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EUTROPHICATION IN COASTAL AQUATIC SYSTEMS AND THE UTILITY OF NUTRIENT BUDGETS IN THEIR MANAGEMENT

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RESUMO

A Eutrofização em Sistemas Aquàticos Litorais e a Utilização do Orçamento dos Nutrientes em seu Gestão

A adição dos nutrientes aos sistemas aquàticos litorais é um processo natural que ocorre por causa da desagregação geològica e occorrências do alto oceano. Em dècadas recentes, êste processo tem sido apressado pela adição de nutrientes generadas pr actividades do homem, pondo em perigo os tais sistemas via eutrofização. Vastas quantidades de nitrogènio e fósforo que são nutrientes importantes de desenvolvimento constituem a razão mais importante. Varias definiçoes do termo *eutrofização* sugerem as simptomas biològicas que nòs temos que controlar, e as respostas dos directores para prevenir ou reduzir êste desenvolvimento precisam minimizar a entrada dos nutrientes tais como nitrogènio e fósforo. O orçamento dêstes nutrientes é capaz de determinar as respostas dos Directores e tambem a importância relativa de vàrias fontes. Estas anàlizes utilizadas com a informação do orçamento de carbònio, sal e agua são capazes de indicar a productividade primària deste sistema e consequentemente a capacidade da carga dos sistemas econòmicos tais como aquicultura. Sendo sujeitas a incertezas, os orçamento de nutrientes deve se utilizado para promover a discussão e não como bases para a formulação de políticas.

Palavra-chaves: eutrofização, sistemas aquàticos litorais, orçamento de nutrientes, nitrogènio, fósforo, gestão

ABSTRACT

Eutrophication in Coastal Aquatic Systems and the Utility of Nutrient Budgets in their Management

The addition of nutrients to coastal aquatic systems is a natural process that occurs due to geological weathering and ocean upwelling events. This process has been hastened in recent decades through nutrient input from several human activities, posing a threat to such systems from eutrophication. The major cause is the delivery of high quantities of nitrogen and phosphorus, important plant growth promoting nutrients, to these systems. Various definitions of the term 'eutrophication' imply the biological symptoms to be monitored for, and management responses to prevent/mitigate this development take into account approaches for reducing nutrient input, especially that of nitrogen and phosphorus. Nitrogen and phosphorus budget analyses can be used to determine the effectiveness of these management responses, and also the relative importance of different input sources. These analyses, used along with carbon, salt, and water budget data, can also indicate the system's primary productivity, and therefore its carrying capacity for economic ventures such as aquaculture. Though being subject to uncertainties, budgets should be used to promote discussion and not as bases for policy formulation.

Keywords: eutrophication, coastal aquatic systems, nutrient budgets, nitrogen, phosphorus, management

CONTENTS

SUMMARY

The addition of nutrients to coastal aquatic systems is a natural process that occurs from geological weathering and inputs from ocean upwelling events. In recent decades, this process has been hastened by various human activities, through the elevated nutrient loading especially of nitrogen (N) and phosphorus (P) compounds, resulting in eutrophication.

Eutrophication has been variously defined by several workers, and has implications in the monitoring of systems. It has been understood as the following:
Eutrophication is an active change in the metabolic resources of a system that results through its fertilisation with previously scarce nutrients (N and P) that causes an increase in organic content. This process can also occur through the delivery of decomposable organic matter, though less commonly. A deviation is caused in the structure, function, and stability of the inhabitant organisms. Phanerogams are replaced by short-lived nuisance macroalgae or phytoplankton, with concurrently higher chlorophyll a concentrations. Shifts also occur in the algal and benthic communities. Turbidity and low dissolved oxygen content are other symptoms. These changes are undesirable as they decrease the human benefits that are gained from such ecosystems. The effects of this condition are now on the rise, and have large-scale implications.

N loads are critical in influencing the extent of eutrophy in an ecosystem as it is generally the limiting nutrient in coastal waters. On the other hand, in some systems or seasons, P may be the limiting nutrient. With anthropogenic nutrient loading, increases in organic matter content can cause the sediments to become more reducing in nature,

thus releasing higher quantities than usual of P into the water column. This can increase the likelihood of N limitation of the system with increased nutrient loading.

Global warming can intensify the effects of eutrophication: elevated temperatures, associated with climate change, reduce the capacity of the water to hold dissolved oxygen, and increase the diversity of species that occupy coastal waters.

Bricker et al (1999, 2007) have classified estuarine ecosystems based on primary symptoms (chlorophyll a and macroalgal abundance) and secondary symptoms (low dissolved oxygen, loss of phanerogams, and nuisance/toxic algal blooms), and eutrophic conditions rated from low to high based on scores assigned to the primary and secondary symptoms. Rabalais (2002) has described the changes that occur in marine ecosystems, where the symptoms are similar as those in estuarine systems, but does not occur by degrees, but rather is stepwise with sudden shifts.

Nutrient loading of N and P has several drivers such as the high use of synthetic fertiliser, atmospheric deposition, agriculture (including wastes and runoff), wastewater, and mariculture. The time that elapses between the pressures on the system and restoration may cause changes in baselines that make the system deviate from simply reverting from the changes encountered during the eutrophication phase. The presumption that management can return the system to its "pristine" state should be replaced by targets that securely perpetuate key ecosystem functions, and hence the steady supply of benefits to society. Management responses that take these considerations into account can be evaluated by budgeting exercises, a useful framework in this regard.

Management strategies have to take into account a monitoring plan. Efforts have to be put into reducing N and P input from point and nonpoint sources such as agroecosystems, animal production, fossil fuel consumption, urban and suburban sources, and wastewaters. Natural sinks (wetlands) should be enhanced. Mariculture, which has been encouraged to supplement declining wild fisheries, is another source whose waste can be controlled through ecological engineering methods, multitrophic aquaculture, and biofiltration utilising plants.

Budgeting exercises stimulate thought about the fluxes and drivers of the state of a system and the relative importance of the various nutrient sources and sinks in the system. They can be used to determine the efficiency of management plans and find use as environmental indicators of nutrient use efficiency, and are therefore useful in policy and regulation. Nutrient budgets, along with salt, water, and carbon budgets, also finds use in aquaculture management.

Nutrient budgets are simple and flexible, but are subject to uncertainty from various sources. To minimise inconsistencies and uncertainties when analysing and interpreting nutrient budgets, it is advisable to observe standard data conventions and guidelines, to utilise established datasets of known quality (where available), and to employ consistent methodology for evaluating budgets and their components.

The “snapshot” analysis that is often used in budget studies may not have much value in the real world as they can present a fixed view of the situation as opposed to average behaviour or trends relevant to management, and so should be reappraised over several years. They should not be viewed as final statements on which policies can be made, but should be used to promote discussion, bringing assumptions and uncertainties to the forefront.

CHAPTER 1: INTRODUCTION

This thesis is in partial fulfilment of the academic requirements of the Erasmus Mundus Master in Water and Coastal Management. The topic chosen is appropriate because it describes several interpretations of the term 'eutrophication', and the symptoms and conditions that occur in this process. It also reviews the use of nutrient budgets in evaluating the success of strategies for the management of coastal aquatic systems to mitigate and/or prevent the occurrence and or effects of eutrophication.

The addition of nutrients to aquatic systems occurs naturally from geological weathering and due to inputs from ocean upwelling. However, in recent decades, a large growth of populations has occurred and related nutrient sources, such as agriculture, wastewater treatment plants, urban runoff, fossil fuel utilisation (atmospheric deposition), have increased nutrient inputs (of especially nitrogen and phosphorus) to these systems to several times their natural levels, hastening eutrophication (Bricker et al 1999, 2007). Nutrient additions are a threat to biota, as well as to the human benefits obtained from these resources like aesthetics, fishing opportunities and success, tourism, and real estate value (Bricker et al 2007).

Besides nitrogen (N) and phosphorus (P), other nutrients such as carbon and silica may affect the onset of symptoms, but their role is not well understood. Apart from increased nutrient loads, other human and natural influences may affect the expression of eutrophic symptoms. These include engineered water flow, development, dedging, overfishing and disease. Climate change may also have a role in the development of eutrophication symptoms. (Bricker et al 1999, 2007). The term 'eutrophication' has been variously defined, and these definitions list the symptoms and conditions to be monitored for in susceptible systems.

A precise quantitative framework of N and P and other nutrients in a system can be provided by nutrient budgets. Budgets are mass balance predictions of variables such as N and P within defined geographic areas and fixed durations of time (Oenema et al 2003, Artioli et al 2008). They are based on the principle that matter can neither be created nor destroyed. Budgets utilise simple accounting criteria and have moderate data requirements (Öborn et al 2003, Swaney et al 2008). From such budget calculations, information is obtained with regard to the rate of delivery and removal of material (N or P) to/from the system under study, and in this way it illustrates how material mass changes within a system (Öborn et al 2003, Artioli et al 2008; Swaney et al 2008).

Nutrient budgets present a wide perspective from which the importance of various processes that occur within a system can be assessed more effectively (Nixon 1981). It is a useful tool in the analysis of eutrophication as it allows the determination of the various external sources of nutrient input to coastal systems (which are the main cause) and their relative importance in terms of content, and also the system’s internal biogeochemistry and water exchange (Artioli et al 2008).

Objectives

The aims of this thesis are

1. To gain an understanding of the occurrence of eutrophication.
2. To understand the role of nutrients, especially N and P, in this process.
3. To highlight the utility of nutrient budgets as a tool in the management of sensitive or eutrophied coastal aquatic systems, and the measures that can be used towards prevention or mitigation of nutrient enrichment.

CHAPTER 2: EUTROPHICATION

2.1 Eutrophication – definitions and effects

The word ‘eutrophication’ has its origins in the Greek language, where ‘eu’ signifies ‘well’, and ‘trope’ means nourishment (Ærtebjerg et al 2003, Andersen et al 2006). In recent years, taking a more modern standpoint, eutrophication has come to be associated with the traits of ecosystem sensitivity. There is no single established threshold for eutrophication in terms of primary production; the threshold varies with variations in abiotic factors, including climate (Cognetti, 2001). Several workers as described below have variously defined eutrophication:

Nixon (1995) characterised eutrophication as being an increase in the rate of supply of organic matter to an ecosystem, and emphasised that it is an active process of the change in metabolic resources on the whole and not of a trophic state. Nixon (2009) does not assume a stoichiometric equivalence between the addition of nutrients to an ecosystem and the addition of carbon to the same. Nixon (1995) proposed a trophic scheme for estuarine and coastal ocean ecosystems, where the organic carbon may be autochthonous or allochthonous:

Trophic Scheme	Organic Carbon Supply g C m^{-2} y^{-1}
Oligotrophic	< 100
Mesotrophic	100 – 300
Eutrophic	301 – 500
Hypertrophic	> 500

In some cases, total system production might not increase with higher nutrient input, but rooted macrophytes may be replaced by macroalgae or phytoplankton, thus changing the types and relative abundance of the primary producers (Nixon 1995). Differences in internal N processing can have an impact on the effects of N enrichment in estuaries. Biogenic silica (BSi) enrichment in lake sediments is a sensitive proxy for P enrichment because BSi production by diatoms integrates the use of silica on an annual timescale, silica is recycled slowly relative to P, and sedimented BSi is focussed into depositional zones (Smith et al 2006).

According to Howarth et al (2000), the fertilisation of lakes, rivers, or coastal waters with previously scarce nutrients, such as N or P, results in an excessive increase in the production of organic matter. It subsequently causes changes in nutrients, light, and oxygen, thereby favouring some species over others, and causing shifts in the structure of phytoplankton, zooplankton, and benthic communities. Ærtebjerg et al (2003) concur in their interpretation of eutrophication as intensified inputs of nutrients and organic matter leading to changes in primary production, biological structure and turnover and resulting in a higher trophic state. The causative factors are: elevated inputs of nutrients from land, atmosphere or adjacent seas, elevated winter DIN- and DIP concentrations, and increased winter N/P-ratios compared to the Redfield Ratio. The effects directly attributed to eutrophication are increased primary production, elevated levels of biomass and chlorophyll a concentrations, shift in species composition of phytoplankton, and shift from long lived macroalgae to short lived nuisance species. The secondary or indirect effects include increased or lowered oxygen concentrations, and changes in species composition and biomass of zoobenthos. Low oxygen concentrations in the bottom water (oxygen depletion, hypoxia) can further affect the fish, benthic invertebrates and plants. Total oxygen depletion (anoxia) can result in the release of hydrogen sulphide from the sediment, causing extensive death of organisms associated with the sea floor. As only a few species can survive these extreme conditions, and as it takes time for plants and animals to recolonise damaged areas, eutrophication can result in impoverished biological communities and impaired conditions.

Andersen et al (2006) have put forward the definition of eutrophication as 'the enrichment of water by nutrients, especially nitrogen and/or phosphorus and organic matter, causing an increased growth of algae and higher forms of plant life to produce an unacceptable deviation in structure, function and stability of organisms present in the water and to the quality of water concerned, compared to reference conditions'. And as highlighted by other workers, it is anthropogenic eutrophication that is of interest. An indicator often used for assessment of eutrophication and as a proxy for primary productivity, nutrient status or phytoplankton biomass is chlorophyll a (Chl a). Andersen et al (2006) recommend some caution when using this indicator, and the information inherent in Chl a measurements should be interpreted as what it is: a Chl a concentration and nothing more.

As stated by Bricker et al (1999, 2007), eutrophication is the process in which the addition of nutrients, especially N and P, stimulates the growth of algae This otherwise natural process has been greatly accelerated by human activities in recent decades. Increased nutrient inputs foster a variety of symptoms, starting with intense algal growth, which may lead to more serious symptoms. The first stages of water quality degradation due to eutrophication can be indicated by high abundance of algae, epiphytes, and/or macroalgae. In many estuaries, primary symptoms may lead to secondary symptoms, such as the loss of submerged aquatic vegetation, nuisance and toxic algal blooms, and low dissolved oxygen. The onset of secondary symptoms, even at moderate levels, indicates a more serious problem.

