



Ecohydrology as a new tool for sustainable management of estuaries and coastal waters

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Abstract

Throughout the world, estuaries and coastal waters have experienced degradation. Present proposed remedial measures based on engineering and technological fix are not likely to restore the ecological processes of a healthy, robust estuary and, as such, will not reinstate the full beneficial functions of the estuary ecosystem. The successful management of estuaries and coastal waters requires an ecohydrology-based, basin-wide approach. This necessitates changing present practices by official institutions based on municipalities or counties as an administrative unit, or the narrowly focused approaches of managers of specific activities (e.g., farming and fisheries, water resources, urban and economic developments, wetlands management and nature conservationists). Without this change in thinking and management concept, estuaries and coastal waters will continue to degrade, whatever integrated coastal management plans are implemented. To help in this process of change there is a need to (1) develop a profound understanding of the effects of biota and biotic processes on mediating estuary response to changing hydrology, sediment and nutrient flux and of the biota on hydrology at the river basin scale, and (2) to develop science-based remediation measures at the river basin scale, with elements of ecohydrology and phytotechnology at their core, to strengthen the ability of the biota to sustain and adapt to human-induced stresses.

Introduction: degraded estuaries and coastal waters

Throughout human history, the coastal plains and lowland river valleys have usually been the most populated areas over the world. This is partially

due to their very high biological productivity sustaining a high level of food production. It is also due to the use of the rivers and estuaries as transport routes, fundamental for economic and social development and improving quality of life. At present, about 60% of the world's population live

along the estuaries and the coast (Lindeboom 2002). Coastal waters, including those covering continental shelves, supply about 90% of the global fish catch. The increase of human populations in the river basins, from natural growth and internal migration within the hinterland, has resulted in a doubling of the population along many coasts over the last 20 years. This is degrading estuarine and coastal waters through pollution, eutrophication (water quality degradation caused by excessive nutrients), increased turbidity, overfishing and habitat destruction. The pollutant supply does not just include nutrients; it also includes mud from eroded soil, heavy metals, radionuclides, hydrocarbons, and a number of chemicals including new synthetic products.

The impact on estuaries is commonly still not considered in environmental impact studies when dams are proposed on rivers. In this case, the coastal environment, coastal fisheries, and the local coastal people may be regarded as an expendable consequence of river basin development. For instance, as we wrote this article, the dam on the Guadiana River, Portugal, which will form the largest man-made lake in Europe, is being completed without a detailed audit of its impact on the estuary and the coastal zone, and without any estuary remediation measures being planned.

Estuaries are often regarded as sites for future development and expansion, and have been increasingly canalized and dyked for flood protection, and their wetlands infilled for residential areas. For example, almost all estuarine marshes have already been reclaimed in Japan and in the Netherlands. All these factors impact on the biodiversity and productivity and, hence, the overall health and function of estuaries and coastal waters. They increasingly lead humans away from the possibility of ecologically sustainable development of the coastal zone.

Human impact on the ecological health of estuarine and coastal waters is dependent on several factors. Water circulation in some estuaries readily flushes away pollutants to the open ocean, whereas other estuaries retain the pollutants. Unfortunately many estuarine environments that are especially attractive for human settlements, such as wetlands, lagoons, harbours, and fjords, are often poorly flushed. In these systems, through physical and biological mechanisms, estuarine and

coastal waters retain much of the nutrients and other pollutants resulting from human activities. In some ecosystems, the level of extra nutrient is small enough that it may generate an increase of biological productivity without dramatic modification of biodiversity (e.g., Zalewski 2002). More commonly, however, the load of nutrients and pollutants is so high that it degrades water quality, ecological 'services', biodiversity, and productivity of coastal waters. This degradation is not restricted to estuaries, it is commonly observed already in poorly flushed, semi-enclosed water bodies. In the Baltic Sea, for example, the limited depth, low salinity of surface water, and the supply of saline oxygenated water from the North Sea at the bottom, lead to water stagnation, eutrophication, and anoxia (Gren et al. 2000). This impacts upon the commercial cod fisheries and generates increasingly common toxic algal blooms (Figure 1a); in turn, this may impact on tourism. The northwestern Black Sea ecosystem is similarly degraded by direct human impacts not only on the coastal ecosystem but also on the drainage basin of the Danube River that receives the effluents from eight European countries (Zaitsev 1992; Lancelot et al. 2002). A similar degradation is also observed in poorly flushed Seto Inland Sea, Osaka Bay, and Tokyo Bay, Japan, where toxic algal blooms occur 100 days per year (Figure 1b; Okaichi and Yanagi 1997; Takahashi et al. 2000). Better flushed and larger systems also suffer from environmental degradation through eutrophication, as is made apparent by some beaches of the North Sea being covered by foams of decaying algae and protozoa, mainly *Phaeocystis* and *Noctiluca*, and by hypoxia in the Gulf of Mexico (Richardson and Jorgensen 1996; Rabalais et al. 2002).

Until now, the solution was believed to depend on reducing the amount of waste and relying on hard technology, namely the construction of sewage treatment plants and the modification of farming practices and technology. While there are exceptions, for instance, the partial restoration of some ecological functions in the Rhine and Thames estuaries, in general this technological fix has not restored the ecological health of estuaries in both developed and developing nations worldwide. The reasons for failure are simple. First, integrated coastal zone management plans are drawn up but, in the presence of significant river input, they are

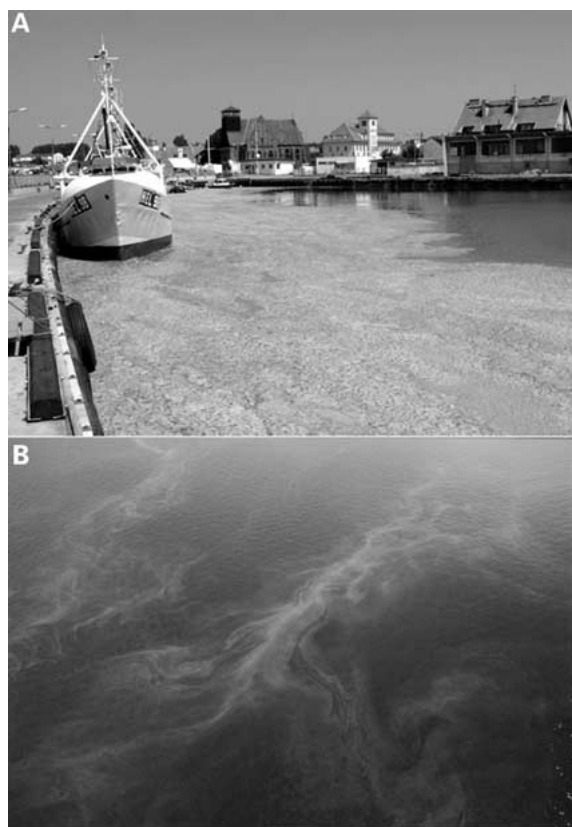


Figure 1. (A) A photograph of a toxic algal bloom in Hel Harbour, Polish Baltic Sea, in the summer of 2001. (B) An aerial photograph of a 3–4 km long toxic algal bloom of *Protoperidinium* sp. at Tokyo Bay, Japan. These toxic algal blooms are caused by eutrophication ((A) is courtesy of K. Skóra and (B) is courtesy of the National Institute of Land and Infrastructure Management (NILIM), Japan (K. Furukawa)).

bound to fail because in nearly all countries these management plans deal only with local, coastal issues, and do not consider the whole river catchment as the fundamental planning unit. It is as if the land, the river, the estuary, and the sea were not part of the same system. When dealing with estuaries and coastal waters, in most countries land-use managers, water-resources managers, and coastal and fisheries managers do not cooperate effectively due to administrative, economic and political constraints, and the absence of a forum where their ideas and approaches are shared and discussed. A typical case is the river disposal of waste from industrial swine and poultry production in North Carolina, USA, and the resulting pollution of estuaries downstream (Mallin 2000). Second, estuarine

environment degradation is often seen as ‘normal’ and nothing to be alarmed at. Most people do not demand better ecosystem health and function, possibly because in poor countries there are no resources to restore the estuarine health (and ecological issues come second to those of economic development), and, in wealthy countries, people have not seen their estuaries in a pristine, healthy state and thus they may think that what they see is normal. Further, and most commonly, solutions are simply not applied because of socio-economic constraints but as well because of national policies, planning and management structures. Finally, there is still no cost-effective technology and political will to treat effluent from diffuse sources in rural activities, except possibly using wetlands as filters; a practice that is effective but rarely used (Moore et al. 2001). The problem remains unresolved as to how, using only limited financial resources, to preserve, restore and manage critical habitats, and reduce the load of nutrients, sediments and pollutants in the face of increasing human population and its aspirations.

In this paper, the concept of ecohydrology is introduced as a holistic approach to the management of estuaries and coastal zones within entire river catchments, by adopting science-based solutions to management issues that restore or enhance natural processes as well as the use of technological solutions. Ecohydrology principles can be applied to control and reduce the impact of nutrients and pollutants along the rivers and the estuaries. Ecohydrology is more than integrated river basin management; its principles provide the guidance for the development of low-cost technology for mitigating the impact on the coastal zone of human activities throughout the river basin, using or enhancing the natural capacity of the water bodies to absorb, or process with no resulting estuary degradation, the nutrients and pollutants. In situations where socio-economic and political constraints do not allow the use of technology to prevent the degradation of estuarine and coastal waters, or to restore estuary quality and health, we argue that catchment-wide ecohydrology principles should instead be applied to help rescue or restore these water bodies. Ecohydrology is, in essence, a question of science-based, integrated management of the river basin ecosystem from upland waters to the seas.

Ecohydrology: the new tool for sustainable development of coastal zones

Ecohydrology is the science of the interplay between hydrology and biota (Zalewski 2000, 2002). Ecohydrology is based on a set of principles. The first principle (hydrological) defines the river basin as a template of hydrological processes determining when and where stable and predictable aquatic biota form and start to play a stabilizing role on water quality. The same principle applies to estuaries. In the second principle, ecohydrology considers the entire river basin as a 'super organism'; this simply means that each of the inhabitants of the basin is an inherent component of the entity, for example, somewhat like a cell in a body. In practice, this means that if an inhabitant discharges pollutants into the system, sooner or later this pollutant will impact on another inhabitant. Such pollutants include eroded soil, nutrients, heavy metals, and DDT and PCB residues that are now found in virtually every ecosystem of the world. Moreover, some pollutants can be bio-accumulated ('biomagnified') in the food chain by several orders of magnitude, potentially impacting on fish and people. In the third principle, the capacity of the system to cope with stresses from human activities (e.g., eutrophication) can be enhanced by managing the biota and by regulating hydrological processes (Mitsch 1993; Jorgensen 1997; Jorgensen and de Bernardi 1998; Zalewski 2000, 2002). An example is the creation of wetlands to trap nutrient and pollutants, and convert nutrients to plant biomass; this biomass can provide bio-energy for society and create employment. Another example is controlling water levels *via* the biota in wetland-fringed water bodies to regulate water quality and enhance fish stocks. An additional example is the use of macrophytes to avoid toxic algal blooms in rivers and lakes. Toxic algal blooms do not appear for $P-PO_4 < 30 \mu g l^{-1}$ (Dunne and Leopold 1978). Reaching such a small concentration means changing farming practices and, in practice, this is economically an unrealistic goal. However, it is possible to avoid toxic algal blooms by reducing $P-PO_4$ concentrations to only $120 \mu g l^{-1}$, provided macrophytes are present. This option is attractive because establishing macrophytes in an aquatic ecosystem may be considerably less expensive than reducing nutrient input from human activities; it

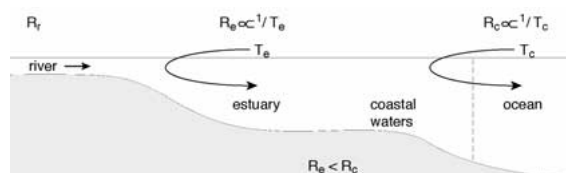


Figure 2. Sketch of the robustness (R) of a river (r) and its estuary (e) and coastal waters (c) as a function of the residence time T .

enables a lower reduction in the nutrient discharge from rural areas, while eliminating toxic algal blooms without any need for an expensive technological solution or large-scale change in land-use practices.

Extending this concept to the coastal zone, the whole river catchment, from the headwaters down to the estuary and coastal waters, must be viewed as a single ecological entity (Figures 2 and 3). Therefore, in order to maintain healthy, productive estuaries and coastal waters, it is necessary to manage human impacts throughout the river catchment, along the river and in the coastal hinterland, and to enhance the natural capacity of the river and the estuary to sequester or degrade pollutants, and to convert the excess nutrient into plant biomass. Ecohydrology thus relies only on low-cost technology; it requires innovative solutions based on an understanding of feedback mechanisms between hydrology and biota within the estuary and coastal waters. By relying on natural ecological processes, it can lead to sustainable development and environmental management. In essence, these principles have been used to good effect in managing coastal dune complexes where appropriate species of grasses and trees are used to manipulate sediment throughput and deposition, but with careful consideration of the hydrological demands of coastal dune communities, for example, Ainsdale Dunes, UK (Wheeler et al. 1993).

The paper describes a number of key processes of estuaries. For convenience, these processes are separated into physical, biological and human-related processes. They all impact on the estuary, and successful sustainable management depends on integrating these processes. Clearly, the prevailing natural conditions and human impacts within the river catchment and its coastal hinterland, the hydrology, the biology and the buffering capacity, and 'robustness' of each estuary (see below), vary

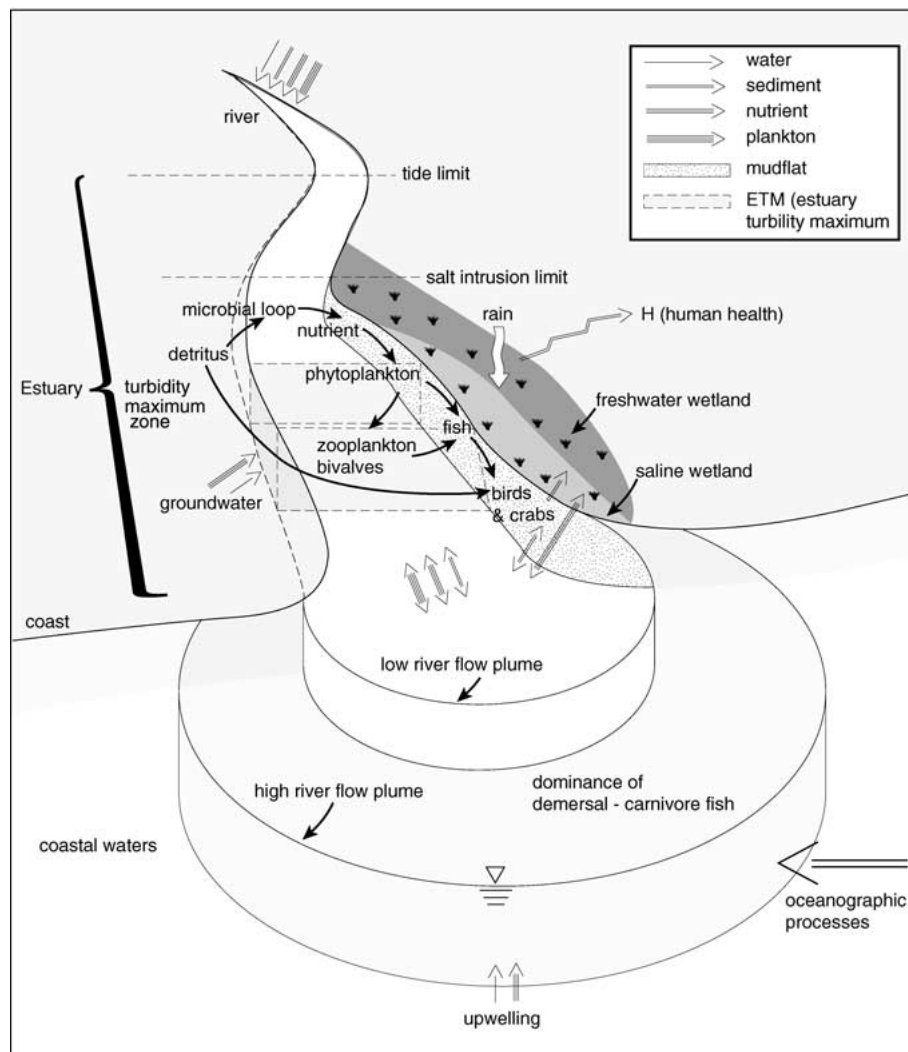


Figure 3. Sketch of the dominant pathways of water, fine sediment, nutrients, and plankton in an estuary, together with the impact on human health.

from river catchment to river catchment and from estuary to estuary in both time and space. Hence, there is no universally applicable formula to integrate these processes in an ecohydrology model of an estuary, but generic ecohydrologic principles may be employed to consider the integration of these factors from one estuary to another.

This paper starts with a consideration of the robustness of an estuary. Robustness (Figure 2) is controlled by the rate at which water in an estuary is flushed; the shorter the flushing time-scale, the higher the robustness, that is, the lower the expected water quality problems. However, the

flushing rate of water is not the only variable determining robustness, and the following sections describe the three other key determinant processes. These are: first, the rate at which fine sediment – soil eroded from the river catchment – is flushed; second, the efficiency with which organic matter is processed by the biology within the water column; and third, the trapping of sediment, nutrients and pollutants in inter-tidal wetlands (mangroves and saltmarshes). A section describes severe estuarine degradation resulting from human activities up-river to illustrate cause and effect. The next section describes aspects of human health linked with

estuaries; once again this is dependent on human activities up-river, but it is also dependent on wetland buffering capacity. The final section describes how a holistic approach is needed to understand the health of the ecosystem. The section also suggests that this holistic approach can be used to manage an estuary and coastal waters on an ecologically-sustainable, low-cost basis.

Estuary robustness

The role of robustness

Estuaries and coastal waters are subject to constant changes in the wind, solar input, rainfall, wind, currents offshore, sea level, freshwater runoff, varying climate, and human influences. These changes occur at time-scales of minutes (turbulent mixing), hours and days (tides), months (hydrologic and seasonal changes), and decades (climate). Thus, estuaries are highly dynamic environments that exhibit strong temporal and spatial changes in their physical, chemical and biological variables. In turn, this dynamism leads to a large variability in estuarine water quality, chemical and biological characteristics. Further, no two estuaries are identical; each estuary (and coastal water body) has its own size, shape and bathymetry, its own length of tidal influence, tidal range, wave climate, freshwater inflow, turbidity and residence (or flushing) times, and its own sediment properties of grain size, carbon-to-nitrogen ratio, organic carbon content, and water-column turbidity. Thus, each estuary is unique and there is no general parameter to readily assess the 'health risk' of an estuary from natural and human influences.

The ecological health of estuaries depends on the 'successful' interaction – and the limits to the possible interaction that are imposed by the temporal and physical variability – between organisms and variations in salinity, currents, waves, suspended particulate matter (SPM), bed sediments, temperature, air exposure, hypoxia, wetland contaminants and biodiversity.

Like the health of a living organism, the health of an estuary or a coastal water body, cannot be measured by one single variable. A number of variables are important (Balls 1994); these include the dissolved nutrients, the dissolved oxygen

concentration, other solutes, and SPM. To some degree, all these variables are controlled by the residence time, that is, by the rate at which these components are flushed out to the sea.

Thus, the residence time of an estuary or coastal water body is an important parameter determining its robustness and its ability to cope with human-induced stresses (Figure 2). Well-flushed estuaries are intrinsically more robust than poorly flushed systems. The residence time is most critical in the upper reaches of an estuary because this is where contaminant accumulation and increased turbidity from human influences are most likely to occur. As a result, environmental degradation is most often apparent during periods of reduced freshwater inflows, for example, during drought or when human activities reduce the freshwater flow. Indeed, human activities in the river catchment may cause significant reductions in dissolved oxygen, mainly through eutrophication. There is no other environmental variable of such ecological importance to coastal marine ecosystems that has changed so drastically due to human influences in recent decades (Diaz and Rosenberg 1995). At times this has resulted in severe deficits of dissolved oxygen, leading to hypoxia and anoxia. Environmental degradation problems associated with the occurrence of low oxygen are increasing on a global scale.

Residence time is a key physical variable determining if a particular estuary will or will not suffer from eutrophication. For example, the residence time of the Tweed Estuary, UK, is generally less than 13 h and dissolved oxygen levels are high throughout the year (Uncles et al. 2000). Rapid flushing ensures both low turbidity (because fine sediments are unable to accumulate in the estuary) and insufficient time for local oxygen depletion to occur, regardless of whatever biochemical oxygen demands are present. The reverse is true in the Humber-Ouse Estuary, UK, which has a residence time of several weeks (Uncles et al. 1998a, b); as a result, this estuary experiences high turbidity and suffers from dissolved oxygen depletion.

The flushing of estuarine waters

The simplest way to quantify the residence time is through the use of a single compartment, tidally averaged box model. This method is very coarse in

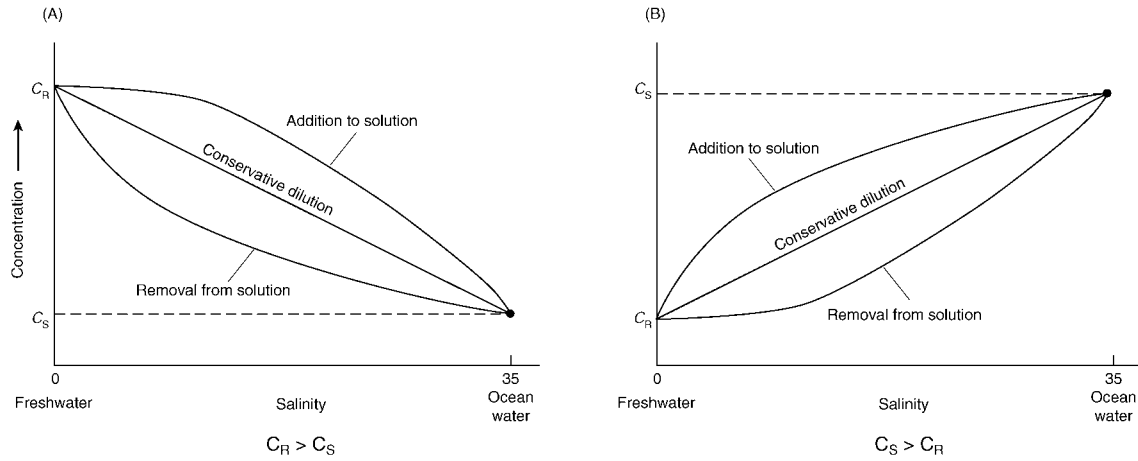


Figure 4. Mixing diagram of a material in an estuary. Salinity is taken to be 0 in freshwater and 35 in ocean water. C_R is the concentration of the material in the river, and C_S that at sea. A straight line indicates conservative mixing (no losses or gain); concave and convex lines indicate, respectively, a loss and a gain of that material in the estuary. When the concentration of material in the river is greater than that at sea, a negative slope exists. Conversely, a positive slope results when the concentration of material in the river is smaller than that at sea.

space and time, but is useful in acquiring a qualitative, conceptual understanding of the time-dependent solute dynamics for a range of estuarine systems in which transients and residence times are important.

