring stocks. Landings exceeded 100,000 tonnes for the first time in 1965 and peaked at 0.9 million tonnes in 1967 (McIntyre 1988) at which point it is estimated that around 50% of the spawning biomass were caught. Decline of the stock followed rapidly. Recruitment has been consistently poor since the good year of 1969 and populations have not recovered subsequently, remaining outside safe biological limits.

Currently, (Serchuk *et al.* 1996) of the major roundfish stocks (cod, haddock, whiting and saithe) all are intensively or over exploited and as a result the size of the stocks and catches have become highly dependent on young fish. With all four stocks, large quantities of juvenile fish are landed or discarded . The spawning stock biomass has declined to very low levels, threatening future recruitment into the fisheries. With the exception of whiting, all the stocks have recently been or continue to be outside of safe biological limits, where the spawning stock biomass is so low as to potentially compromise reproductive success. This has been followed by warnings that cod stocks are in imminent danger of collapse (Cook *et al* 1997). This situation is highly unlikely to improve without curbs on fishing effort and associated fishing mortality. Total Allowable Catches, since they constrain only landings and not actual catches, appear to be an ineffective regulatory instrument. Moreover, landings have often exceeded agreed TACs and widespread discarding of catches has taken place in these essentially mixed fisheries when TACs were reached. Since 1993, the spawning stock biomass of plaice has been outside minimum biologically acceptable limits owing to large increases in catches (see: Rijnsdorp & Millner 1996). It was considered possible that sole could fall below this level after the 1987 and 1991 year classes passed through the stock.

The most recent ICES assessments have identified problems with a number of stocks in the areas covered. North Sea sole are, indeed, now considered to be outside safe biological limits and a reduction in fishing mortality of 25% over 1996 levels has been recommended (ICES 1997). Celtic Sea sole have also fallen below this limit. Cod in the North Sea, Eastern English Channel and the Skagerrak are close to or outside safe biological limits, as are cod in the Western English Channel, Celtic Sea and West of Scotland. The North Sea component of the mackerel stock is outside safe biological limits while SSB has fallen in the Western component and this fishery should be highly restricted. North Sea, Skaggerrak and Celtic Sea plaice are close to safe biological limits while the population in the Eastern English Channel has fallen below this level together with the North Sea stock . The status of the North Sea and E. Channel whiting stocks are currently uncertain but those in the Celtic Sea and Western Channel are outside safe limits. Similarly, saithe stocks in the North Sea , Skagerrak, West of Scotland/Rockall have fallen below SBL. In the case of industrial species, sandeel stocks in the Scottish fishery decreased dramatically after 1982 and the fishery was closed from 1991 to 1994 (IMM 1997a).

Elsewhere in the ICES area, Barents Sea capelin spawning stock is well below the threshold level of 500,000t after collapsing over the period 1991-1994 and still cannot sustain fishery activity in 1998. This fishery fell from 7 million tonnes in 1980 to 100,000 tonnes in 1987. It rose again to 7 million tonnes between 1989 and 1991, falling to just under 200,000 tonnes in 1994 (Barret & Krasnov 1996). Redfish (*Sebastes mentella)* in Norwegian waters have fallen below safe biological levels as have some Baltic and Greenland stocks. Icelandic cod SSB has shown a declining trend since 1955, reaching record lows in the late 1980s and is still at a relatively low level. Southern shelf hake populations are considered to be outside safe biological limits.

There is little doubt that the root cause of the declines observed in the North Sea and in many other fisheries is overfishing. A great deal of research has been directed at the problems caused by this and indeed it has now been widely recognised that unless exploitation of stocks is better controlled, sea fishing is unlikely to prove sustainable (BGPSD 1995). A recent UK Government Report states "Most stocks in the waters of EC member states are over exploited and some stocks are at historically low levels" (DoE 1995). Key to this is the recognition that many North Sea fisheries are mixed-species fisheries and this requires that management takes account of technical interactions between them (Lewy & Vinthner 1994). The industrial fisheries, particularly, which exploit sand-eel and Norway pout are very poorly managed even though these species are important components of North Sea ecosystems.

The focus of fisheries management upon single species of commercial importance has meant that for the most part, the role of fish in the ecosystem as a whole is poorly understood. Indeed, Daan (1989) points out that the major effect of EC Fishery Policy introduced in 1983 has been a deterioration in the quality of catch statistics which further undermines the scientific evaluation of the status of existing stocks. It has been suggested that that if stocks continue to be poorly managed then larger species such as cod, haddock and sole will disappear from the North Sea. Crucially, the ICES (1993) report stated that despite limits on total allowable catch, desired reductions in fishing mortality had not been achieved and the focus has now been shifted towards a reduction in fishing effort. Even so, fishing mortalities have not declined significantly and are actually increasing for some stocks, implying that alternative, more rigorous management strategies are required. In recent years, failure to follow scientific advice in formulating management approaches has been largely responsible for problems in the North Sea fisheries as exemplified by recent developments in the North Sea herring stock (Toresen 1997).

b) Canadian Cod Stocks

Atlantic cod have been fished in the waters of Newfoundland since the late 15th century and the fishery itself played an important role in historical terms. Like the North Sea, oceanographic conditions work together to make areas such as the Georges Bank on of the world's most productive shelf ecosystems (Franks and Chen 1996). The trade which sprang up in salted products of the industry working both the Newfoundland and more southerly waters over the shallow banks extending as far south as Massachusetts played a significant part in the economic development of Eastern North America, exporting fish to Europe, Africa and the Caribbean (Kurlansky 1998) and trading in indigo, cotton, tobacco and sugar. The Canadian stocks formed part of the Atlantic cod stocks taken off Iceland and Norway and which are supported by primary and secondary production taking place at fronts where warm currents meet cold currents flowing from the north. The fishery for the northern cod serves as a regional example of the global trend in fisheries outlined above and was the most severely impacted of the cod populations which collapsed in Canada.

Originally, in the northern cod fishery, fish were taken with hook and line until cod traps were introduced in the late 1860s although longlining still persists (Kenchington 1996). Bottom Trawlers were introduced in the early 1900s and nylon gill nets followed in the 1960s. As a result of technological development landings rose from 100,000 tonnes to 150,000 tonnes between 1805 and 1850. and from 1900-1960 were frequently above 250,000 tonnes. Long distance factory freezer ships were introduced from Europe in the 1950s and catches continued to rise from 360,000 tonnes in 1959 to a historical maximum of 810,000 tonnes in 1968. By the time that Canada extended its jurisdictional zone to 200 miles, preventing most long distance fishing, landings had fallen to around 140,000 tonnes. At the same time, the domestic industry had expanded in anticipation of increased resource availability following from the exclusion of foreign boats. The landing figures increased under Canadian management to 270,000 tonnes in 1991. The following year the fishery was closed (Hutchings & Myers 1994). Over the period from the early 1960s onwards the spawning stock biomass was reduced from around 1.6 million tonnes to 22,000 tonnes in 1992. Overall, this corresponded to around 3 billion fewer fishable cod in 1992 than thirty years earlier. This collapse mirrored developments in other fisheries. In addition to the collapse of the six Canadian stocks, the cod stocks of the Eastern USA are at unprecedented low levels (see: Myers *et al.* 1996). In 1994 Iceland reduced its cod catches by 50% in one year and the stock is currently severely depleted (Baldursson *et al.* 1996; Thorarinsson & Johannesson 1997).

There were severe economic impacts as a result of the collapse in Canada. Some 30,000 fishermen and fish plant workers were put out of work when a moratorium on fishing the cod stocks was announced in 1992 (Kurlansky 1998). Demographic changes are occurring as businesses fail in the aftermath of the collapse with around 7,500 people a year leaving a province with an annual economic deficit of some $700 million. Central government has committed to a $2 billion social adjustment programme over five years (McGinn 1998). In the analysis of the collapse, a fierce debate which became acrimonious (see: Doubleday *et al.* 1997) has raged as to whether the collapse was caused by excessive overfishing or by excessive natural mortality, or indeed other environmental factors (Hutchings 1996).

Despite the highly developed management system, important aspects of the life-cycle of the Canadian stocks, relevant to their management were relatively poorly researched at the time of the collapse, including migration pathways and spawning sites (Rose 1993). Northern cod, like some other populations, show a seasonal migration pattern moving inshore during summer and offshore in the autumn to overwinter and spawn over the continental shelf although some adults remain in the deeper bays. The nearshore zone is considered the nursery ground for juveniles and summer feeding grounds for adults (Wroblewski *et al.*1994; Wroblewski *et al. 1995;* Godo 1995). There is some evidence that the pattern of migrations has changed from those observed prior to the collapse (Kulka *et al.* 1995). It is possible that the distribution of the stocks could permanently change due to the loss of older individuals from the population. A hypothesis has been put forward that migration routes may be partially learned by juveniles from older specimens and that loss of these older fish could change migration and spawning patterns (Rose 1993). This will obviously impact upon regeneration of the stocks.

The simultaneous decline of cod and plaice together with other groundfish populations in the area has been cited as evidence of environmental factors at play, but plaice occur in the same areas as cod and may have been discarded in substantial quantities by the fishery thus accounting for this phenomenon. Similarly, poor recruitment was proposed, but this was not backed up by reanalysis of the data. Seal predation could be ruled out as a significant factor. Increased natural mortality could not be substantiated as a contributor (Myers & Cadigan 1995). Other workers suggested that low sea temperatures and low salinity could be reducing food resources (Conover *et al.*1995), but this too can be rejected on statistical grounds. On the other hand, significant misreporting of catches, particularly of discards of undersize cod, was established as a contributor to the overall problem. Estimates of recruitment produced by VPA declined at a greater rate than those estimated by survey implying that pre-reproductive cod were being discarded and hence not included in the data used in VPA. Further support for overfishing as a sole cause was provided by analysis of tagging data (Myers *et al.* 1996). This showed a high fishing mortality in the late 1980s. The following scenario fits the known facts. Recruitment was not below average for the fish that should have contributed to the fishery at the time of the collapse. High fishing mortality was possible because population abundance was overestimated. Fishing mortality was underestimated and as a result quotas were set too high. Discarding of juveniles increased thereby reducing numbers of fish entering the fishery. Fishing mortality was kept high as populations declined through fleet overcapacity. The populations fell to the level of commercial extinction and low SSB has inhibited recovery. Significantly, a similar pattern can be established for some 20 depleted cod stocks in the North Atlantic (Myers *et al.* 1996).

The Canadian cod stock collapse, therefore, took place against a background of elaborate fisheries management relying on detailed scientific studies to establish TACs. There was a wide scale consultation with the fishing industry with provincial authorities allocating the TACs between the fishing interests. A well trained and equipped monitoring control and enforcement system existed. In short, all of the ingredients of a successful management process were in place (Maguire 1997). It has been argued that the conflictual relationship between the commercial interests of the industry and the bureaucratic, scientific fishery management process was responsible. Certainly, there was a failure to adequately regulate the national fishing fleet after exclusion of foreign boats from the 200 mile EEZ. This scenario was reproduced in the United States after limits were extended and regulation formulated under the 1976 Magnuson Act (McHugh & Hasbrouck 1990). TACs were regularly exceeded through dumping, discarding and various forms of misreporting. The quality of the data used in stock assessments fell, followed by the credibility of the assessments themselves. The collapse has also raised the question of whether the fishery will actually recover. To date there has been little evidence that stocks are actually rebuilding. In fact biomass levels of the northern cod stock are predicted to continue to fall due to foreign fleet activity outside the EEZ. It has been predicted that this population will only regenerate slowly at scales of decades since they represent remnant populations requiring generations and warmer conditions to rebuild. Genetic diversity may have been irreversibly lost. The southern stocks should rebuild more quickly given warmer conditions.

c) Southern Bluefin Tuna

Southern Bluefin Tuna [SBT] (*Thunnus maccoyii*) grow to 200cm in length and reach a weight of 200kg. Living in excess of 20 years (the most recent estimates suggest 40 years), these highly migratory fish are the target of an extensive fishery in the oceans of the southern hemisphere. The species has a general circumglobal distribution between 30S and 50S. There is one known spawning ground between 7S and 20S in the north eastern Indian Ocean south of Java. Some of the known aspects of the natural history of bluefin tuna are summarised by Caton (1994). Recent research has been unable to provide evidence of stock structuring in this species (Proctor *et al.* 1995) implying that SBT are comprised of a single population over the whole of their range.

Eggs and larvae of Southern Bluefin Tuna are assumed to drift south from the spawning ground with the Leeuwin Current. The fish reach the southwest corner of Australia as young-of –the-year juveniles. They then migrate in large schools eastwards along the temperate shelf. The juveniles are thought to spend between four and six years in the temperate region moving towards south-east Australia. Young adults subsequently move south into Sub-Antarctic waters. Fish of reproductive age (thought to be around 8 years) migrate to the Indian Ocean spawning area between September and March. Duration of spawning and individual spawning frequency are not known (Proctor *et al.* 1995; Caton 1994; Jenkins *et al* 1991).

The surface schools of fish found in Australian waters have been targeted for many years. They are captured principally by pole and line and purse seine gears. Since 1990, however, activity has shifted from surface operations to longlining activity, usually in joint ventures with Japanese fleets, catching fish up to eight years old. In addition, Japanese longlining has targeted adult fish over the whole of their range since 1952. This fishery takes adults from eight years of age to in excess of twenty years of age. The domestic Zealand industry was originally based on troll and handline fisheries but diversified into small scale longlining and subsequently included activity by four or five chartered Japanese vessels from 1989 onwards although this has now ended. This fishery is the smallest and targets mostly large adults (Caton 1994).

The Southern Bluefin Tuna fishery is extremely valuable. Top quality fish, in common with other species of tuna, are prized for sashimi in Japan (Bartlett & Davidson 1991) and a single specimen can be worth over $30,000 (Ward *et al.* 1995). The total value of the fishery in 1994-1995 was estimated to be in the region of $500 million on an agreed catch quota of 11,750 tonnes emplaced in 1989-90. Actual catches over this period were 11,523 tonnes (McLoughlin *et al.*1996) with Australia using a system of ITQs and New Zealand and Japan following a system of TACs. Quotas have been imposed as a result of declining catches. Total catch tonnages fell from around 80,000 tonnes in 1960 to less than 14,000 tonnes in 1991 (see: Caton 1994; Ward *et al.* 1995). The spawning stock is considered to be at an historically and critically low level at around 6-11% of its 1960 size. Serious concerns now exist about the long-term sustainability of the fishery. Coupled with the fact that fishery activity largely takes place in waters outside specific national jurisdictions, this led to agreed quotas being set for the fishery. Initially, these quotas were set informally between Australia, New Zealand and Japan but agreements were formalised in 1994 with the ratification of the Convention for the Conservation of Southern Bluefin Tuna (CCSBT). A total allowable catch was set for member countries of 11,750t. This was originally applied informally in 1989. In addition to developing and implementing management strategies agreed by the original three signatory nations, the Commission is also working to involve non-signatory nations active in the exploitation of the fishery. The stated objective of the formalised trilateral arrangement is to set catch limits which will allow the stocks to rebuild. This is a standard aim in the management of depleted stocks.

In addition to the Australian, New Zealand and Japanese fisheries, significant landings are also made by Taiwanese vessels specifically longlining for adult tuna. By-caught juveniles are landed by vessels fishing for albacore. Korean and Indonesian longliners also take southern bluefin tuna. Anecdotal evidence, coupled with known imports to Japan suggest that the 1991 Taiwanese catch alone was in the order of 1300 tonnes while that of Indonesia could have been around 250 tonnes (Caton 1994Declared Indonesian figures for 1995 were 724 tonnes of which an estimated 361 tonnes were exported to Japan. Japanese import figures showed 221 tonnes imported from this source (Anon 1996a). This same source notes that details of catches by Taiwan and Korea are poorly known). Most recent estimated figures (CCSBT 1997) suggest that the total non-CCSBT party catch was between 4136 and 4937t. It is not thought that southern bluefin tuna comprise a significant component of Pacific Island indigenous fisheries (see: Dalzell *et al.* 1996).

The depletion of the Southern Bluefin Tuna stock is largely attributable to protracted periods of heavy fishing pressure. In particular, the Japanese longlining fleet during the 1960s and 1970s in fishing down parental stocks and the expansion of the Australian fishery for juveniles in the 1970s and 1980s are considered to be responsible (Murray and Burgess 1991). Caton (1994) states that since the 1960s the most important area targeted by the fishery has been the area south of Africa. In the longline fisheries global effort was in the region of 100-125 million set hooks annually between 1980 and 1990 when it declined to 63 million hooks.

The situation with respect to Southern Bluefin Tuna is markedly different to that pertaining to the North Sea or the Canadian Grand Banks. In these fisheries the great majority of the catch from managed fish stocks is landed at relatively few large fishing ports. In these cases the basic informational requirements for Virtual Population Analysis (VPA) are theory comparatively easy to fulfil. In the case of SBT, the basic data are demonstrably deficient. This calls into question the robustness of the VPA techniques applied to the population. The total catch statistics for SBT are not accurately known and this in turn will flaw estimates of the fishing effort (as catch per unit effort) and estimates of the size and age structure of the catches. This is particularly problematical for SBT since different age classes are targeted by different fisheries. This requires the collation of data from all fisheries in order to estimate population parameters. As noted earlier, data are almost entirely absent from some sectors of the fishery. That this may actually be a significant problem is hinted at by data supplied by the Indonesian SBT industry. This suggested that the fish taken were largely mature, up to 35 years old and in excess of 170cm fork length (Anon 1996a). No reliable data are available for the Taiwanese or Korean fisheries which target a wide range of tuna species (Haward & Bergin 1996), but if the Indonesian pattern applies, then current and future estimates of the age structure of the population are likely to be imprecise. Current estimates (McLoughlin *et al.* 1996) suggest that catches by non-CCSBT nations amount to around 2000 tonnes *per annum*.

More significantly, the Indonesian data were from fish taken on the spawning ground and could also throw into doubt the accuracy of age at reproductive maturity as noted in Anon (1996a). Again this has significant implications in the assessment of the spawning biomass of the population. In mitigation of these defects in the data it is true that the greatest proportion of the SBT catch is taken by the three signatory nations to the CCSBT. The limitations nonetheless imposed by incomplete total catch data must be recognised as of great potential significance particularly in this intensively exploited fishery.

In longlining operations effort is generally calculated by referring to the number of hooks set (see: Rothschild & Suda 1977). The distribution of effort in the Japanese longlining fishery has shown considerable change in the period between 1960 and 1990. After an initial and rapid expansion of effort over most of the Southern Ocean between 1960 and 1969, progressive focussing of effort has taken place since 1975 into the South African and Australian sectors of the fishery (see: Weimerskirch *et al.* 1997) reflecting progressive falls in catch per unit effort (CPUE) over the whole range of the fishery. Refocusing of effort has helped to confound stock estimates by reducing the amount of data gathered from areas no longer actively fished. (McLoughlin *et al.* 1996. In the New Zealand sector CPUE figures are 10% or less than those recorded before the decline of the fishery started (Murray & Burgess 1991). Indeed, it was concluded that the recorded changes in nominal CPUE for SBT exhibit remarkably linear declines and together with proximity to zero catch rates in most fished areas gives little cause for optimism either about data quality or the long term prospects for the fishery.

Nominal CPUE based upon hook deployment by sector undoubtedly reflects the overall decline in the fisheries and the progressive refocusing of effort. Some concerns, however, have recently been raised (Anon 1996a) concerning the use of CPUE in resolving population dynamics. These arise in part from changes in retention practices by longliners. The Japanese currently discard fish less than 25kg in weight corresponding to fish up to four years of age. The survival of non-retained fish is not known but may not be high. Accordingly, a potentially significant proportion of juveniles may be removed from the population without being recorded. The problem is magnified by the fact that retention practices over the whole fishery are poorly known. Difficulties have been encountered in standardising CPUE data for the SBT fishery to poorly understood retention practices, while different interpretations have given different pictures of the stock. Interpretation of CPUE data remains as a major source of uncertainty in Virtual Population Analyses conducted for the stock. Clearly, there is a need to arrive at a robust interpretation of CPUE before stock assessments can be regarded as valid.

In addition to difficulties standardising the CPUE data resulting from different retention practices within the fishery, size and age structure of tuna populations are also difficult to determine. This has been indicated as a high priority research area (Anon 1994a) by the CCSBT Scientific Committee. Difficulties in resolving basic tuna population structures arise from their large size, migration over large distances and the value as landed catch. Together, these factors have combined to limit accessibility to specimens. Even novel methods based upon examination of otoliths have provided, at best, ambiguous information (Proctor *et al.* 1995) concerning stock structure and larval growth and development (Jenkins & Davis 1996).

The models applied to SBT population and stock analyses have been largely based upon cohort analysis (VPA) and other age structured models which require as part of the input an estimate of the factor M, the instantaneous rate of natural mortality. A variety of estimated values have been used but the most common has been a value of 0.2/year. This figure has also been arrived at by analysis of tagging experiment data (Hampton 1991) with a higher estimated value of 0.4/year. Values reported by CCSBT Scientific Committee (Anon 1996a) suggest higher mortality in the one year old group than in the 2-3 year old group. Estimates ranged up to 0.49/year for the 1 year old group. The Committee noted that preliminary tagging of juvenile fish and direct ageing of older ones suggested a wide range of interpretations of how natural mortality changes with age. The lack of a robust, validated figure for M represents a serious obstacle to the derivation of a reliable population analysis. There is also a lack of validated catch at age data. Polachek *et al.* (1995) note that for SBT no direct age estimates are available, but estimates of the size/weight distribution are recorded. Anon (1996) however notes that inconsistencies exist in size composition data recorded by different fleets fishing the spawning grounds, compromising the precision of size/weight indices. Kalish *et al* (1996) note that age estimation for SBT is complicated by difficulties in interpreting marks on otoliths and vertebrae used in age analyses. Further problems accrue from the mobility of the species and uncertainties about longevity.

Otolith analysis has suggested ages well in excess of the accepted life of 20 years but these have not been validated by tag recapture data. Kalish *et al.* (1996) suggest on the basis of their analysis that a large percentage of fish greater than 1.8m fork length are at least 20 years old and that longevity may be in excess of 30 years. Estimates up to 40 years have been suggests (McLoughlin *et al.* 1996). Significantly, this work suggested that it is not feasible to estimate age from simple length or weight data for fish over 1.8m fork length fact hinted at by Caton (1994). This has significant consequences for the precision of data produced by current reporting activities. In short the problems noted by Rothschild & Suda (1977) namely that, ageing by length may be confounded by a continuum of ages despite the appearance of modal groups in the fishery, have not been fully resolved by the development of alternative methods for the SBT fishery.

A further confounding factor is the possibility that Northern Bluefin Tuna (NBT) *Thunnus thynnus orientalis* are taken with Southern Bluefin stocks. The two species are morphologically similar and may be misidentified (Ward *et al.* 1995). In the Australian fishery specimens of Northern Bluefin are treated as a part of the Southern Bluefin quota. Smith *et al.* note that in the New Zealand fishery in 1989 6.7% of the catch was recorded as Northern Bluefin falling to 1.4% in 1991 and 2.0% in 1992. It is possible, of course, that these figures are the product of misidentification. Nonetheless, the two species may overlap in their ranges off South Africa, Venezuela and Brazil (see: Scott *et al.* 1993). Given the possibility of misidentifying Northern Bluefin, it is also possible that inclusion of this species in SBT catches could be corrupting the data used for SBT stock assessments. This possibility does not appear to have been fully examined by fishery scientists and deserves further scrutiny. One possibility is that substantial numbers of misidentified large NBT could bias estimates of parental biomass of SBT if taken in the same fishery.

It can be concluded that estimates of the size and age structure of the SBT population are subject to a number of substantial uncertainties. These have implications for Virtual Population Analyses conducted using such data. Indeed, these factors have been noted by the CCSBT Scientific Committee and particular concerns have been raised relevant to the catch at age matrix. Growth overfishing occurs when the fishing mortality exerted upon a population is beyond the point of maximum yield per recruit to the population (Gulland 1977; OSPAR 1993). This has undoubtedly occurred in the past in the SBT fishery. This is distinguished from recruitment overfishing which occurs when the year classes fall well below recruitment occurring when adult stock is high (see: Serchuk *et al.* 1996). It is clear that recruitment overfishing of SBT stocks is no occurring Little is known of the reproductive biology of SBT (Caton 1994). Recent studies have investigated various aspects of larval growth (Davis *et al.* 1990a&b; Jenkins *et al.*1991) and aggregation of larvae and adults at eddy and thermal fronts (Reddy *et al.* 1995; Young *et al.* 1996; Moser & Smith 1993). Accordingly, it is not clear to what extent larval abundance has been affected by reduction of the spawning stock biomass, nor to what extent this has contributed to poor recruitment.

As noted above, for some stocks in the North Sea fishery it has been possible to define a Minimum Biologically Acceptable Level (MBAL) of spawning stock although this estimate is subject to the same potential errors as other components of stock assessments. They are useful, nonetheless as a yardstick for assessment of the condition of the stock and as a regulatory parameter. This is the level of spawning stock below which the probability of poor recruitment increases as the spawning stock size decreases. In other words, the stocks may be in danger of severe depletion if not allowed to rebuild as quickly as possible. This in turn is likely to lead to eventual collapse of the stock, roughly defined as a decrease to 5% of the unexploited stock size (IMM 1997). In these cases the recommendation is clear: The MBAL should not be regarded as a target but as an indicator of an emergency situation, requiring immediate action with a long term goal of restoring stocks to beyond the MBAL to reach a maximum sustainable yield. In cases such as cod populations where problems have been identified recommendations to reduce fishing effort by 70% or fishing mortality by 30% have been made (see: Cook 1997) to protect future recruitment.

A range of VPAs have been conducted on SBT in connection with the work of the CCSBT Scientific Committee (see: Polachek *et al.* 1995; Klaer *et al.* 1995) These have investigated the sensitivity of the analyses and have focussed on uncertainties in the data inputs. The major areas of uncertainty identified included interpretation of CPUE data, estimates of natural mortality, the reliability placed on estimates of fishing mortality derived from recent tagging experiments, the relationship between fishing mortality and fishing effort and the methods used to estimate the abundance of fish greater than 12 years of age. As noted above, the catch at age matrix attracted particular concerns.

Hence a variety of outputs have been derived from VPAs depending upon the values used to tune the analyses, in turn gathered predominantly from catch data. Sensitivity tests using the same values for M and CPUE showed that different models can yield appreciably different estimates of trends in parental biomass. The discrepancies are highlighted by the differences in estimates of the possibility that the stock will recover to 1980 parental biomass levels by the year 2020 using an agreed range of uncertainties and factor weightings. Australia arrived at a probability of 15%, New Zealand 29%, Japan 79% while scientists external to the Committee estimated a probability of 69%. As noted by the Committee, the single largest source of uncertainty in these estimates of the probability of recovery was the current process of assigning the weights to different options in the range of uncertainties. The specification of the weighting applied to such factors will remain a major source of uncertainty until an agreed and objective method of determining weights is developed. In fact, disagreement on stock estimates between CCSBT signatories has been a recurring theme of negotiations conducted in this forum.

The species continues to show evidence of decline characteristic of recruitment overfishing. The CCSBT Scientific Committee (Anon 1996a) notes that the trend in recruitment to the fishery with time estimated by VPA is uncertain, with significant differences among VPAs for estimated absolute recruitment levels. Parental biomass estimates varied between 25 and 39% of 1980 levels and between 5% and 8% of 1960 levels. There has been an increased catch of 6-8 year old fish in recent years indicating that the stock may be rebuilding in sequential terms. The magnitude of the rebuilding is highly uncertain however and dependent upon interpreted CPUE data as a measure of abundance. The catch rate for 3-6 year old fish in high seas areas seems to have continued to decline as have catches of the youngest cohorts in the Tasmanian winter fishery and in New Zealand waters. Overall (Anon 1996b) predictions of parental stock recovery appear to have been overly optimistic in that they have tended to overestimate parental biomass changes by a factor of 10-60% for the following two years. As noted by Anon (1996b) the very high level of uncertainty in the assessments (and other factors) means that clear scientific conclusions about stock status will not be possible until after the population is well into recovery or collapse.

As in the case of North Sea roundfish subject to high degrees of exploitation SBT catches are now highly dependent upon recruiting year classes. Available data for these year classes provide a highly equivocal picture of the future of the stock. Declines have been recorded in most year classes over 1995-1996 (CCSBT 1996a) following from a number of years of poor recruitment (Polachek *et al.* 1995). These declines are thought to be due to continued declines in spawning stock, which at the current low level could still result in a recruitment collapse. Overall, it is evident from the data, however, that the 1995 parental biomass is considerably lower than the 1980 level. As noted by the CCSBT Scientific Committee (CCSBT 1996a) "The 1980 level of parental biomass corresponds to commonly used thresholds for biologically safe parental biomass. Below that threshold, the risk of poor recruitment is expected to increase while the stock becomes more likely to experience abrupt and unpredictable recruitment declines". On the basis of this statement and the fact that VPA data suggest a decline to around 5-8% of 1960 values for spawning stock biomass the stock can be regarded as well outside Safe Biological Limits. Indeed, by many definitions can be said to have already collapsed.

A recent Australian evaluation of the SBT stocks has concluded that there is no real evidence that stocks have started to rebuild and that concerns arise from declining catch rates of recent cohorts as they approach maturity (McLoughlin *et al* 1996). This evaluation confirms that spawning stock is severely depleted and it is not clear that it will rebuild at current catch rates. Projections have been made which suggest that at current rates of fishing there is a 30% chance that SSB will be reduced to zero by the year 2020. At 110% of the current catch this would rise to 42%. If a 25% underestimation of catches is the case (by no means impossible) such that catch rates are 125% of current values, the probability of the SSB falling to zero rises to 59% (Klaer *et al.* 1996). In fact, the position may well be even more serious since these figures are based upon a CCSBT catch of 11,750t annually together with a non-CCSBT estimate of 2,600t. In fact, the non-CCSBT catch has recently been estimated at between 4136 and 4937t (CCSBT 1997). In addition, Japan has declared an intention to conduct an Experimental Fishing Programme, taking 2,010t of SBT in addition to the allocated quota of 6,065t each year for the next three years. This will bring the total catch to above 18,550t. According to Klaer *et al.* (1996) this translates to a probability of over 60% that SSB will be reduced to zero by the year 2020. This would result in commercial extinction of the species, if not actual extinction.

It seems quite clear that the Southern Bluefin tuna fishery is in a critical state. The parallels between this fishery and the North Sea and Canadian fisheries is striking even though the data is very much less robust. There is little doubt that the single identified population of Southern Bluefin Tuna stock has been seriously overfished as evidenced primarily through the nominal CPUE data. On the basis of comparison of pre-exploitation state of stocks with current stocks, it can be regarded as collapsed. The fact that it is not as yet commercially extinct is due to refocusing of fishing effort by contracting the overall areas fished. In continuing to fish while the stocks are outside safe biological limits and with the parental stock at best stable and at worst continuing to decline, there is a substantial risk that the stock will not rebuild. This is reflected in the extremely low probabilities attached to targets of population recovery in the next 25 years by the CCSBT. Significant doubts attach to the robustness of VPA, largely on account of uncertainties in the input data. Catch per unit effort; age at capture; spawning stock biomass; recruitment; age at spawning: all are subject to substantial uncertainties which can be regarded as fatally compromising the predictive value of VPAs in relation to this species.

Despite the extensive fishing effort and the importance of the fishery, the southern bluefin stock remains relatively poorly researched in comparison to other heavily exploited fishery stocks. Given the overall poor level of understanding and high levels of uncertainty attached to population parameters, in order to guarantee the survival of the stock, both in economic and actual terms, suspension of the fishery is essential. In addition to protecting stocks of SBT, suspension of the fishery would also have significant benefits to other species impacted by fishing activities, some of which are close to extinction, and allow the development of suitable and adequate protective strategies.

d) Orange Roughy

Development of fisheries now appears to be proceeding in the direction of exploiting deep water species. The orange roughy *Hoplostethus atlanticus* has a worldwide distribution and lives in water 700-1,500m in depth. These fish are very slow growing, reach a length of 55cm and live for well over 100 years reaching maturity at between 20 and 42 years. Fecundity is around 90,000 eggs per female. Fisheries for this species began in the early 1980s, increased markedly in the next few years and then fell markedly by 1990 (Clark & Tracey 1994; Mcloughlin *et al 1996*). Current biomass in the New Zealand Challenger Plateau Fishery is now estimated at around 20% of the unexploited stock, while in Australia biomass in the stocks had fallen to around 24% of unfished levels. Although the stock definition in this species has proven difficult to determine, all the indications are that this fishery is now outside safe biological limits. The fishery itself developed while substantial gaps in knowledge of life-history remained unresolved. The New Zealand stock has shown all the symptoms of an overfished population. Size at maturity in male fish declined significantly and evidence has been found of reduced genetic diversity. It seems that little has been learned from problems encountered in exploitation of continental shelf stocks

4.4 Ecosystem Impacts

A further dimension to fishery activity, and one which has been largely ignored by single species fishery management is that target species do not live in isolation. As part of an ecosystem, it is only relatively recently that whole ecosystem impacts have received significant consideration. Impacts may be divided into two broad categories. Those caused by removing the target and by-catch species and those caused by degradation of the habitat arising from the use of fishing gear. In addition there may be interactive impacts of other factors such as pollution. All these factors need to be taken into account in fisheries management, but most management schemes consider only the target species themselves. Some of the best documented habitat degradation occurs in coastal environments impacted by aquaculture (see: Section 5) and in coral reef environments.

Ultimately an environment may become so degraded by environmental insults that that major changes in the balance of species present in the ecosystem may occur (Browder 1988). In the Black Sea ecosystem, major shifts in the taxonomic composition have been observed. This situation may favour species such as comb jellies and jellyfish which cannot be used as a commercial resource or as a food resource for fish (Tett & Mills 1991; Mutlu *et al.* 1994) Not only will commercial fisheries collapse, but the stocks are then effectively prevented from regenerating by lack of suitable food. In addition, populations of other species associated in the ecosystem will also be jeopardised.

a) Ecosystem Interactions

Single species management systems tend not to account for the fact that the targeted fish comprise one element of ecosystems with highly complex interconnections. Nor do they tend to accommodate the problem that many commercial species are taken in mixed fisheries. Consequently, it is only in recent years that the impact of fishery activity has begun to be considered in whole ecosystem terms. Fishing activity can be viewed as a form of predation and populations of fish subject to severe fishery pressure show changes in population structure. Associated populations can, therefore, also be expected to change.

It has long been appreciated that exploitation of commercial species may lead to changes in species composition. By the early 1980s, it had become clear that replacement of long lived species of fish by smaller short lived ones had taken place in the Gulf of Thailand, the North Sea and off the West Coast of Africa (Garcia & Newton 1994). The current abundance of Alaska pollock in the Bering Sea may have come about as a result of overfishing (see: Stump & Batker 1996). Collapse of the ocean perch stock may have removed competition allowing the faster growing pollock to exploit the opened biological niche. This was part of a systematic overfishing of many fish stocks in the region. In the Baltic, the greatly increased exploitation of intermediate trophic levels by fisheries after 1945 was made possible by the near extinction of competitors at the highest trophic levels, coupled with a broadening of the trophic base, increased input of nutrients and by intensification of techniques and use of energy (Hammer *et al.* 1993).

Data from the North Sea while far from complete, are better than for many other marine fisheries. Fishing can be regarded as a form of predation and at high levels predation can alter food web structure. While the evidence that food web alterations cause changes in fish stock recruitment are ambiguous, major stock changes must at least affect the flow of energy through the food web (Tett & Mills 1991). Attempts to describe North Sea food webs have proven difficult. This partially because the factors that govern plankton dynamics are only poorly understood and these are a crucial determinant of fish populations. The most recent assessments suggest that four key species made up between 97% and 100% of planktivorous fish biomass. These were the Norway pout, herring, sprat and sandeel, all which are taken in quantity (20% or less of annual production) by the industrial fisheries. More than 50% of the food of saithe and whiting and between 10% & 30% of the food of cod, mackerel and haddock consisted of sandeel, norway pout and sprat. The piscivorous demersal fish are also in turn subject to severe predation as juveniles, perhaps explaining in part their vulnerability to fishing mortality (Greenstreet *et al.* 1997).