Finnveden and Potting (1999) define eutrophication as increased production based on the Redfield ratio of nutrients, where greater production is achieved when the balance of nutrients in the system approaches that of the Redfield ratio. According to Rabalais (2002), it is the increased accumulation of organic matter usually as a result of increased nitrogen and phosphorus inputs, but also states that the condition could also result from the external supply of excessive decomposable organic carbon. Where excess carbon is produced and accumulates, secondary effects of eutrophication often occur such as noxious algal blooms (which may be toxic), decreased water clarity, and low dissolved oxygen. The ultimate symptom is the loss or degradation of habitat with repercussions on marine biodiversity, and changes in ecosystem structure and function such as cycling of elements and processing of pollutants. Over the last two decades, it has become increasingly apparent that the effects of eutrophication are not minor and localised, but have large-scale implications and are spreading rapidly.

The US National Estuarine Eutrophication Assessment study (ASSETS http://www.eutro.org/) defined the term as a natural process whereby the productivity of an aquatic system increases, in terms of organic matter, as an outcome of high nutrient inputs. These inputs occur naturally, but have been enhanced by human-related activities. The resulting impacts are interrelated, and include nuisance and toxic algal blooms, lowered dissolved oxygen, and loss of submerged aquatic vegetation and benthic fauna. They are seen as having negative consequences on water quality, ecosystem health, and human uses.

An alternate definition provided by ASSETS is that of a process that encourages algal growth. Human activities have hastened nutrient inputs, causing excessive growth of algae and resulting in degraded water quality, and thereby hindering human use of estuarine resources.

Ferreira et al (2009) have described eutrophication as a process hastened by the enrichment of water with nutrients, especially N and/or P. this leads to situations of increased growth, primary production, and biomass of algae. Changes occur in the balance of organisms and the water quality is degraded. The outcomes of eutrophication are not desirable if they degrade ecosystem health to a considerable extent and/or the provision of goods and services in a sustainable manner.

Elliott and de Jonge (2002) point out that hypertrophication, distinct from eutrophication, is of particular importance and is evident in estuaries that have high levels of nutrients. In hypertrophic systems, excess algal growth is restricted due to light limiting conditions such as high turbidity. In coastal systems that harbour dense populations of bivalves, the turbidity related to algal density may be controlled by the cropping of the phytoplankton, as grazing by the bivalves is a major factor in controlling their growth (Gerritsen et al 1994, Newell 2004).

At the opposite end of the trophic scale, an oligotrophic environment can be defined by a low nutrient change of a fraction of C per litre per day (<100 g C m^{-2} y^{-1}), and by low total concentration of nutrients (Nixon 1995, Cavicchioli et al 2003). Low primary production (≤ 5 µg l^{-1}) as mentioned above can also occur due to other reasons (Bricker et al 1999, 2003).

2.2. Nitrogen, phosphorus, and eutrophication of coastal systems

N loads are crucial in influencing the extent to which a system becomes eutrophied, as N is generally the limiting nutrient in coastal waters (Bricker et al 2007). It is the most prevalent limiting nutrient in temperate marine systems (Finnveden and Potting 1999, Howarth et al 2000, Howarth 2008, Wolanski et al, 2008, Nixon and Fulweiler, 2009). Unlike N, P that is deposited into marine systems and accumulated into marine sediments is often annually remineralised from the sediments and returned to the water column (Conley 2000). Estuarine systems also receive oceanic waters, apart from that from terrestrial sources, and have a low overall N concentration because of the denitrification that occurs on the continental shelves and the low numbers of N fixers in estuarine waters. This situation is compounded by their slow growth rates and grazing mortality by zooplankton and benthic organisms (Correll 1998, Howarth and Marino 2006).

On the contrary, in some systems or seasons, P may be the limiting nutrient (Bricker et al, 2007). P limitation may be more common in freshwater systems (Finnveden and Potting 1999, Rabalais 2002, Wolanski et al, 2008); and in tropical coastal systems that have carbonate sediments, that can strongly bind P through the interaction with Fe, thereby leaving it largely unavailable to organisms (Conley 2000, Howarth et al 2000, Howarth and Marino 2006, Nixon and Fulweiler, 2009) and the high degree of N fixation in the benthos by cyanobacterial mats, epiphytes, and symbionts (Howarth and Marino 2006). As anthropogenic nutrient enrichment of tropical waters increases, the rate at which sediments adsorb P decreases, making a greater proportion of it biologically available and this may make the system N limited (Howarth et al 2000, Howarth and Marino 2006). The storage of P in estuarine sediments is generally lower than in freshwater sediments because of higher sulphate concentrations that lead to higher rates of sulphate reduction, and consequently greater sequestration of iron by sulphides (Correll 1998, Howarth and Marino 2006). P is adsorbed to a smaller extent on iron sulphides than on other iron compounds, though the importance of this process varies among estuaries. With increased eutrophication, there is usually more P released into the water column since more organic carbon leads to more reducing sediments, thereby increasing the likelihood of N limitation of the system with increased nutrient loading (Howarth and Marino 2006).

For an aquatic system, three main factors establish whether N or P will be the limiting nutrient:

(1) The N:P ratio in external nutrient inputs

(2) The preferential of N or P from the photic zone as a outcome of biogeochemical processes such as denitrification, sedimentation of N in zooplankton faecal pellets, or adsorption of P into sediments

(3) The degree to which any deficiency in N relative to P is made up through biological N fixation

As a result of all these factors, estuaries and coasts are more likely to be N limited than are freshwater systems (Howarth 1988).

As stated by Hecky and Kilham (1988), the concomitant limitation by multiple nutrients has not been demonstrated for any unialgal culture in a chemostat. This phenomenon is not one that is expected because one macronutrient cannot substitute another in its biochemical functions. However, diverse algal species cultured in chemostats can be limited by multiple nutrients due to differences among species for the optimum nutrient ratios for growth. Nutrient levels below optimum lead to limitations, for which reason ratios in nutrient loads can have a strong selective effect on the species in the phytoplankton communities and can affect the biomass yield.

Along the estuarine to marine continuum, multiple nutrient limitations can occur among N, P, and silicon (Si) along the salinity gradient as well as by season (Howarth et al 2000, Rabalais 2002, Wolanski et al 2008). Increased N loads cannot be analysed without taking into consideration other nutrients that are essential for plant growth, namely P and Si, and micronutrients such as iron (Fe). Elevated levels of nutrients along with altered nutrient ratios cause several complex changes in aquatic ecosystems (Rabalais 2002).

Relative to the greater flux of N and P to the coast, which are generally associated with human activity (urbanisation and agriculture), loads of dissolved silica (Si), primarily associated with the weathering of silica-bearing minerals have remained the same or have lessened, such that the proportions of Si:N or Si:P in river effluents have decreased over time while the N:P ratio increased. The dissolved Si:N ratio is inversely correlated to indices of landscape development such as population or agricultural intensity. When the Si:N ratio approaches the Redfield ratio of 1:1, shifts in the dominant species of phytoplankton communities and harmful algal blooms may occur. A significant deviation from the Redfield ratio of Si:N:P of 16:16:1 indicates a growth-limiting deficiency of any one of these elements. It was proposed that for the same quantity of N, as the N:P, Si:P, and Si:N ratios simultaneously approach the Redfield ratio, phytoplankton production becomes less dependent on any single growth limiting nutrient and increases (Finnveden and Potting 1999, Rabalais 2002). As diatom production increases upon nutrient enrichment, Si availability decreases. Si becomes further unavailable when it is trapped in bottom sediments as diatoms die and sink. A decline in available Si can limit growth of diatoms or cause a shift from heavily silicified to less silicified types of diatoms. When waters are N limited, the algal community is dominated by diatoms, which tend to sink to the bottom, activating decomposition processes that consume dissolved oxygen creating hypoxia. In contrast, when P limits the primary production, the phytoplankton community is dominated by smaller or lighter algal species and comparatively little sinks to the bottom (Howarth et al 2000). Macrophytes such as *Ulva* and its epiphytes have the capacity to take up and immobilise available N and P, slowing their recycling. These conditions seem to enhance the release of soluble reactive P from the anoxic sediment leading to an imbalance in recycling between N and P (Varioli et al 1996). Increased nutrient inputs may increase the macroalgal biomass to the point where grazers cannot control them (Finnveden and Potting 1999, Bricker et al 2007, McQuatters-Gollop et al 2009, Nixon and Fulweiler, 2009). These blooms are capable of lasting for months at a time and shade light dependent submerged aquatic vegetation, causing the deposition of organic matter in the benthic zone. Upon the death of the organic matter, dissolved oxygen is consumed and N and P are also released which can then be reused to fix more organic matter. This recycling may occur several times before being flushed out of the system (Nixon and Fulweiler 2009).

Threshold effects occur in marine systems when nutrient loads exceed their capacity for assimilation, and degradation of water quality occurs with adverse effects on the components of the system and on ecosystem functioning. Nutrient inputs initially result in a series of fisheries yields that increases to a maximum as nutrient load increases, following which a decline in fisheries occurs as seasonal hypoxia and permanent anoxia become established features of the system. Eutrophication of surface waters accompanied by oxygen deficiency in bottom waters can lead to change in the dominance of fish varieties from demersals to pelagics. The accumulated loads of organic matter and the internal loads of inorganic and organic nutrients in the sediments below eutrophic waters maintain the conditions of eutrophication since they continue to be processed by normal geochemical processes in the sediment. A decrease in the nutrient loads to estuarine and coastal systems does not result in an immediate shift in eutrophic conditions, partially because of the continued remineralisation of labile carbon and regeneration of nutrients (Rabalais 2002).

For many systems, while there is some nutrient input from natural sources, most input is now human-related, from concentrated localised sources like wastewater treatment plants, or diffuse sources such as urban runoff, agriculture and atmospheric deposition. An understanding of the volumes of present loads and that expected in the future would provide greater insight into the application of management measures and their being successful (Bricker et al 2007).

The occurrence of hypoxia and/or anoxia is the best understood and most severe impact of eutrophication, and is the link between N, or sometimes P, inputs and the increased production of organic matter that results in low dissolved oxygen them (Nixon and Fulweiler, 2009).

2.3 Hypoxia and anoxia

A common indication of eutrophication is hypoxia (typically defined as dissolved oxygen concentration DO < 2 mg l^{-1}, Wolanski et al 2008) and anoxia (DO = 0). When the DO is less than a critical value (typically 2mg l^{-1}), mobile animals such as demersal fish, crabs and shrimp migrate away from the area. Resident animals die when the DO < 1 mg l^{-1} (Wolanski et al, 2008).

The displacement of pelagic organisms and the loss of particular demersal and benthic organisms are some evident effects of hypoxia/anoxia. Hypoxia also affects optimal growth rates and reproductive capacity (Wolanski et al 2008). As oxygen levels fall from saturation through optimal levels, towards exhaustion, behavioural and physiological damage is caused in the animals residing in the water column, in the sediments or attached to hard substrates. With the drop of oxygen levels from 0.5 mg L^{-1} towards 0 mg L^{-1}, the decrease in benthic diversity, abundance and biomass becomes reasonably linear (Rabalais 2002, Wolanski et al 2008).

Aerobic bacteria utilise oxygen when decomposing the excess organic matter that sinks to the seabed from the upper water column. An overall loss of oxygen will occur in the lower water column if the rate of oxygen consumption is greater than that of its diffusion to the bottom waters from the surface (Rabalais 2002, Wolanski et al 2008). With increasing eutrophication, there is a corresponding increase in the concentrations of organic carbon and nitrogen, microbial biomass, microbial decomposition potential of substrates, and community oxygen consumption, but this does not occur linearly. A redox potential discontinuity layer shifts upward towards the sediment-water interface, sulphate respiration takes over oxygen respiration, H_2S is generated from the sediments, and oxygen does not penetrate as deeply into the sediments from the bioturbation potential of the macrofauna as their numbers are lowered from sulphide toxicity or a lack of sufficient oxygen. The sediment becomes more susceptible to resuspension and adds to the turbidity of the overlying waters, and this in turn reduces the potential for the growth of the photosynthetic microphytobenthos and generation of oxygen into the lower water column (Rabalais 2002).

As overlying waters become anoxic, many of the microbially mediated processes in the surface sediments get altered, often with negative feedback that leads to continued deterioration of water quality. The nitrification/denitrification cycle of estuarine and continental shelf sediments, which returns N_2 to the atmosphere is a mitigating mechanism to excess reactive N, but the nitrification portion of this cycle is disrupted by the lowered availability of oxygen in overlying waters (Rabalais 2002). With the shift in redox potential in the sediments with decreasing oxygen concentration, high NH_4^+ concentrations usually arise. The anoxic conditions inhibit nitrification and the NH_4^+ released from the sediments cannot be nitrified to NO_3^-. Release of PO_4^{-3} from the benthos into the overlying water is also magnified under low dissolved oxygen concentrations (Correll 1998, Heijs et al 2000, Pinckney et al 2001, Rabalais

2002). These inorganic nutrients become available to encourage further phytoplankton production in the overlying water. The degree to which these nutrients are circulated upward through the water column and across strong pycnoclines is not known (Rabalais 2002).

2.4. Impacts of climate on coastal eutrophication

Certain factors associated with climate change are expected to have the greatest impacts on coastal eutrophication (Bricker et al 1999, 2007):

Elevated temperatures may lead to an increase in the range of species that occupy coastal waters, as well as an increase in the number of macrophytes and their associated growth period. Hypoxia in coastal waters would be aggravated due to the lower capacity of warmer waters to hold dissolved oxygen.

