Eutrophication, Water Borne Pathogens and Xenobiotic Compounds: Environmental Risks and Challenges

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Recent advances in pollution control and monitoring technologies, improved analytical capability, changes in government priorities and results of scientific studies have substantially changed our views and perceptions towards marine pollution in the last two decades. Globally, the problems caused by eutrophication, water borne pathogens and xenobiotic compounds are likely to be exacerbated and pose significant ecological and/or public health risks in the coming years, especially in developing countries. The large amount of anthropogenic input of nutrients has caused major changes in the structure and function of phytoplankton, zooplankton, benthic and fish communities over large areas, and such a trend is likely to continue in many coastal waters. Escalated public health risks associated with the increases in frequency and severity of toxic algal blooms are also of growing concern. Reduction of nutrient input through changes in land-use and farming practises, and the development of cost-effective methods for nutrient removal are required. Water borne pathogens affect large numbers of people through consumption of contaminated seafood and direct contact with contaminated water, and such problems are much more serious in developing countries. Current techniques in monitoring bacterial indicators in water and shellfish have clear limitations and cannot afford adequate protection to safeguard public health. Emerging molecular techniques, such as multiplex PCR and specific gene probes, are likely to provide new and cost effective tools for monitoring water borne pathogens in the coming years. Nowadays, xenobiotic compounds can be found almost everywhere in any marine ecosystems. Although these compounds normally occur at very low concentrations and their effects are not well understood, there is growing concern about the chronic exposure and bioconcentration/biomagnification of xenobiotic compounds. In particular, endocrine disrupters which may cause reproductive dysfunction and threaten species survival, are of growing concern. At present, most of our knowledge on toxic effects of xenobiotic compounds is derived from short-term exposure of a single species to high (environmentally unrealistic) and uniform concentrations under laboratory conditions. Data so derived are largely inadequate in predicting ecological effects in the field, in which multi-species are being exposed to varying, low concentrations under an interacting and complex environment. NOEC and LOEC for population/community/ecosystem, as well as the time required for population/community/ecosystems to recover after toxicant insult, are poorly known. These important topics will become the major endeavours for ecotoxicologists in the years to come. © 1999 Elsevier Science Ltd. All rights reserved.

A number of significant changes in the last two decades have provided new insights and contributed new knowledge to marine pollution studies. This has substantially changed our views and perception towards marine pollution. For example, technological advancement in pollution control (e.g. membrane technology, resin exchange and plasma arc technology) enables the removal of most pollutants from industrial discharges. The introduction of more stringent safety measures for tankers and ocean going vessels has greatly reduced the risk of oil pollution, although accidents still happen from time to time. The practices of clean technology and environmental accounting and audit reduce the amounts of pollutants generated and their input into the marine environment. Likewise, advances in pollution monitoring techniques (e.g. telemetry, remote sensing techniques using satellite images and multi-spectral images) provide a vast amount of water quality data covering large areas over prolonged periods, which could not be done in a cost effective way in the past. Global monitoring programs, such as the “Mussel Watch”, have contributed to our understanding of temporal and spatial trends in trace metals and toxic organics over large coastal areas. Remarkable improvements in analytical capability in recent years have lowered the detection limits of contaminants by several orders of magnitude (e.g. $10^{-12}$, as compared with $10^{-6}$ in the 1970s). Thus, we are now able to detect trace amount of environmental contaminants, which were not detectable in the past. QA/QC procedures have become much more sophisticated and
elaborate, and the validity and reliability of some of the earlier data (e.g. in 1970s) has now become questionable. Increasing environmental awareness of the general public makes both the government and private sectors accord a much higher priority to environmental protection, which has led to a substantial reduction in pollutant input in many areas. Finally, the increased efforts in marine pollution research have contributed tremendously to our understanding of the scientific aspects of marine pollution. Because of these significant changes, scientists have substantially changed their view towards marine pollution. For example, heavy metals and oil, which were once considered important, are of less threat and concern now, and significant effects of these two pollutants are now considered to confined to localised areas near significant discharges and major spillages (GEASMP, 1990). On the other hand, eutrophication, water borne pathogens and toxic xenobiotic compounds are likely to be the most pressing problems in the coming years.

Risk assessment and risk management are more commonly used as powerful tools in environmental management nowadays, especially when zero discharge or no development are not the best option. Risk is a function of hazards and the probability of a hazard being realised. Risk of certain pollutants may be expressed as ecological risk (i.e. their potential and probability in causing dysfunctions of population, changes in structure and function of ecosystems, spatial scale of change etc.), and public health risk (e.g. in terms of number of people affected, severity of disease etc.). Economic loss (in terms of monetary loss and resources loss), as well as reversibility and time for recovery from pollution effects are also important elements in risk assessment. This paper attempts to review the ecological and public health risks associated with eutrophication, water borne pathogens and toxic organic compounds. The global trends, economic loss, recovery time as well as challenges in dealing with these pollution problems in the coming years will also be considered and discussed.