While the hard evidence for changes in trophic interactions due to fisheries until recently remained equivocal, there is some evidence that the rapid and large increase in the populations of gadoid fish in the early 1960s (Pope & Macer 1996), was due in part to decreased predation on larval and juvenile stages and/or reduced competition for food by these fish following the collapse of the herring and mackerel populations. Although the evidence is not consistent , the recent period of below average recruitment of the gadoids in the North Sea coincides with improved recruitment in herring populations and penetration of the Western mackerel stock into the North Sea (Hislop 1996). Given the relative lack of data concerning populations of the industrially fished species, it is not possible to establish the interactive effects of fishing pressure upon fish populations. Models have shown, however, that reduction in fishing pressure would result in an increase in SSB of most species except for haddock and Norway pout due to species interactions (Gislason 1994).

Although effects upon food webs have proven difficult to discern, effects upon the biological parameters of fish subjected to heavy fishing pressure in the North Sea have been identified. These have shown changes in size distributions and of maturation in the longer term which suggest that genetic selection has taken place (OSPAR 1993). Plaice, subject to extremely heavy fishing pressure in the North Sea now mature at a younger age and a smaller size than at the beginning of the century (Rijnsdorp 1989). Length- frequency distributions of fish caught by research vessels has shown that catches in the North Sea contain a greater proportion of small fish than samples from the Faroe and Georges Banks which were less heavily exploited. Size frequency distributions of both roundfish and flatfish have shown a trend towards smaller fish (Rijnsdorp *et al.* 1996) with smaller sized species such as whiting and dab increasing in relative abundance at the expense of species such as cod and plaice. This may be evidence of reduced predation on small fish as a result of depleted demersal fish populations.

A survey of ten non-target fish species has indicated that many of these species increased in number between 1970-1993 in the North Sea. Although no statistically robust inferences could be drawn as to causal effects, cluster analysis showed a grouping of fish species which have increased in number and which also mature at a smaller size than commercially exploited demersal species. This could be indicative of changes in the North Sea ecosystem (Heessen & Daan 1996). Greater weevers (*Trachinus draco*) and blue-fin tuna (*Thunnus thynnus*) have already virtually disappeared from the North Sea even though they were never specifically fished but taken in mixed fisheries (Daan 1990). Other declines have been reported (OSPAR 1993). These include two species of dogfish, rays (Walker & Heessen 1996), sturgeon and conger eel, but in the case of the greater weever, climatic factors in the form of the severe winter of 1963 could be partially responsible (Rijnsdorp *et al.*1996). The increasingly frequent summer appearance of scad in the North Sea may exemplify the behaviour of an opportunistic species. Unfortunately, clear trends in fish population dynamics related to possible ecosystem change are confounded by changes which can also be plausibly related to environmental factors. In many cases the precise mechanisms remain obscure (Rogers & Millner 1996; Philippart *et al.* 1996).

Changes in populations of rays, which have a low reproductive rate, also provide evidence of impacts of fisheries upon ecosystem balance and of the impact of fisheries themselves. As a bycatch of present demersal fisheries they have limited commercial value. Of the 11 species found in the North Sea only 5 are considered to represent stocks in the area (ICES 1997). The common skate has been almost extirpated, while the thornback and spotted rays are outside biological limits. The status of the cuckoo ray is marginal. The starry ray is considered to be within biological limits. Significantly, the starry ray is a small fish, and is usually discarded. In some cases, fish have become larger in contradiction to the trends. Sole, for example, have been found to grow larger at a given age than was the case in the 1960s. Paradoxically, this has taken place at a time when the stock began to fall (Millner & Whiting 1996). The most plausible explanation for this is the effect of beam trawling increasing food availability both by disturbing benthic infauna and by selecting for smaller more resilient infaunal species.

One frequently ignored aspect of fishing activity is the possibility of damage being inflicted on fish which escape the trawl gear and the subsequent fate of these fish. A study of mortality in vendace (*Coregonus albula)* escaping from a trawl was studied by holding them in cages at the depth of capture. It was found that on average 50% of the escapees subsequently died. Fish from night net hauls suffered the worst mortality at 60-80% of the escaped fish (Suuronen *et al.*1995). In a similar experiment on herring 70% of escapees under 12cm were estimated to have died subsequent to escaping together with 30% of fish above this size. Seine netting of 12-17cm herring resulted in mortality of 9-13% and specimens caught this way showed less skin damage than cod-end escapees. Highest mortalities were recorded at night (Suuronen *et al.* 1996 a &b). By contrast cod appear to be somewhat more resilient (Suuronen *et al.* 1996c) to effects of capture in trawls. Bacterial infections due to skin damage may explain the high rates of mortality, but the impact of stress also appears to be highly significant (Ludemann 1993). These findings throw into doubt the efficacy of regulating catches of target species by mesh size, and raise the question of the true impact of fishing activities upon the fish population as a whole.

The most compelling evidence of general global scale ecosystem disturbance has come with a recent re-analysis of FAO catch statistics (Pauly *et al.* 1998). This has shown (Section 4.1) that progressive fishing down food chains has occurred as commercial targeting responds to depletion in the original target stocks. A consistent downwards trend in the trophic level occupied by commercially fished species has been identified in global fisheries as predatory demersal fish are superseded as targets by pelagic fish which feed lower in the food chain. The potential impacts of this are noted by reference to Norway pout in the North Sea. Norway pout serve as a food source for most of the important demersal fish species including cod and saithe. Norway pout is also the most important predator on euphausids (krill) in the North Sea system. It might be expected that increased fishing effort directed at Norway pout will have a net positive effect on krill populations which normally feed on copepods. The copepods are an important resource for adults and juveniles of commercial species. It is possible, therefore, that fishing for Norway pout could result in a build up of krill populations. These are much less important as a food resource for fish. The net effect could be to seriously hamper stock recruitment of some species by diminishing the copepod food available to them at critical points in their life history.

While this scenario is somewhat conjectural, the observed pattern of exploitation is not, and the possibility of changes in the ecosystem need to be accounted for in assessments. As noted above (Tett & Mills 1991), fishing pressure undoubtedly causes the flow of energy through ecosystems to change. Ecological theory suggests that communities can exist in several stable configurations. Although there is a certain amount of resilience to disturbance, if this is severe enough the changes may become permanent. One hypothesis that has been put forward is that under combined environmental pressures, the Southern North Sea could become dominated by jellyfish species as top predators. Ecological regime shifts would undoubtedly vary according to locale, but the end result could be a complete shift away from commercially exploitable fish populations.

b) Physical Habitat Degradation

Some fishing gear and methods are highly damaging to marine habitats. Dredges and bottom trawls scrape or plough the seabed, disrupting sediment, destroying habitat and killing large numbers of benthic (bottom-dwelling) organisms. It has been estimated that fish trawlers and scallop dredges swept tracks across the Canadian continental shelf, approximately 4.3 million km in length in 1985 (Messieh *et al.* 1991). Impacts include scraping and ploughing of the seabed and the destruction of bottom-living plant and animal life (Jones 1992). Tube worms, sponges, anemones, urchins and other seabed life may be caught, crushed, uprooted or displaced . The scale of the problem can be illustrated from North Sea Beam Trawl operations. Over the last 20 years both the numbers and sizes of vessels engaged in this method of fishing have increased, together with the size weight and towing speed of the gear employed. In the period 1972-1990 total engine power of the Dutch beamtrawl fleet increased from around 250,000 horsepower to 600,000 horsepower, largely due to increases in vessels of 1500HP (Witbaard & Klein 1994). In some areas with a high trawl intensity, areas could be fished 3-5 times annually in 1975, rising to seven times in 1982 (Bergman & Hup 1992). It has been reported that up to 90% of resident bivalves (*Arctica islandica)* were damaged when caught by commercial trawler. Mortality of these was estimated at 74-90%. Tickler chains increase the numbers of shells damaged. In another study, by comparing samples taken before and after trawling three times and allowing a two week stabilisation period, it was found that representative benthic organisms decreased in density by between 10% and 60% for polychaete worms and brittle stars. Although these results suggest that beam trawling can seriously disturb benthic populations, the lack of areas which have not been subject to trawl activity over the last thirty years makes it difficult to find a control area representative of a pre-disturbance state.

Similar results were obtained for a study of scallop dredging in Port Phillip Bay, Australia showed that the abundance of seven out of the ten most common species changed after the dredge was dragged over the bottom. Six species decreased in abundance by between 28% and 79%, while one, a scavenger, increased in abundance. While some of the organisms rapidly recolonised, persistent changes in the benthos resulted from organisms failing to recolonise. Differences were detected up to 14 months after dredging. Prior to the experiment, dredging was virtually zero for three years, while in the sector as a whole over the preceding eight years only between 9% and 24% of the bottom was dredged. Nonetheless, scallop dredging operations had taken place in the Bay as a whole for some 30 years, again making it difficult to find a true control site. Nonetheless, this is a much lower level of disturbance than in North Sae fisheries Extrapolating the above findings to known annual trawling levels suggests that in combination with natural mortality, elimination of some species from natural habitats could occur (Dayton *et al.*1995).

Although the broad distribution of benthic communities in the North Sea is relatively well characterised (Heip & Craeymeersch 1995; Heip *et al.* 1992; Huys *et al.* 1992; Basford *et al.* 1993) impacts of fisheries remain unresolved. In part this is due to the limited time series covered by the data and poor understanding of the other environmental factors involved in determining benthic community structure. Other confounding factors exist. Marine aggregate extraction now removes some 25 million tonnes of aggregate in the UK alone, a threefold increase as compared to 1965. Substantially reduced biomass was recorded at one experimental site 24 months after extraction of 50,000 tonnes of gravel. Dominant species recolonised quickly, the rarer species did not (Kenny & Rees 1996). The disposal of dredged material can also modify benthic fauna, and marine development projects have been found to change the primary production in sediments.

One model collated the available life history data for eight species of fish. The model simulated a one-off 50% reduction in the survival of fish larvae at the first year. The model predicted that on average, equilibrium to 88% of pre-impact levels would take ten years (Schaaf *et al.* 1987). Later work (Schaaf *et al.* 1993) modelled the effect of habitat destruction for Atlantic menhaden. This model takes into account potential impacts on adults. These spend winters away from the estuarine habitats which serve as breeding and nursery grounds. The results were startling. If 1% of the Atlantic menhaden habitat was destroyed, the model predicted that the population would fall by 8% over ten years. If this habitat loss took place in the estuaries, affecting juveniles also, then in ten years the population would be driven down to 58% of its normal level.

Destruction of habitat is also associated with coral reef fisheries. The use of cyanide poison to stun fish can kill most reef organisms, leaving much of a reef dead and degraded. The method is used to capture valuable reef-fish for the growing market in live food-fish and for aquarium displays. These earn fishers 4-8 times the amount of the species when dead. Due to the large sums of money involved, it is expected that this fishing method will drive numerous species to collapse in the Philippines, Indonesia and the Maldives. It is predicted that reefs in the wider Pacific will subsequently come under similar pressure (McGinn 1998). The use of hypochlorite bleaches and dynamite for fishing on coral reefs also cause severe damage (Richmond 1993). Removal of fish from coral reefs can cause severe ecosystem shifts. The coral/invertebrate dominance can shift to an algal dominance by fleshy and filamentous algae as a result of removal of grazers. This then can be followed by bio-erosive processes (Roberts 1995). A relatively low fishing intensity by open water standards may prove enough to cause significant changes in the reef fish community. In a study carried out in Fiji, removal of 5% of fish biomass caused long term decreases in overall fish biomass on the reefs (Jennings & Polunin 1996). Similarly in the Seychelles, survey work found that the diversity of target fish was higher at unfished reef sites than those being exploited. No evidence was found of replacement of target species by non-target species (Jennings *et al* 1995). This pattern was confirmed in Hawaiian reefs where unfished sectors supported higher biomasses of fish than fished areas. The unfished populations were less wary of humans, increasing their potential as a tourist resource. Abundance of fish was found to actually increase in the vicinity of treated sewage outfall pipes, partly as a result of construction creating a more complex habitat (Grigg 1994). Fishing appeared to be the primary determinant of reef fish abundance and biomass. Recovery of some species of reef fish after exclusion of fishing activity from a reef in Kenya was found to be slow due to competition from sea urchins which had become established during times of heavy fishing activity. In most other cases fish species showed good recovery (McClanahan & Kaunda-Arara 1995).

c) Bycatch of Fish and Other Species

i) Cetaceans

As noted in Section 4.1 a variety of organisms other than the target fish species may be taken in commercial fisheries. Large numbers of marine mammals, sea turtles and seabirds are caught in commercial operations. Marine mammals are taken unintentionally in a variety of fisheries and best estimates suggest a current global total of between 65,000 and 85,000 mortalities world wide for cetaceans and several hundred thousand when other marine mammals are taken into account (Alverson *et al* 1994). The best known of these are the tuna purse seine fisheries. In the Eastern Tropical Pacific region, for example, it is estimated that between 1959 and 1972 nearly 5 million dolphins were killed in purse seine fisheries for tuna (Wade 1994). Harbour porpoise populations have been seriously affected by gill-netting operations. The Gulf of California Harbour porpoise, endemic to an area about 30 miles in radius has been impacted to such an extent that it could become extinct (Castello 1996). Of the few hundred remaining, it is estimated that 30-40 are lost each year in gill nets. It is estimated that 2000 harbour porpoise are caught and drown in each year in bottom set gill nets in the Bay of Fundy, Canada. Mortalities have also been reported of cetaceans in the coastal fisheries of British Columbia (Stacey *et al.* 1997). Between 25,000 and 45,000 small cetaceans are taken in the modernised gill net fishery in Sri Lanka annually while drift netting for squid took an estimated 19,000 dolphins in 1989 (Mulvaney 1996). In the US East Coast swordfish fishery an unknown number of cetaceans from at least 11 different species have been recorded as caught by the pelagic drift nets used (Read 1996). German fisheries in the Baltic and North Sea fisheries regularly take harbour porpoise, but it is possible that many of the recorded strandings in the region result from by-catch (Kock & Benke 1996).

In the intensively fished North Sea it is estimated that populations of harbour porpoise have fallen by between 53,000 and 89,000 animals since the 1950s. The upper estimate is considered the more reliable (Reijnders 1992). Over the last 30-40 years between 1,000 and 2,000 animals were caught annually in the area bounded by Den Helder-Grimsby and Hook of Holland-Harwich by the Danish gill net fishery alone. Causes of death in 41 harbour porpoises found dead on the English coast showed that entanglement in fishing gear was the single greatest cause of death, accounting for 24% of casualties (Baker & Martin 1992). Theoretical studies of Canadian harbour porpoise have shown that, probably, they would be unable to withstand 4% additional mortality in the population (Woodley & Read 1991). If this is applicable to other populations then current dolphin bycatch in gill nets in the Celtic Sea is a serious concern. It has been estimated (Tregenza *et al.* 1997) that around 2200 dolphins are caught and drowned each year. This represents around 6% of the population.

Impacted populations are showing signs of changes in biological parameters more usually associated with over-exploitation. In the Bay of Fundy, Canada, calf sizes were smaller between 1969-1973 than in the period 1985-1988, with females maturing at a slightly younger age (Read & Gaskin 1991). Similar observations have been made in the North Sea. Sexual maturity is now attained when animals are 10kg lighter and 10cm shorter than in the 1940s (Clausen & Andersen 1988). Animals weighing 60kg or more now make up around 1% of the population as compared with 35% in 1941-43. Animals exceeding 1.6m in length comprise only 2% of the population, down from 26%. This is regarded as evidence that incidental mortality is having the same effect as over-exploitation if this was considered as a target species. (Law *et al.* 1992).

ii) Turtles

Turtles are a significant bycatch component in various fisheries but were not considered specifically in detail in the FAO evaluation of bycatch (Alverson *et al.* 1994). In the Gulf of Mexico all five sea turtles found in the US are listed as endangered, yet estimates of the mortality range from 11,000 to 44,000 annually. (Griffin *et al.* 1996) and is thought to kill more sea turtles than all other human activities combined. The catch of juvenile loggerhead turtles contributes to preventing recovery of this threatened species. It was estimated that the shrimp fishery took some 50,000 loggerhead and 5,000 Kemp's ridley turtles annually in the 1980s. The adult female breeding population of the latter is estimated at a critically low level of less than 1,000 worldwide. Leatherback, green sea (both listed as endangered) and loggerhead turtles were taken in the North Pacific squid driftnet fisheries, but the threat that this represented to the population as a whole is unknown. Turtle exclusion devices, though resisted by the shrimp-fishing industry appear to be employed more frequently in the industry, but this has not prevented the occurrence of turtle strandings associated with the fishery in the US. (Griffin *et al.* 1996; Callout *et al.*1996).

Paned prawn fishing in Australia is also associated with turtle mortality. On the basis of a two year survey of the Queensland East Coast Otter Trawl Fishery turtle bycatch from 50 commercial fishing boats, was estimated at 5295 animals annually. Of these around 1% were dead. although including comatose turtles raised this percentage to 6.8% (Robins 1995). This is a low mortality compared to other similar fisheries, probably because trawl tow durations are short in this sector. In the Northern Australian Prawn Fishery, 6-10% mortality was recorded in the 5730 animals estimated to be caught annually. Although the precise numbers are not known it appears that turtles are caught incidentally in fisheries off West Africa (Carr & Carr 1991). In the South China Sea, turtle by-catch mortality has been estimated at 21,200 in the Western Pacific and South China Sea alone (Alverson *et al.* 1994) due to longlining. Many species of sea turtle are endangered or threatened, and continuing fishery associated mortality is likely to hinder the recovery of these populations, or push them further towards extinction.

<u>iii) Birds</u>

The interaction of seabirds with fisheries was considered in the FAO Assessment (Alverson *et al.* 1994) and bycatch and mortality of seabirds is emerging as an issue of global importance. In particular, longline fisheries have attracted attention. Longlining is regarded as a conservation oriented fishing method insofar as it tends to catch mainly target species, causes no destruction of the bottom habitat and catches fish of high quality with low fuel consumption (Lokkeborg 1998). Countering this perception, however, are the serious documented impacts upon birds and non target fish species. These may be exemplified by the impacts recorded in the Southern Bluefin Tuna (SBT). It has become evident that Albatross suffer significant mortality as a result of attempting to steal bait from longline hooks as they are set (Brothers 1991). Of the 14 species of albatross, ten are confined to the Southern Ocean, roaming widely and coming ashore only to breed on remote oceanic islands. An average longline vessel sets between 2400 and 3000 barbed steel hooks on 40m unweighted branch lines attached to the mainline which is approximately 100km in length. The hooks are baited with whole fish or squid and sink to a depth of between 60m and 150m after being thrown from the stern of the vessel steaming at around ten knots. Albatrosses are caught on the baited hooks during the period when the bait is sinking. Although this is a relatively brief period, observations have shown that an average catch rate of 0.41 birds per 1000 hooks set can occur. The actual catch rate is likely to be considerably higher since not all albatrosses which were caught were hauled aboard, being either eaten by fish or torn off the hooks during hauling. A number of species may be present during line setting. All species of albatross are susceptible to being caught in this way. The actual catch rate varies on a regional basis and is higher around New Zealand than off Tasmania, for example.

If the observed results are extrapolated over the whole SBT fishing range then it can be estimated that some 44,000 albatrosses of five species are killed annually. Albatrosses mature in some cases only at twelve years of age and, moreover, may breed only once every two years. Hence, the mortalities associated with longlining activities can be expected to exert a severe effect on the integrity of populations of these birds. Prince *et al.* (1994) note that population declines in colonies of albatross which they studied appeared to be due to higher juvenile mortality and that this has increased since the 1960's. Coupled with the observation that a high proportion of birds hooked in longlining activities are juveniles, this provided some measure of confirmation that fisheries associated mortality is a highly important factor. Fisheries related adult mortality is thought to explain the decline of wandering albatrosses on Crozet and South Georgia Islands. As pointed out by Weimerskirch *et al.* (1997), there is no absolute proof that the decline of Possession Island albatross populations has been caused by the Japanese longline fishery.

Nonetheless, the correlations found between (1) changes in fishing operation and population size, (2) fishing distribution and effort with albatross survival rates, and (3) estimated numbers of birds killed by the fishery, together with observed changes in the number of breeding birds, lead to the conclusion that long line fisheries are at least partly responsible. Of these, the largest by far is the Japanese SBT fishery. The current situation is acute. The population of wandering albatross on Macquarie Island has been almost entirely extinguished (de la Mare & Kerry 1994). The entire population of the Amsterdam albatross has been reduced to between five and eight pairs breeding each year on a single island and faces a high probability of extinction (Weimerskirch *et al.* 1997). While some measures can be taken which mitigate albatross mortality such as increasing the weighting of the lines, laying them in hours of darkness and using streamer lines, it is likely that such practices will conflict with commercial considerations. Accordingly, in the absence of a reliable enforcement and reporting regime it is probable that the effectiveness of such measures will only be measurable *post hoc* by monitoring breeding population numbers. A point of growing concern is the developing fishery for the Patagonian toothfish by longlining, since this will intensify the numbers of hooks set in areas frequented by these endangered birds (Weimerskirch *et al.* 1997).

Longline fisheries for groundfish in the North Atlantic are conducted by a fleet of several hundred vessels (Lokkeborg 1998). In these fisheries, Northern fulmar (*Fulmarus glacialis*) comprise the great majority of seabirds caught incidentally by longlining. Although this does not appear to have significantly affected the population of these birds as a whole, the loss of up to 70% of baits to birds is a cause for commercial concern. A number of measures have been proposed to reduce incidental catches of birds in longline fisheries. The baited hooks can be guide into the water below the surface through submerged funnel, or by setting baits in the hours of darkness. Birds may be scared away from bait setting operations using visual or noise scarers or lured away from them by discharging fish offal as a more easily exploited food resource. The discharge of fish offal is reported as having greatly reduced seabird capture in the Patagonian toothfish (*Distostichus eleginoides)* fishery conducted in Kerguelen waters. Testing of funnel devices and scarers has also shown that these alone cannot entirely prevent incidental seabird catches. Discharge of offal, the use of fully thawed bait or the weighting of lines may not be possible or involve practical problems in some fisheries.

Other direct impacts upon birds have been noted in the literature. Entanglement of seabirds in gill nets probably causes the heaviest impact together with other fixed nets that entangle diving seabirds. Even in the relatively well researched North Sea area there are not enough data to evaluate the extent of the problem, but it is regarded as being sporadic and localised in this region (OSPAR 1993). Elsewhere more serious problems have been identified. It is estimated (Alverson *et al.* 1994) that between 21 and 9.3% of various populations of gannets are discarded in Newfoundland gill net fisheries. In the early 1980s 12% of the breeding population of razorbills were killed each year in gill nets. Drift nets were also responsible for the death of large numbers of seabirds in the squid fisheries off North American coasts. The impact of these kills is unknown, but threats to can result when the numbers killed are a high proportion of the breeding birds in a population.

Correlations between the availability of fish and seabird populations appear to be complex and this is due in part to a low turnover due to their delayed maturity and low fecundity. In the Barents Sea, only when the stocks of capelin and herring reached the minimum were declines noted in the numbers of seabirds (Barret & Krasnov 1996). The search for precise causal relationships, however, is confounded by incomplete data concerning the stock of other prey species taken by the birds. There is considerable plasticity of diet in the bird species studied. Breeding success in the surface-feeding kittiwakes, however, appears to depend on the abundance of capelin in the chick diet and it appears that this species is now having difficulty maintaining breeding success in the wake of the latest capelin crash. Guillemots and puffins are pursuit diving feeders able to take advantage of returning herring in the area and their populations are currently increasing. The trends in populations of herring gull in the Gulf of St Lawrence are clearer. The overall population of these birds increased in proportion to the intensity of fishing in the area over the period 1925-1975 as a result of utilising discards. Although some populations appeared to be more dependent upon discards than others , collapse of the cod fishery in 1993 after a period of decline was reflected in a fall in the herring gull population from 14,000 pairs in 1988 to 3000 pairs in 1993 (Chapdelaine & Rail 1997) .

Visible effects of over exploitation of shellfisheries were documented in the case of eider ducks. In the Dutch Wadden Sea, overfishing of the mussel and cockle stocks resulted in major mortality in this species. Scoters were found more often at sea than previously, indicating difficulties in obtaining food from their normal sources. A recent study also reports a severe reduction of knots in areas of the Wadden Sea (*Calidris canutus*), probably caused by mussel and cockle fisheries (Piersma *et al.* 1993). In the period of 1964 to 1992 there were on average 10.000 knots each year between August and October on Grient. This small island serves as a key resting place for several migratory bird species. Since 1988 a severe decline in abundance of knots was reported, and in the period of August-October 1992 only 700 knots were observed. The intensive cockle fishery north of Grient in 1988 not only depleted a food source for the knots, but also severely damaged the sediment structure. This has led to a steady decline in density of the bivalve *Macoma balthica*. This is the main food source of knots.

An analysis of the potential impacts of fishing activities on seabirds in the North Sea (Garthe *et al.* 1996) concluded that offal was responsible for supporting large numbers of fulmars, gulls and other scavenging seabirds. Many of these have increased in number since the beginning of the century. Offal provides an important feeding resource in most areas of the North Sea. Discards are considered to be an important cause of increase in great skuas, great and lesser black backed gulls, black headed gulls, herring gulls and gannets (OSPAR 1993). Certainly, black headed gulls and herring gulls comprise a high proportion of the ship-following birds (Garthe 1997) and together with great black-backed gulls, show high densities around trawlers. While fulmars profit from the availability of trawler waste, their distribution does not appear to be determined primarily by shipping activity (Camphuysen & Garthe 1997). There also appears to be some variation in the composition of the birds following trawlers by region and possibly by season (Garthe & Huppop 1994). The discards from shrimp trawlers were found to be taken primarily by herring gull (Walter & Becker 1997). In the North Western Mediterranean, trawler discards were utilised by a variety of seabirds and were estimated as a substantial proportion of the food requirements of the Ebro Delta populations of Audouin's gull (Oro & Ruiz).

Trawl discards have been found to be an important component of seabird diet on the Barrier Reef in Australia and have changed the diets of the birds (Blaber *et al.* 1995). Although no differences were found in breeding behaviour of the studies species, in the Mediterranean, diet differences were caused by a trawling moratorium affecting some populations of black-backed gulls. This in turn was reflected in reduced breeding success (Oro 1996). On the basis of the various studies, it appears that scavenging species of seabirds respond more predictably to the availability of discards than other species. It follows that any limitation on fishing activity can be expected to impact on the populations of these species.

iv) Fish and Discarded Bycatch

Bycatch has been recognised as a serious problem in a number of fisheries and, indeed, is a problem of global dimensions (Alverson *et al.* 1994). Total discards in the global fishing industry are estimated (with high levels of uncertainty) at some 27 million tonnes. A large proportion of discards (37.2%) are associated with various shrimp fisheries, which in tropical regions is composed of small fish. Shrimp fisheries have almost universally high discard rates. In temperate waters a significant proportion of the discard is juveniles and adults of commercially valuable species. Many of the groundfish fisheries in temperate waters are associated with significant levels of discard. Discarding generally takes place in fisheries for human consumption, discarding from industrial fisheries is thought to be rare (Gislason 1994) although there is leakage of fish as they are pumped on board.

The reasons for discarding fish are various and include specimens too small to be commercially exploited, those which are below the official minimum landing size, those of species for which no demand exists or fish which are in excess of allotted quotas. Where quotas exist, discarding may take place to maximise the value of the catch ("high grading"). Survival of discarded species is likely to be very low. Of the 40 million whiting discarded each year, all are thought to die (Alverson *et al.* 1994). Survival of tropical fish such as grouper and snapper are thought to be low after discarding. Halibut discard mortality is variable ranging from 2% in longline fisheries to 100% in some trawl operations. The picture that emerges is one of mortality being dependent upon the biology of the fish, environmental conditions art the time of capture as well as the type of gear employed and length of trawl time.

In assessing the significance of discards, part of the problem is that systematic records of discards are not kept for many fisheries. In the North Sea for example, only discards of haddock and whiting are systematically reported. There is considerable variation between individual fisheries and variation with the state of the stocks. In addition gear types influence the total bycatch and hence discards. In the North Sea it is quite clear that the highest amounts of discard are generated by the beam-trawl sector, consisting of large amounts of benthic organisms mixed with inorganic matter. Overall, discards of commercial species equal or exceed the amounts of fish actually landed (Gislason 1994). Other estimates for the North Sea suggest that offal disposed of overboard amounts to some 62,800 tonnes, roundfish account for a further 262,000 tonnes, flatfish discards amount to 299,300 tonnes and elasmobranchs 15,000 tonnes. 149,700 tonnes of benthic invertebrates are also discarded each year (Garthe *et al.* 1996). The beam trawl fleet accounts for around 50% of the total discards in the North Sea and in the 1980s and early 1990s discarded 6010kg of fish for every kilogram of sole landed. Between 1991 and 1993, some 56,000 tonnes of haddock and 36,000 tonnes of whiting were discarded by North Sea gadoid fisheries. Other estimates have suggested that 15,000 tonnes of cod and about 10,000 tonnes of plaice were discarded from gadoid fisheries (Garthe *et al* 1996) in the North Sea.

This pattern is repeated in other fisheries globally. In the Australian Northern prawn fishery, some 240 different species are known to be among the discarded catch. In the Bering Sea pollock trawl fishery, records show that 130 species were discarded in 1992. These included some 100 million pollock, 8.5 million rock sole, 3.2 million Pacific cod and 2.3 million flounders. In other Bering Sea fisheries, another 200 million pollock were reported as discards. This represents 1.6% of the exploitable population and has risen from a figure of around 0.5%. This implies that recruitment overfishing is taking place. On a species level, discarding in the Bering Sea ranged from 1.9% of the fishing mortality on sablefish to 83% of the fishing mortality on flounder. These are important mortalities and indicate the need to take discards into account in fishery management programmes. Nonetheless, impacts of discarding on populations and species assemblages have been described. In haddock and whiting, discards exert a strong influence on sub-adult population numbers. Reductions in populations of non-target fish have been identified as a result of discarding practices. Similar observations have been made for populations of fish discarded in the Gulf of Carpentaria fisheries in Australia.

The potential impacts of longlining and other fishing techniques upon elasmobranch species are receiving increased attention (Alverson *et al.* 1994; Berrow 1994). Sharks in common with other elasmobranch fish have generally low growth rates, low reproductive rates and a close stock/recruitment relationship make them particularly vulnerable to overfishing (see: Walker & Heesen 1996). One of the better documented examples of impacts upon sharks relates to SBT longline fisheries. Data reported by Stevens (1992) suggests that 34,000 blue sharks, (around 275 tonnes live weight) mostly immature females, are caught in Tasmanian waters of the Australian Fishing Zone annually by longliners fishing for SBT. The carcasses are usually discarded after removal of the fins to conserve freezer space. Data reported from southern New Zealand in the same paper indicate that longlining in a single fishery took an annual average by-catch of around 74,000 blue sharks between 1980 and 1989. Overall in New Zealand, the annual average by catch from three longlining fisheries was almost 79,000 blue sharks. There is a substantial discrepancy between observer data and fishing vessel logbook data suggesting a severe under reporting of by catch. In addition various other species of shark are also caught. There appears to be no complete data for shark by catch attributable to CCSBT signatory nations while by catch for non-signatory nations appears to be virtually undocumented. As pointed out by Stevens (1992) the effects upon blue shark populations of removing significant numbers of non breeding females is unclear. Little is known about blue shark biology and their stock structure. Overall, there is an urgent need to formulate specific management and conservation plans for these fish. These in turn will need to be based upon accurate landing statistics including by-catch and upon a thorough understanding of the stock structure. At present, this information is not available.

The impact of discarding juveniles of target species is noted above (Section 4.3b) in relation to the confounding of data used to manage the Canadian groundfish stocks which collapsed in 1992. Similar problems have emerged in the North Sea (IMM 1997a) where actual fishing mortalities have always been much higher than intended. The models used produce underestimates of mortality as a result of misreported and unreported landings and because of high discarding rates (IMM 1997a). In fact the targeting of fish in mixed fisheries is a particular problem in North Sea gadoid stocks where the target fish mature at different ages and lengths but are caught in the same gear (Lewy & Vinther 1994). Fishing gear is often set to catch as many whiting as possible above the minimum landing size, leading to substantial discards of undersized haddock and cod, increasing pressure on these populations for no commercial gain (Alverson *et al.* 1994). There is clearly a need to account for discards on a global basis and to formulate management practices which account for fishery specific aspects of the problem (Hall 1996b; Alverson & Hughes 1996)).

e) Potential Impacts of Pollution on Fish Populations

Despite the immense pressure on fish stocks caused by over-exploitation, other factors also influence fish populations. Water temperature is, perhaps, the most obvious one yet even this factor is very poorly understood. Temperature can affect the abundance of fish from year to year, and not always in a predictable manner. In a study of flatfish in the River Severn in the UK the abundance of sole was found to be positively correlated with water temperature at the time of spawning of the adult fish the previous year. By contrast, in dab and flounder abundance was negatively correlated with temperature at this time (Henderson & Seaby 1994). The precise reason for this remains unknown but may relate to changes in predator activity in relation to temperature. Direct negative effects upon the reproduction of fish have also been shown in populations exposed to thermal discharges from power stations. Egg producing capacity in female roach was severely reduced by exposure to the hot effluent (Blaxter 1992; Luksiene & Sandstrom 1994). Fish populations, therefore can be extremely sensitive to changes in environmental conditions and even slight changes in such basic factors such as the temperature of the sea can produce wide variations of response.

Recent research has shown that chemical pollution of the sea can adversely affect both individual fish and hence, potentially, whole fish populations. Chemical pollution, for example, can result in disease and reproductive disturbance. The information concerning pollution effects upon fish is reasonably accessible in the scientific literature. Nonetheless, This aspect does not appear to be routinely considered in depth in regional evaluations of fish biology (see: e.g. OSPAR 1993) Two important questions arise in relation to chemical pollutants in marine environments. Firstly, what effect is pollution having on the fish themselves? Secondly, how could this affect fish populations and their recovery from over exploitation?

Pollution can, of course, exert catastrophic effects upon fish populations. Massive fish kills often result from spillages of chemicals and oils, while agricultural pollution is an increasing cause of fish kills in rivers. A fire in a chemical warehouse on the River Rhine in 1986 caused a massive mortality of fish and invertebrates (Capel *et al.*1988). If such kills take place during the spawning season at a sensitive site then a whole year class of fish may be lost to the population. This was a real concern in the case of the *Braer* oil spill which took place in the Shetlands. This happened during the spawning of the sand-eel population which is a highly important food resource for local bird populations. The impacts on sand-eels are not clear but the effects of the spill were highly visible in the intertidal zone (Newey & Seed 1995). Such incidents contrast with normal marine exposures to chemicals which are typically over long periods but at low concentration. Sensitive analytical chemistry techniques have shown that in the North Sea, for example, chemical contamination is detectable throughout (OSPAR 1993).

Many more fish, therefore, are exposed to chronic pollution than are affected by catastrophic spills of chemicals and oil. While there are obvious health concerns attached to the human consumption of contaminated fish (Falandysz 1994b), fish from the North Sea are still regarded as fit for human consumption. The effects of chemicals upon the fish themselves give immediate causes for concern. The chemicals to which fish populations in the North Sea are exposed are present as a complex mixture. Even regulatory agencies admit that the toxicological impacts of chemical mixtures are impossible to predict given the current state of scientific knowledge (Matthiessen *et al.*1993). Some impacts of chemical pollution identified in fish, such as skin damage and liver malfunction, have been known for some time. Other, newly identified effects are more subtle. Suppression of the immune system of fish, damage to their reproductive systems and reduced viability of fish larvae have received little attention from regulatory authorities. Yet these impacts may well be contributing significantly to the decline in fish populations.