In this method, the estuary is treated as a single compartment of fixed volume, V . The compartment is subjected to river (and groundwater) freshwater inputs upstream, and to an equal seaward flow at the mouth. In addition there is mixing with seawater across its down-estuary face. The rate at which the fraction of freshwater within the estuarine compartment, f , changes with time ($\partial_t f$) is given by:

$$\partial_t f = v_f - v_r f \quad (1)$$

where $v_f = Q_f/V = \tau_f^{-1}$ and $v_r = (Q_f + Q_*)/V = \tau_r^{-1} \geq v_f$.

The freshwater ($f = 1$) flow rate into the estuary compartment is Q_f , which is equal to the flow rate out of the compartment to sea, where $f = 0$. The volume-mixing exchange rate with the coastal waters is $Q_* \geq 0$, so that the estuarine residence time of freshwater within the compartment, τ_r , is such that $\tau_r \leq \tau_f$ (from their definitions in Equation (1)). If the estuary were completely full of freshwater then τ_f would equal the estuarine (in this case tidal river) residence time, τ_r , that is, the time required by the river to replace the entire estuarine

basin volume with 'new' fresh water. The coastal salinity is S_o (typically 33–35) and the estuarine salinity, S , is given by:

$$S/S_o = 1 - f \quad (2)$$

A conservative (i.e., no growth or decay) dissolved nutrient with freshwater concentration, N_f , enters the estuary with the freshwater and is mixed into the coastal sea, where, for illustration, its concentration is taken to be zero. The evolution of estuary nutrient concentration, N , is given by:

$$\partial_t N = v_f N_f - v_r N \quad (3)$$

Residence time and therefore v_r in Equations (1) and (3) can be specified empirically in terms of tides and runoff. In the special case of tidally averaged, steady state conditions, during which variables do not change with time ($\partial_t \equiv 0$), the freshwater fraction is given by (from Equation (1)):

$$f = v_f/v_r \quad (4)$$

The estuarine nutrient concentration for this special case is then (from Equations (2)–(4)):

$$N = N_f f = N_f (1 - S/S_o) \quad (5)$$

This is the classic, linear, estuarine 'mixing-diagram' relationship between a solute and salinity (Figure 4). For matter passively carried by water, the mixing diagram shows a linear line. An estuarine nutrient input per unit volume of estuary and

per unit time, $I_N \geq 0$, can be included in the single-compartment estuary model (see Equation (3)):

$$\partial_t N = v_f N_f - v_r N + I_N \quad (6)$$

In the special case of a steady state estuary with a constant estuarine nutrient input, the corresponding steady state estuarine nutrient concentration would then be (see Equation (5)):

$$\begin{aligned} N &= \tau_r I_N + N_f f = \tau_r I_N + N_f (1 - S/S_o) \\ &\geq N_f (1 - S/S_o) \end{aligned} \quad (7)$$

This demonstrates that the estuarine nutrient concentration for this ‘input’ case lies above (i.e., convex upward) the linear, ‘conservative’ line defined by the river and sea end-members (Figure 4). A concave line suggests a loss of the matter within the estuary, for example, by adsorption on mud. A convex line suggests the addition of matter, for example, matter gained by degradation of organic material or desorption from mud. Thus, if there were *in situ* nutrient removal from the estuary (i.e., $I_N < 0$, such as might occur during an algal ‘bloom’) then the estuarine nutrient concentration for this ‘removal’ case would lie below (i.e., convex downward) the linear, ‘conservative’ line defined by the river and sea end-members. In particular, the influence of the local estuarine input, or removal, is directly proportional to the residence time τ_r , and is likely to be much less important in rapidly flushed estuaries (small τ_r and potentially small $\tau_r I_N$) than in slowly flushed estuaries (large τ_r and potentially large $\tau_r I_N$).

Estimates of the residence time

The estuarine residence (or flushing) time of a solute depends strongly on tides, freshwater runoff and morphological size, especially length. Residence times, which generally are calculated from freshwater runoff and estuarine freshwater content, tend to increase with tidal length (TL) and decrease with increasing tidal range. Uncles et al. (2002) derived the following simple, linear, multiple regression relationship for the logarithm of the maximum residence time (τ_r , in days) in terms of the logarithm of the mean spring tidal range (MSTR, in m) and the logarithm of the tidal length TL (in km),

$$\tau_r = 0.23(\text{MSTR})^{-0.4}(\text{TL})^{1.2} \quad (8)$$

The residence time is, therefore, mainly affected by TL, as would be expected for a diffusive system (RT proportional to the square of TL) or an advective system (i.e., proportional to TL).

Equation (8) ignores two processes that can modify the residence time in an individual estuary. These are, first, the currents driven by a difference in density between fresh and salt water – this is particularly important in deep fjords, bays and semi-enclosed seas where the bottom waters can be nearly stagnant and where water quality can degrade severely. Another key process is water storage and buffering by intertidal wetlands, mainly saltmarsh or mangrove vegetation that flank the main estuary and results in drag to the flow and temporary storage of waters. These points are discussed in more detail below in the sections on saltmarshes and mangroves.

The effects of residence time on dissolved substances

An important chemical variable is dissolved oxygen, because of its necessity to the metabolism of aerobic organisms, and its uptake, sometimes to hypoxic and anoxic levels, by the respiration of sewage and other waste inputs. Because of the influence of varying residence times and other environmental factors, such as wind speed, there is usually no simple relationship between the oxygen demand of waste effluents and reductions in oxygen concentrations. The oxygen deficit depends on water flow, turbidity, and oxygen supply and demand, and this varies from estuary to estuary (Owens et al. 1997).

Nutrient loads in many rivers have increased markedly over recent decades and this increase is thought to have been partly responsible for the changed eutrophic status of a number of estuaries and coastal seas (Reid et al. 1994). Once in the estuary, non-conservative behaviour of nutrients can be pronounced. As with the dissolved oxygen deficit, the fate of nutrients varies from estuary to river as a function of water flow, turbidity and biota (Figure 3). In muddy estuaries, the turbidity varies spatially and temporally (see the section on ‘The fate of fine sediment in the estuary’). As a result of all these influences, phosphate usually behaves non-conservatively in turbid estuaries (this behaviour is particularly marked within the

upper reaches of muddy estuaries), whereas it may be less so for silicate concentrations, while nitrate may be essentially conservative (Morris et al. 1981). Systems with very long residence times can export much less of these nutrients to the coastal zone than systems with very short residence-times. For example, the degree of denitrification in an estuary (i.e., loss of nitrogen to the atmosphere) increases with water residence time (i.e., for N, see Nixon et al. 1996). There is no straightforward relationship between trace metal transport and water residence time; this appears to be due to non-linear metal chemistry and absorption/desorption processes of metal on mud (Morris 1988; Liu et al. 1998; Salomons and Forstner 1984). Once the metal is absorbed on mud, it can become buried under the substrate of the estuary or in the fringing wetlands. This metal can still be remobilised, and threaten the biota, by a number of processes such as bioturbation, natural erosion and channel migration, or through dredging (French 1996).

High nutrient inputs to estuaries and associated eutrophication can lead to algal blooms. In turn, this results in the consumption of dissolved oxygen by decaying algae, and in the formation of foams and toxins. The residence time, and thus the rate of flushing of nutrients from estuaries, is therefore an important factor, but not the only one, in determining whether excessive nutrient inputs are likely to lead to algal blooms and oxygen sags.

Eutrophication is not restricted to estuaries; it can also impact on coastal waters. The link between land-use and coastal water pollution can be quantitatively determined using models. For instance, a 50% reduction in artificial fertilizer applications within the Humber drainage basin, UK, would give a 10–15% decrease in nitrogen loads to the North Sea, relative to the 1994–1995 input (Tappin, pers. comm.). To a certain extent, residence time influences which part of the coastal zone is at risk from degradation or eutrophication – whether coastal wetlands have sufficient time to sequester the nutrient reservoir, or whether nutrients make their way to the shelf without significant loss. This has important consequences for marine sanctuaries and world heritage sites which flank intensively used catchments, for example, Monterey Bay, USA, and Greater St. Lucia Wetland, South Africa.

Table 1. Comparison of the drainage areas, the sediment load and the yield for various rivers.

| River | Area (10 ⁶ km ²) | Load (10 ⁶ tonne year ⁻¹) | Yield (tonne km ⁻² year ⁻¹) |
|--------------------|--|--|--|
| Yangtze | 1.9 | 480 | 252 |
| Amazon | 6.1 | 1200 | 190 |
| Mississippi | 3.3 | 210 | 120 |
| Ganges/Brahmaputra | 1.48 | 2180 | 1670 |
| Mekong | 0.79 | 170 | 215 |
| Fly | 0.076 | 116 | 1500 |
| Cimanuk | 0.0036 | 15.7 | 6350 |
| King Sound | 0.12 | 6 | 50 |
| La Sa Fua | 5 × 10 ⁻⁶ | 2.4 × 10 ⁻³ | 480 |

The effect of deforestation on estuaries is much more rapid in the tropics than in temperate zones and the difference is mainly due to the intense rainfall in the tropics. The Cimanuk and La Sa Fua river catchments are small and profoundly modified by human activities in the tropical islands of Java and Guam, respectively (Wolanski et al. 2003b).

The fate of fine sediment in the estuary

Riverine sediment inflow

Associated, though not exclusively, with the fresh-water inflow to estuaries is an input of SPM (which includes and is usually dominated by sediment) derived from erosion of the drainage basin. The SPM concentration of river waters typically exhibits great variability as a function of river inflow, although it is generally larger during high river flow periods, especially during floods. Sediment inputs into estuaries vary greatly for different rivers. For illustration, the Mekong and Fly rivers form large estuaries in the wet tropics. To place these rivers in perspective with other major rivers in the world (see Table 1), the Mekong, Vietnam, has a smaller drainage basin than the Ganges/Brahmaputra (53%), Yangtze (41%), the Mississippi (24%) and the Amazon (12%). The Mekong sediment yield is about twice that of the Mississippi, 85% that of the Yangtze, 12% larger than that of the Amazon, but only about 1/7 that of the Fly and the Ganges/Brahmaputra. The Fly River, Papua New Guinea, has the highest sediment yield, about 10 times higher than the Amazon and the Mississippi. Erosion increases in the catchment as a result of human activities, which leads to an increase in river SPM. When rivers are dammed, much of the coarse sediment (e.g., gravel, sand and coarse silt) may

remain trapped in the reservoirs, which then experience siltation.

Seaward sediment export from estuaries

The export of fluvial SPM varies from estuary to estuary. In practice, there are estuaries that trap essentially all the sediment, other estuaries export seaward all the riverine sediment to the sea, and other estuaries trap some sediment and export the rest. For instance, in the Mekong Estuary, Vietnam, river floods last several months and freshwater may be found throughout the estuary, up to the river mouth. In this case, fine sediment is not stored in the estuary; instead, it is discharged directly into the shallow, coastal waters (Wolanski et al. 1996). A fraction of that sediment returns in the estuary during the low flow period (Wolanski et al. 1997). In the Fly River, Papua New Guinea, river floods are rare and much of the riverine mud remains trapped in the estuary (Wolanski et al. 1998a). In the Fitzroy Estuary, Australia, the mud is deposited slightly offshore during river floods; during the rest of the year, this mud is resuspended by strong tidal currents and transported back into the estuary, so that over a time scale of a year most of the riverine sediment is effectively trapped in the estuary (Wolanski and Spagnol 2003).

Formation of mudflats

The nature, degree and extent of sediment deposition in the estuary are determined by its morphology and sediment characteristics (Figure 3). These vary widely from estuary to estuary, according to the importance of wave activity, tidal variations in water level and current speed, freshwater inflow, shape and size of the estuary, local geology and sediment availability and mineralogy. Many tidal estuaries have large areas of intertidal mudflats, which comprise large amounts of clay and silt mixed with varying fractions of coarser sediments. These mudflats are exposed to the atmosphere at tidal low water (LW) and border the main, deeper channels. They probably formed as turbid waters inundated the near-shore banks on a rising tide and deposited sediment in the relatively slack waters there, especially over high water (HW) slack, and then failed to suspend and remove all of the deposited sediment on the subsequent falling tide (Pethick 1984).

These mudflats are often flanked, in temperate climates, by localised saltmarshes that are inundated at HW of spring tides or, in the tropics, by mangroves. Much of the muddy sediment is derived from tributary rivers and the coastal sea, and was deposited as rising sea levels of the Holocene drowned former river valleys (e.g., Chappell and Woodroffe 1994; Roberts 1998). These vegetated mudflats play a critical role in determining the robustness of the estuary, by selectively trapping fine sediments, influencing the water residence time, sequestering nutrients and pollutants, and converting excess nutrients within the water column into plant biomass. These points are described in separate sections on saltmarshes and mangroves, respectively.

The size, elevation and stability of non-vegetated mudflats depend to a considerable degree, but not entirely, on the biology. There are relationships between sediment erosion properties and the biological characteristics of intertidal mudflats that demonstrate the importance of the surface diatom layer (biofilm) in modifying the initial erosion of a surface mud layer (e.g., Sutherland et al. 1998a; Widdows et al. 2000; Riethmüller et al. 2000; Anderson 2001). Benthic diatoms produce 'sticky' extracellular polymeric substances within these biofilms (Paterson 1994; Sutherland et al. 1998b), which then provide a degree of binding for mudflat and mudbank surfaces. It has been hypothesized that the erosion properties of cohesive, intertidal mudflats depend on a balance between the physical and biological processes of stabilization and destabilization (Widdows et al. 2001). The density of microphytobenthos, algal mats, higher plants, worms, mussel beds and other biological factors can affect bio-stabilization of sediments. Bio-destabilization mainly results from the bioturbation caused by burrowing and deposit-feeding animals, such as bivalves, polychaetes and crustaceans.

Estuarine turbidity maximum zone

Once riverine-derived SPM enters the estuary, it can be trapped within an estuarine turbidity maximum (ETM) zone (Figure 3). The ETM is frequently located in the very low salinity reaches of an estuary, the position of which is dependent on tidal range, river flow and physical dimensions, and, therefore, shows great variability. Continuous exchange of primary sediment

particles and aggregates between water column and bed within the ETM generates a shoal of mobile bed sediment, which moves down-estuary with increased river inflow and exhibits a seasonal migration (Bale et al. 1985).

Long, strongly tidal estuaries tend to have greater SPM concentrations within their ETM regions than either shorter estuaries (with comparable tidal ranges at their mouths) or weakly tidal estuaries (Uncles et al. 2002). An empirical relationship between the maximum, depth-averaged SPM concentration at HW within an estuary, TL and tidal range (TR) was derived from a systematic comparison of SPM concentrations at spring tide, low freshwater inflow conditions for 44 estuaries world-wide. A simple, although crude, linear, multiple regression relationship for the logarithm of the maximum SPM concentration (mg l^{-1}) in terms of the logarithm of the TR (in m and specified at the estuary mouth) and the logarithm of the TL (in km) gave:

$$SPM = 0.0055(TR)^{2.8}(TL)^{1.7} \quad (9)$$

Longer estuaries possess faster tidal currents for a given tidal range at their mouth; this in turn produces more erosion and suspension of bed sediments, and therefore higher SPM concentrations (Uncles et al. 2002).

Water quality aspects of SPM and deposited sediment

Sediment particles and aggregates within the ETM can give rise to marked changes in water quality. Fine particles can adsorb metal ions and organic macro-molecules from solution to such an extent that some metals can be completely removed from solution within a strong ETM (Ackroyd et al. 1986; Salomons and Forstner 1984). On longer time scales, these adsorbed metals become incorporated into anoxic, reducing bed sediments. Here, heavy metal and radionuclide pollution may be regarded as being effectively bound in an immobile sediment sink. Indeed, considerable success has been achieved in using down-core trends in metal and radionuclide concentration as a faithful record of the industrial history of estuaries (e.g., McCaffrey and Thomson 1980; Oldfield et al. 1993; Cundy et al. 1997; Plater et al. 1998; Fox et al. 1999). However, these metals may become solubilised into pore water or have the potential for physical

recycling following increased wave activity or tidal currents with rising sea level or increased storminess.

The affinity of organic molecules for particle surfaces and the incorporation of biogenic material within aggregates that constitute the ETM make it a suitable environment for particle-associated bacteria. The associated, localised bacterial respiration can cause significantly dissolved oxygen depletion within the ETM (Morris et al. 1982; Uncles et al. 1998a). Traditionally, low dissolved oxygen levels have been attributed to industrial discharges and pollution from tributary rivers and direct effluent discharges to the estuary, especially from sewage treatment works located in the major population centres. However, in the Loire Estuary, urban and industrial effluents represent a very small part of the total oxygen demand and the occurrence of anoxia is associated with the stock of organic material in the ETM region (Thouvenin et al. 1994).

Estuaries as sieves for nutrients

The strongly tidal estuaries generally exhibit a tolerance to eutrophication with respect to nitrogen-containing nutrients, despite (in some cases) high nitrogen loadings in their inflowing rivers. The mean annual chlorophyll 'a' levels are significantly lower in the strongly tidal estuaries than in the weakly tidal estuaries with similar nitrogen concentration (Monbet 1992). Tidal range, of course, acts as a surrogate parameter to describe tidal mixing, current velocity, sediment suspension and turbidity (Equation (9)) and therefore light penetration. Larger tides ensure that accumulated sediment is regularly suspended, leading to high turbidity and low light levels with less potential for bloom conditions, regardless of nutrient levels.

In non-impacted environments, the transport of nitrogen (N) and phosphorous (P) from a drainage basin to its watercourses is dependent on the chemical and mechanical weathering of soil minerals, whereas in impacted environments it is thought that agriculture is the largest contributor to river nutrient concentrations (Tappin 2002). The fluvial fluxes of total N and P depend both on water and SPM fluxes and the N and P species associated with them. These species include dissolved (nitrate, nitrite, ammonium, organic N, inorganic P,

organic P) and particulate (organic N, organic P, inorganic P) components. Particulate species tend to dominate the load, although nitrate and phosphate become more important in populous regions.

Once nutrients enter an estuary, non-conservative behaviour can be pronounced as is described in the section on 'The effects of residence time on dissolved substances'. Key processes responsible for this non-conservative behaviour include burial in sediment reservoirs, such as tidal saltmarshes and mangrove forests, and loss to the atmosphere, largely by bacterial denitrification of nitrate N to gaseous N_2/N_2O .

Nitrogen compounds may exert a significant oxygen demand in estuaries through microbially mediated nitrogen transformations (Owens 1986; Grabemann et al. 1990). Organic nitrogen is hydrolyzed to ammonia and additional ammonia is input to estuaries *via* tributary rivers and wastewater discharges. Ammonia utilizes dissolved oxygen during nitrification to produce nitrate, *via* nitrite (e.g., Owens 1986). In conditions of low dissolved oxygen, nitrate may be reduced to enable oxidation of organic carbon to proceed (e.g., Gameson 1981). Other nutrients are discharged to estuaries. For example, river and estuarine waters are often enriched by phosphate from urban and industrial wastewater discharges and from land runoff (e.g., Grabemann et al. 1990; Müller et al. 1991; Hager and Schemel 1996) and they receive silicate from tributary river inflows *via* rock weathering and soil leaching. The river inflow, estuarine mixing and biological uptake are all important to the distribution of dissolved silicate.

SPM and sediments as repositories of organic carbon

Particulate organic carbon (POC) in estuaries and coastal waters is derived from a number of sources and, as such, its proportion in SPM varies in both time and space. The observed POC distributions in the estuary are derived from three main sources: first, fluvial plankton, both living and detrital; second, terrestrial POC detritus, mostly carried by the non-tidal, freshwater river during floods (typically 3% by dry weight of SPM; Meybeck et al. 1988); third, marine plankton with POC content up to 20%.

Estuaries often show a POC decrease from the non-tidal, freshwater river to the estuarine ETM.

POC decreased from a range of 4.6–1.5% in the Garonne River, France, to 1.5% in the Gironde Estuary's ETM (Etcheber 1983). In the Delaware Estuary, USA, the POC content was 9% in the river and 2.3% in the estuary's ETM (Biggs et al. 1983). In the Ems Estuary, Netherlands, POC decreased from 6.1% in the non-tidal river to 2.8% in the ETM (Eisma et al. 1982). Similarly, in the Tamar Estuary, UK, POC decreased from 5% in the river to about 2.5% within the estuarine ETM (Morris et al. 1982). Considering carbon to nitrogen ratios, computed POC/PN ratios (P means particulate here) for the Humber Estuary, UK, are generally >10 and higher than many data reported elsewhere (which are typically 8–12). Similarly high values have been reported for the very low salinity reaches of some North Carolina estuaries, USA, where the POC is characteristic of particulate terrestrial plant material (>15; Matson and Brinson 1990), and for the Vellar Estuary, India (4–140; Sivakumar et al. 1983). High ratios occur in the Humber Estuary, UK, when PN levels become very small, rather than when POC becomes very high. This might be a consequence of a significant coal fraction of the POC resulting from historical coal mining activity in the Humber's drainage basin and estuarine-borne coal transportation.

Temporal variability in fluvial inputs of suspended sediment also impacts on POC in estuaries. In the northern reaches of San Francisco Bay for example, the POC fraction of SPM during freshwater floods was 1–2%; POC was higher (mostly 2–4%) during low runoff periods (Schemel et al. 1996). Algal production probably accounted for some of the additional POC during summer (Tipping et al. 1997). The first major freshwater flood of winter that followed a period of relatively low runoff could have higher POC levels, due to mobilisation of terrestrial organic matter produced over the previous summer. Phytoplankton production and fluvial loadings accounted for 85% of the organic carbon sources (Jassby et al. 1993). Even if only 10% of the fluvial loadings were biologically available, these two sources would still account for 62% of the total.

In addition to observed seasonality, freshwater inputs of POC are also susceptible to catchment-derived organic pollution. Bed sediment samples from the Cochin Estuary, India, had POC contents of 0.7–3.8%, with a mean of 1.6%

(Sankaranarayanan and Panampunnayil 1979). In general, values were highest during monsoon periods of large freshwater runoff and they related this feature to the deposition of water-borne plant and animal matter from terrestrial sources. A similar study two decades later found a substantial increase in maximum POC levels that ranged from 0.2% to 6.2%, with a mean of 3.6%, due to organic pollution (Seralathan et al. 1993). Mud samples accounted for the highest POC values and muddy sand samples the least – the result of a strong, negative correlation between POC content and particle size. A similarly strong correlation between POC and bed-sediment particle size was observed in the sewage-polluted estuary of Tolo Harbour, Hong Kong, where POC reached about 2% (Thompson and Yeung 1982).

