Environmental threats and environmental future of estuaries

MICHAEL J. KENNISH*
Institute of Marine and Coastal Sciences, Rutgers University, New Brunswick, New Jersey 08901-8521, USA

SUMMARY

Estuaries exhibit a wide array of human impacts that can compromise their ecological integrity, because of rapid population growth and uncontrolled development in many coastal regions worldwide. Long-term environmental problems plaguing estuaries require remedial actions to improve the viability and health of these valuable coastal systems. Detailed examination of the effects of pollution inputs, the loss and alteration of estuarine habitat, and the role of other anthropogenic stress indicates that water quality in estuaries, particularly urbanized systems, is often compromised by the overloading of nutrients and organic matter, the influx of pathogens, and the accumulation of chemical contaminants. In addition, the destruction of fringing wetlands and the loss and alteration of estuarine habitats usually degrade biotic communities. Estuaries are characterized by high population densities of microbes, plankton, benthic flora and fauna, and nekton; however, these organisms tend to be highly vulnerable to human activities in coastal watersheds and adjoining embayments. Trends suggest that by 2025 estuaries will be most significantly impacted by habitat loss and alteration associated with a burgeoning coastal population, which is expected to approach six billion people. Habitat destruction has far reaching ecological consequences, modifying the structure, function, and controls of estuarine ecosystems and contributing to the decline of biodiversity. Other anticipated high priority problems are excessive nutrient and sewage inputs to estuaries, principally from land-based sources. These inputs will lead to the greater incidence of eutrophication as well as hypoxia and anoxia. During the next 25 years, overfishing is expected to become a more pervasive and significant anthropogenic factor, also capable of mediating global-scale change to estuaries. Chemical contaminants, notably synthetic organic compounds, will remain a serious problem, especially in heavily industrialized areas. Freshwater diversions appear to be an emerging global problem as the expanding coastal population places greater demands on limited freshwater supplies for agricultural, domestic, and industrial needs. Altered freshwater flows could significantly affect nutrient loads, biotic community structure, and the trophodynamics of estuarine systems. Ecological impacts that will be less threatening, but still damaging, are those caused by introduced species, sea level rise, coastal subsidence, and debris/litter. Although all of these disturbances can alter habitats and contribute to shifts in the composition of estuarine biotic communities, the overall effect will be partial changes to these ecosystem components. Several strategies may mitigate future impacts.

Keywords: Estuaries, biotic communities, anthropogenic effects, pollution, habitat loss and alteration, future impacts

INTRODUCTION

Among the most important environments of the coastal zone are estuaries, which constitute transition zones or ecotones, where fresh water from land drainage mixes with seawater, creating some of the most biologically productive areas on Earth. Along the coast of the USA alone, nearly 900 estuaries cover approximately 1.09×10^7 ha (Biggs 1982; Kennish 1986); together with lagoon systems, they occupy 80–90% of the Atlantic and Gulf of Mexico coasts and 10–20% of the Pacific coast (Emery 1967). These complex systems vary considerably in geomorphology, hydrography, salinity, tidal characteristics, sedimentation, and ecosystem energetics (Table 1). As a result, biotic communities also differ substantially in estuarine systems.

When proceeding from above the head to below the mouth of an estuary, as many as six distinct zones are recognized based on salinity distribution (i.e. the Venice System; Table 2). The limnetic zone is found in the area where a river enters an estuary. The oligohaline zone generally lies at the head, and the mesohaline zone, at the upper reaches of the system. The polyhaline zone usually occurs at the middle and lower reaches of an estuary, and the euhaline zone at the estuarine mouth. The hyperhaline zone is characteristic of negative estuaries where elevated rates of evaporation result in excessively high salinities.

Estuaries have been defined repeatedly in the literature. The definition of Pritchard (1967) states that an estuary is a semi-enclosed coastal body of water with a free connection to

Table 1 Types of estuaries.

Criteria	*Estuarine type*	*Example*
Geomorphology/ physiography	Drowned river valley	Chesapeake Bay, USA
	Bar-built estuary	Pamlico Sound, USA
	Fjord-type estuary	Sognefjord, Norway
	Tectonic estuary	San Francisco Bay, USA
Circulation	Salt-wedge estuary	Tees Estuary, UK
	Partially-mixed estuary	Narragansett Bay, USA
	Well-mixed estuary	Coos Bay, USA
Salinity	Positive estuary	Severn Estuary, UK
	Negative estuary	Laguna Madre, USA
Tidal	Microtidal estuary	Zaire River Estuary, Africa
	Mesotidal estuary	Sado Estuary, Portugal
	Macrotidal estuary	Ord River Estuary, Australia
Sedimentation	Positive filled estuary	Hangzhou Bay, China
	Inverse filled estuary	Wadden Sea, Netherlands
	Neutral filled estuary	Skjomen Fjord, Norway
Ecosystem energetics	Natural stressed systems	Barnegat Inlet, USA
	Natural tropical ecosystems of high diversity	Biscayne Bay, USA
	Natural temperate ecosystems with seasonal programming	Delaware Bay, USA
	Natural Arctic ecosystems with ice stress	Skelton Inlet, Antarctica
	Emerging new systems associated with man	Liverpool Bay, UK
	Migrating subsystems that organize areas	Aransas Bay, USA

Table 2 Venice System for the classification of estuaries.

	Venice System	
Section of estuary	*Salinity (‰)*	*Zone*
River	< 0.5	Limnetic (freshwater)
Head	0.5–5	Oligohaline
Upper reaches	5–18	Mesohaline
Middle reaches	18–25	Polyhaline
Lower reaches	25–30	Polyhaline
Mouth	30–40	Euhaline
–	> 40	Hyperhaline

the open sea and within which seawater is measurably diluted with fresh water derived from land drainage. This widely used definition excludes coastal lagoons, which may be temporarily cut off from the sea and periodically inundated with seawater at irregular intervals. Alongi (1998) alludes to coastal bodies of water in arid and semi-arid areas which sporadically receive freshwater input, such as in north-western Australia, parts of Africa, and Mexico. Tropical estuaries, where the mixing boundaries frequently are not distinct, do not conform well to Pritchard's (1967) classic definition. Kjerfve (1989, p. 3), therefore, generated a more comprehensive and viable definition of an estuary, placing it in the context of the entire coastal zone: 'An estuarine system is a coastal indentation that has a restricted connection to the ocean and remains open at least intermittently. The estuarine system can be subdivided into three regions: (a) a tidal river zone – a fluvial zone characterized by lack of ocean salinity but subject to tidal rise and fall of sea level; (b) a mixing zone (the estuary proper) characterized by water mass mixing and existence of strong gradients of physical, chemical, and biotic quantities reaching from the tidal river zone to the seaward location of a river mouth or ebb-tidal delta; and (c) a nearshore turbid zone in the open ocean between the mixing zone and the seaward edge of the tidal plume at full ebb tide.'

Numerous anthropogenic perturbations affect estuarine environments, contributing to habitat alteration and changes in the structure and dynamics of biotic communities. Environmental problems encountered in these systems invariably stem from overpopulation and uncontrolled development in coastal watersheds, as well as human activities in the estuarine embayments themselves. Nearly all estuaries are influenced in some way by anthropogenic activities, which will likely become more widespread and acute during the next 25 years, because the coastal population is expected to approach six billion people by 2025 (approximately the total global population today; Weber 1994; Hameedi 1997). More of these impacts will probably emerge in the coastal zone of developing countries in Asia, Africa, and South America, where government regulatory controls are less stringent or lacking compared to those of most developed countries.

The purpose of this article is to review the current state of the estuarine environment, assess the extent of pollution and other anthropogenic impacts in estuaries, and predict what the condition of the estuarine environment might be in the year 2025 based on examination of published literature. This work is motivated by the observation that human activities are significantly altering estuarine systems worldwide, threatening their resources and the commercial and recreational uses dependent on them. It is necessary, therefore, to identify the major anthropogenic activities impacting estuaries and to recommend remedial actions for improving the health and viability of these systems. Included among the most serious stress factors in the estuarine environment are pollution inputs (e.g. nutrient enrichment, organic carbon loading, and chemical contaminants) and several other anthropogenic factors (e.g. habitat loss and alteration, overfishing, freshwater diversions, and introduced species). It is worth examining these in detail in order to develop effective management strategies to mitigate their impacts. The importance of early detection of human-induced alteration of estuarine environments cannot be overstated, because the success of cost-effective remedial measures clearly depends on addressing the problem expeditiously before it becomes intractable.

ENVIRONMENTAL FORCING FACTORS

Contemporary anthropogenic impacts

Human activities have degraded extensive coastal habitat areas worldwide (Fig. 1) (National Research Council 1994; Viles & Spencer 1995; Alongi 1998; Kennish 2001*a*). Accelerated population growth and development in the coastal zone, accompanied by increasing urbanization and industrialization, are closely coupled to these anthropogenic impacts, which have compromised the ecological integrity of many estuaries. Approximately four billion people now inhabit land areas within 60 km of the world's coastlines, and they have placed considerable pressure on sensitive estuarine habitats. Various human activities and conflicting uses in coastal watersheds and neighbouring estuarine waters have contributed greatly to impaired water quality, habitat loss and alteration, and diminished resources (Clark 1992; Goldberg 1995; McIntyre 1995; Kennish 1997, 2000, 2001*a*, *b*). Of particular note are anthropogenic disturbances associated with urban development and tourism, land reclamation and waste disposal, water extraction and diversion, agriculture and aquaculture, oil and gas production, electric power generation and transmission, dredging and filling, marine transportation and shipping, harbour and marina operations, and recreational and commercial fishing.

Pollution remains a major problem worldwide, because a wide array of point and non-point pollution sources promotes nutrient and organic carbon loading, pathogen impairment, and chemical contamination of estuarine waters (Table 3) (Clark 1992; Kennish 1997, 2001*a*). Overharvesting of commercially and recreationally important finfish and shellfish populations can severely deplete resources beyond the limits of sustainability, thereby directly impacting humans. Introduced species frequently outcompete and eliminate indigenous forms. Physical alteration of habitats also creates persistent and serious environmental problems, such as large-scale modifications of coastal watersheds (e.g. deforestation and construction, marsh diking and impoundment, bulkheading and lagoon formation) and estuarine basins (e.g. dredging and dredged material disposal, channel and inlet stabilization, and harbour and marina development), which adversely affect estuarine organisms (Table 4). Habitat restoration programmes have been implemented in many areas to overcome some of these impacts (Zedler 2001).

Marine pollution

The Joint Group of Experts on the Scientific Aspects of Marine Pollution (GESAMP), an international committee sanctioned by the United Nations (UN), has periodically assessed the global state of the marine environment (GESAMP 1990; Kennish 1997; Pinet 2000). GESAMP has prepared reports on the health of marine waters, specifying the priority problems that need to be addressed by governing bodies worldwide (GESAMP 1993, 1995). These reports

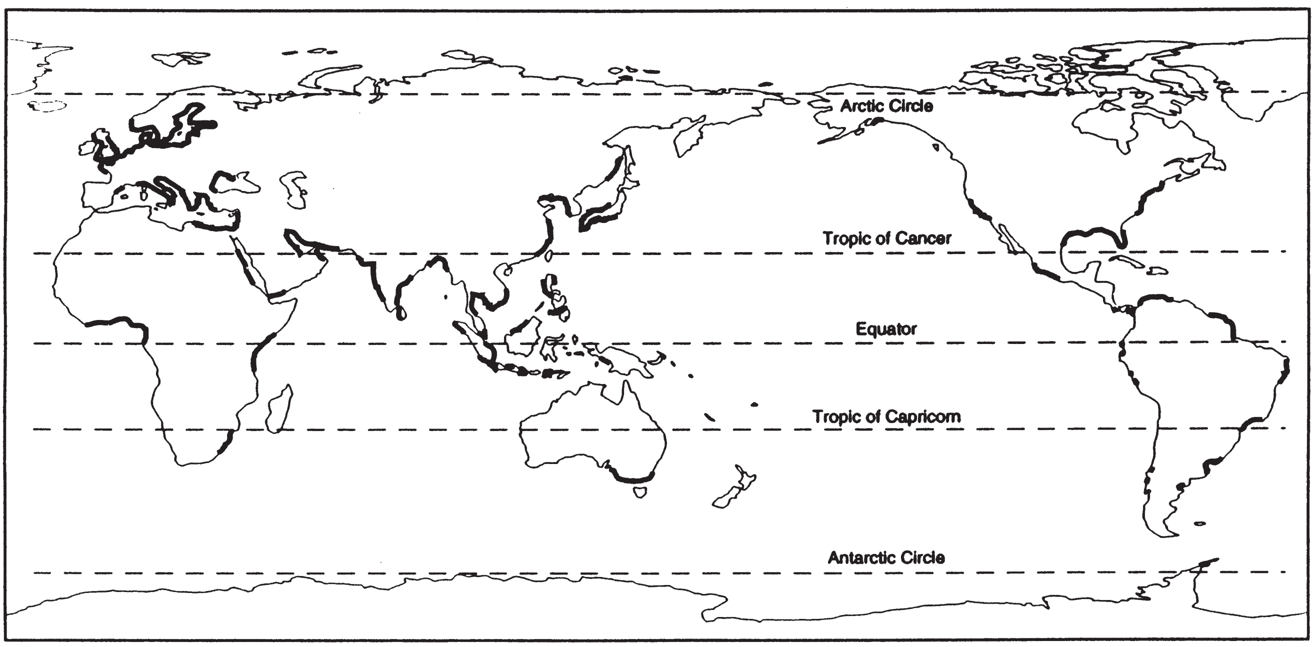


Figure 1 World map showing existing coastal areas (in bold) significantly impacted by human activities. Note coastlines bordering heavily populated, industrialized nations of the northern hemisphere exhibit more extensive impacts than those of the southern hemisphere. Escalating coastal impacts during the 21st century are expected to shift to developing nations of the tropics, where population growth and attendant urbanization are rapidly increasing (after Alongi 1998).

Table 3 Point and non-point sources of pollution in estuarine and marine waters (after US Environmental Protection Agency 1986). BOD = biochemical oxygen demand.

Sources	*Common pollutant categories*
Point sources	
Municipal sewage treatment plants	BOD, bacteria, nutrients, ammonia, toxic chemicals
Industrial facilities	Toxic chemicals, BOD
Combined sewer overflows	BOD, bacteria, nutrients, turbidity, total dissolved solids, ammonia, toxic chemicals
Non-point sources	
Agricultural run-off	Nutrients, turbidity, total dissolved solids, toxic chemicals
Urban run-off	Turbidity, bacteria, nutrients, total dissolved solids, toxic chemicals
Construction run-off	Turbidity, nutrients, toxic chemicals
Mining run-off	Turbidity, acids, toxic chemicals,
Septic systems	Bacteria, nutrients
Landfills/spills	Toxic chemicals, miscellaneous substance
Silvicultural run-off	Nutrients, turbidity, toxic chemicals

Table 4 The possible effects of physical disturbance at various levels of biological organization (after Knox 2001).

Level of organization	*Possible effects*
Individual	Increased possibility of death or injury Energetic cost of re-establishment Effect of reproductive development Effects on food availability Exposure to predation or displacement Provision of colonizable space Competitive release
Population	Changes in density Changes in recruitment intensity and/or variability Changes in dispersion patterns
Community	Changes in species diversity Changes in overall abundance Changes in productivity Changes in the patterns of energy flow or nutrient recycling

have formed the basis for policy decisions within the UN agencies, as well as at the national level (Taylor 1993). Since 1993, GESAMP has been identified as the Group of Experts on the Scientific Aspects of Marine Environmental Protection.

GESAMP (1982, p. 1) has defined marine pollution as the 'introduction by man, directly or indirectly, of substances or energy into the marine environment (including estuaries) resulting in such deleterious effects as harm to living resources, hazards to human health, hindrance to marine activities including fishing, impairment of quality for use of seawater and reduction of amenities.' By far, the greatest inputs of pollution to marine environments derive from that portion of the world's population (about 20%) living in developed, industrialized countries (Taylor 1993). The work of GESAMP and scientific investigators from different countries around the world indicates that there are six primary pathways by which pollutants enter estuarine environments: (1) non-point source run-off from land; (2) direct pipeline discharges; (3) riverine inflow; (4) atmospheric deposition; (5) maritime transportation; and (6) waste dumping at sea (McIntyre 1992, 1995; Goldberg 1995; Kennish 1997). Land-based sources of pollution predominate in estuarine and coastal marine environments, with run-off responsible for an estimated 44% of the pollutant inputs, atmospheric deposition 33%, maritime transportation 12%, waste dumping 10%, and offshore production 1% (GESAMP 1990). As noted by Kennish (1997), numerous pollutants have been reported in estuaries from these sources, most conspicuously:

(1) Excessive nutrients causing progressive enrichment and periodic eutrophication problems.
(2) Sewage and other oxygen-demanding wastes (principally carbonaceous organic matter) which promote anoxia and hypoxia of coastal waters.
(3) Pathogens (e.g. certain bacteria, viruses, and parasites) and other infectious agents often associated with sewage wastes.
(4) Petroleum hydrocarbons originating from oil tanker accidents and other major spillages, routine operations during oil transportation, effluent from non-petroleum industries, municipal wastes, and non-point run-off from land.
(5) Polycyclic aromatic hydrocarbons (PAHs) entering estuarine and marine ecosystems from sewage and industrial effluents, urban stormwater run-off, oil spills, creosote oil, combustion of fossil fuels, and forest fires.
(6) Halogenated hydrocarbon compounds (e.g. organochlorine pesticides) principally originating from agricultural and industrial sources.
(7) Heavy metals accumulating from smelting, sewage-sludge dumping, ash and dredged-material disposal, antifouling paints, seed dressings and slimicides, power station corrosion products, oil refinery effluents, and other industrial processes.
(8) Radioactive substances generated by uranium mining and milling, nuclear power plants, and industrial, medical, and scientific uses of radioactive materials.
(9) Thermal loading of natural waters, owing primarily to the discharge of condenser cooling waters from electric generating stations.

(10) Debris/litter and munitions introduced by various land-based and marine activities.
(11) Fly ash, colliery wastes, flue-gas desulphurization sludges, boiler bottom ash, and mine tailings.
(12) Acid mining wastes.
(13) Drilling muds and cuttings.
(14) Pharmaceuticals and alkali chemicals.
(15) Pulp and paper mill effluents.
(16) Suspended solids, turbidity, and siltation/sedimentation.

Among the most severe pollution problems observed in estuaries today are those ascribable to nutrient enrichment, organic carbon loading (e.g. sewage), oil spills, and toxic chemicals (e.g. PAHs, halogenated hydrocarbons and heavy metals). Nutrient enrichment, organic carbon loading, and pathogen inputs can significantly compromise water quality. Oil spills directly degrade habitats. Toxic chemicals, in turn, endanger the health of estuarine organisms, causing lethal as well as sublethal impacts on biotic communities.

The accumulation of floatable debris in estuarine waters and litter along estuarine shorelines is an escalating problem. Debris/litter is not only aesthetically displeasing but also may be hazardous to organisms that ingest or become entangled in it. Seabirds and wildlife entangled in debris/litter are very susceptible to predation, suffocation, and drowning (Shaw & Day 1994).

Nutrient enrichment and organic carbon loading. Anthropogenic nutrient enrichment and organic carbon loading have been linked to the eutrophication of estuarine waters (Dederen 1992; McComb 1995; Nixon 1995; National Estuary Programme 1997*a*, *b*; Valiela *et al.* 1997; Smith *et al.* 1999). Enhanced eutrophication or hypereutrophication can cause serious imbalances in the trophic structure of these waters (Ingrid *et al.* 1996; Livingston 1996, 2000). The influx of nutrient elements (nitrogen and phosphorus) from anthropogenic activities stimulates primary production of microphytes and macrophytes, leading in some cases to excessive plant growth and elevated biomass that culminate in widespread and recurring hypoxia ($<2\,\mathrm{mg\ l^{-1}}$ dissolved oxygen) and anoxia ($0\,\mathrm{mg\ l^{-1}}$ dissolved oxygen) due to heightened benthic respiration. Other adverse effects associated with increased nutrient supply and accelerated primary production, as well as organic carbon enrichment (e.g. sewage, mariculture, and seafood processing wastes) are periodic toxic or nuisance algal blooms, shading effects, build up of toxins (e.g. sulphides), mortality of benthic and pelagic species, reduced biodiversity, diminished secondary production, the diminution of recreational and commercial fisheries, as well as altered species composition, abundance, and distribution (Weston 1990; Costello & Read 1994; Alongi 1998; Kennish 1998*a*, 2000; Holmer 1999; Howarth *et al.* 2000). Shading produces unfavourable conditions for bottom-dwelling plants and commonly causes a reduction in the areal coverage of submerged aquatic vegetation (e.g. seagrasses), which serves as critical benthic habitat in many shallow estuarine systems.