Fish exposed to sub-lethal industrial pollution may become diseased and stressed. Like all organisms, fish suffer naturally from disease and infestation with parasites. More recently, however, it has become apparent that exposure to chemical pollution can also exert serious effects upon the health of individual fish and hence upon fish populations (Myllyvirta & Vuorinen 1989; Vethaak 1993). The longer term exposure of fish populations to certain industrial effluents has resulted in much more subtle effects than the catastrophic kills often associated with pollution incidents. The observed effects range from alterations in the behaviour of fish, to the appearance of deformities and to the appearance of obvious diseases of the skin, liver and other organs. It has been shown in laboratory studies that the capacity of immune cells of fish exposed to cadmium is significantly compromised (Hutchinson & Manning 1996). Although the apparent seriousness of these effects spans a wide spectrum, each can negatively affect individual fish and ultimately whole fish populations.

Fish have a number of biochemical responses to toxic chemicals. One of the most studied responses of fish to chemical pollutants involves the functions of the liver. The mechanisms for chemical detoxification are sited in this organ which can produce enzymes known as the mixed function oxidases. For example induction of the cytochrome P4501A1 enzyme system, may take place as fish deploy natural mechanisms in an attempt to detoxify chemicals (Jiminez *et al.*1990). The activity of these liver enzymes increases with increasing exposure to the chemicals until the system becomes overwhelmed (Collier & Varanasi 1991; Elksus & Stegeman 1989; Goksoyr *et al.*1989) and damage to the liver results.

The induction of liver enzymes by toxic chemicals has been proposed as an indicator of chemical exposure in fish (Stebbing *et al.* 1992a; Burgeot *et al.* 1994). Bottom dwelling flatfish seem to be particularly affected by constant exposure to chemicals in the sediments. Positive correlations have been established between the occurrence of liver cancer in fish and the levels of toxic chemicals in sediment (Landahl *et al.* 1990; Vethaak & ap Reinhallt 1992; Vethaak *et al.* 1992). An increase in liver abnormalities and liver enzyme activity in dab has also been documented along a North Sea pollution gradient extending out as far as the Dogger Bank where chemical pollutants build up in the sediments which are deposited there. The detoxification of chemicals in the liver involves enzyme systems that are normally involved in hormone, steroid and fatty acid metabolism (Goksoyr & Forlin 1992). This then allows the direct interaction of foreign chemicals with physiological processes. Some chemicals such as the polynuclear aromatic hydrocarbon benzo[a]pyrene (B[a]P) are converted to their most carcinogenic form by these processes (Hinton 1989). The chemical metabolite of B[a]P can combine directly with the genetic material DNA. The severity of liver lesions in flatfish in Europe and the US have been shown to closely correlate with the accumulation of a range of chlorinated, toxic and carcinogenic chemicals (Kohler 1990; Myers *et al.* 1991).

One disease of finfish has attracted attention because its unsightly appearance affects fish marketability. Highly visible ulcerations of the skin appear to be pollution related. Skin ulcers occur on a wide range of fish and have been described from the North Sea and the eastern seaboard of the US. The species distribution and occurrence of these lesions appears to be expanding and in the US at least they are considered to be a problem of regional scale (Levine *et al* 1990). Skin lesions have been reported from Atlantic menhaden, weakfish, southern flounder and American eels in the US, while in the North Sea the problem has been identified in dab exposed to metal rich wastes from the titanium dioxide industry (Vethaak & van der Meer 1991).

Monitoring activities conducted by the International Council for the Exploration of the Sea (ICES) have revealed that ulcerative lesions are more numerous in flatfishes from coastal waters adjacent to large urban populations, and these findings were paralleled in the US (Murchelano 1990). Nonetheless, the effects of ulcerative lesions on the reproductive viability of commercial stocks are unknown although the stressed conditions of the fish could affect reproduction by affecting survival and metabolic processes. There is also some evidence that the conditions favouring the development of these skin ulcers in fish also suppress the immune system (Bucke *et al.* 1989). Immune responses of a variety of marine organisms are known to be affected by pollution (Moles *et al.*1994). In turn, this may affect the ability of fish to resist bacterial disease and parasitic infection although relatively little empirical evidence of this has been gathered to date (Kahn & Thulin 1991).

Alterations to the behaviour of schools of fish have been observed following exposure to industrial effluents. The schooling behaviour of vendace (*Coregona albula*) is changed by exposure to pulp and paper bleachery effluents. By increasing predation pressure on the fish, such changes may be of great significance to fish populations (Myllyvirta & Vuorinen (1989). As noted earlier much larger numbers of fish are exposed to sublethal levels of chemicals than are acutely exposed. Sub-lethal levels of pollutants are known to interfere with the sensory systems of aquatic animals (Blaxter & ten Hallers-Tjabbes) affecting their ability to avoid predators and their ability to locate food. Under some circumstances, the feeding rate of fish exposed to chemicals may be reduced (Sandheinreich & Atchinson 1990). This is independent of pollution effects which may be exerted directly upon the food organisms themselves. This may alter the availability of preferred food organisms to the fish. Importantly, fish may not show avoidance behaviour towards areas where they are chronically exposed to pollution. Some juvenile fish will elect to live on contaminated sediments rather than clean ones on the grounds of preferring the size of the contaminated sediment grains (Moles *et al.*1994). This behavioural pattern will maximise their exposure to pollutants and increase the likelihood of other pollution related impacts.

Changes in the quality and quantity of food resources available to fish, together with changes in their ability to locate and utilise food resources have profound implications for individual fish and their populations. The growth and maturation of fish depends very greatly on the food that is available to them. Any fall in the quality of their food is likely to stress fish making them less likely to breed successfully. If their numbers are reduced by predation following changes in their schooling behaviour, then this too could mean that fewer individuals will reach reproductive age and be capable of regenerating depleted stocks.

Gross external deformities have been observed in fish exposed to the chemicals present in pulp and paper effluents. Baltic perch exposed to chlorine bleach effluents have been shown to suffer from fin erosion (Lindesjoo & Thulin 1990). Fin erosion has also been observed in fish living in the highly contaminated waters of the New York Bight (Weis & Weis 1994). Pike exposed to chlorine bleach effluents are found to suffer from severe skeletal deformities of the jaw and similar deformities have been recorded in other fish species chronically exposed to pollution. Such deformities may affect the ability of fish to escape predators or to exploit food resources and thus affect the size of the breeding population.

Overall, several clear links have emerged between disease in fish and their exposure to pollution. Gross deformity and behavioural alterations lie at opposite ends of a spectrum of demonstrated effects of pollution on fish health (Shugart *et al.*1992). In most cases gross disease in fish is relatively easy to detect by examination of the whole fish or their internal organs. It is possible that ill health, by reducing the condition of the fish will affect their ability to avoid predation. Fish in poor health due to pollution exposure are less likely to breed successfully and the population will suffer as a result. Overall, sub-chronic pollution can stress individual fish to a high degree. By impairing the immune system and making the fish more vulnerable to disease, and by compromising the function of the liver, pollution stress could seriously impair the condition and reproductive capacity of a fish population.

One aspect that has been little investigated to date, but which is of great potential importance is the possible effect of pollution on the genetic diversity of fish populations. Genetic changes not only lead to cancerous diseases of organs such as the liver. In many species of aquatic animals exposed to pollution, genetic diversity is reduced. Pollution effectively acts as a factor in natural selection by selecting for populations of animals which can withstand the polluted conditions. The characteristics which allow adaptation to other changes in environmental conditions may be selected against. Hence, while fish may appear to adapt to polluted conditions it may be at the expense of the long term ability of the population to accommodate other changes in environmental conditions. Although this may be important, (Depledge 1994) such changes in genetic diversity in fish have received very little attention, and so any conclusions are, necessarily, highly speculative. At worst, it could lead to a form of insidious chemical extinction of natural fish populations.

The embryos and larvae of fish are known to be much more highly sensitive to the effects of pollution than the adults (Sprague 1971; Weis & Weis 1994). One study outside a petrochemical complex has shown that fish recruitment into contaminated areas is lower than at reference sites (Jacobsson & Neuman 1991). Individual fish from adult populations close to the complex were larger in size but much smaller in number as a result. Developmental defects in fish larvae have also been reported and attributed to pollution. Aberrant development of Baltic fish was found to range between 18% and 44% depending upon species for larvae sampled in the western Baltic (von Westernhagen *et al.*1988). Malformation rates of fish larvae in the Eastern North Sea were found to be highest in the centre of the German Bight and off the Dutch coast. Both areas are highly impacted by contaminant input from rivers (Dethlefsen 1989). In addition, (Kingsford & Gray 1996), pollutants may not disperse but may become locally concentrated away from the point of release by various physico-chemical processes thus exacerbating the effects.

In the North Sea, the number of deformed fish embryos was positively correlated with contaminant levels along a transect from the inner German Bight to the Dogger Bank. Abnormalities fell with increasing distance offshore, rising again over the Dogger Bank, where pollutants tend to accumulate in the sediments (Stebbing *et al.*1992b). Similar results have been reported from California (Matsui *et al.* 1992) and the New York Bight (Longwell *et al.*1992) where the highest rates of larval malformations were reported from the most seriously polluted areas. Subsequent further evaluation in the North Sea suggested that fish embryo deformation could possibly occur as a result of low temperature acting as a sensitiser to the effects of pollutants (Dethlefsen *et al.* 1996). Earlier research work identified the marine microlayer as a critical component of the ecosystem. This thin layer of lipid rich material at the sea surface can concentrate organic pollutants which in turn can adversely affect the fish larvae which develop in contact with it (von Westernhagen *et al.*1987). There is some evidence that the survival of fish larvae is determined to some degree by the exposure of the adults to toxic chemicals. For example, in a polluted area of San Francisco Bay, more highly exposed female starry flounder laid fewer viable eggs. These were less successfully fertilised and aberrant embryological developments were more common than in eggs laid by less exposed fish (Spies *et al.*1988). Such findings imply considerable disruption of biochemical and physiological controls and recent findings have highlighted the ability of some chemicals to disrupt the hormone systems which control fundamental aspects of fish reproduction.

A growing number of chemicals have been shown to interfere with hormone pathways in a wide range of species, including humans. This should come as no surprise in the case of many pesticides which are actually designed to interfere with these systems. The trichlorophenoxyacetic acid herbicides (2,4-D; 2,4,5-T; MCPA), for example work by disrupting the normal growth hormone systems of plants. Various insecticides also disrupt the hormonal systems controlling moulting and development of insects (Hassal 1990). TBT is another well known chemical which interferes with the hormonal systems of marine molluscs (Weis & Weis 1994). The herbicides atrazine and simazine have also been shown to exert powerful disruption of oestrogen hormone systems (Stevens *et al.* 1994). Disruption of fish endocrine systems has been detected in fish exposed to pollutants. The reproduction of affected fish has been found to be particularly seriously affected.

Many chemicals are capable of disrupting endocrine systems (Colborn & Clement 1992; Colborn 1994; Toppari *et al* 1995) partly as a result of their ability to induce liver enzymes normally involved in regulating the oestrogen-like hormones or by otherwise affecting levels of circulating hormones. A recent extensive review has concluded that the effects of chemicals upon fish include lesions, haemorrhage or malformations in the gonads. Malformations of these organs together with the pituitary and liver can inhibit the secretion, production and metabolism of hormones by the endocrine system (Kime 1995). A wide range of environmental contaminants, in fact, are known to have the potential to disrupt endocrine systems. The major known group are representatives of the class of organochlorines although these are by no means the only ones. Accordingly, in benthic fish exposed to contaminated sediments levels of circulating hormones were found to be reduced and the females found to be less likely to spawn (Johnson *et al* 1993).

Winter flounder exposed to contaminated sediments were less able to manufacture egg yolk, vital to the survival of the larvae (Pereira *et al.*1992). This in turn,could lead to the reduced egg and larval viability noted above (Spies & Rice 1988; Collier *et al.* 1993). These effects can be maintained over long periods. Consistent low egg viability, and larval size has been observed over three consecutive years in one population of winter flounder. Hence, this is not a transient, but a chronic long term, response (Nelson *et al.* 1991). The contaminants present were varied in nature but included PAHs, other hydrocarbons and PCBs.The chlorinated dioxins too, are powerful endocrine disrupters and act on liver enzyme pathways. In higher vertebrates at least twenty two other groups of halogenated chemicals with similar properties have also been identified (Geisy *et al.* 1994).

Not all such effects are produced by induction of liver enzymes and subsequent interference with hormonal pathways. Some chemical effects may be direct. Perhaps the most dramatic example of endocrine disruption in fish has been documented downstream of sewage effluent discharges into river systems (FWR 1992). In this case, male fish acquired pronounced female characteristics, in particular, the ability to produce a precursor to egg yolk. This process is normally under the control of oestrogens produced in the ovaries of females. The chemical or chemicals causing Have been identified as synthetic hormones reactivated after passage through sewage treatment works, but the same effect has also been shown by some industrial chemicals discharged to rivers. Detergent breakdown products (nonyl phenols) are one of a number of known endocrine disrupters. The detergents based on nonyl-phenols are in wide use and are known to be able to influence a wide range of biological functions, generally with negative effect (Cserhati 1995). Large quantities of surfactants are released in sewage discharges, and the nonyl-phenols are also released in large quantities by the factories which manufacture them (Law *et al.* 1991;Kuhnt 1993).

In addition, reproduction may be affected by the ovaries of fish taking up chemical contaminants directly. Even cod from relatively remote areas are known to have organochlorine contaminants in their ovaries (Hellou *et al* 1993) as well as flatfish from more polluted areas (Loizeau & Abarnou 1994). Many organochlorines such as PCBs, DDT, toxaphene and mirex are known endocrine disrupters but may also act directly on the reproductive tissues. In the North and Baltic Seas positive correlations have been made between the levels of these contaminants and reproductive success. Contamination of the ovaries of fish by organochlorines has been correlated with a lower hatching success in a variety of fish species in the Baltic Sea (von Westernhagen *et al.* 1987; von Westernhagen *et al.* 1987). The gametes themselves are known to be highly sensitive to toxic chemicals (Kime 1995).

There is no doubt, therefore, that pollution is capable of affecting fish in a variety of ways. The most important impacts are upon health and reproduction. This could compromise the ability of populations to replace and maintain themselves. A full understanding of stock-recruit relationships has proved elusive but reproduction and larval viability is regarded as a fundamental determinant of fish stocks. Impairment of reproductive potential of fish species, therefore could result in depletion of fish stocks. Population crashes have been recorded in birds of prey and seabirds as a result of pesticide use. Effects could be exerted on two levels. Reduction in the fecundity of breeding fish could result in fewer viable larvae. Pollution effects could also be exerted upon the juvenile stages which are the most sensitive to pollution (Sprague 1971; Weis & Weis 1994).

The degree to which these established effects are influencing fish stocks is a matter for conjecture. Lack of hard evidence at the population level in marine waters is merely a reflection of the natural size and variability of fish populations coupled with a lack of suitable techniques for the reliable detection of changes. It is likely that detection of any effects will result only from catastrophic decline of any given fishery which is not under fishing pressure. It is unlikely that the involvement of pollution as a causal factor in the rebuilding of a depleted stock could be isolated from the multifarious other environmental factors in operation. Modelling techniques however allow some appreciation of the scope of problems following acute toxic incidents.

Pollution of marine habitats can prevent fish reaching their full reproductive potential. Modelling exercises have shown, unsurprisingly that large scale removal of larval and juvenile fish will result in a decline in adult stock if removed physically (Nisbet *et al.* 1996). The same could be true of removal by pollution. The possible impacts could be sufficient to not only prevent stocks recovering but in some cases to actually drive stocks down. The pollution of seas and coastal waters is a continual process and the progressive degradation of marine habitats in this way poses a threat to fish populations of considerable scale.

Official evaluations of the state of the marine environment have signally failed to include a consideration of the potential effects of chronic exposure to low levels of toxic chemicals upon fish populations. This is a serious omission given the fact that disease syndromes observed in both laboratory and field studies clearly demonstrate the potential negative impacts of pollution upon fish. Demonstrated effects include behavioural alterations, developmental abnormalities, liver disease, cancer, interference with sensory mechanisms and skin disease. There is a possibility also that reduction in the genetic diversity of fish populations exposed to toxic chemicals may also be an important factor. Recent findings of major importance to the population dynamics of fish include a reduced production of eggs from adult fish exposed to toxic chemicals, reduced viability of eggs and larvae and an increase in the numbers of deformed larvae. Research into the consequences of long term low level exposure of fish to toxic chemicals is still very much in its infancy. Provisional modelling exercises, however, predict significant impacts upon fish populations and these are likely to be most serious in populations already under pressure from over fishing since they will further reduce recruitment.

4.5 Alternative Management: The Precautionary Approach to Fisheries

Considered in relation to the four provisional principles of sustainability (Section 1) it is clear that current fisheries are disturbing ecosystems to a considerable degree. Evidence for these changes can be seen generally in the trend of fishing down trophic levels in ecosystems on a global scale. Regional examples of collapsed fisheries abound together with increasingly wide documenting of local impacts of various kinds. These have as a foundation a widespread failure of management which in turn is rooted in poor or incomplete understanding of the exploited ecosystems. These shortfalls result in problems with the precision and accuracy of the predictive tools which underpin of management decisions. In addition, this results in fishery resources not being used in a fair and efficient manner. The four case studies illustrate the need for a management system which allows sustainable use of fisheries resources.

As the problem of overfishing grows in scale, it is becoming increasingly appreciated at both the scientific and political level that the current management paradigm is fatally flawed. It has become clear that simple extension of jurisdiction to 200 miles does not solve the problems of over-exploitation if adequate control is not exerted on the remaining, domestic fisheries. This recognition has led to scientists and politicians alike calling for fisheries management to be established on a different footing (Stephenson & Lane 1995; IMM 1997a). The elements of this fall into several categories. Making uncertainty in stock estimates needs to be made explicit as well as defining operationally feasible strategies. It should be recognised that fisheries management has a different focus from fisheries science. Closing the gap between the two is necessary. Major improvements will require more than a subtle modification of the *status quo.* Conceptual and organisational change is necessary so that decisions are based upon rigorous analysis and predefined objectives which accommodate the full context of biological, socioeconomic, operational and political elements. Quantification and explicit definition of the various interacting factors should be carried out (Stephenson & Lane 1995; Lane & Stephenson 1995; Charles 1995; Clay & McGoodwin 1995). A further element in successful fisheries management is a robust means of enforcement of the imposed management regime (see Caddy 1995). It is necessary too, to move away from the single species management approach to accommodate the complexities of whole ecosystems (Botsford *et al.* 1997). In addition it will be necessary to recognise the limitations inherent in multi species models together with the interactions of a variety of ecological factors (Conover *et al.*1995).

In broad terms what is being defined by these requirements is a precautionary approach to fisheries. The need for such an alternative management strategy was explicitly recognised in the 1995 UN Treaty for the Conservation and Management of Sraddling Fish Stocks and Highly Migratory Fish Stocks. Specifically, Article 5 contains "General Principles" for fisheries conservation and management. The UN FAO has produced a Code of Conduct for Responsible Fisheries (Garcia & Newton, 1994). In 1997, North Sea States (IMM 1997b) agreed to apply a precautionary approach to management of living marine resources following the UN initiatives. These are significant steps, and represent the first political acknowledgements that considerable uncertainties exist in fisheries models and management techniques. In applying a precautionary approach, the aim is to provide a "buffer" for uncertainties in order to safeguard the integrity of the population or ecosystem in question (Dayton 1998).

As with chemical regulation, the intention is to reverse the burden of proof such that exploitation is not carried out unless a very high degree of certainty exists that it will be sustainable. Some comparisons of existing management schemes have been made with schemes designed to introduce precaution into management (Caddy 1997) and it is clear that precaution as a concept in fisheries is some way behind the concept as it is now applied to chemical contaminants. These management schemes will need to develop rapidly in order to regulate fisheries within defined precautionary limits. Much depends upon defining uncertainty and reducing TACs as uncertainties increase. The danger is that unless carefully formulated, this could merely represent a "business as usual" scenario rather than a mechanism through which the predominance of socio-economic considerations in exigent situations can be eliminated. The essential element of a precautionary approach is to restrict levels of fishing effort and the impact upon marine populations so that the character of the ecosystem does not change as a result of fishing. This applies to target fish stocks, by-catch and habitat disturbance. Fisheries should not be allowed to operate in the absence of data on the target population and catches should be set lower where data quality is poor. A well planned response strategy will need to be evoked if limits are exceeded or unanticipated events take place.

In order to achieve stability in global fisheries and put them on a sustainable footing under a precautionary paradigm, some actions that need to be taken stand out clearly from the data. The global fishing capacity needs to severely reduced. Given current overcapacity, this reduction should be in the order of 50% for the large scale industrial fleet including factory trawlers, longliners and large multi purpose vessels. This should be implemented as soon as possible. Regional efforts to cut capacity should be strengthened. Subsidies for the industry should be eliminated as part of the drive to reduce overcapacity initially focused on destructive fishing practices, large scale factory trawlers, distant water fleets and new boat construction. Reflagging should be prevented. As part of these efforts, the UN Agreement on Straddling Stocks should be ratified and implemented with some urgency, while the FAO Code of Conduct should be effectively applied in practice.

5. Aquaculture

5.1 The development of aquaculture

Some forms of aquaculture have been practised for several thousand years, particularly in freshwater systems (Iwama 1991). Historically this took the form of measures to enclose and protect stock, often utilising the natural productivity of the system, in order to enhance returns of important subsistence foodstuffs. In some areas, such extensive systems remain important sources of local foodstuffs. More recently, however, aquaculture has also become big business, making significant contributions to the global economy and providing high proportions of national product, especially in the economies of South East Asia. The rapid growth in demand for certain aquaculture products, particularly high value marine products such as salmon and shrimp, has led to widespread development of intensive production facilities in coastal regions of Northern Europe, North America, Latin America and Asia.

In 1985 the FAO estimated global aquaculture production at 10.6 million tonnes y^{-1} and conservatively predicted a doubling of production by 2000 (Iwama 1991). Most recent estimates from the FAO (1996) indicate total aquaculture production in 1994 reached a record 25.5 million tonnes, and contributed $39.83 billion to the global economy. This represents an increase in production of 11.8% over 1993, indicative of the continuing rapid expansion of the industry. Much of this expansion remains centred in Asia which now dominates the global aquaculture market, supplying approximately 80% of total production by mass (FAO 1996). Freshwater finfish remains the largest bulk of aquaculture production (44%), much of this produced in China, with freshwater and marine molluscs adding a further 17% and plant culture (seaweeds and higher plants), 27%. Culture of marine finfish and crustaceans account for relatively small proportions of total production by mass (1.8% and 4.2% respectively) but, given the high commercial value of the products, contribute more substantially in economic terms. Crustaceans, for example, represent just over 4% of global production by weight, but contribute 18% of total economic value (FAO 1996). In Malaysia, intensive shrimp culture generates 4.3% of total by mass, but accounts for 44% of national income from the aquaculture sector (Omar *et al.* 1994). In addition, as the vast majority of marine species aquaculture is necessarily located on the coastal fringe, in estuaries or in shallow coastal waters, potential and actual impacts at this sensitive interface can be substantial.

To date, much of the research and information on developments, practices and impacts of marine species aquaculture relates to salmonid production in the temperate waters of Northern Europe and North America (Iwama 1991, Wu 1995). This remains an expanding industry. Enell (1995) reported an increase in overall aquacultural production in the Nordic countries from 15800 tonnes in 1974 to 250 000 tonnes in 1994. Much of this expansion took place in Norway; intensive cage production of Atlantic salmon (Salmo salar) in Norway rose from 8000 tonnes in 1980 to 160 000 tonnes in 1990 (Johnson *et al.* 1993, Heggberget *et al.* 1993). Expansion has also been rapid in the UK over the past 25 years (Neiland 1990), total production increasing 10 times between 1984 and 1991. Virtually all sea lochs in Scotland are now used for culture of Atlantic salmon (Thompson *et al.*1995), generating 18.2% of global salmon production in 1991. Following earlier expansion of the industry in Denmark, the number of salmon farms declined from 800 in 1974 to about 500 today, largely a result of the introduction of relatively stringent controls on husbandry and environmental impact (including the banning of wet feed under Statutory Order of 1985, Environmental

Protection Act 1974). Despite the decline in farm numbers, production continued to increase until 1989 as technical developments allowed increased stocking densities (Iversen 1995).

The development of marine aquaculture in other regions, particularly Asia and Latin America, has been extremely rapid over the last 10 to 20 years. To some extent, coastal aquaculture is traditional in South East Asia (Eng *et al.* 1989). Cage culture is practised for some high value finfish species (see Chou and Lee 1997), and culture of bivalves is locally significant. Nevertheless, much of the expansion in these regions has been in intensive and semi-intensive production of shrimps (*Penaeus* spp.) in artificially constructed coastal ponds (Gujja and Finger-Stich 1996, Beveridge *et al.* 1997b). The pattern of development in Malaysia has been typical in this regard, the focus shifting from freshwater fish in the 1950's and 60's, to marine and raft culture in the 1970's and finally to shrimp in the 1980's and 90's (Omar *et al.* 1992).

Beveridge *et al.* (1997a) estimate that shrimp production in Asia has increased from 100 000 tonnes in 1984 to 650 000 tonnes in 1994, accounting for a large proportion of the estimated 733 000 tonnes y^{-1} shrimp cultured globally (Gujja and Finger-Stich 1996). Of this, Thailand was the single largest producer, responsible for 225 000 tonnes, up 45% on 1993. Production in Taiwan, Indonesia and the Philippines reached 20 000, 40 000 and 80 000 tonnes respectively in 1995 (Kongkeo 1995). Development has also been extremely rapid in Sri Lanka (Corea *et al.* 1995) and in many Latin American countries.

Expansion has resulted both from the explosive increase in the number of intensive culture operations in coastal areas and improvements in yield per unit area through the application of high value feedstock and disease control agents. Traditional extensive methods of shrimp production, involving entrapment in tidal ponds and growth sustained on natural levels of productivity, yields between 100 and 500kg shrimp per ha. Intensive pond culture can increase this to 1000-10000 kg per ha (Gujja and Finger-Stich 1996).

Development of the industry in Thailand, and in Asia as a whole, is primarily a response to demand for high value fish products from developed world (Flaherty and Karnjanakesorn 1995). Extensive shrimp culture began in Thailand in the 1930's, largely supplying local needs. Semi-intensive shrimp culture was first promoted and supported by the Thai Government in 1973, resulting in rapid expansion. In 1972, there were 1154 intensive facilities, covering little over 9000 ha and generating under 1000 tonnes shrimp y^{-1}. By 1990 this had increased to 15 278 farms, occupying 64 081 ha of coastal land and yielding 118 227 tonnes. Much of this increase took place between 1987 and 1991. Although 30% of Thai shrimp production occurs inland (Kongkeo 1995), production remains focused in coastal regions. The majority of intensive operations in Thailand are owned and managed by urban businesses, employing local people from coastal communities world (Flaherty and Karnjanakesorn 1995). Traditional economies and local habitat are frequently negatively impacted. In general, little consideration has been given to protection of coastal habitat or water quality or to the development of overarching management plans on a regional basis. The common goal has been to maximise profits in short-term intensive operations, often involving the abandonment of ponds as soon as yields begin to fall, frequently as a result of inadequate waste management, poor husbandry and the inevitable spread of disease. This model of resource intensive production, likely to be typical of intensive operations in developing countries, is ultimately unsustainable. In Taiwan, for example, which developed intensive shrimp culture earlier than other Asia states, production crashed in the late 1980's due to high costs, pollution and disease (Flaherty and Karnjanakesorn 1995).

Intensive production is also expanding in other aquaculture sectors. Chou and Lee (1997) suggest that further intensification of cage culture may be essential to meet demands for finfish in Singapore. Increasing demand for agar has led to moves towards pond culture of the seaweed *Gracilaria* in Namibia (Critchley *et al.* 1991). Welcome (1992) recognised that many environmental problems are associated with practices aimed at enhancement of aquaculture production in the coastal zone, including reduction of water quality, habitat destruction, depletion of wild stock and impacts on wild breeding populations. Clearly the problems in any one region will depend on the type and location of aquaculture operations, and the management techniques adopted (Wu 1995), but there is little doubt that the spread of marine aquaculture has contributed in some measure to the poor state of coastal ecosystems worldwide. Despite the fact that much of existing research relates to finfish and bivalve culture in Europe and North America, environmental impacts even in this sector are complex and remain difficult to predict (Silvert 1992). As Iversen (1995) notes, despite heavy investment in technology, the problems experienced in the Danish aquaculture industry are substantial. Ironically, information on impacts and investment in control strategies in Asia and Latin America, where perhaps the most severe and rapidly increasing problems are being observed, remain extremely limited (Iwama 1991).

5.2 Habitat loss and consequences for biodiversity

Aquaculture operations have the potential for a range of direct and indirect impacts on habitat and biodiversity, including habitat destruction or modification, species selection and depletion of natural populations through collection of brood and feedstock and the control of predator and disease vector organisms. In their review of aquaculture operations globally, Beveridge *et al.* (1994) note that the impacts on biodiversity are "rarely positive, sometimes neutral but usually negative to some degree".

The potential for habitat destruction is perhaps best exemplified by the large-scale clearance of mangroves in Asia and Latin America for the construction of intensive aquaculture facilities. Mangroves are sensitive habitats which are integral to the cycling of carbon and nutrients in many tropical and sub-tropical coastal ecosystems. Processes in coastal seas may be closely linked to mangrove systems; for example, carbon balance in sea grass beds appears to be tightly coupled to coastal mangrove dynamics (Hemminga et al 1994). Mangroves provide spawning and nursery areas for a wide range of fish and crustacean species (Laegdsgaard and Johnson 1995, Flaherty and Karnjanakesorn 1995), including wild populations of the penaeid shrimp (Newell *et al.* 1995) from which pond cultures are stocked. For an overview of biodiversity and ecology of mangroves in Latin America and Asia, see Lacerda *et al.* (1993) and associated references. In addition, mangroves represent important buffer zones at the land-sea interface. Their removal can greatly upset water balance in the coastal zone, lead to saline intrusion to soils inland and increase exposure of coastal communities to severe weather events.

Of the 80 592 ha of mangrove converted in Thailand between 1961 and 1986, 38% was turned over to aquaculture, the single largest contributor (Flaherty and Karnjanakesorn 1995). Intensive shrimp production is the fastest growing sector. Dierberg and Kiattisimkul (1996) estimated that of the total area of mangrove destroyed in Thailand between 1979-93, 16-32% was converted to shrimp culture. Other estimates suggest that shrimp ponds accounted for 64% of the total mangrove area developed between 1986 and 1989 (Flaherty and Karnjanakesorn 1995). In the Philippines, aquaculture has contributed significantly to the removal of 75% of the original mangrove habitat (Iwama 1991). Similarly, in Ecuador, the rapid growth of shrimp production had claimed 12 000 ha of virgin mangrove by 1987 (Wilks 1995, in Bruce's).

The lifetime of intensive ponds is relatively short, primarily as a result of the accumulation of pathogens and greatly increased likelihood of disease outbreaks even after a small number of production cycles. Moreover, ponds reclaimed from mangrove become too acidic to support penaeid shrimp stocks within few harvests (Eng *et al.* 1989). Dierberg and Kiattisimkul (1996) estimate a mean lifetime for Thai ponds as 7 years, although substantially shorter lifetime's are possible. Catastrophic or gradual decline in pond utility inevitably leads to abandonment and pressure to clear and develop new areas. In the north of Thailand, 10 120 ha of intensive shrimp ponds developed from mangrove were abandoned between 1989 and 1991 as a result of declining water quality or disease outbreaks (Flaherty and Karnjanakesorn 1995). Similar problems are now being observed in the south. The development of mangroves for aquaculture involves, therefore, short-term investment from which high returns are essential in the first few years. Such unsustainable "shifting aquaculture" operations supply a large proportion of global shrimp production.

While the areas of the ponds themselves are substantial, the total area impacted through development of shrimp culture may be much larger. Beveridge *et al.* (1997b) noted the need to consider the areas used for construction of roads, dykes, canals and other infrastructure related to mangrove development. Moreover, much larger areas are generally used for the collection of broodstock to seed the ponds. Larsson *et al.* (1994) calculated that the "ecological footprint" of an intensive shrimp facility in Columbia may be 35-190 times larger than facility itself. Indeed, as much as 20-50% of the total mangrove area in Columbia has been exploited to supply postlarval forms (Larsson *et al.* 1994).Exploitation of mangrove systems can result in serious impacts on other systems, particularly the estuaries and coastal waters which the mangroves fringe (Eng *et al.* 1989). Flaherty and Karnjanakesorn (1995) highlight the potential for negative impacts on inshore fisheries through removal or modification of nursery grounds. Indeed it has been estimated that the production of 120 000 tonnes of shrimp in Thailand may have resulted in a loss of potential yield from inshore fisheries of as much as 800 000 tonnes (Wilks 1995, in Bruce's). Eng *et al.* (1989) also point out that the destruction of mangroves and the collection of broodstock from wider areas can have severe impacts on non-target species. By-catch of non-target larvae or post larval forms associated with shrimp culture in the Bay of Bengal Sanderbans, India, can be as high as 64-99% (Beveridge *et al.* 1997b).

Sea grasses (*Zostera* spp.) also provide important nursery grounds for finfish and shellfish in shallow coastal waters (Short and Wyllie-Echeverria 1996). Stands of sea grass have been in decline globally for a number of years, resulting from a variety of environmental pressures including the onward effects from the destruction of mangroves. Some aquaculture operations can also have more direct impacts on sea grass stands. For example, commercial culture of oysters in the Pacific Northwest of the USA has resulted in dramatic reductions in cover of submerged aquatic vegetation in the vicinity of the operations (Everett *et al.* 1995). Elimination of sea grasses effects, in turn, sediment transport, nutrient and carbon cycling and biodiversity over a wider area. Both stake and rack type oyster culture operations can cause rapid reduction in sea grass cover through a combination of physical disturbance during placement and harvesting, increased siltation downstream of the cultures and increased shading. Sea grass cover was reduced by as much as 75% within the first year in the vicinity of new rack culture operations, with complete elimination within 17 months (Everett *et al.* 1995). Oyster culture is a growing sector of the aquaculture industry in the US, so damage may be expected to become increasingly widespread in the future.

Although perhaps the most obvious and devastating impacts of aquaculture operations on habitat relate to intensive pond culture of shrimps and other organisms in coastal areas, cage culture of marine species may also have significant indirect impacts. The spread of salmonid culture in Scottish sea lochs has inevitably led to the disturbance of natural populations of seabirds and marine mammals (Thompson *et al.* 1995). Commercial interests in protecting stock may result in conflict with potential predators. Although reporting of predator control measures is generally poor, shooting, trapping and destruction of surrounding habitat may be common in some regions (Price and Nickum 1995). Impacts of coastal aquaculture operations on biodiversity may also be apparent further offshore, particularly as feedstock for cage culture operations are frequently taken from offshore fisheries. Thompson *et al.* (1995) note that capelin, sprat and sand eel are commonly exploited as feedstock for the culture of salmonids in temperate waters. Globally, aquaculture feedstock may account for between 13 and 15% of total fish oil and meal production (Beveridge *et al.* 1997b), contributing to the overall decline in stocks of sand eel and other species in some regions. Aspects of marine aquaculture operations other than physical habitat modification and destruction may also have significant impacts on biodiversity; these are discussed in detail below.