Eutrophication

At present, some 65% of existing large cities (with more than 2.5 million people) are located along the coast. By the year 2000, world population will exceed 6 billion, of whom 60% (3.6 billion) will live within 100 km of coast (UNEP, 1991; The World Resources Institute, 1992). It is highly likely that a substantial proportion of wastewater generated from this population will be directly discharged into the coastal marine environment with little or no treatment, thereby adding to the already high nutrient input. Various studies have attempted to estimate the anthropogenic input of nutrients into the marine environment (e.g. Fogel and Paerl, 1993; Gabric and Bell, 1993; Paerl, 1993; Cornell et al., 1995). Although the various estimates do not agree with each other, there is little doubt that the magnitude of global anthropogenic flux of N and P is comparable to that of natural flux, indicating that human activities have already caused major disturbance in the distribution and balance of nutrients on earth.

Although eutrophication has been recognised as a significant problem in both freshwater and marine environments for more than three decades, this problem has not been solved thus far. On the contrary, the problem of eutrophication is likely to be exacerbated in the coming years for the following reasons:

- Coastal waters are often N limited, while both non-point source discharge and atmospheric fallout of N are significant and difficult to control. It is noteworthy that the present anthropogenic emissions and deposition of nitrogen to the North Atlantic Ocean is about five times greater than pre-industrial time (Prospero et al., 1996). At present, atmospheric deposition of N contributes some 10–50% of the total anthropogenic N input (20–100 mmol N/m²/yr), and this is expected to further increase in the coming years (Paerl, 1993).
- There is a world wide increase in irrigation in arid areas, large scale clearing of land vegetation, and deforestation, which contribute enormously to terrestrial runoff.
- Intensive farming results in overgrazing, ammonia emission, and farm waste disposal problems. Nutrient export from crop and pasture lands are typically an order of magnitude greater than those from pristine forest (Gabric and Bell, 1993). Mariculture activities has increased dramatically in many coastal areas in the last decade, and such a trend will continue (FAO, 1992). This will further augment the nutrient input into coastal environments, since some 80% of N input into a mariculture system will be lost into the marine environment (Gowen and Bradbury, 1987; Handy and Poxton, 1993; Wu, 1995).
- The volume of wastewater generated by human populations is typically large, and removal of nutrient from such huge amounts of wastewater is expensive. The cost of secondary treatment (which only removes some 30–40% of N and P) for example, is some 3–4 times more expensive that of primary treatment. Due to the high construction and recurrent costs, it is unlikely that building of sewage treatment facilities can match population growth and GNP in developing countries.

Ecological risk

Eutrophication is the process of nutrient enrichment (primarily N and P) that stimulates algal blooms, primary productivity and massive growth of macrophytes. Blooming and finally collapse of algae may lead to hypoxia/anoxia and hence mass mortality of benthos and fish over large areas. Sensitive species may be eliminated and major changes in ecosystem function may occur. Deteriorating environmental quality adversely affects an amenity, recreational values and the tourist industry. Indeed, increases in nutrient concen-
Eutrophication, phytoplankton biomass and productivity, alternation of nutrient ratios, change of species composition, and large scale hypoxia/anoxia affecting hundreds and thousands of km² have been reported in many areas all over the world (Rosenberg, 1985; Rabalais et al., 1991; Rosenberg et al., 1992; Justic et al., 1995; Diaz and Rosenberg, 1995). Eutrophication is now regarded as the most important pollution threat to marine waters (GEASMP, 1990).

Change in structure and function of ecosystems

Eutrophication has been shown to cause major changes in species composition, structure and function of marine communities over large areas. The general response of phytoplankton communities to eutrophication involves an increase in biomass and productivity (Vollenweider, 1992; Rieger, 1995). A general shift from diatoms to dinoflagellates, and also down shift in size in phytoplankton towards a dominance of small size nanoplanктон (e.g. microflagellates and coccoliths) is generally observed (Oviatt et al., 1989; Kimor, 1992a,b; Vollenweider, 1992; Hallengraef, 1995). A similar response is observed in zooplankton communities, with herbivorous copepods being replaced by small-size and gelatinous zooplankton (Regner, 1991; Zaitsev, 1992).

In Hong Kong, changes in phytoplankton composition have been clearly observed in Tolo Harbour and Channel since nutrient input into this semi-land locked system have increased, with diatoms being gradually replaced by dinoflagellates (Lam and Ho, 1989; Hodgkiss and Yim, 1995). Eutrophication also promotes proliferation of macroalgae and filamentous algae (e.g. Ulva and Enteromorpha). This often becomes a nuisance, and may affect benthic fauna, nursery and feeding of fish, amenity, recreational uses and tourism (Rieger, 1995; Rosenberg et al., 1996).

Eutrophication (and induced hypoxia) also alters the structure, diversity as well as trophic structure and food web of benthic and fish communities (Wu, 1982, 1988; Baden et al., 1990; Kerr and Ryder, 1992; Rosenberg et al., 1992; Zaitsev, 1992; Rieger, 1995). A decrease in dominance of predatory gastropods in the benthic community (Wu, 1982) and a shift from demersal fish species to pelagic species in response to eutrophication (Regier et al., 1988) have been reported. Changes in species composition of macrobenthos in response to eutrophication have also been reflected in the diet of demersal fish in Sweden waters (Phil, 1994). Mass mortality of fish and benthos and destruction of marine communities caused by large scale hypoxia/anoxia and eutrophication have been reported in numerous studies all over the world (e.g. Wu, 1982; Stachowitsch, 1984, 1992; GEASMP, 1990; Gomoiu, 1992; Desprez et al., 1992; Adnan, 1992; Diaz and Rosenberg, 1995).