Sea-level rise due to climate change may cause changes in water balance and circulation patterns in coastal systems. Sea level rise will gradually submerge coastal lands, causing greater erosion and sediment delivery to water bodies, and potentially flooding wetlands that act as sinks for nutrients.

Changes in precipitation may affect land runoff, stratification of and oxygen concentration in bottom waters, and water circulation patterns.

2.5. Assessment of eutrophication

Bricker et al (1999, 2007) used three primary symptoms to define the first stages of water quality degradation in estuaries – algal abundance (Chl a as the indicator), epiphyte abundance, and macroalgae. In many cases, the primary symptoms lead to secondary symptoms, such as the loss of submerged aquatic vegetation, nuisance and toxic algal blooms, and low dissolved oxygen. The classification of coastal systems based on these primary and/or secondary eutrophication symptoms has been carried out in the USA:

An increase in two of the primary symptoms – chlorophyll a (phytoplankton biomass) and macroalgal abundance – denotes the first possible stage of water quality deterioration associated with eutrophication. Because of the high variability of short-term measurements, N and P concentrations cannot be used as a measure of eutrophication (Bricker et al 1999, 2007).

Three secondary symptoms portray more serious consequences: low DO levels, loss of submerged aquatic vegetation, and the occurrence of nuisance/toxic algal blooms. Secondary symptoms can sometimes exist without having been the outcome of primary symptoms. This may occur when toxic algal blooms are transported into the system from an external source like the ocean (Bricker et al 1999, 2007).

The overall eutrophic conditions for the estuaries studied by Bricker et al (2007) use the categories of “low”, “moderate”, and “high”, based on scores assigned to the primary and secondary symptoms described earlier. Through this method, systems with moderate to high primary symptoms and low secondary symptoms have well - developed problems associated with elevated chlorophyll a and/or macroalgal blooms are in the early stages of eutrophication and may be on the verge of developing more serious conditions.

There are many reasons for systems with low primary symptoms and moderate to high secondary symptoms: blooms may be produced offshore and then be transported into the coastal system. Additionally some blooms have no direct relation to the nutrient conditions (Bricker et al 2007).

Alternatively, water quality in relations may have improved recently, but the response time for the reduction of secondary symptoms is longer than that for primary symptoms. The secondary symptoms still evident could be lingering conditions that might improve as nutrient concentrations continue to decrease. Secondary conditions also occur without necessarily being related to nutrient enrichment. For example, local dredging operations can cause the loss of submerged aquatic vegetation due to physical removal or disruption of habitat. In warmer climates, DO concentrations may be lower than cooler climates due to the lowered oxygen solubility at higher water temperatures (Bricker et al 2007).

Rabalais (2002) has described in marine ecosystems dominated by macroalgae (as opposed to the earlier description of the process in estuaries), a series of shifts in species occur as eutrophication increases:

In uneutrophied marine or brackishsh allow coastal waters, the dominant producers are usually perennial benthic macrophytes, such as seagrasses and other phanerogams on soft bottoms, or long-lived seaweeds on hard substrates.

In the stages of slight to medium eutrophication, increased nutrient loading favours the growth of bloom-forming phytoplankton and fast- growing, short-lived epiphytic macroalgae over slow-growing, long-lived macrophytes. Phanerogams and perennial macroalgal communities gradually decrease along with changes in the structure (species composition, coverage, or depth distribution limits) and their function (production and reproduction).

With increased nutrient loads towards hypereutrophic conditions,free-floating macroalgae, in particular 'green tide' forming taxa such as *Ulva* and *Enteromorpha* alternate with dense phytoplankton blooms in dominance and replace the perennial and slow-growing benthic macrophytes until they become extinct.

Under hypereutrophic conditions, phytoplankton constitute the dominant primary producers and benthic macrophytes disappear completely. This sequence of events does not occur by degrees, but is stepwise with sudden shifts (Rabalais 2002).

Information pertaining to the levels and trends of eutrophic symptoms and of nutrient input is necessary for the assessment of overall eutrophic conditions, establishment of baseline conditions, and tracking of the conditions of an estuary. This would be highly useful in gauging the success of management actions. The information would also be required to determine the influencing factors so that appropriate managements plans can be implemented (Bricker et al 1999).

CHAPTER 3: DRIVERS OF NUTRIENT LOADING, AND REGIME SHIFTS IN COASTAL ECOSYSTEMS

3.1. Drivers of nutrient loading

In the past few decades there has been an enormous increase in coastal eutrophication on a global scale, resulting in widespread hypoxia and anoxia, habitat degradation, alteration of food web structure, loss of biodiversity, and increased frequency, distribution, and duration of harmful algal blooms. N loading drives a substantial level of eutrophication, although pollution from P can also contribute to coastal eutrophication (Howarth 2008).

Aquaculture development has been encouraged on the basis that it can compensate for the insufficiency in food production due to decreasing wild fisheries (Primavera 2006). Near-shore aquaculture, which has been actively developed over the past few decades, introduces organic and inorganic wastes into the environment increasing the risk of eutrophication (by modifying N:P ratios) and pollution, where organic enrichment of the sediments would hasten benthic oxygen consumption. Apart from this is the possibility of introducing disease, altering the endemic biodiversity, and altering system resilience and carrying capacity (Buschmann et al 2008).

3.1.1 Drivers of N loading

N is fundamental for life and yet biologically available forms are a small percentage of the Earth's nitrogen, and for this reason primary production is limited by N in many ecosystems around the world. Before the industrial revolution, N was biologically available only through bacterial fixation and chemical reaction with oxygen during infrequent reactions associated with energy sources such as lightning and volcanic eruptions. Over the past few decades, human activities have generated reactive N at a pace that rivals the natural rate of production globally, and is identified with an "explosive increase" in the incidence of coastal marine eutrophication worldwide. Therefore, taking into account natural fixation on land and in the oceans, human activity has raised the total rate of production of reactive N on a global scale by 35 – 55% (Howarth 2008).

Human activity produces reactive N by three means:

Assistance of biological fixation associated with agriculture

Production of synthetic N fertiliser

Unintended creation of reactive N by oxidation of atmospheric N when fossil fuels are burned

(Rabalais 2002, Howarth 2008)

Among these, the greatest change during the 20th century was in the rate of production of synthetic N fertiliser. It allowed for considerable expansion of agriculture with associated decreases in hunger and malnutrition (Howarth 2008).

Reactive N compounds such as ammonium ions and ammonia have residence times of approximately a day due to quick assimilation into living organisms. The consequences of the acceleration of the nitrogen cycles by humans varies on the regional scale because most reactive N moves only over short distances and due to the variation in the quantity of reactive N produced from region to region around the world. Temperate regions, the major sites of industry and agriculture, are where most of the human production and use of reactive N occurred during the 20^{th} century, though most of the biological N fixation occurs in the tropics (Howarth 2008).

A measure of how humans have altered the N cycle is the increase in the rate of N deposition from the atmosphere since the industrial revolution. Another measure of the magnitude of change in the cycle is the increase in the flux of N in rivers to coastal areas and oceans. Climate can also have an effect on riverine N fluxes, which are higher in regions having more precipitation and freshwater discharge (Howarth 2008).

Human inputs of N have several sources that vary among the different regions of the world. On a worldwide scale, synthetic N fertiliser production is the single biggest alteration of the N cycle by humans, and in many regions and watersheds, agriculture dominates the N flux. The extent to which fertiliser is used differs by about 1000-fold between countries tend to apply fertiliser more intensively, and increasing per capita income in developing countries along with agricultural training programmes is likely to increase the use of fertiliser in these regions (Nixon 1995, Howarth 2008).

In those areas where agriculture does not predominate, the single largest input of N is atmosphere deposition of oxidised N compounds, with the N originating from the combustion of fossil fuels. The amount of N inadvertently fixed during the combustion of fossil fuels is small when compared to the rate at which synthetic fertiliser is produced globally. However this can be a significant source of N pollution in some regions. Greater urbanisation and use of motor vehicles increases the emission of N oxides to the atmosphere in large cities, most of which are located in the coastal zone. Atmospheric nutrient inputs to the coastal ocean may also arise from natural sources like deserts. This input may be episodic but periodic occurrences of high deposition may have significant effects on the phytoplankton communities. Direct deposition is most significant in large systems, or in those coastal systems with a small ratio of watershed to surface water area (Nixon 1995, Jickells 1998, Pinckney et al 2001, Bricker et al 2007, Howarth 2008). The importation of food and feeds to the region is the second largest input (Howarth 2008).

Over these large extents, the flux of riverine N to the coast is dependent on net anthropogenic N inputs. Agriculture plays a role in N pollution of surface waters from both, the direct runoff from agricultural lands and from the N in animal wastes. In the United States, most of the crop that is not exported is used as animal feed, and this is a major driver in using synthetic N fertiliser in agriculture (Howarth 2008). Increased consumption of animal protein also contributes to N pollution, and being related to per capita income, is rising in developing countries (Nixon 1995). N in animal feed is not effectively converted to N in animal protein; so much of the N is excreted in the animal waste and into the environment (Nixon 1995, Howarth 2008). Of the N contained in the animal wastes, an estimated 30% is volatilised to the atmosphere as ammonia, most of which gets deposited at the site of emission. The fate of most of the N in animal wastes is not well known, but it contributes via direct or indirect means towards water pollution. With the availability of synthetic N fertiliser, the utilisation of animal waste to fertilise soil has now been made no longer necessary (Howarth 2008). *Pfiesteria piscicida*, a dinoflagellate responsible for massive fish kills in Pamlico-Albemarle Sound (North Carolina, USA), and one that poses serious neurological human health risks, is associated with high organic loading from sewage or hog/chicken wastes. Thus there is evidence for the linkage of nutrient over-enrichment to some toxic species that kill or disable higher organisms and the formation of non-toxic but noxious blooms that can lead to other habitat impairments (Rabalais 2002).

3.1.2. Drivers of P loading

Anthropogenic sources of P may be delivered to coastal systems through highly concentrated point sources where the P is in soluble form, and diffuse sources where a greater proportion is particulate (Withers and Jarvie 2008).

According to Withers and Jarvie (2008), wastewaters from sewage treatment plants and industrial sites contribute towards the load of P in rivers. In rural catchments, septic tanks may contribute significantly to riverine P as a concentrated source if they discharge directly into streams or do not have adequate soakaway facilities (a hole dug into permeable ground that is filled with granular material and usually covered with soil, which allows collected water to soak into the ground).

Runoff from urban areas, including atmospheric deposition, urban and industrial debris, detergents, and lubricants, are other sources of riverine P. Though the concentration of

P in runoff may vary greatly, it increases with population level. Farmyard runoff is high in soluble P, especially from livestock manure (with higher concentration from animals supplemented with P-enriched feed), sheds, and paths. Drainage from agricultural lands includes soil and fertiliser manufactured from mined P. Storm events would cause soil erosion that would increase the quantity of such materials carried to rivers. Since P inputs to soils (especially as fertiliser) exceed its outputs in farm products, and unlike N, will not be lost to the atmosphere (denitrification), it can accumulate in the soil with the potential for increased runoff from the land to surface waters (Rabalais 2002, Carpenter 2005, Withers and Jarvie 2008).

Much P is adsorbed to soil particles, and so riverine sediment transport of eroded soils also represents a flux of P. the greater flux of P eroded from land or carried through wastewaters to rivers have increased the global flux of P to the oceans almost 3-fold above historic levels of ~8 million Tg P yr^{-1} to the present load of ~22 Tg yr^{-1}. The accumulation of P in the landscapes of developed countries has started to decline, but that of developing countries is increasing (Rabalais 2002).

While direct P loading to estuarine and coastal systems may not be as significant as N loading, when conditions of eutrophication are enhanced, oxygen in coastal waters can decline, and the bottom sediments become more reducing in nature. This can result in an increase in the quantity of P released to the water column than would usually occur (Howarth and Marino 2006) as has been described earlier in chapter 2, section 2.2.

3.2 Regime shifts

The increase in nutrient inputs is not the only driver of changes in algal biomass in coastal waters, and reduction in nutrient inputs does not always result in the expected outcomes. Along with the time that has elapsed between the pressure and restoration, the response of coastal systems to minimisation of anthropogenic nutrient inputs might be more complex than expected as changes in other control factors may occur at the simultaneously. The ongoing change in these baselines (reference status) that indicates that a coastal system faced with having to return to oligotrophy from eutrophy may deviate from simply reverting from the changes encountered during the earlier phase of eutrophication (Duarte et al 2009). These are very significant changes, and involve several basic factors involved in the functioning of coastal ecosystems, that, according to Andersen et al (2008) and Duarte et al (2009), they would have forced, separately and collectively, the coastal systems to move away from the reference status even in the absence of direct anthropogenic inputs, and would result in abrupt changes on several trophic levels. This is a 'regime shift' involving the resistance of the ecosystem to return to its original status with the imposed oligotrophication, due to hysteresis resulting from nonlinearities in the ecosystem. This scenario considers coastal ecosystems to flip between alternative stable states; while during the development of shifting baselines no state is necessarily stable (Duarte et al 2009).

The degradation of coastal systems around the world has promoted efforts to determine the pressures that are responsible and the identification of a reference status to which the system is supposed to return on reduction of these pressures. Progress has been achieved scientifically with the growing understanding of anthropogenic impacts on coastal systems; however, the definition of the reference status that is to be attained upon the minimisation of the direct anthropogenic pressures is still to be elucidated (Duarte et al 2009). Duarte et al (2009) suggests taking into account the possibility of shifting baselines and regime shifts in the ecosystems to evaluate attainable outcomes of restoration efforts. With increasing human pressures on coastal ecosystem, the need for analytical methods for identifying ecological thresholds and regime shifts becomes more urgent (Andersen et al 2008) and the means of identifying them without having to cross them are required (Duarte et al 2009).