5.3 Eutrophication – Organic and Nutrient Enrichment

Many intensive aquaculture operations, either in cages or coastal ponds, utilise high loadings of nutrient rich feedstock in order to maximise production. The quantities of waste produced, in the form of uneaten food, faeces and dissolved organic and inorganic matter can be substantial. Much of the information on the quantities of waste produced and resulting impacts relates to temperate cage culture of salmonids, although some of the problems may be common to all intensive operations. A high proportion of the feed applied to salmonid cages is lost from the system unassimilated as dissolved or particulate waste (Sandifer and Hopkins 1996). Salmonid feed can contain more than 7% nitrogen by weight, 67-75% of which may ultimately be lost to the environment (Iwama 1991). Handy and Poxton (1993) estimate that, in some operations, up to 95% of the nitrogen supplied in feed may go unassimilated, 30% being egested in faeces. The quantities of organic matter and dissolved nutrients lost to the surrounding water column or to the underlying sediments, even from relatively small operations, can, therefore, be substantial. Ackefors and Enell (1994) estimate that 2.5 tonnes of organic waste may be generated for every tonne of cage-farmed salmon produced, while Penczak *et al.* (1982, in Thompson et al 1995) calculated that as much as 100 kg nitrogen and 23 kg phosphorus may be released to surrounding waters per tonne of rainbow trout in freshwater culture. The accumulation of organic matter beneath salmonid cages can exceed 5 kg m^{-2} y^{-1}.

Recent developments of reduced nutrient feed (Kolsater 1995), prompted in some instances by the tightening of regulatory controls (e.g. the banning of wet feed in Denmark, Iversen 1995), may help reduce emissions of nitrogen, phosphorus and organic matter, this does not eliminate the problems. For example, Enell (1995) reported that operations utilising reduced nutrient feed may still release 55 kg nitrogen and 4.8 kg phosphorus per tonne of salmon to receiving waters. Total nutrient loadings from cage culture operations in Nordic states to the Baltic and Skaggerack (13 750 tonnes N and 1 200 tonnes P y^{-1}) may represent a relatively small contribution to overall anthropogenic enhancement of nutrient budgets but may be locally significant nevertheless. The significance of releases from individual farms may be greatly enhanced by virtue of the common location of such cage farms in relatively sheltered embayments and coastal waters.

Data on feeding regimes and the generation of dissolved and particulate wastes are limited for aquaculture operations in developing countries, although it is unlikely that the strict controls in some Nordic countries would be applied routinely (Eng *et al.* 1989). It is recognised that many cage and pond farms rely heavily on imported food material in order to maximise stocking densities (Kongkeo 1997) and the accumulation of dissolved inorganic nitrogen, particularly ammoniacal nitrogen, is a significant problem in intensive shrimp ponds (Lorenzen *et al.* 1997). Ammoniacal nitrogen can be highly toxic to many marine species, and frequent water exchange, releasing high levels of nitrogen to receiving waters, may be essential to prevent its build up in pond waters.

The generation of total ammoniacal nitrogen (TAN) in intensive cage and pond cultures is difficult to model and control even in temperate systems, requiring accurate estimates of stock size and density and frequent monitoring regimes (Kelly *et al.* 1994, Hennessy *et al.* 1996). Rates of TAN generation tend to increase with temperature, so that problems may be exacerbated in sub-tropical and tropical systems. Furthermore, Lorenzen *et al.* (1997) note that the stocking densities and water exchange regimes commonly practised in intensive shrimp pond operations tend to be ineffective in controlling the accumulation of TAN or nitrite, also toxic to marine life, or their release to receiving waters. Even if the excess nitrogen is assimilated by phytoplankton before leaving the ponds, this will still result in enrichment of receiving waters when the ponds are flushed.

The large quantities of organic matter which accumulate at the bottom of intensive ponds (as much as 1 tonne ha^{-1}) are also rich in nitrogen and phosphorus. It is estimated that the 40 000 ha of intensive shrimp ponds in Thailand generate more than 16 million tonnes (dry weight) of organic rich sediment every year (Dierberg and Kiattisimkul 1996). In some instances, where access and finance permit, the sediment is excavated from the ponds for disposal on land, although very little information exists as to the final fate of such material. Frequently, however, the sediment is simply washed from the ponds with high pressure water hoses into estuaries and coastal waters (Flaherty and Karnjanakesorn 1995), adding substantially to the nutrient loadings of the systems. Dierberg and Kiattisimkul (1996) estimate that the total quantities of nutrients released to water from current levels of shrimp production in Thailand are equivalent to that expected for sewage discharges from between 3.1-3.6 million (N) and 4.6-7.3 million (P) people. The rapid development of pond culture over the last few years may, therefore, have an impact equivalent to an increase in coastal human population, without sewage treatment, of between 50 and 100%. While the impacts of such discharges have not been studied in any detail, they may be expected to be substantial in some areas.

Much of the research on the impacts of high organic and nutrient releases from aquaculture relates to cage culture in temperate waters. Organic enrichment of marine sediments is readily detectable beneath and down current from marine fish farms in Norwegian waters (Johnsen *et al.* 1993, Black *et al.* 1996), although clearly the extent of accumulation will depend greatly on local hydrographic conditions. Holmer and Kristensen (1992) reported that sediment metabolic activity may be increased by a factor of 10 during the annual production cycle beneath freshwater cages and that such elevation may continue throughout the winter season. Persistence of enrichment has also been reported in marine sediments (Johnsen *et al.* 1993). The accumulation of sulphides in anoxic sediments beneath cages, and their subsequent release to overlying waters can, in turn, have negative impacts on the growth of the fish in the cages (Black *et al.* 1996).

Although both cage and pond aquaculture undoubtedly lead to local enhancement of nutrient loading, information on direct impacts is extremely limited. Perhaps the clearest evidence relates to the stimulation of macrophyte (seaweed) growth in shallow embayments. Downstream from a rainbow trout farm in Denmark, natural populations of the brown seaweed *Fucus vesiculosus* were progressively overtaken by fast growing green algae (e.g. *Enteromorpha spp*.), either as epiphytes on the fucoids or, closer to the farm site, as the dominant benthic algae (Ronnberg *et al.* 1992). Similarly, De Casabianca *et al.* (1997) reported a clear gradient of enrichment associated with shellfish culture in the Thau lagoon (France). Natural eel grass communities showed greatly increased epiphyte growth and, closer to the cultures, were replaced by the ephemeral green species *Ulva rigida*, or the red genus *Gracilaria* where shading was also significantly enhanced.

Impacts of enrichment from aquacultural activities on plankton productivity may be expected, particularly in waters where exchange rates are relatively low, but direct evidence is very limited. Some studies have suggested an association between aquaculture activities and the increasing incidence of nuisance algal blooms in coastal waters (Kaartvedt *et al.* 1991, Arzul *et al.* 1994), although the relationship appears complex and uncertainties remain high. It has been known for some time that a relative abundance of ammoniacal nitrogen over other inorganic forms can favour the selection of red tide dinoflagellates (Takahashi and Fukazawa 1982). Kaartvedt *et al.* (1991) suggested that the Swedish fish farms which were devastated by the extensive blooms of the toxic flagellate *Prymnesium parvum* in the late 80's may, in turn, have been partially responsible for the stimulation of the blooms in the first instance. A higher incidence of the toxic dinoflagellate genus *Gymnodinium* was reported by Arzul *et al.* (1994) in areas of high density intensive cage farming. Indeed, Arzul *et al.* (1996) have since demonstrated that elutriates from certain aquaculture feedstocks can stimulate *Gymnodinium* spp., although other genera showed inhibition. It is currently unclear which components of the feed were responsible for species selective effects. Graneli *et al.* (1993, in Wu 1995) noted that vitamin B12, a component of some artificial feeds used in cage culture of salmonids, stimulated growth of the toxic species *Chrysochromulina polylepis* and *Heterosigma akashiwo*. In addition to overall enrichment, therefore, aquaculture operations may initiate fundamental shifts in plankton community structures (Arzul *et al.* 1996). It has also been suggested that increasing prevalence of the predatory dinoflagellate *Pfiesteria* spp. in coastal waters may be linked to localised eutrophication, including that arising from intensive aquaculture operations (Burkolder *et al.* 1995, Pelley 1997).

Impacts may be even more difficult to detect over wider areas and in deeper water. Complex hydrographic and biological regimes generally make the processes of identifying long-term stimulation of phytoplankton production and of the enrichment sources responsible extremely difficult. There is some evidence that intensive mussel culture on the Atlantic coast of Spain may have resulted in progressive enrichment of over 300 km2 of the continental shelf and the dampening of the effects of periodic upwelling of nutrient rich water (Tenore *et al.* 1982), but such examples are rare. Indeed, the debate on the significance of coastal aquaculture as a source of nutrient enrichment and eutrophication has, on occasion, been a heated one (Folke *et al.* 1994, 1997, Black *et al.* 1996). In summary, Folke *et al.* (1997) stress that the impacts of enhanced inputs may not always be manifest as an increase in the standing stock of phytoplankton biomass, as a proportion of the increased productivity may be carried through to higher trophic levels or to the marine microbial community. The authors conclude that, taking a more generic definition of eutrophication as an increase in the presence and production of organic matter in a system (Nixon 1995), coastal aquaculture undoubtedly contributes to the eutrophication of coastal waters.

5.4 Energy and Water Consumption

Intensive culture of finfish and shellfish is, by definition, expensive in terms of energy use. Construction and maintenance of cages, ponds and associated infrastructure, collection and preparation of broodstock and feed species, control of disease and predators, water exchange, waste management and final harvesting of product all demand substantial energy inputs which may, in total, greatly exceed the energy yield of the aquaculture product itself. For example, Larsson *et al.* (1994) calculated that for every Joule of edible shrimp protein produced in intensive pond cultures, 295 Joules of energy were expended. More than 80% of the required energy has to be imported to the pond, often from long distances (e.g. feedstock). Overall, intensive shrimp culture is one of the most resource intensive food production systems in operation.

Coastal pond culture of marine and brackish water species also presents high demands for water. Daily water exchange from intensive cultures can be as high as 40% by volume (Dierberg and Kiattisimkul 1996), largely in order to control the accumulation of dissolved and particulate wastes in the pond. Gujja and Finger-Stich (1996) estimate that the production of 1 tonne of shrimp can utilise and, therefore, contaminate 50-60 million litres of seawater, or a mixture of sea and freshwater. Beveridge *et al.* (1997b) present estimates of a similar order (29-43 million litres $tonne^{-1}$). Where species are grown in brackish water, common practice for some penaeids, high demands for freshwater may also exist. Abstraction of freshwater for shrimp culture in the Philippines (Kongkeo 1997) involves high energy expenditure and has led to extensive land subsidence, saline infiltration and potential conflict with other water users.

5.5 Use of Chemicals for Disease Control

Intensive aquaculture greatly increases the risks and consequences of disease outbreaks by concentrating many individuals in a small volume, allowing wastes to accumulate in ponds or beneath cages and maintaining continuous production cycles for many years (e.g. Pearson and Inglis 1993, Buchmann *et al.* 1995. These diseases which, in natural populations, may account for significant mortality can, in turn, devastate stocks of intensively farmed finfish or shrimp over wide areas. For example, almost half of Peru's shrimp farms were recently devastated by hepatopancreatitis before the agent could be identified and corrective measures taken (Lightner and Redman 1994). Similarly, shrimp production in India and Indonesia has suffered recent declines due largely to widespread outbreaks of disease (Gujja and Finger-Stich 1996). Moreover, the likelihood of large-scale disease outbreaks increases year on year for intensive ponds (Dierberg and Kiattisimkul 1996). High rates of disease and mortality are also characteristic of farmed finfish, particularly those utilising hatchery reared broodstock (Buchmann *et al.* 1993, 1995).

A wide range of chemical agents may be added to aquaculture cages and ponds, or used to pre-treat broodstock, in order to control viral, bacterial, fungal and other pathogens. These chemicals, which include antibiotics, various pesticides, detergents and other chemicals used to control water quality, are used in large quantities by the aquaculture industry globally (Wu 1995). Although there are some controls on use, information on the types and quantities of chemicals employed in many countries is very scarce. The final fate of many of these chemicals is even less well understood, although it is likely that, for some, a high proportion is lost to the environment during water exchange, harvesting or disposal of waste sludges (Sandifer and Hopkins 1996). Data on the quantities of chemicals used in shrimp aquaculture and their impacts are extremely limited, especially for Asia where much of the production is centred (Tacon *et al.* 1995). Nevertheless, information available from other aquaculture sectors, particularly marine fish cages, provides a useful illustration of the problems involved.

Many of the chemicals used to control disease are toxic to a range of species, persistent in the environment and able to accumulate in biological tissues. In addition to the direct hazards to those applying the treatments, the potential for contamination of the surrounding environment is therefore high. Furthermore, the accumulation of residues from chemical treatments in the products themselves presents serious concerns for public health.

a) Antibiotics

A range of antibiotics are used worldwide to prevent and treat bacterial conditions in aquaculture stock. The tetracyclines, especially oxytetracycline, and the quinolones, including oxolinic acid and flumequine, are among the most widely used in commercial aquaculture. Others, including chloramphenicol, are now banned in Europe because of the hazards of the development of antibiotic-resistance in human pathogens (Robert 1996), but are still in use in other parts of the world. Antibiotics may be administered directly to marine cages, pond water or, increasingly, through the food supply (see, *e.g.* Nelis *et al.* 1996). Data on quantities used in aquaculture in many countries are extremely limited. Although ICES provides listings of chemicals developed or offered for use in aquaculture, reporting of chemical use, even in ICES states, remains very poor (Ackefors and Enell 1994). Again, those data which are available relate principally to cage culture of marine finfish in temperate waters.

Between 430 and 500 grammes of antibiotic may be used in the production of each tonne of farmed salmon in marine cage culture (Ackefors and Enell 1994, Wu 1995). Repeated application over extended periods is frequently necessary in order to keep populations disease free (Lightner and Redman 1994), especially in areas hit by widespread disease. In such situations, antibiotics such as oxytetracycline may be supplied at 0.2-0.4% by weight in feed. It is estimated that 37 tonnes of antibiotics were used in marine fish cages in Norway in 1990 alone (Johnsen 1994). The figure for Denmark in 1995 was nearer 3.5 tonnes, but this nevertheless represented almost one tenth of the total quantity of antibiotics administered for therapeutic purposes to the whole of Denmark's 5.2 million inhabitants (Halling-Sorensen *et al.* 1998). The sulfa-drugs trimetoprim and sulfamethizol account for the majority of the antibiotics used in aquaculture in Denmark, although oxolinic acid is also prominent.

Very little information is currently available on impacts of most of the pharmaceuticals released to the environment, be they from therapeutic use, use in agriculture or aquaculture (Halling-Sorensen *et al.* 1998). Although some of the antibiotics used in aquaculture may break down rapidly in the culture ponds, a proportion of that added will inevitably accumulate in shrimp tissue and pond sediments or be released to the environment with wastewaters (Johnsen 1994, Smith 1996). Accumulation in sediment probably accounts for only 1% of the total oxytetracycline added to marine fish cages. As Smith (1996) points out, the precise fate of the remaining 99% remains unknown. The novel fluorinated antibiotic Florfenicol, currently marketed in Japan and on trial in Scotland and Norway, provides effective treatment against the fungal disease furunculosis, but there are currently few data available on its environmental fate (Nordmo *et al.* 1994).

Contamination of the surrounding ecosystem can lead to toxic effects in other organisms and the development of bacterial strains resistant to antibiotics. A very high proportion of wild fish caught in the vicinity of Norwegian fish farms has been found to contain significant residues of oxolinic acid and flumequine (Ervik *et al.* 1994). In the US, the red rock crab accumulates particularly high concentrations of oxytetracycline downcurrent from marine fish farms (Capone *et al.* 1996). This is also likely to be a significant problem in waters receiving effluents from coastal shrimp ponds. Antibiotics have also been recorded in the sediments beneath marine fish cages, where they may persist at elevated concentrations for 10 months or more (Johnsen 1994, Capone *et al.* 1996, Kerry *et al.* 1996, Smith 1996, Smith and Samuelsen 1996). Degradation of oxytetracycline is slow enough to allow significant contamination of other areas (Samuelsen 1989 in Pearson and Inglis 1993). The quinolone sarafloxacin is particularly resistant to degradation (Hektoen *et al.* 1995). Likewise, a proportion of the antibiotics applied to shrimp ponds will simply accumulate in bottom sediments. Some antibiotics, including the commonly used oxytetracycline, break down extremely slowly in very organic rich, anaerobic sediments, such as those in intensive shrimp ponds (Lai *et al.* 1995). In turn, they can inhibit the bacteria responsible for the natural breakdown of organic matter and, when the sediments are finally disposed of, are then released to the wider environment. Sensitive bioassays are currently under development to allow detection of antibacterial agents in seawater (Pearson and Inglis 1993) and other media. However, although these techniques may provide rapid indications of the presence of antimicrobial activity, they will yield no information on the active agents responsible.

Some antibiotics are toxic to other marine life and may, if applied without sufficient control, also be hazardous to the handlers and even toxic to the feedstock or to the shrimp. The commonly used quinolones and sulfonamides are toxic to humans and may cause severe allergic reactions (*e.g.* Clegg *et al.* 1997). Both oxolinic acid and flumequine cause stress responses in some fish species (Moutou *et al.* 1997) and flumequine appears to be quite toxic to the brine shrimps which are frequently used as feedstocks in aquaculture (Migliore *et al.* 1997). The quinolone enrofloxacin, although of apparently low toxicity itself, has been shown to enhance the toxicity to fish of other contaminants, especially PCBs (Williams *et al.* 1997). Toxicity can be quite species specific; for example, oxytetracycline inhibits the growth of the diatom *Chaetoceros* sp. and the dinoflagellate *Gymnodinium* sp., but stimulates higher growth rates in *Alexandrium minutum* (Arzul *et al.* 1996).

Resistance to antibiotics is very low in natural populations. In contrast, in sediments in the vicinity of aquaculture operations, antibiotic resistance is frequently very high (Herwig *et al.* 1997). Indeed, such resistance generally covers a wider area, and persists for much longer periods, than detectable levels of the antibiotics themselves (Husevag *et al.* 1991, Kerry *et al.* 1996). Spanggaard *et al.* (1993) demonstrated that increased antibiotic resistance generally does not result from shifts in species composition but from fundamental changes to the resistance of existing communities and may, therefore, take some time to revert. The potential for long-term changes to ecosystems, even from the use of antibiotics which break down rapidly, is therefore significant. Moreover, the presence of antibiotic resistant strains of bacteria in the open environment, some of which may be pathogenic to marine species or even humans, raises very serious concerns.

In addition, the accumulation of antibiotic residues in aquaculture products themselves is increasingly recognised as a public health concern (Reilly and Kaferstein 1997). The accumulation of sulfonamide drugs in cultured channel catfish to levels well in excess of the US FDA limit of 0.1 parts per million (ppm) has been recognised for some time (Xu *et al.* 1996). Similarly, trimethoprim and sulfamethoxyzole have been recorded in the tissues of intensively cultured shrimp at levels over 35 ppm (Chair *et al.* 1996). These contaminants are then passed on to the consumer. Furthermore, the development of antibiotic resistance over time, following repeated applications of antibiotics, could result in the transmission to humans of pathogens which do not respond to antibiotic treatment (Reilly and Kaferstein 1997). The development of resistant *Salmonella* strains may be of particular concern (Daoust 1994)

<u>b) Pesticides</u>

As for antibiotics, although a wide range of pesticides are reportedly used to control disease and pest species in commercial aquaculture, data on precise use patterns are very scarce. Many of the chemicals employed are toxic to marine life and present substantial hazards to humans during application. Chlorinated organophosphorus pesticides have been used for many years to control sea lice infestations in farmed marine fish and reportedly are also used to control disease carrying organisms in shrimp ponds. Again these compounds may be used in large quantities; the annual use of trichlorfon in Norway increased from 4.9 tonnes to 28.3 tonnes between 1981 and 1985. Although this fell to 3.2 tonnes in 1988, this was due in part to its replacement by the similar pesticide dichlorvos ("Aquagard", Grave *et al.* 1991a). Even in Norway, compliance with recommended procedures for application of the pesticides was poor (Grave *et al.* 1991b). Controls in shrimp aquaculture are likely to be even less stringent.

McHenery *et al.* (1996) suggest that the use of dichlorvos in marine fish cages may present limited hazards to the surrounding environment provided the chemical is well dispersed by currents, although impacts on sensitive larvae may be expected close to the site of application. However, these authors did not appear to consider the potential for this pesticide to accumulate in sediments. That dichlorvos is toxic to a range of marine crustaceans and bivalves has been recognised for some time (Egidius and Moster 1987, in Thompson *et al.* 1995). Impacts on rocky shore communities adjacent to fish cages are reported to be sub-lethal but significant (Thompson *et al.* 1995). Dosing rates of dichlorvos are often greatly elevated in winter in order to account for the temperature dependent action and degradation of the pesticide, which could result in higher levels of exposure of natural communities in the vicinity of cage farms.

Moreover, the use of dichlorvos in an enclosed embayment or pond, followed by release of waters to mangroves or other low energy coastal environments, may give little opportunity for dispersal of the pesticide. The possibility for accumulation of pesticide residues in fish and shrimp tissue would also be enhanced. Several of the other pesticides under evaluation for control of sea lice in caged salmonids (Roth *et al.* 1993) may present similar problems to non-target organisms. For example, the pesticide ivermectin, used increasingly for lice control in Europe, has been shown to persist in sediments and to accumulate in infaunal polychaetes (Black *et al.* 1997). Long-term fate and impacts of exposure to this agent are poorly described.

Among other pesticides in use for shrimp aquaculture, trifluralin and malathion give particular cause for concern. In addition to its reported effects on the thyroid, testes and pituitary in mammals (Danish EPA 1995), exposure to trifluralin for as little as 1 day may be sufficient to induce bone deformities in salmon and trout (Wells and Cowan 1982). Although pesticide use is undoubtedly widespread practice in intensive aquaculture, there appear to be very few studies on the impacts of such use on the environment or on the accumulation of residues in the final product. Moreover, guidelines for the monitoring of the effects of aquaculture wastes prepared by GESAMP (1996) note the importance of sedimentation and of loading of coastal systems with degradable organic matter and nutrients but take no account of the potential for contamination with persistent organic compounds, heavy metals and other chemotherapeutants. Methods to detect residues, and therefore to protect the consumer, are improving but are far from complete.

c) Other Chemicals

The range of chemicals used as therapeutic agents or for other purposes in commercial aquaculture is extremely wide. For example, malachite green and formalin have been used in Europe to treat fungal infections in salmonids (Iwama 1991). Similarly, formalin, malachite green and copper sulphate have been used to disinfect shrimp ponds between cycles and to treat specific diseases in stock (Wu 1995). Indeed, Kongkeo (1997) reports continued use of formalin and hypochlorite bleach in order to treat incoming and outflowing water and for the treatment of contaminated sediments in shrimp aquaculture. The use of chloramine for similar purposes is also reported (Kongkeo 1997). Tributyl tin (TBT) was used extensively in Europe as an antifouling agent on boats as well as on fish cages, until its demonstrated ecotoxicity, notably to molluscs, led to bans or restrictions on use in many countries. For a more in depth discussion, see section 7.1d. Since national restrictions were introduced, including a ban on the use of the agent in aquaculture, levels of TBT in the environment have declined and impacted populations have shown signs of recovery in both the UK and Ireland (Matheissen *et al.* 1995, Minchin *et al.* 1997). Nevertheless, data on uses of TBT in developing economies is scarce. Other chemicals may be introduced to aquaculture cages or ponds, and subsequently to the environment, as additives or contaminants in feedstock. For example, butylated hydroxytoluene (BHT) and diphenylamine, used as antioxidants in artificial feeds, are known to accumulate to some degree in the sediments beneath marine cages (Samuelson *et al.* 1988, in Iwama), although the impacts on benthic organisms are unknown. In addition, certain vitamin additives have been demonstrated to stimulate phytoplankton growth (see section 5.3).

It is likely that various other chemicals are employed in aquaculture, either on a regular or *ad hoc* basis. However, information on such use, particularly in developing countries, remains very poor or non-existent.

5.6 Impacts of Cultured Organisms on Wild Populations

In addition to the introduction of contaminants, in the form of organic pollution, nutrients and production chemicals, aquaculture has also led to widespread introductions of cultured organisms, and frequently non-indigenous species, to the wild. Such individuals may be capable of interbreeding with wild populations, introducing foreign genetic material to local gene pools, or may simply compete with wild stocks for limited resources. The potential for introduction of novel diseases is also of serious concern. Again, the most extensive information on the impacts of introductions or escapes of cultured organisms relates to salmonid culture in temperate waters (Beveridge *et al.* 1996). It is well recognised that the culture of salmonids inevitably leads to escapes to the wild at all life stages (Hansen *et al.* 1997). If such escapes involve a large number of individuals, or if escapees breed successfully in the wild, their numbers relative to wild stocks can become very significant. Populations of salmon of aquaculture origin have risen steadily in Norwegian waters with the growth of marine cage culture (Heggberget *et al.* 1993). Indeed, Thompson *et al.* (1995) estimated that up to 20% of salmon caught by Norwegian fisheries may be escapees; this figure may be as high as 30-40% in some years (Lund *et al.* 1991). Figures for Scotland (Webb and Youngson 1992) and the Faroes (Hansen *et al.* 1992) are similar. While noting that farmed salmon are much less frequent in Greenland waters, Hansen *et al.* (1997) warn that, in many areas, the rise in farm escapees may be masking an underlying decline in wild stocks.

The precise fate of the majority of escapees remains unclear (Hansen *et al.* 1997), although it has been recognised for some time that some interbreeding will undoubtedly occur (Youngson *et al.* 1993). Even stock reared in hatcheries for aquaculture may be capable of interbreeding with wild stocks, although the survival of hatchery fish is generally lower (Unwin 1997, Jonsson 1997). Brannon (1993) recognised the dangers associated with the introduction of genes from outside the natural gene pool through hatchery fish introduction programmes. This is also a significant problem with fish reared from wild stock as there is considerable transboundary trade in live stock. For example, McGinnity *et al.* (1997) note that farmed salmon in Ireland and Scotland are predominantly derived from Norwegian stock. Escapes and interbreeding can therefore dramatically alter the local gene pool through the introduction of genetic traits selected for under different environmental conditions. Ultimately this may result in the progressive erosion of local adaptations and a reduction in resistance to disease and/or severe conditions in impacted "wild" stocks.

Survival and reproductive success of farmed salmon are generally lower than for wild populations (Jonsson 1997) and fish tend to be behaviourally less mature. Aggressive responses appear more frequent, leading to increased incidence of wounding of individuals in both the escapees and wild fish. Interbreeding of farmed and wild fish can, therefore, lower the fitness of wild stocks, causing long-term and potentially irreversible damage at population level (McGinnity *et al.* 1997). Even without interbreeding, behavioural differences and resource competition may lead to progressive exclusion of wild fish. Damage to wild stocks may be further enhanced by the collection of juveniles as broodstock for aquaculture operations.

The introduction of diseases and parasites to natural populations from escapees presents further problems. Natural immunity to diseases which are not native to a region is generally very low (McVicar 1997), putting wild populations at great risk from novel pathogens and parasites. In some instances, diseases "imported" with broodstock sourced from other regions have led to widespread losses of both farmed and wild stocks. While the potential clearly exists for pathogens to move both from wild to farmed and farmed to wild fish, the importance of farmed salmonids as a source of infection has long been recognised. McVicar (1997) provides an extensive review. The fungal disease furunculosis, introduced from *Onchorhynchus mykiss* stocks imported from Denmark in 1964, was first detected in Atlantic salmon in the wild in 1966, downstream from a cultured population of *O. mykiss*. The outbreak reached its peak in the late 1960s and had largely died out by 1979. However, the disease was again introduced to Norway in 1985, on this occasion *via* infected broodstock from Scotland. From 16 infected farms in central Norway in 1985, the disease had spread to 32 by 1988 and 171 farms by 1989, including some farms on the West coast. Between 1989 and 1992 the number of infected farms increased to over 500 and the disease was found to be impacting wild populations (Heggberget *et al.* 1993). McVicar (1997) describes similar outbreaks in other areas, stressing that the risk of infection may be significant within a radius of up to 10km from a single cage site.

Knowledge of the dynamics of parasite transfer to wild populations is quite limited (Bristow *et al.* 1993), although undoubtedly it occurs. In recent years, for example, as many as 30 natural populations of Atlantic salmon may have been lost through parasitic introductions in Norwegian waters alone (Heggberget *et al.* 1993). Furthermore, information on interactions in other aquaculture sectors is extremely scarce. Again, introductions are likely but the impacts, both potential and actual, have yet to be studied in any depth.

5.7 Genetically Modified Organisms

The risks and potential complexity of interactions may be further enhanced by the increasing development of genetically modified organisms, designed to have higher resistance to certain diseases, return higher yields or produce specific food qualities. For example, the development of transgenic marine organisms has moved beyond research and biotechnology applications (see review by Chen *et al.* 1996) to direct application for the production of disease-resistant shrimp broodstock (Miahle *et al.* 1995). Such developments will allow what Sin (1997) describes as a "fast-track" for disease control in intensive systems. While hailed as a solution which will reduce the use and release chemical disease-control agents, the consequences of release, including the potential for incorporation of novel genetic material in natural populations, appear to have been largely overlooked. Although such risks are likely to be substantial (Hallerman and Kapuscinski 1995), use of genetically modified organisms remains poorly documented and regulated. Following a global survey of legislation, Bartley and Hallerman (1995) reported that policies on use of genetically modified organisms were lacking in many developed countries and were almost absent from developing countries. This is despite the fact that such countries present the largest potential market for genetically modified broodstock. Where legislation does exist, it is frequently voluntary and there has been little effort to developed a harmonised regional or global approach. For example, legislation in the USA relies largely on voluntary compliance whereas in neighbouring Canada, legislation is much stricter (Hallerman and Kapuscinski 1995). Legislation in Norway does not currently preclude the use of genetically modified organisms for aquaculture, although the Norwegian government appears to be adopting a precautionary stance.

5.8 Overview

Aquaculture, therefore, exerts numerous and substantial environmental pressures. Despite development of regulations and technological improvements, problems in all regions, and particularly in developing economies, remain significant. Continued economic pressure for growth of the aquaculture industry will only magnify these problems further and increase their geographical sphere of influence. Financial pressures can force higher stocking densities and increased use of chemical disease control agents, leading in turn to the need for an unsustainable system of "shifting aquaculture" (Flaherty and Karnjanakesorn 1995). As Bodvin (1997) notes, waste reduction measures for new and existing operations are possible, but the technology remains expensive, resource intensive and, in many cases, is still under development.

Tacon *et al.* (1995) recognise the need for much greater control of the use of chemicals in aquaculture, particularly in Asia where shrimp aquaculture is most prominent. This was also stressed by Goebbels (1990) in connection with control of residue levels in products. Techniques to identify specific pathogens in order better to target treatments are generally in their infancy, especially for shrimp (Miahle *et al.* 1995). Chemicals tend, therefore, to be used more or less routinely and indiscriminately as "preventative" treatments, with little regard for the impact on the surrounding environment or, indeed, for the accumulation or chemical residues in the final product and the transmission of antibiotic-resistant pathogens to the human food chain. Tacon *et al.* (1995) note that antibiotics may frequently be used as substitutes for more sustainable management practices in intensive aquaculture in order to maximise harvest over the short-term. Efficacy and economy may be more important determinants of the chemicals used (Schnick 1991), rather than concerns over residues or impacts.

Gempesaw *et al.* (1995) note that current consumer unwillingness in the West to buy certain aquaculture products stems more from unfamiliarity with the product than any concern over chemical residues or the environmental impacts of the production systems. Following a decline in consumption in the late 1980s, the aquaculture industry has recognised the need to market their products more effectively (Hanson *et al.* 1994). Given that controls on the use of chemicals in aquaculture remain totally inadequate, careful control of the use and release of transgenic organisms seems highly unlikely. What is likely is that disease control in intensive aquaculture will continue to present a high demand for antibiotics and other chemotherapeutants and, therefore, present significant risks to the environment, aquaculture workers and the consumer for some time to come.

6. Whales and Whaling

The parallels between the over-exploitation of whales and over exploitation in conventional fisheries are striking. In particular, the failure to devise adequate conservation measures is a problem that is common to both activities Systematic whaling was probably first started by the Basque people around 700AD (Lean & Hinrichsen 1992) exploiting the Right whales passing through the Bay of Biscay. Overexploitation of these stocks led to expansion beyond the Faroe Islands to Spitzbergen and into the Atlantic. In the 17th Century the Dutch and British replaced the Basque fleet. The Dutch fleet of 300 ships and 18,000 men targeted the bowhead whale, clearing the Arctic Ocean east of Greenland and moving towards American waters. The whaling industry centred around Cape Cod targeted right whales and subsequently sperm whales. The whaling ships moved increasing further south in the Atlantic, and rounding Cape Horn began to exploit the stocks of sperm whales present in the Pacific Ocean.

The development of mechanically powered vessels in the early 20th century, together with harpoon guns and inflation lances to inflate carcasses with compressed air and keep them afloat, allowed the targeting of fast whales such as the blue and sei whales. In addition it allowed the scale of operations to be greatly expanded, ensuring that whale products were kept at a price low enough to sustain a market. After stocks became depleted in the northern seas, exploitation began in the Antarctic on the whale feeding grounds where the animals congregated. Norway, the pioneer country was joined by the UK and in the 1930s by

Japan and Germany. The initial targets of the fast catcher boats were the blue whales. By 1925, 2,500 were being taken annually. In 1926, the first factory ship, serviced by a fleet of fast catcher vessels was introduced. This innovation removed even the imperfect controls which could be exerted on a land-based industry. By 1930, the catch of blue whales had reached 30,000. Overall, between 1910 and 1966 over 330,000 blue whales were killed. After these became scarce, fin whales were targeted and caught at a rate of almost 30,000 per year in the 1950s. Technological advances and a continuing market demand for whale oil ensured that whaling continued to be a growth industry (Knauss 1997). Much of the growth was in the Antarctic. About 2,000 whales were taken in the region in the 1907-08 season, rising to 14,000 in 1927-18, until in 1937-38 before the Second World War 46,000 animals were killed. This accounted for around 85% of the total number of whales caught in that year. The sei whales were targeted in the 1960s and the minke whales in the 1970s. The history of whaling is represented in 6 showing the successive targeting, and over exploitation of one species after another (Evans 1987), particularly in Antarctic regions.

In total, over 1.5 million whales were killed in the fifty years after 1925. Conservation measures were generally taken in response to commercial concerns such as the whale oil glut of 1931 which prompted whaling companies to voluntarily reduce catches to maintain prices. While biological extinction of species has apparently not occurred, the Atlantic grey whale population has been extirpated and the blue, fin and humpback whale have been reduced to extremely low levels relative to those which existed prior to exploitation. (Stroud 1996). The 1931 Convention for the Regulation of Whaling was formulated as a multilateral arrangement which regulated whaling in all waters. It was, in the view of some, formulated as a response to the perception that the industry itself needed protecting from extinction and prevented the killing of right whales (reduced to 3% of the original population), the killing of calves and of females accompanied by calves. The Convention came into force in 1935 and was amended in 1937 to protect grey whales (reduced in numbers to an estimated 250) and to set close seasons and limits on hunting areas. The following year, humpback whales were given added protection.