Effects on coral reefs

Corals are evolved and adapted to low nutrient, warm waters where phytoplankton biomass is typically low, and are therefore not adaptable to eutrophic conditions. Both laboratory and field experiments have clearly demonstrated that corals are adversely affected by elevated nutrient levels (Bell, 1991; Stambler et al., 1991; Jokiel et al., 1994; Dubinsky and Stambler, 1996). Onset of eutrophication in coral reefs is estimated at approx. 1 μM DIN and 0.1–0.2 μM phosphate and 0.5 mg/m³ Chl. a. (Bell, 1992), while significant adverse effects have been observed at 10–100 μM ammonia and 1–10 μM phosphate. An increase in plant biomass in water reduces light intensity and alters the light spectrum, thereby affecting photosynthesis of zooxanthellae (Tomascik and Sander, 1985). Proliferation of macrophytes may also overgrow corals (Genin et al., 1995). Zooxanthellae are N-limited, and under normal conditions most of the photosynthate they produce is passed to the coral host. High nitrogen causes an imbalance in nutrient exchange between the host and the zooxanthellae, in which case zooxanthellae retain their own photosynthate for their own biomass production and reduce coral growth (Dubinsky and Stambler, 1996). Tomascik and Sander (1985) demonstrated a reduction of adult coral growth in response to eutrophication under field conditions. Hunte and Wittenberg (1992) demonstrated a lower coral settlement as well as a lower ratio of coral recruits to adult coral in eutrophic areas, providing good evidence that coral settlement may be retarded under eutrophic conditions.

The steady increase in nutrient concentration and phytoplankton biomass along the west coast of Barbados has eliminated sensitive species and led to a reduction in coral diversity and an alternation of community structure of scleractinian coral (Tomascik and Sander, 1985). Likewise, large scale eutrophication is considered to be a significant threat to the 2000 km Great Barrier Reef coast (Bell and Elmetri, 1995).

Algal and toxic blooms

The major impact of eutrophication is the stimulation of algal growth and the production of toxic algal blooms. Increases in the frequency and spatial scale of algal blooms, toxic blooms and red tides have been reported all over the world in the last two decades (Vollenweider, 1992; Viviani, 1992; Qi et al., 1993). Large scale algal blooms cause serious ecological damage and economic loss, while toxic blooms pose additional public health threats. For example, blooms of the toxic dinoflagellate Gymnodinium aureolum (> 1 million cells/L) were observed in 13 out of 25 years in Norwegian waters (Dahl and Tangen, 1993). In 1988, blooming of the small flagellate Chrysochromulina polylepis (50–100 million cells/L) in Norway covered an area of 75,000 km², and toxin produced caused great damage to seaweed, invertebrates and feral fish and cultured salmon along a 200 km stretch, with fisheries losses of more than US$10 million (GEASMP, 1990).

More than 160 red tides have been reported in Chinese waters from 1980 to 1990, and the frequency,
magnitude and geographic extent of red tides along the coast of China has increased in the last decade. The area covered ranged from 10 to 6100 km² and over 60 causative red tide species have been reported (Qi et al., 1993). In 1989, a red tide affected 1.5 × 10³ ha. shrimp farms in Bohai, and the total loss was estimated at US$40 million (Xu et al., 1993).

In Hong Kong, a red tide caused by a persistent bloom of Gonyaulax polygramma (> 50 million cells/L) occurred continuously for three months in Tolo Harbour and Channel, covering an area of some 80 km², and all fish and benthos were killed in this incidence. In 1998, a major and extensive red tide outbreak occurred along the coast of Hong Kong and S. China, covering an area of more than 100 km². Over 80% (3400 ton) of mariculture fish were killed, and the total loss was over US$40 million.

**Public health risks**

Biotoxins produced by toxic algae are of major public health concern world wide. Although there is no firm evidence to directly link toxic algal blooms with eutrophication, there is little argument that the probability of having toxic algal blooms and red tides are much higher in eutrophic waters. Public health risk associated with eutrophication is mainly through consumption of seafood (mostly filter feeding shellfish) contaminated with red tide toxins, although direct contact with toxic algae may also cause dermatitis and conjunctivitis. Shellfish poisoning caused by toxic algae may be grouped into four major categories (Table 1). The various types of shellfish poisoning have caused significant public health problems world wide. For example, 24 people died and more than 1000 suffered from PSP in a Pyrodinium bloom in the Philippines (Estudillo and Gonzales, 1984). 441 cases of PSP and 15 deaths were reported after consuming the hard clam Meritrix meritrix and the fish Sardinella spp. during three major PSP outbreaks in Indonesia (Adnan, 1993). In France, some 3000 people were affected by DSP intoxication in 1983, after consumption of mussels exposed to a bloom of Dinophysys spp. (Lassus et al., 1991; Belin, 1993). In Hong Kong, levels of PSP toxins in shellfish collected from Tolo have tripled from 1984 to 1987 (Wu, 1988; Morton, 1989).