Significantly, POC plays an important role in oxygen utilisation, and hence water quality, within the estuary. For example, suspended particles in the Mississippi River Estuary were observed to have an average POC content of 1.8%, with offshore particles having higher POC fractions (Trefry et al. 1994). There was a positive correlation between SPM concentration and apparent oxygen utilisation, which is due both to respiration of POC by bacteria attached to the SPM, and to the existence of hypoxic waters on the Louisiana Shelf.

Saltmarshes

The importance of saltmarshes

Traditionally, saltmarshes have been largely considered in the narrow sense of the vegetated part of the tidal flat. Recent studies have been extended to include the whole ecosystem including the vegetated creeks and the mudflats occurring at lower levels at the seaward edge of the marsh. The same consideration must be given to the upper edge of the saltmarsh where there are gradations to terrestrial communities and, more importantly, gradations to brackish and fresh water marshes still with a tidal influence. These intermediate habitats have long been recognised in major river systems such as the Mississippi, but they also occur in both a floristic and functional sense in many very much smaller systems that have transitions from salt water to fresh. While usually the freshwater

input is through riverflow it can also occur in the form of non-saline groundwater movements. There are exceptions however, such as saltmarshes without any direct links to, or connections with, fresh water systems. These marshes can still be seen to be fully functional in all respects but the potential magnitude of the fluxes that they can generate, in respect of sediment, mineral and organic matter is inevitably reduced. They are generally much smaller systems although their proportional contribution can still be very significant.

Saltmarshes (including the associated freshwater marshes) provide a living link and buffer between land and sea (Figure 3; Boorman 2000). They are able to withstand much of the erosive power of the sea during periods of storms. The erosion losses are subsequently made good by increased rates of accretion during calmer periods (Pethick 1992). The possibilities for the export of organic matter, produced in the saltmarsh, to adjoining marine ecosystems have also been widely recognised (Adam 1990; Lefeuvre and Dame 1994). The extent to which export occurs appears to vary considerably from marsh to marsh (Dame and Lefeuvre 1994). Saltmarshes can also act as sources or as sinks for specific components; these components range from organic pollutants, heavy metals and radionuclides through the various mineral nutrients to the organic and inorganic components of sediments themselves (Gueuné and Winett 1994). However, future climate change, sea-level rise and changing storm magnitude and/or frequency may result in erosion of this sediment and remobilisation of its potentially toxic pollutants. Past erosive phases have been interpreted from morpho-sedimentary evidence of historical records of metal pollution in the Severn estuary (Allen 1990a). The significance of future remobilisation lies in the fact that many of these saltmarsh sediment stores contain heavy metals, radionuclides and organic pollutants in concentrations well above currently accepted safety limits (Oldfield et al. 1993; Valette-Silver 1993; Williams et al. 1994; Leggett et al. 1995).

The fate of organic material produced by the saltmarsh vegetation is determined by various environmental factors, such as the local water currents and the maturity and the age of the saltmarsh (Troccaz et al. 1994; Lefeuvre and Dame 1994; Boorman et al. 1994a; Lefeuvre 1996).

Nutrient exchanges

The fluxes of mineral nutrients are dependent both on the external nutrient loading of the estuary, that is the degree of eutrophication, and on the release of nutrients by the decay of organic matter within the saltmarsh. Within a saltmarsh system there are considerable seasonal variations in the concentrations of inorganic nitrogen and these variations are even more marked when exchanges between the different forms of nitrogen ($\text{NO}_3\text{-N}$, $\text{NO}_2\text{-N}$, and $\text{NH}_4\text{-N}$) are considered. Overall, in studies in France, England and the Netherlands, there appeared to be a net export of dissolved nitrogen out of the saltmarshes (Boorman et al. 1994a). The fate of phosphorus was less clear; phosphate may be exported for at least part of the year. Most of the differences in the fate of nitrogen and phosphate can be explained by their different sorptive behaviour because P is strongly adsorbed onto clay and N is not. The concentrations of both nitrogen and phosphorus in saltmarsh creeks will depend on the balance between the supply from inside and outside the marsh and the rate of uptake by the growth of saltmarsh vegetation.

It appears that low concentrations of phosphorus can be the main limiting factor in determining the productivity of saltmarsh vegetation within eutrophic estuaries such as the Westerschelde, The Netherlands, and the Tagus, Portugal, having enhanced productivity levels (Lefevre 1996). This is in contrast to the situation regarding the primary productivity of estuarine waters where, under saline conditions, phosphorus is released from terrigenous sediments upon contact with seawater and nitrogen becomes the limiting factor (Doering et al. 1995). There is a complex pattern of variation in phosphorus levels with, on the one hand, the release of phosphate as described above and, on the other hand, the large scale removal of phosphorus by its take-up from solution during algal blooms. Where adequate levels of both phosphorus and nitrogen occur, other elements, such as silicon, can become limiting (Jacobsen et al. 1995). It is clear, however, that saltmarshes are characterised by their large nutrient storage capacities and that, under certain circumstances, these stores can become 'leaky' with subsequent nutrient releases (Turner 1993). While generally the release of nitrogen and phosphorus from the saltmarsh

occurs during the processes of the decomposition of organic matter, direct losses by the leaching of nitrogen, phosphorus and also carbon, from live plant tissues can occur as well (Turner 1993). The amounts released are high enough to account for significant increases in the activity of the estuarine plankton community and consequently are of potential significance for many other estuarine communities.

Saltmarsh sedimentation

The transport of sediment to a saltmarsh by the tidal flow of sediment-rich estuarine waters into and over the marsh is a crucial part of the vertical development of the surface of the marsh. Models of saltmarsh deposition consider two primary mechanisms of accretion: the tidal ramp and tidal creeks. In the ramp model, decreasing tidal flat elevation in a seaward direction increases the frequency and duration of tidal inundation (=hydroperiod). In this case, sedimentation rate is a decreasing function of elevation within the tidal frame (Pethick 1981; Allen 1990b; French 1993). Saltmarsh accretion *via* creek processes considers the incoming sediment-laden waters to be funnelled by the creek network, which then overspill once the altitude of the fringing saltmarsh is exceeded. In this example, sedimentation is focused in the immediate vicinity on the creek margins in the form of levees, and vertical accretion is achieved by lateral creek migration over the long term (Bridges and Leeder 1977). Providing that the saltmarsh is not limited by an insufficient supply of sediment or spatially constrained by levees preventing its landward movement, a saltmarsh can respond to sea-level rise *via* an landward retreat and/or enhanced sediment accretion.

The magnitude and direction of sediment transport depends on the velocity of the water flow and the size and density of the sediment particles. The bulk flow of the water along a creek is the major mode of sediment transport but there can also be a significant transport of material by near-bed movements of hyper-concentrated suspended sediment. Techniques have been developed to determine both these components (Troccaz et al. 1994; Hemminga et al. 1996a). The effect of water velocity on the behaviour of the suspended sediment load is crucially affected by the particle size of the sediment

involved. Studies at Stiffkey, Norfolk, UK, showed a significant relationship between sediment load and peak tidal velocity ($r^2 = 0.85$, $p < 0.001$) whereas at Tollesbury, Essex, there was no significant relationship between water velocity and sediment load. This reflects the much higher proportion of coarser sediment particles at Stiffkey (10% $> 63 \mu\text{m}$ and 40% $< 2 \mu\text{m}$) compared with Tollesbury (2% $> 63 \mu\text{m}$ and 60% $< 2 \mu\text{m}$). The pattern of distribution of the deposited sediment on the marsh surface is also determined by the sediment particle size (Pitman 1993; Boorman 1996). With the coarser sediments at Stiffkey there was a relationship between the quantity of sediment deposited in a single tide and the distance from the nearest creek, leading to the build-up of distinct levees. Similar results have been obtained from a *Juncus roemerianus* marsh in Florida (Leonard et al. 1995). At Tollesbury, however, the quantity of sediment deposited was independent of the distance from the creek and the deposition of the predominantly fine sediment was largely limited to the short period at high tide when the water movements were very slow. Fine grained deposition may, therefore, be dependent both on hydro-period and distance to the creek (Pethick 1992; Pethick et al. 1992).

Rates of sediment accretion recorded at Tollesbury averaged $4.27 \text{ mm year}^{-1}$ over two years of observation (Boorman 1996) but there were considerable fluctuations from month to month with up to $6.2 \text{ mm month}^{-1}$ of erosion and $4.1 \text{ mm month}^{-1}$ of accretion. Calculations of the annual rate of accretion from the tidal sediment fluxes gave a potential accretion rate of 3.9 mm year^{-1} , suggesting that the regular tidal inputs formed the major mode of sediment addition to the marsh surface. At Stiffkey the rates of accretion of the marsh surface were slightly smaller ($3.08 \text{ mm year}^{-1}$). Here, however, the estimated tidal sediment fluxes were only able to account for 0.3 mm year^{-1} , a finding suggesting that the major sediment input comes with episodic storm events. Similarly, high rates of accretion in low *Spartina foliosa* marsh were almost entirely limited to episodic storm-related events (Cahoon et al. 1996). On the short time scale, rates of saltmarsh accretion can be huge, reaching 1 m year^{-1} (Li et al. 2000; Plater et al. 2002). However, when time-averaged, these rates are much smaller, typically a few millimetre per year.

Sediment trapping by plants

Distinct patterns of sedimentation and vegetation dynamics start in the initial stages of colonisation by plants (Richards 1934; Gray 1992). Both topography and vegetation affect sedimentation dynamics in a feedback process. Sedimentation is increased when the vegetation is taller than 8 cm (Boorman et al. 1998). Among the taller species, *Spartina anglica* enhances sedimentation in European saltmarshes (Thompson 1990; Gray et al. 1991; Sanchez et al. 2001). *Puccinellia maritima* is a perennial grass species whose populations reach their maximum extent on stabilised lower marshes. It also occurs in the pioneer zone, possibly where sedimentation has started.

These processes have been studied at three sites of Mont St Michel Bay (Langlois et al. 2003). In this area, *S. anglica* is quite rare in the lower marsh and *P. maritima* is abundant. *P. maritima* plays a role in the sediment dynamics and in the establishment of micro-topography in the pioneer zones of lower saltmarshes. At the beginning of plant colonisation, on mudflats, the community was composed of *S. anglica*, *Salicornia fragilis* and few clones of *P. maritima*. The micro topography is absent. This plant community changes rapidly in composition and in topography. We can observe after 2 years a low marsh with a large cover of *P. maritima* associated with *Suaeda maritima* with a low cover and *Aster tripolium*. For example, the cover of *P. maritima* increased significantly from 20% up to 95%. This plant community is localised mainly on hummocks. In the presence of *P. maritima*, sedimentation was significantly higher than on bare mudflats. Ninety five per cent of the of the *P. maritima* colonies were recorded on hummocks and only 1% on the mudflats. By contrast, all the *S. anglica* was encountered on mudflats, as well as all the annual *S. fragilis*. At the same sites, 99% cover by *S. maritima* was recorded on hummocks and only 1% on the mudflats. No plants were found in the erosion zones in the low marsh.

The sedimentation dynamics in lower saltmarshes are related to both abiotic (e.g., disturbances) and biotic factors (e.g., the vegetation type, its cover and height) (Adam 1990; Pethick et al. 1992; Moeller et al. 1997; Dijkema 1997; Boorman et al. 1998). Sedimentation is largest in the lowest parts of the marsh provided that vegetation is present

(Richards 1934; Kamps 1962; Randerson 1979; Stumpf 1983). In the Mont St Michel Bay, significant sedimentation only occurred in areas covered by *P. maritima* (Langlois et al. 2003). *Puccinellia* sp. was also involved in the geomorphogenesis of the lower marsh, as evidence by the positive linear relation between the area of hummocks and the abundance of *Puccinellia* sp., leading to the formation of hummocks on which it is dominant. The sedimentation rate reached 82 mm year⁻¹ in the pioneer zone (Langlois et al. 2003). Similar observations were reported for Danish and Dutch saltmarshes (Jakobsen 1954; Jakobsen et al. 1955; Kamps 1962; Andresen et al. 1990; Scholten and Rozema 1990; Dijkema 1997).

On sandy beach plain type saltmarshes, *S. anglica* initiates the formation of small dunes that are rapidly invaded by *P. maritima* (Scholten and Rozema 1990). Once the micro-topography has been established, the rates of rise of the substrate and vegetation succession both accelerate.

Where sand dominates, the ability of *Puccinellia* sp. to stabilise the sediment is especially related to the density of its root system and the speed of its spatial spread by rhizome propagation, as well as its tolerance to burial (Wohlenberg 1933; Richards 1934; von Weihe 1979; Langlois et al. 2001). This is a perennial species that is relatively abundant in the pioneer zones, and which has a very fast rate of spreading (Langlois et al. 2003). *S. anglica* is renowned for its ability to accrete large volumes of sediment, limiting the erosion and enhancing sediment trapping (König 1948; Ranwell 1972; Thompson 1990; Gray et al. 1991; Sanchez et al. 2001). However this species has a limited abundance, typically less than 10% in sandy sediments, because sand is not optimal for *Spartina* sp. especially when in competition with *Puccinellia* (König 1948; Chater and Jones 1957; Dijkema 1983; Scholten and Rozema 1990). This could be due to waves which inhibit the natural spread of *S. anglica*, and to ergot fungus, which affects seed viability and germination ability (Morley 1973; Groenendijk 1984; Gray et al. 1990).

Saltmarshes and fluxes of organic matter

Primary production and decomposition rates are high within the saltmarsh – often equivalent to those of tropical rainforests. The fate of the organic

matter after plants die remains little known. Some of this organic matter is moved by the tidal flow in and out of the marsh, and the end result depends on the form of the organic matter. Three components exist: dissolved organic matter (DOM) that is operationally defined as the fraction of organic matter not retained on 0.45 µm filters; fine (suspended) particulate organic matter (POM); and coarse (>0.2 mm) organic matter (COM), which is generally the same as the floating material. The behaviour of the first two components, DOM and POM, is essentially the same as that of the suspended sediment and is based on the flux of the tidal water flow. The routes and destinations of the floating material (COM) under quiet weather conditions are also similar. Under strong wind conditions, floating material tends to be wind-driven rather than following the normal tidal routes.

The magnitude and routes of export of COM is very variable. At the Tollesbury marshes, UK, export of COM only amounted to 7–8% of the total net above-ground primary production (NAPP; Lefeuvre et al. 1994). These marshes are at a level where they are subjected to regular inundation and, thus, there are regular opportunities for the exchange of organic matter. When the marshes are at a higher level, such as those at the Slufter, The Netherlands, the export of COM tends to be dominated by the occasional storm tide when the nett export can be 6 times as high as the largest import (Lefeuvre et al. 1994). Up to 1/3 of the leaf production of *Atriplex portulacoides* is exported from its point of origin (Bouchard 1996).

A key question remains un-answered on what is the final destination of this material exported from the saltmarsh, whether it is re-deposited elsewhere in the marsh replacing local losses, or whether it is deposited along the driftline at the marsh edge or whether it is exported to coastal waters. In both Europe and America, studies have indicated that usually less than 1%, but occasionally up to 10%, of the organic matter produced is exported as COM, that is, as floating litter, to coastal waters (Hemminga et al. 1996b; Wolff et al. 1979; Dankers et al. 1984; Dame and Kenny 1986). The export of POM and DOM is better known. One major factor is the relative level of the saltmarsh and the corresponding frequency of inundation. This probably explains the apparent major differences that have been observed between the marshes of the east

coast of North America, where exports of over 40% of NAPP are cited, and the much lower values obtained from European saltmarsh sites (Dame and Lefeuvre 1994). There can also be indirect exports of organic matter in the form of fish and other organisms that come in from the sea to feed within the saltmarsh and then return to the sea. There are also other links between the terrestrial and the marine ecosystems. For example, when tidal water flows over a saltmarsh there is a marked increase in the bacterial component of the plankton; mussel beds remove 1/3 of this enhanced production (Newell and Krambeck 1995). The emerging picture is that, in European saltmarshes, the primary production is not easily exported because the generally high level of marshes reduces the frequency of tidal flooding; nevertheless at least a part of primary production is integrated within marine trophic webs (Lefeuvre 1996).

*The primary productivity of saltmarshes:
conversion of excess nutrients into plant biomass*

The above-ground productivity of saltmarsh vegetation is probably the most visible aspect of estuarine productivity and yet at the same time its determination has proved to be difficult. Much of the research has followed the classical method involving the monthly cutting and harvesting of plots during the growing season and comparisons of the density of the living and dead material (Linthurst and Reimold 1978; Boorman et al. 1994b; Lefeuvre 1996). There is an ambiguity in the method because some workers separate the dead material into standing dead and litter, using the standing dead in the determination of productivity. The inclusion of litter in the dead material can result in an apparent increase of net productivity by up to 10% (Bouchard 1996). Measurements are usually carried out at monthly intervals, and do not account for the loss from the dead biomass between the monthly sampling as a result of flushing of the marsh by occasional, short-lived wind-driven currents. Thus, the measurement errors may be large (Kirby and Gosselink 1976; Boorman et al. 1994b). Despite these complications, this determination of the primary productivity of saltmarsh communities, based on monthly sampling, can provide a reasonable picture of the range of productivity of northern temperate saltmarsh vegetation

(Groenendijk 1984; Boorman et al. 1994a; Lefeuvre 1996).

Measurements show that the saltmarshes of Western Europe generally produce more than $1 \text{ kg m}^{-2} \text{ year}^{-1}$ of above-ground dry matter and that this high productivity is achieved both by pioneer and mature saltmarsh plant communities (Boorman et al. 1994a, b; Lefeuvre 1996). Lower productivities than this have been recorded in pioneer communities, which may be due to the incomplete development of plant cover in the earlier stages of colonisation and, thus, do not represent the full potential for those species or communities. On the other hand, the pioneer communities dominated by *S. anglica* are particularly productive, producing up to $1.5 \text{ kg m}^{-2} \text{ year}^{-1}$ of above-ground dry matter. This is very much in line with the high productivities achieved by *Spartina alterniflora* in comparable New World marshes (Kirby and Gosselink 1976). Many of the middle level marsh communities are very productive and this is particularly true for the shrubby species *Atriplex portulacoides* ($\text{NAPP} > 2 \text{ kg m}^{-2} \text{ year}^{-1}$). However, low productivity can also be observed in middle level communities where productivity is limited by a combination of low nutrient levels and a high degree of inter-specific and probably also intra-specific competition. Not all saltmarsh systems have the full range of marsh types, but where they occur the mature communities of the upper saltmarsh can also be very productive, with *Elytrigia aetherica* producing as much as $1.5 \text{ kg m}^{-2} \text{ year}^{-1}$ (Boorman et al. 1994a).

Four major components of productivity can be recognised in an estuarine ecosystem, namely the production of the saltmarsh vegetation of higher plants, the production of the algae associated with the saltmarsh, the production of the microalgae on the surface of the inter-tidal flats, and the productivity of the algae suspended in the water column. In some estuarine systems there are also submerged communities of aquatic plants (sea grasses) but in the systems under consideration their distribution is limited and their overall contribution is small. It has to be noted, however, that in certain communities, where conditions of sediment stability and the clarity of the water favour their growth, they can make a significant contribution to the overall productivity (Heip et al. 1995). Saltmarsh vegetation has both above- and below-ground production.

In terms of the contribution of the marsh to the estuary as a whole it is the above-ground productivity that is more important, because the products of above-ground productivity can be transported by water currents from the wetlands to the estuary. The products of below-ground production are largely retained within the saltmarsh subsystem (see the section on 'Saltmarsh impact from the flow of ground water').

The mean productivity of saltmarshes has been estimated as $500 \text{ g m}^{-2} \text{ year}^{-1}$ at Tollesbury, UK, and $450 \text{ g m}^{-2} \text{ year}^{-1}$ at Stiffkey, UK (Boorman 1996). These values correspond to carbon production of 200 and $180 \text{ g C m}^{-2} \text{ year}^{-1}$, respectively. The growth of algae within the saltmarsh is included within these values; it probably accounts for $5\text{--}15 \text{ g C m}^{-2} \text{ year}^{-1}$. The value is low because of the relatively small contribution of algae to the total biomass. However, in local areas dominated by macroalgae, algal production can be similar to that of the vascular plants of the saltmarshes themselves. The overall productivity of the English saltmarshes is generally similar to those in France and The Netherlands, although the marshes in the highly eutrophic Westerschelde have a markedly higher productivity (Lefeuvre 1996). The microalgae of the intertidal flats are important for their productivity and for their contribution to the stabilising of sediments through the excretion of carbohydrates (Holland et al. 1974; see the section on 'Formation of mudflats').

The productivity of north temperate microphyto-benthic communities is in the range $30\text{--}230 \text{ g C m}^{-2} \text{ year}^{-1}$ (Heip et al. 1995). A typical value, and close to that quoted for several English and Dutch sites, would be of the order of $150 \text{ g C m}^{-2} \text{ year}^{-1}$. This is lower than the estimate of $200\text{--}300 \text{ g C m}^{-2} \text{ year}^{-1}$ for the Delaware Estuary, USA, and is in line with the estimate of $125 \text{ g C m}^{-2} \text{ year}^{-1}$ for the Westerschelde (Pennock and Sharp 1986; Nienhuis 1993; Jacobsen et al. 1995).

Saltmarsh impact from the flow of ground water

The dominant factors in saltmarsh development are the regular tidal coverage by seawater and the varying salinity. An additional factor is the outflow of fresh groundwater (Figure 3). This can lead to anomalies in the vegetation, exemplified by the local increase of less salt-tolerant species. This

phenomenon is often seen in estuaries, such as those in drowned river valleys, where a saltmarsh abuts high ground with the possibility for artesian flows. Such flows are common also in the uplands (Jones 2002). Because many saltmarshes have layers of coarser sediments, the tidal movement in the main creeks can be reflected by changes in the ground water levels and flows at considerable distance from the creeks. Seepages can frequently be seen in the banks of saltmarsh creeks and while these are generally local saline flows, the existence of non-saline groundwater is likely to result in a significant freshwater contribution to the marsh system.

The contribution of groundwater flows to saltmarshes can be significant. For example, during periods of low precipitation artesian groundwater can account for essentially all of the freshwater input to a marsh ecosystem in North Carolina (Gramling et al. 2003). In studies in South Carolina, the ground waters in the marshes are mixtures of sea water and fresh water and measurably mediate land-sea carbon fluxes (Cai et al. 2003). One of the species indicative of the presence of freshwater inputs into saline environments is the common reed, *Phragmites australis*, and the spread of this species has been linked to fresh groundwater flows reducing salinity and the concentration of sulphide-reducing soils (Chambers et al. 2003). The concentrations of the macronutrients N and P influence the growth and development of saltmarsh vegetation, and they are influenced by groundwater flows (Corbett et al. 2002).

It is clear that saltmarsh nutrient fluxes can be affected by the hydrological situation, particularly the magnitude and status of groundwater flows (Anderson et al. 2001; Sutula et al. 2001). Studies on the links between land and sea as mediated by coastal ecosystems need to take this additional factor into consideration. Hydrological dynamics must be seen as a crucial factor for ecological patterns and processes and, indeed, as the key overall linkage between soil, climate and vegetation (Rodriguez-Iturbe 2003). Methods are available for quantifying groundwater discharge at the land-sea interface (Gramling et al. 2003).