Overfertility is a major problem in coastal areas worldwide, and the number of estuarine systems experiencing oxygen deficiency in bottom waters due to this effect is increasing (Holmer 1999). Case studies of estuarine eutrophication presented for the Dutch Wadden Sea, the Netherlands (De Jonge 1990; Kennish 1998*a*), Harvey-Peel Estuary, Australia (McComb 1995), Chesapeake Bay, Maryland, USA (Malone *et al.* 1996), and Perdido Bay, Florida, USA (Livingston 2000) convey the global seriousness of this problem. Livingston (2000) has demonstrated unequivocally in long-term studies how anthropogenic nutrient loading in estuarine waters of the north-eastern Gulf of Mexico causes phytoplankton blooms, destabilization of plankton populations, unbalanced food webs, and general reduction of useful productivity. As more people settle in coastal watersheds, the severity of nutrient enrichment in estuarine waters escalates, persisting at some locations for many years (Balls *et al.* 1995). These impacts are most acute in shallow, poorly flushed systems.

Estuaries receive most nutrients via allochthonous transport systems, notably surface water inflows (streams, rivers, and direct land run-off), groundwater discharges, and atmospheric deposition (both wet and dry deposition). A large fraction of the nutrients derives from multiple allochthonous anthropogenic sources (e.g. fertilized lawns and farmlands, municipal and industrial wastewaters, combined sewer overflows, and malfunctioning septic systems). The release of untreated or partially treated sewage can significantly increase nutrient levels in estuaries. This waste input also degrades water quality by raising the biochemical oxygen demand, promoting eutrophication problems, and delivering chemical contaminants and pathogenic microorganisms (i.e. bacteria, viruses, protists and helminths) to estuarine systems (Chapman *et al.* 1996; Kennish 1997; Eganhouse & Sherblom 2001). Pathogens from sewage pose a serious threat to human health, being responsible for cholera, hepatitis, as well as gastroenteric diseases; hence, they are the targets of water quality monitoring programmes for assessment of the closure of bathing beaches and shellfish growing waters.

Chemical contaminants. There are three major classes of chemical contaminants which rank among the most dangerous in terms of potential impacts on estuarine organisms: (1) PAHs; (2) halogenated hydrocarbons; and (3) heavy metals (Kennish 1997, 2001*a*). Some of the substances comprising these classes are widespread, persistent, and detrimental to most biotic groups. They tend to bioaccumulate in aquatic organisms, and in the case of certain compounds (e.g. the organochlorines DDT and PCBs) biomagnify in food chains to reach highest concentrations in top carnivores of the system. Toxic substances which undergo biomagnification in estuarine food chains also pose a potential health threat to humans who consume contaminated seafood. Various estuarine fisheries have been closed because of unacceptably high heavy metal and organochlorine contaminant concentrations in seafood products and habitats (Lalli & Parsons 1993). Mercury is a particularly problematic toxic agent in estuaries around the world. In the USA, it has become a serious concern in systems along the mid-Atlantic and New England coasts, as well as the Gulf of Mexico and in California (Locarnini & Presley 1996; Kannen *et al.* 1998; Robert J. Livingston, Florida State University, personal communication 2001). The National Oceanic and Atmospheric Administration's (NOAA's) National Status and Trends Programme has documented elevated concentrations of mercury at a number of estuarine sites in the USA (O'Connor & Beliaeff 1995, Fig. 2).

Estuarine organisms exposed to high levels of chemical contaminants exhibit an array of adverse responses (Table 5). As a result, biotic communities affected by the contaminants can experience significant changes, including the loss of rare or sensitive species, decreased species abundance, shifts in the age structure of populations, and altered trophic interactions (Howells *et al.* 1990). Entire ecosystems can be disrupted by these changes.

Low-molecular-weight (one to three ring) PAH compounds are acutely more toxic than high-molecular-weight (four to seven ring) PAHs which have greater carcinogenic, mutagenic, and teratogenic potential for a wide variety of organisms (Eisler 1987; Kennish 1992; Fernandes *et al.* 1997). Although bacteria, fungi, and algae naturally synthesize some PAHs, most of the contaminants in this class originate from anthropogenic sources (e.g. oil spills, municipal and industrial discharges, marinas, fossil-fuel combustion and waste incineration). Polluting oil is a significant source of PAHs in estuarine environments. The pyrolysis of organic matter represents a primary pathway of PAH genesis and explains, in part, the wide distribution of PAHs in aquatic environments (Wild & Jones 1995). The concentration of PAHs in estuarine organisms is largely a function of the bioavailability of the contaminants and the capacity of the organisms to metabolize them. PAHs invariably attain high levels in bottom sediments of urbanized systems, where they may persist relatively unaltered for long periods. These contaminants have been shown to have deleterious effects on the structure and function of benthic communities (Carman *et al.* 1995; Kennish 1998*a*; Bennett *et al.* 2000). The occurrence of fish immune syndrome has been attributed to PAH exposure.

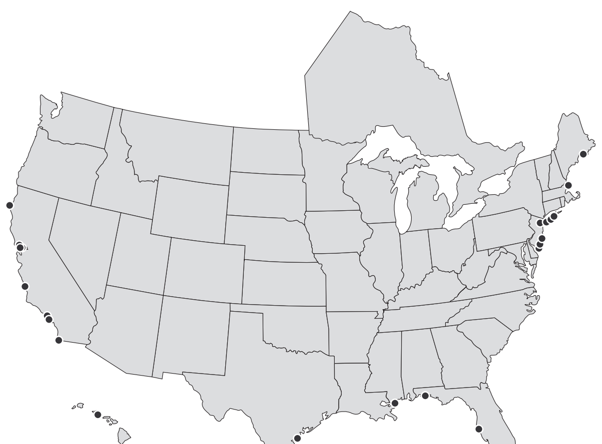


Figure 2 Map of the USA displaying sites of elevated mercury concentrations in estuarine systems based on analysis of whole soft-parts of mollusc (mussel and oyster) samples by NOAA's National Status and Trends Programme. Locations of 'high' mercury concentrations include those with mean contaminant levels exceeding 24 $\mu g\ g^{-1}$ in the samples. Note the heavily populated north-east region, Gulf of Mexico, and California coast have the most sites with elevated mercury measurements (after O'Connor & Beliaeff 1995).

The halogenated hydrocarbons or organochlorines include some of the most toxic synthetic organic compounds found in estuarine environments. Among the notable contaminants in this group are a number of insecticides (e.g. DDT, aldrin, chlordane, dieldrin, heptochlor, mirex and toxaphene), herbicides (e.g. chlorophenoxy compounds, 2,4-D and 2,4-T), and industrial chemicals (e.g. PCBs, chlorinated dibenzo-*p*-dioxins and chlorinated dibenzofurans). They enter estuaries via multiple routes such as industrial and municipal wastewater discharges, urban and farmland run-off, and atmospheric deposition (Dickhut & Gustafson 1995; Kennish 1997; Wania *et al.* 1998). The high-molecular-weight halogenated hydrocarbons are lipophilic and readily accumulate in tissues of estuarine organisms. They appear to be responsible for a wide range of diseases in these organisms (e.g. blood disorders, immunosuppression, altered endocrine physiology, aberrant developmental patterns, reproductive abnormalities, skin and liver lesions, and cancer; Kennish 1992).

Heavy metals (e.g. copper, cobalt, lead, mercury, and zinc) are a concern because of their toxicity to estuarine organisms above a threshold availability. They have been implicated in the development of the following biotic disorders: (1) feeding, digestive, and respiratory dysfunctions; (2) aberrant physiological, neurological and reproductive activities; (3) tissue inflammation and degeneration; (4) neoplasm formation; and (5) genetic derangement (Kennish 1992, 1997). Heavy metals in estuaries derive from both natural and anthropogenic sources. Anthropogenic inputs to estuarine waters originate from mining and smelting operations, refining and electroplating, dye and paint manufacture, and fossil-fuel burning. River discharges, urban run-off, and atmospheric deposition transport the bulk of the total heavy metal burden that accumulates in estuaries (Bryan & Langston 1992; Atrill & Thomas 1995; De Groot 1995; Bothner *et al.* 1998; Kennish 1998*b*; Rozan & Benoit 1999; Cearreta *et al.* 2000; Turner 2000).

Of all trace metals, mercury is most toxic. Methylmercury, the organometal form, poses a particular danger to estuarine and marine organisms, as well as humans who consume contaminated seafood, because it readily bioaccumulates in aquatic food chains (Wren *et al.* 1995). Adverse

Table 5 Responses of marine organisms to chemical contaminant exposure (after McDowell 1993).

Level	*Effects*	*Responses*
Cell	Toxication	Toxic metabolites available
	Metabolic impairment	
	Cellular damage	Disruption in energetics and cellular processes
	Detoxication	Adaptation
Organism	Physiological changes	Reduced population performance
	Behavioural changes	
	Susceptibility to disease	
	Reduced reproductive effort	
	Decreased larval viability	
	Readjustment in rate functions	Population regulation and adaptation
	Altered immunities	
Population	Changes in age/size, structure, recruitment, mortality, biomass	Negative impacts on species productivity as well as coexisting species and communities
	Adjustment of reproductive output and other demographic characteristics	Adaptation of population
Community	Changes in species abundance, species distribution and biomass	Replacement by more adaptive competitors
	Altered trophic interactions	Reduced secondary production
	Ecosystem adaptation	No change in community structure and function

effects in aquatic biota occur at low parts per billion levels of exposure to mercury. Chronic toxicity to estuarine and marine organisms takes place at concentrations in seawater as low as 1 ppb (Monroe & Kelly 1992). At elevated concentrations, mercury acts as an acute enzyme inhibitor and can impact entire biotic communities (Kennish 1992). Mercury is ubiquitous throughout many estuarine and marine ecosystems, being derived from a wide range of industrial uses, from the combustion of fossil fuels, and from natural sources such as the weathering of mercury-bearing rocks and ores (e.g. cinnabar), the fallout of atmospheric gases from volcanoes and geothermal vents, and the emissions of deep-sea hydrothermal vents (Wren *et al.* 1995; Kennish 1997).

Such chemical contaminants can be found in the sea surface microlayer (i.e. neuston layer), water column, bottom sediments, and organisms of estuaries. They rapidly attach to fine-grained sediments ($<63\ \mu m$ in size) and other particulates (Kennish 2001*a*), often being sorbed to or occluded within the hydrogenous and biogenic phases which coat the particles (Thomas & Bendell-Young 1999; Turner 2000). The sediment-sorbed contaminants tend to accumulate in estuarine bottom sediments which serve as a major repository for the contaminants. They may be later remobilized by various natural processes (e.g. bottom currents, waves and storms) and anthropogenic activities (e.g. dredging, shipping, motorized boating and commercial fishing). Extensive investigations of sediment contamination and toxicity have been conducted in estuaries in the USA (Daskalakis & O'Connor 1995; Long *et al.* 1996; Kennish 1997; Thompson *et al.* 1999; O'Connor & Paul 2000). In heavily-contaminated areas of some urbanized estuaries in the USA (e.g. Boston Harbour, western Long Island Sound, Hudson-Raritan Estuary and Puget Sound), contaminant accumulation in bottom sediments has produced acute as well as chronic impacts on the resident biota (Kennish 1997). Similar impacts have been reported on biotic communities in contaminated sediments of European estuaries such as the Tees Estuary and Tyne Estuary, UK (Tapp *et al.* 1993; Matthiessen 1998) and the Bay of Cádiz, Barbate River Estuary, Odiel River Estuary and Bilbao Estuary in Spain (Drake *et al.* 1999; Cearreta *et al.* 2000).

Other anthropogenic effects

Fisheries exploitation. The overexploitation of fisheries resources in estuarine and coastal marine waters is a growing concern because of the widespread decline in finfish and shellfish populations of recreational and commercial importance (Botsford *et al.* 1997) and because of the potential imbalances arising in ecosystem function and overall community structure (Jennings & Kaiser 1998; Pinnegar *et al.* 2000). Jennings and Lock (1996) and Hall (1999) have reviewed the effects of fishing on the structure and function of marine ecosystems. Pinnegar *et al.* (2000) have discussed how widely trophic cascades in benthic marine systems can be expected to occur when marine communities are subject to intense harvesting.

The size and age structures of exploited fish populations change in time because of the preferential removal of larger and older individuals (Goñi 1998). Fishing reduces population abundance. When a fishery declines dramatically or collapses, it is often because of overharvest. The cessation of fishing, in contrast, increases abundance due to population protection. This process leads once again to changes in the demographic structure of the previously-exploited populations, shifting it towards bigger and older individuals (Sánchez Lizaso *et al.* 2000).

Declining fisheries have been documented in many estuaries along the Pacific, Gulf of Mexico, and Atlantic coasts during the past several decades. For example, major declines in fisheries of San Francisco Bay have been observed since the 1970s, as exemplified by the depletion of delta smelt (*Hypomesus transpacificus*), chinook salmon (*Oncorhynchus tshawytscha*), and striped bass (*Morone saxatilis*) stocks (Monroe & Kelly 1992; San Francisco Estuary Project 1998). Blue crab (*Callinectes sapidus*) landings have dropped significantly in Corpus Christi Bay (National Estuary Programme 1997*c*). In Sarasota Bay, Florida, landings of sea trout (*Cynoscion arenarius* and *C. nebulosus*) have decreased by 50% over the past 40 years. Substantial declines in fisheries have also been noted for the Florida Bay-Florida Keys estuaries (Robert J. Livingston, Florida State University, personal communication 2001). Eight species of commercially and recreationally important finfish and shellfish also appear to be overfished in the Albemarle-Pamlico Sound system of North Carolina (Kennish 2000). Marked declines in shellfish harvests (oysters and clams) have been reported in Chesapeake Bay, Maryland, the Delaware Estuary, Delaware, and the Barnegat Bay-Little Egg Harbor Estuary, New Jersey during the past 50 years (John Kraeuter, Rutgers University, personal communication 2001). Management strategies are being formulated to restore depleted stocks in these and other estuaries and to re-establish viable fisheries that will ensure resource sustainability for future generations.

Introduced species. Another emerging problem in estuaries is the introduction of exotic species which can lead to reduced species diversity in estuarine communities, shifts in trophic organization, infiltration of detrimental pathogens, and alteration of habitats. Introduced species, which often enter new environments on the hulls and in the ballast water of ships, frequently lack natural controls in their adopted estuarine habitats; therefore, they often outcompete native populations to dominate the communities. Other species are intentionally introduced to some environments for mariculture purposes and to establish new recreational fisheries. In the case of aggressive, non-native plant species which invade wetlands and shallow submerged estuarine habitats, secondary impacts that may arise include changes in run-off and erosion, modification of current flow, and alteration of nutrient cycles (Kennish 2000). San Francisco Bay is one of the most heavily invaded estuaries in the world, with nearly 250 species of plants and animals having been introduced (Cohen & Carlton 1998). One notable species introduced here from the Atlantic coast of the USA is the striped bass (*Morone saxatilis*). Introduction of the Asian clam (*Potamocorbula amurensis*) has been particularly problematic because it has decimated the phytoplankton community in extensive areas of the estuary (Cohen & Carlton 1998; San Francisco Estuary Project 1998).

The intentional or accidental introduction of invasive species has also become a serious ecological concern in many other estuaries in the USA (National Estuary Programme 1997*b*, *c*; Kennish 2000). In Puget Sound and Padilla Bay, Washington, for instance, management efforts have been implemented to control the spread of the smooth cordgrass (*Spartina alterniflora*), because of its potential negative effects on native flora and fauna. The edible brown mussel *(Perna perna*), an invasive species in Corpus Christi Bay, Texas, has created significant fouling problems on natural and man-made structures. The introduction of numerous plant species in the Barataria-Terrebonne Estuary, Louisiana, has contributed to substantial habitat loss and alteration and reduced biodiversity in the system. Coastal marsh habitat losses in the Gulf of Mexico increased markedly in the 1930s after introduction of the nutria (*Myocastor coypu*), a native South American mammal (Day *et al.* 1989). The encroachment of the invasive Brazilian pepper plant on native mangroves and other wetland populations remains a formidable problem in Charlotte Harbor and the Indian River Lagoon, Florida. The spread of the invasive common reed (*Phragmites australis*) in marsh habitats of Delaware Bay threatens indigenous flora. The introduction of a parasitic protist (*Haplosporidium nelsoni*) in the Delaware Estuary during the 1950s decimated the American oyster (*Crassostrea virginica*) fishery. The invasion of two subtropical shipworm species (*Teredo bartschi* and *T. furcifera*) in Barnegat Bay, New Jersey, during the 1970s and 1980s caused serious damage to untreated wooden structures (Kennish & Lutz 1984).

While the introduction of invasive species to estuarine systems worldwide continues unabated, more data are needed to effectively assess their long-term environmental impacts. In some estuaries, such as San Francisco Bay, most of the invertebrate taxa are now introduced forms. It is difficult to determine the cumulative effects of the invasive species on the indigenous communities of these impacted systems (Nybakken 1988). However, the introduced forms have the potential to significantly reduce population abundances and biodiversity of the native communities.

Freshwater diversions. Changes in natural flow regimes in riverine-estuarine systems throughout the world are commonly due to multiple human activities in nearby watershed areas. For example, increased coastal development invariably leads to greater impervious cover which accelerates stormwater run-off. Destruction of wetland habitat and channelization for purposes of flood control likewise alter freshwater flows to estuaries. Further inland, the construction of dams and reservoirs in upland drainage basins also modifies freshwater discharges to the coastal zone. Other causes of changes in freshwater flows to estuaries include catchment and retention structures, groundwater withdrawals, and wastewater discharges. Stochastic natural disturbances (e.g. prolonged droughts) can exacerbate the detrimental effects of freshwater deprivation associated with upstream anthropogenic (consumptive) water use.

River diversions greatly reduce freshwater inputs to some estuaries (e.g. San Francisco Bay). These surface water diversions are usually constructed to satisfy agricultural, municipal, and industrial demands for fresh water, but ecological impacts may be acute. For example, decreased freshwater inflow can significantly change the hydrologic, salinity, and sediment regimes as well as the nutrient loadings of an estuary, which directly affects habitat areas, the abundance and distribution of estuarine organisms, and trophodynamics of the system.

In Galveston Bay, Texas, stormwater run-off and surface water withdrawals in watershed areas are the principal factors effecting change in freshwater inflows. The main causes of freshwater flow reduction to Tampa Bay, Florida, are hydrologic modification of numerous tidal creeks and the damming of four major rivers for flood control and water supply development. The diminution of freshwater flow to Sarasota Bay, Florida, is largely attributable to surface and groundwater withdrawals, which are also the principal factors contributing to the flux of freshwater flow to the Albemarle-Pamlico Sound system, North Carolina, and the Delaware Estuary, Delaware. Hydrologic modifications (channelization) and surface water withdrawals are the chief reasons proffered for changes in freshwater flow to Narragansett Bay (National Estuary Programme 1997*b*).

The Apalachicola River and Bay system along the northwest coast of Florida provides an excellent example of how altered freshwater inflows can dramatically affect the biological organization and production of an estuary. Livingston (1997, 2000, 2001) and Livingston *et al.* (1997, 2000) have conducted a long-term (approximately 15 year) study of this system, assessing the trophic structure and production of the estuary within the context of changes in Apalachicola River flow. They determined that the trophodynamics and production of the estuary depend on a combination of variables which are either directly (salinity, light penetration, nutrients) or indirectly (trophic relationships) associated with freshwater inputs modified by wind, tidal factors, and the physiography of the bay (Livingston *et al.* 1997). Acute reduction in river inflow through human activities has the potential to permanently alter the estuarine food web structure, as manifested by the failure of specific estuarine populations (Livingston 1997). Since many other river-estuarine systems worldwide are subject to similar hydrologic alterations ascribable to an array of human activities, diminished river flow and its impact on biotic organization in affected estuarine waters may be an emerging global problem.