**Trends**

Progressive increases in nutrient concentration and altered nutrient ratio (e.g. N/P and Si/N) have been reported from the Baltic Sea, Wadden Sea, North Sea, Black Sea, Adriatic Sea, Dutch Sea, Japan Sea, the coast of Germany, China, Australia and Hong Kong (Justic et al., 1987; Wu, 1988; GEASMP, 1990; Gomoiu, 1992; Goonan et al., 1992; Moss et al., 1992; Vollenweider, 1992; Gabriel and Bell, 1993; Bell and Elmetri, 1995; Justic et al., 1995; HELCOM, 1996; De-Jonge et al., 1996). In many places, the level or input of nutrient has increased several times within the last 20–60 yr. For example, concentrations of nitrate have increased 5 times and phosphate 20 times in the Black Sea from 1960s to 1980s (Gomoiu, 1992). Likewise, levels of N and P in Dutch Seas have increased 4 and 2 times respectively from 1930 to 1980 (GEASMP, 1990). Three to five times increases in N and P export have been reported in Queensland, Australia, in the last 65 yr (Moss et al., 1992). The increase in nutrients has led to a corresponding decrease in dissolved oxygen and an increase in primary productivity. For example, progressive increases in primary productivity and decreases in dissolved oxygen due to eutrophication have been reported in the Baltic Sea from 1958 to 1989 (Nehring, 1992; HELCOM, 1996). A decrease in bottom oxygen was found in northern Adriatic Sea during the period 1911–1984 (Justic et al., 1987). The long-term increase in nutrient in the Baltic has caused an increase in phytoplankton biomass, a decrease in water transparency, proliferation of filamentous algae, and also large scale changes in species diversity of benthic and fish communities (Bonsdorff et al., 1997). Globally, increases in frequency and severity of hypoxia are evident, especially in coastal and estuarine areas; many ecosystems are now near the verge of hypoxia-induced catastrophe (Diaz and Rosenberg, 1995).

In the last two decades, there has been an increased frequency and scale of red tides in coastal waters of Brunei, Malaysia, Philippines and Thailand. In the last two decades, an increase in PSP frequency has been found in both temperate and tropical regions, although part of the increase may be due to an improvement in surveillance or reporting systems (Viviani, 1992).

The high cost involved makes it highly unlikely that construction of sewage treatment facilities can catch up with the rapid population growth in the coming years. Thus, the problem of eutrophication is likely to be exacerbated, especially in developing countries where GNP growth is slower than population growth.

**Recovery**

A paucity of scientific data does not permit an accurate prediction of the recovery of ecosystems after

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<th>Type</th>
<th>Example spp.</th>
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<td>Paralytical shellfish poisoning</td>
<td>Gonyaulax spp. (Alexandrium spp.)</td>
<td>Domoic acid</td>
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<td>Neurotoxic shellfish poisoning</td>
<td>Gymnodinium (Psychodiscus) brev</td>
<td>Brevetoxins</td>
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<td>Diarrhoetic shellfish poisoning</td>
<td>Dinophysys sp. Prorocentrum spp. Gymnodinium aureolum</td>
<td>Saxitoxin, neosaxitoxin or gonyautoxin</td>
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<td>Amnesic shellfish poisoning</td>
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A paucity of scientific data does not permit an accurate prediction of the recovery of ecosystems after
impact from eutrophication. Studies carried out in the temperature region indicated that recovery may take a long time. For example, macrobenthic communities in Gullmarsfjord did not re-established 18 months after a major hypoxic event. (Josefson and Widborn, 1988). Little or no recovery were found after two years in the Kathegat after benthic defaunation caused by hypoxia (Rosenberg et al., 1992). Olsgard (1993) showed that recovery of a benthic community from hypoxia caused by a toxic algal bloom (Chrysochromulina polylepis) in Norway took about 2 yr. Diaz and Rosenberg (1995) opined that no large system has recovered after development of persistent hypoxia to date, while small systems are able to recover once pollution ceased. Tolo Harbour in Hong Kong is a highly eutrophic system, and experiences periodic defaunation in the summer followed by rapid winter recovery. Field studies and field manipulation experiments provide evidence that recolonization and recovery of the benthic community after defaunation is rapid, and may range from several months to one year (Wu, 1982; Gamenick et al., 1996; Wu and Shin, 1998; Lu and Wu, 1998).

Challenges

While there is no shortage of techniques for nutrient removal, the main problem lies in the cost-effectiveness of these methods. Nutrient removal is expensive (e.g. capital and recurrent costs of secondary treatment is some 3–4 times higher than that of primary treatment). Reduction of nitrogen by 50% from a population of 85 million people in the catchment of the Baltic countries is estimated at US$20 billion (HELCOM, 1996). So far, no adverse effects are observable for nutrient disposal into open waters, and such disposal methods remain a cost-effective and acceptable option (Vollenweider, 1992). Shallow, stratified coastal systems with high nutrient input and limited dispersing capacity are, however, at risk. Such vulnerable areas along coasts should therefore be identified, and their carrying capacity and ecological characteristics determined. The carrying capacity (e.g. “trigger level” of eutrophication) needs to be estimated, if nutrients are disposed of in these water bodies. For example, critical values from oligotrophy to eutrophy in the Baltic Sea have been estimated at somewhere between 0.05 and 0.25 g-P/m²/yr and 1–3.5 g-N/m²/yr (Vollenweider, 1992). Statistical analysis of long-term water quality data in 28 embayments suggests that the trigger level for algal blooms in Hong Kong coastal waters is about 1 mg-N/L (Ove Arup et al., 1988).