Saltmarshes as fish habitats

The primary productivity of saltmarshes is largely assimilated though the degradation and release of

organic matter, particularly through the activities of marine and terrestrial invertebrate species. Associated with this are complex food chains leading to the larger vertebrates, including economically significant species of fish. Fish form an important component in many estuarine ecosystems. There are numerous fish species; for example, as many as 50 in the medium-sized Mira Estuary, Portugal, where, in terms of number and of biomass, the Common Sole (*Solea vulgaris*) is the most abundant species (Costa 1988; Costa et al. 1996). Pristine habitats support an extremely rich biodiversity that can support commercial fisheries, the success of which is closely linked to continued habitat conservation (e.g., Al-Mohanna and Meakins 2003). Fish are an indicator of saltmarsh ecosystem health; indeed, the success of habitat restoration can be judged by the extent to which the fish populations have recovered (Tupper and Able 2000). A well-developed creek system is important in this recovery (West and Zedler 2000).

Mangroves

There are many similarities between physical and biological processes in mangroves and saltmarshes. Similarities include their role in trapping sediment, converting nutrients to plant biomass, trapping pollutants and serving as a habitat for fish and crustaceans (Figure 3).

Mangrove sediment dynamics

During spring tides, mangroves are fully inundated at high tide. A deep tidal channel, fringed by intertidal, mangrove forest, is usually maintained by self-scouring; this is due to a difference between peak water velocities at rising and falling tides (Wolanski et al. 1980).

Within the mangrove forest, the velocities are slightly larger during the rising tide than during the falling tide (Furukawa et al. 1997). At their study site near Cairns, tropical Australia, the suspended sediment concentration peaked at about 150 mg L^{-1} on the rising tide and at about 30 mg L^{-1} on the falling tide, indicating import and trapping of sediment. Sediment was only imported at spring tides; there were zero fluxes into the mangroves at neap tides because the swamp was not

inundated. The total import flux of sediment at rising tide was $12 \text{ kg m}^{-2} \text{ tide}^{-1}$, and the total export flux of sediment at falling tide was $2 \text{ kg m}^{-2} \text{ tide}^{-1}$. This implies for this area a net import in the mangrove forest of fine sediment of $10 \text{ kg m}^{-2} \text{ tide}^{-1}$. Most of this sediment settled in the mangroves within 50 m of the creek. That the mud was able to advance so far into the mangroves and did not deposit within a few m of the banks, was due to the fine-scale turbulence of the flows through the vegetation keeping the fine sediment in suspension. Deposition was observed to occur just before slack high tide and lasted only about 30 min.

Wolanski et al. (1998a) repeated such studies in the mangroves of Hinchinbrook Channel, Australia, and found that the mangroves accumulated mud at a rate of about $1000 \text{ ton km}^{-2} \text{ year}^{-1}$. To determine how this sediment-trapping rate varies with suspended sediment concentration in the rivers, further studies were undertaken in the mangroves of the Ngerikiil and Ngerdorch estuaries in Palau, Micronesia (Victor et al. 2003). These two rivers drain adjacent, similar, mountainous areas of comparable area in the wet tropics. The former catchment is being deforested, the latter is still forested. This results in a factors of 10 differences between the sediment yield in these two catchments. The mangroves comprise 3.8% of each of these two river catchments, they flood semi-diurnally, and in both cases they trap about 30% of the riverine fine sediment. This mud trapping efficiency, while helpful, is however not sufficient by itself to prevent degradation of coastal coral reefs from excessive sedimentation from extensive land clearing and poor farming practices; better land care is still required in the river catchment.

Mangroves are, thus, very efficient mud traps, and this may help mangroves to keep up with the expected rise in sea level. The level of success will depend on the availability of sediment from riverine inflow and from the coastal zone (Chappell and Woodroffe 1994; Wolanski and Chappell 1996). As sea level is predicted to rise, history may be a good guide to predict how mangroves will fare in the future. Following the last glaciation, the sea level rose at a rate of up to 1 cm year^{-1} . At those sites where sediment accumulation rate kept up with the sea level rise, the mangrove forests survived. At other sites mangrove migrated landwards. Elsewhere the mangroves were submerged and perished. Their

burial sites are still recognisable by the mangrove mud at the bottom of some tropical coastal seas. This old mangrove mud is often capped by a layer of new sandy or calcareous sediment brought in by wave action on a sandy beach (Grindrod and Rhodes 1984). After sea level stabilized about 5000 years ago, mangroves reestablished themselves at some, but not all, sites where they still exist. At other sites, the old mangrove forest has perished and has been buried by recent freshwater sediment from riverine inflow (Chappell 1993; Chappell and Woodroffe 1994). In mud-poor, reef environments, mangroves were unable to keep up with sea level rise and drowned. They recolonised the area only when sea level stabilized (Fujimoto et al. 1996). In other cases such as in the Gulf of Carpentaria, tropical Australia, isostatic adjustments lifted the mangroves out of the water, transforming them into a hypersaline mudflat (Rhodes 1980).

Recent studies by Furukawa (pers. comm.) in mangroves fringing mud-poor reef water of Iriomote Island, Japan, show that small wind waves resuspend the mud, which is then exported at ebb tide. As a result the mangroves are nutrient-starved and stressed. They are stunted and weakened and woodborers extensively attack them, because stressed trees generate less tannin than healthy trees.

Nutrients in mangrove sediments: trapping and mobilization

The capacity of mangroves to trap fine sediments is essential for the ecosystem. This includes among others the role of sediments as a physical substrate and as a nutrient source for forest development. The role of biogeochemical transformations within the sediment is relatively well understood (i.e., microbial processes, plant/sediment relationships); however, less is known about the role of system-shaping agents such as inundation dynamics on nutrient trapping or mobilization.

Few researchers treat these issues in a simultaneously detailed but integrative manner, building causal chains between hydrology, sediment properties, and vegetation structure. Numerous observations have been summarized within empirical rules such as '*Avicennia* prefers high places', but relatively seldom has anyone looked beyond correlations into

causality (Boto and Wellington 1983, 1984; Boto et al. 1989; McKee 1993). There is a pressing need to integrate existing knowledge to understand how a mangrove ecosystem works. The key concepts include: flushing time, inundation frequency, residence time, degree of water-logging and stagnancy, hyper-salinization of soils, acid sulfates, water depth and stagnancy. All of these are interrelated variables that can drastically change sediment characteristics. Topographic differences of a few cm can generate major differences between sites with contrasting sediment and vegetation characteristics. For example, in North Brazil, in transition areas between mangroves and saltmarshes, elevation increases of ~10 centimetre can be reflected in a decrease of average inundation frequency from 60 to 40 days year⁻¹. This produces differences in sediment quality causing the existence of an ecotone bordered by a monospecific *Avicennia* forest and an herbaceous plain. Mangroves react quickly and dramatically to variations in relative sea level (Lara et al. 2003, and references therein).

In addition to inundation dynamics, other factors are influenced by changes in relative land-sea level. The degree of exposure of sediments to air is important for determining, for example, the nutrient availability and the salinity. Tidal inundation produces a diffusion of oxygen from the flooding water into the sediment and influences salt transport (R. Lara, pers. comm.). The frequency of inundation determines soil salinity and the frequency of anoxia (Boto and Wellington 1984; Leeuw et al. 1991). In turn, this controls nutrient availability because a drier surface layer with lower permeability to gas exchange causes more hypoxic conditions.

Mangrove sediments are commonly anaerobic, organic-rich intertidal/estuarine muds. Yet, there are also substantial differences in the textural and physicochemical properties according to forest type and inundation dynamics. In contrast to muds in temperate saltmarshes, mangrove muds are less anaerobic, probably because of crab holes and the low bioavailability of organic matter (Alongi and Sasekumar 1992). Crabs are a primary means for incorporation of significant amounts of organic matter from leaf litter fall deep into mangrove sediments as well as causing an active turnover of detritus at the sediment surface (Schories et al. 2003). Key characteristics of mangrove

sediments such as redox potential or sulphide levels can be also modified by the root systems of adult trees (McKee 1993).

Other typical features of mangrove sediments are relatively low concentrations of dissolved inorganic nutrients, for example, nitrate, ammonium and phosphate in porewater, and the presence of tannins derived from leaching and decomposing roots and litter (Alongi 1987; Boto et al. 1989). The reasons for low nutrient concentrations in porewater are rapid turnover, such as high uptake by trees and bacteria, as well as redox conditions and the recalcitrant character of organic matter. Ammonium is the main form of inorganic N in mangrove soils because nitrification is prevented due to the lack of oxygen to oxidize it into nitrite/nitrate. Concentrations of ammonium are higher in muddy than sandy soils and are influenced by the degree of tidal wetting, rainfall, and biological uptake and production. Concentrations of dissolved organic N can be higher than dissolved inorganic N. The high sodium level in most mangrove soils displaces ammonium, which is thus mostly found in the interstitial waters where it can be leached by rainfall or tidal inundation or drained (Alongi et al. 1992). Tidal inundation generates a nutrient exchange between sediment and estuarine waters (Dittmar and Lara 2001).

Interactions between inundation dynamics, sediment characteristics, and vegetation structure in the basin can be particularly well illustrated with reference to phosphorus mobility. For example, changes in porewater phosphate concentrations in sediments beneath mangrove communities in the Indian River, USA, seemed to be linked to variations in tidal inundation frequency (Carlson et al. 1983). Due to its dependence on tidal height, P can become limiting in elevated mangrove forest areas (Boto and Wellington 1983). Concentration of dissolved inorganic P in mangroves is generally low (Alongi et al. 1992; Lara and Dittmar 1999). Bacteria and microalgae in mangrove muds are also probably P-limited. A close microbe-nutrient-plant connection may serve as a path to conserve scarce nutrients necessary for the existence of these forests (Alongi et al. 1993).

Retention of dissolved phosphate and long-term P storage in wetlands is probably controlled by adsorption by Al and Fe oxyhydroxides and by precipitation of Al, Fe and Ca phosphates

(Walbridge and Struthers 1993). Particularly redox potential has a significant effect on the P availability in mangrove sediments. Increasing inundation frequency elevates redox potential in the sediment, that is, it creates more oxidative conditions by diffusive aeration, and elevates the concentration of available P. At lower elevation sites, the higher flooding frequency accounts for a higher total P content. Higher P availability is reflected in vegetation biomass (Boto and Wellington 1984) and P content in leaves (Lara, unpubl. data).

Freshwater flood plain ecosystems are controlled by a combination of topographic, hydrographic and chemical processes (Townsend and Walsh 1998); an equivalent understanding of mangrove ecosystems is largely missing (Cohen and Lara 2003).

Estuarine food webs

When summarising key biological processes within the water column, it is particularly difficult to keep a balance between a simplistic description that highlights the interplay between components of the system, and a scientifically thorough approach that does justice to the ecological complexity, where the general picture can be lost. One can find in the scientific literature dozens of figures describing estuarine food webs for different estuaries. The description of estuarine food webs often gets bogged down in local details. Some of these figures show simplistic food webs, others show extremely complex food webs with numerous feedbacks; other figures focus on biomass and highlight phytoplankton dominance, while yet others emphasise the key role of detritus.

Generic description of estuarine food webs

A generic approach is used to describe food webs in estuaries (Figure 3). Estuaries are converters of living phytoplankton to detrital particles; they are also conveyors of both allochthonous and autochthonous detrital particles to the sea. Herbivorous fishes transfer energy and matter from estuarine plants to upper trophic levels and to the coastal zone. However, direct grazing by herbivores consumes only a small proportion of the macrophyte and macroalgal production. The great bulk of the organic matter produced (sometimes 90%) is

processed through the detrital system. Zooplankton, planktivorous fish, interstitial micro and meiofauna, surface deposit-feeding molluscs, fishes and polychaeta, and filter-feeding invertebrates consume a much greater proportion of the primary production of the phytoplankton and benthic microalgae. Annual plant growth and decay provides continuing large quantities of organic detritus. In addition, there is often a considerable input of detritus from river inflow. Detrital particles and their associated microorganisms provide the basic food source for primary consumers such as zooplankton, most benthic invertebrates and some fishes. Many of the estuarine consumers are selective and indiscriminate feeders on particles in suspension in the water column, or in the sediment that they ingest. Thus, most of the biota that inhabits estuaries can be considered as particle producers (microalgae and detritus derived from plant growth) and particle consumers. The first trophic level in the estuarine ecosystem is therefore best described as a mixed trophic level of detritus consumers, which in varying degrees are herbivores, omnivores or primary carnivores (Knox 1986).

Environmental factors affecting estuarine and coastal biota

Estuaries are highly productive ecosystems because they are often nutrient rich and have multiple sources of organic carbon to sustain the population of heterotrophs. The nutrient sources include riverine and waste inputs and autochthonous primary production by vascular plants, macroalgae, phytoplankton and benthic microalgae (Cloern 1987). These nutrients sustain a trophic structure that varies from estuary to estuary (Degan et al. 1994). Inorganic nutrient loading stimulates primary production and generates a shift to larger-sized phytoplankton (Boynton et al. 1982). Organic matter loading stimulates bacterial production, in different ways depending on the form of dissolved organic matter. Microbes, through the microbial loop (Figure 3) serve as N-remineralizers when metabolizing N-rich organic compounds because they take up inorganic nutrients when organic inputs are deficient in N or P (Kirchman et al. 1989). Increased riverine imports of organic matter in an estuary may result in decreased nutrients availability for algae that may limit phytoplankton

production (Rhee 1972). This situation may occur in estuaries with large fringing saltmarshes and with little external input of inorganic N. More commonly though, estuarine waters are often enriched with N and P from rural, urban and industrial wastewater discharges (Uncles et al. 1998a, b).

The fate of nutrients depends on a number of physical, biological and chemical processes. Physical processes include mixing, flushing and sedimentation. Chemical processes include absorption and desorption. Biological processes include stripping of dissolved and particulate nutrients, primarily by bacteria and phytoplankton (Alongi 1998). Biological and chemical processes increase in importance with increasing values of the residence time because, when the residence time is large, there is more time for these processes to act (see the section on 'The flushing of estuarine waters').

For example, organic N entering the estuary is degraded to ammonia and further nitrified by aerobic bacteria; this increases nitrate concentration, resulting in a convex curve of nitrate versus salinity (see the section on 'Estuaries as sieves for nutrients' and Figure 4). These biological processes consume oxygen and lead to depletion of oxygen content if the water column is not well aerated. Phytoplankton consumes the produced nitrate rapidly, in which case a removal of nitrate may be observed.

High loads of N and P may lead to depletion of silicate (see the section on 'Estuaries as sieves for nutrients') during phytoplankton blooms, conditioning the successive phytoplankton assemblages during the remaining productive period. Subsequent low river discharges may provide an environment with low Si:N and N:P relative availability, and this may favour dominance of cyanobacteria and even toxic algal blooms (Rocha et al. 2002). Pulses of freshwater discharges (and their nutrients) stimulate the development of a more diversified phytoplankton assemblage that supports zooplankton (Rey et al. 1991), that is, top-down control, and this hinders harmful algae blooms.

Geochemical and biological transformations are concentrated in the ETM (see the section on 'SPM and sediments as repositories of organic carbon' and Simenstad et al. 1994). The long retention time of SPM in the ETM – and the nutrients it contains – enables bacteria to consume particulate nutrients and to incorporate them into primary levels of the

estuarine food web by increasing the importance of heterotrophic processes (Reed and Donovan 1994; Thayer 1974). However, this may also result in depleting the dissolved oxygen (see the section on 'SPM and sediments as repositories of organic carbon').

Light penetration is a limiting factor in the ETM, inhibiting phytoplankton production (see the section on 'Estuaries as sieves for nutrients'). This decrease of primary production by phytoplankton results in minimal excretion of photosynthetic products that are a food source to bacteria (Goosen et al. 1999). When SPM exceeds 50 mg L^{-1} , phytoplankton photosynthesis is severely limited (Cloern, 1987). In estuaries where primary production rates are less than $160 \text{ g C m}^{-2} \text{ year}^{-1}$, phytoplankton are light-limited and these systems can be considered heterotrophic (Heip et al. 1995). The phytoplankton biomass can also be regulated by vertical stratification in salinity (Cloern 1987).

Microphytobenthos comprises the microscopic, photosynthetic eukaryotic algae and cyanobacteria that grow on, or close to, the sediment–water interface in illuminated inshore areas. Microphytobenthos, anaerobic and chemosynthetic bacteria may also contribute to microphytobenthic production. Because they are found at the sediment interface, the microphytobenthos also modulates the exchange of nutrients between the sediment and water column (Serôdio and Catarino 1999).

Freshwater phytoplankton and bacterioplankton in the river are subject to salt stress when freshwater mixes with saltwater (Flameling and Kromkamp 1994). The freshwater microbial populations die in this zone (Goosen et al. 1995). Salinity changes also affect estuarine plankton and some invertebrate species (e.g., bivalves). Salt-hardy invasive species are increasingly invading this ecological niche. For instance, salinity changes in the Guadiana Estuary, Portugal, favour the introduced, salt-resistant, Asian clam (*Corbicula fluminea*) that is increasingly out-competing the salt-intolerant indigenous clam (*Anodonta cygnea*) (L. Chicharro, pers. comm.).

Estuarine fishes suffer less impact from salinity changes due to their ability to utilize osmotic regulation (Whitfield and Wooldridge 1994; Thayer et al. 1999). In the ETM, primary consumers are generalist and omnivorous feeders, capable of exploiting both autotrophic and heterotrophic food web pathways. The presence of higher standing

stocks of prey species attracts secondary and tertiary consumers in this region (Simenstad et al. 1990).

Salinity changes in the estuary induce a physiological stress in organisms that must then spend extra energy in physiological adaptation changes. This may reduce growth and body size in organisms. In fact, because of the extra energy spent in physiological adaptation, brackish water organisms are usually smaller than their marine relatives (Harris et al. 2000). Production and consumption are determined by the relative availability of food (Simenstad et al. 1990). The critical process determining the impact of hydrology on estuarine and coastal food webs is related to organism size. Size is a determinant feature in prey–predator interactions. Thus different food webs for the same species can exist in estuaries and coastal areas. Similarly, natural and human-induced changes in river flows lead to different food webs.

Feedback between biological and physical processes

There are numerous feedbacks between physical and biological processes in considering food webs; a few dominant ones are summarized here. Larval and juvenile species exist in estuaries, both in the water column and the bottom. Coastal species, such as the anchovy *Engraulis encrasicolus* or the crustacean *Crangon crangon*, use the estuary as a nursery ground (Chicharro et al. 2002). In turn, these groups affect the uptake and release of organic and inorganic materials (from excretion products) and influence fluxes between water and sediment.

On the mud and sand flats, detritivorous macrobenthic invertebrates benefit from the deposition of particulate organic matter that they use as a food source (Snelgrove and Butman 1994). These benthic organisms change the physical and chemical properties of sediments (see the section on 'Formation of mudflats'), modifying, in turn, the bottom shear stress, currents and flushing of the estuary (Jumars and Nowell 1984; Brown et al. 1998). Through bioturbation, these organisms can also cause sediment mobilization or, conversely, increase stability as in the case of dense mats of worm tubes, or by enhancing the deposition of fine particles through biodeposition (Schaffner 1990; Luckenbach et al. 1988; Widdows et al. 1998; Noji and Noji 1991; Ayukai and Wolanski 1997).

The amphipod *Corophium volutator* (Pallas) may reduce sediment binding by grazing benthic diatoms that contribute to sediment aggregation (Gerdol and Hughes 1994). Birds, by feeding on this amphipod, decrease the erosion of the sediment (Brown et al. 1998).

Enhanced erosion and resuspension of sediment through bioturbation may result in greater sediment accretion on adjacent saltmarshes (Brown et al. 1998). Bivalves also play an important role in reducing phytoplankton densities and controlling algal blooms (Takeda and Kurihara 1994). Attrition of particles in the gut, together with selective feeding of fine particle sizes by suspension and deposit-feeders and incorporation into faecal pellets, results in deposition of finer mineral and organic particles in the surrounding sediment, with an increase of absorbed and adsorbed nutrient and trace elements concentrations (Brown 1986). The resulting availability of these nutrients enables the growth of macrophytes and macroalgae, which are contributors to the primary production in estuaries (Simenstad et al. 1990).

The transition between estuaries and coastal zones

Compared with the extensive production and transport of detrital carbon, the consumption by detritivores constitutes only a minor reduction of the organic matter exported to the nearshore coastal waters. Detritus can be removed *via* grazing by zooplankton, epibenthos and benthic fauna. The exchange of zooplankton between estuary and adjacent coastal waters results in the export of organic carbon (Simenstad et al. 1990). The export of nutrients from the estuary also contributes to primary production in coastal zones.

The coastal fish community in the Guadiana Estuary, Portugal, changed from planktivorous during high river inflow periods, to demersal-carnivorous during low river discharge periods (Figure 3; L. Chicharo, pers. comm.). During periods of high river inflow, the increase in primary productivity and consequent availability of food for planktivorous pelagic fish species, like anchovy (*Engraulis encrasicolus*) or sardine (*Sardina pilchardus*), may explain higher fishery catches in the coastal zone (Soberón-Chávez and Yáñez-Arancibia 1985; Ray 1996; Sklar and Browder 1998; Chicharo

et al. 2002). Strong outwelling or river plumes are important for these species because they require turbidity or salinity signals for orientation to find adequate spawning areas (Kingsford and Suthers 1994).

Human health and aquatic ecosystems

Ecological integrity is central to human health (Epstein 1999). Threats to human health arising from human interaction with aquatic ecosystems can originate from multiple factors, which might be broadly grouped into three main categories:

- derived from the effects of water pollution (chemical, microbiological, radioactive, thermal) on humans and on the physiology of individual organisms;
- those resulting from management of aquatic resources (e.g., wetland drainage, aquaculture, dam construction);
- effect of global changes affecting climate and the hydrological cycle (e.g., habitat degradation, warming, increased rainfall, storms).

These categories are interlinked.

Examples of health threats derived from water pollution

A wide range of activities on land contributes to the release of contaminants (typically organic chemicals, heavy metals, microorganisms, sewage, nutrients) into the sea by rivers, run-off, groundwater and the atmosphere. The bulk of these contaminants remains in coastal waters, particularly in poorly flushed areas. Water related diseases cause millions of preventable deaths every year, especially among children (UNDP 1998). Greater incidence of illness due to consumption of contaminated fish and shellfish is an increasing concern. Harmful algal blooms in many coastal regions in the world cause a number of diseases, including poisoning, neurological disorders and gastroenteritis (HEED 1999; UNEP 1999). Increased flooding as a result of changes in precipitation may cause contamination of water supplies, which leads to greater incidence of faecal-oral contamination (WHO 1996). Typhoid and paratyphoid fevers, produced by *Salmonella typhi* and *S. paratyphi* A, B, C can occur in areas with poor sanitation and inadequate sewerage systems. They are transmitted by contaminated water

and food. Crop irrigation with untreated sewage can cause higher *Salmonella* infection rates among children living in the wastewater-spreading area (Melloul and Hassani 1999).