Shoreline development and dredging. The most direct physical impacts on estuaries are those associated with the construction of shoreline structures (e.g. bulkheads, boat ramps, docks and piers), activities at marinas and harbours, and dredging to maintain navigable waterways. Recreational activity at boat ramps, docks, piers and other structures produces localized impacts in estuaries. Boat and ship traffic at marinas, in harbours, and along shipping channels (e.g. Houston Ship Channel) generally modifies bottom habitats over widespread areas due to prop scarring and engine emissions of chemical contaminants (e.g. uncombusted fuel/oil mix, PAHs and other non-aromatic structures), which accumulate in the benthos. Some heavy metals (e.g. arsenic, copper and chromium) diffuse from wooden shoreline structures chemically treated with chromated copper arsenate, and these metals likewise concentrate in nearby estuarine bottom sediments (Weis *et al.* 1993).

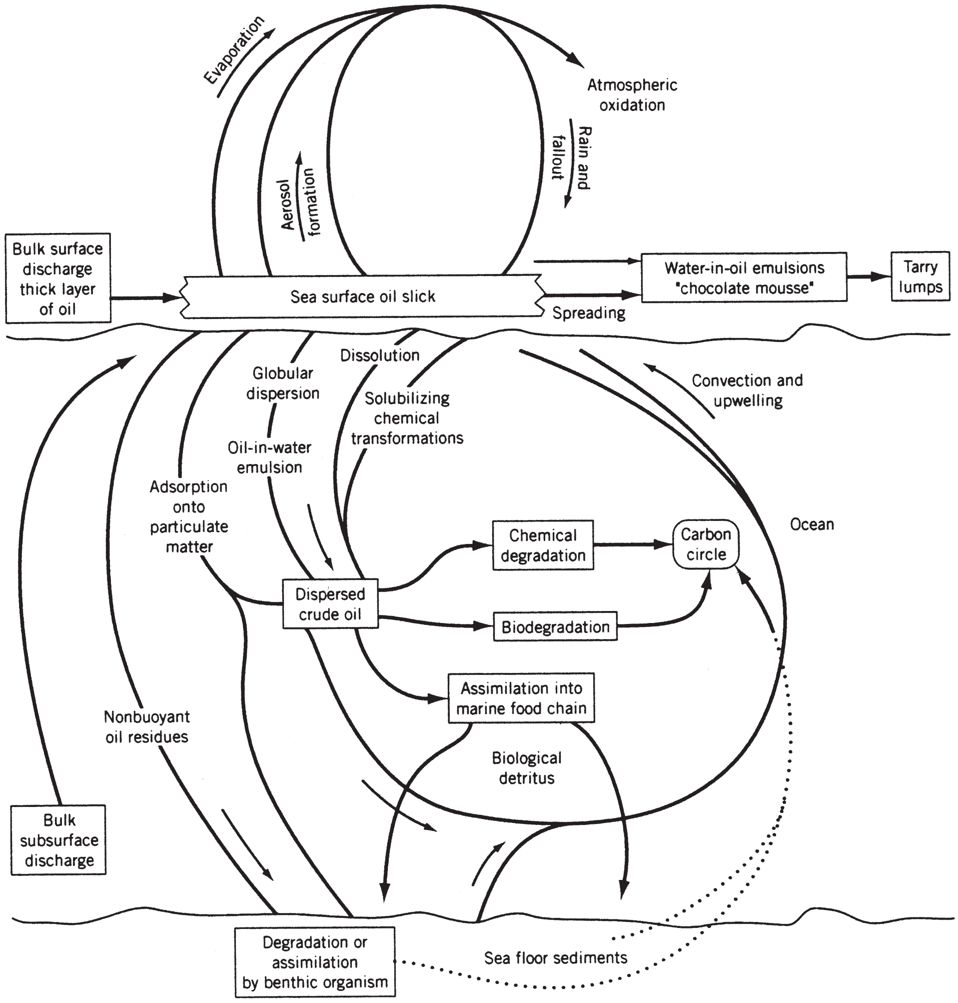
Along heavily developed shorelines in particular, motorized watercraft also adversely affect estuarine systems by increasing erosion and sediment resuspension, as well as remobilizing sediment-bound nutrients and chemical contaminants via propeller and pressure wave effects. Heavy metals may diffuse from antifouling paints and primer bases used on boat and ship hulls. All of these factors degrade water quality conditions. The accumulation of chemical contaminants in bottom sediments and their remobilization in the water column can cause both lethal and sublethal impacts on organisms when they occur at sufficiently high levels and are readily bioavailable. In addition, sediment resuspension by watercraft attenuates light, although this effect is usually ephemeral, except during periods of heavy vessel traffic.

The input of oil via marine transportation activities, leakages of fixed installations (e.g. refineries and marine terminals), urban and suburban run-off, and oil spills is hazardous to estuarine organisms and habitats (Kennish 1992, 1997; Pezeshki *et al.* 2000). It affects all components of estuarine systems (Fig. 3). The impacts are directly coupled to physical (smothering and reduced light), habitat (altered pH, decreased dissolved oxygen, and decreased food availability), and toxic actions of the polluting oil. They may persist for decades at locations of major oil spills, such as heavily oiled wetland sites.

There are positive and negative effects associated with dredging and dredged material disposal in estuarine environments. The beneficial effects include: (1) increased nutrients that can enhance productivity of a system; (2) improved circulation; (3) increased recreational and commercial usage of the water body; and (4) sediment supply for beach nourishment, landfill projects, and soil improvement. The adverse effects are: (1) destroyed bottom habitats; (2) impaired water quality; and (3) direct or indirect impacts on organisms, most conspicuously those inhabiting the benthos (Kennish 1997).

Adverse effects of dredging clearly involve the removal of bottom sediments, which destroys the benthic habitat. Operation of the dredge also increases mortality of benthic organisms by mechanical injury of the dredge, as well as by the smothering of sediment during dredging and dredged material disposal. Recolonization of a dredged site is usually protracted, often requiring a year or more. The impairment of water quality associated with dredging and dredged material disposal is ascribed to the remobilization of nutrients and chemical contaminants from the benthos (Kennish 1997, 2001*a*).

Figure 3 Physical, chemical, and biological processes which affect the distribution and accumulation of polluting oil in estuarine environments. Most notable among these processes are evaporation, dissolution, photochemical oxidation, biodegradation (bacteria, fungi and yeast), advection and dispersion, emulsification, and sedimentation. Many oil components ultimately accumulate in bottom sediments and organisms, where they can pass through food chains (after Burwood & Speers 1974).



Habitat loss and alteration. In many estuaries worldwide, the physical modification and destruction of habitat exert an even greater impact than pollution on ecosystem health. Anthropogenic activities in coastal watersheds are pervasive, and some have deleterious consequences for estuarine biotic communities. Changes in the communities frequently can be traced directly to activities in estuarine watersheds and along the estuarine shoreline. Others develop in response to physical disturbances in estuarine embayments themselves.

A number of anthropogenic activities and structures are particularly damaging to estuarine habitats. For example, poorly planned domestic and industrial construction not only destroys natural habitat in estuarine watersheds but can also increase non-point source pollution to estuarine waters. Where development is most extensive, large areal coverage of impervious surfaces facilitates surface run-off, which promotes pollutant transfer to waterways. Construction and other anthropogenic activities in catchments may hasten sediment loading which can impact estuarine benthic communities. For example, in Tasmania, Edgar and Barrett (2000) determined that anthropogenic activities in catchments covering an area of about 1 000 km^2 have accelerated silt loading to neighbouring estuaries, thereby converting sandflats to mudflats and altering the benthic community structure to one dominated by infaunal species. Along other coasts, in contrast, dam construction and river diversions have ameliorated sediment delivery to coastal systems (Aubrey 1993). Effective, long-term development planning in coastal watersheds can minimize habitat destruction in estuarine systems.

The hydrology of some wetlands has been modified by an array of human-mediated structural changes that also influence water quality and salinity regimes in nearby estuarine waters. Diking, ditching, canal construction and impoundments in coastal marshes invariably interfere with normal tidal flooding and drainage to estuaries and contribute to the deterioration of marsh plant communities and their filtering functions. Detrital inputs to adjacent tidal creeks and embayments likewise may be altered. In addition, dredge-and-fill activities directly convert wetland areas to open water or upland habitats, and they generate higher turbidity levels in tidal creeks and nearshore estuarine waters, culminating in secondary impacts on biota. Boesch *et al.* (1994) ascribed more than 25% of the total coastal wetland loss in Louisiana between 1955 and 1978 to these types of modifications.

The reclamation of wetlands for agriculture and other human needs results in critical habitat and economic losses associated with declining recreational and commercial fisheries in adjacent estuarine and coastal marine waters. Wetland reclamation eliminates valuable finfish spawning, feeding and nursery grounds. Approximately 50% of the original tidal saltmarsh habitat in the USA has been destroyed by human activities (e.g. impounding and draining for agriculture; Kennish 2001*b*). In an extreme case, more than 95% of the wetland habitat surrounding San Francisco has been lost since 1950, mainly due to human impacts (Pinet 2000). Anthropogenic activities along the mainland coast of the Dutch Wadden Sea have been responsible for almost the complete disappearance of natural saltmarshes that were extensively diked and drained (i.e. converted to polders) in past centuries. Hence, much of the original mudflat and saltmarsh habitat along the Wadden Sea has been reclaimed by physical alteration of the coastal marshlands. In addition, reclamation, silviculture, and mariculture have accounted for a loss of about 75% of the mangrove forests in Puerto Rico and more than 50% of the mangrove forests in south-east Asia (Eisma 1998).

On a regional scale, coastal subsidence coupled with groundwater, oil and gas withdrawal, as well as sediment compaction have caused rapid estuarine shoreline retreat in some locations (e.g. Galveston Bay). Subsidence in the Dutch Wadden Sea has been attributed to gas withdrawal. Along the Louisiana coast, subsidence is generally rapid because of sediment compaction and downwarping of underlying crust (Finkl 1994). Global sea level rise is also a factor in estuarine shoreline retreat along the northern coast of the Gulf of Mexico (Kennish 2001*b*). The net effect of the shoreline retreat has been the submergence of extensive wetland areas and the conversion of marsh habitat to an open water system. For example, nearly 60% of the wetland loss observed along the northern Gulf of Mexico is attributed to this process (White & Tremblay 1995). According to Eisma (1998), relative sea level rise along the Louisiana coast increased from a net of 1–2 mm yr^{-1} to 1 cm yr^{-1} or more during the 20th century, accounting for an annual loss of up to 73 km^2 of wetland area (Fig. 4).

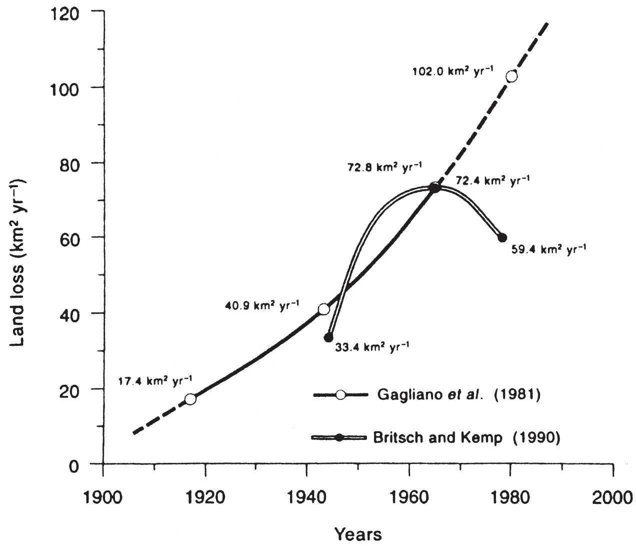


Figure 4 Long-term loss of coastal wetlands recorded during the 20th century in the Mississippi Delta Plain due to a relative sea level rise from 1–2 mm yr^{-1} to 1 cm yr^{-1} (after Britsch & Kemp 1990; Eisma 1998).

Habitat restoration. Efforts are underway in many estuarine systems worldwide to restore degraded habitats. Ecological restoration is defined as the process of repairing damage caused by human disturbance to the diversity and dynamics of indigenous ecosystems (Jackson *et al.* 1995). In estuarine systems, restoration efforts have focused on fringing wetlands (saltmarshes, mangroves and seagrasses), where environmental degradation is often most severe. The principal goal of these restoration efforts has been to return an altered or disturbed habitat to a previously existing natural condition. Some degree of habitat manipulation is necessary, because restoration of wetlands entails the re-creation of a specific vegetation association at a site (Kennish 2000). Attempts to restore these damaged habitats are not always successful because of the complex structure and function of the habitats and the difficulties of dealing with the vagaries of multiple environmental factors that affect them.

Coastal wetland habitats worldwide have been subject to considerable anthropogenic impacts associated with agricultural, domestic, industrial, and recreational activities. Natural habitat conversion here usually leads to a reduction in biodiversity (Dobsen *et al.* 1997). In the USA alone, the loss of wetlands occurred at a rate of approximately 225 000 ha yr^{-1} between the mid-1950s and mid-1970s (Ciupek 1986; Moy & Levin 1991). Enactment of major environmental legislation (i.e. National Environmental Policy Act of 1969; Marine Protection, Research, and Sanctuaries Act of 1972, and Clean Water Act of 1977) sharply curtailed this habitat destruction in the 1980s and 1990s.

Coastal tidal wetlands have been altered frequently by the construction of levees, dikes, impoundments, and ditches, as well as dredge-and-fill operations and inappropriate development. These actions have not only physically destroyed many hectares of sensitive habitat area but have also modified hydrologic conditions, causing changes in wetland flooding and drying that affect water quality and biotic communities (Turner & Lewis 1997). The most successful restoration and enhancement projects in the USA have involved saltmarsh systems. Various mitigation methods have been employed in these settings such as marsh creation through transplantation or enhancement processes of natural revegetation, restoration of filled marshes to intertidal elevations, re-establishment of tidal flow to impounded areas, mitigation banking, and increased accessibility of estuarine and marine organisms to marsh areas (Moy & Levin 1991; Zedler 2001).

By comparing the flora and fauna of the project site with those of the natural community as a reference, the success of a restoration programme can be effectively assessed. The

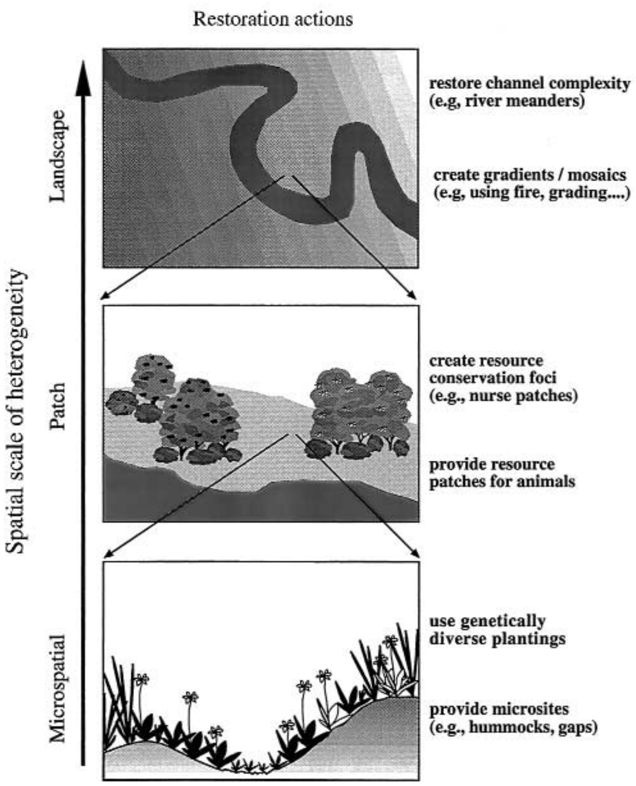


Figure 5 Restoration actions in coastal wetland systems focusing on natural habitat heterogeneity at different spatial scales (microspatial, patch and landscape) that influence key ecological processes and accelerate wetland development. A successful restoration framework should consider ecological processes and structures at a variety of spatial scales (after Vivian-Smith 2001).

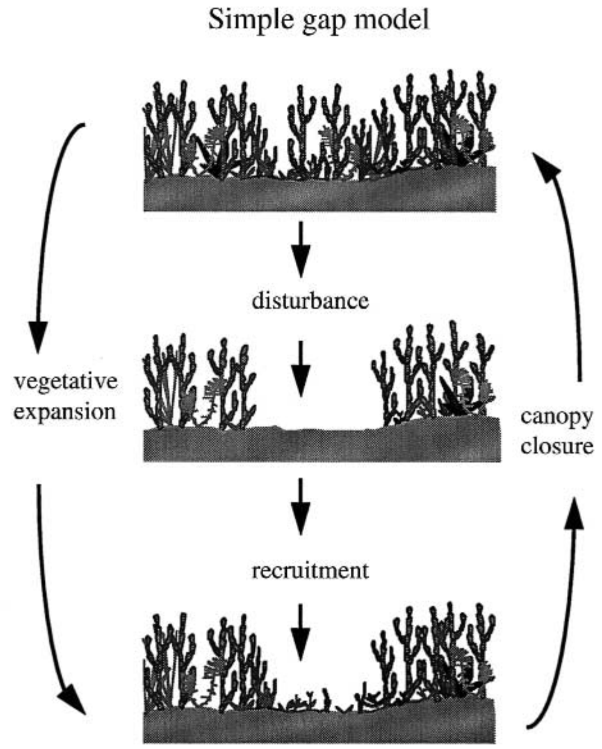


Figure 6 Saltmarsh canopy gap formation and eventual closure by the early colonization of a gap-favouring species, such as *Cordylanthus maritimus*. The colonization and expansion of gap-favouring species are key elements for restoring damaged habitat in coastal wetlands (after Vivian-Smith 2001).

greater the similarity of the communities in composition and function, the higher is the probability of restoration success (Lewis 1994). According to Vivian-Smith (2001), a restoration framework should consider ecological processes and structure at different spatial scales. Heterogeneity increases the probability that some optimal habitat will exist at the restoration site for an intended species. Observations should be made at scales less than 5 m (microspatial and patches), as well as at larger landscape scales encompassing a number of habitats (e.g. mudflat, channel and marsh, Fig. 5).

Zedler (1996) stresses the need for landscape-level approaches to habitat protection and restructuring. Wetland restoration requires the careful formulation of strategies to deal with the constraints imposed by environmental variation. It also necessitates vigilant, long-term tracking of wetland restoration efforts (Zedler & Callaway 1999).

An understanding of gap dynamics is valuable for restoring diversity of degraded wetland habitat. For example, Vivian-Smith (2001) has demonstrated that the recolonization of gaps in disturbed saltmarsh habitat can be facilitated by the introduction of gap-favouring species, such as saltmarsh bird's-beak (*Cordylanthus maritimus*) (Fig. 6). The growth of *C. maritimus* enables canopy gaps to be filled in relatively short periods of time, promoting rapid stabilization of the habitat.

The faunal colonization of constructed wetland habitat is often more problematic. For instance, Minello and Webb (1997) recorded lower infaunal density and species richness in constructed *Spartina alterniflora* marshes than in reference marshes up to 15 years after their construction. Sacco *et al.* (1994) also reported lower faunal densities in six constructed *S. alterniflora* marshes (1–17 years old) than at reference sites in the Newport River Estuary, North Carolina. Scatolini and Zedler (1996) and Williams and Desmond (2001) likewise documented lower epifaunal abundance in four-year-old constructed *S. foliosa* marshes in San Diego Bay than in nearby natural marshes. These studies reflect the difficult challenges involved in attempting to restore degraded wetland habitat.