Chapter 4: MANAGEMENT RESPONSES

As reviewed in the previous section, the return of an ecosystem to a past state may not be the expected outcome because of the occurrence of shifting baselines in various factors involved in the functioning of the system, and the presumption that management can restore ecosystems to “pristine” states should be replaced by targets that securely perpetuate key ecosystem functions, and hence the steady supply of beneficial services to society (Duarte et al 2009). Management responses that take the above considerations into account can be evaluated by budgeting exercises, a useful framework in this regard.

In the subsequent sections, the utility of budgets in nutrient management is addressed, as are some general approaches, involving monitoring, and technological and policy responses, for the management of nutrient loading to coastal system. This is followed by means for improving the efficiency of nutrient use in agroecosystems, an important diffuse source of nutrients to coastal systems. Technical concepts for reducing the runoff and leaching of N to coastal areas from watersheds and concepts for the removal of P from wastewaters are discussed as the removal of N or P or both along the freshwater-marine gradients is required. Management approaches follow in the last section for waste generated from mariculture, an important means of supplementing wild fish catches.

4.1 Budgets as a tool in nutrient management

Whole system nutrient budgets have been used as quantitative schemes to analyse estuarine N and P inputs, transformations, and fate. The results from budgeting exercises encourage the users to think about the fluxes and drivers of the state of a system in a quantitative manner, and not only about the apparent symptoms. They provide a quantitative comparison of the nutrient fluxes in the system and stimulate thought about the relative importance of various sources and sinks of nutrients in their biogeochemical cycling, which provide support in settling management priorities (Le Tissier et al 2006, Boynton et al 2008). By applying budgeting methods before and after the implementation of management plans, they can be used to determine whether the plans have proven effective in reducing nutrient loads. They find use as environmental indicators of nutrient use efficiency and are therefore useful in policy and regulation (Oenema et al 2003). Budgeting can also demonstrate whether interactions occur between the system being monitored and adjacent marine waters. An input of nutrients from adjoining water bodies into the system under consideration may nullify the effect of the management responses. Conversely, the N and P that might be exported from the system to the adjoining water bodies may also be determined (Boynton et al 2008).

The value of budgets as a performance indicator also depends on reference or baseline levels against which the nutrient balances can be evaluated. As has been done by Bricker et al (1999), such levels can be set by reviewing data on a national scale and by consulting local and regional experts.

Suitable management plans, after an overall assessment of the system, must take into account its eutrophic condition (low through moderate to high), the sources of nutrients to it, and the expected reductions in the loads should be guided by and also reflected in the validity of the assessment results (ASSETS http://www.eutro.org/). These plans should include measures to prevent/mitigate the effects of anthropogenic nutrient loading to coastal systems and the use of nutrient budgets to assess their effectiveness. The following sections outline the possible measures that can be utilised towards this end.

4.2 General approaches to system management

Coastal systems show great variations in their sensitivity to nutrient pollution because of differences in the size of the watersheds (reflected in the differing nutrient loads), different physical mixing regimes, and differences in ecological structure (Howarth 2000). Nutrient management efforts have halved riverine P loads to the northern Adriatic and coastal North Sea, from 14 t km^{-3} y^{-1} total P (Northern Adriatic Sea) and 32.3 t km^{-3} y^{-1} total P (coastal North Sea) to 8 t km^{-3} y^{-1} and 12.9 t km^{-3} y^{-1} respectively. This has come about mainly from the intervention on point sources, especially the ban on P in detergents and the treatment of urban wastewater. A similar trend in riverine P loads has not been observed for the Baltic Sea, despite policy initiatives and severe disruption of agriculture. This may be because of ‘memory effects’ due to nutrient saturation with release of dissolved inorganic nutrients from sediments. Such cases may require some time before reduced loads are observed (Artioli et al 2008).

Bricker et al (1999) recommend the management of a system from a watershed perspective, and that it be focussed on nutrient sources that can be controlled. They also suggest that strategies be adapted to individual watershed characteristics so that the potential for improvement is the greatest possible. These strategies should also consider overall eutrophic conditions and the human and natural factors influencing the level of expression in each estuary, such that efforts can be focussed on those estuaries with the highest susceptibility that will most profit from the control of nutrients. The natural factors to be considered include nutrient sources and watershed alterations affecting nutrient delivery such as loss of wetlands. Estuaries with low eutrophic conditions but high susceptibility should have precedence for preventive managements

4.2.1 Monitoring methods

The first step in a management plan should be an assessment of the given system, the basis for which is a monitoring plan. Another important step is the setting of a reference status against which ecological quality can be assessed. Andersen et al (2006) define a reference condition as a description of the biological elements that exist, or would exist, at very low or no disturbance from human activity. This has to be derived from scientific knowledge (Bricker et al 1999).

With the development in the definitions of eutrophication, monitoring methods should include the measurement of primary production. In shallow waters, data on area coverage by macroalgae and/or seagrasses can be used as a proxy for production (Nixon 2009). Other methods that can be used to measure increased growth include the monitoring of Chl a (McQuatters-Gollop et al 2009, Nixon 2009), Secchi depth, phytoplankton community composition, N:P ratios to assess the possible effects of imbalanced nutrient enrichment on the phytoplankton (McQuatters-Gollop et al 2009). Andersen et al (2006) suggest the use of ^{14}C to precisely determine the uptake of C into phytoplankton or that incorporated into organic matter. As this would greatly increase the expense of regular monitoring programmes, it would find use in providing information about a system when utilised after the initial symptoms of eutrophication have been observed

4.2.2 Technological responses

Technologies for treating animal wastes for the removal of nutrients and to prevent the volatile losses of ammonia from these wastes to the atmosphere are necessary on a large scale, as are control strategies to reduce fertiliser runoff from agriculture and emission of N compounds from fossil fuel combustion. Stronger regulation and incentives for compliance will all be required if these control strategies are to be better directed towards solving pollution problems of coastal systems (Howarth et al 2000). Some symptoms of eutrophication in estuaries may be partially mitigated by ecological engineering measures for nutrient reduction such as the building of treatment ponds, dredging, creating man-made channels for improved circulation by increased communication with the sea, and wetland restoration and creation (Wolanski et al 2008).

4.2.3 Policy responses

N reduction policies have had varying degrees of success. Reduction in riverine N loads is more difficult to achieve than P because N loads have relatively diffuse sources, particularly farming. N emission is more closely related to human lifestyles, especially diet and the associated significance for agriculture and livestock activities. Atmospheric transport can also deposit N originating from distant powerplant and automobile emissions directly on coastal waters. A significant proportion of N discharged by rivers flows into groundwater, and due to its long retention time, efforts in reducing N emissions may not produce an immediate reduction in loads and the system may take several years to respond to such remedial action (Artioli et al 2008).

It is important to focus on those sources of nutrients that can be controlled in the system under consideration. Also important is that the level of susceptibility, eutrophic condition, and future outlook be used to set management priorities (Bricker et al 2007). Considerable reductions in N and P may be achieved through a decrease in point source loading, especially through wastewater treatment (Carstensen et al 2006). For example, in order to combat an event of widespread anoxia in the Danish straits, an agenda was adopted recommending that the government act to reduce discharges and losses of N (by 50%) and P (by 80%) from agriculture, municipal wastewater treatment plants, and individual industrial discharges. This plan was formulated with sector-specific reduction objectives and targets. The specific objectives and targets were difficult to achieve during the original time frame and another plan for sustainable agriculture was put into action. Considerable time lags occurred between changes in agriculture practice and water quality responses, but these reductions would not have been achieved if periodic assessment for reduction assent had not been carried out and if the national monitoring programme had not remained focussed on eutrophication. Reduction in nutrient over-enrichment of coastal ecosystems will rely on the implementation of an adaptive management framework (Carstensen et al 2006).

Long-term (i.e., decadal) nutrient management strategies will also have to take into consideration climatic and hence hydrologic fluctuations and the resultant shifts in in-stream N and P processing. Thus nutrient management should be highly adaptive, taking into account short and longer-term patterns and trends, as well as keeping in mind relevant scales (Howarth et al 2000). A “one shoe fits all” appears inappropriate for eutrophication, and different targets need to be met for systems of different scales (Artioli et al 2008). It is also worth considering that while management practices for reducing N pollution can be effective for P control, the converse does not hold true. This is because N moves with surface and groundwater far more readily in dissolved forms than does P and is more subject to atmospheric transport (Howarth 2005).

4.3 Improvement of nutrient use efficiency in agroecosystems

Agroecosystems should be managed at several temporal and spatial scales to reduce the need for chronic addition of excess nutrients. Landscape-level strategies that integrate wetlands and other types of riparian buffers into agricultural landscapes are very effective in protecting sensitive natural ecosystems and are essential for ecosystem-scale nutrient management. The incorporation of strategically located plant communities that can act as nutrient sinks in managed landscapes can effectively capture particulate and soluble nutrients before they reach adjacent waterways or aquatic systems. Including cover plants in rotation with the food/cash/biofuel crop incorporates more fertiliser into the microbial biomass than is leached from the soil as compared to the continuous crop. This is due to the effect of the cover crop in increasing the soil C content in proportion to other nutrients, which has a positive effect on microbial uptake (Drinkwater and Snapp 2007).

4.4 Technical approaches for the control of nitrogen loads from watersheds

In some estuarine systems, P may be the limiting nutrient in the spring (Cowan and Boynton 1996, Correll 1998, Conley 2000), and so the greatest effects of P reduction would be observed during this season. This is also an important time of the year for the seasonal deposition of organic matter to the sediments. The reduction of P during this time would make it more limiting, resulting in decreased accumulation of organic matter. For deep estuaries that are P limited, the conditions of anoxia observed during the summer that is spurred by spring production can be alleviated by reductions in P loads (Conley 2000).

N limitations occur in some estuaries in the summer and fall (Cowan and Boynton 1996, Correll 1998, Conley 2000), and N reduction strategies would serve to lower summer Chl a concentrations and also reduce the growth of epiphytes that would shade the submerged aquatic vegetation (Conley 2000). If only P reductions were attempted such that it becomes the limiting nutrient throughout the year, the retention of excess N in estuarine systems would be lowered and N would be delivered to the adjoining N-limited coastal systems, thus only transferring the problem (Conley 2000, Rabalais 2002). Therefore a coordinated effort for managing both N and P is required as P can influence the delivery of N to the coast.

The biogeochemical cycle of Si is also altered upon excessive inputs of N and P, and dissolved silicate limitation may occur more frequently. Increases in the N and P concentrations in the water column lowers the ratio of dissolved silica, and increased diatom production leads to increased accumulation of biogenic silica in the sediments, thus lowering its content in the water column. The change in Si availability causes diatoms to be replaced by phytoplankton that do not require it for growth, which may have negative impacts on the food web (Conley et al 1993, Conley 2000).

Many technical solutions are available for the management of N to coastal systems. The following list of N sources and solutions is summarised from Howarth (2005):

1. Leaching and runoff from agricultural lands: in the United States, about 20% on average of the new N input to agricultural fields leach to surface or ground waters. Climate is important, and N losses are greater in wet years and in areas with higher rainfall. Approaches with good potential for reducing the leaching of N from agricultural fields include

- Growing perennial crops rather than annuals, as perennials conserve N in the rooting zone and greatly reduce its loss to ground water.
- Planting winter cover crops that would reduce the leaching of nitrate to groundwater in the spring, which is when most leaching occurs

- Applying N fertiliser as close as possible to the time of crop requirement, which is also an opportunity to reduce the quantity of fertiliser used. Increasing the application of fertiliser increases crop yield up to a point, but once the crop requirements are saturated, further fertilisation has no effect on production. The N that is not taken up by the crop is then available for leaching to surface and ground water.
- Artificial wetlands that intercept drainage from agricultural fields can provide a sink for N, if the landscape is such that it does not cause the fields to become flooded. Such wetlands can substantially reduce the flow of N to surface water. Buffer strips, which effective at capturing P (which is largely particle-bound), are not effective at trapping N from drainage systems unless the drain water is fully intercepted by the buffer.

2. Animal production and concentrated animal feeding operations: this is a significant sector as the consumption of animal protein is increasing in countries with high or improving economies

- Composting is a good method for reutilising wastes, however a lot of NH_3 is volatilised to the atmosphere during this process
- Biogas production from animal wastes

3. Fossil fuel sources: the emission of nitrogen oxides (NO_x) to the atmosphere from fossil fuel combustion is an important source of N, most of which is deposited onto the landscape in rain and as dry deposition. In the US, the deposition of N from fossil fuel combustion is the major input of N to most coastal rivers and bays on the eastern seafront through export from their

watersheds. NO_x emissions are critical to ozone and smog formation in the lower portions of the atmosphere, and also contribute towards acid rain

- Encouraging less driving and the use of more energy efficient vehicles
- Employing stricter emission standards to larger vehicles, especially trucks and off-road vehicles, is also significant.
- The generation of electricity from fuel cells rather than traditional combustion methods would completely cancel out NO_x emissions from that source

4. Urban and suburban sources: sewage and other urban sources (especially human wastes) are very significant to the N loads in many coastal rivers and bays. Treatment beyond the secondary type (biological oxygen demand) is necessary to remove the N from these wastes

- The separation of storm sewers from sanitary sewers allows for further treatment of each, including using artificial wetlands
- Increasing water storage capacity of the combined sewers

5. Enhancing N sinks: wetlands, ponds and riparian zones are particularly effective at trapping N that serve to sediment out particulate N as well as converting reactive biologically available N into harmless N_2 gas through denitrification. Enhancing them would increased the benefits that can be gained

- Conversion to wetlands a portion of the river basins that are a source of pollution to the rivers and the reestablishment of the flow of water through these wetlands could also have an important role in reducing N to the coast.

The cost for implementing strategies to reduce N inputs may be considerable. For example, the potential cost for reducing the levels of N from only point sources in the Long Island Sound (USA) watershed as estimated in 1998 would be about 2.5 billion US dollars (Bricker et al 1999).