The 1931 Convention failed to protect the great whales. Problems, now familiar from failed fisheries management schemes, included lack of scientific knowledge and failure of some whaling nations to sign the Convention, while those that did failed to emplace stringent compliance enforcement mechanisms (Stroud 1996). These failures were not rectified by the 1946 International Convention for the Regulation of Whaling (ICRW) under whose auspices the International Whaling Commission (IWC) was established (Rose 1996). There followed a further quarter century of intensive whaling activity. Whatever

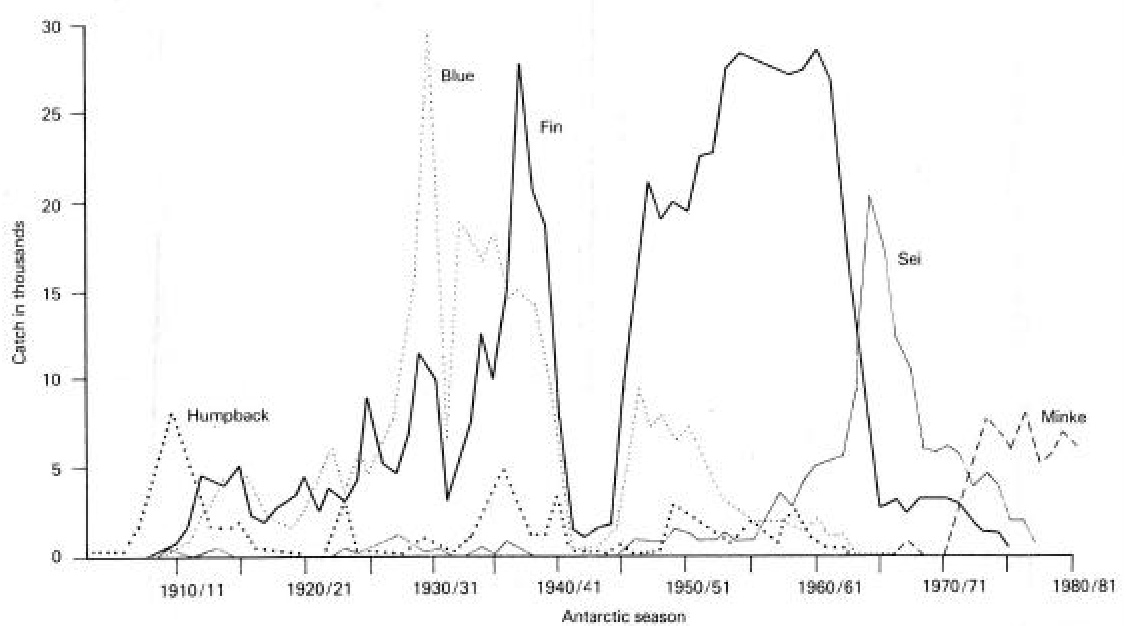


FIGURE 6: Catches of Great Whales in thousands with season. The curves clearly show the sequential targeting of the most

commercially valuable species as populations of individual species were successively over exploited (Evans 1987).

improvements have resulted in knowledge of whale population dynamics, catch statistics and in promoting research it has undoubtedly failed in the primary goal of maintaining sustainable populations (Knauss 1997). At the time of the 1972 Stockholm Conference which called for a 10 year moratorium on whaling activities, the use of management techniques based on a "blue whale equivalent value" had proven disastrous. Allowable catches in Antarctica had been reduced to less than 25% of levels agreed to when the IWC was first established. Blue and humpback whales had been placed on the protected list. The fin whale was placed on the Antarctic protected list three years later. The IWC's Objection system whereby any member was able to exempt itself from any decision with which it did not agree, ensured that any protective measures were emplaced only after populations became heavily depleted Although highly contentious, provisional figures (Lean & Hinrichsen 1992) suggested that sperm whales were reduced by whaling from around 1,000,000 to around 10,000. Humpbacks have been reduced from 200,000 to 4,000. Fin whales have been reduced to 2,000 from 100,000 and blue whales from 250,000 to around 500. In terms of the cetacean biomass, because the large species were taken first, this is estimated to have declined from an estimated 45 million tonnes to 9 million tonnes over the period 1904-1973 (Bowen 1997).

The call for a ten year moratorium by the 1972 Stockholm Conference was initially ignored by the IWC and a New Management Plan based on the concept of Maximum Sustainable Yield emplaced in 1975. Negotiations between the interested parties in the IWC continued to be characterised by major disputes over science and quotas (Skare 1994). Ironically, two years later the concept of MSY for fisheries management was first discredited (Larkin 1977). The all too familiar situation (in marine capture fisheries management) of uncertainties in population estimates generated by the Scientific Committee of the IWC and strong pressure exerted on economic grounds by the industry, led Commissioners to ignore the scientific advice and to approve larger catches than were justified. After quotas fell from 32,000 in 19774 to 12,000 in 1981-82, the IWC finally agreed to a total moratorium on commercial whaling in 1982 effective from 1986. There are two exceptions were made to the moratorium: subsistence whaling and scientific whaling. By 1990. a comprehensive assessment of impacts of the moratorium on whale stocks was due to be initiated, while over the period subsequent to 1986, a Revised Management Plan (RMP) was developed and adopted in 1994. The RMP was intended as a highly conservative model based approach towards the regulation of whale exploitation. At the same time the Southern Oceans were declared as a sanctuary for all great whales from 1994 onwards.

The initial purpose of the moratorium, to allow a period of recovery against which whale stock recoveries could be assessed has been undermined somewhat by continued whaling under the guise of scientific programmes allowed under the Convention. Whaling under Objection has accounted for some 14,000 whales between 1986 and 1996 (Stroud 1996) Undoubtedly, however, while the moratorium has not been completely effective, it has undoubtedly resulted in catches being kept at much lower levels than would otherwise have been the case (Stroud & Simmonds 1998). Pressure has continued from two directions. The whaling nations have continued to argue for the lifting of the moratorium. Conservationists have argued for a permanent end to commercial whaling operations. On the one hand the pro-whaling lobby cite scientific evidence that some populations of whales could support commercial exploitation. On the other hand the conservationists argue that notwithstanding the precautionary provisions of the RMP, numerous factors are not taken into account, including the potential impacts of climate change and stratospheric ozone depletion (see: IWC 1996), the potential impacts of chemical contaminants (Colborn & Smolen 1996; Johnston *et al.* 1996) and difficulties in establishing stock sizes (see: Stroud 1996; Knauss 1997). In addition, attention has been drawn to the historical resistance of the whaling industry to regulation (Simmonds 1997). Numerous examples exist of regulations being flouted (Stroud 1996). A 1993 DNA analysis of whale meat on sale in Japanese markets, for example showed that several species of whale, supposedly under full protection through IWC provisions, were on sale. Breaches of import and export regulations imposed by the IWC have also been recorded.

Compounding suspicions about the tractability of the whaling industry to regulation is the fact that, despite provisions for independent observation of operations under the RMP, the statistics such as catch data and abundance estimates used to operate the plan will be largely derived from the industry itself. Substantial under-reporting of catches by Soviet vessels of around 80,000 whales took place between 1948 and 1980, with over 48,000 humpback whales actually taken as oppose to the reported number of 8,000 (Stroud 1996). Errors introduced through under reporting and misreporting of catches into the RMP input data sets could compromise the objectives of the plan in the same way that similar errors in data input to fisheries models have compromised the operation of management systems in marine capture fisheries. In addition to these pragmatic considerations, whaling also touches on a number of ethical issues which are much harder to define. The simplest way to protect against deliberate abuses or accidental errors in the management is simply to remove the need for such management by emplacing a permanent ban on commercial whaling activities. This would also remove the need to police the markets in whale products. It would constitute a truly precautionary approach to the conservation of the great whale species and prevent mismanagement driven by commercial considerations.

7. Impacts of Shipping

a) Operational and Illegal Discharges

Currently, some 40,000 merchant vessels are involved in shipping activities around the world (ICS 1997) with a gross registered tonnage of approximately 520Mt. Considering all vessels of 100 gross registered tons (GRT) or more then the total rises to around 85,000 ships with an aggregate tonnage of 753 Mt. The miscellaneous category (Lloyds Register pers comm.) numbers around 40,000 vessels but accounts for only 25 M gross registered tonnes. Other statistics (Corbett & Fischbeck) suggest that including military vessels the world fleet numbers around 105,000 ships with a deadweight tonnage of 771Mt. Military vessels account for 19% of the worlds ships and 11% of this tonnage.

These ships are involved in the transport of materials representative of the products of all branches of industry and commerce. It has been estimated that more than 50% of the cargoes carried can be regarded as hazardous or harmful from a safety or environmental point of view. In addition, many materials used in the routine operation of ships have the potential to damage the environment (ICS 1997) and operational ships produce routine emissions to both air and water. Moreover, there are accidental losses of both ships and cargo. Between 1990 and 1995, total losses of ships totalled 1461 ships with a gross tonnage of 8.2 million tons (Lloyds Register 1996). Between 1986 and 1995 world seaborne trade increased by around 45% from 14 billion tonne miles to 20 billion tonne miles. Oil and petroleum products increased disproportionately from 5.8 billion tonne miles to 9.3 billion tonne miles, an increase of 60%.

The total loss of ships is a relatively low percentage of the world fleet on an annual basis, amounting to about 1.3 million tonnes per annum over the period 1973-1995, totalling 29.3 million tonnes. In conflicts, losses of vessels can be very much higher. In the two world conflicts a total of 28 million tonnes of merchant shipping was sunk on the Allied side alone. Many of these ships, whenever lost, contained potentially harmful cargoes (Angel & Rice 1996).

Marine transport accounts for around 12% of contaminants entering the worlds oceans (GESAMP 1990) and this figure appears to be accepted by the industry (ICS 1997). The precise contribution of individual contaminants by the shipping industry remains obscure. There is a great deal of uncertainty with such figures due to the quality of the data from which they are derived. Moreover, the estimates do not appear to take account of emissions to atmosphere. The best documented emissions are those of oil entering the oceans. Operational discharges of oil from shipping now comprise around four fifths of the total of 568,000t of oil estimated to enter the marine environment annually from shipping (IMO 1990; ICS 1997) given that on average 120,000t are lost through accidental spills. In turn this is estimated at around 24% of the oil from all sources into the sea. Surprisingly, the single largest estimate of oil discharges is comprised of around 186,000t of fuel oil sludge. A further 64,000t is discharged from machinery spaces. This compares to 158,000t discharged from tanker cargo residues.

Control of operational discharges from ships falls under the mandate of MARPOL 73/78, the International Convention for the Prevention of Pollution from Ships. This Convention has been credited with helping to substantially decrease the amounts of oil entering the sea from maritime transport activities (ICS 1997).

Oil entering the sea from these sources has fallen from 1.47 million tonnes to 568,000 tonnes between 1981 and 1989. Improvements in engine room technology and changes in tanker loading practices have helped to bring about the decrease. In addition, oil entering the sea as a result of accidental discharges has fallen by about two thirds over the period from the 1970s to the 1990s. There is tremendous variability in the amounts of oil lost accidentally from year to year, however, since one large incident can significantly increase annual losses. Attempts to calculate operational losses from details of loads carried are confounded by the fact that the existing database carries details of only 30% of the crude oil carried by sea (IoP 1997). It is not possible to distinguish between oil lost to the sea and that lost to the air by volatilisation given that systematic errors also occur in the data.

Illegal discharges of oil are particularly problematical in areas where there is intense shipping traffic. In the North Sea it has been estimated that anywhere between 15,000-60,000t are discharged annually, mainly in the shipping corridor between the Strait of Dover and the German Bight (OSPAR 1993). Individual slicks were estimated to contain up to 2000t of oil. Other areas where there is large scale oil transportation and production such as the Red Sea and the Arabian Gulf show high levels of beach contamination with oil. Extensive oil contamination is also found in the Caribbean and in Indonesia and the Philippines (GESAMP 1990) although, in general, this problem is thought to be improving. The use of "load on top" procedures and crude oil washing of tanks aboard tankers, together with the segregation of ballast water have all made important contributions (ICS 1997).

Chemical tankers represent another source of pollutants into the sea. Although in the North Sea area tank washings from vessels carrying the most dangerous chemicals must discharge to shore based reception facilities, less hazardous chemicals from tank washings can be discharged at sea. The carriage of chemicals has increased markedly in recent years (GESAMP 1990) although it still regarded as low in comparison to oil transport. Nonetheless, the scale of the global trade can be gauged from European figures which suggested that some 41 potentially hazardous chemicals were imported into North Sea ports in estimated quantities ranging from 12-812kt per annum (Hurford *et al.* 1989). These included: chlorobenzene (15kt); dichloromethane (38kt); dichloroethyl-ether (18kt); ethylene dichloride (94kt); perchloroethylene (17kt) and trichloroethylene (17kt). These comprise just 184kt of a total of 5887kt of the 41 listed chemicals. Estimates from 1985 (GESAMP 1990) suggest that total movements of chemicals by ships was some 25Mt with some 80% of this accounted for by 22 products. Half of the volume is comprised of petrochemicals. Western Europe and the United States are both leading exporters and importers. Latin America with Asia/Oceania are leading importers of liquid chemical products. It has been noted that such trading is notoriously difficult to analyse since the chemical industry is secretive about its activities. A wide range of chemicals have been lost at sea around the UK where recording of such incidents is undertaken (Johnston *et al.* 1997). These include acrylonitrile, o-cresol, xylene, uranium hexafluoride and the pesticides dinoseb and nemagon.

b) Packaged Goods

It is estimated that up to 15% of all goods carried in packaged form by conventional dry cargo ships are dangerous to some degree (ICS 1997). The carriage of pesticides has attracted attention in recent years owing to some highly publicised losses of containerised goods at sea (Johnston *et al.* 1997). This is a growing problem. Between 1977 and 1987 pesticide sales increased susbstantially and the agrochemical market doubled in size. Exports from the US represent around a quarter of the world market and exports are estimated at 400-600 million pounds sterling *per annum*.. The trade is also highly developed in Europe. Export tonnages of pesticides from Germany are over twice the domestic consumption. The carriage of pesticides are regulated by the United Nations International Maritime Organisation Dangerous Goods Code (IMDG Code) with respect to packaging and labelling, but this does not necessarily allow successful attempts at salvage should containers be lost at sea. The impacts of losses could be severe. It has been estimated that a spillage of 10t of the organophosphate pesticide pirimphos-ethyl into the English Channel could extend over an area of 10,000 km^2 given that the “safe” concentration of this pesticide is assumed at 20ng l^{-1}. This would eradicate or deplete stocks of crustaceans over a wide area, with major effects on the fishery. This could take up to five years to recover while the social and financial costs could run into tens of millions of pounds.

Containers are routinely lost at sea, but the lack of a comprehensive database recording incidents involving pesticides and other hazardous chemicals means that only for those incidents attracting publicity are details available. Even well managed operations can suffer losses in adverse weather. One operator reported 18 lost containers over an eighteen month period from 1995 out of a total of 2.5 million carried. These relatively low losses contrast with the 88 containers lost in a single incident during heavy weather in the English Channel by the MV *Sherbro* in 1993. Ten of these containers held pesticides, and the resulting beach clean-up was estimated to have cost US$2.7 million. The MV *Perintis* was lost in 1989 together with deck cargo containers of pesticides. Although some of these were salvaged, the location of around 6t of lindane is unknown. Failure to salvage this cargo led to the need for costly routine surveillance monitoring to be put in place in the area. The losses of hazardous cargoes are made more likely by the practice of stowing them on deck in order to maximise safety for crew on the vessels which transport them. There is clearly a need for much better documentation of such incidents together with a need for measures to reduce the numbers of such incidents. Measures need to be taken to enhance the prospects for location and recovery of hazardous cargoes lost at sea.

c) Energy Use and Atmospheric Emissions

Shipping is responsible for the carriage of some 80% of the volume of goods traded globally (ICS 1997). It is regarded as a safe and efficient form of transport. It produces lower emissions of atmospheric pollutants per tonne of goods moved than any other major mode of transport and is more energy efficient on this basis than road or rail (ICS 1997). Nonetheless, attention has been drawn recently to the fact that per tonne of fuel consumed, ships engines are among the worlds highest polluting combustion sources. This problem arises in part from the nature of fuels used on board vessels. These are to large extent residual oils which replaced crude oil after the oil crisis in the 1970s. The residual oils which remain from the intensive refining of crude oils contain enhanced concentrations of contaminants such as sulphur and heavy metals (Corbett. & Fischbeck 1997). Analysis of exhaust emissions from ships shows that they are significant contributors to atmospheric pollution. On a global basis, ships account for 4% of emissions of sulphur oxides (SO_x), 7% of nitrogen oxides (NO_x) and 1.4% of carbon dioxide emissions (CO_2) (Mathiesen 1994). When considered purely in terms of petroleum usage, then ship emissions account for around 14% of nitrogen emissions from fossil fuels and 16% of all sulphur emissions. Carbon emissions are around 2% of the global total. These emissions occur mostly in the Northern Hemisphere (Corbett & Fischbeck 1997) where shipping densities are heaviest. By comparison, ship nitrogen and sulphur emissions are equivalent to 42% and 35% of those from North America. It follows that these emissions and those of the associated heavy metals are of global significance.

d) Antifouling Paint and TBT

The relative efficiency in energy terms of marine transport is maintained in part by the use of hull coatings designed to prevent the growth of fouling organisms which increase the water resistance of the hull. The most commonly used antifouling preparations use the organo-tin compound tributyl-tin (TBT). This has been described perhaps the most toxic chemical ever deliberately introduced into natural waters (Stewart & de Mora 1990). TBT is known to be acutely toxic to a wide variety of marine macro and micro-organisms and thus far, a no-effect threshold has not been identified. It has been suggested that phytoplankton communities can be affected by TBT (Petersen & Gustavson 1998). Shell malformations characteristic of TBT in commercial oysters have been found in waters where TBT could not be detected using current instrumental analytical methods. The economic loss in the Arcachon Bay (France) oyster fishery due to TBT contamination has been estimated at around 147 million US dollars. Importantly, extremely low concentrations of TBT can exert sublethal effects which do not directly cause mortality but can cause population declines by affecting reproduction. The most studied of such effects is that of imposex in the marine mollusc *Buccinum undatum* where females develop male characteristics which suppress and ultimately prevent reproduction. This can take place at water concentrations as low as 1ng l^{-1} of TBT. The effect appears to be due to interference with the mechanism of testosterone production and metabolism in gastropod molluscs (Matthiessen & Gibbs 1998) and available evidence suggests that more than 100 species of prosobranch molluscs are affected by imposex worldwide.

In areas of heavy shipping activity such as the English Channel shipping lanes where some 420,000 shipping movements are recorded annually (OSPAR 1993) dog whelk (*Nucella lapillus*) populations have been eliminated from busy coastal areas. A high incidence of imposex was found in whelks (*Buccinum undatum*) from offshore deepwater shipping lanes, while a much lower incidence was found in the area of the Dogger Bank, where shipping activity is lower (Ten Hallers-Tjabbes *et al.* 1994). It has been estimated that the gross annual input of TBT to the North Sea could theoretically be as high as 240t annually, although estimates based upon evaluation of individual inputs suggest a gross input of around 68t annually. (Davies *et al.* 1998).

Current inputs are largely from large shipping since a ban has been placed on the use of TBT paints on boats of less the 25m in length under the IMO Resolution MEPC.46 (30) to address the severe problems associated with leisure craft marinas. Historical records have shown that dog-whelks were previously distributed much more widely in the North Sea than at present and that continued exposure to TBT is likely to exert catastrophic effects on populations of these animals (OSPAR 1993). The findings in the North Sea are mirrored in local situations where monitoring has been carried out. Dog whelk populations were found to be declining in the vicinity of the Sullom Voe oil terminal in Shetland . and absent from the most heavily impacted areas. Even after the phase-out of free association TBT coatings in the late 1980s and their replacement by self polishing co-polymer paints, decline continued (Harding *et al.*1997) probably as a result of delayed effectiveness of the phase out due to ship maintenance cycles. More recently the population at Sullom Voe has started to recover. Reduction of water concentrations of TBT through use of co-polymer paints will not prevent continued build up of TBT in sediments due to its extremely low degradation rate (Stewart & de Mora 1990). In Ireland, restrictions on the use of TBT on aquaculture nets and small boats resulted in recovery of *Nucella* in salmon farming areas and areas with small boat activity. Continued decline was observed in the vicinity of shipping ports (Minchin *et al.* 1995). Similar findings have been made for coastal waters of France, the UK and US (Evans *et al.* 1994 Evans *et al.* 1995; Evans *et al.* 1996).

The recovery of coastal populations of gastropods on a local basis as a result of limited restrictions does not obscure the fact that there is a wide global dimension to the impacts of TBT. TBT and its degradation products are persistent and can remain in sediments for a considerable period at concentrations likely to inhibit further degradation. (Stewart & de Mora 1990). A wide variety of gastropod molluscs can be affected by both TBT and triphenyl-tin (TPT) (Horiguchi *et al.* 1997). Detectable residues have been found in molluscs and fish in a number of regions including the Mediterranean (Morcillo *et al.* 1997; Kannan *et al.* 1996), The Pacific (Stewart *et al.* 1992; de Mora *et al.* 1995) including the islands (Kannan *et al.* 1995) and in fish sampled in a variety of Asian and Oceanian countries (Kannan *et al.* 1995). There is growing evidence that these contaminants can be bioaccumulated in the tissues of marine mammals (Kannan *et al.* 1997; Kannan *et al.* 1996; Tanabe *et al.*1997) with the highest concentrations found in animals living in the coastal waters of developed countries. Analyses on the liver and kidney of seabirds collected from Japan, Korea, the North Pacific Ocean and the Indian Ocean showed the presence of various butyltins in most samples, with the highest concentrations found in coastal birds. This indicates that organotins are widespread contaminants of higher trophic levels even in remote areas. The ecotoxicological significance of this is unknown.

Despite restrictions in place on organotin usage in antifouling preparations, growth in use of these biocides in this and other applications is reflected in an increase in production of these chemicals from 30,000t to 50,000t between 1982 and 1992 (Kannan *et al.* 1995). Far from conforming to initial perceptions that TBT was a transient environmental problem which could be resolved by a complete or partial ban, TBT continues to pose a major ecotoxicological threat. TBT and its degradation products are now ubiquitous global contaminants and require immediate comprehensive regulatory action. In order to eliminate problems, its use on large vessels as well as small boats needs to be banned.

In addition, some attention needs to be directed at the chemicals likely to be used as replacements. Herbicides are added to copper based antifouling paints to inhibit algal growth. One of these, a triazine herbicide known as IRGAROL 1051 is highly stable in seawater. It has been found at concentrations up to $1.7\mu l^{-1}$ in yacht marinas in the south of France (Readman *et al.* 1993). This herbicide was also found at elevated levels in coastal waters of the UK in areas of high boating activity (Gough *et al.* 1994). Water concentrations of up to 120ng l^{-1} of IRGAROL 1051 were found in the enclosed waters of Plymouth

Sound in the UK (Scarlett *et al.* 1997). This concentration was found to inhibit the growth of algal zoospores in laboratory tests. This potential problem is likely to become more widespread as TBT is replaced with alternative antifoulants. Over 80 products containing IRGAROL 1051 are registered for use as antifouling paints in the UK and survey results suggest that around 20% of small boats in Plymouth were using such paints. 300 boats, including fishing boats were present in the harbour in the category of small boats. Usage of this type of antifouling by larger vessels is unknown.

e) Marine Litter and Debris

Routine ship operations generate waste consisting of ordinary household waste, cargo associated waste, damaged fishing nets and ropes, and used medical products. Most garbage is allowed to be dumped at sea with the exception of plastic materials although some regional prohibitions on other wastes exist in the Baltic and Mediterranean Seas. The distance offshore at which various categories of waste can be dumped is specified under MARPOL 73/78. The amount of food waste generated is in the range of 1.4-2.4 kg/person per day and household refuse is generated at the rate of 0.5-1.5 kg/person/day. Cargo wastes are generated at the rate of approximately 1t per 123t of cargo. This will obviously vary according to the type of ship. Tankers would not expect to generate cargo wastes regulated under Annex V of MARPOL 73/78, while cruise ships are likely to generate large quantities of household refuse. A passenger vessel will also generate around 230l of sewage and grey water per person per day although this falls to around 185l per person if a vacuum system is used Olson (1994).

The regulations imposed by MARPOL which prohibited plastic waste dumping at sea appear to have been only partially successful. In a survey of discarded trawl net deposited on Alaskan beaches it was found that the number of pieces found declined by 60% over the period 1988 to 1992 and that the mesh size of the net pieces decreased (Johnson 1994). In general, plastics continue to cause problems (Goldberg 1995) together with glass, rubber, metal and glass items but not all of these are from shipboard activities. Contamination of Indonesian island beaches up to 45 km away suffer severe beach contamination with plastics from the city of Jakarta and this has increased substantially since 1985. This reflects both the persistence of plastics and their increased use over time (Prulley *et al.*1997). The Antarctic, by contrast, has relatively few direct sources of man made debris. Plastic debris discarded by fishing boats targeting the Patagonian toothfish stocks off South Georgia increased substantially between on Bird Island between 1990 & 1995 corresponding to an increase in fishing effort in the area (Walker *et al.*1997).

Similarly, in an Australian survey of fishing debris it was estimated that some 2500km of monofilament line were discarded annually from longliners operating off the Tasmanian coast. Observers reports suggest that around a third of the boats were not operating in compliance with the provisions of MARPOL. Accordingly, in Tasmania it has been found that 20% of the debris and 40% of plastic debris found originated from the fishing industry (Jones 1995). Overall on Tasmanian beaches, 300 debris items were collected per kilometre of beach. This compares to 262 items km^{-1} in Hawaii, 814 items km^{-1} in California, 1712 items km^{-1} in Texas and 8000 items km^{-1} in Mexico. In these surveys it was estimated that around 59% of the recovered items were potentially hazardous to wildlife. In a survey of Australian beaches, around 70% of the items were thought to be of marine origin (Frost & Cullen 1997) although on certain beaches, litter increased in response to heavier tourist usage. Rainfall can also be important by flushing materials discarded in the watershed into the sea. In one survey of UK beaches, it was found that 22 plastic bottles, 28 cans, 4 packing straps and 11 sanitary items, on average, were present per kilometer of coastline (Rees & Pond 1993) and thirty two source countries were identified for the materials found. The dominant debris items found in beach surveys in Panama were made of plastic (Garrity & Levings 1993). This was also true of debris in St Lucia where plastic items comprised 51.3% of items by number and 38.6% by weight. In Dominica, driftwood was the most common debris item although plastic still comprised a significant proportion of the debris items (Corbin & Singh 1993). Plastics present in the sea consist of weathered particles from larger consumer items, but also pellets from primary manufacture. The weathering of expanded polystyrene creates small particles, which may also act as a source of chemicals present as additives (see: Zitko 1993). Small plastic particles are also added to hand cleansers, cosmetic preparations and are used in blast cleaning as a substitute for sand (Gregory *et al.*1996). Plastic pellets produced by manufacturers for onward processing have achieved a global distribution as evidenced by beach surveys (USEPA 1992a).

Plastics also comprise a substantial element of debris found on in the open ocean in all geographical areas (see: Dufault & Whitehead 1994) with up to 3.5 kg km^{-2} reported from the Pacific, up to 10kg km^{-2} off the coast of South Africa and up to 17.7 kg km^{-2} from the Sargasso Sea. These small items of debris are supplemented by large debris items which average around 2000km^{-2} in the Mediterranean but between 0.23-1.83 in the waters of the Pacific Ocean. In a survey off Nova Scotia up to 112.8 items km^{-2} of large debris were recorded while small debris was found in 80% of the tows carried out. While some items will float other will sink to the bottom. A bottom survey of the Eastern Mediterranean, where the disposal of all garbage except food wastes is prohibited, found 277 items originating from at least 7 countries. The most common debris was paint chips (44%) and plastics (36%). 70% of the benthic trawls performed during this survey contained litter. This compared to 57% in the Gulf of Alaska and 41% in the Bering Sea. Dating of some materials was possible, establishing that they had been dumped recently. This points to a widespread non-compliance with the provisions of MARPOL and the regional legislation (Galil *et al.* 1995). Similar findings were made in a survey of the North Western Mediterranean where up to 924 pieces of debris were recovered by sampling individual sites. Plastics were the most frequently recovered items of debris with a mean percentage of 77.1% in the samples. There was a very significant influence of metropolitan areas on the amounts of debris recovered particularly in the vicinity of Marseille. Crude extrapolation of these results suggests that at least 3000t of plastics are present on the shelf of the NW Mediterranean alone (Galgani *et al.*1995).

These findings highlight the fact that most research concerning marine debris has focussed on shores and beaches. Relatively little work has been directed at elucidating the fate of debris in the sea itself. It has become evident that off-shore sinks exist for marine litter, where plastics and other debris concentrate as a result of hydrographic conditions. A limited study conducted in the Bristol Channel (UK) has demonstrated that this may be a considerable problem and cause economic losses to fisheries (Williams *et al* 1993) through damage to gear or obstructing fishing grounds. The impacts of marine debris upon natural ecosystems are varied. Floating debris can act as a transport substrate for exotic organisms. Debris which sinks can inhibit sediment/water gas exchange causing anoxic regions on the sea floor (Goldberg 1995). Entanglement in discarded fishing net can kill marine mammals. Increase in fishing effort off South Georgia was found to correspond with an increase in the proportions of fur seal entangled in discarded fishing line (Walker *et al.* 1997). An increase in fur seal population may disguise the fact that greater numbers of seals are becoming entangled in discarded fishing gear although as a percentage of the population entanglements appear to have decreased (Arnould & Croxall 1995). Ingestion of debris has been implicated in the death of turtles in the US and elsewhere (Bjorndal *et al.* 1994). Seabirds are also known to ingest debris items. In the case of puffins sampled in Britain over a 24 year period no clear evidence of harm was evident (Harris & Wanless 1994). In a survey comparing birds from the subarctic North Pacific sampled in 1988-1990 showed that plastic ingestion has increased substantially since examination of birds sampled in 1968. This corresponds to an overall increase in plastics manufacture over the period. Plastic production in the US increased from 2.9Mt in 1960 to 47.9Mt in 1985. Ingestion of plastics by seabirds can result in obstruction of the digestive tract and in ulcers as well as affecting feeding behaviour and hormone metabolism (Robards *et al* 1995).

Discarded fishing nets and gear can continue to net fish and birds. One 1500m long section of net recovered from the Pacific Ocean was found to contain 99 seabirds, 2 sharks and 75 salmon after an estimated month adrift. Whales, including endangered humpback and grey whales have been found entangled in netting (USEPA 1992b). In commercial fisheries, “ghost fishing” defined as the capacity of fishing gear to continue to fish after it has been lost and is outside the control of the fisher may significantly add to fishing mortality. Ghost fishing traps were estimated to catch an amount equivalent to 7% by weight of reported landings of the Dungeness crab in the British Columbian fishery. In Kuwait, lost fish traps were estimated to catch between 3 and 13.5% of the total Kuwait landings. The decline of the St. Lawrence queen crab has been partially attributed to mortality associated with lost traps (see: Erzini *et al.* 1997). A study in southern Portugal using simulated lost nets showed that catch rates were initially comparable to normally fished gill nets but decreased with time. The fishing lifetime of a lost net was estimated at between 15 and 20 weeks. and the decline in catch was due to changes in net shape, decreasing net height and increasing net visibility. The catches estimated of between 200 and 300 fish were considered to be much lower than actual catches due to predation and scavenging of entrapped fish by species such as octopus and conger eels. A similar study was carried out in Wales in the UK using 90m sections of gill and trammel nets set in inshore waters. In this case, the nets immediately caught large numbers of dogfish which caused the nets to collapse, reducing fish catches. The large biomass of decaying fish, however, attracted large numbers of invertebrate scavengers, and catches of crustaceans peaked at around 6 weeks after the nets were set (Kaiser *et al* 1996).

Marine debris arises from both land based and ocean based sources. In both categories there is considerable scope for improvement. In terrestrial environments there is a need for wide ranging education of the public in order to build awareness of the wider impacts of litter in the marine environment. In marine systems there is a need to rigidly enforce the provisions of MARPOL Annex V and to promote awareness within marine industries.

7.2 Introduction of Alien Species.

Human activity has resulted in many species being moved from their native aquatic ecosystem to other regions where they are not native inhabitants. A species that is removed from its native ecosystem and successfully reproduces in another is referred to as a non-native, non-indigenous or introduced species. While alien or non-established introductions consist of species that are inoculated to an ecosystem but are unable to establish self-sustaining populations. An introduction of either type of species is defined as a biological invasion irrespective of the number of the organisms involved and their ecological impact (GESAMP 1997a, Eno et al. 1997). It has been estimated that biological invasions of individuals from virtually every major taxonomic group of marine organisms have occurred somewhere in the world (Morton 1996). The consequences of biological invasions as a result of human activity are steadily becoming more serious. Control measures have been ineffective and preventative methods are still under development. Legislation to prevent the spread of non-native species currently focuses on terrestrial and freshwater ecosystems. While coastal and marine habitats have been consistently denied similar protection (Eno et al. 1997).

a) The Nature of Biological Invasions

This is not a new process. Invasions have been taking place since man first navigated across the oceans on ships. However the rate at which successful introductions are taking place has increased. The rate at which they are transported has increased. The susceptibility of the ecosystems which are inoculated with foreign organisms has also been enhanced by independent stressing factors such as pollution (Carlton 1996).

Deficient taxonomic data cripples the extent to which this problem can be assessed. Many species can not be conclusively categorised as non-native or native. Such species should be defined as cryptogenic. However they rarely are, which leads to much confusion in the data. In 1997, Eno et al. were able to document fifty non-native species in British waters. They presented a more accurate impression of the scale of the problem in the UK by also documenting the alien and cryptogenic species that could be found.

The possible economical impacts of such incidents became apparent in the mid to late 1980s as a result of a number of devastating invasions that took place around the world. Two specific biological invasions have become particularly infamous; the introduction of the American comb jellyfish, *Mnemiopsis leidyi,* into the Black Sea and the European zebra mussel, *Dreissena polymorpha*, into the Great Lakes. Both invasions resulted in similarly explosive and huge populations of organisms. However the devastating ecological impact that the *Mnemiopsis* has had on the natural finfish fisheries in both the Black and the Azov Seas has led to unprecedented losses by the local fishermen (GESAMP 1997a). While the zebra mussel will have cost a record $5,000 million in North America by the year 2000, as a result of fouling and blocking water intake pipes. Here the costs were incurred by the many industrial institutions along the banks of the Great Lakes (Morton 1996, Eno et al 1997).

The ecological impacts of a specific introduced species on an ecosystem depends on the condition of the ecosystem prior to the invasion. Whereas the human impacts are dependent on which activities generate the majority of the income in a particular coastal dwelling. Coastal resources are becoming more valued with time which has led to increasing concern about the cost of biological invasions (Morton 1996).

Non-native species that cause the largest ecological impacts often display a similar set of characteristics to each other. They often originate from similar latitudes, in either hemisphere, to the receiving ecosystem. They are commonly small and have rapid reproduction and maturing rates. It has been proposed that a list of the potentially most hazardous organisms could be compiled through identifying these characteristics. This would enable more effective protective measures to be put into place. The value of being able to predict when and where, which organisms will invade has been fully recognised (Daehler & Gordon 1997). However a framework on which these predictions could be based has not yet been agreed by international marine researchers (Carlton 1996, Grosholz & Ruiz 1996).

i) Vectors for Alien Species

The predominant vector for the transport of aquatic organisms from one port to another has been identified as ballast water in shipping. Today 80% of the world's commodities are transported by ship. The nature of ballast water as a transport mechanism has changed as the design of boats have been modified. The production of iron and steel boats introduced the large scale usage of liquid ballast for the first time in 1800s. In the 1990s it has been estimated that 3,000 species are being transported around the world every day (CSBO 1996). Due to the complex composition of the groups of organisms that are transported in this manner, it can be assumed that tens of thousands of species are inoculated into foreign waters on an annual basis (Carlton 1996).

Other factors are known to be involved because corridors have been in place for as long as 100 years before a particular organism successfully invades an ecosystem. As well as pollution-related stress, an ecosystem can become more susceptible to invasion as a result of over-fishing or the deliberate introduction of a non-native species for its economic value (Carlton 1996). Alien species thus have the potential to establish self-maintaining populations as a result of altered conditions in the receiving ecosystem or reaching a certain inoculation concentration. Other important methods of introduction include deliberate commercial introductions, unintentional introductions associated with such mariculture and transport by fouling and clinging onto ships' hulls (Eno et al. 1997).

ii) The Consequences of Biological Invasions

a) Ecosystem effects

Marine biologists are presently reviewing the validity of a traditional belief that humans were unable to cause any marine species to become extinct. This wisdom was based on the size and depth of the ocean and the prolific nature of its inhabitants. The vast geographical ranges that were thought to be generally typical of marine organisms were assumed to ensure the survival of any species that was put under pressure. As somewhere in the ocean's expanse an individual would find a sanctuary to repopulate in. Today complacency about the resilience of marine life is dwindling. The larger more visible marine organisms, such as starfish, crabs and fish, do have large geographical ranges because they produce long-lived larvae that can drift large distances. However the majority of marine life is on a smaller scale than these organisms, have been studied less vigorously and produce larvae that live less long and do not travel so far. Poor taxonomic data has also been accredited with causing this misconception. Recent genetic studies have revealed that widely distributed organisms which were previously believed to belong to the same species actually consist of more than one species with smaller geographical ranges and significantly different life histories. The edible mussel is one such organism; it is found in the North Atlantic and the North Pacific and they were assumed to belong to the same species, however in 1988 they were found to belong to three distinct, sibling species (Malakoff 1997).