Restoration projects have generally been more successful on saltmarshes and mangroves than on seagrass meadows. Mangroves appear to be most amenable to revitalization. The restoration of mangrove forests has been accomplished by transplanting propagules, young trees, or mature trees, with restoration programmes being most successful in India and countries in the Far East (Burma, Malaysia, Thailand, and Vietnam; Alongi 1998). Wetland restoration in the USA has

Table 6 Ranking of future anthropogenic threats to estuarine environments based on assessment of published literature (McIntyre 1992, 1995; Windom 1992; Yap 1992; Jones 1994; Goldberg 1995, 1998; Kennish 1997, 1998*a*, 2000, 2001*a*, *b*).

Stress	*Principal impacts*
1. Habitat loss and alteration	Elimination of usable habitat for estuarine biota
2. Eutrophication	Exotic and toxic algal blooms, hypoxia and anoxia of estuarine waters, increased benthic invertebrate mortality, fish kills, altered community structure, shading, reduced seagrass biomass, degraded water quality
3. Sewage	Elevated human pathogens, organic loading, increased eutrophication, degraded water and sediment quality, deoxygenated estuarine waters, reduced biodiversity
4. Fisheries overexploitation	Depletion or collapse of fish and shellfish stocks, altered food webs, changes in the structure, function, and controls of estuarine ecosystems
5. Chemical contaminants	
(a) Higher priority synthetic organic compounds	Adverse effects on estuarine organisms including tissue inflammation and degeneration, neoplasm formation, genetic derangement, aberrant growth and reproduction, neurological and respiratory dysfunction, digestive disorders and behavioral abnormalities; reduced population abundance; sediment toxicity
(b) Lower priority: oil (PAHs), metals, radionuclides	
6. Freshwater diversions	Altered hydrological, salinity, and temperature regimes; changes in abundance, distribution, and species composition of estuarine organisms
7. Introduced species	Changes in species composition and distribution, shifts in trophic structure, reduced biodiversity, species introduction of detrimental pathogens
8. Sea level rise	Shoreline retreat, loss of wetlands habitat, widening of estuary mouth, altered tidal prism and salinity regime, changes in biotic community structure
9. Subsidence	Modification of shoreline habitat, degraded wetlands, accelerated fringe erosion, expansion of open water habitat
10. Debris/litter (plastics)	Habitat degradation; increased mortality of estuarine organisms due to entanglement in debris and subsequent starvation and suffocation

been most successfully achieved in estuarine saltmarsh systems (Kusler & Kentula 1990; Kennish 2000). The revitalization of damaged seagrass beds has been fraught with difficulties; the success rate of seagrass restoration programmes on seagrasses worldwide is about 45% (Berger 1990; Alongi 1998).

Future anthropogenic threats

Estuaries have been the target of considerable human exploitation because of their formidable resources and economic importance. The expanding population in the coastal zone will exert even greater demands and pressures on estuarine systems during the next 25 years. More than 75% of the world's population may live within 60 km of the coast by the year 2020 (Roberts & Hawkins 1999). Although urbanization and industrialization of developed nations in the northern hemisphere have accounted for a disproportionately large number of the estuarine impacts documented in the past, more rapid population growth and attendant urbanization in developing nations of the tropics and elsewhere in the southern hemisphere will dramatically increase estuarine impacts during the 21st century (Alongi 1998). According to Postel (2000), nearly two-thirds of the world's population will inhabit cities and towns by the year 2025, many in coastal regions, with the population projected to double in urban areas to more than five billion people.

Table 6 provides a ranking of future human impacts on estuaries worldwide by level of importance based on assessment of published literature (e.g. McIntyre 1992, 1995; Windom 1992; Yap 1992; Jones 1994; Goldberg 1995, 1998; Kennish 1997, 1998*a*, 2000, 2001*a*, *b*). It is anticipated that uncontrolled population growth and development in coastal watersheds, as well as escalating suburban sprawl in shore communities, will hasten the degradation of estuarine habitats. The encroachment of residential and commercial development (coastal urbanization) on estuarine shorelines will not only physically alter wetland and aquatic habitats but will also functionally degrade various components of estuarine systems by increasing pollutant inputs, modifying hydrologic regimes, and destroying critically important vascular plant communities (e.g. saltmarshes, mangroves and seagrasses). Development in the coastal zone and associated habitat loss and alteration appear to pose the most serious threat to the future health and viability of estuarine ecosystems.

Potential impacts of nutrient enrichment and sewage inputs are a priority concern. Excessive nutrient loading (e.g. nitrogen and phosphorus) from fertilizers, livestock, mariculture operations, seafood processing wastes, municipal and industrial wastewaters, and fossil-fuel emissions will accelerate autotrophic growth in estuaries, leading to greater incidence of nuisance and toxic algal blooms, hypoxia and anoxia, fish kills, declines in fishery and shellfishery yields, diminishing seabird populations and changes in community structure (Goldberg 1995). Eutrophication of estuarine waters and the occurrence of harmful algal blooms are now widespread problems rather than local issues (Goldberg 1998). Indeed, because of its increasing frequency worldwide, eutrophication may now be seen as a global problem (McIntyre 1995).

Sewage inputs will also exacerbate eutrophication problems in some regions already experiencing nutrient enrichment by further raising the concentrations of nitrogen and phosphorus in receiving waters and thus stimulating autotrophic growth (World Resources Institute 1998). In addition, sewage inputs will increase the amount of organic carbon loading and human pathogens, thereby elevating the biochemical oxygen demand and creating a potential public health threat. Sewage impacts will be most acute in estuaries of developing countries, where effective sewage treatment and waste disposal programmes, as well as government regulatory controls, will be largely deficient. Yap (1992) notes that the inadequate or even total lack of sewage treatment facilities in major population centres of south-east Asia and other parts of the Third World is probably the most significant existing pollution problem in these regions.

As human population growth and settlement increase in the coastal zone during the next two decades, a greater probability exists for the further overexploitation of fisheries resources. The decoupling of an overharvested (piscivorous) fish species from an estuarine system may have serious consequences for the estuarine food web, because it can result in uncontrolled population growth of prey species and increased abundance of competitors (Valiela 1995). These shifts in population abundance may lead to pronounced changes in the composition of the estuarine community (Day *et al.* 1989).

Estuarine water quality will continue to be degraded by toxic contaminant inputs from industrial, agricultural and urban sources. Of greatest concern are synthetic organic compounds (e.g. the chlorinated hydrocarbons DDT and PCBs), organometals (e.g. tributyltin, methylmercury and organically bound copper) and certain heavy metals (e.g. mercury, cadmium, and lead), as well as PAHs. Adverse effects of PAHs and other toxic components in oil will be most evident at oil spill sites (Valiela 1995). Agricultural uses of chlorinated pesticides are expected to rise substantially in developing countries during the next 25 years, and these contaminants will pose an increasing threat to estuarine biota. Heavy metals will remain a local rather than a global problem, attaining peak impacts in the vicinity of metal-rich discharges or mine tailing effluents (McIntyre 1992), although releases of mercury, cadmium, and lead from fossil-fuel combustion may produce more regional problems (Taylor 1993). Similar to heavy metals, radionuclides are seen as less threatening than synthetic organic compounds to estuarine environments (Goldberg 1995; McIntyre 1995). Radionuclide inputs are mainly linked to relatively small releases from nuclear power stations and reprocessing plants.

Other potentially serious problems are coupled to rising sea level. During the past century, eustatic sea level has risen 10–25 cm, principally because of thermal expansion of ocean waters and the melting of glacial ice in response to global warming (Ledley *et al.* 1999). Other factors, such as tectonic and isostatic adjustments (i.e. ocean basin deformation and land subsidence or emergence), have also influenced sea level changes. By the year 2020, sea level is expected to rise an additional 2.6–15.3 cm (IPCC 2001). Such sea level change will result in estuarine shoreline retreat, increased flooding and erosion, and the loss of wetland habitat (Wolanski & Chappell 1996). In addition, tidal prisms and salinity regimes may be substantially altered, which could significantly affect the composition of estuarine communities (Kennish 2000). Changing climate conditions, such as those manifested by El Niño and La Niña events, may exacerbate these effects by escalating storm activity that promotes coastal flooding and erosion, as well as sediment and contaminant resuspension from the estuarine floor (Bijlsma *et al.* 1996).

Three additional stresses on estuaries, namely freshwater diversions, introduced species, and debris/litter, will likewise adversely affect estuarine habitats and organisms during the next 25 years. For example, more freshwater diversions will be constructed to satisfy the agricultural, industrial, and domestic needs of expanding coastal populations. These diversions will reduce freshwater inflow, nutrients and sediment loads, which could greatly influence the salinity regime, productivity and species composition of biotic communities in affected estuaries. The introduction of exotic, non-native species could be even more deleterious in some estuaries, possibly resulting in the complete displacement of outcompeted native forms.

Debris/litter will have a more direct impact by increasing the mortality of organisms which become entangled in it and subsequently drown, or ingest it and later suffocate or starve. Organisms entangled in debris are also more vulnerable to predation. Floatable debris, especially plastics, could increase substantially in estuaries by 2025 concomitant with a burgeoning coastal population and the greater recreational use of these valuable systems.

Natural versus anthropogenic influences

Estuaries exhibit wide variations in physical and chemical conditions, and hence it is often difficult to distinguish natural changes in these systems from changes mediated by anthropogenic effects. Biotic community changes ascribable to natural factors including hurricanes, storm surges, and predation may be more acute than those attributable to chemical contamination or some other anthropogenic impact. In addition, natural processes interacting with anthropogenic influences can obfuscate environmental assessment programmes. For example, major storms commonly roil bottom sediments in shallow water systems, and currents can remobilize chemical contaminants from impacted sites to originally unaffected areas. The significance of natural processes in the dispersal of chemical contaminants is frequently overlooked.

The difficulty of distinguishing an environmental effect associated with natural variability in an estuary from a debilitating human impact is evident when examining certain pollutants. A larger fraction of the total heavy metal load in some systems originates from natural sources (e.g. weathering of rocks, leaching of soils and volcanic emissions) rather than anthropogenic inputs (e.g. industrial and municipal effluents). Similarly, more hydrocarbons may originate from natural sources (e.g. volcanic eruptions, seeps and bacteria) than anthropogenic activities (e.g. oil spills, urban run-off and shipping). Nutrient inputs to estuaries from autochthonous mineralization of organic matter, seawater inflow and atmospheric deposition may also exceed those entering from land-based sources, such as farmlands and municipal discharges.

Table 7 Total system net annual primary production of carbon in selected estuaries (after Knox 2001).

Site	*Production ($gC\ m^{-2}\ yr^{-1}$)*
Beaufort, North Carolina	152
Flax Pond, New York	535
Sapelo Island, Georgia	1445
Grays Harbour, Washington	2817
Barataria Bay, Louisiana	880
Bot River Estuary, South Africa	815
Tidal Pond, Mai Po, Hong Kong	1005
Upper Waitemata Harbour, New Zealand	473

LONG-TERM TRENDS

Organism abundances, production and community composition

Estuaries are highly fertile ecosystems with rich food supplies; thus, they support large densities and biomasses of organisms. However, because estuaries are physically stressed ecosystems subject to wide fluctuations in environmental conditions and frequent anthropogenic impacts, their biotic communities are characterized by relatively low species diversity. Large numbers of organisms use estuarine habitats for nesting, feeding, reproduction, and shelter. The overall biotic production here rivals the most productive systems on Earth (Table 7), according to Alongi (1998) largely for the following reasons:

- Abundant nutrients.
- Conservation, retention and efficient recycling of nutrients among benthic, wetland and pelagic habitats (i.e. coupling of subsystems).
- Consortia of phytoplankton, benthic microalgae and macroalgae, seagrasses, mangroves, and fringing saltmarsh vegetation that maximize available light and space (Table 8; Fig. 7).
- Tidal energy and circulation.

Multiple plant subsystems in estuaries yield high production. For example, the net annual primary production of phytoplankton ranges from 5–530 gC m^{-2} yr^{-1} and averages about 250 gC m^{-2} yr^{-1}. The net annual primary production of benthic microalgae, in turn, ranges from 25–250 gC m^{-2} yr^{-1}, but can be as high as 2000 gC m^{-2} yr^{-1}. Highest values are generally recorded in the top centimetre of mudflat sediments. Above ground net annual primary production of saltmarshes amounts to 200–3 000 gC m^{-2} yr^{-1}; below ground production may equal or exceed these numbers. Somewhat lower net annual primary production figures of

Table 8 Net annual primary production of carbonin estuarine habitats (values expressed in gC m^{-2} yr^{-1} or g dry wt m^{-2} yr^{-1}; after Kennish 1986). [a]Above ground. [b]Below ground.

Plant type	*Location*	*$gC\ m^{-2}\ yr^{-1}$*	*$g\ dry\ wt\ m^{-2}\ yr^{-1}$*
Phytoplankton	Baltic Sea	48–94	
	St Margaret's Bay, Canada	190	
	Cochin Backwater, India	124	
	Ems Estuary, Netherlands	13–55	
	Grevelingen, Netherlands	130	
	Wadden Sea, Netherlands	100–200	
	Loch Etive, UK	70	
	Lynher, UK	81.7	
	Alewife Cove, USA	162	
	Barataria Bay, USA	210	
	Beaufort, USA	52.5	
	Bissel Cove, USA	56	
	Charlestown River, USA	42	
	Core Sound, USA	67	
	Duplin River, USA	248	
	Flax Pond, USA	60	
	Hempstead Bay, USA	177	
	Jordan Cove, USA	66	

	Long Island Sound, USA	205	
	Narragansett Bay, USA	242	
	Niantic River, USA	72	
	North Inlet, USA	346	
Microbenthic algae	Danish fjords	116	
	Wadden Sea, Netherlands	115–178	
	Grevelingen, Netherlands	25–37	
	Ythan Estuary, UK	31	
	Lynher, UK	143	
	Alewife Cove, USA	45	
	Barataria Bay, USA	240	
	Bissel Cove, USA	52	
	Charlestown River, USA	41	
	Delaware Bay, USA	160	
	False Bay, USA	143–226	
	Flax Pond, USA	52	
	Hempstead Bay, USA	62	
	Jordan Cove, USA	41	
	Niantic River, USA	32	
	Sapelo River, USA	180	
Seagrasses			
Halodule wrightii	North Carolina, USA (intertidal)	70–240	
	South Florida – Gulf coast, USA (intertidal)	70–240	
Thalassia testudinum	South Florida – Gulf coast – Texas, USA (subtidal)	580–900	
Zostera marina	Denmark: Leaves	856[a]	
	Denmark: Roots, rhizomes		241[b]
	Alaska, USA	19–552	
	Beaufort, USA	350	
	North Carolina, USA (intertidal)	330	
	Little Egg Harbour, USA		466
	Pacific coast, USA (subtidal)	90–540	
	Puget Sound, USA	58–330	116–680
Saltmarsh grasses			
Carex spp.	Arctic waters	10–120[a]	
Distichlis spicata	Atlantic coast, USA		1 070-3400[b]
	Gulf coast, USA (Louisiana)	1600[a]	
	Pacific coast, USA	300–600[a]	
Juncus gerardi	Maine, USA		485[a]
	Atlantic coast, USA		1620–4290[b]
Juncus roemerianus	North Carolina, USA		754[a]
	Georgia, USA		2200[a]
	Gulf coast, USA (Louisiana)	1700[a]	
	Gulf coast, USA		1360–7600[b]
Puccinellia phrygondes	Arctic waters	25–70[a]	
Salicornia virginica	Atlantic coast, USA		430–1430[b]
Salicornia spp.	Norfolk, UK		867[a]
	Pacific coast, USA	325–1000[a]	
Spartina alterniflora	Massachusetts, USA		3500[b]
	Atlantic coast, USA	220–1680[a]	
	Atlantic coast, USA		220–3500[b]
	Atlantic and Gulf coasts, USA	200–2000[a]	
	Gulf coast, USA (Louisiana)	1300[a]	279–6000[a]
Spartina foliosa	Pacific coast, USA	400–850[a]	
Spartina patens	Maine, USA		5163[a]
	Atlantic coast, USA		310-3270[b]
	Atlantic and Gulf coasts, USA	500–700[a]	
Mixed species	Netherlands	100-500[a]	200–1000[a]
	Europe		11–1100[a]
	Barataria Bay, USA	590[a]	1175[a]
	Gulf coast, USA	500[a]	
	North America	100–1700[b]	300–4000[a]
Mangroves			
Rhizophora mangle	South Florida, USA	400	100–730

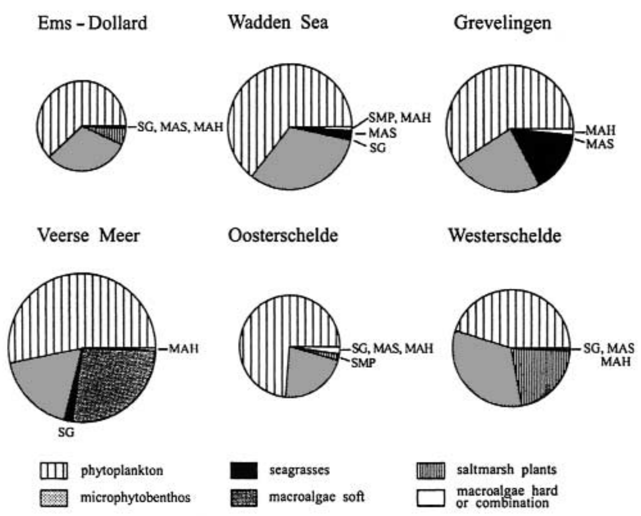


Figure 7 Mean net annual primary production in several Dutch estuaries demonstrating the relative importance of multiple plant subsystems to the total carbon production. Total production values: Ems-Dollard (160 gC m^{-2} yr^{-1}), Wadden Sea (310 gC m^{-2} yr^{-1}), Grevelingen (320 gC m^{-2} yr^{-1}), Veerse Meer (450 gC m^{-2} yr^{-1}), Oosterschelde (240 gC m^{-2} yr^{-1}), and Westerschelde (275 gC m^{-2} yr^{-1}) (after Nienhuis 1992).

50–1500 gC m^{-2} yr^{-1} have been recorded for seagrasses, with those of mangroves being 350–1 000 gC m^{-2} yr^{-1} (Mann 1982; Kennish 1986; Valiela 1995; Alongi 1998; Knox 2001).

Most of the aforementioned vascular plant production passes to the detritus food web, where it is decomposed by bacteria and fungi. Bacteria attain high abundances in estuaries, because of the high primary production and relatively large amount of organic matter that accumulates in these systems (dissolved organic matter = 1–5 mg l^{-1}; particulate organic matter = 0.5–5 mg l^{-1}). Bacterial counts in estuarine waters range from 10^6–10^8 cells ml^{-1}, and they decline seaward (Ducklow & Shiah 1993; Valiela 1995; Pinet 2000). Bacterial biomass in estuarine waters, in turn, is about 1.0 μg C l^{-1} (Kennish 2001*a*). Peak numbers of bacteria exist in bottom sediments, with highest cell counts (10^8–10^{11} cells cm^{-3}) recorded in mudflat and saltmarsh sediments containing elevated concentrations of organic matter. Lower cell counts (10^6–10^8 cells cm^{-3}) are found in subtidal sediments. Bacterial production in saltmarsh, mangrove, and seagrass biotopes ranges from 10–800 mgC m^{-2} day^{-1} (Moriarty 1986; Kemp 1990).

All major animal phyla are represented in estuarine biotic communities. Zooplankton are the principal herbivorous consumers, converting plant to animal matter. They provide forage for numerous benthic and nektonic fauna.

Estuarine benthic fauna are subdivided into four major groups based on size: microfauna (<0.1 mm), meiofauna (0.04–0.5 mm), macrofauna (0.5 mm–20 cm), and megafauna (>20 cm) (Levinton 1982). The microfaunal group mainly consists of protists (e.g. ciliates, zooflagellates and foraminifera), diminutive forms weighing 10^{-6}–10^{-11}g. These organisms can attain densities greater than 10^7 individuals m^{-2} in estuarine bottom sediments and can reach biomasses of about 5×10^{-3} g in 1 g dry weight of detritus (Kennish 1986). They feed heavily on bacteria, although the ciliates also consume zooflagellates, other ciliates and microalgae.