Since a significant proportion of nutrient entering the ocean is from land farming and terrestrial runoff, a fundamental change in land farming practices, and land use and land management in the hinterland is required to reduce input from these important sources (Gabric and Bell, 1993). Natural wetland can be effective in nitrogen removal at low loading rates, and can remove some 40–70% of nitrogen (Nichols, 1983; Short and Short, 1984; Patruno and Russell, 1994), while constructed wetlands can remove 43–65% ammonia in wastewater (Nuttall et al., 1997). Restoration of wetland may be a cost-effective option for nutrient removal, especially in developing countries. The possibility of harvesting nutrients by culturing economic macroalgae (e.g Ulva and Porphyra spp.) in lagoons may also be explored (Cuomo et al., 1993; Wu, 1995).

Water Borne Pathogens

Water borne pathogens in the marine environment are mainly derived from untreated or inadequately treated sewage, although several human bacterial pathogens are indigenous to estuarine and sea waters (e.g. halophilic vibrios). Infection may be through consumption of contaminated seafood or direct contact through bathing or recreational water activities. Gastrointestinal tract infection caused by water borne pathogens remained one of the most wide spread health problems in coastal cities with high population densities (GEASMP, 1990).

Contact with pathogenic bacteria (e.g. Staphylococcus aureus, Pseudomonas aeruginosa) and certain virus in sea water may cause ear, eye and skin infection or respiratory diseases, while halophilic vibrios may cause ear and wound infections. Epidemiological evidence has firmly established the relationship between bathing in sewage polluted waters and pathological incidences (Larbaigt, 1989), and this means that potentially a large number of people may be affected. In Hong Kong, for example, the 42 gazetted beaches are visited by more than 20 million people each year. In 1997, the microbiological standard of 14 of these beaches exceeded the local standard (180 E. coli/100 ml), and 4 had to be closed down. Results of an epidemiological survey showed that swimmers in Hong Kong waters have 1.5 times higher risk than non-swimmers, implying that a very large number of people is at risk.

In tropical and sub-tropical developing countries, a very high percentage of sewage is discharged into the sea without treatment and seafood harvested in these places constitutes a major health hazard (GEASMP, 1990). Epidemics caused by consumption of bivalves have been reported from time to time all over the world, although such problems are much more significant in developing and densely populated countries. High levels of indicator bacteria, pathogenic bacteria and viruses are commonly found in bivalves, sediment and water along the West African coast, in Asian seas, the Pacific and the Caribbean. Correspondingly, a high incidence and epidemic outbreaks of gastro-intestinal infections, diarrhea and viral hepatitis caused by consumption of contaminated shellfish have been widely reported from these areas.

Hepatitis A Virus (HAV) is a major water borne pathogen worldwide (Nasser, 1994). In 1988, a major outbreak of HAV occurred in Jiangsu and Jiangjiang, China. About 400,000 people were infected due to the consumption of polluted shellfish (Aera spp.) (Wu,
16 cases of hepatitis were reported in the first three months of 1988. Of these, 647 cases were confirmed as Hepatitis A and some 80% of the patients had consumed local shellfish prior to infection (Wu, 1988). The incidence of HAV ranged from 264 to 3636 incidences/year in the last ten years. In 1996, for example, 170 out of 264 (64%) of HAV infection were caused by consumption of shellfish. Results of food surveillance programs during 1995–1997 showed that 7.9–12.9% of shellfish in the Hong Kong market exceeded the local microbiological standard (Health Department, Hong Kong Government, unpublished data).

Contamination of seafood is not only a problem restricted to developing countries, but also occurs in developed countries. For example, 57 outbreaks of viral gastroenteritis were reported in New South Wales, Australia in 1990, in which 1750 people were affected, 550 of them (31%) ate oysters (Bird and Kraa, 1995). A high level of enteric virus contamination in shellfish samples was also found from a major urban market in Europe, suggesting that the existing sanitary practices (culture, handling and storage) as well as standards and regulation need to be re-examined (GEASMP, 1990).

Technology for disinfecting water borne pathogens (e.g. UV or ozone treatment) is readily available, and shellfish depuration prior to sale has been practised in some countries (Siewicki, 1988). Regular monitoring of coastal water and shellfish is required in order to protect human health. However, water borne pathogens normally occur in low numbers in water samples, and large errors may therefore be incurred in sampling and enumeration (Fleisher, 1996). Furthermore, culture techniques for specific pathogens are invariably species specific, tedious, time consuming and insensitive (Jaykus et al., 1995). Because of these difficulties, pathogens are not normally enumerated in water quality and shellfish monitoring. Instead, faecal coliforms and E. coli, which occur in high numbers in animal faces, are used to indicate faecal pollution, and hence the risk of potential exposure to pathogens derived from faecal origins.