Also, eutrophication is depriving lakeside residents of good water quality in many densely populated areas of the world (UNEP 1999).

In the developed countries, *Cryptosporidium* and *Giardia* are two of the most common agents of surface water-associated infections and disease outbreaks. These parasitic infections, primarily causing diarrhoea, are difficult to control due to the high resistance of the oocysts and cysts to environmental stress and disinfectants. The level of contamination is particularly dependent on the land use in the water-catchment area, on wastewater discharge, and on weather conditions (Exner et al. 2001).

For decades, a great deal of research has concentrated on the direct influence of contaminants on aquatic biota, their bioaccumulation in trophic webs, and their detrimental effect on humans. Examples of threats to human health arising from the degradation of marine, freshwater and watershed ecosystems are summarised in Table 2. Information about these topics is available in the literature and will not be further related here.

Examples of health threats derived from management of aquatic resources

Wetlands

These ecosystems filter floodwaters and support the genetic, species and functional group diversity that ensures resilience to stress and resistance to pests and pathogens (Epstein 1999). Mangroves, marshes and floodplains have been devastated in the last centuries and decades. In the 20th century alone, some 10 million square kilometres of wetlands have been drained across the globe, an area about the size of Canada (Chivian 2002). This has brought losses of habitat and retention capacity of water and sediments, as well as the filtering function for waterborne nutrients. Interrelated issues affecting aquatic ecosystems such as loss of biological diversity, endangered physical coastal stability, floods, and eutrophication make it a priority to adequately restore these environments and the 'ecological services' they offer.

In some of the tropical regions of the developing world, the incidence of malaria has increased in recent years as the mosquito and the malaria parasite it spreads have evolved more resistance to DDT alternatives and to the medications used to prevent or treat the disease (Thurow 2001). Malaria now strikes more than 300 million people per year and kills about a million of those affected. In some regions (e.g., South Africa) DDT is again increasingly used to control mosquitoes. In general, land claim for agriculture, deforestation and changes in land use most probably bear the greatest blame for the climatic and habitat changes producing this development. Yet, it was primarily the drainage of wetlands for agricultural purposes that contributed most significantly to the eradication of malaria in Europe (Reiter 2000).

It is today often erroneously thought that larvae of malaria-transmitting mosquitoes can only develop in freshwater. Furthermore, malaria is not restricted to the tropics. Only in 1975 did the World Health Organization declare that Europe was free of malaria. About 200 years ago malaria was a leading cause of death in many marshlands communities along the coast of southern England (Reiter 2000). There, extensive salt-marshes provided high-quality grazing for sheep and cattle, but they were also a favoured habitat for a highly effective malaria vector, *Anopheles atroparvus*, which prefers to breed in brackish water along river estuaries and in the presence of abundant algae. Until the 19th century, malaria was a major mortality factor in The Netherlands. By the end of that century transmission had dropped precipitously in the more prosperous countries of North Europe. A major factor contributing to this decline was that the mosquito habitat was eliminated by improved drainage and extensive land claim. However, major epidemics occurred in Russia and Poland in the 1920s, with high death rates reaching regions near the Arctic Circle.

Today, malaria is again common in many parts of Central America, northern South America, tropical and subtropical Asia, some Mediterranean countries and many of the republics of the former USSR. Thus, policies on wetland creation or restoration must carefully consider, besides the benefits of the re-establishment of lost ecological services, the potential consequences of increased areas of slow-flowing or stagnant waters on disease

Table 2. Examples of threats to human health arising from the degradation of marine, freshwater and watershed ecosystems. Modified from UNEP (1999).

| Driving forces | Changing ecological pattern | Influence on human health/example |
|---|--|---|
| Marine | | |
| Pollution from oil, industry, naval operations and sewage discharge | Deterioration of marine ecosystems from imbalances due to dense ship traffic | Decrease in life expectancy, skin and eye diseases (Black/Azov seas, Caspian Sea) |
| Biological/bacterial contamination due to hydrological changes | Effects on fish and algae | Typhoid, malaria, diphtheria (Central Asia) |
| Rising sea levels | Ecosystem destruction and pasture degradation; changes in depth and amplitude of water fluctuations lead to reduction of wildlife habitats | See main text in the frame of climate change |
| Biological contamination of surface water with waste water | | Gastroenteritis, eye and skin infections (UK, France, S. Africa) |
| Biological and chemical contamination: harmful (toxic and nontoxic) algal blooms from the rapid reproduction and localized dominance of phytoplankton | Shellfish poisoning, wildlife mortalities, sunlight penetration prevention, oxygen shortages, reservoirs for bacteria | Poisonings, diarrhoea, dehydration, headaches, confusion, dizziness, memory loss, weakness, gastroenteritis, bacterial infections, swimming related illnesses, neurological diseases, deaths (Florida, Gulf States, S. America) |
| Contamination: cholera contaminated sea plankton due to contaminated ships' hulls | Coastal shellfish and fish contamination | Cholera (Peru and other 16 countries) |
| Freshwater | | |
| Dam construction | Inundating of lands | Schistosomiasis epidemics (Senegal River: Manantali and Diama dams) |
| Changes in hydrogeological cycle | Habitat alteration, new breeding sites for mosquitoes | Intestinal nematode infections (Yangtze River) |
| Changes in water flow: scarcity and degradation of water resources contaminated river systems | High levels of suspended soils and faecal bacterial content | Rift valley fever (Senegal) Faecal infections, intestinal diseases (S. Africa) |
| Pollution | | |
| Pesticides, herbicides, fertilizers and defoliants. Airborne toxic salts | Water quality deterioration | Health problems (general), infant mortality and morbidity (Central Asia) |
| Water contamination with heavy metals (Cadmium) | Accumulation in biochains | Progressive and irreversible kidney damage (Japan) |
| Faecal water pollution | | Diarrhoea, Hepatitis (Bangladesh, India) |
| Declines in ground water levels and groundwater contamination | Declines have been implicated in the widespread arsenic contamination of well water | Poisoning (Bangladesh, India) |
| Degradation, e.g., fallen water tables, chemical contamination | Severe increase in salinity | Cholera outbreak (1995), infectious diseases, intestinal parasites (Gazan aquifer, Palestine, Israel) |
| Catchment | | |
| Deforestation | Erosion, changes in hydrological cycles Firewood shortage, land degradation "slash and burn" use, increased wood transport and processing infrastructure New breeding grounds for insects | Malaria (S. Africa) Trauma, allergic reactions, infections, respiratory diseases, cancer (Gambia, Kenya, South Africa, Zimbabwe) Yellow fever (Ghana, Nigeria, Zaire, Sudan, Liberia, Cote d'Ivoire) |
| Agroecosystem development: e.g., land inundation for rice growing | New breeding grounds for insects | Japanese encephalitis (Japan, China, India) |
| Intensified agriculture, irrigation schemes damaging ecosystems | Insecticide resistance | Malaria (Central America) |

vector proliferation, particularly under a scenario of increasing temperatures, which could allow disease causing organisms to significantly extend their range.

Aquaculture

Threats to human health from aquaculture products can arise from the misuse of therapeutic drugs (e.g., chloramphenicol), chemicals (e.g., tributyltin), fertilizers (e.g., raw chicken manure containing salmonella), and viruses from shrimp farms spreading to wild stocks (US EPA 1998). The abandonment of small aquaculture ponds in tropical countries leads to an extension of mosquito habitats and concomitant increases in malaria. Escape of exotic species and animal pathogens into the environment can have a tragic impact on native aquatic species. The FAO and the WHO recommend that the Hazard Analysis and Critical Control Points (HACCP) concept be applied to fresh water aquaculture programs to control food-borne digenetic nematode infections in humans (Garret et al. 1997; and references therein).

Examples of health threats derived from global changes

Global warming

The effect of global warming on aquatic ecosystems, mainly concerning deforestation of watersheds and plankton blooms, is considered one of the driving forces behind re-emerging diseases such as cholera and malaria. Strong evidence links the outbreak of cholera in recent years to warmer seas caused by El Niño. There is a biological basis for a link between sea surface temperature (SST), marine ecology, and human cholera (Colwell 1996). Zooplankton, which feed on algae, accumulate *Vibrio cholerae* and other enteric pathogens. This may explain why cholera follows seasonal warming of SST that can enhance plankton blooms. *Vibrio* spp. in general are influenced by temperature and salinity (Lipp and Rose 1997). Many other factors promote algal growth, the reservoir of cholera, including pollution, eutrophication, the loss of mangrove filtration systems, and declining fisheries in offshore waters. The result was an epidemic between 1991 and 1993, which started in Asia and may have spread to South America in the bilge

water of ships. As many as 500,000 people were afflicted; there were 5000 deaths.

Habitat degradation

The degradation of ecosystems, such as forests, wetlands and coastal waters, is the single most important factor behind the current mass extinction of species. Compared with the knowledge of the impact of human pressure on aquatic ecosystems, almost nothing is known about the effects of diversity loss in these systems on human health. A new category of extremely serious threats to ecosystem and human health arises from the so-called 'emerging infectious diseases' (EIDs), which in several cases arise from human-induced habitat destruction or invasion. EIDs link health and the fate of humans and wildlife; EIDs are 'one of the most significant, yet underestimated, anthropogenic threats to biodiversity conservation' (Daszak et al. 2000). In turn, disturbances in biodiversity are probably a driving force in major emerging disease threats to humans. These include Hantavirus, influenza, AIDS, Lyme's disease, Ebola virus – they are all zoonotic or have resulted from pathogens switching hosts from wild animal reservoirs to humans.

If knowledge of the above field is just beginning for terrestrial ecosystems, much less is known about human-relevant EIDs in aquatic ecosystems. Mass mortalities due to disease outbreak have recently affected major marine taxa. Emerging diseases are associated with algal toxins, and bacteria and viruses are affecting fish, shorebirds, and mammals. Of great concern also are diseases that attack coral and sea grasses, essential habitats that sustain mobile species (Epstein 1999). An increase in the severity of coral bleaching in 1997–1998 is coincident with high El Niño temperatures. Such climate-mediated physiological stresses may compromise host resistance and increase frequency of opportunistic diseases. New diseases typically have emerged through host or range-shifts of known pathogens. Both climate and human activities may have also accelerated global transport of species, bringing together pathogens and previously unexposed host populations. There is hitherto no evidence of switch in the host-pathogen interactions at the level of aquatic fauna and humans, yet this may be 'absence of evidence, rather than

evidence of absence of a (climate) effect' (Kovats et al. 2001).

Changes in hydrological cycles

Hydrological change is a main threat to human health (WHO 2000). Through the joint WHO/UNICEF programme 'Health Map', specific GIS software was developed for this purpose, combining a standardized geographic database, a data manager and a mapping interface. However, human health is not usually included in interdisciplinary research projects dealing with coastal wetlands or river basin management programs. Increasingly, human health needs to become a major issue to be considered in management strategies.

There are many pathways through which hydrologically relevant events can affect health; notably when a river or stream inundates the flood plains (see the arrow labelled H in Figure 3). This produces changes in mosquito abundance (malaria, dengue fever), or contamination of surface water with human or animal waste, for example, rodent urine (leptospirosis). Conversely, droughts can produce changes in vector abundance, for example, a vector breeding in stagnant river ponds. Flooding may become more intense with climate change and can affect health through the spread of disease (Noji 1997; Menne et al. 1999). Thus, the elaboration of high-resolution topographic models (digital elevation models) of basins is necessary for a better prediction of the dispersal of inundation waters and the location of stagnant waters. Precisely in tropical coastal areas, where the impact of climate change on vector-transmitted diseases is of high concern, there is frequently a lack of topographic information with an adequate resolution for low-lying sectors. In vulnerable regions, the combination of risks to both food and water can exacerbate the health impact of even minor weather extremes such as floods and droughts (Webb and Iskandarani 1998). The only way to reduce vulnerability is to ensure the infrastructure for removal of solid waste and wastewater and to supply potable water. No sanitation technology is 'safe' when covered by floodwaters, as faecal matter mixes with floodwaters and is spread wherever the floodwaters run. Similarly, groundwater quality deterioration by saline intrusion due to climate change and sea-level rise must be considered (Sherif and Singh 1999).

Human impacts in estuaries from upland activities

Direct human impacts on estuaries arise from several sources. They range from the consequences of engineering works (e.g., damming of rivers, abstraction of water for irrigation, dredging, land claim of wetlands and harbour constructions) on the estuarine water residence time and erosion and sedimentation patterns, to the effects of wastewater discharges on the health of biota. Anthropogenic discharges include an array of chemical contaminants, such as metals, organometals, and petroleum hydrocarbons, organic compounds from pesticides, industrial waste products and nutrients. Dams built in uplands for water storage or power generation can result in lower river flows and raised estuarine salinities. Indirect human impacts in estuaries can be just as severe (see below) as a result of poor land-use management in the river catchment and of damming rivers.

Impacts from land clearing, soil erosion and damming

Land clearing, overgrazing and other poor farming practices considerably increase soil erosion (Wolanski and Spagnol 2000). Most of the eroded soil is fine, cohesive sediment. Much of this mud is not retained in reservoirs and flows right through to the estuary. When the excess sediment is deposited in the coastal waters, changes in coastal properties may be very rapid, occurring within a few decades. Examples of this are found in some bays along the Great Barrier Reef, which have become permanently muddy in only a few decades following land clearing and accelerated soil erosion (Wolanski and Duke 2002). The coral reefs are also being degraded by this excess fine sediment (Wolanski et al. 2003a). An example is the La Sa Fua River, in the wet, tropical island of Guam, Mariana Islands (Table 1). This river drains a steep sloping drainage basin, 1.5% that of the Cimanuk River. Due to poor land-use practices in this catchment of highly erodible volcanic soils, the sediment yield is twice that of the Mekong and Yangtze. The river has no delta and discharges directly to a shallow, coral-fringed bay where coral reefs are smothered by mud (Wolanski et al. 2003b).

Much of the eroded soil remains trapped in estuaries, resulting in increased siltation and flooding, increased turbidity, decreased primary productivity, and a loss of aesthetics resulting in an economic loss for the tourism industry. As a result, the estuary can be severely degraded. An example is the Cimanuk River, Indonesia (Table 1). Compared to the Mekong (Table 1), the Cimanuk River is very small. However, due to human activities, its sediment yield is thirty times larger. The catchment is yielding sediment at a rate exceeding 6000 ton mud $\text{km}^{-2} \text{ year}^{-1}$, possibly the highest recorded erosion rates in the world (Milliman and Meade 1983; Wolanski and Spagnol 2000). The river is now profoundly degraded and little more than a drain for eroded soil and human waste. As a result of siltation, flooding is common. To protect the farms from flooding, levees have been constructed; these need to be raised at a rate of 0.05–0.1 m year^{-1} and reach 4 m in height. Other costs involve hard engineering structures along the river to combat erosion of the levees and maintenance-dredging to combat siltation.

The Fly Estuary, Papua New Guinea, is rapidly silting due to increased sediment input from activities in the river catchment, as well as from sediment imported from coastal waters by tidal and wind-driven currents (Wolanski et al. 1998b). The Jiaojiang River estuary, China, is infilling at a rate of about 0.13 m year^{-1} , requiring continuous dredging to maintain navigability (Guan et al. 1998).

In the Mekong Estuary, Vietnam, some of the sediment that is flushed to the coastal zone during freshwater floods may return to the estuary during the low-flow season. Over time-scales of years, the natural system may be at quasi-equilibrium in the sense that the size, shape and depth of the system may change only very slowly, evolving at time-scales of decades. This is important because it gives humans the time to adapt to changing estuaries and coasts. However, there are plans for 100 hydroelectric dams and water diversion structures in the Mekong River and tributaries in riparian countries upstream of Vietnam, and this will disturb this quasi-equilibrium (Wolanski et al. 1996).

Many estuaries are dammed, and these dams trap much of the non-cohesive sediment (e.g., sand). As a result, many temperate estuaries are generally suffering from sediment starvation. The Nile River carried about 110×10^6 ton of sediment

per year before construction of the Aswan High Dam (Khafagy et al. 1992; Fanos 1995; Stanley and Warne 1998). About 90% of this sediment is now trapped in the reservoirs. As a result, coastal erosion is intense – the Rosetta and Damietta promontories are eroding at the rates of 106 m year^{-1} and 10 m year^{-1} , respectively. Similarly, the Ribarroja-Mequinenza dam on the Ebro River, Spain, traps about 96% of the riverine sediment. This has led to coastal recession at the river mouth area – stopping the previous seaward progradation of the delta (Jiménez and Sánchez-Arcilla 1993; Guillen and Palanques 1997). The story is repeated for the Colorado River, USA and Mexico, which once supplied 150×10^6 ton of sediment per year to the Gulf of California before damming (Meckel 1975). Damming has resulted in coastal recession. The Mississippi River suspended sediment load has decreased by about 40% between 1963 and 1989, and this may be the major cause for the recession of the Mississippi deltaic coast (Coleman et al. 1998; Streever 2001). It is also likely that the Three Gorges Dam in China, under construction, will generate coastal erosion and recession, thus reversing the present trend of progradation of the river delta (Chen et al. 1985; Xiqing 1998; Yang et al. 2001). Water diversion from China's Luanhe River has decreased the riverine sediment load by 95% and resulted in its delta's recession at a rate of about 17.4 m year^{-1} (Qian 1994). These problems occur for both small and large rivers. Indeed, similar coastal problems due to water regulation in the catchment of small rivers are degrading coastal embayments (Poulos et al. 1994; Jay and Simenstad 1996). The on-going, rapid shoreline retreat in several segments of Portugal is mainly caused by human activity and not natural fluctuations (Granja and Soares-de-Carvalho 1999; Dias et al. 2000).

In semi-arid regions, river floods are short-lived and can be suppressed by large dams. The balance between scouring of the estuary during occasional river floods, and the regular import into the estuary of coastal zone sediment by tidal pumping, is then disturbed by dams. The estuary may then silt, such as did indeed happen in the Ord Estuary, Australia, which has silted by $30 \times 10^6 \text{ m}^3$ over the last 30 years since the river was dammed (Wolanski et al. 2001).

Mangrove creeks are self-scouring; this effect is proportional to the mangrove area. If the latter decreases because of mangrove land claim, then

tidal creeks will silt in a few years (Wattayakorn et al. 1990).

Changes to estuarine biology and variability

Changes in river flows due to irrigation, damming and water diversion change the entire food web – even up to the level of fisheries – with significant negative socio-economic consequences (see above).

Longer-term climatic variability is also an important factor because it affects the density of important species and influences predator–prey relationships. The ultimate survival of many estuarine species depends both on the environmental state of the individual estuaries where they occur and critically on the continued existence of a chains of habitats (estuaries) to enable them to respond to long-term climatic changes.

Global warming, sea level rise and changes in precipitation and freshwater runoff to estuaries that result from greenhouse-gas emissions will alter not only the physical system but also its biology. At this stage it is difficult to predict these changes, partly because the predictions of economic and social development within the catchment and of climate change remain uncertain.

Ecohydrology as a solution for estuarine and coastal management

Estuary and coastal water degradation

Estuary and coastal water degradation occurs worldwide and is made apparent by the fact that coastal fisheries are collapsing worldwide. Large indicator species, such as sea turtles, sea lions, dugongs and manatees, are disappearing and critical habitats such as saltmarshes, mangroves, coral reefs and seagrass beds are also declining (Pauly et al. 1998; Lindeboom 2002; Hughes et al. 2003; Diop 2003). In many cases, particularly in less developed countries, it is not that traditional management has failed but that management is piecemeal or that there is no management at all; overexploitation is the rule. In countries where management policies and strategies are now in place, but did not exist in the past, the destruction of habitats and decreased water quality may be an important reason for the failure of fisheries to

recover from an earlier collapse due to overfishing (Jackson et al. 2001).

Human populations and economic activities in the coastal zone are increasing in most countries. At the same time, more rivers are dammed worldwide. Deforestation is continuing and in many countries, accelerating, while urban and rural activities are growing. Thus, even though sewage discharge from cities may increasingly be treated, diffuse sources remain unchecked, overall the river flow is increasingly modified, and the load of sediment and nutrients is increasing. No solution has yet been offered of how, using only limited financial resources, to preserve, restore and manage critical estuarine habitats, and reduce the load of nutrients and pollutants in waters in the face of increasing human population and its aspirations.

A solution that managers advocate the most at present is to reduce the amount of waste and to rely on a technological fix, namely to construct sewage treatment plants and to modify farming practices and technology. This technological approach is logical; nevertheless it often fails for reasons outlined in the section on 'Introduction: degraded estuaries and coastal waters'.

An ecohydrology-based solution is needed to restore the health of estuaries. Ecologically, the estuary is part of the river basin. It is the end-user of the river where incremental small-scale impacts become cumulative, and where the summation of minor environmental stresses becomes a critical issue. In practice, people have commonly not considered this integrative, end-member aspect of estuaries. Critically, in most countries, land-use managers dealing with development within the river catchment, water-resource managers dealing with hydrology and water quality, city councils dealing with growth and waste disposal in residential areas, and coastal and fisheries managers do not have the forum or mechanism to cooperate as a result of their various socio-economic and political imperatives.

Ecohydrology as a solution

In comparison with purely technological solutions, ecohydrology works from within ecosystems. Implementation of ecohydrologic solutions requires a profound knowledge of the interdependences between water and biota. By manipulating these

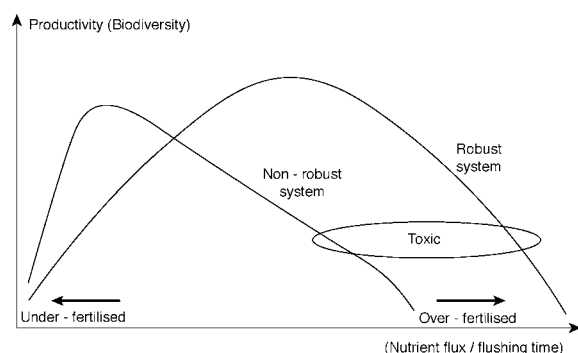


Figure 5. Sketch of the relationship between productivity (and also biodiversity) and the nutrient inflow in an estuary. Extra nutrients in small quantities may generate an increase of biological productivity without dramatic modification of biodiversity. More commonly the load of nutrients and pollutants is so high that it degrades the water quality, the ecological services, biodiversity, and productivity of coastal waters, leading in extreme cases to toxic algae blooms.

interdependences, it is possible to control ecosystem characteristics and increase system robustness.

Estuarine food webs show a strong dependence on river flow, the ecosystem shifting as a result of changing river inflow and salinity regimes. Therefore, it is necessary for estuary managers to take into account in their decision-making the dependence of food webs on natural (hydrologic and oceanic) factors, as well as on human-induced changes of river inflows of water, nutrients, pollutants and eroded soils. Key processes are sketched in Figures 3 and 5. Robust systems are less likely to become degraded than non-robust systems from disturbances such as fertilisation and changes in hydrology. Changes in estuarine and coastal zones biota will affect humans by degrading water quality and reducing its uses, and also by impacting on fisheries, with consequent socio-economic impacts.