The meiofaunal group includes juvenile stages of the macrofauna (temporary meiofauna) and permanent members such as gastrotrichs, kinorhynchs, nematodes, rotifers, archiannelids, halacarines, copepods, ostracods, mystacocarids, tardigrades, and representatives of the bryozoans, gastropods, holothurians, hydrozoans, oligochaetes, polychaetes, turbellarians, nemertines, and tunicates. Most individuals inhabit the upper 2–5 cm of sediments, and they exhibit a patchy distribution. Their densities average about 10^5 individuals m^{-2} in subtidal sediments, but can exceed 10^7 individuals m^{-2} in intertidal mudflats (Kerby 1977). Mean standing crop biomass amounts to 1–2 g m^{-2} (Coull & Bell 1979). Estimates of meiofaunal production range from 0.2-20 gC m^{-2} yr^{-1} (Warwick *et al.* 1979).

Benthic macrofauna of estuaries are far less abundant than microfauna and meiofauna but have much greater biomass. As stated by Kennish (1986, p. 267): 'For an entire estuary, the mean biomass of the macrobenthos generally ranges from 10–25 g ash-free dry wt m^{-2}, and the mean production approaches 50 g ash-free dry wt m^{-2} yr^{-1}. Dense aggregations of animals, such as mussels and oysters, yield annual production values of 200-300 g ash-free dry wt m^{-2} yr^{-1}. Locally, the mean biomass of these organisms attains levels of 2000 g g ash-free dry wt m^{-2} or more.'

Fishes, marine mammals, marine reptiles, swimming molluscs and crustaceans (e.g. swimming crabs), together with wading birds and shorebirds, represent the highest-trophic-level organisms of estuaries. Individuals in this group range in size from less than 20 mm to more than 20 m. Among the larger members of this group are piscivorous fish (e.g. bluefish and striped bass), cuttlefish, cetaceans, pinnipeds (sea lions and seals) and sea turtles.

Fish populations dominate estuarine nektonic communities in terms of numerical abundance and biomass, and they play a significant role in energy flow of the system (Fig. 8). The most abundant forms are juveniles, which use the estuary as a nursery area. Only a relatively few species dominate estuarine fish faunas, and they tend to have broad tolerances and wide ranges of adaptations. Most fishes found in estuaries are not permanent residents, but seasonal visitors that mainly enter from the nearshore ocean. A number of marine species are estuarine dependent (Claridge *et al.* 1986). Some species (e.g. anadromous and catadromous forms) use these shallow environments strictly as migratory pathways between feeding and spawning grounds.

Estuarine fish faunas can be grouped into five distinct assemblages based on their abundance and occurrence in the estuary, migratory habits and reproductive patterns. These assemblages are: (1) freshwater species; (2) diadromous forms; (3) marine migrants; (4) residents; and (5) adventitious visitors (Kennish 1986; 2001*a*). Estuarine fish

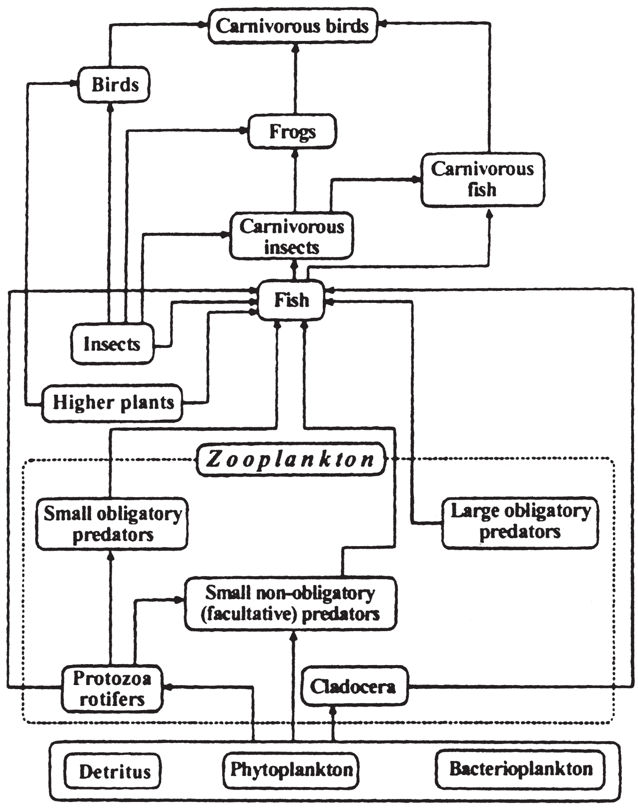


Figure 8 Generalized flow diagram applicable to the estuarine food chain, illustrating the significant role of fish as consumers in the system (after Bukata *et al.* 1995).

communities display relatively low species diversity, although the abundance and biomass of fish species typically are high. Estimates of annual fish community production in temperate estuaries range from 5–150 g m^{-2} yr^{-1}. These values are comparable to those recorded in subtropical and tropical systems (5–125 g m^{-2} yr^{-1}; Day *et al.* 1989).

Estuarine nektonic communities also include marine mammals and reptiles. Of particular note are the cetaceans (whales, porpoises, and dolphins), pinnipeds (seals, sea lions and walruses), carnivorans (sea otters), and sea turtles (loggerhead, leatherback and green turtles). Many of these large animals periodically visit estuaries in search of food, foraging heavily on fish and invertebrates. Numerous land mammals, reptiles and amphibians also use wetland and shoreline habitats surrounding estuaries, consuming organisms on land, in tidal creeks, and in shallow estuarine waters. Examples of such animals are racoons, muskrats, foxes, frogs, crocodiles, alligators, lizards and snakes. Some wildlife species also prey on the eggs of shorebirds, turtles and other organisms, and can produce imbalances in the trophic organization of estuarine systems (Kennish 2000).

Avifaunas are often overlooked when assessing estuarine biotic communities, but they are important food web components capable of controlling prey abundance or denuding extensive areas of submerged aquatic vegetation. Bird excretion also delivers faecal nitrogen and pathogens that can compromise water quality. In addition, some waterfowl species (e.g. ducks and geese) are estuarine game birds with significant value to sport hunters.

Bird populations inhabiting estuaries belong to three main groups: (1) seabirds (e.g. terns, skimmers, gulls and auks); (2) waders (e.g. egrets, ibises and herons); and (3) waterfowl (e.g. ducks, geese, mergansers and scoters). Some shorebirds intensively crop benthic invertebrates on mudflats and sandflats. For instance, shorebirds migrating northward from South America gain as much as 50% of their body weight over a 10–14-day foraging period along the Delaware Bay shoreline (Kennish 2000). Some seabirds rely greatly on estuaries for food, and their population densities vary in response to fluctuations in prey availability. Human activities (e.g. motorized boating and personal watercraft use) frequently disturb nesting shorebirds and other avifaunas, which can reduce the abundance of populations.

Ecosystem structure and energy flow

Lalli and Parsons (1993), Pinet (2000) and Knox (2001) have reviewed the general structure and dynamics of estuarine ecosystems. Proceeding from the head to the mouth of temperate and boreal estuaries, the saltmarsh community proliferates in the intertidal zone, being comprised of halophytic grasses, sedges and succulents. Dominant genera include *Spartina, Carex, Distichlis, Juncus, Salicornia* and *Puccinellia* (Fig. 9). The saltmarsh community is replaced by mangroves in the intertidal zone of subtropical and tropical regions, with the dominant genera consisting of *Avicennia, Bruguiera* and *Rhizophora*. Benthic algae also may contribute substantial amounts of primary production to both the saltmarsh and mangrove communities.

The seagrasses occupy the lower intertidal and shallow subtidal zones of estuaries from boreal to tropical latitudes, seaward of saltmarsh and mangrove communities. In the temperate latitudes of Europe, North America, Asia, and Australia, the most widely distributed seagrasses are *Zostera*

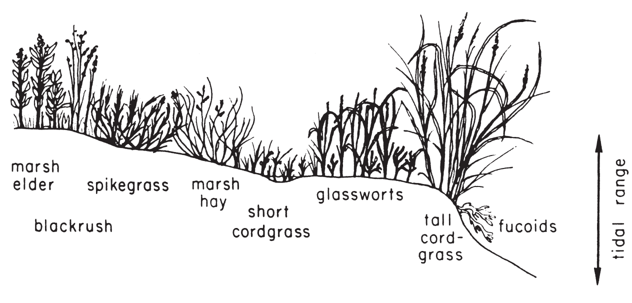


Figure 9 Plant zonation in a typical New England saltmarsh system. Tidal marsh zonation is based on the frequency of tidal flooding. Marsh plant species present from left to right in the diagram include *Iva frutescens, Juncus gerardi, Distichlis spicata, Spartina patens, Salicornia europaea* and *Spartina alterniflora*. *Fucus* and *Ascophyllum* represent the fucoids. Note tidal range on right (after Valiela 1995).

and its subgenus *Zosterella*. In the tropical latitudes of North America, South America, Africa, Asia and Australia, the genera *Thalassia* and *Cymodocea* predominate. Seagrass, saltmarsh and mangrove communities are dominated by detritus-based food chains. Much of the plant material in these macrophyte communities is refractory; thus, more than 90% of the plant production passes to detritus and is used by detritivores.

In some areas, depending on the current and tidal regime, intertidal mudflat and shallow subtidal sand bar communities occur nearly continuously on the seaward side of the aforementioned intertidal macrophyte communities. Epipsammic algae (largely diatoms and dinoflagellates) are the dominant autotrophs of the mudflats and sand bars. Meiofauna and some macrofauna also attain high densities in tidal flat sediments. However, as noted by Eisma (1998, p. 15), 'The distribution of benthic fauna in intertidal sediments is much less well known worldwide than the distribution of vegetation.'

In the middle and lower estuary, phytoplankton is usually the major primary producer group. Zooplankton, including holoplankton and meroplankton, and pelagic fishes account for most of the faunal biomass in the water column. Infaunal and epifaunal benthic communities are well established here. Shorebirds, waterbirds, marine mammals, and marine reptiles use the rich estuarine food supply.

Estuaries are characterized by a complex food web structure consisting of two major interlocking components of energy flow: the detritus and grazing pathways (Kennish 1986; Lalli & Parsons 1993; Valiela 1995; Knox 2001). In the detritus-based food web, particulate and dissolved organic matter principally derived from vascular plant remains, as well as from biodeposits, faeces, pseudofaeces and remains of animals, serve as the energy base. Detritivores consume the detritus, but appear to obtain most energy from microbes (bacteria) attached to the detrital substrate. The detritivores, in turn, provide a food source for secondary consumers (e.g. larger invertebrates and fish) that constitute prey for tertiary consumers. Because some of these upper-trophic-level organisms are omnivorous, consuming organisms at more than one trophic level, a complex network of feeding relationships typifies estuaries (Fig. 8). Detritus-based food webs are most conspicuously developed in shallow coastal bays and lagoon-type estuaries harbouring extensive saltmarsh, mangrove and seagrass subsystems.

In grazing food webs, phytoplankton (diatoms, dinoflagellates and other microalgae) forms the energy base. Zooplankton consumes this plant material and is fed upon by secondary consumers (e.g. anchovies and silversides). Larger invertebrates (e.g. blue crabs) and fish (e.g. bluefish and striped bass) represent the tertiary consumers. A microbial loop, in which bacteria process organic matter while being grazed by protists, is also coupled to this classic food web of phytoplankton, zooplankton and fish. Grazing-based food webs predominate in deeper, clearer waters down-estuary.

The separation between grazing and detrital pathways becomes obscured among organisms at upper trophic levels, consisting of heterotrophs which derive energy from both pathways. It is at the primary producer and primary consumer levels, therefore, where the distinction between the grazing and detritus food webs clearly exists (Odum & Biever 1984). In both types of food webs, the transfer efficiencies are relatively low, with energy losses between trophic levels amounting to 80–90%.

POTENTIAL STATES IN 2025

Habitat loss and alteration

Human activities impact estuaries over all temporal and spatial scales, being responsible for population, community, and ecosystem changes that can have long-lasting, detrimental consequences. Most of the adverse anthropogenic effects on estuaries are directly coupled to increasing population growth and the expansion of human settlements in coastal areas. Habitat modifications rank among the most serious of all anthropogenic impacts on estuaries. Human interference with coastal processes not only affects estuarine biota but also renders many coastal communities more vulnerable to stochastic natural events, such as hurricanes, extratropical storms, storm surges, and coastal flooding (Kennish 2001*a*).

The evidence is that human occupancy of the coastal zone will continue to increase during the 21st century, which will lead to greater land development, habitat alteration, resource use, and multiple stresses on estuaries and watersheds. Coastal habitat destruction will likely be greatest in developing countries, where government regulatory controls are weak. Sprawled development will continue to create more fragmentation, isolation, and functional degradation of wetland and upland complexes, thereby accelerating the loss and alteration of habitat for wildlife populations and the deterioration of water quality in surrounding areas (Shabman 1996). Impervious surfaces in these developed regions will facilitate nutrient and contaminant transfer to estuaries. They will also promote the erosion and transport of sediments and other particulates to these systems (Kennish 2000). Larger coastal human populations will place greater pressure on limited commercial and recreational fisheries resources. The cumulative impact of multiple stresses associated with escalating population growth and human activities in the coastal zone (e.g. habitat loss and alteration, point and non-point source pollutant inputs, nutrient and organic carbon loading, overfishing, freshwater diversions, and introduction of exotic species) will impair estuarine environments by reducing species diversity and abundance, altering community structure and functional processes, and decreasing system productivity and overall habitat quality (Kennish 1992, 1997; Weinstein 1996).

Excessive population growth and unplanned or poorly planned development in coastal regions are a growing source of the most serious stresses that threaten the long-term health and viability of estuaries worldwide (Kennish 2000). Physical modification of habitat, pollution and other perturbations associated with human activities in watersheds and on the water bodies themselves may be as important as natural influences in effecting change at all scales of biological organization (Harding 1992; McIntyre 1992, 1995).

However, some stresses have more pervasive effects than others do in terms of intensity and scale of ecological consequences. For example, habitat destruction and eutrophication thoroughly restructure the function and controls of marine ecosystems at all temporal and spatial scales (Valiela 1995). Nutrient enrichment effects are mediated through bottom-up controls (i.e. resource supply), although they interact with top-down controls in shallow water systems. Nutrient enrichment is emerging as a major force driving alterations in many estuarine systems worldwide (National Academy Press 1993; Kennish 1997; National Estuary Programme 1997*c*; Livingston 2000). By comparison, the invasion of exotic species, freshwater diversions, and subsiding coastal areas affect estuaries on local or regional scales.

Eutrophication

One of the most compelling aquatic environmental problems is nutrient enrichment, which on an aggregate basis has caused global-scale changes to estuarine systems (Valiela 1995). Increased nutrient inputs are occurring in marine environments worldwide, with anthropogenic sources accounting for 50% or more of the total nutrient influx (Windom 1992). Greater nutrient influx to estuarine and coastal waters correlates with more frequent episodes of phytoplankton blooms, particularly harmful algal blooms (HABs; Anderson 1989; Smayda 1990). According to Hallegreaff *et al.* (1995), red tide blooms increased sevenfold worldwide between 1965 and 1976 during a period when nutrient loading (principally from sewage and industrial wastes) increased twofold. During this period, global fertilizer use also increased by about 50% from approximately 50 000 Mt yr^{-1} to 75 000 Mt yr^{-1}. Toxic red tides have severely impacted many fish, seabird, and marine mammal populations worldwide (World Resources Institute 1998).

Hypoxic and anoxic events in estuaries have also continued to increase in recent years (Kennish 1997, 2000; Howarth *et al*. 2000). However, it has been difficult to unequivocally link these episodes to specific anthropogenic nutrient inputs. As stated by Livingston (2000, p. 2), 'thresholds of nutrient loading leading to toxic blooms remain largely undetermined ... and the combination of nutrient inputs and other anthropogenous stressors remains largely unknown.' Nevertheless, there is a growing body of evidence that the input of nutrients via various anthropogenic activities plays a significant role in the development of estuarine eutrophication problems and that in river-dominated systems, phytoplankton are critical in eutrophication processes (Livingston 2000).

In a recent comprehensive study of eutrophication problems in estuaries of the conterminous USA, Bricker *et al.* (1999) identified 44 highly eutrophic systems and another 40 moderately eutrophic estuaries, mainly along the Atlantic and Gulf of Mexico coasts, and they also projected that by the year 2020 eutrophic conditions would be worse in 86 estuaries. The systems most susceptible to eutrophication will be those having a high capacity to retain nutrients (Livingston 2000).

The outlook for estuarine eutrophication in other countries also appears to be problematic. For example, eutrophication of coastal waters is on the rise in China (e.g. Yellow and East China Seas), Japan, Korea and smaller developing nations of the Far East as the intensity of agriculture, aquaculture, and industrial activity increases (Galloway *et al.* 1994; Valiela 1995). The frequency and severity of eutrophication have also been escalating along the coastal zone of India, Pakistan and the Caribbean (Alongi 1998).

Global fertilizer use has risen steadily with world population growth during the past 50 years and is an important pathway for nutrient influx to estuarine systems. Between 1970 and 1990, it more than doubled to more than 100 000 Mt yr^{-1}, while the global population grew to more than five billion people (Windom 1992). Human-generated nitrogen now totals about 210 Mt yr^{-1}, with natural sources yielding only about 140 Mt yr^{-1} (World Resources Institute 1998). Because fertilizer consumption continues to increase with a burgeoning world population (with projected use being $>$200 000 Mt yr^{-1} by 2005) and because fertilizer is a major source of nutrients to coastal waters, eutrophication poses a greater potential threat than other anthropogenic factors to estuarine systems worldwide. Eutrophication remains one of the most effective anthropogenic agents of change to these sensitive ecosystems (Valiela 1995; Livingston 2000).

Sewage waste inputs as well as intensive mariculture operations and livestock rearing will also promote eutrophication of estuarine waters during the next 25 years, especially in developing countries. These inputs likewise represent priority problems, because they augment nutrient concentrations originating from other sources. In the USA, UK, and other developed nations where tighter government controls now exist on the release of sewage wastes to aquatic systems, the potential for nutrient enrichment of estuaries from land-based municipal sources and boats has decreased appreciably during the past 25 years and is expected to decline further during the next 25 years (Office of Technology Assessment 1986; National Academy Press 1993; National Estuary Programme 1997*b*, *c*; Kennish 2000). However, the projections are not as favourable for many developing countries in Asia, Africa, and South America, where sewage waste disposal in coastal waters continues unabated. This problem may become even more acute as the coastal population in these countries expands during the 21st century.

Fisheries exploitation

As in the case of eutrophication, overfishing is a major anthropogenic agent mediating global-scale change to estuarine and coastal marine systems. Many estuaries which formerly provided large fish harvests are now overexploited, with some fisheries in a state of near collapse. High-value species have been the first to be depleted, leaving less desirable forms to be exploited. Of the 28 estuaries that have been designated as National Estuary Programme sites in the USA, declines of fish and wildlife populations are now considered to be high- or medium-priority problems in 22 of them.

Estuaries have long supported a wide array of commercially and recreationally-important finfish and shellfish species, largely because of their rich food supply. They are primary nursery areas for many common commercially important species (Day *et al.* 1989). Numerous juvenile marine species grow at exceptional rates in estuaries, inhabiting wetlands, submerged aquatic vegetation, and open water habitats.

Marine fisheries worldwide are producing at or near their global maximum sustainable yield, and both developing and developed countries contribute significantly to the total world catch (Fig. 10). However, many individual fisheries are declining or are depleted (Sissenwine & Rosenberg 1996). This situation is exemplified by the large (> 75%) decreases in total fishery landings of demersal fish caught off the New England coast in USA waters between 1960 and 1992. Fishers in these waters are catching fish at more than twice the estimated sustainable yield (Valiela 1995), and this unsustainably high harvest has decimated fisheries stocks.

A comparable condition has been reported in estuaries in various regions of the world (Day *et al.* 1989; Valiela 1995; Hall 1999). Because of excessive harvesting pressure in these coastal systems, the catch of many fishery resource species is unsustainable. This situation is evident for both fish and shellfish populations.