4.5 Technical approaches for the removal of phosphorus from wastewaters

Many of the solutions for N reduction discussed in the previous section also apply to P, but specific technologies apply for the removal of P in wastewater. The treatment of wastewater is necessary before its discharge into the environment, and the removal of P is essential. Present treatments remove it by converting the ions in the wastewater into a solid fraction. Several methods are available for carrying out this process (de-Bashan and Bashan 2004):

1. Precipitation of P with metal salts: chemical precipitations with iron, alum, or lime are the main commercial processes currently used for P removal through adsorption and precipitation. Calcium-phosphorus precipitation is a commonly used method because of its low cost and easy handling, and may also be used as a pre-treatment of urban wastewaters before treatment by biological processes

2. Biological removal of P by the use of microorganisms in bioreactors

- Certain strains of cyanobacteria, purple non-sulphur bacteria, and *Staphylococcus* are able to remove P from wastewater when grown in batch reactors or when immobilised on cellulose or ceramic beads.

- Suspensions of microalgae immobilised in alginate beads used in combination with plant growth-promoting bacteria can remove a significant proportion of phosphate as well as ammonium ions.
- Enhanced biological P removal (EBPR) is a method that takes advantage of bacteria that have the capable of internally accumulating biopolymers during periods when the environment is unfavourable and utilise them as during carbon-rich phases. This capacity is applied to the release of orthophosphate from the wastewater and the sludge in treatment plants and its uptake into the bacterial cells. This technique is the most efficient of the biological methods, and also stimulates the growth of denitrifying P removing bacteria, thereby removing both P and N together.

3. Constructed wetlands through which wastewaters are made to flow are another method for the removal of P and its uptake into plant biomass

4. Recovery of P from the sludge of wastewater treatment plants and its use as fertiliser can be used presently only for sludge from biological nutrient removal as other commercial processes remove P along with undesirable compounds. The most promising compound for recovery from wastewater plants is struvite (ammonium phosphate hexahydrate), containing both P and N, for use as a fertiliser.

4.6 Mariculture waste management

Shellfish and finfish mariculture, carried out in the open ocean on lines, and in cages in farmed areas, areas, are the two main types carried out commercially. Sites for mariculture are chosen based on hydrodynamic conditions and infrastructure to optimise the growth rate of the cultivated species, while ensuring appropriate flushing of the farmed area and sufficient depth that prevents deposition of wastes that would reduce water and bottom quality. If the chosen site does not allow proper water circulation, submarine blowers can be used to disperse the residues or finfish cages can be suspended from single moorings (Buschmann et al 2008).

4.6.1 Ecological engineering of mariculture

The aim of ecological engineering is to prevent adverse effects on the environment from human activities or to recover from these effects. To avoid disruption from the introduction of aquaculture species into coastal areas, the comparative abundance of organisms that have differing ecosystem functions can be balanced using new technologies in this field. The rotation of farming sites would allow for their recovery with time (Buschmann et al 2008).

Fish provide an ecological service, and the farming sites could serve as nurseries as well as refuges. The coupled use of such sites as both farming areas and refuges. The coupled use of such sites as both farming areas and reserves requires a large amount of information that varies from area to area. Potential interactions of wild fish and other species (especially predators) should be considered. Another emerging method for reducing wastes is the development of artificial reefs in which a significant biomass of extracting organisms would be important. Artificial reefs also provide a habitat for diverse organisms having different ecological functions and services. Since the reefs would be placed on the bottom of the faming site, where the water quality is reduced and light limited, algal growth would be limited and because of this, biofiltration may not be increased. Further testing will be required before this method can be implemented (Buschmann et al 2008).

For the use of these approaches, regulatory and financial incentives are necessary to recognise their benefits.

4.6.2 Integrated multitrophic aquaculture (IMTA)

Integrated mariculture is bioremediation using integrated concepts to restore water quality to its natural state (Troell et al 2003). The development of open IMTA systems requires well-established knowledge of the oceanographic features of a water body used for aquaculture, which can be used for determining the impact of the IMTA layout in reducing wastes. For this purpose, coastal water circulation models are important, along with complementary models.

'Bag' techniques for aquaculture that isolate an operation from its surroundings by developing closed systems at sea, though useful, have not as yet been applied on a commercial scale due to cost and technological issues. This technique would not discharge the waste generated into the sea and would be contained within the closed systems (Buschmann et al 2008).

Land-based IMTA methods can be inclusive, adaptable to several fish/shrimp, shellfish, and seaweed culture combinations at any level of intensification (Buschmann et al 2008). Ammonia is the form of N that is released by fish and shellfish in aquaculture, and this is the preferred N source for seaweeds (Troell et al 2003). High quality edible seaweed can thus be generated from such systems that can be used as feed for the cultured fish, which would reduce the pressures on the ecologically important phanerogams. This land-based method has high economic benefits (Troell et al 2003, Buschmann et al 2008).

Land-based recirculation systems that use bacteria as biofilters have a high capacity for denitrifying and transforming wastes. This method has not yet developed at the commercial stage since bacterial transformation of wastes does not yield any product of commercial value, such as the seaweed in IMTA systems, and currently remains complex and costly (Troell et al 2003, Buschmann et al 2008).

IMTA at sea has been developed as a means of increasing the environmental and economic sustainability of marine open-water systems. Unlike land-based IMTA, these are capable of achieving high efficiency (Buschmann et al 2008). If oxygen levels remain stable, the particulate matter that settles onto the sediments could be removed from the system by cultivating scavenger organisms of commercial value, such as crabs, lobsters, and sea cucumbers (Troell et al 2003, Buschmann et al 2008).

4.6.3 Biofiltration by plants

Plant biofilters, being assimilative in nature, can greatly reduce the overall environmental impact of fish culture and stabilise the culture environment. Algae, especially seaweeds, are suitable for biofiltration as they are highly productive and have economic value (Neori et al 2004).

Because of its complexity, biofiltration poses significant financial costs. To raise its revenues, the productivity per unit of feed must be increased. The waste nutrients from the fish culture are used for the auxiliary production of new plant biomass, which can be used as feed in shellfish culture, or marketed itself for human consumption, phycocolloids, feed supplements, agrichemicals, nutraceuticals, and pharmaceuticals. *Ulva, Gracilaria*, and *Porphyra*, the most commonly used seaweeds in mariculture biofiltration, have been used for these purposes (Neori et al 2004).

The ability to remove ammonia from the effluent is a requisite of the seaweed biofilter species, as is resistance to epiphytes and small herbivores. *Ulva* is the most preferred species in this respect, but once a different species has been selected as the biofilter, proper management of the culture can lessen the epiphyte problem, such as by filtration. The main management aspects are proper harvesting frequency and water flow, changes in water level, pulse feeding (to promote growth), and a high stocking density. The daily rate of areal uptake of total N as ammonia is an economically important feature in seaweed biofilters – the uptake capacity is inversely proportional to the cost and size of the biofiltration system. The uptake efficiency is also an important point (Neori et al 2004).

In open mariculture systems, the efficiency of nutrient uptake by seaweed has been low due to the nature of the water flow, and requires further study, though saline aquaculture effluents can also be treated using this method through land-based recirculation. This technique removes P from effluents as well, though to a smaller extent than N (Neori et al 2004).

CHAPTER 5: OVERVIEW OF LOICZ BUDGETS DATABASE AND CASE STUDIES

5.1 An overview of the LOICZ budget database

The Land – Ocean Interactions in the Coastal Zone (LOICZ) has a catalogued database of sites around the world where the budgeting method has been applied. An analysis of this database has provided a profile of the number of sites over each continent (figure 5.1), with the global distribution as shown in figure 5.2.

A very small proportion of the budgeted sites were shown to have balanced production, where, through the LOICZ budgeting procedure, the system was found to be neither a source of nor a sink for nutrients. The autotrophic and heterotrophic sites are present in the ratio of 1.277, and their distribution is as shown in figures 5.3 and 5.4.

Among these sites are those that have been budgeted twice – 4 in Asia (Calaug Bay, Manila Bay, San Miguel Bay, and BangPakong Bay), and 4 in Europe (Bothnian Bay, Bothnian Sea, Baltic Proper, and S'Ena Arrubia Lagoon):

- Calaug Bay in the Philippines was budgeted in 1999 and 2000 and was found to be heterotrophic both times.
- Manila Bay, The Philippines, was found to be heterotrophic in both 1998 and 2000. More attention was given to waste loading in 2000 than in the earlier year.
- San Miguel Bay, The Philippines, was budgeted in 1999 and 2000, and was autotrophic on both occasions.

- BangPakong Estuary in Thailand was found to be heterotrophic in both the years it was budgeted, and the nutrient sources were better described in 1999 than in 1997.
- The Bothnian Bay and Bothnian Sea in Europe were both autotrophic when budgeted in 1997 and 2001.
- The Baltic Proper in Europe was autotrophic when budgeted in 1997, but found to be mildly heterotrophic upon the budgeting in 2001.
- S'Ena Arrubia Lagoon in Italy was budgeted in 1994 and 1995 and was autotrophic.