Recent epidemiological evidence has shown that water borne pathogens as well as pathogenic incidence do not correlate well with total coliform, faecal coliform, or E. coli counts. For example, incidence of bathing related GI diseases is much better correlated with Enterococcus than E. coli. (Cabelli, 1989), and not related with total faecal coliform counts (GEAAMP, 1990; Fleisher, 1990; Lund, 1994), or with fecal streptococi (Larbaigt, 1989). Likewise, it has been shown that levels of Vibrio cholerae O1 in oysters were not related to either fecal coliform levels in either oysters or ambient waters (Motes et al., 1992). Furthermore, survival of these bacterial indicators (e.g. E. coli and fecal coliform) in the marine environment is very different from that of other pathogens (Matsumoto and Omura, 1980). For example, $T_{90}$ of coliform, faecal coliform and E. coli. in sea water ranges from a few hours to a few days, depending on species, temperature, sunlight, turbidity and salinity (Pompeup et al., 1992), while $T_{90}$ of enteric virus (e.g. adenovirus causing gastroenteritis in children) and HAV is more than a month (Nasser, 1994). The above evidence clearly demonstrates that the use of E. coli or faecal coliforms as indicators for water borne pathogen are largely inadequate. Boher et al. (1995) pointed out that the present use of bacteria indicators for sanitary standards is largely unsatisfactory, and cannot afford adequate protection to public health.

A recent advent in molecular biology offers a variety of new techniques potentially applicable to direct monitoring of water borne pathogens. For example, Kong et al. (1995) used multiplex PCR techniques for concomitant detection of six pathogenic bacteria in a single water sample. Their current work is being extended to couple multiplex PCR techniques with fluorescent quantitative PCR and ELISA, to provide a quantitative or semi-quantitative measure of water borne pathogens (Kong et al., in preparation). Johnson et al. (1995) used the PCR technique to determine Cryptosporidium in water samples. Likewise, PCR has been used to detect Vibrio cholerae (Karunasagar et al., 1995) and Norwalk virus and HAV in shellfish (Atmar et al., 1995). Radioimmunoassay and RT-PCR have been used to detect HAV and enteric virus in shellfish (Crance et al., 1995; Jaykus et al., 1995), and attempts have been made to use cDNA and RNA gene probes to detect virus in shellfish samples (Chaput and Margolin, 1991; Appaire-Marchais et al., 1995). These new molecular techniques appear to be extremely promising in monitoring water borne pathogens in water and shellfish samples, and confer great advantages over the current use of bacterial indicators. First, these techniques are highly species-specific and accurate. Second, unlike the use of bacteria indicators, several types of water borne pathogens can be directly measured in a single water/shellfish sample, thereby providing a much more accurate risk assessment. Third, determination is much quicker while the cost is much lower. For example, the PCR technique yields results in several hours, while the existing culture techniques for E. coli and faecal coliforms take at least 48 h.

Xenobiotic Organic Compounds

Many synthetic organic chemicals (e.g. organochlorines, organophosphates, PAHs and organometals) are of growing environmental concern, because of their high toxicity and high persistence in the environment and in biological systems. Furthermore, the high lipophilicity of many of these xenobiotics greatly enhances their bioconcentration/biomagnification, thereby posing potential health hazards on predators at higher trophic levels (including human beings). Nowadays, persistent xenobiotic compounds have been found in every part of...
the ocean: from arctic to antarctic, and from intertidal to abyssal. For example, PCBs, HCH and DDT (and its derivatives) were found in rat-tail fish collected at 3000 m depth in the Atlantic and arctic seals long after the ban of DDT and PCBs in the early 1970s, indicating the persistence of these chemicals in the marine environment (GEASMP, 1990).

Most xenobiotic compounds occur only at very low concentrations in the environment, and their threats to marine life and public health are still not well understood. However, sub-lethal effects of these compounds over long-term exposure may cause significant damage to marine populations, particularly considering that some of these compounds may impair reproduction functions of animals while others may be carcinogenic, mutagenic or teratogenic.

Ecological risk

The major ecological concern of xenobiotics is their ability to impair reproductive functions and subsequently threaten survival of the species. For example, white croaker inhabiting contaminated areas near Los Angeles have higher body burdens of chlorinated hydrocarbons, lower fecundity and lower fertilization rates (Cross and Hose, 1988). Likewise, endocrine dysfunction and reduced gonad size were reported for the yellow perch (Perca flavescens) exposed to sediments in the St. Lawrence River contaminated with PAHs and PCBs (Hontela et al., 1995). Epidemiological and experimental evidence exist that reproductive failure and population decline of the common seal (Phoca vitulina) in the Wadden Sea were attributable to their PCB body burden (Reijinders, 1986). However, PCBs in common seal and killer whales (Orcinus orca) in Puget sound are amongst the highest in the world, but no correlation has been found between birth rate and body burden. High body burden of DDT and PCB also led to skeletal and uterine deformations in seals in the Baltic. Following the ban of DDT and PCBs, the average percentage of seals with uterine deformation decreased from 36% in 1977–1986, to 25% in 1987–1993 (HELCOM, 1996). High body burden of organochlorines found in seals and sea birds in the Baltic has been related to reduced egg hatching (HELCOM, 1996).

There is growing evidence that exposure to very low (nanogram) levels of certain xenobiotic organic compounds (e.g. halogenated hydrocarbons, PCBs, DDT, TBT) may disrupt normal metabolism of sex hormones (including gonadotropins) in fish, birds and mammals. This in turn, may lead to reproductive dysfunction such as reduction in fertility, hatch rate, alternation of sex behavior and viability of offspring (Crews et al., 1995). Perhaps one of the most well studied endocrine disrupters is estrogen. Exposure to very low levels of TBT (0.5–3 ng/l) or a body burden of only 10–20 ng TBT/g wet tissue has been shown to cause a significant disruption in sex hormone metabolism/testosterone level, which subsequently leads to malformation of oviducts and suppression of oogenensis in female whelks (e.g. Nucella lapillus, Thais claviger and T. brommi (Bettin et al., 1996; Gibbs, 1996). Secondary male characteristics, such as induction of spermatogenesis and development of a male penis and/or vas deferens, begins to develop in the females. This phenomenon, known as imposex, has been reported in some 50 species of gastropods all over the world in areas with high marine traffic/activities or where TBT has been used. The frequency of imposex in field populations shows a clear relation to environmental TBT levels, and sex imbalance causes a decline and species extinction in some natural populations (Cadee et al., 1995).