The projected outlook for fisheries exploitation in estuaries is not favourable considering the rapid population growth expected in the coastal regions of the world, the increasing demand for fish products, and the policy of open access to fisheries promoted by most countries. Although the world demand for fish remains high (about one billion people obtain most of their protein from this source), fish production is likely to decrease during the next 25 years. It is anticipated that the world demand for fish will increase to 110–120 Mt yr^{-1} by 2010 (Food and Agriculture Organization 1996), and even higher demands can be expected by 2025. However, the supply is unlikely to meet the demand, unless aquaculture production can be increased substantially. Currently, aquaculture provides about 25% of the fish consumed worldwide (World Resources Institute 1998).

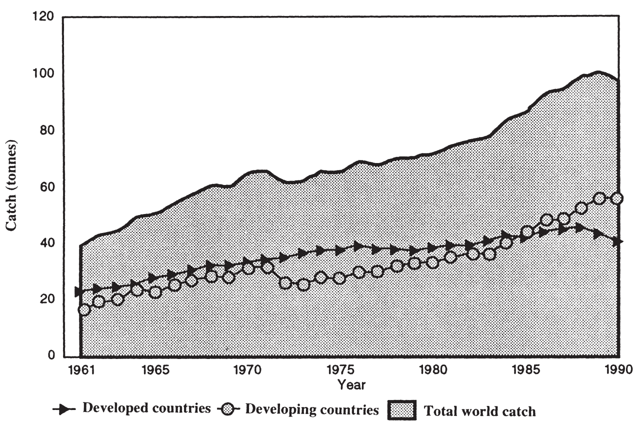


Figure 10 Global marine fish catch by developed and developing countries from 1961 to 1990. Catch includes aquaculture production. Data suggest that the maximum sustainable fish yields may have been reached in the late 1980s (after Alongi 1998).

Where fisheries access has been restricted through individual transferable quota systems (e.g. Canada, Iceland, and New Zealand) or through a complete moratorium on fisheries harvest (e.g. striped bass fishery in the mid-Atlantic states of the USA during the 1985–1990 period), recovery of fisheries stocks has been demonstrated (Day *et al.* 1989; Sissenwine & Rosenberg 1996). Nevertheless, for most countries the trend of depleted fisheries stocks due to overfishing continues. Many developing countries do not have effective fisheries management programmes, and fishers generally oppose initiatives that limit fishing effort. Thus, it is likely that overfishing problems in estuaries will become more serious during the next 25 years, manifested by greater numbers of depleted or collapsed fisheries stocks and perhaps by other, more far reaching changes in the structure of the impacted systems.

Chemical contaminants

Estuaries are efficient filters of particle-reactive chemical contaminants (PAHs, halogenated hydrocarbons and heavy metals), which mainly enter these systems via land run-off, river inflow, and atmospheric deposition. Although population and community impacts mediated by toxic chemicals can be severe (see Kennish 1992, 1997, 2000, 2001*a*), they are more localized relative to the problems associated at larger spatial scale with eutrophication and overfishing. The most acute effects of chemical contaminants are observed in spatially-limited, urbanized estuaries. However, chronic contaminant inputs over broader areas can cause problems that are more widespread.

Current restrictions on waste dumping and wastewater discharges in response to more stringent government controls have significantly reduced contaminant concentrations in many developed countries. The future trend is for continued tight controls on chemical contaminants released to estuarine environments in these countries. In contrast, contaminant inputs in developing nations are expected to increase in concert with the expansion of agriculture and industry.

Assessment by GESAMP (1990) of the status of chemical contaminants in the marine environment indicates that synthetic organic compounds, especially organohalogens, pose a greater threat to biotic communities than oil, heavy metals (particularly lead), radionuclides and other substances. Several reasons have been given for GESAMP's lessened concern regarding oil, heavy metals, and radionuclides, including the implementation of national and international regulations, declining use and production of heavy metals and other contaminants, and improvements in technology (Alongi 1998). Mercury is an exception among heavy metals, because it is becoming a major toxic agent in estuaries worldwide (Robert J. Livingston, Florida State University, personal communication 2001). The greater focus placed on synthetic organic compounds stems from the fact that some of these substances (e.g. DDT, PCBs and chlordane) are extremely toxic, persistent, and widespread, having been widely dispersed in the environment for agricultural production (e.g. pesticides), disease prevention (e.g. DDT), and industrial applications (e.g. PCBs) (Windom 1992; McIntyre 1995). While many synthetic organic compounds are now banned in developed countries, they are still used in developing countries, most notably in tropical regions. In addition, new synthetic organic compounds are introduced each year.

The use of persistent organochlorine pesticides and certain industrial chemicals is increasing in developing countries at a rate of 7–8% per year (United Nations Environment Programme 1992). The environmental application of some toxic pesticides will quadruple in the environment of these countries by 2025, with estuarine bottom sediments being the ultimate sink for much of them. The projected impacts of these contaminants are obfuscated by the paucity of scientific information available on their fate and effects in the estuarine environment. Hence, there is serious concern related to uncertainties surrounding potential estuarine impacts of synthetic organic compounds during the next 25 years.

Freshwater diversions

Escalating human settlement in coastal regions during the next 25 years will result in increasing modification of natural flow regimes in estuarine tributaries. Greater volumes of fresh water will be needed, not only to meet the domestic demands of larger numbers of people in coastal communities, but also to provide for requirements of agricultural and industrial interests. As demands rise, more fresh water will be diverted from influent systems, thereby reducing freshwater inflows to existing estuaries. The diverted flows will also decrease sediment loads and alter the inputs of nutrients and other chemical constituents. Freshwater influx will decline even more significantly downstream of storage reservoirs and dams.

Diverted freshwater flows can have serious repercussions for estuarine systems. For example, diverted flows will decrease the residence time of water in estuaries and reduce suitable habitat for biota (Valiela 1995). These changes can significantly affect water quality and living resources of an estuary. Diverted flows will also affect important physical and chemical processes. Freshwater inflow, for example, influences the capacity of an estuary to dilute, transform, and flush contaminants (Kennish 2000).

Human alteration of freshwater inputs to estuaries will increase substantially in the USA and other developed nations by 2025. The ecology of affected estuaries (e.g. Charlotte Harbor and Tampa Bay, Florida, and San Francisco Bay, California) will continue to be impacted by the alteration of freshwater inflow (Kennish 2000, 2001*a*). Reduced freshwater input will cause major changes in the trophic structure and biological production of some systems (see Livingston 1997, 2000; Livingston *et al.* 1997, 2000). As coastal settlements expand in developing countries, the interception of freshwater flows for human use will escalate as well. The volume of diverted freshwater flows will vary in response to the agricultural, domestic, and industrial needs of specific regions. Since the 1950s, freshwater discharge from continents has declined by 15%, an amount equal to approximately 50% of the annual rate of sea level rise (Newman & Fairbridge 1986; Valiela 1995). This trend will be likely to continue.

Introduced species

Nearly every estuary is affected in some way by the introduction of exotic species (Carlton 1989; Carlton & Geller 1993). Some estuaries (e.g. San Francisco Bay) are more greatly impacted than other systems by the introduction of non-indigenous forms. Many of these organisms are widely dispersed from endemic environments via accidental transport in ballast water of ships (Nybakken 1988; Carlton 1989; Cohen & Carlton 1998). Other non-native species have been deliberately introduced for recreational or commercial reasons (Nybakken 1988; Day *et al.* 1989; Kennish 2000). Subsequent to invading a new environment, introduced species are often remarkably successful, outcompeting native populations and attaining high abundances. Ecological disruptions commonly arise by the introduction of invasive species, being manifested by changes in species composition, distribution, and abundance, as well as shifts in trophic organization (Carlton 1989; Cohen & Carlton 1998). In extreme cases, introduced species may totally displace native forms. Despite these alterations, the invaded environments usually remain highly productive (Cohen & Carlton 1998; San Francisco Estuary Project 1998).

It is anticipated that the introduction of exotic species to estuarine systems will increase worldwide during the 21st century because of greater shipping activity, especially in developing countries. With increased trade among nations, there will be more opportunity for the accidental dispersal of exotic species via ship hulls and accidental inclusion in shipments of mariculture materials (Valiela 1995; Cohen & Carlton 1998). Mariculture ventures in developing countries will also promote the introduction of non-native aquatic species from distant waters.

Sea level rise

Research during the past several decades has demonstrated that emissions of radiatively active substances from human activities, such as greenhouse gases or aerosol particles (e.g., carbon dioxide, methane, nitrous oxide, chlorofluorocarbons and other halocarbons), are affecting climate by raising the mean global surface air temperature (Wuebbles *et al.* 1999). Because of its large greenhouse effect, carbon dioxide is the most important agent of potential future climate warming. In addition to raising the mean surface air temperature, higher levels of carbon dioxide and other greenhouse gases will increase the mean rates of precipitation, evaporation, and sea level rise (Ledley *et al.* 1999). Computer models predict that significant biosphere changes will develop as a result (Kickert *et al.* 1999).

Global mean surface temperature has increased 0.6 ± 0.2°C during the past century, and it appears to be largely responsible for the concurrent rise in sea level of 10–25 cm (IPCC 2001). The 1980s and 1990s were the warmest decades on record; during this time, the rate of mean sea level rise was about 1.8 ± 0.3 mm yr^{-1} (Douglas 1991; Baltuck *et al.* 1996; IPCC 1996). Based on increases of greenhouse gases alone, models predict that, relative to the present, the global mean temperature will rise 0.5–1.7°C by 2040. Global sea level rise, in turn, is projected to increase 5.6–30.0 cm by 2040 and 9–88 cm by 2100 (IPCC 2001).

These global climate and sea level changes may affect estuaries in several ways, although the precise impacts are somewhat nebulous because the Earth's climate system is highly complex and dynamic and the responses of coastal zones to rising sea level may be obscured by human intervention. As conveyed by Jones (1994), the physical characteristics of the estuary will also play an important role. For example, factors that must be considered are the size and shape of the estuary, its orientation to fetch and local currents, the areal distribution of wetlands, the geology of the neighbouring watersheds, and land use in upland areas. Hence, local conditions will be most critical to the sea level effects observed in estuaries.

The greatest impact of global sea level rise will occur in estuaries located between the equator and 20°N and 20°S latitude. Detailed modelling efforts predict more extreme weather conditions in this region as a consequence of global warming, with greater frequency of severe droughts and periods of excessive precipitation. This climatic variability will elicit marked changes in freshwater inflow to estuaries over relatively short time spans, thereby altering sediment and nutrient inputs and the salinity regime in estuarine embayments. Greater incidence of storms and storm surges will also increase the likelihood of estuarine shoreline erosion, the loss of fringing wetland habitat, and coastal flooding. Rising sea level will exacerbate these conditions, leading to inland migration of wetlands (where possible) and the reduction of estuarine beach habitat. Upstream penetration of saltwater will cause the salinization of freshwater habitats and coastal aquifers. Drinking water supplies of coastal communities could be irreparably altered. In addition, the composition of freshwater plant communities near estuarine boundaries will gradually shift to plant communities dominated by salt-tolerant forms (Jones 1994). These habitat changes, in turn, will influence the species composition of animal communities living there.

Rising sea level could also reduce the areal coverage of intertidal estuarine habitat in some areas, depending on the slope of the adjoining land. This decrease in habitat would reduce the abundance of benthic intertidal fauna that provide forage for larger consumers, notably birds. Such changes are likely to significantly affect the trophic interactions of the system.

Jones (1994, p. 12) states that, 'In the context of a world-wide rise in sea level relative to the land, estuarine salt marshes and mangroves are, therefore, increasingly disadvantaged unless land can be made available above current high water for replacement salt marsh and mangrove habitats.' Alongi (1998) predicts that most mangroves will not be threatened by the projected rise in sea level, and they will readily adapt to a landward progression. Furthermore, the higher temperatures generated by global warming will shift the geographical distribution of mangroves somewhat, enabling them to extend into higher latitudes. In regard to saltmarshes, sediment accretion in many systems along the Atlantic coast of the USA is keeping pace with sea level rise, and thus these habitats will also be likely to survive or even expand over the next 25 years (Kennish 2001*b*). In other saltmarshes lacking comparable accretion, however, a 2.6–15.3 cm rise in sea level over the next 20 years could be devastating. Therefore, it is necessary to carefully examine the physical settings of each estuary and the configuration of the bordering wetlands to accurately predict the precise impacts of sea level rise.

Many individuals are predicting substantial changes in the areal distribution and ecology of the world's estuaries during the next century in response to rising sea level. Some of these changes will be mediated by the large-scale conversion of wetlands to open water habitat. There will be a widening and deepening of estuaries with a concomitant increase in tidal prism and tidal ranges. Salinity increases will result in changes in the species composition of biotic communities in many of these systems.

Coastal subsidence

Estuaries particularly vulnerable to the impacts of rising sea level are those characterized by high wave/tidal energies, high probabilities of major storms, erodible (soft sediment) substrate, shoreline retreat and subsidence (Kennish 2000). In some estuaries, land subsidence rather than eustatic sea level rise is the major process affecting the configuration of the basin as well as the inundation and stability of bordering wetland habitat. For example, areas along the Louisiana coast of the USA are experiencing extremely high submergence rates due to sediment compaction (DeLaune & Pezeshki 1994). In the region surrounding Galveston Bay, Texas, gas and oil withdrawal from subsurface formations has contributed greatly to land submergence and the expansion of the estuarine perimeter.

The effects of coastal lowland subsidence on estuaries due to withdrawal of groundwater, oil and gas will take on greater significance during the next 25 years both in developed and developing nations. It is anticipated that these effects will be most prevalent and conspicuous in urbanized systems near large metropolitan centres on modern coastal plains, where domestic and industrial demands for groundwater supplies are high. Examples of current problem areas with significant subsidence are Tokyo, Japan (4.6 m), Po Delta, Italy (3.2 m) and Houston, Texas, USA (2.7 m). Cities in Third World countries (e.g. Jakarta, Rangoon and Manila) are also experiencing significant submergence problems (Baeteman 1994).

The withdrawal of excessive amounts of groundwater from aquifer systems to meet the demands of coastal communities worldwide will create an array of problems in addition to lowland subsidence. For example, saltwater intrusion problems will become more common in these communities. Excessive groundwater removal will also reduce base flow to area streams resulting in diminished freshwater input to estuarine tributaries. These changes could lead to dramatic shifts in salinity and hydrologic regimes in affected estuaries, as well as major modifications of biotic communities.

In contrast to global changes in coastal systems associated with eustatic sea level rise, subsidence effects are local or regional in nature, often impacting areas surrounding coastal metropolitan centres. Estuaries affected by subsidence will typically exhibit accelerated shoreline retreat, loss or alteration of fringing wetlands, and an increase in open water habitat. Perimeter areas may also show greater rates of erosion. Changes in biotic communities will be most evident in intertidal and shallow subtidal zones, as well as in adjoining wetland habitat.

Debris/litter

Most floatable debris found in estuaries consists of improperly discarded waste material and litter from land-based sources, as well as boats. Because the coastal population is increasing rapidly in size and accessing estuarine environments more frequently, the volume of floatable debris in estuaries is on the rise. Most significant are plastics which now comprise the bulk of all litter identified in some systems. Up to 95% of the litter collected at some sites in the Bay of Biscay and an average of 80% of the debris identified in Seine Bay consist of plastics (Goldberg 1995). Among the great variety of debris types, plastics are most dangerous to estuarine environments because they seriously foul habitats, interfere with ecosystem functions, and pose a threat to birds, marine mammals, and other organisms that ingest or become entangled in them (Goldberg 1994; Kennish 1997). In addition, plastics essentially do not degrade in these environments and therefore are a long-term pollution hazard.

The use of plastic products in developed countries has risen to an all-time high. It is also increasing rapidly in developing countries, with peak amounts of the debris recorded in waters off large cities (Goldberg 1995). These trends will persist or even worsen during the next several decades concomitant with greater production and use of plastic materials worldwide.

CONCLUSIONS

Estuaries rank among the most heavily impacted aquatic systems on Earth. Because of their exceptional resources and economic value, these coastal systems are sites of intense human activity, and thus many serious environmental problems have arisen primarily in response to pollution inputs and habitat degradation. These environmental problems largely stem from activities of an ever-expanding coastal population, which is expected to approach six billion people by 2025. Of chief concern are impacts associated with habitat loss and alteration, nutrient enrichment, fisheries overexploitation, chemical contaminants, freshwater diversions, introduced species and sea level rise. These priority problems are expected to pose the greatest threats to the future health and viability of estuarine systems worldwide.

Although the literature is replete with studies of human impacts on estuaries, significant data gaps exist. For many estuarine systems, especially in developing countries, baseline data on biotic communities and anthropogenic stresses are not sufficient to address informational needs. Of particular note is the paucity of long-term, integrated, ecosystem-level studies to assess anthropogenic effects (Livingston 2000, 2001). These deficiencies are largely due to inadequate financial and scientific resources. The same problems extend to developed countries as well, however, where numerous estuaries also have not been adequately investigated for environmental quality. As discussed by Harding (1992), most measures of marine environmental quality have focused on the lower levels of biological organization such as organism health and population variables of indicator organisms. A major part of pollutant monitoring efforts, in turn, has involved studies of exposure, that is, measuring levels of contaminants in water, sediments, and organisms. In general, the problems are more pervasive and complex involving multiple stresses associated with physical habitat changes and an array of pollutants. In these cases, it is necessary to measure attributes of entire ecosystems (e.g. nutrient cycling, primary productivity, species diversity and contaminant effects across trophic levels) to effectively assess ecosystem changes. Data are required on causes and effects of stress at different scales of biological organization from the individual to the ecosystem.

In estuarine areas where high inputs of agricultural, domestic, and industrial wastes occur, the small number of population studies that usually has been conducted, especially in the benthos, is an impediment to assessing other pollution problems (Goldberg 1998). Even in estuaries intensively studied for years, data on the distribution of contaminants and the alteration of biotic communities and habitats may be incomplete (McIntyre 1995; Kennish 1997). In addition, the identification of the factors causing a pollution event is often challenging. For example, the conditions giving rise to algal blooms are problematic and frequently elusive (Goldberg 1995; Livingston 2000).

Some problems in estuaries require long-term systematic programmes of study to obtain relevant and essential data. Such is the case involving eutrophication in which measurements can be made over large areas and over time scales of decades (Goldberg 1995). Clearly, many investigations of anthropogenic impacts on estuaries are expensive and labour-intensive. Thus, data gaps are not uncommon.

It is evident that similar anthropogenic problems affect estuaries in developed as well as developing countries (Yap 1992; Kennish 2000). For example, the United Nations Environment Programme (1990) provided the following ranking of the most severe long-term anthropogenic problems plaguing the coastal environment of developing regions in south-east Asia:

(1) Habitat destruction (coral reefs and mangroves),
(2) Industrial pollution,
(3) Sewage pollution,
(4) Siltation/sedimentation,
(5) Agricultural pollution,
(6) Fisheries overexploitation,
(7) Hazardous waste,
(8) Oil pollution,
(9) Sea level rise,
(10) Coastal erosion,
(11) Algal blooms/red tides,
(12) Natural hazards.

However, developing countries are commonly overwhelmed by problems of overpopulation and poverty that hinder them in effectively addressing environmental issues. Hence, concerted international management initiatives are encouraged which will deal with these problems from a regional or global perspective (Töpfer 1990).

Estuarine problems and actions for environmental conservation are managed at national, regional, and international levels. A well-integrated, global, estuarine and marine management framework is lacking, although the United Nations Convention on the Law of the Sea, which went into effect in 1994, serves as a strong beginning. As emphasized by Côté (1992, p. 18), pertinent global environmental initiatives consist of 'a patchwork of international and regional conventions, protocols, and agreements, as well as national laws, regulations, and guidelines'. Among important international conventions and protocols that have been ratified are the International Convention for the Prevention of Pollution of the Sea by Oil (1954), the International Convention Relating to Intervention on the High Seas in Cases of Oil Pollution Casualties (1969), the International Convention on the Establishment of an International Fund for Oil Pollution Damage (1971), and the International Convention for the Prevention of Pollution by Ships (MARPOL) (1973) (amended by MARPOL 1973/1978). The Global Ocean Observing System, developed by the Intergovernmental Oceanographic Commission, the World Meterological Organization and the United Nations Environment Programme, is involved in formulating a coordinated marine pollution monitoring system.