Figure 5.1: Global distribution and trophic status of budgeted sites

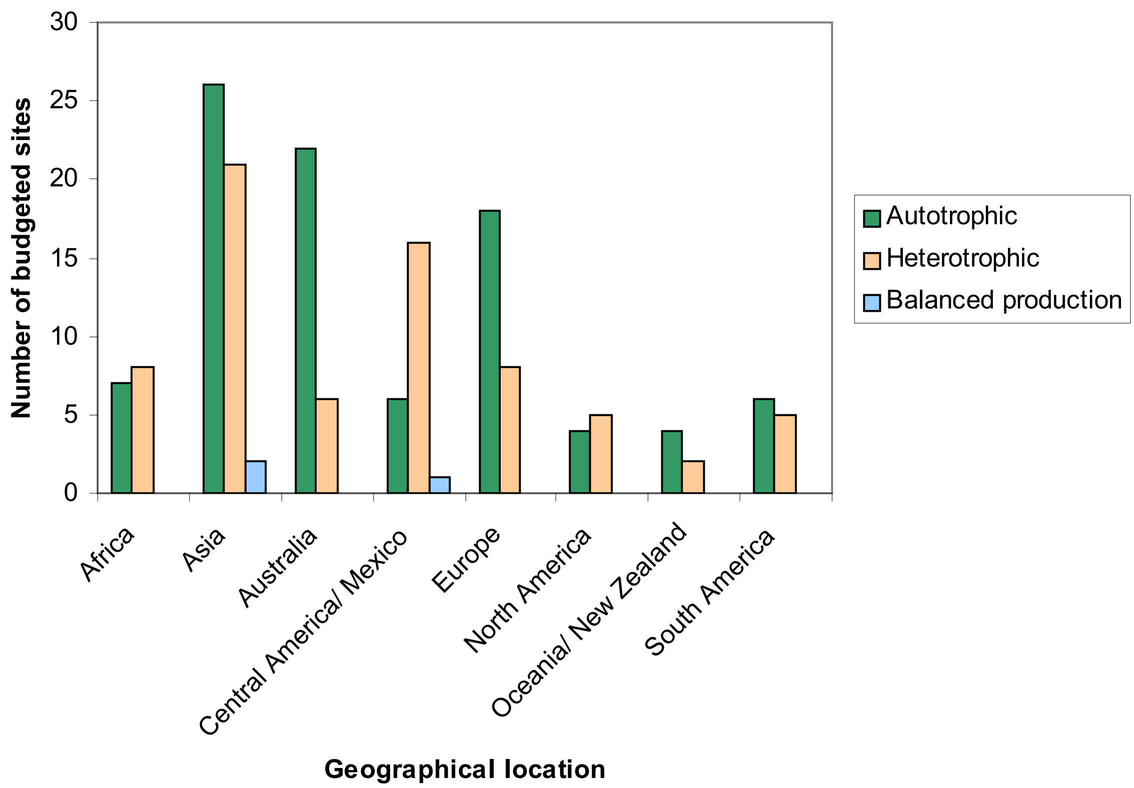


Figure 5.2: Map of budgeted sites around the world

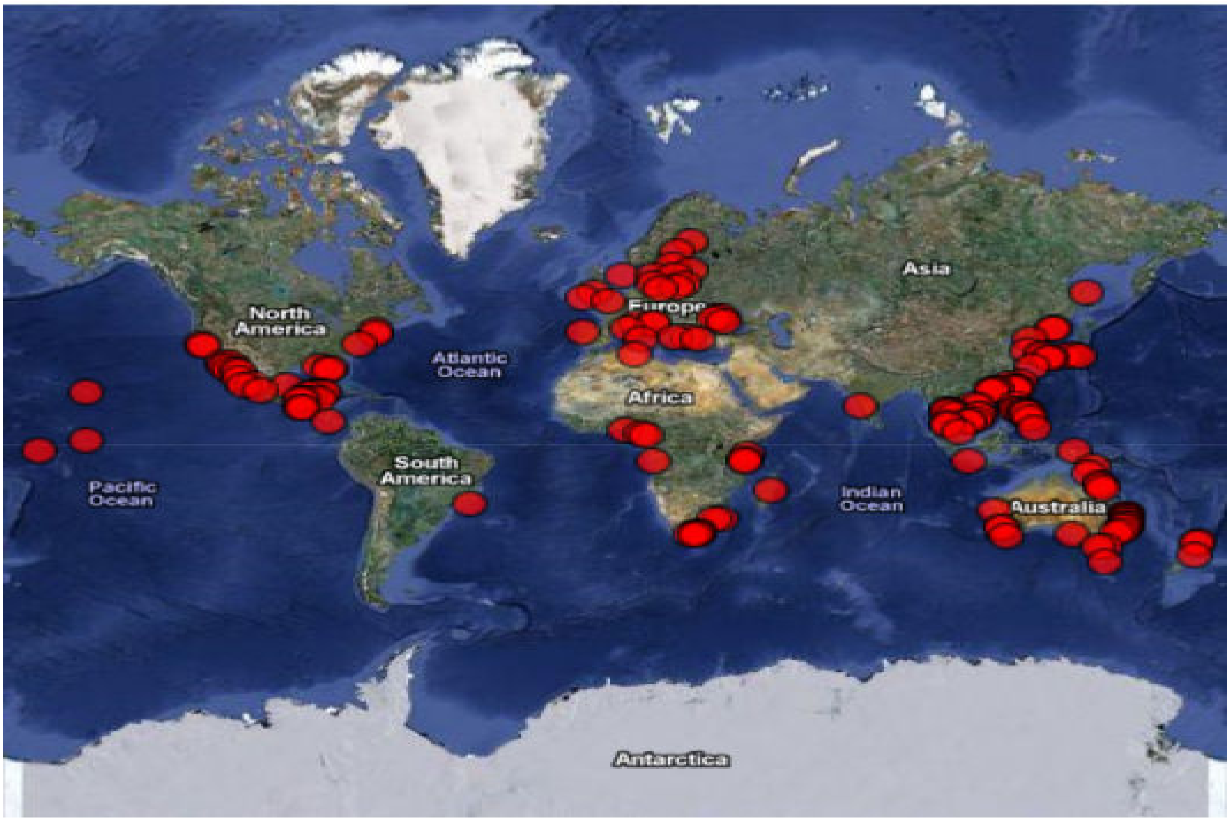


Figure 5.3: Map of autotrophic sites around the world

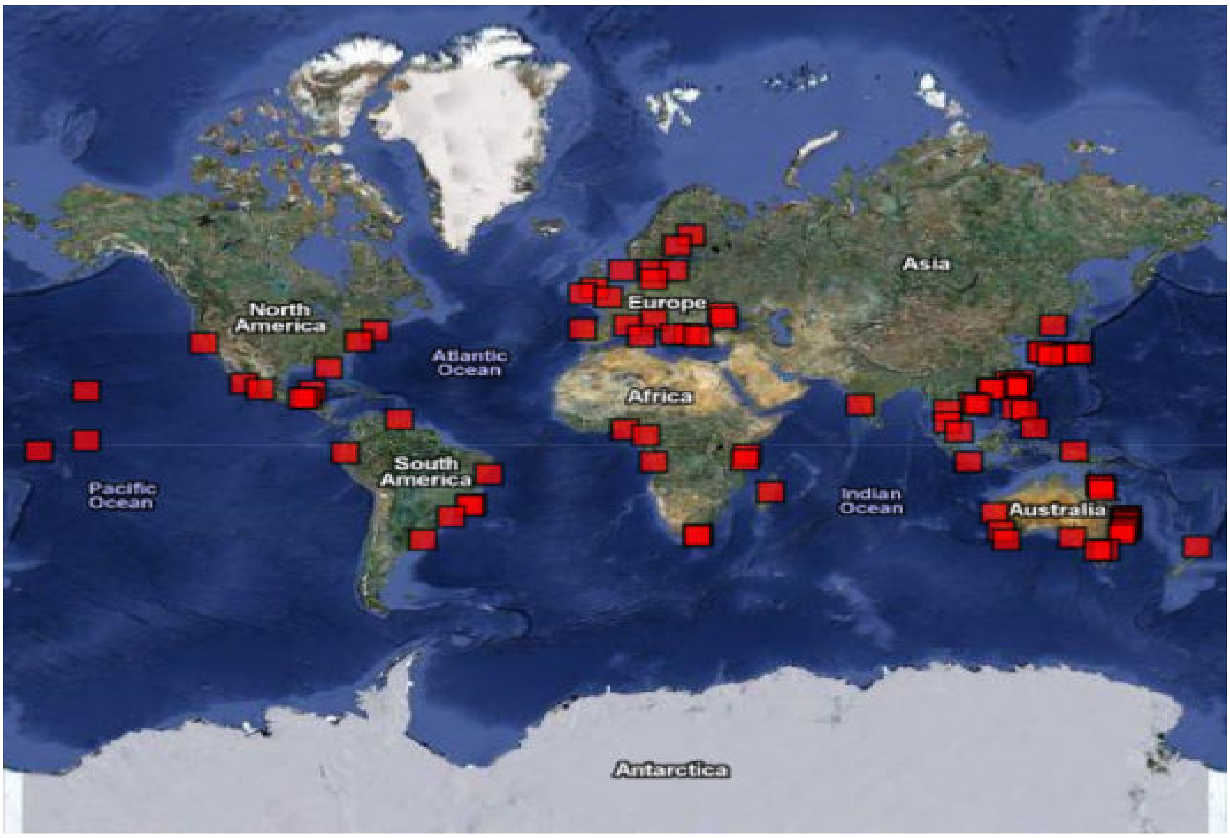
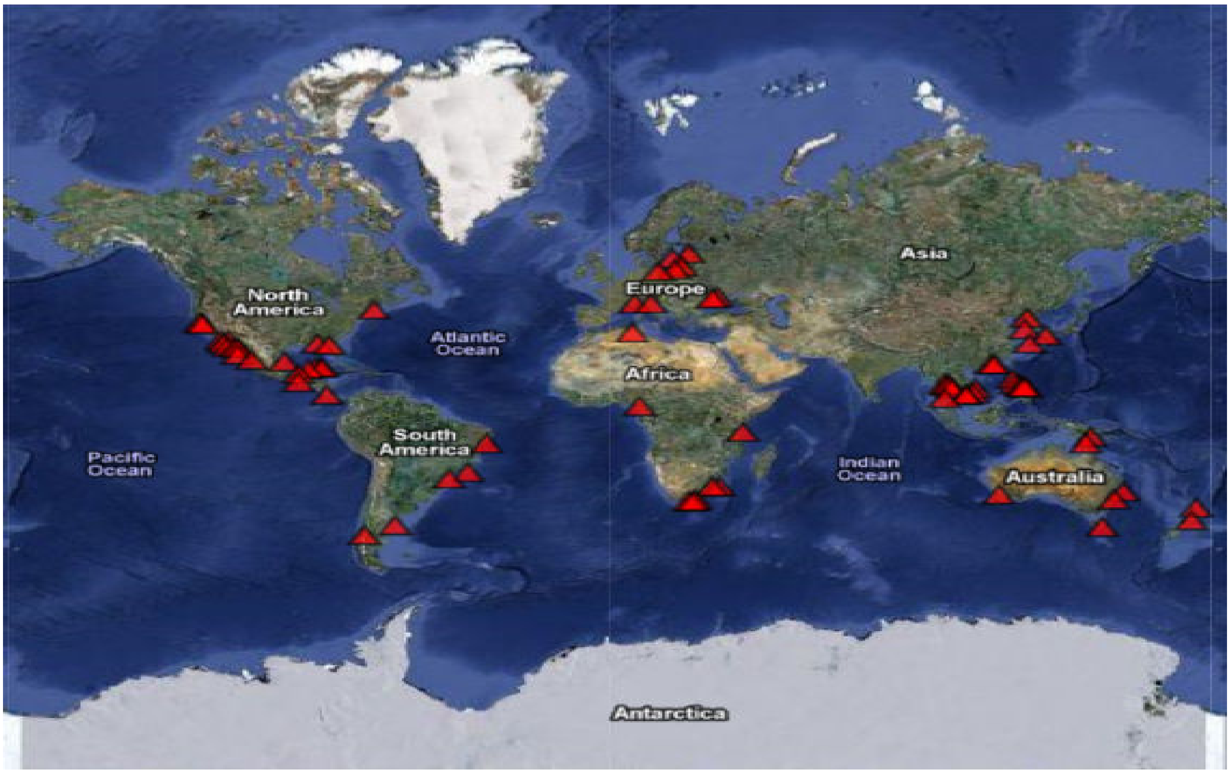


Figure 5.4: Map of heterotrophic sites around the world



5.2 New developments in the use of nutrient budgets

5.2.1. Application to muddy systems

Many coastal ecosystems around the world contain high levels of sediment, either naturally or through human intervention. The presence of such sediments (or mud) in these waters lowers the quantity of light that penetrates through, decreasing the rate of primary productivity in that system, and through this controlling life in that water column. The flux of nutrients between the sediments and the water column is controlled by bacteria, and the partition coefficient that plays a major role in the partitioning between particulate and dissolved forms of P is controlled by mud suspension. A new LOICZ budget model was developed for application to estuaries with muddy waters.

With the use of this new model, the sensitivity and health of such estuaries can be ascertained (LOICZ Deltas).

5.2.2. Link to ASSETS

Simple biogeochemical models, such as nutrient budgets that provide descriptions of the biochemical and ecological dynamics of estuaries, are utilised in determining the degree of sensitivity of an estuary to external drivers since field observations on their own may not be sufficient to describe the system without the understanding of an fundamental mechanism occurring in that system. (Swaney et al 2008). The assessment of estuarine trophic status (ASSETS) utilises this to learn the health of the system, as well as its sensitivity to an increase or decrease in external nutrient loads.

5.2.3. Link to conceptual diagrams

Conceptual diagrams aid in the understanding of processes related to the transformation of N and P in coastal and other waters (Hamilton et al), and provide insights into how ecosystems respond to global changes (IGBP). Conceptual diagrams also find use in setting up appropriate monitoring programmes, and identifying management priorities (IAN). It is a tool that can therefore be used in conjunction with nutrient budgets in the management of nutrient enrichment.

5.2.3. Link to fisheries

The nutrient dynamics of an ecosystem are important for productive fisheries in sustaining phytoplankton and peak rates of primary production in the spring. Upwelling that occurs during the summer/winter monsoon provides an allochthonous source of P relative to N (Liu et al 2009), and this may reduce any limiting effects of the nutrients on phytoplankton growth. Large-scale aquaculture aquaculture also discharges considerable quantities of nutrients into the surrounding waters. While nutrients are lost from the ecosystem through commercial and recreational fisheries, an input of nutrients may result in algal blooms that have detrimental effects on aquatic life.

5.3 Case study: S'Ena Arrubia lagoon, Italy.

S'Ena Arrubia lagoon is located along the centre-western coast of Sardinia, Italy (39.83 °N and 8.57 °E). It is 1.2 km^2 wide, has a mean depth of 40 cm, and is the residue of the wider Sassu Pond that was reclaimed in 1937 (Trebini et al information sheet).

Freshwater is input from the watersheds of two canals: Rio Sant'Anna that drains an area of 78.4 km^2, and the Canale delle aquae basse, which drains an area of 50 km^2 that is extensively utilised for farming and zootechnics. The lagoon communicates with the sea through a channel. The climate of this region is mediterranean with long, hot summers and short, mild, and rainy winters. Precipitation, averaging 65 cm annually, is low, and therefore the level of the water flowing into the lagoon is also low (Trebini et al information sheet).

The lagoon is highly eutrophic and dystrophic crises and fish kills occasionally occur. Intense phytoplankton blooms occur in the spring when terrestrial runoff is the highest (Fiocca et al 1996, Trebini et al information sheet), especially from *Cyclotella atomus* (Husted) and *Chlorella* sp. Macroalgae, consisting mostly of *Ulva* sp. And *Enteromorpha flexuosa* (Kützing) DeToni, become abundant in late spring to summer (Fiocca et al 1996, Trebini et al information sheet) and, having a high affinity for N, may compete with the development of the phytoplankton (Fiocca et al 1996).

The N:P ratio is low in the spring-summer period (Fiocca et al 1996,Trebini et al information sheet), and very high having abrupt peaks in the autumn-winter periods. These dynamics are dependent on the quantity and the way water is input from the Canale delle aquae basse, which provides a substantial percentage of the freshwater flowing into the lagoon. High fluctuations in oxygen levels are also characteristic (Trebini et al information sheet).

The lagoon is a Ramsar site that is included in its listing for important bird areas. It is a wintering and nesting area for several species of waterfowl. It and also has great importance for its fauna. The main human activities in this wetland are fishing (which is economically important), outdoor recreation, and educational and scientific research (Trebini et al 2005, Trebini et al information sheet).

Through the application of the LOICZ (Land-Ocean Interactions in the Coastal Zone) model to this lagoon, it was observed that there is a general trend for the prevalence of nutrient storing processes over nutrient mobilisation, and that N fixation dominates over denitrification. Production appears to be the dominant process over most of the year. These results prove that in terms of its net metabolism, S'Ena Arrubia lagoon is to be considered autotrophic (Trebini et al information sheet).

On many counts, coastal lagoons are characterised by large nutrient inputs from anthropogenic sources and also by limited seawater exchange, low water turnover, and long residence time. Because these systems are ecologically and economically important, several attempts have been made to improve their water quality by increasing tidal flushing. The sea mouth was widened in 2000 using engineering means to improve its flushing and hydrodynamics and reduce the high trophic levels. The change in the lagoon morphology was expected to better the water exchange, lower nutrient concentrations, and limit algal blooms (Trebini et al 2005).

When the conditions of the system before and after modification were compared, a larger number of higher salinity values was recorded after. Differences in the species of phytoplankton present were observed. The LOICZ model drew attention to the residence times of the water in the lagoon before and after the widening of the sea mouth. On the contrary, there was no relevant difference in the total and reactive P concentrations or in the concentrations of the various inorganic forms of N between the two periods. The expected reductions in nutrient exchange were not observed probably because the lagoon still receives heavy inputs from the catchments. The widening of the sea mouth in systems with high trophic levels might make its environmental conditions more marine, which would seriously modify the system's ecology (Trebini et al 2005).

Trebini et al (2005) have suggested that for proper management of the S'Ena Arrubia lagoon, it is necessary to first ascertain what the lagoon is to be used for, and then apply suitable strategies to achieve this. Data has to be established on important ecosystem variables to (1) enable the correct availability of fresh water and permissible nutrient inputs to the lagoon to maintain a good level of fish production (2) define the quantity of water that should be exchanged with the sea to obtain good vivification of water and avoid anoxia. However, the great variability in the structural components of such systems makes it impossible to accurately predict the results and consequences of any particular management strategy.