The ecologically sustainable solution to estuary management is to adopt ecohydrology as the underpinning principle to guide the management of the entire river basin from the headwaters down to the coastal zone (Figure 6). Environmental degradation can only be remedied by restoring some of the working of the ecosystem and helping the partially restored system improve itself naturally. In the freshwater part of the river and especially in impoundments, the aquatic ecosystem is the least robust (Figure 2). It can be, and in most

countries it has been, degraded by human activities that alter the hydrologic regime, increase the nutrient load from the catchment, and destroy wetlands. Some of the aquatic ecosystem functions can be restored by a combination of engineering, ecohydrology, and phytotechnology (i.e., using plants) solutions. As described in the previous sections (see a summary in Figure 3), the estuary ecosystem is more robust than the river ecosystem, especially if the latter is impounded (Figure 2). However, while the system is more robust, it is more difficult to restore the system after it has been degraded, because the range of options is more limited. Under human influences, the level of degradation of the estuarine ecosystem will depend on two key factors, namely (1) the ratio between the nutrient flux and the flushing time, and (2) the robustness of the system. The robustness of the estuary is controlled both by a number of parameters, including the residence time, the estuarine food webs within the water column, and the buffer effect and the habitats provided by the fringing wetlands (principally mudflats, saltmarshes and mangroves). In the estuary it is generally not possible to use engineering solutions to solve environmental degradation problems, though in some cases it is possible to diminish the residence time by dredging, opening new river mouths, and, for dammed rivers, creating artificial river floods. Generally estuarine environmental restoration can only be carried out by enhancing the biotic integrity (Figure 3); restoring intertidal wetlands is one such key option. For non-robust estuaries (Figure 5) none of these solutions will be adequate and the health of the estuary can only be restored by adopting a basin-wide ecohydrology solution. This solution requires (1) regulating basin-wide human activities that impact on the river, and (2) manipulating the river ecosystem to decrease its impact on the estuary (Figure 6). The coastal zone is the most robust (Figure 2) and the only option available to restore environmental health of the coastal zone is to adopt a basin-wide ecohydrology solution (Figure 6).

A holistic, basin-scale approach is necessary because the whole river basin is a functioning, inter-related ecosystem (Zalewski 2002). UNEP recently advocated an integrated approach to the hydrological environment, emphasising links between river and coastal systems (Coccossis et al. 1999; Burt 2003). The Mersey Basin Campaign,

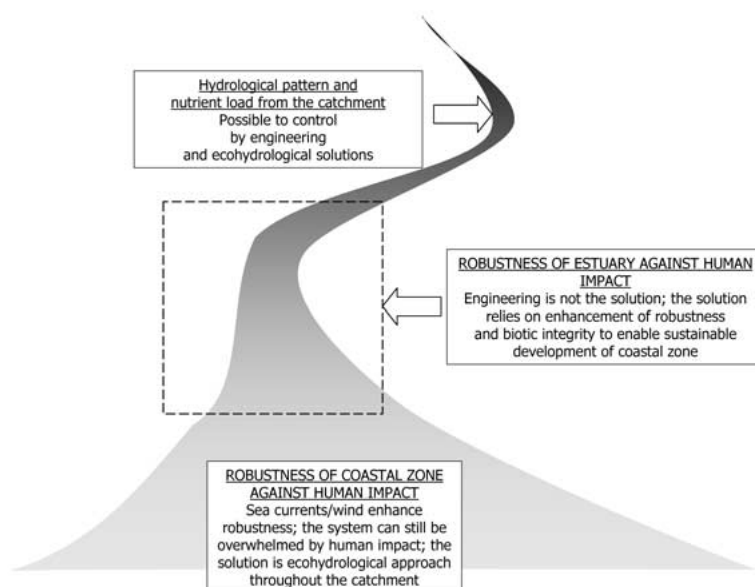


Figure 6. Sketch of the ecohydrology-based management of a river basin that is necessary to enable sustainable development of an estuary.

UK, is one of a number of recent examples in Europe, North America and Australasia where this basin-wide approach has been taken in environmental management, but even here coastal waters are not considered explicitly in the campaign's main objectives. Similarly, ecohydrology principles may be adopted in the restoration of specific tracts, watercourses and habitats, but they do not guide the over-arching ethos of the management strategy. Until this is changed, estuaries and coastal waters will continue to degrade.

Significantly, ecohydrology is a low-cost, uniformly applicable principle. Ecohydrology is a crucial element of future river basin and coastal management where growing human populations are putting increasing pressure on resources and the environment, particularly in the developing world, and where, in addition, the coast is also threatened by climate change and sea-level rise.

Ecohydrology relies on using our understanding of biota and process-interaction to moderate all elements of catchment and estuary robustness (e.g., flow rates, residence times and productivity) to enhance the ability of the river and coastal waters to recover from impacts. At the more specific level, phytotechnology is an element of ecohydrology that relies on using plants to improve water

quality by converting excess nutrients into plant biomass, and by sequestering pollutants.

The interplay between hydrology and biota, and associated fluxes and exchanges of sediment, nutrients and pollutants, can be used in the development of sustainable ecological solutions for water management. A number of practical steps are possible, the first of which is to reduce the discharge to the river of excess nutrients, sediment and pollutants from rural and urban areas through environmentally sensitive land-use and lifestyles in addition to high cost technology. More importantly, ecohydrology-based management relies on the use of feedback mechanisms between hydrology and biota as a management tool by using and manipulating plants and trees within the rivers, lakes, wetlands and flood plains, and, indeed, the coastal zone, to address specific problems of water, sediment, nutrients and pollutants. For example, wetlands can be used as a remediation tool to sequester excess nutrients into biomass, that is, production of bio-energy or timber, to limit their delivery to the estuary. Similarly, aquatic biota may be manipulated to control toxic algae. Woodlands and wetlands may even be used to regulate water flow, water quality and sediment delivery to the coast. The level at which these activities are implemented

will be dependent upon the robustness of the estuary. Riverine woodlands are another example of how biota may be used to regulate catchment-dependent fluxes; much of which has been lost in lowland Europe and North America to intensive agriculture and urban development and these need to be restored where possible (Peterken and Hughes 1995). Floodplain woodlands have drawbacks, such as limited control of floodwaters and increased evapotranspiration. Nevertheless, they offer improved sediment and pollutant retention and nutrient sequestration and, indeed, reduced diffuse discharge as a result of displacing floodplain farmland. They help reduce peak flood flow and maintain low flow; they increase habitat diversity and landscape quality, and timber production (Kerr and Nisbet 1996). They need restoration to improve the health of estuaries downstream.

In estuaries and coastal waters, ecohydrology-based solutions call for the restoration and creation of wetlands, including mudflats, mangroves and saltmarshes, because of their ability to trap sediment and pollutants, to convert excess water-born nutrients into plant biomass, to provide habitats for demersal and pelagic species, and, in some cases, to protect the coast from increased erosion following sea-level rise and sediment starvation from damming. This is an important element of estuarine ecohydrology where the 'knock-on' consequences of catchment land-clearance may be exceeding the capacity of intertidal wetlands to 'absorb' influxes of sediment, nutrients and pollutants, which then by-pass the estuary and enter coastal waters. Wetland creation of this kind necessitates a dynamic coast, using the balance between prevailing tidal levels and accommodation space to determine the pace at which intertidal wetlands may be manipulated to address degradation problems from the mild to the acute. These wetlands must be managed to reduce human health problems. In coastal waters, further applications of ecohydrology are the use of macrophytes to enhance the internal consumption rate, and benthic suspension feeders, such as bivalve molluscs, sponges, tunicates, and polychaetes, to filter and pelletize excess nutrients and plankton. There are, thus, several steps that may be taken in parallel.

The ecohydrologic approaches considered above have an important element of seasonality. Where possible, plant biomass may need to be harvested

annually from wetlands to remove nutrients. Such wetland manipulations go hand-in-hand with basing remediation measures on biological function and productivity. The practical steps that can be taken vary from estuary to estuary. Thus, the use of ecohydrology and phytotechnology in solving practical issues in an estuary, depends on the wise choice of appropriate management practices. These must reconcile the different spatial and temporal scales controlling nutrient and pollutant supply, the robustness of the estuary, the hydrologic regime, the tidal patterns and wave climate, and the biota; these are all factors that can be managed and balanced to enhance the self-purification ability and capacity of estuaries and coastal zones (Figure 6).

Also, it is absolutely necessary to consider the environmental and socio-economic factors of the whole river catchment, including the estuary and the coastal zone, when proposing developments such as land clearance, urbanisation, intensive agriculture and/or river damming. The environmental impacts of such developments must be fully evaluated and quantified before being included in the final cost-benefit analysis.

The issue of human health is critical when creating and managing wetlands, to seek equilibrium between the necessary water residence time for efficient nutrient sequestration and the minimisation of diseases (e.g., bilharzia and malaria) and vector breeding grounds. In intertidal wetlands, this may be less of an issue than in freshwater wetlands, depending on the flushing rate – which can, of course, be manipulated using ecohydrologic principles.

Thus, the successful management of estuaries and coastal waters requires an ecohydrology-based, basin-wide management, which considers the river basin as the fundamental unit of territorial management (Zalewski 2002). This necessitates changing present practices by official institutions based on municipalities or counties as an administrative unit, or based on managers of specific activities (e.g., farming, water resources, fisheries, urban developments). Generally, at present, the limits of these administrative or usage units do not coincide with basin boundaries. The ecohydrology approach also necessitates a high level of collaboration and opportunity to share approaches and experience on a range of forums to develop best practice. Without these changes, estuaries and

coastal waters will continue to degrade, possibly more slowly than at present, whatever local integrated coastal management plans are implemented.

To help in this process, research is needed better to understand the effects of hydrology on biotic processes, and of the biota on hydrology at the water basin scale. In addition, ecohydrologic management decisions need to be based on sound data regarding trends in degradation. Long-term ecosystem research and monitoring (LTERM), making use of data from monitoring programmes, historical documentary evidence and sedimentary records, is necessary to provide measures of baseline conditions and natural variability, and to inform decisions on ecosystem management (Parr et al. 2003). This is particularly true of hydrological systems where long data series are required to resolve slow, subtle signals in a 'noisy' dataset, or to identify the properties of rare events (Burt 1994). Long-term estuarine and wetlands research is needed to understand processes of change and to be able to quantify them so as to distinguish natural variability from human impact in marine ecosystems that are still in the hunter-gatherer stage of exploitation (Hawkins et al. 2003). Estuarine ecohydrologic principles must, therefore, be underpinned by high-quality LTERM, linked with remote sensing to enable broad scale detection, and with social science expertise to inform public, planning, policy and practice (Parr et al. 2003). Furthermore, in the developing world, these baseline data will need to be established at low cost, necessitating cheap, robust approaches that may be applied to sedimentary and biotic records to determine the nature and extent of recent ecosystem change.

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References

- Ackroyd D.R., Bale A.J., Howland R.J.M., Knox S., Millward G.E. and Morris A.W. 1986. Distributions and behaviour of dissolved Cu, Zn and Mn in the Tamar Estuary. *Estuarine, Coastal and Shelf Science* 23: 621–624.
- Adam P. 1990. *Saltmarsh Ecology*. Cambridge University Press, Cambridge.
- Al-Mohanna S.Y. and Meakins R. 2003. Endemic macrofauna of the Sulaibikhat Bay salt marsh and mudflat habitats, Kuwait: status and need for conservation. *Journal of Arid Environments* 54: 115–124.
- Allen J.R.L. 1990a. Salt-marsh growth and stratification: a numerical model with special reference to the Severn estuary, southwest Britain. *Marine Geology* 95: 77–96.
- Allen J.R.L. 1990b. Constraints on measurement of sea-level movements from salt-marsh accretion rates. *Journal of the Geological Society* 147: 5–7.
- Alongi D.M. 1987. The influence of mangrove derived tannins on intertidal meiobenthos in tropical estuaries. *Oecologia* 71: 537–540.
- Alongi D.M. 1998. *Coastal Ecosystem Processes*. CRC Press, Boca Raton, 419 pp.
- Alongi D.M. and Sasekumar A. 1992. Benthic communities. In: Robertson A.I. and Alongi D.M. (eds), *Tropical Mangrove Ecosystems*, American Geophysical Union, Washington, DC, pp. 137–171.
- Alongi D.M., Boto K.G. and Robertson A.I. 1992. Nitrogen and Phosphorus Cycles. In: Robertson A.I. and Alongi D.M. (eds), *Tropical Mangrove Ecosystems*, American Geophysical Union, Washington, DC, pp. 251–292.
- Alongi D.M., Christoffersen P. and Tirendi F. 1993. The influence of forest type on microbial-nutrient relationships in tropical mangrove sediments. *Journal of Experimental Marine Biology and Ecology* 171: 201–223.
- Anderson T.J. 2001. Seasonal variation in erodibility of two temperate, microtidal mudflats. *Estuarine, Coastal and Shelf Science* 53: 1–12.
- Anderson I.C., Canuel E.A. and Macko S.A. 2001. Nitrogen cycling through a fringing marsh-aquifer ecotones. *Marine Ecology Progress Series* 210: 25–39.
- Andresen H., Bakker J., Brongers M., Heydemann B. and Irmeler U. 1990. Long term changes of salt marsh communities by cattle grazing. *Vegetatio* 89: 137–148.
- Ayukai T. and Wolanski E. 1997. Importance of biologically mediated removal of fine sediments from the Fly River plume, Papua New Guinea. *Estuarine, Coastal and Shelf Science* 44: 629–639.
- Bale A.J., Morris A.W. and Howland R.J.M. 1985. Seasonal sediment movement in the Tamar estuary. *Oceanologica Acta* 8: 1–6.
- Balls P.W. 1994. Nutrient inputs to estuaries from nine Scottish East Coast rivers: influence of estuarine processes on inputs to the North Sea. *Estuarine, Coastal and Shelf Science* 39: 329–352.
- Biggs R.B., Sharp J.H., Church T.M. and Tramontano J.M. 1983. Optical properties, suspended sediments, and chemistry associated with the turbidity maximum of the Delaware

- Estuary. Canadian Journal of Fisheries and Aquatic Science 40(Suppl. 1): 172–179.
- Boorman L.A. 1996. Effects of environmental change on European salt marshes. In: Lefeuvre J.C. (ed.), Structure, Functioning and Exchange Potentialities with Marine Coastal Waters, vol. 5, University of Rennes, France, 50 pp.
- Boorman L.A. 2000. The functional role of salt marshes in linking land and sea. In: Sherwood B.R., Gardiner B.G. and Harris T. (eds), British Saltmarshes, Forrest Text for the Linnean Society, pp. 1–24.
- Boorman L.A., Hazelden J.H., Loveland P.J. and Wells J.G. 1994a. Comparative relationships between primary productivity and organic and nutrient fluxes in four salt marshes. In: Mitsch W.J. (ed.), Global Wetlands, Old World and New, Elsevier, Amsterdam, pp. 181–189.
- Boorman L.A., Hazelden J., Andrews R. and Wells J.G. 1994b. Organic and nutrient fluxes in four north-west European salt marshes. In: Dyer K.R. and Orth R.J. (eds), Changes in Fluxes in Estuaries: Implications from Science to Management, Olsen and Olsen, Fredensborg, pp. 243–248.
- Boorman L.A., Garbutt A. and Barrat D. 1998. The role of vegetation in determining patterns of the accretion in salt marsh sediment. In: Sedimentary Processes in the Intertidal Zone, Geological Society, Special Publication No. 139, pp. 389–399.
- Boto K.G. and Wellington J.T. 1983. Phosphorus and nitrogen nutritional status of a northern Australian mangrove forest. Marine Ecology Progress Series 11: 63–69.
- Boto K.G. and Wellington J.T. 1984. Soil characteristics and nutrient status in a northern Australian mangrove forest. Estuaries 7: 61–69.
- Boto K.G., Alongi D.M.M. and Nott A.L. 1989. Dissolved organic carbon – bacteria interactions at sediment water interface in a tropical mangrove system. Marine Ecology Progress Series 51: 243–251.
- Bouchard V. 1996. Production et devenir de la matière organique des halophytes dans un marais salé Européen en système macrotidal (Baie de Mont St. Michel). PhD thesis, University of Rennes.
- Boynton W., Kemp M. and Keefe C. 1982. Phytoplankton-bacteria interactions: an apparent paradox? Marine Ecology Progress Series 25: 23–30.
- Bridges P.H. and Leeder M.R. 1977. Sedimentary model for intertidal mudflat channels with examples from the Solway Firth, Scotland. Sedimentology 23: 533–552.
- Brown S.L. 1986. Faeces of intertidal benthic invertebrates: influence of particle selection in feeding on trace element concentration. Marine Ecology Progress Series 28: 219–231.
- Brown S.L., Warman E.A., McGrorty S., Yates M., Pakeman R.J., Boorman L.A., Goss-Custard J.D. and Gray A.J. 1998. Sediment fluxes in intertidal Biotopes: Biota II. Marine Pollution Bulletin 37: 173–181.
- Burt T.P. 1994. Long-term study of the natural environment: perceptive science or mindless monitoring? Progress in Physical Geography 18: 475–496.
- Burt T.P. 2003. Monitoring change in hydrological systems. The Science of the Total Environment 310: 9–16.
- Cahoon D.R., Lynch J.C. and Powell A.N. 1996. Marsh vertical accretion in a Southern California estuary. Estuarine, Coastal and Shelf Science 43: 19–32.
- Cai W.J., Wang W.C., Krest J. and Moore J.S. 2003. The geochemistry of dissolved inorganic carbon in a surficial groundwater aquifer in North Inlet, South Carolina, and the carbon fluxes to the coastal ocean. Geochimica et Cosmochimica Acta 67: 631–639.
- Carlson P.R., Yarbrow L.A., Zimmermann C.F. and Montgomery J.R. 1983. Pore water chemistry of an overwash mangrove island. In: Taylor W.K. and Whittier H.O. (eds), Future of the Indian River System, First Symposium, Melbourne, Florida, pp. 239–250.
- Chambers R.M., Osgood D.T. and Kalapasev N. 2003. Hydrological and chemical control of Phragmites growth in tidal marshes of SW Connecticut, USA. Marine Ecology Progress Series 239: 83–91.
- Chappell J. 1993. Contrasting Holocene sedimentary geologies of lower Daly River, northern Australia, and lower Sepik-Ramu, Papua New Guinea. Sedimentary Geology 83: 339–358.
- Chappell J. and Woodroffe C.D. 1994. Macrotidal estuaries. In: Carter R.W.G. and Woodroffe C.D. (eds), Coastal Evolution: Late Quaternary Shoreline Morphodynamics, Cambridge University Press, Cambridge, pp. 181–218.
- Chater E.H. and Jones H. 1957. Some observations on *Spartina townsendii* in the Dovey estuary. Journal of Ecology 45: 157–167.
- Chen J.Y., Zhu H.F., Dong Y.F. and Sun J.M. 1985. Development of the Changjiang Estuary and its submerged delta. Continental Shelf Research 4: 47–56.
- Chicharo L., Chicharo M.A., Esteves E., Andrade P. and Morais P. 2002. Effects of alterations in fresh water supply on the abundance and distribution of *Engraulis encrasicolus* in the Guadiana Estuary and adjacent coastal areas of south Portugal. Ecohydrology and Hydrobiology 1: 195–200.
- Chivian E. 2002. Biodiversity: its importance to human health. Interim Executive Summary, Center for Health and the Global Environment. Harvard Medical School, Harvard.
- Cloern J.E. 1987. Turbidity as a control on phytoplankton biomass and productivity in estuaries. Continental Shelf Research 7: 1367–1381.
- Coccossis H., Burt T.P. and van der Weide J. 1999. Conceptual framework and planning guidelines for integrated coastal area and river basin management. United Nations Environment Programme, Mediterranean Action Plan, Priorities Actions Programme, Split, Croatia.
- Cohen M. and Lara R.J. 2003. Temporal changes of mangrove vegetation boundaries in Amazônia: Application of GIS and remote sensing techniques. Wetlands Ecology and Management 11: 223–231.
- Coleman J.M., Robert H.H. and Stone G.W. 1998. Mississippi River Delta: an overview. Journal of Coastal Research 14: 696–716.
- Colwell R.R. 1996. Global Climate and Infectious Disease: The Cholera Paradigm. Science 274: 2025–2031.
- Corbett D.R., Dillon K., Burnett W. and Schaefer G. 2002. The spatial variability of nitrogen and phosphorous concentration in a sand aquifer influenced by onsite sewage treatment and disposal systems: a case study on St. George Island, Florida. Environmental Pollution 117: 337–345.
- Costa M.J. 1988. The Tagus and Mira estuaries (Portugal) and their role as spawning and nursery areas. Journal of Fish Biology 33(Suppl. A): 249–250.