Many countries have adopted integrated coastal zone management (ICZM) strategies that offer advantages over purely sectoral approaches. To achieve the conservation and sustainable multiple use of the coastal zone, ICZM provides a coordinated strategy for the allocation of environmental, sociocultural and institutional resources (Hildebrand & Norrena 1992). A key feature of ICZM is that it integrates and coordinates activities of existing coastal resource users (e.g. various levels of government, economic sectors and resource conservation programmes; Clark 1996). The goal is to improve the management of coastal land and marine resources significantly.

Nordstrom (2000) has shown that coastal management policy is typically a tiered structure of national, state/provincial, and local programmes. While it is essential to establish a sound, comprehensive, and global estuarine and marine environmental framework, local action must remain an integral long-term component to resolve anthropogenic problems. National or regional governments usually set policy and establish strategic planning, whereas local governments implement the programmes consistent with national policy (Hildebrand & Norrena 1992).

Hoss and Engel (1996) have proposed the following management objectives to mitigate pollution and habitat impacts in coastal ecosystems: (1) enactment, coordination and enforcement of land-use plans, so that development can be planned and orderly; (2) promotion and encouragement of efforts to control and limit non-point source run-off; (3) encouragement of contributors (e.g. homeowners, farmers, municipal governments and managers of golf courses) of inputs of pesticides, herbicides, and fertilizers to use them in a prudent manner; (4) increased efforts by the scientific community to demonstrate in an understandable way the importance of coastal wetland habitat to fisheries; and (5) reinforcement of the fact that everyone contributes to the contamination problems affecting the environment and that each is environmentally responsible. The ultimate goal is to mitigate or eliminate these problems while also promoting sustainable estuarine resources.

As the coastal population increases over the next two decades, anthropogenic impacts on estuaries will likely escalate unless effective management strategies are formulated. Best management practices must be initiated to protect freshwater and coastal wetlands, to minimize input of toxic agents, nutrients and disease vectors to receiving water bodies, to mollify physical alterations of river-estuary systems that could lead to adverse changes involving nutrient transfer and salinity distribution, and to maintain adequate freshwater input to sustain natural productivity and the important nursery function of the systems (Livingston 2001). It will also be advantageous to limit shoreline development, reduce invasive species and prevent overfishing. These measures may entail adapting strict management guidelines.

Remedial programmes to address water quality and habitat problems in estuaries must consider specific actions to be taken in both watershed areas and the estuarine basins. In most countries, improved point and non-point source pollution controls are needed to ameliorate water and sediment quality problems in estuaries. These controls may involve more stringent wastewater treatment processes and effluent discharge standards for industrial and municipal facilities, as well as more effective management practices in watersheds such as the use of alternative landscaping (e.g. replacing lawns with ground covers, shrubs, trees and other natural vegetation) and the modification of agricultural operations (e.g. application of new methods to reduce erosion, run-off and sedimentation). When pollution source controls cannot be used, structural controls (e.g. constructed wetlands, detention facilities and filtration basins) may be the best alternative, particularly when dealing with stormwater run-off problems.

For habitat impacts in tidal wetlands and subtidal areas of an estuary, restoration programmes should be instituted to revitalize the system. These programmes involve a construction process which consists of designing, building, and maintaining new wetland habitat. The goal is to produce a vegetation association (saltmarsh grasses, mangroves or seagrass beds) that will achieve ecological function rivalling natural communities in the system. The success of a restoration effort is contingent upon the development of both floral and faunal populations equivalent to those of reference wetland subsystems.

The revitalization of estuarine environments is a multifaceted, long-term problem that requires the focused attention of academic institutions, government agencies, local citizen groups, business and industry. These entities must work collaboratively to ensure that water quality, habitat, and biotic communities in estuaries are protected and maintained. Perhaps most importantly, the key to successful estuarine revitalization hinges on education programmes that inform the public of its responsibility as a steward of these valuable coastal systems.

ACKNOWLEDGEMENTS

This is Contribution Number 01-2 of the Jacques Cousteau National Estuarine Research Reserve and the Institute of Marine and Coastal Sciences, Rutgers University.

References

Alongi, D.M. (1998) *Coastal Ecosystem Processes*. Boca Raton, USA: CRC Press: 419 pp.

Anderson, D.M. (1989) Toxic algal blooms and red tides: a global perspective. In*: Red Tides: Biology, Environmental Science, and Toxicology*, ed. T. Okaichi, D.M. Anderson & T. Nemoto, pp. 11–16. Netherlands: Elsevier Scientific.

Atrill, M.J. & Thomes, K.M. (1995) Heavy metal concentrations in sediment from the Thames estuary, UK. *Marine Pollution Bulletin* **30**: 742–744.

Aubrey, D.G. (1993) Coastal erosion's influencing factors include development, dams, wells, and climate change. *Oceanus* **36**: 5–9.

Baeteman, C. (1994) Subsidence in coastal lowlands due to groundwater withdrawal: the geological approach. In: *Coastal Hazards: Perception, Susceptibility, and Mitigation*, *Journal of Coastal Research, Special Issue No. 12,* ed. C.W. Finkl Jr, pp. 61–75. Lawrence, USA: Coastal Education and Research Foundation.

Balls, P.W., Macdonald, A., Pugh, K. & Edwards, A.C. (1995) Long-term nutrient enrichment of an estuarine system: Ythan, Scotland (1958–1993). *Environmental Pollution* **90**: 311–321.

Baltuck, M., Dickey, J., Dixon, T. & Harrison, G.A. (1996) New approaches raise questions about future sea-level change. *Eos* **77**: 385–388.

Bennett, A., Bianchi, T.S. & Means, J.C. (2000) The effects of PAH contamination and grazing on the abundance and composition of microphytobenthos in saltmarsh sediments (Pass Fourchon, LA, USA): II. The use of plant pigments as biomarkers. *Estuarine, Coastal and Shelf Science* **50**: 425–439.

Berger, J.J., ed. (1990*) Environmental Restoration: Science and Strategies for Restoring the Earth*. Washington, DC, USA: Island Press: 265 pp.

Biggs, R.B. (1982) Estuaries. In: *The Encyclopedia of Beaches and Coastal Environments*, ed. M.L. Schwartz, pp. 393–394. Stroudsburg, USA: Hutchinson Ross.

Bijlsma, L., Ehler, C.N., Klein, R.J.L., Kulshrestha, S.M., McLean, R.F., Mimura, N., Nicholls, R.J., Nurse, L.A., Pérez Nieto, H., Stakhiv, E.Z., Turner, K. & Warrick, R.A. (1996) Coastal zones and small islands. In: *Climate Change 1995: Impacts, Adaptations and Mitigation of Climate Change*, *Contribution of WGII to the Second Assessment Report of the Intergovernmental Panel on Climate Change*, ed. R.T. Watson, M.C. Zinoyowera & R.H. Moss, pp. 289–324. Cambridge, UK: Cambridge University Press.

Boesch, D.F., Josselyn, M.N., Mehta, A.J., Morris, J.T., Nuttle, W.K., Simenstad, C.A. & Swift, D.J.P. (1994) *Scientific Assessment of Coastal Wetland Loss, Restoration, and Management in Louisiana*. *Journal of Coastal Research*, *Special Issue No. 20.* Lawrence, USA: Coastal Education and Research Foundation: 103 pp.

Bothner, M.H., Buchholtz ten Brink, M. & Manheim, F.T. (1998) Metal concentrations in surface sediments of Boston Harbor – changes with time. *Marine Environmental Research* **45**: 127–155.

Botsford, L.W., Castilla, J.C. & Peterson, C.H. (1997) The management of fisheries and marine ecosystems. *Science* **277**: 509–515.

Bricker, S.B., Clement, C.G., Pirhalla, D.E., Orlando, S. P. & Farrow, D.R.G. (1999) National estuarine eutrophication assessment: effects of nutrient enrichment in the nation's estuaries. Technical Report, National Oceanic and Atmospheric Administration, National Ocean Service, Special Projects Office and the National Centers for Coastal Ocean Science, Silver Spring, Maryland, USA: 71 pp.

Britsch, L.D. & Kemp, E.B. (1990) Land loss rates: Mississippi Delta Plain. Technical Report, GL/90/2, US Army Corps of Engineers, Vicksburg, Mississippi, USA.

Bryan, G.W. & Langston, W.J. (1992) Bioavailability, accumulation, and effects of heavy metals in sediments with special reference to United Kingdom estuaries: a review. *Environmental Pollution* **76**: 89–131.

Bukata, R.P., Jerome, J.H., Kondratyev, K.Y. & Pozdnyakov, D.V. (1995*) Optical Properties and Remote Sensing of Inland and Coastal Waters*. Boca Raton, USA: CRC Press: 362 pp.

Burwood, R. & Speers, G.C. (1974). Physical–chemical processes affecting the distribution of polluting oil in the marine environment. *Estuarine and Coastal Marine Science* **2**: 117–127.

Carlton, J.T. (1989) Man's role in changing the face of the oceans: biological conservation of near-shore environments. *Conservation Biology* **3**: 265–273.

Carlton, J.T. & Geller, J.B. (1993) Ecological roulette: the global transport of nonindigenous marine organisms. *Science* **261**: 78–82.

Carman, K.R., Fleeger, J.W., Means, J.C., Pomarico, S.M. & McMillin, D.J. (1995) Experimental investigation of the effects of polynuclear aromatic hydrocarbons on an estuarine sediment food web. *Marine Environmental Research* **40**: 289–318.

Cearreta, A., Irabien, M.J., Leorri, E., Yusta, I., Croudace, I.W. &

Cundy, A.B. (2000) Recent anthropogenic impacts on the Bilbao estuary, northern Spain: geochemical and macrofaunal evidence. *Estuarine, Coastal and Shelf Science* **50**: 571–592.

Chapman, P.M., Paine, M.D., Arthur, A.D. & Taylor, L.A. (1996) A triad study of sediment quality associated with a major, relatively untreated marine sewage discharge. *Marine Pollution Bulletin* **32**: 47–64.

Ciupek, R.B. (1986) Protecting wetlands under Clean Water Act 404: EPA's conservation policy on mitigation. *Natural Wetlands Newsletter* **8**: 12–15.

Claridge, P.N., Potter, I.C. & Hardisty, M.W. (1986) Seasonal changes in movements, abundance, size composition, and diversity of the fish fauna of the Severn estuary. *Journal of the Marine Biological Association of the United Kingdom* **66**: 229–242.

Clark, J.R. (1996) *Coastal Zone Management Handbook*. Boca Raton, USA: Lewis Publishers: 694 pp.

Clark, R.B. (1992) *Marine Pollution*, Third Edition. Oxford, UK: Clarendon Press: 427 pp.

Cohen, A.N. & Carlton, J.T. (1998) Accelerating invasion rate in a highly invaded estuary. *Science* **279**: 555–562.

Costello, M.J. & Read, P. (1994) Toxicity of sewage sludge to marine organisms: a review. *Marine Environmental Research* **37**: 23–42.

Côté, R.P. (1992) Marine environmental management: status and prospects. *Marine Environmental Pollution* **25**: 18–22.

Coull, B.C. & Bell, S.S. (1979) Perspectives of marine meiofaunal ecology. In: *Ecological Processes in Coastal Marine Systems*, ed. R.J. Livingston, pp. 189–217. New York, USA: Plenum Press.

Daskalakis, K.D. & O'Connor, T.P. (1995) Distribution of chemical contaminants in US coastal and estuarine sediment. *Marine Environmental Research* **40**: 381–398.

Day, J.W., Jr, Hall, C.A.S., Kemp, W.M. & Yáñez-Arancibia, A., eds. (1989) *Estuarine Ecology*. New York, USA: John Wiley & Sons: 558 pp.

Dederen, L.H.T. (1992) Marine eutrophication in Europe: similarities and regional differences in appearance. In: *Marine Coastal Eutrophication*, ed. R.A. Vollenweider, R. Marchetti & R. Viviani, pp. 673–682. Amsterdam, the Netherlands: Elsevier Science.

De Groot, A.J. (1995) Metals and sediments: a global perspective. In*: Metal Contaminated Aquatic Sediments*, ed. H.E. Allen, pp. 1–20. Chelsea, USA: Ann Arbor Press.

De Jonge, V.N. (1990) Response of the Dutch Wadden Sea ecosystem to phosphorus discharges from the River Rhine. *Hydrobiologia* **195**: 49–62.

DeLaune, R.D. & Pezeshki, S.R. (1994) The influence of subsidence and saltwater intrusion on coastal marsh stability: Louisiana Gulf coast, USA. In: *Coastal Hazards: Perception, Susceptibility, and Mitigation*, *Journal of Coastal Research*, *Special Issue No. 12,* ed. C.W. Finkl Jr, pp. 77–89. Lawence, USA: Coastal Education and Research Foundation.

Dickhut, R.M. & Gustafson, K.E. (1995) Atmospheric inputs of selected polycyclic aromatic hydrocarbons and polychlorinated biphenyls to southern Chesapeake Bay. *Marine Pollution Bulletin* **30**: 385–396.

Dobson, A.P., Bradshaw, A.D. & Baker, A.J.M. (1997) Hopes for the future: restoration ecology and conservation biology. *Science* **277**: 515–522.

Douglas, B.C. (1991) Global sea-level rise. *Journal of Geophysical Research* **96**: 6981–6985.

Drake, P., Baldó, F., Sáenz, V. & Arias, A.M. (1999) Macrobenthic community structure in estuarine pollution assessment on the Gulf of Cádiz (SW Spain): is the phylum-level meta-analysis approach applicable? *Marine Pollution Bulletin* **38**: 1038–1047.

Ducklow, H.W. & Shiah, F.-K. (1993) Bacterial production in estuaries. In*: Aquatic Microbiology: An Ecological Approach*, ed. T. E. Ford, pp. 261–277. Boston, USA: Blackwell Scientific Publications.

Edgar, G.J. & Barrett, N.S. (2000) Effects of catchment activities on macrofaunal assemblages in Tasmanian estuaries. *Estuarine, Coastal and Shelf Science* **50**: 639–654.

Eganhouse, R.P. & Sherblom, P.M. (2001) Anthropogenic organic contaminants in the effluent of a combined sewer overflow: impact on Boston Harbor. *Marine Environmental Research* **51**: 51–74.

Eisler, R. (1987) Polycyclic aromatic hydrocarbon hazards to fish, wildlife, and invertebrates: a synoptic review. Biological Report 85(1.11), US Fish and Wildlife Service, Washington, DC, USA.

Eisma, D. (1998) *Intertidal Deposits: River Mouths, Tidal Flats, and Coastal Lagoons*. Boca Raton, USA: CRC Press: 525 pp.

Emery, K.O. (1967) Estuaries and lagoons in relation to continental shelves. In: *Estuaries*, ed. G.H. Lauff, pp. 9–11. Washington, DC, USA: Publication 83, American Association for the Advancement of Science.

Food and Agnculture Organization (1996) The state of world fisheries and aquaculture 1996. Technical report, Food and Agriculture Organization of the United Nations, Rome, Italy: 36 pp.

Fernandes, M.B., Sicre, M.-A., Boireau, A. & Tronczynski, J. (1997) Polycyclic hydrocarbon (PAH) distributions in the Seine River and its estuary. *Marine Pollution Bulletin* **34**: 857–862.

Finkl, C.W., Jr, ed. (1994) *Coastal Hazards: Perception, Susceptibility, and Mitigation. Journal of Coastal Research*, *Special Issue No. 12.* Lawence, USA: Coastal Education and Research Foundation: 372 pp.

Gagliano, S.M., Meyer-Arendt, K.J. & Wickerm, K.M. (1981) Habitat loss in the Mississippi river deltaic plain. *Transactions of the Gulf Coast Association of Geological Societies* **31**: 295–300.

Galloway, J.N., Levy, H., II & Kasibhatia, P.S. (1994) Year 2020: consequences of population growth and development on deposition of oxidized nitrogen. *Ambio* **23**: 123–123.

GESAMP (1982) Scientific criteria for the selection of waste disposal sites at sea. Reports and Studies No. 16, Inter-Governmental Maritime Consultative Organization, London, UK.

GESAMP (1990) The state of the marine environment. Report No. 39, GESAMP, IMP, London, UK.

GESAMP (1993) Report of the twenty-third session. Reports and Studies No. 51, Food and Agriculture Organization, London, UK.

GESAMP (1995) Report of the twenty-fifth session. Reports and Studies No. 56, Food and Agriculture Organization, London, UK.

Goldberg, E.D. (1994) Diamonds and plastics are forever? *Marine Pollution Bulletin* **25**: 1–4.

Goldberg, E.D. (1995) Emerging problems in the coastal zone for the twenty-first century. *Marine Pollution Bulletin* **31**: 152–158.

Goldberg, E.D. (1998) Marine pollution – an alternative view. *Marine Pollution Bulletin* **36**: 112–113.

Goñi, R. (1998) Ecosystem effects of marine fisheries: an overview. *Coastal & Ocean Management* **40**: 37–64.

Hall, S.J. (1999) *The Effects of Fishing on Marine Ecosystems and Communities*. Oxford, UK: Blackwell Science: 274 pp.

Hallegreaff, G.M., Anderson, D.M. & Cembella, A.D. (1995*) Manual on Harmful Marine Microalgae*. Paris, France: IOC Manuals and Guides No. 33, UNESCO.

Hameedi, M.J. (1997) Strategy for monitoring the environment in the coastal zone. In: *Coastal Zone Management Imperative for Maritime Developing Nations*, ed. B.U. Haq, pp. 111–142. Rotterdam, the Netherlands: Kluwer Academic Publishers.

Harding, L.E. (1992) Measures of marine environmental quality. *Marine Pollution Bulletin* **25**: 23–27.

Hildebrand, L.P. & Norrena, E. J. (1992) Approaches and progress toward effective integrated coastal zone management. *Marine Pollution Bulletin* **25**: 94–97.

Holmer, M. (1999) The effect of oxygen depletion on anaerobic organic matter degradation in marine sediments. *Estuarine, Coastal and Shelf Science* **48**: 383–390.

Hoss, D.E. & Engel, D.W. (1996) Sustainable development in the southeastern coastal zone: environmental impacts on fisheries. In: *Sustainable Development in the Southeastern Coastal Zone*, ed. F. J. Vernberg, W.B. Vernberg & T. Siewicki, pp. 171–186. Columbia, USA: University of South Carolina Press.

Howarth, R.W., Anderson, D.M., Church, T.M., Greening, H., Hopkinson, C.S., Huber, W.C., Marcus, N., Nainman, R.J., Segerson, K., Sharpley, A.N. & Wiseman, W.J. (2000) *Clean Coastal Waters: Understanding and Reducing the Effects of Nutrient Pollution*. Washington DC, USA: Ocean Studies Board and Water Science and Technology Board, National Academy Press: 391 pp.

Howells, G., Calamari, D., Gray, J. & Wells, P.G. (1990) An analytical approach to assessment of long-term effects of low levels of contaminants in the marine environment. *Marine Pollution Bulletin* **21**: 371–377.

Ingrid, G., Andersen, T. & Vadstein, O. (1996) Pelagic food webs and eutrophication of coastal waters: impact of grazers on algal communities. *Marine Pollution Bulletin* **33**: 22–35.

IPCC (1996) *Climate Change 1995: Impacts, Adaptations, and Mitigation of Climate Change. Contribution of WGII to the Second Assessment Report of the Intergovernmental Panel on Climate Change*, ed. R.T. Watson, M.C. Zinoyowera & R.H. Moss. Cambridge, UK: Cambridge University Press: 878 pp.

IPCC (2001) *Climate Change 2001: Impacts, Adaptation, and Vulnerability*, ed. J.J. McCarthy, O.F. Canziani, N.A. Leary, D.J. Dokken & K.S. White. Cambridge, UK: Cambridge University Press: 1000 pp.

Jackson, L.L., Lopoukhine, N. & Hillyard, D. (1995) Ecological restoration: a definition and comments. *Restoration Ecology* **3**: 71–75.