5.4 Case study: Sacca di Goro lagoon, Italy

The Po river drains a large part of northern Italy, an area with a surface of 67000 km^2 and about 17 million inhabitants. The Po di Volano canal-Sacca di Goro lagoon is a small hydrographic system partially located in the southern portion of the Po delta, and accounts for about 1% of the total area of the Po river catchment. Only the eastern portion of the watershed belongs to the delta, whereas the western portion is for most part a human-regulated system that is the outcome of a reclamation that is centuries old. The entire Sacca di Goro watershed is located in an area that lies below sea level (Viaroli et al 2006).

The Po di Volano-Sacca di Goro watershed has an area of 860 km^2 and is composed of three sub-basins: the Po di Volano, the Canal Bianco, and the Giralda canals. All these three sub-basins are human-regulated, and the Canal Bianco and the Giralda are partially connected with the Po di Volano. Freshwater inflow is partially independent of precipitation as the overall hydrographic system is human-regulated. During the rainy period, from late summer through late spring, the canals are kept empty in order to avoid the risk of flooding. In the summer, the drainage system is filled with fresh water that was released from the Po di Goro river for irrigation purposes. The water that drains from the agricultural systems is then delivered to the Po di Volano, the Canal Bianco, and the Giralda that leads to summer peaks of freshwater discharge into the lagoon (Viaroli et al 2006).

The Sacca di Goro lagoon is a shallow water embayment of the Po river delta (44°47'-44°50' N and 12°15'-12°20' E). The lagoon is surrounded by embankments and freshwater input is mainly from the Po di Volano, the Canal Bianco, and the Giralda canals. The average water retention time ranges from 1-4 days, although stagnation can occur in the sheltered portions. Water is presently exchanged between the lagoon and the adjacent Adriatic Sea through two sea mouths in the southern sand barrier. The Sacca di Goro, with minor aquatic ecosystems, is important for the protection and conservation of waterfowl (Viaroli et al 2006).

The population density of the Po di Volano watershed is relatively low, and has been negatively inclined during the last decade. The main economic activities are agriculture (utilising 80% of the watershed) and aquaculture. Changes are occurring in the agricultural and zootechnical practices as a result of the population decrease. Pesticides and fertilisers are used in crop cultivation, and that of rice requires flooded soils, which keeps the pesticides and fertilisers in solution, thereby increasing the risk of pollution to the lagoon. Livestock is of minor relevance because of a decline in poultry and cattle rearing (Viaroli et al 2006).

Sacca di Goro is currently one of the best production sites in Europe for the Manila clam *Tapes philippinarum* turning out an annual crop of several thousand tonnes. This occupation generates a high economic income of 50-100 million euros each year. Through the year, young clams (5-10mm in shell length) are collected from along the southern sand barrier and sown in the licensed areas of the lagoon. The clams grow to commercial size within a few months because of the high food availability. Market demand determines the quantities harvested. As the clams are harvested by manual dredging, this causes alterations at those points in the natural stratification of the sediment, and also an increase in sediment resuspension, nutrient release and the mobility of reduced compounds (Bartoli et al 2001, Viaroli et al 2006). Massive clam mortality has occurred due to blooms of *Ulva rigida* and related events. Evidence from research has shown that clams themselves can play a role in oxygen depletion and increasing organic nutrient cycling and the deposition of organic matter in the surface sediments, which stimulates sulphide production through the reduction of sulphate. In the farmed areas of the lagoon, the sediment was enriched with biogenic carbonates that are recognised in the control of P recycling through adsorption and coprecipitation. The dominance of the clams within the community also caused other organisms to be displaced with the loss of their associated functions (Viaroli et al 2006).

The green nitrophilous seaweed *Ulva rigida* has replaced the original vegetation of Sacca di Goro that was composed mainly of submerged aquatic phanerogams, for which N loading was considered responsible. Blooms of the red macroalga *Gracilaria* *verrucosa* were important during the late 80s and early 90s and that of the *Cladophora* species occurred in the summers of 1996-98, whereas at other times monospecific *Ulva* blooms dominated. With favourable climatic conditions, *Ulva* reached its maximum biomass, followed by a sudden collapse of the mats that decomposed causing anoxia and the release of sulphide into the water column. A general principle is that the relative abundance of the main nutrients, especially N, controls the macroalgal productivity and the onset of biomass collapse. In systems with high nitrate input like the Sacca di Goro, the nitrate was considered one of the key factors in controlling macroalgal blooms. Though the growth of *Ulva* can also regulate the retention and recycling of nutrients within the ecosystem, but the N accumulation was shown to be only a temporary sink. Upon the crash of the biomass, the nutrients are rapidly recycled (Viaroli et al 2006).

In the Sacca di Goro lagoon, the manual harvesting of the clams makes use of rakes to dredge the sediment. The harvested are represents less than 1% of the entire farmed area. Field experiments have shown that dredging increased the oxygen demand of the sediment twenty fold and the levels of ammonia and reactive P approximately 50 and 100 times over (Viaroli et al 2006). Carbon dioxide, ammonia, reactive P, and silica fluxes were all strongly stimulated by the presence of the clams. This happens because of the reducing conditions at the sediment surface that get established with the excretion and rapid degradation of organic matter at the sediment-water interface. Silica was regenerated about nine times faster probably from the feeding of the clams on silicaceous diatoms and active bacterial metabolism of the excreted matter (Bartoli et al 2001).

The overall metabolism of this lagoon was analysed using the LOICZ biogeochemical model. The residence time of the water ranged from 1-4 days, depending on inputs of freshwater and exchanges between the adjacent sea. Data for 1997 demonstrate that the internal fluxes of DIP and DIN correspond to the seasonal trend of macroalgal nutrient uptake during the growth phase and release during the subsequent disintegration. The net ecosystem metabolism follows a similar pattern with an autotrophic phase during algal growth; however on an annual basis, the lagoon is considered heterotrophic with a net ecosystem metabolism of about - 50 mmol Cm^{-2} d^{-1} (Viaroli et al 2006).

For the mitigation of water circulation in the lagoon, the canals were dredged (Viaroli et al 2006). A short-term bloom alleviation strategy used is to harvest the *Ulva* biomass when it approaches the threshold level, and Cellina et al (2003) suggest the use of a larger number vessels for this purpose as soon as the threshold is reached as it is more efficient in controlling anoxic crises and more economic. Organic alternatives for the agriculture practised on the watershed are promising ecologically and financially (Viaroli et al 2006).

5.5 Case Study: Budget analysis for the Mandovi Estuary – Goa, India

The budgeting method has earlier been applied to the Mandovi estuary located in Goa, India, as catalogued in the LOICZ database. The analysis has been repeated using data collected from literary sources:

Estuaries are an important part of the watershed-coastal zone continuum, and are affected by both natural and anthropogenic forcing. The Mandovi estuary (15º28'N;

73º52'E) (figure 5.5) is a tide - dominated estuary, situated in the state of Goa on the western coast of India, which drains into the Arabian Sea (i.e. the north-western Indian Ocean). It has, on the average, an area of about 29 km^2 and a depth of 5 m^2 (Alagarsamy 2006). The estuary has thick mangrove forests, and several small streams that originate in the Sahyadri Hills empty into this region. The hydrological characteristics of the estuary are steered by the monsoon regime (Alagarsamy 2006) and a majority of the freshwater discharge occurs only during the southwest monsoon (June – October), the runoff during the remaining months being almost negligible (Kessarkar et al 2010).

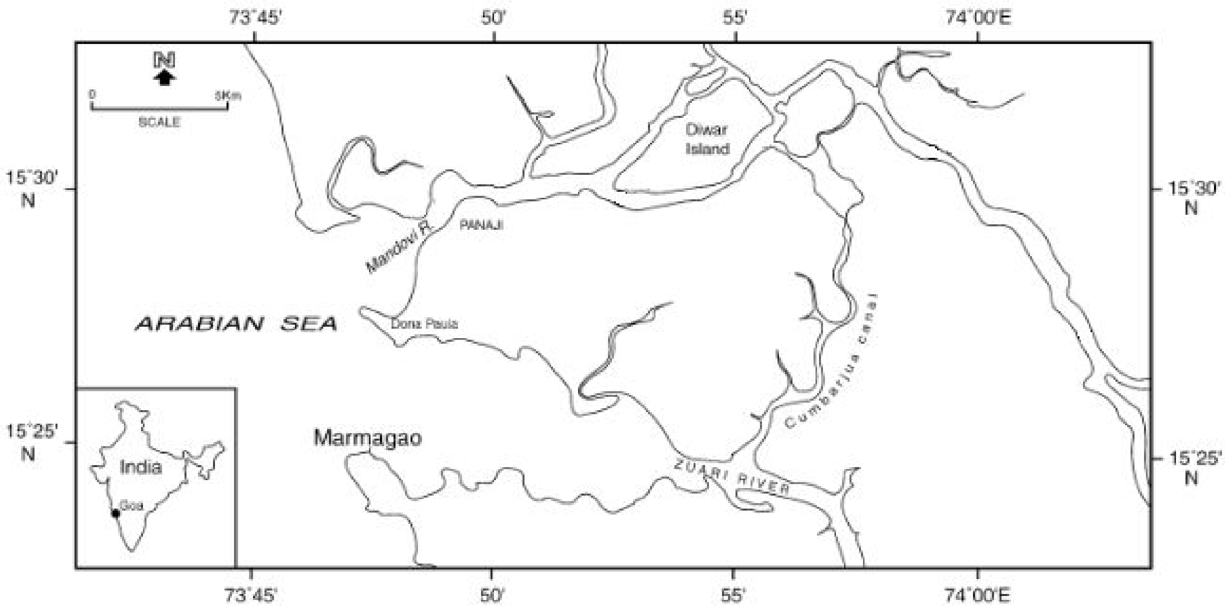
Goa is a significant source of iron ore and ferromanganese in India, and the mining and export of these minerals play an important role in the state's economy. The estuarine channel of the Mandovi is used for transportation of the mined minerals by barges to the Mormugao Harbour on the coastal Arabian Sea. Mining and industrial activities discharge nutrients, heavy metals, and inorganic and organic industrial wastes into the estuary (Kessarkar et al 2010). The estuary also receives municipal and agricultural wastes from the urban and farmed portions of the watershed. Other conspicuous activities in the estuary include the navigation by fishing trawlers, passenger ferries, and recreational activities such as boat cruises and casinos for tourists.

The Arabian Sea, being semi-enclosed, has a climate, hydrography and circulation, and biogeochemical processes that are not commonly observed. It has zones of upwelling, with widespread open-sea and coastal oxygen deficiency. NO_3^- was observed by Naqvi et al to be present in high concentrations in the euphotic zone of the western Indian shelf, where primary production was high only at the surface. The production of reactive nitrogen has also been on the rise in south Asia in the last few decades, due to which the level of atmospheric deposition of anthropogenic nitrogen into the northern

Indian Ocean is among the highest globally (Naqvi et al 2010). This may lead to the intrusion of seawater into estuaries that is already high in N and P, hence increasing their nutrient loads.

This study focuses on inferring the metabolism of the Mandovi estuary by utilising the modified Land – Ocean Interactions in the Coastal Zone (LOICZ) biogeochemical model for muddy systems to construct hydrographic budgets for the estuary. The parameters used for construction of the nutrient budgets are presented in table 5.1. The estuary has an average salinity of 25 ppt during the dry season, and 18.85 ppt during the wet season. The higher salinity during the dry season is due to heightened evaporation, lowered freshwater input from the tributaries, and seawater intrusion. The estuary is well mixed throughout the year (Divya et al 2009) and is therefore considered to be single – layered.

Figure 5.5: River Mandovi on the west coast of India (adapted from Alagarsamy 2006)



5.5.1. Water and salt balances:

The water and salt budgets for the Mandovi estuary in the dry and wet seasons are illustrated in figure 5.6. Freshwater inputs to the estuary, and residual and mixing flows were estimated from the salt balance. The volume of water entering the system during the dry season was observed to be slightly higher than the residual flow, whereas during the wet season, they are balanced. The freshwater influx of 0.162% during the dry season and 1.502% during the wet season indicates that sea water intrusion into the system is predominant throughout the year. Inflow of freshwater is through precipitation (Vp), while other sources, such as groundwater (Vg) and other (Vo), were not considered significant. As compared to the freshwater flow, the degrees of precipitation over and evaporation from the estuary were not significant. The water residence time varied from 60.97 days during the dry season to 2.45 days during the wet season.

5.5.2. Budgets of nonconservative materials:

The nonconservative fluxes for dissolved inorganic phosphorus (DIP) and dissolved inorganic nitrogen (DIN) and stoichiometrically derived net apparent biogeochemical processes in the Mandovi estuary are given in table 2. The system appears to be autotrophic in both seasons, but denitrifying during the dry season and nitrogen fixing during the wet season. There is a net loss in ΔDIP in both seasons, which is attributed to organic matter regeneration processes. The net ecosystem metabolism (NEM) is high during the wet season, with *p-r* = 4.89 x 10^3 mmol m^2 d^{-1}, and the value obtained for *Nfix – Denit* is 33.2 mmol m^{-2} d^{-1}. From this, the system appears to be a sink for DIP and DIN during the wet season.

5.5.3. Management of nutrient enrichment:

Possible strategies that may be used to mitigate nutrient loads in this estuary are:

- Monitoring and assessment of the ecological status of the estuary, with the determination of the appropriate target to be achieved.
- Formulation of plans for the management of both N and P that are suited to each sector (urban/agricultural/industrial wastes), such as the treatment of industrial and urban wastes by biological means, the use of treatment ponds, precipitation, and the use of cover plants and buffer/riparian zones in cultivated sections of the watershed.
- Intervention on point sources, with a focus on those that can be easily controlled. An instance is in the production of biogas from solid wastes from urban and agricultural sources.
- Budgeting of the estuary after a period of a few years would prove the efficiency of the management plans put into action.