Increasing evidence from laboratory and field studies has shown that trace amounts of many chlorinated hydrocarbons (e.g. PCBs), organophosphates and di-ethyliiibestrol in the environment may cause significant endocrine disruption and reproductive disturbance/failure in invertebrates, fish, birds, reptiles and mammals. Chronic exposure to low levels of diethylstilbestrol or pentachlorophenol alters steroid hormone metabolism of the water flea Daphnia magna and reduces their fecundity in the second generation (Baldwin et al., 1995; Parks and LeBlanc, 1996). Exposure to very low levels of certain organophosphate pesticides (e.g. elsan, carbaryl) has been shown to inhibit gonadotropin releasing hormones and reduce gonad development in the fish Chauna punctatus (Bhattacharya, 1993). Likewise, the pesticide kepone has been shown to arrest sperm maturation and block a variety of “oestrogen-like” effects on female reproductive systems in many fish, birds and mammals (Srivastava and Srivastava, 1994). Endocrine disruption found for the above freshwater species may well be applicable to marine species. A recent mesocosm study showed a significant elevation of testosterone and 17-β-oestradiol in the flounder Platichthys flesus exposed to polluted dredged spoil (Janssen et al., 1997).

Common seals (Phoca vitulina) fed with PCB contaminated fish and grey seals (Halichoerus grypus) with high body burdens of PCBs and DDT had significantly lower levels of retinol and free thyroxin. The disturbance in hormonal systems was also related to an increase in microbial infections and reproductive disorders in natural seal populations in the Baltic and North Seas (Brouwer et al., 1989; Jenssen, 1996). Disruption of neuroendocrine functions after exposure to Aroclor 1254 has been demonstrated in Atlantic croaker Micropogonias undulatus (Khan and Thomas, 1996). In the Baltic Sea, high levels of DDT, PCBs and organochlorines markedly reduced the hatching rates of eggs (from 72% to 25%) and the nesting success of the fish eating White-tailed eagle (Haliaeetus albicilla) in 1960s and 1970s (from an average of 1.8–1.2 fledging per nest). Nesting and reproductive success showed a steady increase following the ban on DDT and PCBs, and in 1994, the values almost resumed those prior to the occurrence of organochlorines (HELCOM, 1996). Delayed sex maturity, smaller gonads, reduced fecundity and a
depression in secondary sexual characteristics were reported in fish populations downstream of bleached kraft pulp mills, and these changes were confirmed in fish exposed to treated effluents under laboratory conditions. The changes were closely related to alternations in endocrine systems controlling the production of sex steroid hormones. Improved reproductive performance was found in feral fish at 5 sites after the mills improved their waste treatment and pulping processes (Munktittrick et al., 1997).

Public health risk

Fisheries product in several parts of the world (e.g. middle Atlantic Bight of USA) contain residues of unacceptable levels of PCBs (up to 84 ppm in lobsters and 730 ppm wet wt. in finfish) (GEASMP, 1990). Results of food surveillance programs in Hong Kong from 1995 to 1997 showed that 2.7–7.3% of shellfish in the market exceeded the local standard (PCB < 2 ppm, ΣDDT < 5 ppm.) A recent study by Connell et al. (1998) showed that PCBs, ΣDDT and total HCH pose a significant risk to the marine ecosystems and pose a health hazard to seafood consumers. In general, however, current levels of toxic organic residues in seafood do not appear to pose a significant public health risk, and so far significant health effects have been confined to exceptional cases (GEASMP, 1990; Kannan et al., 1994; Kennish and Ruppel, 1996; Roots, 1996).

Trends

Results of long-term monitoring programs show a general decrease in environmental levels of DDT and PCB in many coastal waters. For example, the annual geometric means of DDT, PCBs and PAHs in mussels at 154 sites in coastal waters of the USA showed a general decrease of 1986 to 1993 (Beliaeff et al., 1997). Likewise, Blomkvist et al. (1992) showed a significant decrease in ΣDDT and PCB in the blubber of 109 specimens of ringed seals (Phoca hispida botica), grey seals (Halichoerus grypus) and harbour seals (Phoca vitulina vitulina) in Swedish waters since the early 1970s. No increase in the concentration of non-ortho coplanar PCBs could be found in liver oil of cod from the southern Baltic Sea during 1971–1989 (Falandyssz et al., 1994). Analysis of sediment core samples in Clyde estuary, UK showed a significant decrease in PAH deposition over time (Hursthouse et al., 1994). The decreased concentration of xenobiotics in the marine environment reflects the general reduction in the use and discharge of these compounds in the northern hemisphere. Unfortunately, very few long-term studies have been carried out in tropical and sub-tropical coastal waters. The decreasing trend observed in temperate regions may not be applicable to tropical and sub-tropical waters, since reduction in use and disposal of toxic organic chemicals in the latter regions may not be the same.