Jennings, S. & Lock, J.M. (1996) Population and ecosystem effects of fishing. In*: Reef Fisheries*, ed. N.V.C. Polunin & C.M. Roberts, pp. 193–218. London, UK: Chapman and Hall.

Jennings, S. & Kaiser, M.J. (1998) The effects of fishing on marine ecosystems. *Advances in Marine Biology* **34**: 201–352.

Jones, G. (1994) Global warming, sea level change and the impact on estuaries. *Marine Pollution Bulletin* **28**: 7–14.

Kannan, K., Smith, R.G., Jr, Lee, R.F., Windom, H.L., Heitmuller, P.T., Macauley, J.M. & Summers, J.K. (1998) Distribution of total mercury and methylmercury in water, sediment, and fish from South Florida estuaries. *Archives of Environmental Contamination and Toxicology* **34**: 109–118.

Kemp, P.F. (1990) The fate of benthic bacterial production. *Reviews in Aquatic Science* **2**: 109–124.

Kennish, M.J. (1986) *Ecology of Estuaries: Physical and Chemical Aspects*. Boca Raton, USA: CRC Press: 391 pp.

Kennish, M.J. (1992) *Ecology of Estuaries: Anthropogenic Effects*. Boca Raton, USA: CRC Press: 494 pp.

Kennish, M.J., ed. (1997) *Practical Handbook of Estuarine and Marine Pollution*. Boca Raton, USA: CRC Press: 524 pp.

Kennish, M.J. (1998*a*) *Pollution Impacts on Marine Biotic Communities*. Boca Raton, USA: CRC Press: 310 pp.

Kennish, M.J. (1998*b*) Trace metal–sediment dynamics in estuaries: pollution assessment. *Reviews in Environmental Contamination and Toxicology* **155**: 69–110.

Kennish, M.J., ed. (2000) *Estuary Restoration and Maintenance: The National Estuary Programme*. Boca Raton, USA: CRC Press: 359 pp.

Kennish, M.J., ed. (2001*a*) *Practical Handbook of Marine Science*, Third Edition. Boca Raton, USA: CRC Press: 876 pp.

Kennish, M.J. (2001*b*) Coastal saltmarsh systems: a review of anthropogenic impacts. *Journal of Coastal Research* **17**: 731–748.

Kennish, M.J. & Lutz, R.A., eds. (1984) *Ecology of Barnegat Bay, New Jersey*. New York, USA: Springer-Verlag: 396 pp.

Kerby, C. (1977) Life in the bottom-meiobenthos. In: *Coastal Ecosystem Management*, ed. J.R. Clark, pp. 656–677. New York, USA: John Wiley & Sons.

Kickert, R.N., Tonella, G., Simonov, A. & Krupa, S.V. (1999) Predictive modeling of effects under global change. *Environmental Pollution* **100**: 87–132.

Kjerfve, B. (1989) Estuarine geomorphology and physical oceanography. In: *Estuarine Ecology*, ed. J.W. Day, Jr, C.A.S. Hall, W. M. Kemp & A. Yañez-Arancibia, pp. 47–78. New York, USA: John Wiley & Sons.

Knox, G.A. (2001) *The Ecology of Seashores*. Boca Raton, USA: CRC Press: 557 pp.

Kusler, J.A. & Kentula, M.E., eds. (1990*) Wetland Creation and Restoration: The Status of the Science*. Washington, DC, USA: Island Press: 477 pp.

Lalli, C.M. & Parsons, T.R. (1993) *Biological Oceanography: An Introduction*. New York, USA: Pergamon Press: 301 pp.

Ledley, T.S., Sundquist, E.T., Schwartz, S.E., Hall, D.K., Fellows, J.D. & Killeen, T.L. (1999) Climate change and greenhouse gases. *Eos* **80**: 453–458.

Levinton, J.S. (1982) *Marine Ecology*. Englewood Cliffs, USA: Prentice-Hall: 526 pp.

Lewis, R.R. (1994) Enhancement, restoration, and creation of coastal wetlands. In: *Applied Wetlands Science and Technology*, ed. D. M. Kent, pp. 167–191. Boca Raton, USA: Lewis Publishers.

Livingston, R.J. (1996) Eutrophication in estuaries and coastal systems: relationships of physical alterations, salinity stratification, and hypoxia. In*: Sustainable Development in the Southeastern Coastal Zone*, ed. F.J. Vernberg, W.B. Vernberg & T. Siewicki, pp. 285–318. Columbia, USA: University of Carolina Press.

Livingston, R.J. (1997) Trophic response of estuarine fishes to long-term changes of river run-off. *Bulletin of Marine Science* **60**: 984–1004.

Livingston, R.J. (2000) *Eutrophication Processes in Coastal Systems: Origin and Succession of Plankton Blooms and Secondary Production in Gulf Coast Estuaries*. Boca Raton, USA: CRC Press: 327 pp.

Livingston, R.J. (2001) Management of the Apalachicola River estuary: relationships of river flow and river-bay productivity. Technical Report, Florida State University, Tallahassee, Florida, USA.

Livingston, R.J., Lewis, F.G., Woodsum, G.C., Niu, X.-F., Galperin, B., Huang, W., Christensen, J.D., Monaco, M.E., Battista, T.A., Klein, C.J., Howell, R.L., IV & Ray, G. L. (2000) Modelling oyster population response to variation in freshwater input. *Estuarine, Coastal and Shelf Science* **50**: 655–672.

Livingston, R.J., Niu, X.-F., Lewis, F.G. & Woodsum, G.C. (1997)

Freshwater input to a Gulf estuary: long-term control of trophic organization. *Ecological Applications* **7**: 277–299.

Locarnini, S.J.P. & Presley, B.J. (1996) Mercury concentrations in benthic organisms from a contaminated estuary. *Marine Environmental Research* **41**: 225–239.

Long, E.R., Robertson, A., Wolfe, D.A., Hameedi, J. & Sloane, G.M. (1996) Estimates of the spatial extent of sediment toxicity in major US estuaries. *Environmental Science and Technology* **30**: 3585–3592.

Malone, T.C., Màlej, A. & Smodlaka, N. (1996) Trends in land-use, water quality, and fisheries: a comparison of the Northern Adriatic Sea and the Chesapeake Bay. *Periodicum Biologorum* **98**: 137–148.

Mann, K.H. (1982) *Ecology of Coastal Waters: A Systems Approach*. Berkeley, USA: University of California Press: 405 pp.

Matthiessen, P., Bifield, S., Jarrrett, F., Kirby, M.F., Law, R.J., McMinn, W.R., Sheahan, D.A., Thain, J.E. & Whale, G.F. (1998) An assessment of sediment toxicity in the River Tyne estuary, UK by means of bioassays. *Marine Environmental Research* **45**: 1–15.

McComb, A.J. (1995) *Eutrophic Shallow Estuaries and Lagoons*. Boca Raton, USA: CRC Press: 240 pp.

McDowell, J.E. (1993) How marine animals respond to toxic chemicals in coastal ecosystems. *Oceanus* **36**: 56–63.

McIntyre, A.D. (1992) The current state of the oceans. *Marine Pollution Bulletin* **25**: 1–4.

McIntyre, A.D. (1995) Human impact on the oceans: the 1990s and beyond. *Marine Pollution Bulletin* **31**: 147–151.

Minello, T.J. & Webb, J.W. (1997) Use of natural and created *Spartina alterniflora* saltmarshes by fishery species and other aquatic fauna in Galveston Bay, Texas, USA. *Marine Ecology Progress Series* **151**: 165–179.

Monroe, M.W. & Kelly, J. (1992) State of the estuary. Technical Report, San Francisco Estuary Project, Oakland, California, USA.

Moriarty, D.J.W. (1986) Measurement of microbial growth rates in aquatic systems using rates of nucleic acid synthesis. *Advances in Microbial Ecology* **9**: 245–272.

Moy, L.D. & Levin, L.A. (1991) Are *Spartina* marshes a replaceable resource? A functional approach to evaluation of marsh creation efforts. *Estuaries* **14**: 1–16.

National Academy Press (1993) *Managing Wastewater in Coastal Urban Areas*. Washington DC, USA: National Academy Press: 451 pp.

National Estuary Programme (1997*a*) Nutrient overloading. Technical Report, US Environmental Protection Agency, Washington, DC, USA: 21 pp.

National Estuary Programme (1997*b*) Management approaches being used to address critical issues. Technical Report, US Environmental Protection Agency, Washington, DC, USA: 16 pp.

National Estuary Programme (1997*c*) Declines in fish and wildlife populations. Technical Report, US Environmental Protection Agency, Washington, DC, USA: 22 pp.

National Research Council (1994) *Priorities for Coastal Ecosystem Science*. Washington DC, USA: Academy Press: 106 pp.

Newman, W.S. & Fairbridge, R.W. (1986) The management of sea level rise. *Nature* **320**: 319–321.

Nienhuis, P.H. (1992) Eutrophication, water management, and the functioning of Dutch estuaries and coastal lagoons. *Estuaries* **15**: 538–548.

Nixon, S.W. (1995) Coastal marine eutrophication: a definition, social causes, and future consequences. *Ophelia* **41**: 199–219.

Nordstrom, K.F. (2000) *Beaches and Dunes of Developed Coasts*. Cambridge, UK: Cambridge University Press: 338 pp.

Nybakken, J.W. (1988) *Marine Biology: An Ecological Approach*, Second Edition. New York, USA: Harper & Row.

O'Connor, T.P. & Beliaeff, B. (1995) Recent trends in coastal environmental quality: results from the Mussel Watch Project. NOAA Technical Report, National Oceanic and Atmospheric Administration, Rockville, Maryland USA.

O'Connor, T.P. & Paul, J.F. (2000) Misfit between sediment toxicity and chemistry. *Marine Pollution Bulletin* **40**: 59–64.

Odum, E.P. & Biever, L.J. (1984). Resource quality, mutualism, and energy partitioning in food chains. *American Naturalist* **124**: 360–374.

Office of Technology Assessment (1986). Pollutant discharges to surface waters in coastal regions. USA Congress, Office of Technology Assessment, Washington, DC, USA.

Pezeshki, S.R., Hester, M.W., Lin, Q. & Nyman, J. A. (2000) The effects of oil spill and clean-up on dominant US Gulf coast marsh macrophytes: a review. *Environmental Pollution* **108**: 129–139.

Pinet, P.R. (2000) *Invitation to Oceanography*, Second Edition. Boston, USA: Jones and Bartlett Publishers: 555 pp.

Pinnegar, J.K., Polunin, N.V.C., Francour, P., Badalamenti, F., Chemello, R., Harmelin-Vivien, M.-L., Hereu, B., Milazzo, M., Zabala, M., D'Anna, G. & Pipitone, C. (2000) Trophic cascades in benthic marine ecosystems: lessons for fisheries and protected-area management. *Environmental Conservation* **27**: 179–200.

Postel, S. (2000) Entering an era of water scarcity: the challenges ahead. *Ecological Applications* **10**: 941–948.

Pritchard, D.W. (1967) What is an estuary: physical standpoint. In: *Estuaries*, ed. G.H. Lauff, pp. 3–5. Washington, DC, USA: American Association for the Advancement of Science, Publication 83.

Roberts, C.M. & Hawkins, J.P. (1999) Extinction risk in the sea. *Trends in Ecology and Evolution* **14**: 241–246.

Rozan, T.F. & Benoit, G. (1999) Heavy metal removal efficiencies in a river-marsh system estimated from patterns of metal accumulation in sediments. *Marine Environmental Research* **48**: 335–351.

Sacco, J., Seneca, E.D. & Wentworth, T.R. (1994) Infaunal community development of artificially established saltmarshes in North Carolina. *Estuaries* **17**: 489–500.

Sánchez Lizaso, J.L., Goñi, R., Reñones, O., García C., Galzin, R., Bayle, J.T., Sánchez J., Pérez, R. & Ramos, A.A. (2000) Density dependence in marine protected populations: a review. *Environmental Conservation* **27**: 144–158.

San Francisco Estuary Project (1998) State of the estuary: 1992–1997. Technical Report, San Francisco Estuary Project, Oakland, California, USA.

Scatolini, S.R. & Zedler, J.B. (1996) Epibenthic invertebrates of natural and constructed marshes of San Diego Bay. *Wetlands* **16**: 24–37.

Shabman, L. (1996) Land settlement, public policy, and the environmental future of the southeast coast. In*: Sustainable Development in the Southeastern Coastal Zone*, ed. F.J. Vernberg, W.B. Vernberg & T. Siewicki, pp. 7–24. Columbia, USA: University of Carolina Press.

Shaw, D.G. & Day, R.H. (1994) Color-and form-dependent loss of plastic micro-debris from the North Pacific Ocean. *Marine Pollution Bulletin* **28**: 39–45.

Sissenwine, M.P. & Rosenberg, A.A. (1996) Marine fisheries at a critical juncture. In: *Oceanography: Contemporary Readings in Ocean Sciences*, Third Edition, ed. R. G. Pirie, pp. 293–302. New York, USA: Oxford University Press.

Smayda, T.J. (1990) Novel and nuisance phytoplankton blooms in the sea: evidence for a global epidemic. In: *Toxic Marine Phytoplankton*, ed. E. Granéli, L. Sundström, L. Edler & D.M. Anderson, pp. 29–40. Amsterdam, the Netherlands: Elsevier Scientific.

Smith, V.H., Tilman, G.D. & Nekola, J.C. (1999) Eutrophication: impacts of excess nutrient inputs on freshwater, marine, and terrestrial ecosystems. *Environmental Pollution* **100**: 179–196.

Tapp, J.F., Shillabeer, N. & Ashman, C.M. (1993) Continued observations of the benthic fauna of the industrialized Tees estuary, 1979–1990. *Journal of Experimental Marine Biology and Ecology* **172**: 67–82.

Taylor, P. (1993) The state of the marine environment: a critique of the work and role of the Joint Group of Experts on Scientific Aspects of Marine Pollution (GESAMP*). Marine Pollution Bulletin* **26**: 120–127.

Thomas, C.A. & Bendell-Young, L.I. (1999) The significance of diagenesis versus riverine input in contributions to the sediment geochemical matrix of iron and manganese in an intertidal region. *Estuarine, Coastal and Shelf Science* **48**: 635–647.

Thompson, B., Anderson, B., Hunt, J., Taberski, K. & Phillips, B. (1999) Relationships between sediment contamination and toxicity in San Francisco Bay. *Marine Environmental Research* **48**: 285–309.

Tiner, R.W. & Burke, D.G. (1995) Wetlands of Maryland. US Fish and Wildlife Service, Region 5, Hadley, Massachusetts and Maryland Department of Natural Resources, USA.

Töpfer, K. (1990) Assistance to developing countries in the implementation of international rules. *Marine Policy* **14**: 259–263.

Turner, A. (2000) Trace metal contamination in sediments from UK estuaries: an empirical evaluation of the role of hydrous iron and manganese oxides. *Estuarine, Coastal and Shelf Science* **50**: 355–371.

Turner, R.E. & Lewis, R.R. (1997) Hydrologic restoration of coastal wetlands. *Wetlands Ecology and Management* **4**: 65–72.

United Nations Environment Programme (1990) Report of the fourth meeting of experts on the east Asian seas Action Plan. Quezon City, Philippines, 3–6 December 1990. UNEP 9OCA/EAS WG.4/6. UNEP OCA/PAC, Nairobi, Kenya.

United Nations Environment Programme (1992) Persistent organic pollutants. In: *The World Environment 1972–1992*, ed. M.K. Tolba & O.A. El-Kholy, pp. 257–277. London, UK: Chapman and Hall.

US Environmental Protection Agency (1986) National water quality inventory. Technical Report, US Environmental Protection Agency, Washington, DC, USA.

Valiela, I. (1995) *Marine Ecological Processes*, Second Edition. New York, USA: Springer-Verlag: 686 pp.

Valiela, I., Kremer, J., Lajtha, K., Geist, M., Seely, B., Brawley, J. & Sham, C.H. (1997) Nitrogen loading from coastal watersheds to receiving estuaries: new method and application. *Ecological Applications* **7**: 358–380.

Viles, H. & Spencer, T. (1995*) Coastal Problems, Geomorphology, Ecology and Society of the Coast*. London, UK: Edward Arnold: 350 pp.

Vivian-Smith, G. (2001) Developing a framework for restoration. In*: Handbook for Restoring Tidal Wetlands*, ed. J. B. Zedler, pp. 39–88. Boca Raton, USA: CRC Press.

Wania, F., Axelman, J. & Broman, D. (1998) A review of processes involved in the exchange of persistent organic pollutants across the air–sea interface. *Environmental Pollution* **102**: 3–23.

Warwick, R.M., Joint, I.R. & Radford, P.J. (1979) Secondary production of the benthos in an estuarine environment. In: *Ecological Processes in Coastal Environments*, ed. R.L. Jefferies & A.J. Davy, pp. 429–450. Oxford, UK: Blackwell Scientific.

Weber, P. (1994) It comes down to the coasts. *World Watch* March/April: 20–29.

Weinstein, J.E. (1996) Anthropogenic impacts on saltmarshes – a review. In*: Sustainable Development in the Southeastern Coastal Zone*, ed. F.J. Vernberg, W.B. Vernberg & T. Siewicki, pp. 135–170. Columbia, USA: University of Carolina Press.

Weis, P., Weis, J.S. & Proctor, T. (1993) Copper, chromium, and arsenic in estuarine sediments adjacent to wood treated with chromated-copper-arsenate (CCA). *Estuarine, Coastal Shelf Science* **36**: 71–79.

Weston, D.P. (1990) Quantitative examination of macrobenthic community changes along an organic and organic-enrichment gradient. *Marine Ecology Progress Series* **61**: 233–247.

White, W.A. & Tremblay, T.A. (1995) Submergence of wetlands as a result of human-induced subsidence and faulting along the upper Texas Gulf coast. *Journal of Coastal Research* **11**: 788–807.

Whitfield, A.K. (1998) *Biology and Ecology of Fishes in Southern African Estuaries*. Grahamstown, South Africa: Ichthyological Monographs of the J.L.B. Smith Institute of Ichthyology, No. 2: 223 pp.

Wild, S.R. & Jones, K.C. (1995) Polynuclear aromatic hydrocarbons in the United Kingdom environment: a preliminary source inventory and budget. *Environmental Pollution* **88**: 91–108.

Williams, G.D. & Desmond, J.S. (2001) Restoring assemblages. In: *Handbook for Restoring Tidal Wetlands*, ed. J.B. Zedler, pp. 235–269. Boca Raton, USA: CRC Press.

Windom, H.L. (1992) Contamination of the marine environment from land-based sources. *Marine Pollution Bulletin* **25**: 32–36.

Wolanski, E. & Chappell, J. (1996) The response of tropical Australian estuaries to a sea level rise. *Journal of Marine Systems* **7**: 267–279.

Wren, C.D., Harris, S. & Harttrup, N. (1995) Ecotoxicology of mercury and cadmium. In: *Handbook of Ecotoxicology*, ed. D.J. Hoffman, B.A. Rattner, G.A. Burton Jr & J. Cairns Jr, pp. 392–423. Boca Raton, USA: Lewis Publishers.

World Resources Instute (1998) *World Resources 1998–1999*. New York, USA: Oxford University Press: 384 pp.

Wright, L.D. (1995) *Morphodynamics of Inner Continental Shelves*. Boca Raton, USA: CRC Press: 241 pp.

Wuebbles, D.J., Jain, A., Edmonds, J., Harvey, D. & Hayhoe, K. (1999) Global change: state of the science. *Environmental Pollution* **100**: 57–86.

Yap, H.T. (1992) Marine environmental problems: experiences of developing regions. *Marine Pollution Bulletin* **25**: 37–40.

Zedler, J.B. (1996) Coastal mitigation in southern California: the need for a regional restoration strategy. *Ecology Applications* **6**: 84–93.

Zedler, J.B. & Callaway, J.C. (1999) Tracking wetland restoration: do mitigation sites follow desired trajectories? *Restoration Ecology* **7**: 69–73.

Zedler, J.B. (2001) *Handbook for Restoring Tidal Wetlands*. Boca Raton, USA: CRC Press: 439 p.