This type of taxonomic data is essential for identifying, predicting and quantifying the impacts of non-indigenous species which are successfully introduced into ecosystems via human-mediated processes. Although it seems likely that the overall effect of non-indigenous species to date will have been considerable, the lack of information about marine ecosystems before boats travelled across the oceans ensures that the effects will only ever be estimated. This in itself should urge policy-makers to adopt a precautionary approach to the question of disrupting our marine ecosystems.

Current concerns focus not only on the ecological changes but also on the genetic influence such introductions have on introduced species. Non-endemic pathogens also pose a direct threat to all levels of the food-web (Morton 1996).

b) Human health effects

In July and September of 1991 and June of 1992 ballast water was transported from Latin America containing a strain of the toxigenic cholera bacteria (*Vibrio cholerae* O1 serotype Inaba, biotype El Tor) in 5 separate cargo ships to ports of the U.S. Gulf of Mexico. This pathogen posed a serious threat to human health via contaminated seafood such as oysters and fish. The U.S. Coast Guard and the FDA issued an advisory to shipping requesting that ballast tanks were flushed twice in the open oceans. Monitoring of the seafood from this area and the environment continued until October 1992 and no more contamination was found (McCarthy and Khambaty 1994). Cholera bacteria can coexist in bilge or ballast water with algae and plankton in a dormant form and thus be transported across large distances (Johnston et al. 1993).

Another type of threat to human health takes the form of exotic species that support parasites from their native part of the world. The Chinese mitten crab (*Eriocheir japonicus*) is one such species that was first observed in San Francisco Bay and the Sacramento-San Joaquin Delta region in 1994. It has raised concerns because it is an important second host for the Oriental lung fluke (*Paragonimus westermani*). Since many infected people have immigrated from Asian to California the arrival of this crab completes another link in the parasite's life cycle. If the mitten crab becomes abundant it may also serve as a host for the North American lung flukes *P.kellinotti* and *P.mexicanus*. In this way this particular biological invasion may cause an increase in the potential for human infection by lung flukes (Lafferty & Kuris 1996).

b) Understanding biological invasions

Marine ecosystems are extremely poorly documented. Only 275,000 marine species have been described. While the ocean's coral reefs and deep sea floors have been estimated to support 19 million species alone (Malakoff 1997). Thus any debate about extinctions or impacts is hampered by lack of information. At present marine invasion studies are usually inferential and are rarely able to combine extensive descriptive data with quantitative or experimental results (Grosholz & Ruiz 1996). It seems inevitable that many introductions as well as extinctions will have gone unreported (Carlton 1993).

Models of various kinds are being investigated however the priority at this stage has to be to learn from previous invasions. By developing a list of particularly dangerous organisms it would be possible to focus control measures and improve their efficiency. However, without legislation, introductions will continue to take place for short-term gain at the expense of long-term ecological and economic harm (Daehler & Gordon 1997).

A knowledge of community structures is essential. Suchanek (1994) describes how the use of "keystone species" could enable a framework for conservation efforts to be constructed. He maintains that the level of marine diversity is not inherently was needs to be preserved. Instead the ecosystem and how in functions needs to be protected through the dynamic set of interactions that maintain it. The integrity and long-term stability of these communities is reliant on the complex relationships that constructs them. Simplification is what needs to be prevented.

It is unlikely that it will be possible to entirely prevent further introductions from occurring. Instead the aims of contemporary research into this subject include:

(i) to reduce the rate at which successful invasions take place.

(ii) to predict which invasions are likely in any particular port, estuary or economically valuable ecosystem.

(iii) to develop effective management techniques to use when successful invasions do occur (CSBO 1996).

c) Methods of Control

i) Pre-Invasion Control

a) The Use of an International Inventory

Daehler and Gordon (1997) estimate that if "repeat offenders" were excluded or at least restricted the risks associated with introductions would be substantially reduced. We have a lot to learn from repeat offenders. Thus it is essential that a comprehensive inventory of the invasions that have taken place around the world is compiled, updated regularly and made accessible to the relevant decision-makers at the ground level. The information provided needs to be as detailed and include as much quantitative data as possible. This would provide the international marine research community with the basis to start understanding, managing, predicting and eventually preventing some of the more detrimental biological invasions (Grosholz & Ruiz 1996, Carlton 1996).

This inventory, for instance, should include seasonal information because the type of organisms transported and the receiving ecosystem varies with time and weather conditions resulting in differences in susceptibility to invasion. If patterns of invasion can be identified in a particular port then more efficient control measures can be implemented (Locke et al 1991).

b) Shipboard methods

The IMO published a set a guidelines in 1993 which is due for a much needed update. The Draft Assembly Resolution MEPC 40/21 which is due to replace the *Guidelines for the control and management of ships' ballast water to minimize the transfer of harmful aquatic organisms and pathogens;* Resolution A.774(18) includes some precautionary approaches for the management of ballast and more responsibilities for port authorities to monitor their harbours and keep shipowners informed of potentially dangerous organisms that are available for uptake.

The most significant recommendations include those which aim to reduce the number of dangerous organisms taken up into ballast water at source. This involves not taking ballast water onboard in areas where sewage outflow or harmful organisms (e.g. phytoplankton blooms) are known to be present, near dredging operations, or a tidal stream is known to be turbid or tidal flushing poor, in darkness or in shallow water. More rigorous reporting procedures are also due to be implemented.

Compliance remains to be predominantly on a voluntary basis which is unacceptable considering the risks that are involved. The Joint Nature Conservation Committee has called for European legislation which is thoroughly precautionary because biological invasions represent a 'self-regenerating form of pollution which may have irreversible effects on natural ecosystems and other species' (Eno et al. 1997)

c) Changing ballast at sea

The IMO guidelines presently hinge on the use of ballast water exchange in the open ocean to increase the salinity of the ballast tanks to levels above that which any freshwater species aboard can survive. Freshwater organisms released into the open ocean are thought to have little chance of survival. As a practice it certainly reduces the number of species that survive to have a chance to propagate in the destination port or estuary. However it has been found to be less than 100% effective, for a number of reasons. Locke et al. (1991) studied the ballast water of 59 foreign ocean-going vessels and estimated that the practice of mid-ocean exchange was only 67% effective. This inefficiency is attributed in part to incomplete exchange which leads to intermediate saline concentrations in the ballast tanks. Consequently, organisms with wide salinity tolerances such as the estuarine calanoid, *Eurytemora affinis*, were seen to survive. Sediment was present on almost all ships and this is also seen to pose a threat. Zooplankton are able to enter dormant stages in response to adverse conditions. As such they are able to survive in sediments in the form of eggs or cysts for hundreds of days. These sediments remain to be a source of alien species even after the tanks have been emptied as much as is feasible. For this reason, Locke et al. recommend that ships which declare that they are not carrying ballast water should also be monitored because when they next fill up and discharge ballast water it is possible that non-indigenous species might be introduced. Clams and mussels are also able to survive adverse conditions for up to seven days by closing their valves tightly. A comparable study of cargo ships in Australian ports estimated that 40% of the vessels carried viable dinoflagellate cysts in the sediment contained in their ballast tanks (Locke et al. 1991).

Locke et al. (1991) highlighted that although guidelines in the Laurentian waterway specifically about performing mid-ocean exchange practices were complied with at an adequate rate (89%) it was much more difficult to gain information in the form of ballast water exchange reports (BWER). A return rate of 39% was reported. If a complete picture of global water exchanges is to be tracked these BWERs should be compulsorily submitted. Compliance is very difficult to achieve without a universal law enforcement agency.

Safety considerations have also limited the ability of mid-ocean exchange to become fully applicable because weather conditions and individual boat designs can sometimes cause the procedure to become hazardous for the crew of the ships (IMO MEPC 40/21; Annex 6).

d) Other Shipboard Control Methods

Other methods of controlling the number of species transported in ballast tanks include filtration, biocides (oxidising and non-oxidising), thermal treatment, electric pulse/ pulse plasma treatment, ultraviolet, acoustic, magnetic, deoxygenation, biological and anti-fouling coatings. These treatments are well understood and presently used in wastewater treatment plants. However their use in this context is limited by the space and power available on board ship and the large volumes of water that need to be treated without increasing the hazards associated with the ship and crew. The Committee on Ships' Ballast Operations (CSBO) within the National Research Council (1996) evaluated these techniques and no single technology shone out.

The most promising technique however is physical filtration. The most feasible of which used a 1-5ìm mixed media fuzzy filters fitted with a backwash cleaning system to restrict organisms and sediment entering the ballast tanks. It would remove the vast majority of organisms whether in the water column or sediment load of the ballast water and thus allowing the discharge of ballast water to take place anywhere without risk. Filtered matter might need to be stored and disposed of at a shore-based facility. However the principle of shore-based treatment for ballast contaminated with oil has already been accepted, thus an expansion of these facilities should not be unobtainable.

Biocides and thermal treatment were also thought to be viable methods for control on certain types of vessels. The scientific community, however, is currently rejecting the use of chemical addition to inactivate micro-organisms for a number of reasons despite its practicability. Some of the reasons are listed here:

- reluctance to use a chemical that might be released into the sea or destination water system
- uncertain effectiveness in targeting nuisance species
- hazards associated with handling chemicals on board ship.

Most boats are unlikely to have the energy production ability to enable thermal treatment of ballast water. It could also upset the equilibrium of coastal ecosystems by altering their average temperature (GESAMP 1998, CSBO 1996).

ii) Post-Invasion Control

a) Eradication or control of introduced species

Possible methods include:

Mechanical control **-** such as harvesting or removal. However the feasibility of this method is dependent on the scale of the invasion, the type of organism and the availability of manpower. It did prove efficient enough to control the spread of the Chinese mitten crab in Germany in the 1920s which was causing substantial erosion in the Lower Rhine Valley. An intensive trapping program succeeded in extirpating them. It can also be used in conjunction with other methods (Lafferty & Kuris 1996).

Chemical control **-** the use of pesticides. At present very few aquatic pesticides exist. They are unlikely to become as widely used a their land-based equivalents because the pest is often physiologically similar to the resource (e.g. fish, crustacean, mollusc) that the control method is supposed to be protecting. Large amounts of money and time would have to be invested in order to produce chemicals with high enough specificities. Dilution and dispersion also cause problems with the use of pesticides in aquatic systems. The concentrations that are applied have to be increased accordingly and the area of application can not be restricted. The environmental consequences of this control method are thus increased to an unacceptable level (GESAMP 1998, CSBO 1996).

Biological Control **-** introducing a non-indigenous predator to control the nuisance species. This method is currently receiving a certain amount of credibility based on the premise that marine pests do not need to be controlled to the low levels required by farmers in agriculture. Outbreaks simply need to be prevented (Lafferty & Kuris 1996). However the dispersal, host preference, genetic variation of proposed biological control agents and the interaction of these factors with the environment need to be studied extremely carefully in order to ensure the safety of native species. It must also be considered that these predators could feasibly become pests should these studies not be performed diligently enough (Daehler & Gordon 1997).

d) Overview

The community of marine biologists of today have an immense task on their hands. They have so many foundations of understanding to lay in order to catch up with their terrestrial counterparts. It is essential that previously incomplete studies are re-evaluated to correct for the sampling biases that have existed during previous generations. The only way successful changes will be implemented in time to prevent massive losses in the integrity of our marine ecosystems will be through comprehensive international prioritisation and collaboration. As Brundtland wrote last year, “close cooperation between scientists and politicians is the only way forward” specifically in the case of ocean management. She also warned of compromising on scientific evidence or facts because the costs of repairing nature would become “enormously costly”. Assuming that it would even be possible during a timescale relevant to mankind.

Meanwhile, it is essential that all sensible precautions are taken immediately. The guidelines, published by the IMO, with respect to ballast water procedures must be enforced by ship captains and harbour masters with rigour and diligence. In this way it may be possible to reduce the number of biological invasions that occur by the greatest amount possible. We must act now to protect our coastal and marine ecosystems from the uniformity that biological invasions promise to cause eventually. A precautionary approach affording saline ecosystems the same aegis that is given to terrestrial and freshwater systems from non-native species has to be implemented without further delay.

Human activity has resulted in many species being moved from their native aquatic ecosystem to other regions where they are not native inhabitants. A species that is removed from its native ecosystem and successfully introduced into another is referred to as an alien, non-indigenous or introduced species. An introduction of this nature is defined as a biological invasion irrespective of the number of the organisms involved and their ecological impact (GESAMP 1997a). It has been estimated that biological invasions of individuals from virtually every major taxonomic group of marine organisms have occurred somewhere in the world (Morton 1996). The consequences of biological invasions as a result of human activity are steadily becoming more serious.

8. Operational Discharges from the Offshore Industry

8.1 Introduction

To date the use and release of chemicals by the offshore industry has been relatively poorly controlled and documented. Although some of these agents are chemically inert, many are hazardous or contain hazardous components. There is increasing evidence that the discharge of cuttings and chemicals during normal exploration and production operations has had significant impacts on the chemistry and biology of the marine environment surrounding test, operational and redundant wells, even at distances of several kilometres from well sites. In some cases, effects have been reported to persist for many years.

8.2 Discharge of drill cuttings

Besides the physical impact, concerns have been raised over the chemical content and toxicity of oil residues in discharged cuttings. Elevated concentrations of oil in sediments have been detected up to 5-10km away, even from small groups of wells using oil-based muds (Daan *et al.* 1992, Olsgard and Gray 1995). Olsgard and Gray (1995) reported contaminated areas of between 10 and 100 km^2 surrounding the Veslefrikk and Valhall oil fields respectively. Caged mussels 5 km down-current from cuttings discharges have been found to contain higher than background tissue oil concentrations (Zevenboom *et al.* 1992).

The toxicity of oil-based mud (OBM) residues to marine fauna has been fairly well documented, and includes impacts on feeding, settlement and reproductive development in bivalves (Cranford and Gordon 1991, Stromgren *et al.* 1993, Plante-Cuny *et al.* 1993) and on immune response in fish (Tahir *et al.* 1993). While major impacts on benthic organisms may generally be restricted to a limited area around platforms, effects on populations of more sensitive species, especially sea urchins, have been recorded several kilometres away from the discharge. Effects on overall community structure have been detected as far as 2-6 km from disused platforms in the North Sea (Olsgard and Gray 1995).

Drill cuttings which are not recovered after disposal can be redistributed on currents, the area contaminated continuing to grow for as long as 6-9 years after discharge has ceased (Daan *et al.* 1992). At the same time, degradation of oil residues appears to be inversely related to the level of contamination (OSPAR 1997a). Within the cuttings pile, biodegradation may be negligible, even over extended periods (Daan *et al.* 1996). Although accelerated degradation may occur at the periphery, this could in turn result in localised enhancement of oxygen depletion, or of toxicity, bioavailability or dispersal of contaminants. Washing of OBM cuttings prior to discharge may reduce the severity of effects, but generally has little influence on the area impacted or the persistence of contamination and effects (Daan *et al.* 1992).

Concerns over the toxicity of OBM led to PARCOM Decision 92/2 on the Use of Oil-based Muds. Alternatives to OBM include fluids using organic compounds (synthetic muds, SM) or water (water-based muds, WBM) as base agents.

SM bases may also be persistent and/or exert toxic effects on benthic organisms. Although fluids with an olefin (alkene) base are not considered as OBMs, persistence can be similar to, or even greater than, that for a mineral oil base (OSPAR 1997a). In addition, many of the chemicals in SMs have a high potential for bioaccumulation (OSPAR 1997b). For those synthetic bases which do degrade more rapidly, most notably the ester bases, toxicity and oxygen depletion remain as problems. For example, Daan *et al.* (1996) reported impacts of ester-based muds on sensitive species (sea urchins) even 1 year after drilling had ceased.

Exposure of larval and adult marine invertebrates even to water-based muds in use off southern California revealed some negative impacts on brown cup corals and effects on settlement of red abalone (Raimondi *et al.* 1997). This latter response would not be detected using standard toxicity tests, although the potential implications for community structure could, nevertheless, be severe.

8.3 Discharge of produced water

Produced water can contain high concentrations of salts, heavy metals, radioisotopes, dispersed and dissolved oils, detergents and other production chemicals. The 160 million m^3 of produced water discharged by North Sea offshore operators in 1991 contained more than 5000 tonnes of oil, 52 000 tonnes of total organic compounds and 250 tonnes of heavy metals (E & P Forum 1994). PARCOM regulations limit the concentration of dispersed oil to 40 parts per million (ppm), but little attention is paid to other contaminants. Volumes of produced water recovered increase with the age of a well as oil or gas is depleted. In the North Sea, volumes are expected to increase to 340 million m^3 by 1998 (E & P Forum 1994).

Laboratory-based investigations of produced water toxicity have generally focused on lethal effects and have overlooked a wide range of sub-lethal and behavioural effects which may be seen at concentrations of produced water chemicals as low as 5-15 parts per billion (ppb). Chronic effects of produced water discharges have been observed on planktonic and benthic organisms and fish in the field following discharge of produced water from shallow water platforms in the US (Slager *et al.* 1992).

A number of recent studies have highlighted the significance of ionic concentration as a contributor to overall produced water toxicity (Douglas and Horne 1997). Nevertheless, overall toxicity frequently cannot be described in terms of ionic content alone . In a study of 14 produced water samples from sources in the US, Sauer *et al.* (1997) identified a range of compounds with the potential to contribute to toxicity, including hydrocarbons, ammonia, hydrogen sulphide and more poorly characterised acidic and basic organic compounds, volatile organic compounds and particulates. Such contaminants may be introduced from drilling fluid or other production chemical formulations or are recovered from the well itself.

8.4 Use and discharge of production chemicals

The range of production chemicals employed, and which may subsequently appear as contaminants in produced water or drill cuttings, is large and chemically diverse, including emulsion breakers, corrosion inhibitors, scale inhibitors, coagulants, flocculents, acidising chemicals, paraffin control chemicals, bactericides and water clarifiers (Sauer *et al.* 1997). Information on formulations and quantities used is generally confidential. In addition, reporting of chemical usage has historically been very poor and remains extremely limited, particularly for non-oil chemicals.

It is clear that the use of production chemicals can contribute to contamination of drill cuttings and produced water. For example, in addition to n-alkanes, the production chemical derivative nonylphenol was identified in samples of sediment collected adjacent to three drilling rigs in the North Sea in 1997 (OSPAR 1997c). Concentrations varied from 0.4-30 mg/kg (dry weight) in contaminated sediments; levels at a reference site were below limits of detection.

Recognition of the toxicity and persistence of the alkylphenols led to PARCOM Recommendation 92/8 to phase out use of nonyl and octylphenol ethoxylates in household (by 1995) and industrial (by 2000) detergents. More recently, concerns have been raised over the ability of the alkylphenols to interfere with hormonal communication and sexual development in fish (Jobling *et al.* 1996), including estuarine and marine species (Lye and Frid 1997, Hylland *et al.* 1997).

8.5 Regulation of the offshore industry

In comparison to land-based industry in Europe, the offshore industry is relatively poorly regulated. Even if current legislation were strictly adhered to, substantial contamination of, and effects upon, marine communities may still be expected. PARCOM regulations on produced water, for example, cover only dispersed oil, taking no account of the water soluble fractions which are often more toxic. Contamination from historical use is also significant as there is no requirement to recover contaminated cuttings to land.

In the mean time the discharge of offshore production chemicals in drill cuttings continues to be poorly documented and regulated. Under PARCOM Decision 96/3, all applications to use or discharge offshore production chemicals should be accompanied by detailed composition and hazard information according to the Harmonised Offshore Chemical Notification Format (HOCNF). The CHARM model, which is central to the process of ranking of chemicals under HOCNF, is limited in scope, relying heavily on lethal toxicity tests with organisms which are unlikely to be the most sensitive components in the ecosystem (Power and McCarty 1997). Genetic variation within populations (Evenden and Depledge 1997) may present further problems in assessment of potential impact. Existing bioaccumulation tests under HOCNF may be inappropriate for many surface active agents included in drilling fluid formulations (OSPAR 1997b). In the past, implementation of CHARM has also suffered from lack of information on chemicals in use as a result of commercial confidentiality of proprietary formulations.

If, as intended, implementation of HOCNF proceeds through mandatory toxicity tests on a chemical by chemical basis, discharge of certain hazardous chemicals can be expected needlessly to continue for some time to come. The range and sheer complexity of drilling fluid formulations will render such an approach laborious and one which is unlikely to yield information useful for the prediction of toxicological interactions in chemical mixtures. Poor data availability in relation to chemical content, hazard and environmental fate of produced waters and other discharges, in addition to a limited understanding of interactions in complex chemical mixtures, will continue to present fundamental limitations to risk assessment procedures.

Clearly there is a requirement for greater transparency and control of offshore operations. While development of waste treatment technologies may contribute to reduction of discharges of some chemicals, this solution has limited application overall, particularly with more persistent contaminants. A review and tightening of current national and international legislation on discharges is essential, including rigorous application of the precautionary principle and recognition of zero-discharge as a goal for existing operations. There is an urgent need within international agreements to recognise the importance of ecosystem protection as the central goal of environmental regulation, based in turn on an holistic appreciation of the diversity and complexity of ecosystem fluxes and processes.

9. Global Changes

9.1 Greenhouse Gases

The Industrial Revolution, the start of major industrial activity, marked the point at which humans began to significantly and rapidly alter a number of biogeochemical cycles. These include the chlorine, sulphur, nitrogen and carbon cycles. Some of these biogeochemical cycles had been influenced inevitably by deforestation and evolving agricultural practices and indeed still are. The wide scale burning of fossil fuels to provide energy, however, has heavily and directly influenced the natural carbon and sulphur cycles. Some of the energy produced has been used, in turn, to convert nitrogen and chlorine to other chemical forms, thus influencing the global cycle of these elements (IPCC 1996a; Andrews *et al.* 1996).

Carbon dioxide, methane and nitrous oxide concentrations have risen steadily in the atmosphere as compared to preindustrial times. From the end of the last ice age 10,000 years ago, ice core records show that the concentration of carbon dioxide in the atmosphere was relatively constant until the beginning of the 18th century. The preindustrial concentration of around 280ppm had risen to 358ppm by 1994 (IPCC 1996a), an increase of around 30%. Methane, another component of the carbon cycle, also rose in atmospheric concentration by some 145% from around 700ppb to 1720ppb over the same period while nitrous oxide has increased by around 15% from around 275ppb to 312ppb. In addition, from the 1940's onward industrially produced gases, based on chlorine chemistry and with only very minor or non-existent natural sources, have been released into the atmosphere. Releases have been in kiloton quantities annually and they are now present at ppt levels. These gases include the CFCs, which are responsible for stratospheric ozone depletion and their substitutes the HCFCs and HFCs. Recently, it has been found that fluoroform, a by-product of HCFC manufacture has been steadily increasing in the atmosphere over the period 1978-1995, and total atmospheric loadings are now equivalent to 1.6 billion tonnes of carbon dioxide in terms of their impacts upon global atmosphere. Sulphur hexafluoride is another industrial gas which has increased in the atmosphere in recent years (Oram *et al.* 1998). In addition, NO_x from aircraft flying at high altitude also contribute to an unknown extent (Skodvin & Fugelstvedt 1997).

These gases all have in common the fact that they act as "greenhouse" gases. The atmosphere is relatively transparent to most of the incoming solar radiation which heats the surface of the earth. By contrast, energy emitted from the earth is absorbed by water and "greenhouse" gas molecules and retained within the atmosphere. Carbon dioxide is the largest contributor to the effect, but the contributions of the other gases are also important. Although they are present in the atmosphere at lower concentration, molecule for molecule they are more effective at trapping heat emitted from the Earth. The scope for increasing carbon dioxide levels in the atmosphere is immense unless emissions from the burning of fossil fuels are brought under control. Estimates suggest that in preindustrial times the atmosphere contained 590 billion tons of carbon as carbon dioxide. Between 1765 and 1991 this rose to 755 billion tons (Moore and Braswell 1994). Estimates of recoverable fossil fuels vary, but 10,000 billion tonnes as carbon is a reasonable figure. The Intergovernmental Panel estimates that current emissions are around 7.1 billion tons of carbon annually. This broadly agrees with the current estimated rates of carbon dioxide concentration increase of around 0.4% per year (IPCC 1996a).

The potential for industrialisation and the burning of fossil fuels to increase atmospheric concentrations has been appreciated since at least the turn of the century following the work of Svante Arrhenius (see: Arrhenius 1997). Arrhenius with an outlook typical of the cultural optimism at this time suggested that there would be generally beneficial effects. This view has been considerably modified in modern times by valid concerns about possible unwanted consequences. Mathematical models predict that as levels of these greenhouse gases increase in the atmosphere, a greater proportion of the heat radiated from the earth is retained in the lower atmosphere, while the higher atmospheric layers cool (Andrews *et al.*1996). This model is supported by temperature measurements in both the lower and upper atmosphere (IPCC 1996a). Computer models can also be used to predict the scale of the temperature rise. Using these, the Intergovernmental Panel on Climate Change has estimated that the increase in global average surface air temperature in the year 2100 relative to 1990 will be between 1 and 3.5C. It is estimated that air temperatures have increased by between 0.3 and 0.6C since the late 19th century and by 0.2-0.3C over the last forty years. Increased surface air temperatures of the scale projected for 2100 are extremely large considered in historical terms, larger in all probability than any following the emergence of the earth from the last ice-age 10,000 years ago. One key area of uncertainty is the value of climate sensitivity, that is the actual temperature rise which will result from a doubling of atmospheric carbon dioxide. The most recent estimates (Tol & de Vos 1998) suggest that this could be as high as 3.8ºC ±0.9ºC. This is somewhat higher than the estimates used by the IPCC in their projections although values anywhere between 0.3 and 5.3ºC have been estimated by various workers at different times.

Since climatic processes are largely driven by temperature changes and differences, any increase in temperature of the scale estimated will be associated with major climatic changes around the world. The prediction of the precise changes in climate which are likely to result from increased temperatures is highly uncertain. Limitations exist in the models due to our poor understanding of total global processes and consequent failure to consider all the possible influencing factors. Many of the potential controlling feedback mechanisms are also only poorly understood. Natural variability masks many trends and this can only be compensated for by considering data obtained over a long time period. The difficulties in deriving robust models for climate change are considerable (see Howe & Henderson-Sellers 1997) and much effort has been directed at refining them (McBean 1994). Superimposed upon any long term trends may be decadal variations. These seem to be partly associated with the North Atlantic Oscillation (Hurrell & van Loon 1997) and with the El-Niño Southern Oscillation (Zhang *et al.* 1998). Solar variability may also be of importance. Long term variation in the total irradiance of the sun has been identified which corresponds with the 11 year solar cycle (Reid 1997). Other work has shown apparently good correlations between northern hemisphere land surface temperatures and sun-spot activity (Lassen & Friis-Christensen 1995; Friis-Christensen 1993). Nonetheless, such cyclic phenomena are unable to explain all of the observed temperature rise (Cubasch *et al.* 1997). The pattern of recent observed warming agrees better with the greenhouse warming pattern: the observed temperature rise over the last 30 year and 100 year periods is larger than the trend due to solar forcing alone in modelled simulations.

On the basis of the evidence available the Intergovernmental Panel (IPCC 1996a) considered that "climate has changed over the past century" and that "the balance of evidence suggests a discernible human influence on global climate" a conclusion strengthened by the developing body of observational data (see e.g. Kaufmann & Stern 1997). Studies of extended data sets of historical climate indices suggest that solar irradiance levels partially explain the cold period between the mid 17th and mid 18th centuries and the gradual warming from the early 19th to mid 20th century. A peak correlation emerges during the mid 19th century and is followed by a sharply positive correlation with greenhouse warming in the 20th century (Mann *et al.* 1998). On the basis of these and other observations, Mann *et al.* (1998) consider that "it is reasonable to infer that greenhouse-gas forcing is now the dominant external forcing of the climate system." This concurs with the results of various modelling studies although it is emphasised that uncertainties need to be resolved.

Among the conclusions of the IPCC (1996a) scientific analysis of trends was that the global mean surface temperature increase of 0.2-0.6°C recorded since the late 19th Century has been accompanied by decreased temperature ranges, with nightime temperatures increasing more than daytime values. Minimum temperature values have increased by twice the value of maximum temperatures over the past 40 years. In some regions, this seems to correspond to changes in atmospheric circulation patterns. The reduced daily temperature range observed in northern Europe is consistent with increased westerly flow of the atmosphere (Easterling *et al* 1997). There have also been changes in precipitation over land areas. Overall levels have been falling since 1980 with decreases being noted after 1960 in subtropical and tropical areas. Limited data suggests that over the equatorial Pacific Ocean, precipitation has increased over the past few decades. Cloud cover seems generally to have increased over the oceans since the 1950s as it has over many terrestrial areas (IPCC 1996c). Sea level has risen by between 10 and 25cm over the last century with melting of ice caps and glaciers contributing to this. At the same time there is clear evidence of changes in extremes of climate and in indices of variability of climate on regional scales. Some of these trends are towards greater variability, while for other indices variability has decreased.

Although some of the changes in climate and climate indices appear at first sight to be contradictory, they are broadly in agreement with the output of mathematical models designed to simulate global climate processes (DETR 1997; IPCC 1996c; Karl *et al.* 1997). The role of the oceans as a store of heat, carbon and CO_2 and as a major source of water to the atmosphere can be incorporated into "coupled" models and unsurprisingly, this improves the physical realism of the simulations. In particular, it helps resolve many issues concerning the timing and regional distribution of anthropogenically driven climate change (IPCC 1996c). Generally, the continuing refinement of the mathematical models used has provided further strong evidence that the observed trend in global temperature over the last century is unlikely to be solely of natural origin (Kaufmann & Stern 1997). Analysis of historical data has shown that since credible instrumental global temperature records began 140 years ago, 1997 was the warmest year on record . Overall, nine of the past eleven years have been the warmest on record (Quayle *et al.* 1998). Northern hemisphere mean annual temperatures for three of the past eight years are warmer than any other since at least 1400AD (Mann *et al.* 1998). Large inter-annual variations are superimposed upon long term trends together with inter-decadal variations. In part these variations are caused by changes in solar output and by cooling due to dust ejected into the atmosphere by volcanic eruptions. When these factors are taken into consideration, it becomes apparent that the last two decades have been unusually warm (DETR 1997). By including the effects of cooling sulphate aerosols, models suggest that observed patterns of temperature change across the surface of the earth and through the depth of the atmosphere can be explained through the influence of human activities. Undoubtedly, as temperatures continue to rise impacts will become increasingly obvious and the trends will become more clearly defined and more easily identified.

a) Impacts upon Ocean Ecosystems

Much work to date has concentrated upon the impacts of climate change upon terrestrial systems. For example, models have been used to predict the likely impacts upon natural vegetation and upon agricultural systems. It is thought that by the year 2080, many tropical grasslands and tropical forests will be at risk of decline. There is likely to be both a loss of areal extent and of biomass, although many temperate and boreal forests may increase as a result of projected changes. After about 2070, it is estimated that as the tropical forests are adversely affected , this major carbon sink will no longer act as a "brake" on increasing levels of carbon dioxide in the atmosphere. Major changes are also expected in agricultural systems at a regional level, particularly in the arid and sub-humid tropics, although global food production is expected to remain sufficient for human needs (DETR 1997). There are also likely to be extensive changes in water resources on a global basis as a result of changes in precipitation regimes. The impacts upon the oceans themselves have been less well evaluated, but climate change has the potential to significantly affect biological diversity in ocean and coastal areas. It could cause changes in the population sizes and distributions of species. These changes may come about as a result of the combined effects of sea level rise and changes in sea surface temperature, together with increases in the frequency of storms and other extreme events (Michener *et al.* 1997). Increased precipitation could lead to increased run-off of pollutants and nutrients due to increased erosion.

As noted by IPCC (1996a) through their fluid motion, high heat capacity and biotic processes, the oceans play a pivotal role in shaping the earth's climate and its variability. Ocean circulation redistributes heat and water. The ocean thermohaline circulation moves water from the surface to the deep ocean. In addition, the oceans absorb carbon dioxide and other atmospheric gases and exchange these gases with the atmosphere. Changes in climate are likely to both be driven by oceanic processes and to drive the changes affecting them in response to a number of identified feedback loops. In addition temperature regimes are an important determinant of organismal and whole ecosystem function and changes in temperature are likely to alter the species composition and geographical extent of habitats and ecosystems and markedly increase the rate of species extinctions (Reid and Miller, 1989). As a result of increasing temperatures at the poles and a consequent decrease in the temperature gradients from pole to equator, it has been suggested that trade winds will weaken. The strength of upper ocean currents and the intensity of coastal and oceanic upwelling is likely to be reduced. Hence, it is possible that there will be a general decrease in the biological productivity of the oceans as climate change progresses (IPCC 1996b).

Much of the evidence of the potential for wide reaching changes in ocean ecosystems is provided by limited studies of defined regions covering a relatively small diversity of habitats and physical regimes. Each of these studies, however, provides further evidence against which projected trends may be ultimately confirmed. Studies of planktonic organisms over relatively long time scales have shown that these may be changing in some areas. Since 1951 the biomass of macro-zooplankton in the waters of the California current has declined by some 80%. This coincides with a warming of the surface water over the same period of up to 1.5°C. This reinforces the thermocline and reduces the availability of nutrients to planktonic organisms. Although the impacts in California may well be explained by regional as well as global trends, if a temperature rise of 1-2°C takes place globally and ocean thermal stratification increases, then the biological impacts are likely to be devastating (Roemmich & McGowan 1995). Studies of a Californian rocky shore community have indicated that annual mean shoreline temperatures increased by 0.75°C during the past 60 years. The mean summer maximum temperatures over the period 1983-1993 were 2.2°C warmer than over the period 1921-1931. Shifts in faunal and floral community structure were also recorded over the same period representing an integrated response to long term change. Species with a southern distribution increased in number while those associated with cooler, northerly habitats declined (Barry *et al.*1995).

Preliminary results of studies of phytoplankton in the North Atlantic using continuous plankton recorder data from the period 1948-1955 have shown a decline north of 59°N while further south, the length of the growing season and phytoplankton abundance appear to have increased. The decline in northern latitudes has been tentatively attributed to changes in deep water convection in turn caused by increased formation of Arctic surface water in the Greenland Sea and/or increased export of freshwater from melted ice (Reid *et al.* 1998). Evidence for Arctic warming is considered in Section 9.1c below. In parallel with these marine observations, there is evidence that photosynthetic activity in terrestrial plants has increased over the period 1981-1991 between 45°N and 70°N coinciding with marked increase in springtime warming and an early disappearance of snow (Myneni *et al.* 1997).