- Costa M.J., Lopes M.T., Domingos I.M., de Almeida P.R. and Costa J.L. 1996. Final Report of the Portuguese Fauna Working Group – Tagus and Mira sites. In: Lefeuvre J.C. (ed.), Effects of Environmental Change on European Salt Marshes. Structure, Functioning and Exchange Potentialities with Marine Coastal Waters, vol. 3, University of Rennes, France, pp. 95–174.
- Cundy A.B., Croudace I.W., Thomson J. and Lewis J.T. 1997. Reliability of salt marshes as 'geochemical recorders' of pollution input: a case from contrasting estuaries in southern England. *Environmental Science and Technology* 31(4): 1093–1101.
- Dame R.F. and Kenny P.D. 1986. Variability of *Spartina alterniflora* production in the euhaline North Inlet estuary. *Marine Ecology Progress Series* 32: 71–80.
- Dame R.F. and Lefeuvre J.C. 1994. Tidal exchange, import–export of nutrient and organic matter in New World and Old World salt marshes: conclusion. In: Mitsch W.J. (eds), *Global Wetlands. Old World and New*, Elsevier, Amsterdam, pp. 303–305.
- Dankers N., Binsbergen N., Zegers K., Laane R. and Van der Loeff M.R. 1984. Transportation of water, particulate and dissolved organic and inorganic matter between a salt marsh and the Eems-Dollard estuary, the Netherlands. *Estuarine, Coastal and Shelf Science* 19: 143–165.
- Daszak P., Cunningham A.A. and Hyatt A.D. 2000. Emerging infectious diseases of wildlife-threats to biodiversity and human health. *Science* 287: 443–449.
- Degan L.A., Finn J.T., Hopkinson C.S., Giblin A.E., Peterson B.J., Fry B. and Hobbie J.E. 1994. Flow model analysis of the effects of organic matter–nutrient interactions on estuarine trophic dynamics. In: Dyer K.R. and Orth R.J. (eds), *Changes in fluxes in estuaries: implications from science to management*, Proceedings of ECSA22/ERF Symposium, 13–18 September 1992, Plymouth, Olsen and Olsen, pp. 273–281.
- Dias J.M., Boski A., Rodrigues T. and Magalhães F. 2000. Coastline evolution in Portugal since the last glacial maximum until present – a synthesis. *Marine Geology* 170: 177–186.
- Diaz R.J. and Rosenberg R. 1995. Marine benthic hypoxia: a review of its ecological effects and the behavioural responses of benthic macrofauna. *Oceanography and Marine Biology: an Annual Review* 33: 245–303.
- Dijkema K.S. 1983. The salt-marsh vegetation of the mainland coast, estuaries and halligen. In: Dijkema K.S. and Wolff W.J. (eds), *Flora and vegetation of the Wadden Sea Islands and Coastal Areas*, Stichting Veth, Leide, pp. 185–220.
- Dijkema K.S. 1997. The influence of salt marsh vegetation on sedimentation. In: Eisma D. (ed.), *Intertidal Deposits: River Mouths, Tidal flats and Coastal Lagoons*, CRC Press, Boca Raton, FL, pp. 403–414.
- Diop S. 2003. Vulnerability assessments of mangroves to environmental change. *Estuarine, Coastal and Shelf Science* 58: 1–2.
- Dittmar T. and Lara R.J. 2001. Driving forces behind nutrient and organic matter dynamics in a mangrove tidal creek in North Brazil. *Estuarine, Coastal and Shelf Science* 52: 249–259.
- Doering P.H., Oviatt C.A., Nowicki B.L., Klos E.G. and Reed L.W. 1995. Phosphorus and nitrogen limitation of primary production in a simulated estuarine gradient. *Marine Ecology Progress Series* 124: 271–287.
- Dunne T. and Leopold L.B. 1978. *Water in Environmental Planning*. W.H. Freeman, San Francisco.
- Eisma D., Cadée G.C., Laane R. and Kalf J. 1982. Preliminary results of Aurelia and Navicula cruises in the Rhine and Ems estuaries, Jan. – Feb. 1982. *Mitteilungen aus dem Geologisch-paleontologischen Institut der Universität Hamburg, SCOPE/UNEP Sonderband*, 52: 633–653.
- Epstein P.R. 1999. Climate and health. *Science* 285: 347–348.
- Etcheber H. 1983. *Biogéochimie de la matière organique en milieu estuarien: comportement, bilan, propriétés. Le cas de la Gironde*. Thèse Doctorat d'Etat, Bordeaux, France.
- Exner M., Gornik V. and Kistemann T. 2001. Characterization, risk assessment and prevention of waterborne parasitic infections. *Bundesgesundheitsblatt, Gesundheitsforschung, Gesundheitsschutz* 44: 358–363.
- Fanos A.M. 1995. The impacts of human activities on the erosion and accretion of the Nile Delta coast. *Journal of Coastal Research* 11: 821–833.
- Flameling I.A. and Kromkamp J. 1994. Responses of respiration and photosynthesis of *Scenedesmus protuberans* (Fritsch) to gradual and steep salinity increases. *Journal Plankton Research* 16: 1781–1791.
- Fox W.M., Johnson M.S., Jones S.R., Leah R.T. and Copplestone D. 1999. The use of sediment cores from stable and developing salt marshes to reconstruct historical contamination profiles in the Mersey Estuary, UK. *Marine Environmental Research* 47: 311–329.
- French J.R. 1993. Numerical simulation of vertical marsh growth and adjustment to accelerated sea level rise, North Norfolk UK. *Earth Surface Processes and Landforms* 18: 63–81.
- French P.W. 1996. Long-term variability of copper, lead, and zinc in intertidal salt marsh sediments of the Severn Estuary. *Mangroves and Salt Marshes* 1: 59–68.
- Fujimoto K., Miyagi T., Kikuchi T. and Kawana T. 1996. Mangrove habitat formation and response to Holocene sea-level changes on Kosrae Island, Micronesia. *Mangroves and Salt Marshes* 1: 47–57.
- Furukawa K., Wolanski E. and Muller H. 1997. Currents and sediment transport in mangrove forests. *Estuarine, Coastal and Shelf Science* 44: 301–310.
- Gameson A.L.H. 1981. *The Quality of the Humber Estuary, 1961–1981*. The Yorkshire Water Authority, Leeds, UK, 88 pp.
- Garret E.S., dos Santos C.L. and Jahnke M.L. 1997. Public, animal and environmental health implications of aquaculture. *Emerging Infectious Diseases* 3: 435–457.
- Gerdol V. and Hughes R.G. 1994. Effect of *Corophium volutator* on the abundance of benthic diatoms, bacteria and sediment stability in two estuaries in south-eastern England. *Marine Ecology Progress Series* 114: 109–115.
- Goosen N.K., van Rijswijk P. and Brockmann U. 1995. Comparison of heterotrophic bacterial production rates in early spring in the turbid estuaries of the Scheldt and the Elbe. *Hydrobiologia* 311: 31–42.
- Goosen N.K., Kromkamp J., Peene J., van Rijswijk P. and van Breugel P. 1999. Bacterial and phytoplankton production in the maximum turbidity zone of three European estuaries: the Elbe, Westerhelde and Gironde. *Journal of Marine Sciences* 22: 151–171.
- Grabemann I., Kühle H., Kunze B., Müller A. and Neumann L.J.R. 1990. Studies on the distribution of oxygen

- and nutrients in the Weser Estuary. In: Michaelis W. (ed.), Estuarine Water Quality Management, Coastal and Estuarine Studies, Springer-Verlag, Berlin, pp. 341–344.
- Gramling C.M., McCorkle D.C., Mulligan A.E. and Woods T.L. 2003. A carbon isotope method to quantify groundwater discharge at the land-sea interface. *Limnology and Oceanography* 48: 957–970.
- Granja H.M. and Soares-de-Carvalho G. 1999. Inland beach migration and predictability of change in the northwest coastal zone of Portugal. In: Barthel K.G., Barth H., Bohle-Carbonell M., Fragakis C., Lippiatou E., Martin P., Ollier G. and Weydert M. (eds), Proceedings of Third European Marine Science and Technology Conference (MAST), Luxembourg, European Commission DG12, 315 p.
- Gray A. 1992. Saltmarsh plant ecology: zonation and succession revisited. In: Allen J. and Pye K. (eds), Saltmarshes: Morphodynamics, Conservation and Engineering Significance, Cambridge University Press, Cambridge, pp. 63–79.
- Gray A.J., Drury M. and Raybould A.F. 1990. *Spartina* and the ergot fungus *Claviceps purpurea* – a singular contest?. In: Burdon J.J. and Leather S. (eds), Pests, Pathogens and Plant Communities, Blackwell Scientific, Oxford, pp. 63–79.
- Gray A.J., Marshall D.F. and Raybould A.F. 1991. A century of evolution of *Spartina anglica*. *Advances in Ecological Research* 21: 1–62.
- Gren I.M., Turner K. and Wulf F. 2000. Managing a Sea: The Ecological Economics of the Baltic. Earthscan, London, 138 pp.
- Grindrod J. and Rhodes E.G. 1984. Holocene sea-level history of a tropical estuary: Missionary Bay, North Queensland. In: Thom B.G. (ed.), Coastal Geomorphology in Australia, Academic Press, Sydney, pp. 151–178.
- Groenendijk A.M. 1984. Primary production of four dominant salt marsh angiosperms in the SW Netherlands. *Vegetatio* 57: 143–152.
- Guan W.B., Wolanski E. and Dong L.X. 1998. Cohesive sediment transport in the Jiaojiang River estuary, China. *Estuarine, Coastal and Shelf Science* 46: 861–871.
- Gueuné Y. and Winett G. 1994. The transport of the pesticide atrazine from the fresh water of the wetlands of Brittany to the salt water of the Bay of Mont Sainte-Michel (France). *Journal of Environmental Science and Health* 29: 753–768.
- Guillen J. and Palanques A. 1997. A historical perspective of the morphological evolutions in the lower Ebro River. *Environmental Geology* 30: 174–180.
- Hager S.W. and Schemel L.E. 1996. Dissolved inorganic nitrogen, phosphorus and silicon in South San Francisco Bay. 1. Major factors affecting distributions. In: Hollibaugh J.T. (ed.), San Francisco Bay: the Ecosystem, American Association for the Advancement of Science, San Francisco, pp. 189–216.
- Harris R.P., Wiebe P.H., Lenz J., Skodal H.R. and Huntley M. 2000. ICES Zooplankton Manual. Academic Press, New York, 684 pp.
- Hawkins S.J., Southward A.J. and Genner M.J. 2003. Detection of environmental change in a marine ecosystem – evidence from the western English Channel. *The Science of the Total Environment* 310: 245–256.
- HEED, 1999. Global Change Program. Marine Ecosystems: Emerging Diseases as Indicator of Change. Health, Ecological and Economic Dimensions. <http://heed.unh.edu/heedreport/exec/exec001.html>.
- Heip C.H.R., Goosen N.K., Herman P.M.J., Kromkamp J., Middleburg J.J. and Soetaert K. 1995. Production and consumption of biological particles in temperate tidal estuaries. *Oceanography and marine Biology: An Annual Review* 33: 1–149.
- Hemminga M.A., Huiskes A.H.L., Steegstra M. and van Soelen J. 1996a. Assessment of carbon allocation and biomass production in a natural stand of salt marsh plant *Spartina anglica* using ¹³C. *Marine Ecology Progress Series* 130: 169–178.
- Hemminga M.A., Cattrijsse A. and Wielemaker A. 1996b. Bedload and near-bed detritus transport in a tidal salt marsh creek. *Estuarine, Coastal and Shelf Science* 42: 55–62.
- Holland A., Zingmark R. and Dean J. 1974. Quantitative evidence concerning the stabilization of sediments by marine benthic diatoms. *Marine Biology* 27: 191–196.
- Hughes T.P., Baird A.H. et al. 2003. Climate change, human impacts, and the resilience of coral reefs. *Science* 301: 929–933.
- Jackson J.B.C., Hughes T.P. et al. 2001. Historical overfishing and the recent collapse of coastal ecosystems. *Science* 293: 629–638.
- Jacobsen A., Egge J.K. and Heimdal B.R. 1995. Effects of increased concentration of nitrate and phosphate during a spring bloom experiment in microcosm. *Journal of Experimental Marine Biology and Ecology* 187: 239–251.
- Jakobsen B. 1954. The tidal area in south-western Jutland and the process of the salt marsh formation. *Geographisk Tidsskrift* 53: 49–61.
- Jakobsen B., Jensen K.M. and Nielsen N. 1955. Forlag til landvindingsar-bedjerlangs den somderjyke vadehavskyst. *Geographisk Tidsskrift* 55: 62–87.
- Jassby A.D., Cloern J.E. and Powell T.M. 1993. Organic carbon sources and sinks in San Francisco Bay: variability induced by river flow. *Marine Ecology Progress Series* 95: 39–54.
- Jay D.A. and Simenstad C.A. 1996. Downstream effects of water withdrawal in a small, high-gradient basin: erosion and deposition on the Skokomish River delta. *Estuaries* 19: 501–517.
- Jiménez J.A. and Sánchez-Arcilla A. 1993. Medium-term coastal response at the Ebro Delta, Spain. *Marine Geology* 114: 105–118.
- Jones J.A.A. 2002. Natural soil piping, water quality and catchment management in the British Uplands. In: Cunnane C. and Barrins J. (eds), Celtic water in European framework – pointing the way to quality, Proceedings of the 3rd Inter-Celtic Colloquium on Hydrology and Management of Water Resources, National University of Ireland, Galway, pp. 98–114.
- Jorgensen S.E. 1997. Integration of Ecosystem Theories: a Pattern. Kluwer, Dordrecht.
- Jorgensen S.E. and de Bernardi R. 1998. The use of structural dynamics models to explain success and failures in biomanipulation. *Hydrobiologia* 379: 147–158.
- Jumars P.J. and Nowell A.R.M. 1984. Effects of benthos on sediment transport: difficulties with functional grouping. *Continental Shelf Research* 3: 115–130.
- Kamps L.F. 1962. Mud distribution and land reclamation in the eastern Wadden Sea shallows. *Rijkswaterstaat Communications* 4: 1–73.

- Kerr G. and Nisbet T.R. 1996. The restoration of floodplain woodlands in lowland Britain: a scoping study and recommendations for research. R&D Technical Report W15, Forestry Commission Research Division, 28 pp.
- Khafagy A.A., Naffaa M.G., Fanos A.M. and Dean R.G. 1992. Nearshore coastal changes along the Nile Delta shores. Proceedings of the 23rd Coastal Engineering Conference, pp. 3260–3270.
- Kingsford M.J. and Suthers I.M. 1994. Dynamic estuarine plumes and fronts: importance to small fish and plankton in coastal waters of NSW, Australia. *Continental Shelf Research* 14: 655–672.
- Kirby C.J. and Gosselink J.G. 1976. Primary production in a Louisiana Gulf Coast *Spartina alterniflora* marsh. *Ecology* 57: 1052–1059.
- Kirchman D., Keil R. and Wheeler P. 1989. The effect of amino acids on ammonium utilization and regeneration by heterotrophic bacteria in the subarctic Pacific. *Deep Sea Research* 36: 1736–1776.
- Knox G.A. 1986. *Estuarine Ecosystems: a Systems Approach*, vol. II, CRC Press, Florida, 230 pp.
- König D. 1948. *Spartina townsendii* an der westk üste von schleswig-holstein. *Planta* 36: 34–70.
- Kovats R.S., Campbell-Lendrum D.H., Mc-Michael A.J., Woodward A. and Cox J.S. 2001. Early effects of climate change: do they include changes in vector-borne disease? *Philosophical Transactions Royal Society of London B* 356(1411): 1057–1068.
- Lancelot C., Martin J.-M., Panin N. and Zaitsev Y. 2002. The north-western Black Sea: a pilot study to understand the complex interaction between human activities and the coastal environment. *Estuarine, Coastal and Shelf Science* 54: 279–283.
- Langlois E., Bonis A. and Bouzillé J.B. 2001. The response of *Puccinellia maritima* to burial: a key to understanding its role in salt marsh dynamics? *Journal of Vegetation Science* 12: 289–297.
- Langlois E., Bonis A. and Bouzillé J.B. 2003. Sediment and plant dynamics in saltmarshes pioneer zone: *Puccinellia maritima* as a key species? *Estuarine, Coastal and Shelf Science* 56: 239–249.
- Lara R.J. and Dittmar T. 1999. Nutrient dynamics in a mangrove creek (North Brazil) during the dry season. *Mangroves and Salt Marshes* 3: 185–195.
- Lara R.J., Szlafsztein C.F., Cohen M.C.L., Berger U. and Glaser M. 2003. Consequences of mangrove dynamics for private land use in Bragança, North Brazil: a case study. *Journal of Coastal Conservation* (in press).
- de Leeuw, J.A. van den Dool, W. de Munck, J. Nieuwenhuize and W.G. Beftink 1991. Factors influencing the soil salinity regime along an intertidal gradient. *Estuarine, Coastal and Shelf Science* 32: 87–97.
- Lefeuvre J.C. 1996. Effects of Environmental Change on European Salt Marshes. Structure, Functioning and Exchange Potentialities with Marine Coastal Waters, vols. 1–6, University of Rennes, France.
- Lefeuvre J.C. and Dame R.F. 1994. Comparative studies of salt marsh processes on the New and Old Worlds: an introduction. In: Mitsch W.J. (ed.), *Global Wetlands. Old World and New*, Elsevier, Amsterdam, pp. 169–179.
- Lefeuvre J.C., Bertru G., Burel F., Briant L., Creach V., Guené Y., Levasseur J., Mariotti A., Radureau A., Retiere C., Savouré B. and Troccaz P. 1994. Comparative studies on salt marsh processes: Mont St. Michel Bay, a multi-disciplinary study. In: Mitsch W.J. (ed.), *Global Wetlands. Old World and New*, Elsevier, Amsterdam, pp. 215–234.
- Leggett D., Bubb J.M. and Lester J.N. 1995. The role of pollutants and sedimentary processes in flood defence. A case study: saltmarshes of the Essex coast, UK. *Environmental Technology* 16: 457–466.
- Leonard L.A., Hine A.C. and Luther M.E. 1995. Surficial sediment transport and deposition processes in a *Juncus roemerianus* marsh, West-Central Florida. *Journal of Coastal Research* 11: 322–336.
- Li C., Wang P., Daidu F., Bing D. and Tiesong L. 2000. Open-coast intertidal deposits and the preservation potential of individual laminae: a case study from east-central China. *Sedimentology* 47: 1039–51.
- Lindeboom H. 2002. The coastal zone: an ecosystem under pressure. In: Field J.G., Hempel G. and Summerhayes C.P. (eds), *Oceans 2020: Science, Trends, and the Challenge of Sustainability*, Island Press, Washington, pp. 49–84.
- Linhurst R.A. and Reimold R.J. 1978. An evaluation of the methods for estimating the net aerial primary productivity of estuarine angiosperms. *Journal of Applied Ecology* 15: 919–931.
- Lipp E.K. and Rose J.B. 1997. Le rôle des poissons et des fruits de mer dans les toxi-infections alimentaires aux États-Unis d'Amérique. *Revue Scientifique et Technique office International des Epizooties* 16: 620–640.
- Liu Y.P., Millward G.E. and Harris J.R.W. 1998. Modelling the distributions of dissolved Zn and Ni in the Tamar Estuary using hydrodynamics coupled with chemical kinetics. *Estuarine, Coastal and Shelf Science* 47: 535–546.
- Luckenbach M.W., Huggett D.V. and Zobrist E.C. 1988. Sediment transport, biotic modifications and selection of grain size in a surface deposit feeder. *Estuaries* 11: 134–139.
- Mallin M.A. 2000. Impacts of industrial-scale swine and poultry production on rivers and estuaries. *American Scientist* 88: 26–37.
- Matson E.A. and Brinson M.M. 1990. Stable carbon isotopes and the C:N ratio in the estuaries of the Pamlico and Neuse Rivers, North Carolina. *Limnology Oceanography* 35: 1290–1300.
- McCaffrey R.J. and Thomson J. 1980. A record of the accumulation of sediment and trace metals in a Connecticut salt marsh. *Advances in Geophysics* 22: 83–96.
- McKee K.L. 1993. Soil physicochemical patterns and mangrove species distribution- reciprocal effects? *Journal of Ecology* 81: 477–487.
- Meckel L.D. 1975. Holocene sand and bodies in the Colorado Delta area, northern Gulf of California. In: Broussard (ed.), *Deltas – Models for Exploration*, Houston Geological Society, Houston, pp. 239–265.
- Melloul A.A. and Hassani L. 1999. Salmonella infection in children from the wastewater-spreading zone of Marrakesh city (Morocco). *Journal of Applied Microbiology* 87: 536–539.
- Menne B., Pond K., Noji E.K. and Bertollini R. 1999. Floods and public health consequences, prevention and control measures. Discussion paper presented at the UNECE Seminar on flood prevention, Berlin, 7–8 October 1999.

- Meybeck M., Cauwet G., Dessery S., Somerville M., Goulet D. and Billen G. 1988. Nutrients (organic C, P, N, Si) in the eutrophic river Loire (France) and its estuary. *Estuarine, Coastal and Shelf Science* 27: 595–624.
- Milliman J.D. and Meade R.H. 1983. World-wide delivery of river sediment to the oceans. *Journal of Geology* 91: 1–21.
- Mitsch W.J. 1993. Ecological engineering – a cooperative role with the planetary life-support system. *Environmental Science and Technology* 27: 438–445.
- Moeller I., Spencer T. and French J.R. 1997. Wind wave attenuation over salt marsh surfaces: Preliminary results from Norfolk, England. *Journal of Coastal Research* 12: 1009–1016.
- Møller J.T. 1963. Vadehavet mellem Mando og Ribe A. *Folio Geografica Danica* 8: 1–40.
- Monbet Y. 1992. Control of phytoplankton biomass in estuaries: A comparative analysis of microtidal and macrotidal estuaries. *Estuaries* 15: 563–571.
- Moore M.T., Rodgers J.H. Jr., Smith S. Jr. and Cooper C.M. 2001. Mitigation of metolachlor-associated agricultural runoff using constructed wetlands in Mississippi, USA. *Agriculture, Ecosystems and Environment* 84: 169–176.
- Morley J.V. 1973. Tidal immersion of *Spartina* marsh at Bridgwater Bay, Somerset. *Journal of Ecology* 36: 383–386.
- Morris A.W. 1988. Kinetic and equilibrium approaches to estuarine chemistry. *The Science of the Total Environment* 97/98: 253–266.
- Morris A.W., Bale A.J. and Howland R.J.M. 1981. Nutrient distributions in an estuary: evidence of chemical precipitation of dissolved silicate and phosphate. *Estuarine, Coastal and Shelf Science* 12: 205–216.
- Morris A.W., Loring D.H., Bale A.J., Howland R.J.M., Mantoura R.F.C. and Woodward E.M.S. 1982. Particle dynamics, particulate carbon and the oxygen minimum in an estuary. *Oceanologica Acta* 5: 349–353.
- Müller A., Grabemann I. and Kunze B. 1991. Water quality modelling: prediction of the transport of water constituents in the Weser Estuary (Germany). *Estuarine and Coastal Modeling*, Wastewater Division, ASCE, pp. 405–417.
- Newell S.Y. and Krambeck C. 1995. Responses of bacterioplankton to tidal inundations of a salt marsh in a flume and adjacent mussel enclosures. *Journal of Experimental Marine Biology and Ecology* 190: 79–95.
- Nienhuis P.H. 1993. Nutrient cycling and food webs in Dutch estuaries. *Hydrobiologia* 265: 15–44.
- Nixon S.W., Ammerman J.W., Atkinson L.P., Berounsky V.M., Billen G., Boicourt W.C., Boynton W.R., Church T.M., Ditoro D.M., Elmgren R., Garber J.H., Giblin A.E., Jahnke R.A., Owens N.J.P., Pilson M.E.Q. and Seitzinger S.P. 1996. The fate of nitrogen and phosphorus at the land–sea margin of the North Atlantic Ocean. *Biogeochemistry* 35: 141–180.
- Noji C.I.M. and Noji T.T. 1991. Tube lawns of spionid polychaetes and their significance for recolonization of disturbed benthic substrates. *Meeresforsch* 33: 235–246.
- Noji E. 1997. *The Public Health Consequences of Disasters*. Oxford University Press, New York, 496 pp.
- Okaichi T. and Yanagi T. 1997. *Sustainable development in the Seto Inland Sea – from the Viewpoint of fisheries*. Terra Publishing Company, Tokyo, 329 pp.
- Oldfield F., Richardson N., Appleby P.G. and Yu L. 1993. ²⁴¹Am and ¹³⁷Cs activity in fine grained sediments from parts of the N.E. Irish Sea shoreline. *Journal of Environmental Radioactivity* 19: 1–24.
- Owens N.J.P. 1986. Estuarine nitrification: a naturally occurring fluidized bed reaction? *Estuarine, Coastal and Shelf Science* 22: 31–44.
- Owens R.E., Balls P.W. and Price N.B. 1997. Physicochemical processes and their effects on the composition of suspended particulate material in estuaries: implications for monitoring and modelling. *Marine Pollution Bulletin* 34: 51–60.
- Parr T.W., Sier A.R.L., Battarbee R.W., Mackay A. and Burgess J. 2003. Detecting environmental change: science and society – perspectives on long-term research and monitoring in the 21st century. *The Science of the Total Environment* 310: 1–8.
- Paterson D.M. 1994. Microbiological mediation of sediment structure and behaviour. In: Stal L.J. and Caumette P. (eds), *Microbial Mats*, vol. 35, NATO ASI series, pp. 97–109.
- Pauly D., Christensen V., Dalsgaard J., Froese R. and Torres F. Jr. 1998. Fishing down marine food webs. *Science* 279: 860–863.
- Pennock J.R. and Sharp J.H. 1986. Phytoplankton production in the Delaware Estuary: temporal and spatial variability. *Marine Ecology Progress Series* 34: 143–155.
- Peterken G.F. and Hughes F.M.R. 1995. Restoration of floodplain forests in Britain. *Forestry* 68: 187–202.
- Pethick J.S. 1981. Long-term accretion rates on tidal salt marshes. *Journal of Sedimentary Petrology* 51: 571–77.
- Pethick J.S. 1984. *An Introduction to Coastal Geomorphology*. Edward Arnold, London, 260 pp.
- Pethick J.S. 1992. Saltmarsh geomorphology. In: Allen J.R.L. and Pye K. (eds), *Saltmarshes: Morphodynamics, Conservation and Engineering Significance*, The University Press, Cambridge, pp. 41–62.
- Pethick J., Leggett D. and Husain L. 1992. Boundary layers under salt marsh vegetation developed in tidal currents. In: Thorne J. (ed.), *Vegetation and Erosion Processes and Environments*, John Wiley, London, pp. 113–124.
- Pitman S. 1993. *Vegetation and over-marsh sediment deposition in a backbarrier saltmarsh at Stiffkey, North Norfolk, UK*. MSc thesis, University of Wolverhampton.
- Plater A.J., Ridgway J., Appleby P.G., Berry A. and Wright M.R. 1998. Historical contaminant fluxes in the Tees estuary, UK: Geochemical, magnetic and radionuclide evidence. *Marine Pollution Bulletin* 37: 343–360.
- Plater A.J., Stupples P., Roberts H.M.R. and Owen C. 2002. The evidence for late Holocene foreland progradation and rapid tidal sedimentation from the barrier and marsh sediments of Romney Marsh and Dungeness: a geomorphological perspective. In: Long A., Hipkin S. and Clarke H. (eds), *Romney Marsh: Coastal and Landscape Change through the Ages*, OUSA Monograph 56, OUSA, Oxford, pp. 40–57.
- Poulos S., Papadopoulos A. and Collins M.B. 1994. Deltaic progradation in Thermaikos Bay, northern Greece, and its socio-economical implications. *Ocean and Coastal Management* 22: 229–247.
- Qian C.L. 1994. The impacts of water withdrawal from Luanhe River on the delta. *Acta Geographica Sinica* 49: 158–166 (in Chinese).