Table 5.1: Parameters used in the construction of the nutrient budgets

Parameters	Dry Season	Wet Season
Precipitation ($mm\ mo^{-1}$)	**0.004135** [a]	**0.5931** [a]
Evaporation (annual average)	**485** [b]	**485** [b]
River discharge ($m^3\ yr^{-1}$)	**189.216 x 10^6** [c]	**8136.288 x 10^6** [c]
Salinity (ppt)	System box (average) **25** [c]	System box (average) **18.85** [c]
	Outer box (average) **36.25** [d]	Outer box (average) **36.2** [d]
	River discharge points **12.6** [e]	River discharge points **0.11** [e]
DIN ($\mu mol\ l^{-1}$)	System box **0.81** [f]	System box **12.99** [f]
	Outer box **1** [d]	Outer box **1** [d]
	River discharge points **1.59** [f]	River discharge points **15.63** [f]
DIP ($\mu mol\ l^{-1}$)	System box **0.22** [f]	System box **1** [f]
	Outer box **4** [d]	Outer box (Arabian Sea) **3** [d]
	River discharge points **0.36** [f]	River discharge points **0.68** [f]
SPM ($mg\ l^{-1}$)	System box **8** [g]	System box **13** [g]
	Outer box **12** [h]	Outer box **16.8** [h]
	River discharge points **ND** [g]	River discharge points **13** [g]

[a] CWC 2005; [b] Legates and Mather 1992; [c] Vijith et al 2009; [d] Morrison et al 1998; [e] Divya et al 2009; [f] Maya et al 2010; [g] Kessarkar et al 2010; [h] Bhaskar and Bhosle 2006. ND – not detected.

Table 5.2: Nonconservative nutrients and stoichimetrically derived net apparent biogeochemical processes in the Mandovi estuary during the dry and wet seasons

Season	Δ DIP (10^3 mol d^{-1})	Δ DIN (10^3 mol d^{-1})	p-r (10^3 mol m^{-2} d^{-1})	Nfix-Denit (10^3 mol m^{-2} d^{-1})
Dry	-6.638	-0.746	703.644	3.64
Wet	-46.14	245.32	4890.935	33.92

Figure 5.6: Water and salt budgets for Mandovi estuary during dry and wet seasons. Water flux in 10^3 m^3 d^{-1} and salt flux in 10^6 PSU m^3 d^{-1}.

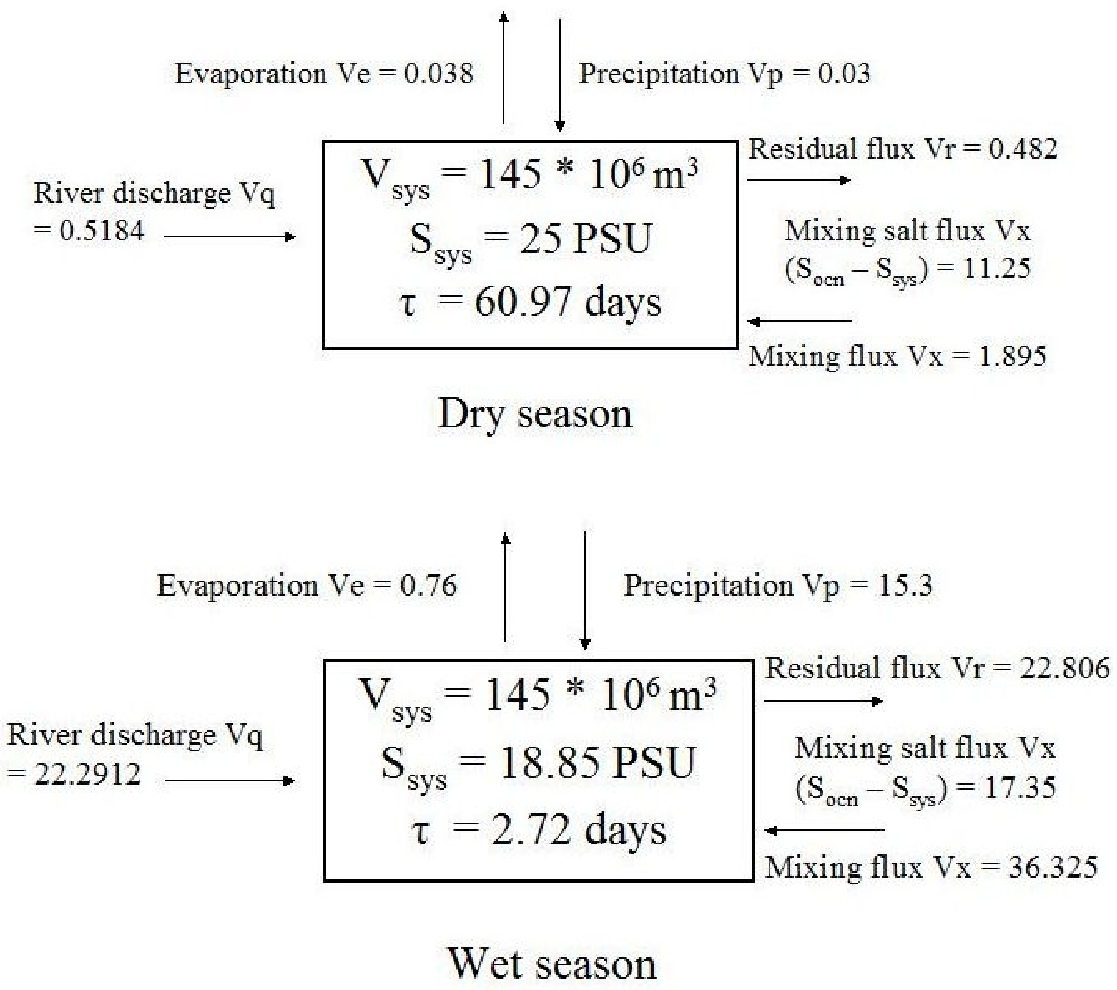


Table 3: Summary of results obtained using the modified LOICZ budget for muddy systems

Result	Dry season	Wet season	Unit
Inflow from land:	0.5184	22.291	10^3 m^3 d^{-1}
Rain inflow to estuary, Vp:	0.003	0.554	10^3 m^3 d^{-1}
Water loss from estuary by evaporation, Ve	0.038	0.038	10^3 m^3 d^{-1}
Advective water loss from the estuary, Vr	0.482	22.806	10^3 m^3 d^{-1}
Diffusive inflow from coastal waters, Vx	1.895	36.323	10^3 m^3 d^{-1}
Mean residence time	60.97	2.45	Days
1-Kd in river	1.00	0.85	
1-Kd in estuary	0.9	0.85	
1-Kd in coastal waters	0.86	0.81	
ΔKd river to estuary	0.10	0.00	
ΔKd coastal water to estuary	-0.04	-0.04	
Inflow DIP from land	0.186	15.158	
Effective DIP inflow from land	0.167	15.158	River flux d^{-1}
Advective DIP loss from estuary	1.019	45.613	Residual flux d^{-1}
Diffusive DIP inflow from coastal waters	7.489	76.596	Mixing flux d^{-1}
Delta DIP = Net budget of DIP for estuary	-6.638	-46.14	10^3 moles d^{-1}
Inflow DIN from land	0.824	348.411	Inflow d^{-1}
Effective DIN inflow from land	0.741	348.411	River flux d^{-1}
Advective DIN loss from estuary	0.437	159.533	Residual flux d^{-1}
Diffusive DIN inflow from coastal waters	0.441	-344.199	Mixing flux d^{-1}
Delta DIN=Net budget of DIN for estuary	-0.746	245.32	10^3 moles d^{-1}
p-r = net ecosystem metabolism	703.644	4890.935	10^3 moles d^{-1}
nfix – denit	3.64	33.92	mmol/m^2/day

CHAPTER 6: DISCUSSION

Eutrophication has been a threat to the health and stability of coastal aquatic systems for several years. While nutrient and organic matter enrichment of coastal waters can occur as a result of natural processes, it has been greatly accelerated by various human activities.

From the definitions of eutrophication listed in chapter 2, section 2.1, the common factor causing the cascade of effects that follow is the high input of plant growth promoting nutrients, especially N and P, from anthropogenic sources. Monitoring of susceptible systems for the appropriate biological indicators is a necessary step in managing responses to excess nutrient loading. High chl a and the presence of macroalgae are initial indicators, and monitoring for these indicators should be constant with investigation into their occurrence once they are observed. Though care should be taken when interpreting the concentration of chl a.

The character of the coastal zone varies from region to region. The extent to which N loading encourages eutrophication differs between marine systems because susceptibility to eutrophication is controlled by several factors such as size, morphology, and flushing and residence times. Systems receiving very high loads of N may develop other nutrient limitations, as described by Paerl and Piehler (2008). Temperature variation also plays an important role, whether as temperature differences between seasons or between climates. This is because it can influence the rate of release or metabolism of nutrients from sources within the system itself (i.e. the sediments and suspended organic matter) and thus changing the balance of nutrient availability.

Temperature can also alter which organisms grow optimally, and lower the oxygen holding capacity of the water (Paerl and Piehler 2008, McQuatters-Gollop et al 2009). Climate change due to global warming may therefore also have effects similar to that of eutrophication, such as increased algal productivity, altered food webs, shifts in phytoplankton community composition, and bottom hypoxia (McQuatters-Gollop et al 2009). Coastal variability within systems and between regions would imply that any single control measure may not have sufficient remedial action, and would probably need to be used in combination with other measures. With climatic change, it may be possible to learn management lessons from other regions that have faced similar situations.

The Redfield ratio of N and P for a system is an average of the internal ratio of the nutrients found in various algal species in the communities of the coastal system, and so the actual ratio of these nutrients in the system waters is determined by the algal communities present there. The average N:P ratio of a system has been used to suggest the most likely limiting nutrient by comparing it to the Redfield ratio. A high winter ratio in comparison with the Redfield ratio can also be indicative of eutrophic conditions. Increased concentrations of N and P in a system decrease the availability of dissolved Si in the water column. Therefore measuring the concentration of Si in ratio with N and P may also be an indication of nutrient enrichment. This provides vital information for nutrient management strategies. But because ecosystem responses to anthropogenic nutrient loading varies from system to system (and region to region) as highlighted by McQuatters-Gollop et al (2009), very similar policies cannot be used in all cases and have to be adapted to the characteristics of each system.

Nutrients reach estuaries and other coastal systems from a number of sources, each of which contributes differing levels of nutrients. Nutrient budget analyses can provide a picture of the relative importance of the various sources, and based on defined reference levels for the system, can provide an overall measure of its capacity for nutrient loss. Öborn et al (2003) suggest that the accurate assignment of these sources that add to the nutrient loads of a system and focussing on management practices to reduce nutrient loss from these sources during vulnerable times of the year instead of total reductions may be more cost-effective to work towards set targets. Budgets at compartment (or source) levels would be advantageous in identifying from where the nutrient surpluses arise that would finally contribute the most to the system being monitored (Oenema et al 2003).

Salt and water budgets, are especially useful in combination with nutrient budgets, because they allow estimates of average residence and flushing times of the system (Le Tissier et al 2006, Zeldis 2008b). The flushing rate is useful in determining if the system can manage the nutrient load delivered to it. The major terms of the water and salt budget also suggest the dominant forcing of the system: whether it receives greater inflow of nutrients from the ocean (including that from upwelling events) or from riverine inputs (Zeldis 2008b). In the S'Ena Arrubia lagoon, the modification of the sea mouth did not improve its conditions, which, as stated earlier, may be because the lagoon still received considerable wastewater from terrestrial sources. The utilisation of wetlands would be beneficial in achieving improvements.

Nutrient budget exercises, including inferences of net productivity, allow the estimation of a system's carrying capacity for aquaculture (such as shellfish or finfish aquaculture).

Budgets can be useful in comparing between different types of aquaculture and their effects on the system, and whether they can be developed to take advantage of complementary behaviour in terms of the effects on nutrient removal/enrichment (for example, the co-culture of algae or benthic deposit feeders) (Zeldis 2008a). The inclusion of IMTA, especially benthic feeders, in the Sacca di Goro lagoon would help to alleviate nutrient conditions.

Nutrient budgets can be credited for being simple and flexible, but as with all quantitative methods are subject to uncertainty from various sources. Uncertainty in nutrient budgets might occur from biases and other errors in measurements, and monitoring of the inputs and outputs of nutrients through time may afford additional checks on the constancy of budgets (Oenema et al 2003). To minimise inconsistencies and uncertainties when analysing and interpreting nutrient budgets, it is advisable to observe standard data conventions and guidelines, to utilise established datasets of known quality (where available), and to employ consistent methodology for evaluating budgets and their components.

Scrutinising the uncertainties in nutrient budgets may shed light on weak areas in determining cause-effect relationships. Due to variability and ambiguity, budgets should be reappraised over several years (Oenema et al 2003) to confirm their conclusions. In addition, assessing budgets on a longer-term basis may balance the effect of short-term variations (lower inputs in some years may be offset by higher nutrient inputs during other years). The "snapshot" analysis that is often used in budget studies may not have much value in the real world as they can present a fixed view of the situation as opposed to average behaviour or trends relevant to management (Scoones and Toulmin 1998).

Nutrient budgets aim to summarise the basic but essential components of complex systems and therefore reflects the assumptions and biases of researchers in determining which elements of the system under study are to be investigated in greater depth than others. Thus nutrient budgets should not be viewed as final statements on which policies can be made, but should be used to promote discussion, bringing assumptions and uncertainties to the forefront (Scoones and Toulmin 1998). The participation of stakeholders in the management of coastal systems is essential, and the right incentives and reasonable alternatives may have to be offered to take remedial action.

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