Recovery

Due to long environmental and biological half lives, recovery from the effects of many xenobiotic compounds is expected to be slow. Indeed, it has been shown that some 15 yr are required to remove the negative effects of DDT on reproduction of white tail eagles (Haliaeetus albicilla) in the Baltics, and another 10 years for the population to recover. Likewise, long recovery times have been reported for harbour, grey and ringed seals in the Baltic. The grey seal (Halichoerus grypus) population in the northern Baltic has shown marked increase since the ban of DDT and PCB in the Baltic region (HELCOM, 1996). Despite a decrease in environmental concentrations, the adverse effects may remain in the ecosystem for a much longer period. In the Baltics, DDE decline to 10% of the original levels in 1984, but increased again afterwards, and the egg shells of fish eating birds, which had begun to return to normal, have recently become thinner again. Thus, the downward trend was halted after the ban, and may be due to the recycling of persistent chemicals in sediment (HELCOM, 1996).

Widespread recovery of dog whelk populations has been reported in European waters several years after the introduction of regulations banning the use of TBT for vessels less than 25 m (Minchin et al., 1995; Evans et al., 1996).

With tighter pollution control and the introduction of clean production technology in developed countries, environmental concentrations of xenobiotic compounds are expected to decrease in coastal areas. However, the same is unlikely to happen in developing countries in the near future, since pollution control technology is less developed and clean technology less widely practised.

The ecological and public health risks of eutrophication, water borne pathogens and xenobiotic compounds are summarised in Table 2.

Challenges

One of the most intractable problem facing ecotoxicologists is the extrapolation of laboratory data to field conditions. Most of our knowledge on the toxic effects of organic chemicals is derived from short-term exposure of a single species to high (often environmentally unrealistic) and uniform concentration under laboratory conditions. Data so derived have been used to predict ecological effects in the field, in which multi-species are being exposed to varying, low concentrations in an interactive and complex environment. While the effects of non-persistent, acutely toxic chemicals in field tests can be accurately predicted from laboratory studies, test results on many toxicants may overestimate the potential effects in the field because some chemicals are subject to photo-oxidation and biodegradation. The bioavailability and persistence of many xenobiotic compounds (and hence toxicity) are likely to be reduced under field conditions.
On the contrary, laboratory tests would tend to underestimate the potential impact of chronic toxicants that require metabolic activation or involve biomagnification. The great variety of physical, chemical and biological factors in natural environments which affect biological effects are difficult, or impossible, to simulate under laboratory conditions. Furthermore, ecological factors and their interactions such as trophic relationships, species interactions, nutrient dynamics, limiting factors, environmental fate of chemicals in the natural systems etc. are poorly understood, and cannot possibly be reproduced in laboratory and mesocosm studies. These uncertainties pose questions on the validity of extrapolation, or making predictions on ecological consequences from laboratory test results.

Most of the early ecotoxicological studies focused on sub-organisimal responses. Relatively few studies have been carried out, and little is known on population/community/ecosystem responses, and current predictive power at the population level and above is relatively weak. While the concept of “no observable effect concentration” (NOEC) and “lowest observable effect concentration” (LOEC) have been commonly used in studying sub-organisimal responses, NOEC and LOEC for population/community/ecosystem, as well as the time required for population/community/ecosystems to recover after toxicant insult, are poorly known. These important topics will become the major endeavours of ecotoxicologists in the years to come. It is obviously impossible to test the effects of every toxic organic chemical. The recent Quantitative Structural Activity Relationship (QSAR) approach appears to be particularly useful in addressing such problems.

**Concluding Remarks**

Eutrophication, water borne pathogens and xenobiotics are imminent marine pollution problems. These problems are likely to exacerbate pose significant ecological risk/public health risk in the coming years, especially in developing countries. Eutrophication has already caused major changes in the structure and function of phytoplankton, zooplankton, benthic and fish communities over large areas. Increases in frequency and severity of toxic algal blooms as a result of eutrophication pose a significant risk to coastal living resources as well as public health. Reduction of nutrient input through changes in land-use and farming practices, and the development of cost-effective methods for nutrient removal are required. A large numbers of people is at risk through consumption of contaminated seafood and direct contact with contaminated water, and such problems are much more serious in developing countries. Current techniques in monitoring bacterial indicators in water and shellfish have clear limitations and cannot afford adequate protection to safeguard public health. Emerging molecular techniques, such as multiplex PCR and specific gene probes, are likely to provide new and cost effective tools for monitoring water borne pathogens in the coming years. Although xenobiotic compounds normally occur at very low concentrations and their effects are not well understood, there is growing concern about their chronic exposure and bioconcentration/biomagnification. There is growing evidence to suggest that some of these xenobiotics may disrupt normal endocrine functions of marine animals, thereby causing reproductive dysfunction and threaten species survival. Most of our knowledge on toxic effects of xenobiotic compounds is derived from short-term exposure of a single species to high (environmentally unrealistic) and uniform concentrations under laboratory conditions. Data so derived are largely inadequate in predicting ecological effects in the field, in which multi-species are being exposed to varying, low concentrations under an interacting and complex environment. NOEC and LOEC for population/community/ecosystem, as well as the time required for population/community/ecosystems to recover after toxicant insult, are poorly known. These important topics will be major challenges in ecotoxicological research.