In addition to direct impacts upon ocean ecosystems of temperature increase, indirect effects are likely to occur. Increased precipitation may change the salinity regime in critical breeding grounds for fish. Changes in water circulation patterns may well modify the physico-chemical parameters which govern the distribution of marine organisms. On the coastal margins much ecologically important habitat is likely to be lost as sea levels rise. The role of the oceans in storing dissolved CO_2 may also change, in turn affecting the rate and magnitude of any predicted changes. The limited studies carried out to date at least illustrate the scope for ecological change in the oceans as a result of climatic change. Otherwise the precise impacts remain largely undefined. Predictions of impacts upon fisheries, for example, suggest that the potential positive impacts conferred by longer growing seasons and lower winter mortality may be offset by changes in reproductive patterns, migration routes and ecosystem relationships (IPCC 1996b).

b) Coral Bleaching and Climate Change

It is also likely that the bleaching of corals will increase as temperature tolerance maxima are exceeded over increasingly lengthy periods (IPCC 1996b). The phenomenon of coral bleaching has been attracting considerable attention in recent years and can take place in response to a number of environmental stressors (D'Elia *et al.* 1991). Corals react to thermal and other stresses by expelling the symbiotic algae which gives them their colour, and prolonged bleaching leads to reduced growth and reproduction. If the symbionts do not become re-established, death of the entire colony results. This may result in increased levels of extinction (Chadwick-Furman 1996) and may be particularly severe where reef overfishing or other human activity has already resulted in large scale ecological phase shifts (Hughes 1994). While corals can recover from coral bleaching events, mortality during severe events can be as high as 100% (Hoegh-Guldberg *et al.* 1996) Thermal stress associated with ENSO events was found to cause 97% mortality in some Galapagos corals (Chadwick-Furman 1996). Mass bleaching of corals on some of the central and northern Great Barrier Reef in 1982 in Australia was estimated to have killed more than 50% of corals on some reefs (Oliver 1985). In addition to killing the corals, bleaching can also diminish the capacity of corals to reproduce and this may have subsequent negative effect on coral recruitment (Glynn 1996).

Coral bleaching can be induced by changes in water salinity, increased solar radiation, pollution and positive or negative changes in sea temperature (Goreau & Hayes 1994; Glynn 1996). Sensitivity to temperature changes appears often to be acute: Differences in average daily seawater temperatures increases of less than one degree centigrade have been shown to induce bleaching in corals off Magnetic Island on the Barrier Reef (Jones *et al.* 1998), this temperature difference broadly defining bleaching and non bleaching years. Globally, mass coral bleaching events appear to have become more frequent, widespread and severe since the early 1980s and high sea surface temperatures have been suggested as the primary cause of these large scale events. Coral in tropical and sub-tropical locations exist at temperatures within 1-2ºC of their upper thermal lethal limit during summer months. The temperature elevation resulting in stress in coral appears to be between 1 and 2ºC irrespective of the local average maximum temperature (Chadwick-Furman 1996). The observed increase in bleaching since the 1980s is regarded as consistent with measured sea temperature increases and as increasing confidence in direct temperature observations (IPCC 1996a). It has also been suggested on the basis of the same observations that coral bleaching may be one of the first major impacts of climate change upon ocean ecosystems becoming manifest (Glynn 1991; Goreau & Hayes 1994).

The Australian Great Barrier Reef occupies a centrally important position in conservation terms. It is relatively pristine when compared to other reefs due to it being remote from high density human populations. As less remote reefs are degraded by direct human pressure, the Barrier Reef and other remote reef formations could have an important function as refugia of coral reef biodiversity (Wilkinson & Buddemeier 1996). Nonetheless, although partially immune from degradative impacts affecting other reef areas, it cannot be protected against elevated temperatures due to climate change and a striking correlation has been observed between elevated sea surface temperatures and bleaching events which have occurred in 1980, 1982, 1987, 1992, 1994 and 1998. (Hoegh-Guldberg *et al.* 1996; GBRMPA 1998). Indeed, the 1998 event appears to be the most severe and extensive ever recorded. Aerial surveys indicate that more than 88% of inshore reefs are suffering from bleaching. More than 60% of corals have been bleached over an area of 25% of the inshore reefs. This is regarded as a severe episode.

Very high coral mortality rates have been recorded around the Palm Islands, north of Townsville

(GBRMPA 1998). As in previous bleaching episodes, the root cause appears to be abnormally high sea temperatures with high levels of solar irradiance and low salinity caused by spate river flows probably contributing. Lack of long term observations mean that it is not possible to identify whether there is an increasing trend over the Barrier Reef as a whole, but certainly in the case of the Magnetic Island reefs, bleaching events of 1979/80, 1981/82, 1986/87 and 1993/94 appear to be related to heatwave conditions (Jones *et al.*1998) with a close correlation with average daily seawater temperatures approaching 32ºC. In the 1994 event, average daily water temperature increase by 2ºC over a period of one week reaching 34ºC on the reef flat and 32ºC over the reef slope. Bleaching was observed shortly afterwards (Jones 1997). It was concluded from the study of these events and of data from the 1930s onwards that bleaching events at Magnetic Island were a comparatively recent phenomenon and were associated with a change in the local climate system. Annual air temperatures in the locality have increased by nearly 1°C since 1942 while across the Barrier Reef region as a whole temperatures have also increased significantly this century (Lough 1997) consistent with observed Australian and hemispheric trends (Lough 1995). In combination, the direct and indirect evidence supports the view that current high sea temperatures, while not unprecedented over the last 300 years, may now be at their highest this century (Barnes & Lough 1996).

Future climate scenarios suggest that by 2070 temperatures could be up to 2.7°C higher than present with a concomitant increase in the incidence of extreme high temperature events. It is doubtful whether corals could adapt to accommodate this rate of temperature rise and the likely associated extreme events. Hence, these projected increases pose a major threat to coralline ecosystems irrespective of whether recent bleaching events are part of an emerging trend. This could result in the widespread decline of the reef and reduced biological diversity (Glynn 1996). Impacts would be expected to be most severe along inshore fringing reefs because of the more direct linkage between elevated air and sea surface temperatures. In addition, the occurrence of extreme rainfall events across Australia appears to have increased over the last century (Plummer *et al.* 1998) and increased freshwater will be likely to decrease salinity and increase sedimentation , both of which phenomena are deleterious to corals. Overall (IPCC 1996a) this would exert major impacts on the Great Barrier Reef Lagoon and lead to extensive degradation of the environment of this system.

In addition to the concerns about elevated temperature, it is possible that increased levels of CO_2 in the atmosphere and hence in seawater will affect the ability of corals to form their calcareous skeleton. Increased concentrations of carbon dioxide in seawater could cause alterations of carbonate chemistry. It has been predicted that doubling of atmospheric carbon dioxide above pre-industrial concentrations could reduce coral calcification rates by between 10 and 30% (Gattuso *et al.* in press), impairing their capacity to keep pace with sea level rise or increased storm damage. To a greater or lesser extent the combined impacts of these changes will affect coral reefs globally. Another key area of uncertainty is the extent to which the El-Niño Southern Oscillation (ENSO), a periodic disturbance of ocean currents in the Pacific Ocean which profoundly influences global systems, will impact upon coral and their associated ecosystems. Examination of coral growth records over 96 years at Tarawa Atoll in the equatorial Pacific has shown that growth was correlated with the strength of the El-Niño index.

c) El-Niño Southern Oscillation

El-Niño is the name given to an irregular but periodic appearance of warm surface water along the equator in the Central and Eastern Pacific. This is associated with a change in atmospheric pressure at Darwin, Australia relative to that at Tahiti. This periodic change in relative atmospheric pressure is known as the Southern Oscillation. The El-Niño/Southern Oscillation is an event in which the strong easterly trade winds which drive the Equatorial Current from east to west to weaken. These trade winds normally push warmer waters into a "pool" in the Western Pacific. The weakened air flow then allows the equatorial current to flow in the opposite direction (Betshill *et al.* 1997) warming the waters of the Eastern Pacific and causing the warm pool to fall in temperature. Once started, the phenomenon becomes self reinforcing through ocean-atmosphere coupling. This weakens the trade winds further. Because this change also perturbs the atmosphere within tropical regions and to a lesser extent outside these regions, a number of "teleconnections" become established. These lead to wide ranging effects upon weather and climate on a global basis. The ENSO phenomenon is described as the single most significant source of inter-annual variability in weather and climate around the world (IPCC 1996b). ENSO events end as the cooled warm pool starts to reheat and the trade winds strengthen. The water in the Eastern Pacific then becomes cooler leading to La Niña conditions. Events generally begin in the northern spring when trade winds are weakest and may last up to 18 months.

El-Niño events occur on average every three to seven years although in recent years their frequency appears to have increased. In the 1980s ENSO events have occurred at about four year intervals. Strong events have occurred about once a decade and the gradual progress of the events can be traced through changes in oceanic and atmospheric variables recorded across the whole Pacific Basin (Tourre & White 1997). The impacts recorded during previous events appear to be comparable to the impacts being documented as a result of the current ENSO event which began in 1997, and seems to be of a record intensity (see: Karas 1997). The 1982-83 ENSO event was associated with droughts in Indonesia, India, Australia and the Philippines. Serious flooding occurred in Peru and Ecuador and serious storms were recorded in California (Yan *et al.* 1997). Retrospective examination of climate indices for the Yangtse Basin in China has shown that all regional extreme climate events in the form of floods and droughts over the past 500 years occurred in conjunction with ENSO events (Chang 1997). ENSO signals have also been detected in sea-ice cover in the Western Antarctic region (Stammerjohn & Smith 1997). At the same time, where upwelling of cold bottom waters is suppressed by El-Niño, this reduces nutrient availability. As a consequence, serious declines of fisheries have occurred. The collapse of the anchovy fishery was an economic disaster for Peru during the 1972 event. The effect of the El-Niño, in combination with other environmental factors, was to concentrate anchovy over a smaller range allowing overfishing to take place on the whole population. At the same time the production centre associated with the Humboldt current shifted southward from north-central Peru to the Peru-Chile border. Other recorded changes include displacement of salmon migration routes and changes in the distribution of whiting in Eastern Pacific waters (IPCC 1996b). The economic impacts of ENSO events can be collectively severe (Karas 1997) running into billions of dollars due to floods, extreme weather and crop losses.

Under conditions of climate change it is possible that ENSO events may become more frequent, more intense or indeed both (see: Karas 1997; Trenberth & Hoar 1996; 1997). There is evidence from the palaeorecord and documented historical sources which shows that ENSO events have changed in frequency and intensity on multi-decadal to century timescales (IPCC 1996b). There appears to have been a distinct change in the behaviour of ENSO events dating back to the mid-1970s and this has been attributed to anthropogenic change by some scientists (Trenberth & Hoar 1996;1997), although others consider it too early to draw such conclusions from the available data (Harrison & Larkin 1997; Latif *et al* 1995). Overall, during the period subsequent to the mid-1970s, El-Niño events have predominated over the El-Niña cold events and a particularly severe event occurred in 1982-83 coupled with an unusually long series of events over the years 1990-95. The motion of the warm water pool prior to the events of 1982-82 and 1986-87 has been found to be very different to that preceding the event of 1991-93 (Yan *et al.* 1997). Attempts to elucidate the mechanisms leading to the onset of the 1991-1993 event have been confounded to some degree by the fact that it could not be entirely accounted for by existing explanations of such phenomena (Kessler & McPhaden 1995; Picaut *et al.* 1997). It is now believed that in the mid-1970s the North Pacific Ocean underwent what has been described as a clear phase transition (Zhang *et al.* 1998). This took the form of a subsurface temperature anomaly which extended its influence outside tropical regions, acting as an ocean "bridge". Whether this is entirely natural or partly driven by anthropogenic activity has not been resolved and the potential relationship with ENSO events is therefore unclear. While the likely behaviour of ENSO under conditions of climate change remains unknown, the phenomenon serves as a good indicator of the potential for other changes in ocean circulation induced by climate change to affect climate on a global basis.

The Southern Oscillation has a counterpart in the Atlantic known as the North Atlantic Oscillation (NAO) this is an index derived by comparing atmospheric pressures over the Azores with those over Iceland. The mid-latitude low and high pressure weather systems are the results of the heat exchange which takes place between the warm and cold water and air masses. This results in semi-permanent low pressure systems centred around the Aleutian Islands and Iceland which bring moist air northwards in the winter. In the Atlantic, the Iceland low pressure system is coupled with a high pressure system centred near the Azores (McCartney 1996). Analysis of Greenland ice cores has revealed large decadal climate variations over the

North Atlantic that can be related to the NAO in similar way to the ENSO events with respect to Pacific Basin climate. Pacific tropical sea surface temperature anomalies appear to be at least partially related to observed warming along the west coast of North America and Alaska and cooling in the Central Northern Pacific. This corresponded to a deepened, eastwards shifted Aleutian low pressure system during winter which persisted through most of the 1980s. The NAO influences atmospheric circulation patterns and hence rainfall and temperature in Europe, The Eastern United States, Arctic Canada, Greenland and the Mediterranean region. Over the last 130 years, the NAO has exhibited considerable variability at almost biennial and decadal time scales with some states persisting over several winters. The decadal variability seems to have become particularly pronounced since 1950. Since 1980, the NAO has tended to remain in a single extreme phase during winters (Hurrell 1995). This in turn has been correlated with observed winter warming in Europe and Eurasia and cooling in the Northwest Atlantic. Wetter than normal conditions over Northern Europe and Scandinavia are also linked to the NAO as are drier than normal wintertime conditions in Southern Europe and the Mediterranean. (Hurrel & van Loon 1997).

d) Sea Level Rise

The most obvious potential impact of climatic change is on sea level as a result of melting of ice masses and thermal expansion of sea water due to increased temperatures. Over the past 100 years, global sea level has risen by between 10 and 25cm and according to IPCC analyses, "it is likely that much of the rise in sea-level has been related to the concurrent rise in the global temperature." Although uncertainties remain, the IPCC suggests that warming and subsequent expansion of the oceans may account for 2 to 7 cm of this rise, while the melting of glaciers and sea-ice may have contributed a further 2 to 5 cm (IPCC 1996a). By the year 2050, the global mean sea level is expected to rise by between 7 and 39cm, with a mean projection of 20cm. By 2100 the rise is projected to be on average 49cm with a range of uncertainty of between 20 and 86cm. The uncertainties stem from imperfect historical data as well as lack of confidence in future projections (Gornitz 1995). Even so, the rate of increase is about two to four times the rate observed over the past 100 years. The increase would not be uniform, with the greatest increases expected in the North Atlantic while in some areas, such as the Ross Sea in Antarctica, sea levels would actually fall (IPCC 1996b) as a result of changes to the Antarctic ice sheet. Even assuming that carbon dioxide levels are ultimately stabilised at twice current atmospheric concentrations there would be a continuing commitment to further sea level rise (DETR 1997) as a result of inertia in the responses of the Greenland and Antarctic ice sheets to temperature changes. The models continue to be refined. Other estimates which take into account the effects of regional and seasonal temperature variations suggest that by 2100 sea level will be around 50cm higher than at present. 13.2 cm of the rise will be attributable to glacier melting and 7.6cm attributable to melting of the Greenland ice sheet (Gregory & Oerlemans 1998). Thermal expansion of the oceans accounts for the balance. This model predicts that in a warmer climate, ice will accumulate on the Antarctic ice sheet.

Although conditions will vary depending on local environments and the effects of other human impacts, it has been suggested that, for each centimetre of sea-level rise, beaches may erode a metre landward; for every 10-cm rise, saltwater wedges in estuaries and tidal rivers may advance a kilometre; and any sea-level rise will increase salinity in freshwater aquifers (NAS, 1987; Ray *et al.*, 1992). Coastal environments that are already under the greatest stress as a result of human activities are the ones which are most likely to be impacted by rising sea levels. For example, in Bangladesh, human activities in the Ganges-Bramaputra-Meghna river system may have rendered the area more susceptible to flooding, while in many Pacific Island states, mining and pollution of coral reefs – which act as natural barriers against marine erosion – are likely to exacerbate the impacts of rising seas in a region already at considerable risk because the land is low-lying. In other areas, the natural response of coastal ecosystems such as mangroves and salt marshes to sea-level rise (an upward and landward migration) is inhibited by flood embankments and other human construction (IPCC 1996b).

Without protective measures, land-loss as a result of rising sea levels is likely to be significant in some countries. Although some coastal nations, such as Argentina , Uruguay, Benin, Mauritius and the United States are predicted to lose only a few tenths of one per cent of their land area, the impacts elsewhere are likely to be far more extreme. Estimates range from 8.4% in Belize, through 12.5% in Kiribati and 17.5% in Bangladesh, to 80% of Majuro Atoll in the Marshall Islands. In addition to the obvious impacts on coastal populations of such land loss, there is concern over the effect on food production. Approximately 85% of the world's rice production takes place in South, Southeast and East Asia, of which 10% is located in areas that are considered to be vulnerable to sea-level rise, thereby endangering the food supply of more than 200 million people (IPCC 1996b).

Although the evidence is conflicting, it has been suggested that higher sea surface temperatures will lead to increasing numbers of tropical cyclones and increases in their intensity. The most recent modelling exercises suggest that for a sea surface warming of around 2.2°C , hurricanes would be between 5% and 12% more intense (Knutson *et al.* 1998) in the Northwest Pacific Basin. with a corresponding decrease of between 7 and 20 millibars in central surface pressure. If such predictions are applicable on a global basis then the impacts of storms and associated storm surges are likely to heighten the risk of storm and flood damage in coastal areas. Around 46 million people, most of them in less developed countries, are estimated to experience flooding as a result from storm surges under present conditions. That number will double if sea level rises 50cm, and almost triple if sea-level rise reaches 1 metre (Barse 1995; Hoozemans *et al*. 1993). One quantitative study concluded that in The Netherlands, the costs of avoiding damage related to an adverse 10% change in the direction and intensity of storms may be worse than those of a 60 cm rise in sea levels (Peerbolte *et al.*, 1991).

e) Polar Systems

i) Sea Ice and Glaciers

A consistent feature of many models of climate change is that they have predicted that the warming process will be intensified towards the polar regions. Hence the Arctic and Antarctic could well act as an area where the first effects of global change will first be unequivocally identified (Chapin *et al.* 1992; Bernes 1996; AMAP 1997; Walsh 1991). In addition, both areas have a significant role in establishing global circulation patterns which in turn influence both weather and climate. There is much evidence that thermal changes are forcing changes in the cryosphere. In the Antarctic, use of whaling records has led to the retrospective finding that the Antarctic summer sea ice edge has moved southwards by 2.8° of latitude between the mid 1950s and the early 1970s. This in turn suggests a decline in the area covered by ice of some 25% and implies that there may have been associated changes in Antarctic deep water formation and upon biological productivity. Both these processes affect atmospheric CO_2 concentrations and pose challenges for the verification of the global circulation models used to project climate change (de la Mare 1997). Other work has suggested that over the period 1978-1996, Antarctic sea ice increased by around 1.3% per decade (Cavalieri *et al.* 1997). Although these findings are apparently contradictory, they can be explained by normal variability in sea ice cover (Smith *et al.* 1998) and are consistent with observed temperature rises in the West Antarctic region (Smith *et al.* 1996) from the 1940s onwards. In addition, regional differences exist. Southern Ocean regions adjoining the South Atlantic, South Indian and Southwest Pacific Oceans show increasing trends in ice cover over the period 1978-1995, while regions adjoining the South East Pacific show decreasing trends in sea-ice coverage, particularly in the summer months. This may be evidence of developing climate differentials and may have important implications for the Southern Ocean heat budget as well as for ice related ecosystems (Stammerjohn & Smith 1997). Nonetheless, it is not clear whether these observations are due to natural variability or to anthropogenically forced change.

At the same time there have been well recorded retreats of ice shelves on the Antarctic Peninsula corresponding with atmospheric warming (Vaughan & Doake 1996) together with a rapid collapse of the Larsen A Ice Shelf in January 1995 (Doake *et al.* 1998). This has been followed by loss of a significant area of the Larsen B Shelf in early 1998 and this may ultimately result in destabilisation of the whole shelf (Scambos 1998). Ice shelves are important contributors to the production of Antarctic Bottom water hence their collapse is of potentially global importance. Not all shelves are likely to respond in the same way. The Filchner-Ronne ice shelf , for example, is located much more to the south than the Larsen shelves. The initial response to climate change is likely to be a reduced rate of basal melting from the shelf accompanied by reduction in the generation of high salinity shelf water and hence in the flow of ice shelf water. This would impact upon the production of Antarctic Bottom Water even though the shelf itself would effectively be reinforced (Nicholls 1997). Satellite images have also revealed decreases in velocity and increases in erosion of the West Antarctic Ice sheet overlying the continent which have taken place since 1963 (Bindschadler & Vornberger 1998).

There is some evidence that between 1978 and 1987 sea-ice extent in the Arctic decreased by around 2.1%-2.9% together with a decrease of open water of 3.5% (Gloerson & Campbell 1991; Cavalieri *et al.* 1997). In 1987, work carried out using a submarine found that the thickness of ice covering 300,000 square km of sea had fallen, decreasing in volume by some 15% (Wadhams 1990). Whether this actually represented an overall trend or normal variation has been called into question (McLaren *et al.* 1992), highlighting the need for additional data. Further work has indicated that over the period 1978-1994 the extent of Arctic sea-ice was reduced by around 4.6% while the sea-ice area contracted by almost 6% (Johannesson *et al.* 1996). More recent analysis from November 1978 through to December 1996 suggests that the areal extent of sea ice in the Arctic decreased by 2.9% ± 0.4% per decade (Cavalieri *et al.* 1997). In the Bering Sea, sea-ice extent has been reduced by about 5% over the last forty years (BSIS 1997) with the steepest decline taking place during the 1970's. Sophisticated statistical analytical techniques, considering the sea-ice extent has shown that the mean extent in summer over most sectors of the Arctic has fallen substantially when the period 1961-1975 is compared with 1976-1990 (Chapman & Walsh 1993) although no trend was seen for winter. Between 1979 and 1996, analysis of spring melt and autumn freeze dates suggests that there has been an increase of 8% per decade in the number of summer melt dates in the Arctic. Taken together, this is evidence the Arctic is undergoing a large scale warming (Smith 1998).

While the changes in themselves are not unequivocal indicators of a trend, they once again concur with the predictions from modelling exercises although these have tended to produce variable results (Hunt *et al.* 1995). Nonetheless, under doubling of atmospheric carbon dioxide concentrations a reduction in sea ice area of between 10-50% has been predicted by one model, together with a reduction in thickness, particularly in summer. When ice volumes are considered, a 50% decrease on average in winter could be followed by a virtual disappearance of sea-ice in the summer. Normal variability in snow cover coupled with likely regional differences in the rate of temperature increase mean that any such trends are likely to be difficult to identify. Even so, directly measured snow cover over the North American Great Plains suggests that there has been an increase over the last century, whereas in the Canadian prairies, snow cover has declined (IPCC 1996b). In Alaska (Walsh 1991), there is evidence that springtime disappearance of snow cover occurred some two weeks earlier in the 1980s than in the 1940s & 1950s. Data from lakes in central Canada show a warming of several degrees C and lengthening of the ice free season by several weeks over a twenty year period beginning in the late 1960s. The modelled predictions (IPCC 1996b) suggest that winter snowlines in the Arctic could move northwards by 5-10 degrees of latitude. Snowfall will tend to begin later and snowmelt will be earlier, extending the snow free season although in some regions, more frequent open water may actually lead to more snowfall downwind due to increased evaporation from the water surface.

A reduction of snow cover and sea-ice will affect one of the most highly important feedback systems in the Arctic Environment. By reducing the reflection of solar energy from the surface of the earth and sea it allows more energy to be absorbed. In turn this leads to an increased flow of heat from the surface to the atmosphere. The resulting temperature rise may bring about further loss of snow and ice, completing a positive feedback loop and reinforcing the original effect. One possible offset against this is the evaporation of water from the surface leading to increased cloud cover, but the relative importance of the two processes is impossible to assess at present (LAII 1997). The sea-ice also controls the transfer of heat from the relatively warm ocean to the cold atmosphere. Hence, ice reduction either in area or thickness could lead to increased heat transfer (IPCC 1996b) and interaction with winds (AMAP 1997). This would allow the air to pick up moisture and make the Arctic cloudier. This in turn would change regional weather patterns. The role of clouds, however, continues to be one of the major uncertainties in climate models.

Some of the observed changes in sea ice and ice shelves are mirrored in the retreat of glaciers in mountain and sub-polar regions. It has been estimated that global glacier area has been reduced by some 6000-8000 square kilometres between 1961 and 1990, around 10% of the total (Dyurgerov & Meier 1997a). The contribution of glaciers to sea level rise has increased steeply since the mid 1980s and this is also in agreement with the rise in global temperatures recorded, allowing for the lag-phase in glacial response (Dyurgerov & Meier 1997b). In the Arctic, there has been a consistent trend of glaciers to retreat although this varies regionally (IPCC 1996 b). It is predicted that the largest Gulf of Alaska glaciers should persist into the 22nd century. This must be set against the fact that Alaskan glaciers have suffered ice thickness decreases of 10m over the last forty years. A retreat of 15% appears to follow from each 1C rise in temperature, on average (BSIS 1997). In the projected sea level rise under changing climatic conditions some models predict that the largest contributions of glacial melt will derive from North West American and Central Asian glaciers. (Gregory & Oerlemans 1998).

The significance of melting glaciers is global. It is estimated that the contribution of glaciers worldwide to sea level rise was some 0.35 mm annually over the period 1890-1990 as an average. Between 1985 and 1993 the average rose to 0.6mm. Total sea level rise over this period was around 18cm. The 3.5cm contribution of glaciers is, therefore significant. The large glaciers of south east Alaska are particularly critical to the question of sea level rise in the future which could range between 13-94cm. The glaciers in this region are projected to account for around 43% of the total melt water volume. Estimates using more refined models which account for regional and seasonal temperature variations suggest that 13cm of sea level rise between 1990-2100 will be attributable to glacier melt (Gregory & Oerlemans 1998). The Greenland ice sheet will contribute a further 7.6cm with the Antarctic ice sheet, paradoxically, remaining stable or increasing in mass as a result of increased precipitation at higher temperatures. Increased precipitation could also cause some glaciers to increase in size, or remain stable, particularly at high latitudes and altitudes. The response will also partly depend upon weather patterns. It is known that high arctic glaciers also melt quickly following certain shifts in the semi-permanent weather systems which become established in the Arctic region (IPCC 1996b). Meltwater, by decreasing the salinity of Arctic seawater could also contribute to changes in the Ocean Conveyor system.

Glaciers show a delayed response to climate change of years to decades. In the case of a very large mass of ice such as the Greenland ice sheet, this lag phase will be longer. The ice sheet does not at present seem to be shrinking but it is predicted that both melting rate round the edges and accumulation rates through increased precipitation in the interior should increase. It is predicted that melting will predominate (IPCC 1996b; Greuell & Konzelmann 1994). The ice sheet can retain ice for up to a million years, making the impacts of climate change extremely hard to predict. Estimates made of the likely contribution of the Greenland ice sheet to sea level rise range from between 1-4cm by the year 2100 but could under some scenarios reach over one metre by the year 2200 (Titus & Narayanan 1994). If the Gulf Stream slows, then it is possible that Greenland would remain stable or even cool. Eventually it would be subject to the same temperature rises as the rest of the Arctic, melting being simply delayed.

ii) Atmospheric temperatures

The significance of climate change with respect to the Arctic Regions is twofold. On the one hand climatic change can be expected to change Arctic ecosystems. On the other, these changes themselves may well induce other changes of global significance. Modelling studies have suggested that the Arctic will warm by more than the global mean temperature rise (IPCC 1996a; Bernes 1996; AMAP 1997). There are still, however, difficulties in differentiating human impacts from natural variation. In addition temperature increases are unlikely to be uniform over the whole area. Models have given results which in some cases are not easily interpreted partly because the likely impact of some factors such as cloud cover are poorly simulated by mathematical models. Regional differences in temperature trends are also predicted to an extent by models. For example, models applied to the Nordic Arctic area have predicted a rate of temperature rise of 0.3C per decade for Iceland and the Faeroe Islands, but 0.45C per decade for eastern Finland and the northern extremes of Sweden. Summer warming is predicted to range between 0.25-0.3C whereas winter warming is likely to range over 0.35C-0.6C per decade for the same areas. Precipitation is likely to increase by between 3-6% per degree of warming (Johannesson *et al.*1995).

It appears that while surface air temperatures have increased by about 1.5C per decade over Central Siberia and Continental North America they have fallen by a similar amount in the Baffin Bay Area, for example. Indications of warming have been seen around the northern continental rims of central and western North America and Central Asia over the past century. For Eastern North America through the North Atlantic there is a cooling trend. Most of the warming in the western Arctic appears to have occurred in winter and spring, with less warming observed in the summer and autumn. Paleoclimate records for the last four hundred years indicate that terrestrial and aquatic Arctic environments have undergone some profound changes already. From 1840 to the mid-20th century the Arctic warmed to the highest temperatures in four centuries (Overpeck *et al.* 1997). In addition to glacial retreat, high latitude permafrost conditions have changed, plants have changed in distribution and aquatic plant communities have changed in composition. The Arctic shows high levels of natural variability and this, together with human forcing of the climate is expected to drive unprecedented change in the region. This will undoubtedly impact upon all aspects of Arctic ecosystems. Generally negative impacts are predicted for marine mammals such as polar bears, ringed seals and walrus which depend upon favourable ice conditions in order to forage. There may also be negative impacts upon migration patterns as a result of decreases in sea-ice extent (Tynan & McMaster 1997).

Other observations have been made which support the predictions of models. Satellite monitoring has shown that the lower atmosphere of the Arctic has become around 0.05C warmer per decade. This is a more pronounced change than the global average. The associated cooling in the upper atmosphere has been determined at -1.01C per decade, with the most rapid decreases over Russia. This change is the most pronounced over the whole planet. Borehole studies in Alaska of permafrost have also shown a temperature rise of between 2-4C over the last hundred years (AMAP 1997).

In Alaska, a warming trend of 0.75C per decade has been identified for the last three decades over land bordering the Bering Sea. Over the Eastern Bering Sea, temperatures have risen by 0.25C per decade (BSIS 1997). From these trends and modelling results, a further rise of 1-2C over 20 years and 4-5C over 100 years can be predicted. Also in Alaska, west of latitude 141, precipitation has increased by 30% between 1968 and 1990 (Anderson & Weller 1996). Elsewhere precipitation in high latitudes has increased by 15% over the last forty years AMAP (1997). On the North American tundra there is a tendency toward earlier snowmelt. The area of land with continuous snowcover over the winter to the south of the subarctic has retreated by about ten percent during the past 20 years. In the Mckenzie River Basin Study (Cohen 1997a & b), an area which includes parts of the Yukon and Northwest territories as well as northern British Columbia, Alberta and Saskatchewan, temperatures have increased by 1.5C this century.

Observed trends, therefore, are all broadly consistent with the results of modelling exercises. The full impacts of climate change on the Arctic, however, are extremely difficult to assess because of the intricate interactions between physical and biological factors. Overall, the hydrological cycle of the Arctic links atmospheric moisture transport, precipitation, river runoff, sea ice and ocean circulation in a single system (LAII 1997) and this in turn drives global ocean circulation. Hence, in considering potential impacts of climate change in the Arctic it is important to also consider how these changes may themselves cause wider impacts and indeed how they may actually reinforce the warming trend. It is predicted that practically all snow and ice features of the Arctic will be affected by continued warming of the earth's atmosphere (LAII 1997; IPCC1996b) as might be expected. The freeze-thaw line is likely to be shifted significantly northwards, and this will be accompanied by dramatic impacts.

f) Arctic Systems & Thermohaline Circulation

The Polar regions are recognised as potential indicators of widespread global change but changes in Arctic regions have been generally better characterised than for Antarctic regions. The importance of the Arctic in global processes is becoming increasingly well understood. Marine systems in the Arctic are characterised by very complex food webs but with only a very few key species directly connecting the various levels (Bernes 1996; AMAP 1997). The Central basins of the Arctic Ocean itself are considered to be one of the least productive major water bodies on the planet but recent work has suggested that they support a significant in situ food web (Pomeroy 1997). Even the pack ice has high productivity with small plants living within it and on it (Horner 1989). The ice organisms live in brine filled spaces between ice crytals where the tempearture may reach values far below the normal freezing point of water with salinity up to four times that of seawater. Light intensity is less than 10% of the incoming radiation. Physico-chemical parameters are subject to strong gradients within the ice. Nonetheless, biomass of algae and protozoans in the brine filled spaces often exceeds that found in open waters and mesocosm experiments have established communities in which at least two trophic levels were present. (Weisssenberger 1998). Intense production takes place elsewhere in Arctic waters. The biologically richest areas occur at the edge of the ice and in the shelf seas. Some of these shelf areas are among the most productive ecosystems in the world.

Perennial pack ice covers about 8-9 million square kilometres of the Arctic Seas, growing to a maximum of 15-16 million square kilometres in the period from November to February before melting sets in. Areas of open water (polynyas) in the ice field may be temporary or persistent and recurring (Maytham 1993). Persistent openings in the ice also act as a focus for intense marine production by allowing earlier penetration of light and facilitating mixing of nutrient poor with nutrient rich water (Schneider & Budeus 1997). The polynya can also be highly important to bird populations (Falk *et al.* 1997) and to wintering mammal populations (Bernes 1996; Stirling 1997). Ice free areas even in the central Arctic can comprise 1-2% of the total area, extending in summer and these areas can also play a significant role in heat exchange between atmosphere and water (IPCC 1996a) which influence local and to a lesser extent regional weather patterns (Schneider & Budeus 1997).

The seasonal accumulation and melting of sea ice results in high productivity at the edge of the ice and in the shelf seas and also contributes to the formation of a salt driven layering of the water. This keeps plants close to the sunlit surface. The ice edge bloom of microscopic plants and bacteria supports a complex food web (Vezina *et al.* 1997; Herman 1989) which follows the retreating ice edge. In some areas, freshwater inflow from north flowing rivers creates highly productive estuarine zones. In the Barents Sea, cooling and ice formation result in a complete turnover of the water column and in other northern sea areas surface water is enriched annually with nutrients from deeper waters. The marine boundary of the Arctic is formed when the water of the Arctic Ocean, cooled and diluted by melting ice meets the warmer saltier water of the southern oceans. The position of this Polar Front is relatively stable from year to year (Bernes 1996; AMAP 1997). To the south of this front nutrients can be brought to the surface by summer storms, while at the front itself, considerable vertical mixing can take place throughout the year. Despite the short growing season, the supply of nutrients forms the basis of a rich fauna supporting highly profitable fisheries.

The significance of the physical and chemical processes taking place in the Arctic region extend far outside it. The polar area has been described as a "refrigerator in the equator to pole transport of energy" (AMAP 1997). As well as being an area where nutrients are recycled and released into the water, the Polar Front region in the North Atlantic plays a fundamental role in the driving of ocean currents. Broadly, the Arctic regions transform and then export Atlantic Water, originally of lower density as high density intermediate and deep waters. Because of high precipitation it also forms and exports surface water of lower salinity and density. At the front near Greenland, Iceland and the Labrador Sea, warm salty water from the North Atlantic is cooled by Arctic waters and by intense heat loss to the atmosphere, becomes more dense and sinks to deeper layers of the ocean. Salt rejected as sea ice forms also increases the density and contributes to the process. Although a slow process, this sinking takes place over a wide area and each winter several million cubic kilometres of water sink and begin moving slowly south along the bottom of the Atlantic Ocean. It is known as thermohaline circulation because it is driven in part by temperature and partly by salinity differences.

The dense, cooled water becomes part of what is termed the Ocean Conveyor and the water eventually returns to the surface in the Indian and Pacific Oceans. As warm water returns to the Atlantic, the current moves polewards as the Gulf Stream and North Atlantic Drift (Bernes 1996; AMAP 1997; Weaver 1993). The circulation processes within the Arctic Ocean are extremely complex, with water exchange processes taking place with both the Pacific and Atlantic Oceans and the circulation as a whole being affected by freshwater inputs from rivers and precipitation (Meincke *et al* 1997; Malmberg & Jonsson 1997). There is some evidence that changes in the Odden, an ice tongue which forms in winter to the east of Greenland may be responsible for changes observed in the renewal of the deep waters of the Greenland Sea by convective processes in recent years (Meincke *et al.* 1997, Wadhams, 1996). This in turn could impact the formation of North Atlantic Deep Water. Global circulation of the water in the ocean currents takes place over time scales of centuries. In addition, the formation of deep-water also dissolves carbon dioxide from the atmosphere and effectively removes. This is of significance in the global cycling of carbon. The Arctic region, therefore, plays a fundamental role in ocean circulation patterns, which in turn determine climate patterns over the rest of the globe.