- Rabalais N.N., Turner R.E. and Scavia D. 2002. Beyond science into policy: gulf of Mexico hypoxia and the Mississippi River. *BioScience* 52: 129–142.
- Randerson P.F. 1979. A simulation model of salt marsh development and plant ecology. In: Knights B. and Phillips A.J. (eds), *Estuarine and Coastal Land Reclamation and Water Storage*, Saxon House, Fanborough, pp. 48–67.
- Ranwell D.S. 1972. *Ecology of Salt Marshes and Sand Dunes*. Chapman Hall, London.
- Ray G.C. 1996. Coastal–marine discontinuities and synergisms: implications for biodiversity conservation. *Biodiversity and Conservation* 5: 1095–1108.
- Reed D.J. and Donovan J. 1994. The character and composition of the Columbia River estuarine turbidity maximum. In: Dyer K.R. and Orth R.J. (eds), *Changes in fluxes in estuaries: implications from science to management*, Proceedings of ECSA22/ERF Symposium, 13–18 September 1992, Plymouth, pp. 445–450.
- Reid P.C., Auger C., Chaussepied M. and Burn M. 1994. Quality Status Report of the North Sea 1993. Report on Sub-Region 9. The Channel, 153 pp.
- Reiter P. 2000. From Shakespeare to Defoe: malaria in England in the Little Ice Age. *Emerging Infectious Diseases* 6: 1–11.
- Rey J.R., Kain T., Crossman R., Peterson M., Shaffer J. and Vose F. 1991. Zooplankton of impounded marshes and shallow areas of a subtropical lagoon. *Florida Scientist* 54: 191–203.
- Rhee G.Y. 1972. Competition between an alga and its aquatic bacterium phosphate. *Limnology and Oceanography* 17: 505–514.
- Rhodes E.G. 1980. Modes of Holocene coastal progradation, Gulf of Carpentaria. PhD thesis, Australian National University, Canberra.
- Richards F.J. 1934. The salt marshes of the Dovey Estuary. IV. The rates of vertical accretion, horizontal extension and scarp erosion. *Annals of Botany* 48: 225–259.
- Richardson K. and Jorgensen B.B. 1996. Eutrophication: definition, history and effects. In: Jorgensen B.B. and Richardson K. (eds), *Eutrophication in Coastal Marine Ecosystems*, Coastal and Estuarine Studies 52, AGU, Washington DC, pp. 1–20.
- Riethmüller R., Heineke M., Küh H. and Keuker-Rüdiger R. 2000. Chlorophyll a concentration as an index of sediment surface stabilization by microphytobenthos? *Continental Shelf Research* 20: 1351–1372.
- Roberts N. 1998. *The Holocene – An Environmental History*. Blackwell, Oxford. 316 pp.
- Rocha C., Galvão H. and Barbosa A. 2002. Role of transient silicon limitation in the development of cyanobacteria blooms in the Guadiana estuary, south-western Iberia. *Marine Ecology Progress Series* 228: 35–45.
- Rodriguez-Iturbe I. 2003. Hydrological dynamics and ecosystem structure. *Water Science and Technology* 47: 18–24.
- Salomons W. and Forstner U. 1984. *Metals in the Hydrocycle*. Springer-Verlag, Berlin, 349 pp.
- Sanchez J.M., San Leon D.G. and Izco J. 2001. Primary colonisation of mudflats estuaries by *Spartina maritima* (Curtis) Fernald in Northwest Spain: vegetation structure and sediment accretion. *Journal of Vegetation Science* 7: 695–702.
- Sankaranarayanan V.N. and Panampunnayil S.U. 1979. Studies on organic carbon nitrogen and phosphorus in sediments of the Cochin Backwater. *Indian Journal of Marine Science* 8: 27–30.
- Schaffner L.C. 1990. Small-scale organism distributions and patterns of species diversity: evidence for positive interactions in an estuary benthic community. *Marine Ecology Progress Series* 61: 107–117.
- Schemel L.E., Hager S.W. and Childers D. Jr. 1996. The supply and carbon content of suspended sediment from the Sacramento River to San Francisco Bay. In: Hollibaugh J.T. (ed.), *San Francisco Bay: the Ecosystem*, American Association for the Advancement of Science, San Francisco, pp. 237–260.
- Scholten M. and Rozema J. 1990. The competitive ability of *Spartina anglica* on Dutch salt marshes. In: Gray A. and Benham P. (eds), *Spartina anglica – a research review*, Institute of Terrestrial Ecology, HMSO, London, pp. 39–47.
- Schories D., Barletta Bergan A., Barletta M., Krumme U., Mehlig U. and Rademaker V. 2003. The keystone role of leaf-removing crabs in mangrove forests of North Brazil. *Wetlands Ecology and Management* 11: 243–255.
- Seralathan P., Meenakshikutty N.R., Asarafe K.V. and Padmalal D. 1993. Sediment and organic carbon distributions in the Cochin harbour area. *Indian Journal of Marine Sciences* 22: 252–255.
- Seródio J. and Catarino F. 1999. Fortnightly light and temperature variability on estuarine intertidal sediments and implications for microphytobenthos primary productivity. *Aquatic Ecology* 33: 235–241.
- Sherif M.M. and Singh V.P. 1999. Effect of climate change on sea water intrusion in coastal aquifers. *Hydrological Processes* 13: 1277–1287.
- Simenstad C.A., Small L.F. and McIntire C.D. 1990. Consumption processes and food web structure in the Columbia River Estuary. *Progress in Oceanography* 25: 271–297.
- Simenstad C.A., Reed D.J., Jay D.A., Baross J.A., Prah F.G. and Small L.W. 1994. Land-margin ecosystem research in the Columbia River estuary: an interdisciplinary approach to investigating couplings between hydrological, geochemical and ecological processes within estuarine turbidity maxima. In: Dyer K.R. and Orth R.J. (eds), *Changes in fluxes in estuaries: implications from science to management*, Proceedings of ECSA22/ERF Symposium, 13–18 September 1992, Plymouth, Olsen and Olsen, pp. 437–444.
- Sivakumar V., Thangaraj G.C., Chandran R. and Ramamoorthi K. 1983. Seasonal variations in carbon, nitrogen and phosphorus in sediments of the Vellar Estuary, Porto Novo, Mahasagar. *Bulletin of the National Institute of Oceanography* 16: 175–181.
- Sklar F. and Browder J. 1998. Coastal environmental impacts brought about by alterations to freshwater flow in the Gulf of Mexico. *Environmental Management* 22: 547–562.
- Snelgrove P.V.R. and Butman C.A. 1994. Animal-sediment relationships revisited: cause versus effect. *Oceanography Marine Biology Annual Review* 32: 111–177.
- Soberón-Chávez G. and Yáñez-Arancibia A. 1985. Ecological control of the demersal fishes: Environmental variability of the coastal zone and the influence in the natural production of fisher resources. In: Yáñez-Arancibia A. (ed.), *Recursos pesqueros potenciales de México: La pesca acompañante del*

- Camarón, Programa Universidad Alimentos, Instituto de Ciencias del Mar y Limnología, Instituto Nacional de Pesca. Editorial Universitaria, UNAM, México D.F., pp. 399–486.
- Stanley D.J. and Warne A.G. 1998. Nile Delta in its destruction phase. *Journal of Coastal Research* 14: 794–825.
- Streever B. 2001. *Saving Louisiana? The Battle for Coastal Wetlands*. University Press of Mississippi, Jackson, 189 pp.
- Stumpf R.P. 1983. The process of sedimentation on the surface of a salt marsh. *Estuarine, Coastal and Shelf Science* 17: 495–508.
- Sutherland T.F., Grant J. and Amos C.L. 1998a. The effect of carbohydrate production by the diatom *Nitzschia curvilineata* on the erodibility of sediment. *Limnology and Oceanography* 43: 65–72.
- Sutherland T.F., Amos C.L. and Grant J. 1998b. The effect of biofilms on the erodibility of sublittoral sediments of a temperate microtidal estuary. *Limnology and Oceanography* 43: 225–235.
- Sutula M., Day J.W., Cable J. and Rudnick D. 2001. Hydrological and nutrient budgets of freshwater and estuarine wetlands of the Taylor Slough in the Southern Everglades, Florida (USA). *Biogeochemistry* 56: 287–310.
- Takahashi H., Hosokawa Y., Furukawa K. and Yoshimura H. 2000. Visualisation of an oxygen deficit bottom water circulation in Osaka Bay, Japan. *Estuarine, Coastal and Shelf Science* 50: 81–84.
- Takeda S. and Kurihara Y. 1994. Preliminary study of management of red tide water by the filter feeder *Mytilus edulis galloprovincialis*. *Marine Pollution Bulletin* 28: 662–667.
- Tappin A.D. 2002. An examination of the fluxes of nitrogen and phosphorus in temperate and tropical estuaries: current estimates and uncertainties. *Estuarine, Coastal and Shelf Science* 55: 885–901.
- Thayer G.W. 1974. Identity and regulation of nutrients limiting phytoplankton production in the shallow estuaries near Beaufort, NC. *Oecologia* 14: 75–92.
- Thayer G.W., Powell A.B. and Hoss D.E. 1999. Composition of larval, juvenile and small adult fishes relative to changes in environmental conditions in Florida Bay. *Estuaries* 22: 518–533.
- Thompson J.D. 1990. *Spartina anglica*, characteristic feature, or invasive weed of coastal salt marshes? *Biologist* 37: 9–12.
- Thompson G.B. and Yeung S.K. 1982. Phosphorus and organic carbon in the sediments of a polluted subtropical estuary, and the influence of coastal reclamation. *Marine Pollution Bulletin* 13: 354–359.
- Thouvenin B., Le Hir P. and Romana L.A. 1994. Dissolved oxygen model in the Loire Estuary. In: Dyer K.R. and Orth R.J. (eds), *Changes in Fluxes in Estuaries, Implications from Science to Management*, Olsen and Olsen, Fredensborg, pp. 169–178.
- Thurrow R. 2001. In malaria war, south africa turns to pesticide long banned in the west. *Wall Street Journal*, July 26.
- Tipping E., Marker A.F.H., Butterwick C., Collett G.D., Cranwell P.A., Ingram J.K.G., Leach D.V., Lishman J.P., Pinder A.C., Rigg E. and Simon B.M. 1997. Organic carbon in the Humber Rivers. *The Science of the Total Environment* 194/195: 345–355.
- Townsend P.A. and Walsh S.J. 1998. Modeling floodplain inundation using an integrated GIS with radar and optical remote sensing. *Geomorphology* 21: 295–312.
- Trefry J.H., Nelsen T.A., Trocine R.P. and Eadie B.J. 1994. Transport of particulate organic carbon by the Mississippi River and its fate in the Gulf of Mexico. *Estuaries* 17: 839–849.
- Troccaz O., Girault F., Bertru G. and Lefeuvre J.C. 1994. Methodology for studying exchanges between salt marshes and coastal marine waters. *Wetlands Ecology and Management* 3: 37–48.
- Tupper M. and Able K.W. 2000. Movements and food habits of Striped Bass (*Morone saxatilis*) in Delaware Bay (USA) salt marshes: comparison of a restored and a reference marsh. *Marine Biology* 137: 1049–1058.
- Turner R.E. 1993. Carbon, nitrogen and phosphorus leaching rates from *Spartina alterniflora* salt marshes. *Marine Ecology Progress Series* 92: 135–140.
- Uncles R.J., Joint I. and Stephens J.A. 1998a. Transport and retention of suspended particulate matter and bacteria in the Humber-Ouse estuary, United Kingdom, and their relationship to hypoxia and anoxia. *Estuaries* 21: 597–612.
- Uncles R.J., Howland R.J.M., Easton A.E., Griffiths M.L., Harris C., King R.S., Morris A.W., Plummer D.H. and Woodward E.M.S. 1998b. Seasonal variability of dissolved nutrients in the Humber-Ouse Estuary, UK. *Marine Pollution Bulletin* 37: 234–246.
- Uncles R.J., Bloomer N.J., Frickers P.R., Griffiths M.L., Harris C., Howland R.J.M., Morris A.W., Plummer D.H. and Tappin A.D. 2000. Seasonal variability of salinity, temperature, turbidity and suspended chlorophyll in the Tweed Estuary. *The Science of the Total Environment* 251/252: 115–124.
- Uncles R.J., Stephens J.A. and Smith R.E. 2002. The dependence of estuarine turbidity on tidal intrusion length, tidal range and residence time. *Continental Shelf Research* 22: 1835–1856.
- UNDP, 1998. *Human Development Report 1998*. United Nations Development Programme, Oxford University Press, New York.
- UNEP, 1999. *Global Environment Outlook 2000*. United Nations Environment Programme, available at: <http://www.unep.org/unep/state.htm>.
- USEPA (US Environmental Protection Agency), 1998. Report on the shrimp virus peer review and assessment workshop. Developing a qualitative ecological risk assessment. Report No. US EPA/600/R-97/136, Washington, DC, 38 pp.
- Valette-Silver N.J. 1993. The use of sediment cores to reconstruct historical trends in contamination of estuarine and coastal sediments. *Estuaries* 16: 577–588.
- Victor S., Golbuu Y., Wolanski E. and Richmond R.H. 2003. Fine sediment trapping in two mangrove-fringed estuaries exposed to contrasting land use intensity in Palau, Micronesia. *Wetlands Ecology and Management* (in press).
- von Weihe K. 1979. Morphologische und ökologische Grundlagen der Vorlandsicherung durch *Puccinellia maritima*. *Helgoländer wiss. Meeresunters* 32: 239–254.
- Walbridge M.R. and Struthers J.P. 1993. Phosphorus retention in non-tidal palustrine forested wetlands of the Mid-Atlantic Region. *Wetlands* 13: 84–94.
- Wattayakorn G., Wolanski E. and Kjerfve B. 1990. Mixing, trapping and outwelling in the Klong Ngao mangrove swamp, Thailand. *Estuarine, Coastal and Shelf Science* 31: 667–688.

- Webb P. and Iskandarani M. 1998. Water insecurity and the poor: issues and research needs. ZEF Discussion Paper, Zentrum für Entwicklungsforschung, Bonn.
- West J.M. and Zedler J.B. 2000. Marsh-creek connectivity: Fish use of a tidal salt marsh in southern California. *Estuaries* 23: 699–710.
- Wheeler D.J., Simpson D.E. and Houston J.A. 1993. Dune use and management. In: Atkinson D. and Houston J. (eds), *The Sand Dunes of the Sefton Coast*, National Museums and Galleries on Merseyside, Eaton Press, UK, pp. 129–149.
- Whitfield A.K. and Wooldridge T.A. 1994. Changes in freshwater supplies to southern African estuaries: some theoretical and practical considerations. In: Dyer K.R. and Orth R.J. (eds), *Changes in Fluxes in Estuaries, Implications from Science to Management*. ECSA22/ERF Symposium, Olsen and Olsen, Fredensborg, pp. 41–50.
- WHO, 1996. *Climate Change and Human Health*. World Health Organization, Geneva.
- WHO, 2000. *Climate Change and Health: Impact and Adaptation*. http://www.who.int/environmental_information/Climate/climchange.pdf.
- Widdows J., Brinsley M.D., Bowley N. and Barrett C. 1998. A benthic annular flume for *in situ* measurement of suspension feeding/biodeposition rates and erosion potential of intertidal cohesive sediments. *Estuarine, Coastal and Shelf Science* 46: 27–38.
- Widdows J., Brinsley M.D., Salkeld P.N. and Lucas C. 2000. Influence of biota on spatial and temporal variation in sediment erodibility and material flux on a tidal flat (Westerschelde, The Netherlands). *Marine Ecology Progress Series* 194: 23–27.
- Widdows J., Huntley D.A., Leeks G.J.L. and Walling D.E. 2001. The intertidal zone. In: Huntley D., Leeks G. and Walling D. (eds), *Land-ocean Interaction: Measuring and Modelling Fluxes from River Basins to Coastal Seas*, IWA Publishing, London, pp. 184–208.
- Williams T.B., Bubbs J.M. and Lester J.N. 1994. Metal accumulation within salt marsh environments: a review. *Marine Pollution Bulletin* 28(5): 277–290.
- Wohlenberg E. 1933. Das Andelpolster und die Entstehung einer charakteristischen abrasionsform im Wattenmeer. *Wiss. Meeresuntersuchungen N.F. Helgoland Band XIX*: 1–11.
- Wolanski E. and Chappell J. 1996. The response of Australian tropical estuaries to a sea level rise. *Journal of Marine Systems* 7: 267–279.
- Wolanski E. and Spagnol S. 2000. Environmental degradation by mud in tropical estuaries. *Regional Environmental Change* 1: 152–162.
- Wolanski E. and Duke N. 2002. Mud threat to the Great Barrier Reef of Australia. In: Healy T., Wang Y. and Healy J.-A. (eds), *Muddy Coasts of the World: Processes, Deposits and Function*, Elsevier, Amsterdam, pp. 533–542.
- Wolanski E. and Spagnol S. 2003. Dynamics of the turbidity maximum in King Sound, tropical Western Australia. *Estuarine, Coastal and Shelf Science* 56: 877–890.
- Wolanski E., Jones M. and Bunt J.S. 1980. Hydrodynamics of a tidal creek – mangrove swamp system. *Australian Journal of Marine and Freshwater Research* 31: 431–450.
- Wolanski E., Huan N.N., Dao L.T., Nhan N.H. and Thuy N.N. 1996. Fine sediment dynamics in the Mekong River estuary, Vietnam. *Estuarine, Coastal and Shelf Science* 43: 565–582.
- Wolanski E., Nhan N.H. and Spagnol S. 1997. Sediment dynamics during low flow conditions in the Mekong River estuary, Vietnam. *Journal of Coastal Research* 13: 1241–1251.
- Wolanski E., Spagnol S. and Ayukai T. 1998a. Field and model studies of the fate of particulate carbon in mangrove-fringed Hinchinbrook Channel. Australia, *Mangroves and Salt Marshes* 2: 205–221.
- Wolanski E., Gibbs R., Spagnol S., King B. and Brunskill G. 1998b. Inorganic sediment budget in the mangrove-fringed Fly River estuary, Papua New Guinea. *Mangroves and Salt Marshes* 2: 85–98.
- Wolanski E., Moore K., Spagnol S., D'Adamo N. and Pattiaratchi C. 2001. Rapid, human-induced siltation of the macro-tidal Ord River Estuary, Western Australia. *Estuarine, Coastal and Shelf Science* 53: 717–732.
- Wolanski E., Richmond R., McCook L. and Sweatman H. 2003a. Mud, marine snow and coral reefs. *American Scientist* 91: 44–51.
- Wolanski E., Richmond R.H., Davis G. and Bonito V. 2003b. Water and fine sediment dynamics in transient river plumes in a small, reef-fringed bay, Guam. *Estuarine, Coastal and Shelf Science* 56: 1029–1043.
- Wolff W., Van Eeden M.J. and Lammens E. 1979. Primary production and import of particulate organic matter on a salt marsh in the Netherlands. *Netherlands Journal of Sea Research* 13: 242–255.
- Xiqing C. 1998. Changjiang (Yangtze) River delta, China. *Journal of Coastal Research* 14: 838–858.
- Yang S.L., Ding P.X. and Chen S.L. 2001. Changes in progradation rate of the tidal flats at the mouth of the Changjiang (Yangtze) River, China. *Geomorphology* 38: 167–180.
- Zaitsev Y.P. 1992. Recent changes in the trophic structure of the Black Sea. *Fisheries Oceanography* 1: 180–189.
- Zalewski M. 2000. Ecohydrology – The scientific background to use ecosystem properties as management tools towards sustainability of water resources. *Ecological Engineering* 16: 1–8.
- Zalewski M. 2002. Ecohydrology – the use of ecological and hydrological processes for sustainable management of water resources. *Hydrological Sciences Bulletin* 47: 823–832.