In one scenario (AMAP 1997), barometric pressure could be reduced in the Arctic. This could give rise to more cyclonic storm systems particularly in winter. Changes in wind patterns could influence temperature and humidity, and hence the formation of sea-ice and the circulation of water in ocean currents. A specific concern is the likely behaviour of the thermohaline circulation. Increased precipitation and melting of ice could make the upper layers of the Arctic less saline. This could change the formation of deep water at the polar front (Rahmstorf 1995). A change in the thermohaline circulation, therefore, could shut down the Ocean Conveyor. Alternatively, based on evidence from sediment cores, the site where deep water forms could shift southwards. This would end the influence of warm waters from the Gulf Stream and North Atlantic Drift along the coast of the Western Atlantic. Paradoxically, this could lead to areas of Europe and Scandinavia cooling markedly. Thereafter, these regions would experience the global temperature rise, but starting from a point where the warming effects of Atlantic waters were absent. This possibility is supported by geological evidence of previous phenomena following large inputs of low salinity water into the Arctic seas (Bernes 1996; Lehman & Keigwin 1992).

Potential influences of climate change upon the thermohaline circulation have been the focus of considerable study. The thermohaline circulation may react sensitively to disturbances, oscillate of its own accord, or even break down temporarily (Rahmstorf *et al.* 1996). The system appears to have been stable over the past 10,000 years, but nonetheless the circulation is in a delicately balanced state with a number of factors contributing to this equilibrium including the flow regime of North Pacific water into the northern North Atlantic (Shaffer & Bendtsen 1994). One of the more important factors appears to be freshwater flux. As freshwater input to the Arctic increases as a result of increased glacial melt, riverine inputs and precipitation, it may trigger instability (Rahmstorf 1995). As yet, mathematical models have been unable to resolve the uncertainties. The palaeorecord suggests that climatic shifts are strongly related to deep water reorganisations. The last interglacial period 118,000-127,000 yeas ago was preceded by, and ended with, abrupt changes in deep water flow. At the end of the period, bottom waters had a much higher proportion of water sourced from southern areas (Adkins *et al.* 1997) implying a breakdown in North Atlantic Deepwater Formation. The palaeorecord also suggests that cold glacial periods in the past were associated with weakened production of deepwater in the North Atlantic (Oppo *et al.* 1998), helping to explain millennial scale climate changes 500,000 to 340,000 years ago.

Evidence has also been found of a cooling event 12,000 years before present which was possibly associated with a temporary shutdown of deepwater formation in the North Atlantic. This event also had impacts in the Pacific Ocean above 30ºN as well as the marked impacts in the North Atlantic (Mikolajewicz *et al.* 1997). Similarly, around 17,200 years before present, discharge of icebergs and meltwater is thought to have shut down North Atlantic thermohaline circulation (McCabe & Clark 1998). The palaeorecord shows that between 10,000 and 80,000 years before present there were millennial oscillations of several degrees in sea surface temperature in the North Atlantic accompanied by abrupt temperature shifts on decadal scales (Benson *et al.* 1997). Interaction between warm ocean surface currents and the Scandinavian ice sheets is thought to be the driving mechanism. The millennial cycles, which produce cumulative cooling, appear to end with a massive discharge of icebergs into the North Atlantic (Heinrich events) and an abrupt shift to a warmer climate in the North Atlantic, although the Heinrich events may act to delay or reverse deglaciation initially (McCabe & Clark 1998). During the cool periods, the thermohaline circulation was progressively weakened due to iceberg surges, moving the polar front to the east and restricting the northward flow of warm water (IPCC 1996b; Hughes 1996).

In addition to the possibility that thermohaline circulation could be stopped, the site at which it occurs could also shift. The results of modelling exercises suggest that changes in ocean circulation at the time of the last glacial maximum around 21,000 years before present involved the formation of North Atlantic Deepwater at a more southerly site at 50ºN than at present. The sea-ice margin is thought to have moved southwards in winter (Ganopolski 1998). Movement of convection sites southwards rather than a complete shutdown of deepwater formation is consistent with isotope studies which suggest that thermohaline circulation at the last glacial maximum was similar to present day magnitudes (Yu *et al.* 1996). Modelling exercises have shown that thermohaline circulation is not only potentially sensitive to the final atmospheric CO_2 concentration, but also to the rate of change of concentration (Stocker & Schmittner 1997). At current rates of increase, to 750ppmv over the next hundred years, then thermohaline circulation is predicted to permanently shut down, whereas if the final CO_2 concentration is attained more slowly, circulation is simply slowed down . At the extreme this could lead to the Antarctic Circumpolar front remaining as the sole source of deepwater. The modelled impacts of increased atmospheric concentrations of CO_2 upon the thermohaline circulation are shown in Fig. 7

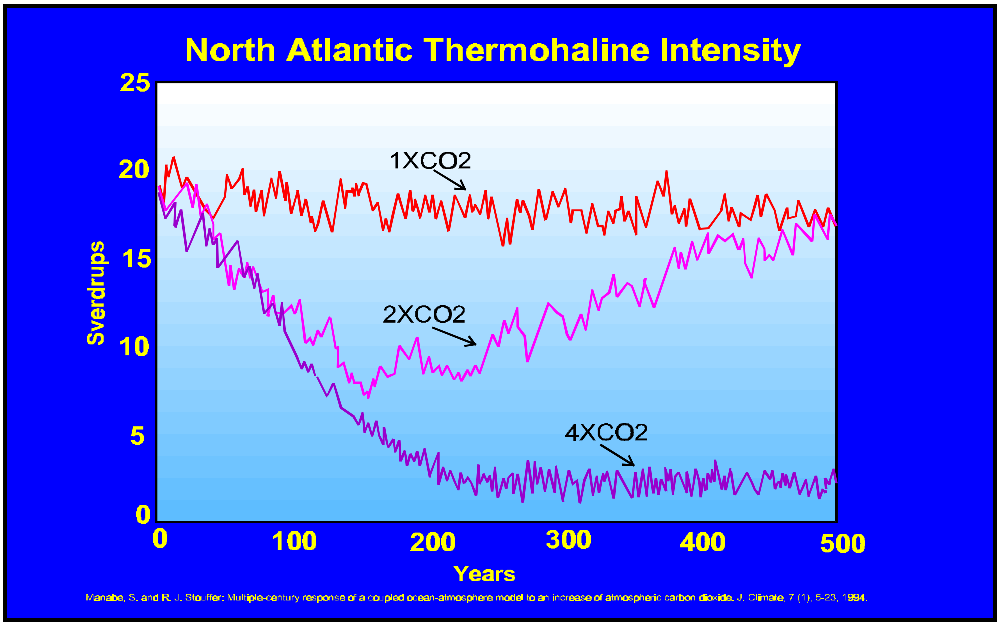


Figure 7: The possible behaviour of the thermohaline circulation under different atmospheric CO_2 emission scenarios. With a doubling of CO_2 concentration, the circulation is predicted to decline sharply over time and recover gradually over 500 years. At quadrupled CO_2 concentrations, the thermohaline circulation could shut down completely. Recent work has suggested that not only is the final atmospheric concentration important, but also the rate at which it increases.

An understanding of the likely behaviour of the thermohaline circulation under global warming is important because the Gulf Stream/North Atlantic Drift currents which transport warm surface water from subtropical regions northwards are responsible for the mild climate which currently prevails over the west of the European land mass. Changes in convection in the Greenland Sea have been linked with unusual southern penetration of low salinity water into the North Atlantic. This in turn can affect the distribution and reproduction of living marine resources (Malmberg & Jonsson 1997). If Arctic currents change or weaken then it is inevitable that climate in the region would be affected. A simple modelling exercise examining impacts on the Gulf Stream has shown that temperatures would become lower with an increase in the number of winters with temperatures below the present averages, while in other seasons temperature would become more variable and cold days would become colder. In short, the temperature range would increase (Klein-Tank & Konnen 1997). In addition to the profound effects that a temperature drop of up to 5C would have upon regional climate and weather in Europe and Scandinavia it would also change the transport of atmospheric carbon dioxide from the atmosphere into the oceans. If atmospheric levels of carbon dioxide were increased by this a positive reinforcement of rising temperatures could occur. Currently it is estimated that CO_2 equivalent to 5-10% of current industrial emissions are conveyed to the deep interior of the Atlantic via thermohaline circulation, although high latitude “sinks” may be balanced by net annual CO_2 releases in tropical Atlantic waters (Takahashi *et al.* 1995).

The thermohaline circulation also drives global water circulation. Hence, in addition to disrupting regional warm water circulation in the North Atlantic wider disruption would result. A key question relates to the likely speed with which any changes are likely to occur. It is possible that rather than a slow decline in circulation, changes could be relatively rapid and could occur within a relatively short time frame under current scenarios of atmospheric CO_2 increase.

g) Mitigation of Potential Impacts

i) The Carbon Logic

The question at present is not one of how climate change can be prevented but rather one of how the impacts can be kept within certain limits. While the provisions of the Kyoto (1997) agreement go some way to promote reductions in the emissions of greenhouse gases it is clear that these are unlikely to prove sufficient. Probably the most rational proposals centre around the restriction of fossil fuel use. By relating the quantity of fossil fuel that leads to a given emission of CO_2 it is possible to calculate the quantities which may be consumed if atmospheric CO_2 is to be kept within certain limits. In turn these limits are defined by the need to keep the rate of climate change constrained within limits that allow ecosystems to adapt to changes in sea level and temperature to the maximum extent possible. Such a scenario has been explored in depth (Hare 1997). Axiomatic to this approach is action to: restrict the long term global temperature increase to less than 1ºC above the pre-industrial average, bring the rate of average global temperature rise to below 0.1ºC per decade within the next few decades, limit the rise in sea level to 20cm above 1990 levels and restrict the rate of rise to below 20mm per decade. The rationale behind these restrictions is that they would allow many compromised ecosystems to adapt to climate change and result in a reduction in the overall risks of catastrophic change.

When these constraints for “manageable” levels of impact are logically (The Carbon Logic) translated into the amount of fossil fuels that may be used , a figure of 225 billion tonnes of carbon is arrived at which represents around 25% of current reserves. There is a need, therefore, to place greater emphasis on the development of renewable energy resources. Plans for new exploration and for the development of unconventional hydrocarbon reserves should be cancelled. In addition to these measures, policies which go far beyond the Kyoto Agreement in reducing CO_2 emissions will need to be emplaced.

In contrast to this rational, pragmatic approach, other less well rationalised, though superficially attractive approaches have emerged. These appear to be driven by the search for a moderate strategic response given the inherent uncertainties in climate change prediction. Equally possibly, such schemes could weaken international resolve for a substantive preventive strategy. In addition, because the solution formulated needs to apply globally, emerging proposals constitute a global environmental engineering solution. Global scale environmental engineering is an issue which requires great caution (Marland 1996). The idea behind such proposals is that they provide a means whereby the rate of carbon dioxide assimilation by the oceans can be increased above natural levels. In this way, it is reasoned, it may be possible to avoid the sharp peak in atmospheric CO2 concentrations predicted for around 2100 under a "business as usual" scenario (GESAMP 1997b).

ii) Ocean Fertilisation

One global engineering proposal which has been theoretically and practically explored is fertilisation of the open ocean. Field experiments have already been conducted in the Equatorial Pacific Ocean, based on the hypothesis that limitation of the micro-nutrient iron acts to limit phytoplankton growth. If phytoplankton growth can be increased, it is reasoned, then greater amounts of carbon can be fixed in oceanic environments (Ormerod & Angel 1998) thus reducing atmospheric concentrations. The Ironex experiment series showed that introduction of iron over an area of about 64 square kilometers as a single injection caused a direct biological response and a small decrease in the partial pressure of carbon dioxide in the fertilised area. A second experiment in the series used multiple introductions of iron. The results showed that both particulate and dissolved organic carbon increased as did oxygen, while CO_2 decreased. There was a differential response of the planktonic organisms resulting in an altered community structure. Accordingly, this change could propagate throughout the food web if iron was added on a regular basis. In fact, the logistic difficulties in using this as a method of increasing carbon sequestration by the oceans are formidable. The impact of added iron in the Ironex experiments lasted less than a week and the maximum benefits of such an approach would be restricted to the southern ocean. If iron additions were not maintained beyond a few decades, models suggest that there would be a premature return to atmosphere of some of the sequestered CO_2 . Perhaps recognising the sensitivity of the experiments the researchers noted that such experiments were designed to investigate limits to productivity in high nitrate/low chlorophyll areas and were "not intended as preliminary steps to climate manipulation" (Coale *et al.* 1994).

Proposals have also been made to fertilise the ocean with macronutrients such as phosphate and nitrate. In general, these schemes have been directed at increasing the yield of fish farms and of macroalgae. The aim is to simulate the high nutrient conditions found in upwelling waters and thereby increase productivity of the waters. The harvesting of resources is not entirely compatible with the permanent fixation of carbon (Ormerod & Angel 1998). Major environmental concerns can be identified in connection with ocean fertilisation schemes. The impacts of eutrophication in fertilised waters could result in undesirable whole ecosystem impacts including shifts in species composition and appearance of nuisance species. Changes in marine communities could have unanticipated consequences. Increased microbial activity could result in oxygen deficiencies in deeper waters, while there could also be an increased production of N_2O, itself an important greenhouse gas.

Overall, ocean fertilisation is an extremely high risk strategy for the mitigation of climate change in terms of the potential ecosystem impacts. In addition, the use of such a strategy could encourage the increased dumping of industrial wastes at sea. Residues from mineral processing such as the manufacture of titanium dioxide generate substantial volumes of iron rich wastes, but these also contain other contaminating elements of ecological significance. Similarly, phosphate rich wastes are generated by the processing of phosphate rock while nitrogen rich wastes arise from manufacture of nitrogen chemistry. Both can contain substantial quantities of ecologically significant contaminants. The dumping of such wastes is regulated under the terms of the London Convention.

iii) Carbon Dioxide Dumping

a) Aquifer Injection

Proposals to directly introduce carbon dioxide into the deep ocean or into sub-seabed formations are at an advanced stage. Following an extensive theoretical evaluation of the practical aspects of such a scheme (Wong & Hirai 1997; Ormerod 1996 a & b; Ormerod 1997; Ormerod & Angel 1996) proof of concept experiments are now planned or in operation (Hanisch 1998). These plans fall into two categories. The first stems from the oil industry practice of reinjecting gas separated from oil at the well-head into oil formations. A large scale seabed CO_2 disposal project was initiated in late 1996 by the Norwegian State Oil Company (Statoil). Around 1 million tonnes of CO_2 annually are being pumped into a porous slat water aquifer some 32,000 square kilometres in extent (IEA 1998). This Utsira sandstone formation lies some 1000 metres below the sea floor. Monitoring activities have been emplaced to follow the movement of the CO_2 through the formation with time by means of seismic data acquisition programmes. A similar programme is being considered by a consortium involving Exxon and the Indonesian State Oil Company Pertamina in the Natuna offshore gas field in the South China Sea (Hanisch 1998). This field is on of the largest in world and if the project goes ahead up to 100 million tonnes of CO_2 would be disposed of in this way annually into a sub-seabed aquifer. This practice clearly differs from standard industry procedures in that gas is being injected into formations other than those from which the oil has been extracted. In time it is expected that CO_2 from land based processing activities will be disposed of into the Norwegian formation as part of Statoil's commitment to reduce emissions by 30% over a ten year period. Ultimately, it is anticipated that emissions from power generation plants could be captured and disposed of in this way. The degree of containment achievable by these techniques is central to their long term viability. Although modelling studies suggest that storage in aquifers could be secure for at least 10,000 years, this is by no means certain. Slow escapes of CO_2 could significantly modify ecosystems over a large area by changing the physico-chemical characteristics of the overlying water.

b) Direct Ocean Disposal

The second scheme proposed for CO_2 disposal into the oceans involves the one of three methods of CO_2 introduction: introduction by pipeline into deepwater followed by dissolution, dispersion following discharge as dry ice blocks or liquid CO_2 from a pipe towed by a ship or isolation resulting from the formation of a lake of liquid carbon dioxide in the deep ocean. The first direct disposal test is due to take place through a fixed pipe into deepwater at a site along the Kona coast of Hawaii in the year 2000 This is a collaborative project involving US, Japanese and Norwegian researchers (Hanisch 1998). Other programmes are under development for injection into deep Norwegian Fjord systems. The major short term impact is perceived to be the changing of seawater pH close to the discharge. A review of these proposals (GESAMP 1997b) notes that for them to comprise a viable mitigation option for climate change, a large proportion of the future CO_2 arisings from fossil fuled power plants would need to be captured and disposed of in this way. There are also substantial scientific uncertainties which have been identified. These include the period over which injected CO_2 can be expected to remain in the ocean, and likely impacts upon biological, geochemical and physical processes.

Theoretical studies show that a considerable proportion of any CO_2 introduced into the deep ocean would return to the atmosphere over a period of several hundred years. Hence, benefits could be reduced in the long term. The time scale over which CO_2 would be emitted would depend upon the method used to dump the gas and the depth of water. In the short term, dumping of carbon dioxide in volumes equivalent to those produced by a single coal fired power station would reduce the pH of seawater to a value of 7 or less (as against a normal value of around 8.2) for a distance of tens of kilometres from the discharge point, with small isolated bodies of water showing such effects hundreds of kilometres from the discharge point. Sustained pH values below 6.5 are lethal to many coastal marine organisms and oceanic and deep benthic organisms are expected to be more sensitive. Interference with biological processes around shelf breaks could have considerable commercial impacts upon marine living resources, while strata of low pH water could act as barriers to vertical migration patterns. A key consideration is the quantity of carbonate ion available to neutralise introduced CO_2 such that no large changes of chemical balance occur (carbonate buff ering). This is an issue whether carbon dioxide is introduced directly or allowed to exchange naturally across the ocean/atmosphere interface (GESAMP 1997b) and the possibility exists that carbonate cycling could be disrupted. The likely carbonate interactions are also critical in determining residence time of introduced carbon dioxide in the oceans, and the proportion likely to be returned to the atmosphere.

In general, ocean disposal of CO_2 represents an irrevocable solution. If unanticipated effects occur, recovery of emplaced or dispersed material, as with all material dumped at sea, is likely to prove impossible. The long term benefits are also in doubt, raising intergenerational liability issues. In addition, this option could misguidedly divert attention away from other mitigation options. Accordingly, the prohibition of ocean disposal under the terms of the London Convention should be maintained and more weight should be given to exploring the mitigation options outlined by IPCC (1996b) with the exception of nuclear power. Nuclear power also represents a high risk mitigation option and is further made unacceptable by the unresolved issue of nuclear wastes. The options include the development of alternative energy sources, and modifications of current land use practices. Ultimately, it should be recognised that reducing the current dependence upon fossil fuels and adopting the “Carbon Logic” constitutes the only robust way forward.

9.2 Stratospheric Ozone Depletion

Every spring for over the last 20 years a sizeable portion, up to 60% of the stratospheric ozone over Antarctica has disappeared. Substantial depletion of the ozone layer over the Arctic is now being recorded as are varying levels of ozone depletion at temperate and tropical latitudes (Stolarski *et al.*, 1992; Gleason *et al.*, 1993; Fioletov *et al.* 1997). Continued declines in stratospheric ozone will occur for at least the next few years and recovery will be slow over much of the next century (Madronich *et al.*, 1995). UNEP's Atmospheric Science Panel, in the 1994 assessment of the environmental effects of ozone depletion, predicted that global UV levels should peak around the turn of the century and gradually decline to pre-ozone depletion levels over the subsequent 50 years. These predictions were made on the assumption of a "fairly optimistic scenario" including full compliance to international agreements, that no ozone-depleting chemicals were overlooked, and that there will be no new threats to the ozone layer (van der Leun *et al.*, 1995).

There are, however, some differences among the predictions of different models, and between predictions and past trends(Madronich *et al.*, 1995; Wayne *et al.*, 1995). Furthermore, Wayne *et al.* (1995) warn that incorrect predictions are likely to occur if as yet unknown processes have been overlooked; they cite, as an obvious past example of this, the gross underestimation during the 1970s of the potential impact of CFCs on the ozone layer. Meanwhile, processes that influence stratospheric ozone concentrations are not completely understood, such as volcanic eruptions (Madronich, 1995) or the possible role of iodine-catalysed ozone depletion (Wayne *et al.*, 1995). There is also much that is unclear concerning the origins and action of atmospheric methyl bromide, believed to be the main source of stratospheric bromine. Bromine, approximately 50 times more effective than chlorine in destroying stratospheric ozone (though having a much shorter atmospheric lifetime), is believed to be responsible for at least 20% of the Antarctic ozone hole (Mano & Andreae, 1994; Butler, 1995). Such uncertainties have led some, e.g. Madronich (1995), to question the validity of extrapolating from past ozone trends to what may occur in the next century. As an example, it is possible that increased stratospheric ozone loss may result from increasing levels of greenhouse gases in the atmosphere as a result of lowered temperatures in the stratosphere. This effect may also act to delay any recovery (Shindell *et al.* 1998).

Though some scientists and politicians remain skeptical about the destructive effect of chlorofluorocarbons (CFCs) on the Earth's ozone layer (Kiernan, 1995a; Brune, 1996) there is overwhelming evidence (e.g. Russell *et al.*, 1996) (and hence widespread scientific agreement) that humankind's production and use of various chlorinated and brominated chemicals are ultimately responsible for the recent and continuing depletion of ozone in the lower stratosphere; and that this depletion has resulted in increased levels of UV radiation incident at the earth's surface. In addition, there is clear evidence that UV-B radiation is detrimental to various marine species in the upper layers of the ocean and that it reduces primary production. However, there is tremendous uncertainty of the significance of this on the scale of communities and ecosystems (Bidigare, 1989; Hardy & Gucinski, 1989; Smith *et al.*, 1992; Hader, 1993; Neale *et al.*, 1994; Hader *et al.*, 1995) and, by extension, to populations of higher level predators -- including the cetaceans. Regardless, the potential impact of UV-B on marine ecosystems is considerable as the oceans cover approximately 70% of the earth's surface and because of the role of the euphotic zone in overall oceanic and atmospheric processes.

Prior to 1978, springtime ozone levels for the Antarctica region averaged approximately 300 Dobson Units (DU) or higher. By 1987, minimum ozone levels during spring had plummeted to under 125 DU covering nearly 50% of the continent. The edges of the 250 DU zone reached the southern tip of South America beyond the northern boundary of the Polar Front (Karentz, 1991). This meant that the entire Southern Ocean was now experiencing increased levels of UV-B radiation. In 1992, the ozone hole itself (212 DU and less) extended over the tip of South America and covered approximately 23 million sq. km (NOAA, 1992), while in 1993 values under 100 DU were recorded for the first time. In 1994, ozone layer thickness at the southern tip of South America was recorded at 151 DU (WMO, 1994). Recently, (Roscoe *et al.* 1997) observations have shown that Antarctic ozone depletion begins in midwinter at the sunlit edges of the polar vortex, rather than in its centre at the austral spring.

The regular occurence of a severe Antarctic ozone hole represents a special case in that this region is experiencing seasonal ultraviolet radiation levels far in excess of those before the reduction in stratospheric ozone concentrations. The ozone depletion cycle in the Antarctic region begins as organisms emerge from a dark winter, presumably a time when vulnerability to UV exposure is at its highest (Karentz, 1994b). Ozone depletion events over the Northern Hemisphere, though not as extreme as the Antarctic ozone hole, are substantial. In 1993, between 45 N and 65 N, there was record ozone depletion with seasonal means some 13% below the norm (Bojkov, 1993). Values were somewhat higher in early 1994 but near-record lows, with ranges from 8.1% to 19.4% below normal, were reported during early 1995 (AES, 1995). More recently, the March 1997 average total ozone concentration in the Arctic was found to be some 30% lower than average concentrations measured between 1979 and 1982 (Newman *et al.* 1997). Ozone levels in the High Arctic have fallen from around 450DU to around 300DU, the lowest levels correlated with extremely low stratospheric temperatures. In turn these low temperatures may be caused by anthropogenically forced climate change (Fioletov *et al.* 1997).

Even without the loss of stratospheric ozone, many marine organisms are exposed and stressed by UV-B on a daily basis. Ambient levels at any latitude are potentially detrimental and there is much evidence to suggest that normal levels of solar UV are an important ecological factor in oceanic processes (Calkins & Thordardottir, 1980). That UV-B has been a global environmental stress for many life forms is made evident by the many protective and recovery mechanisms among diverse taxonomic groups (Karentz *et al.*, 1994).

a) Effects of UV-B Radiation on Planktonic Organisms

The intensity and duration of UV-B exposure of marine organisms depend on numerous factors, including weather conditions (Gautier *et al.*, 1994), the rate and depth of vertical mixing (Helbling *et al.*, 1994), the areal extent and thickness of sea ice (Perovich, 1993), levels of tropospheric aerosols (Vogelmann *et al.*, 1992) and concentrations of tropospheric ozone and other gaseous pollutants (Madronich *et al.*, 1995). Penetration is markedly reduced by the presence of UV absorbing dissolved and particulate matter in the water column (Mitchell *et al.*, 1989); UV-B wavelengths can penetrate to a depth of 65 m in the clear waters of the Southern Ocean (Hader, 1993). In addition, incident UV-B varies with season and time of day. Short-term and long-term variability and interactions among these environmental modifiers confound any accurate determination of exposure levels for organisms in their natural habitats (Karentz, 1994c). Calculations of exposure are further complicated for those species, such as certain motile phytoplankton and krill, which are able to influence their position in the water column.

The final impacts of UV-B on marine organisms depend on natural biological defense systems which can also vary among species. Harmful effects can be countered by protection mechanisms such as UV absorbing compounds (Marchant *et al.*, 1991; Vernet *et al.*, 1994) and recovery mechanisms including DNA repair pathways (Karentz *et al.*, 1991) and the ability to resynthesize damaged proteins and/or pigments (Strid & Anderson, 1994). Phytoplankton can also acclimate to UV by increasing their capacity to correct UV-induced damage (Strid & Anderson, 1994; Riegger & Robinson 1997). However, the existence of such mechanisms does not mean that detrimental effects are thus avoided; there is an associated metabollic cost to protection and recovery mechanisms. The energy reserves required to synthesize UV-absorbing compounds or repair UV-damaged photosynthetic systems may have a significant effect on growth rates or on the long-term survival and fitness of cells (Cullen & Neale, 1994).

i) Bacterioplankton

The viability of bacterioplankton has been shown to be greatly reduced with exposure to UV radiation, primarily in the upper 10 m of the water column, with UV-A exerting far greater effects than UV-B (Sieracki & Sieburth, 1986; Helbling *et al.*, 1995). Nevertheless, ambient UV-B has been found to inhibit bacterial DNA replication, protein synthesis and degredative enzyme activities by as much as 40%. Furthermore, bacterioplankton appear not to have any capacity to protect themselves from UV-B suggesting that there should be considerable suppression of bacterial growth due to UV-B penetration of oceanic surface waters (Herndl *et al.*, 1993).

ii) Phytoplankton

Though the phytoplankton are limited to the euphotic zone (in that they require solar energy for their growth and metabolism) most species are not adapted to unfiltered solar radiation. Highest productivities and abundances are, therefore, generally not found at or close to the sea surface. Though UV-B is more deleterious than UV-A, the majority of natural phosynthetic inhibition is caused by UV-A (Buhlmann *et al.*, 1987) because of its greater fluxes in the ocean. There have been numerous, extensive reviews on the variety of effects of UV-B exposure on phytoplankton (e.g. Vincent & Roy, 1993; Karentz *et al.*, 1994; Smith & Cullen, 1995). Though there is a wide variation in UV-B tolerance among phytoplanktonic species and taxonomic groups, the effects can include reduction in growth and cell division rates, reduction in cell survival and fitness, damage to DNA, enzymes, and membranes, inhibition of photosynthetic rates, destruction of photosynthetic pigments, inhibition of motility and mechanisms for orientation and reduction in nitrogen assimilation. Biological effects have been detected at depths reaching 20-30 m (Karentz & Lutze, 1990; Smith et al, 1992). Work carried out on freshwater plankton has shown that exposure to UV-

B can cause differences in the allocation of fixed carbon between lipid, protein and polysaccharide in a species-specific manner (Arts & Rai 1997). Changes in lipid composition, cell size and cell wall properties can subsequently impact upon organisms which graze on algae (Hessen *et al.* 1997).

That increased UV-B can effect natural populations and primary production has been conclusively shown by Smith *et al.* (1992) in studies on phytoplankton productivity in the marginal ice zone (MIZ) in the Southern Ocean, both within and outside the area of the ozone hole. The MIZ was selected as the region of study due to it being a spring phenomena (as is the ozone hole), because the water column dynamics that promote enhanced productivity in the MIZ would also promote maximal phytoplankton exposure to UV-B, and because of the importance of MIZ productivity to the overall production of the Southern Ocean. Their results showed a minimum 6-12% reduction in MIZ primary production associated with ozone layer depletion. In addition, Smith *et al.* (1992), estimate a yearly production loss for the MIZ at approximately 2-4% though they caution against inferring longer term ecological consequences from short-term observations. Although these results need to be viewed within the context of a presumed natural variability in primary production in the MIZ of +/- 25%, the consequences of a consistent underlying shortfall in production could be substantial.

iii) Zooplankton and Vertebrates

Numerous studies have shown that increased UV-B can cause death, decreased reproductive capacity, reduced survival and impaired larval development in various zooplankton and fish species (Worrest, 1986;Hader *et al.*, 1995). The larvae of many fish and even some benthic invertebrates, have planktonic phases during which they remain close to, or even at, the surface. Exposure to ambient UV-B increased the frequency of underdeveloped and malformed Antarctic sea urchin embryos and caused lethal damage to early life cycle stages Karentz, 1994c). Similarly, a 16% ozone reduction was found to cause increases in mortality of 50%, 82%, and 100% at the 0.5 m depth for anchovy larvae of ages 2, 4, and 12 days respectively (Hunter *et al.*, 1979).

iv) Higher organisms

The impacts of enhanced UV-B exposure on other organisms appear to have been poorly investigated (Karentz, 1994a). Direct impacts on the skin may be limited by pigmentation in most organisms, although consequences of prolonged exposure to elevated levels cannot be ruled out. Furthermore, at least some mammalian species, including cetaceans, can transmit UV radiation as low as 300 nm (i.e. well into the UV-B spectrum) to the retina, including wavelengths which are known to regulate neuroendocrine physiology and suppress pineal melatonin (Brainard *et al.*, 1991; 1994). Melatonin is a ubiquitously acting hormone that adjusts the entire physiology of an organism on a seasonal and, possibly, daily basis, including thyroid physiology, reproductive cycles, skin pigmentation, fat metabolism, and body temperature regulation. The immune system, circadian rhythm and hormone-responsive tumour growth may also be linked to melatonin levels (Reiter, 1991).

b) Trophic Level Interactions and Ecosystem Impacts

As the phytoplankton are the basis of marine food webs any loss in overall biomass or changes in species composition could cause the reduction in biomass at higher trophic levels. Direct effects of UV-B on zooplankton and on fish eggs and larvae could also cause a significant impact at higher trophic levels (see Hardy & Gucinski, 1989; Voytek, 1990; Hader & Worrest, 1991; Hader, 1993).

Given the wide range in UV-B sensitivity among phytoplanktonic species and taxonomic groups (Karentz *et al.*, 1991; Davidson & Marchant, 1994; Karentz & Spero, 1995), UV-B could play an important role in species selection and in lower food web processes. Such selection, conceptually similar to that relating to shifts in nutrient availability, could ultimately result in fundamental changes in community structure at higher trophic levels with far-reaching implications (Worrest *et al.*, 1978, Smith & Cullen, 1995). The precise ramifications of what might at first appear quite subtle changes in rates of, or organisms responsible for, primary production, are practically impossible to predict.

Uncertainties surrounding trophic level interactions contribute further to these difficulties. For example, Bothwell *et al.* (1994) noted that the differential sensitivity to UV-B between algae and herbivores can contribute to "counterintuitive" increases in algae in habitats exposed to UV-B. Furthermore, as smaller phytoplankton cells are generally more sensitive to UV-B than larger ones (Karentz *et al.*, 1991; El-Sayed & Stephens, 1992; Bothwell *et al.*, 1993), increased UV-B could cause a significant change in the size spectrum of phytoplanktonic cells. Zooplankton relying on smaller cells would be dramatically impacted. Smaller phytoplankton species, i.e. the nano- and picoplankton, comprise a substantial proportion (up to 95% or more) of the total biomass of phytoplankton and primary production in the Southern Ocean (El-Sayed & Stephens, 1992). As Bothwell *et al.* (1994) concluded, predicting the response of entire ecosystems to elevated UV-B cannot be made on the basis of single trophic-level experiments.

All marine environments contain naturally high concentrations of viruses which appear to play a fundamental role in food web dynamics and in carbon and nitrogen cycling (Bergh *et al.*, 1989; Proctor & Fuhrman, 1990; Murray & Eldridge, 1994). Viruses in surface waters are likely susceptible to DNA damage by UV-B radiation as they have no protective membranes (Suttle & Chen, 1992; Karentz *et al.*, 1994). However, the study of marine viral ecology and its role in microbial planktonic food webs is in its infancy (Pomeroy, 1992) and the implications of increased UV-B stress on viral-bacterial-algal interactions are not known (Karentz *et al.*, 1994).

<u>c) Impacts on Biogeochemical Cycles</u>

Marine phytoplankton are a major biological sink for atmospheric CO2 and thus any decrease in their overall biomass could conceivably lead to a further increase in atmospheric CO2 concentrations and an enhancement of the greenhouse effect (Zepp *et al.*, 1995). Sulfur cycling may also be affected through impacts on the production of dimethyl sulphide (DMS) by certain marine phytoplankton and its air-sea exchange. The release of DMS may be the primary source of sulfur for cloud condensation nuclei in the remote marine atmosphere and, on a global scale, is responsible for about 15% of the total atmospheric sulfur input (Zepp *et al.*, 1995). The common and widespread phytoplankton species, *Phaeocystis*, an important producer of DMS, responds rapidly and adversely to UV-B exposure and, in this regard, appears to be more sensitive than co-occuring phytoplankton (Davidson & Marchant, 1994; Karentz & Spero, 1995). Significant reductions in DMS production could reduce cloud cover and alter the radiative balance of the atmosphere, weather patterns and the global climate (Hader *et al.*, 1995).

<u>d) Overview</u>

The above uncertainties notwithstanding, the issue of "environmental unknowns" (Myers, 1995) needs to be taken into account. Hader *et al.* (1995) suggest that the consequences of increased UV-B radiation are complicated by unpredicted feedback loops and other shifting conditions such as temperature, salinity, CO2 concentration, and radiation patterns caused by changing cloud cover. Cumulative environmental processes are now recognised as an important component in environmental change and as a phenomena that must be taken into account in environmental impact assessments, planning, research and policy (CEARC & US NRC, 1986; Cocklin *et al.*, 1992). It may well be that the cumulative stresses of long-term increases in UV-B fluxes and/or their unpredictable interactions with other components of global change that may ultimately decide the overall response in marine populations, including those of the higher level predators.

Existing uncertainties mean that effects on natural populations and on ecosystems can conceivably range from minimal to catastrophic (e.g., Roberts, 1989; Voytek, 1990). One worst-case scenario would see changes in phytoplankton populations and communities in the Southern Ocean of a scale that krill survival and reproductive capacity is substantially reduced. This in turn would lead to declines in cetacean populations.

The fact that there has been a significant decline in the ozone layer for over 15 years suggests that any biological and subsequent ecological effects have not only been initiated but are also being continuously modified. The lack of baseline data prior to ozone layer depletion makes it impossible to assess what exactly the overall ecological changes may have been thus far. The tremendous scientific uncertainties remaining about the current and future impacts of the continuing ozone hole, however, make it evident that we are still in the beginning stages of documenting the effects of an uncontrolled global experiment. The impacts, if any, on cetaceans and other higher level predators may already be occuring but remain to be identified and quantified.

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