The concept of 'assimilative capacity' as a management tool in temperate coastal waters of Western Australia

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Preface

Society produces waste and currently most of this waste ultimately enters the environment, often with unknown consequences. At the same time modern communities expect a clean environment. Although it is now technically possible to recycle much of the waste generated by society, even with the best recycling programmes some proportion will always require disposal to the environment. The challenge then is to develop waste management strategies consistent with the maintenance of long-term environmental health.

Ecosystems respond uniquely to waste inputs. Some, such as the open ocean, will be able to accept certain wastes without causing unacceptable changes, while the same amount and type of waste discharged to shallow, poorly flushed coastal lagoons and embayments may result in severe changes to their biological communities. The ability of the environment to accept waste is therefore ecosystem and pollutant specific; a fact that must be recognised for effective environmental management and for the maintenance of acceptable environmental quality in the long-term.

In Western Australia, most of the population lives within 20 km of the coast and, as a result, the waters of the nearshore coastal zone have long been the focus of recreational, industrial and commercial activities. If current trends in industrial and urban expansion continue, pollutant loadings to the coastal zone will inevitably rise as a result of direct waste discharge, urban drainage and groundwater inflow. In the past, a lack of knowledge of the ecological implications of waste discharges to coastal waters resulted in catastrophic changes to the biological communities of Cockburn Sound and Princess Royal Harbour. To ensure this scenario is not repeated in the future, the critical links between pollutant loadings and environmental response must be determined in order to develop appropriate ecologically-based management strategies.

This report outlines the concept of assimilative capacity as a management tool to derive ecosystem-specific, ecologically-based maximum acceptable pollutant loadings to marine ecosystems with particular reference to the temperate nearshore waters of Western Australia. The report is the result of a collaboration between the Environmental Protection Authority of Western Australia, the CSIRO Division of Fisheries and the CSIRO Institute of Natural Resources and Environment and is a contribution to the Southern Metropolitan Coastal Waters Study (1991-1994).

1. Introduction

In Western Australia the increase in the amount of waste discharged to the marine environment is closely correlated to population growth. The discharge of treated wastewater (sewage) to the coastal waters off Perth, the capital city of Western Australia, is a good example of this. Treated wastewater was first discharged to the ocean off metropolitan Perth in the late 1920s when the population was about 200,000. By the late 1960s the population had increased to 650,000 and, by this time, 60 million litres was being discharged to the nearshore coastal environment each day. By the late 1980s, the population had risen to about 1.1 million and treated wastewater discharge had approximately trebled to 200 million litres per day (ML/day). By the end of the current decade the population of Perth is expected to exceed 1.4 million and treated wastewater discharge is predicted to reach about 350 ML/day. Until the recycling of waste is more widespread or alternative disposal methods are implemented this trend is expected to continue.

The impact of chronic discharges of pollutants into some nearshore marine and estuarine environments in Western Australia is well documented. The seagrass meadows (predominantly *Posidonia* spp) and the dependent biological communities of Cockburn Sound, off the southern metropolitan coast of Perth, and Princess Royal Harbour, near Albany, have been severely altered by chronic discharges of pollutants. In each case approximately 90% of the seagrasses that once flourished in these embayments died relatively suddenly (ie. over a period of several years) after almost 20 years of continuous waste discharge. The loss of these seagrass meadows is essentially irreversible (Clarke and Kirkman, 1989) and was directly linked to nutrient inputs to these waters (Anon., 1979; Simpson and Masini, 1990). As a result, the professional and recreational fishing potential and the amenity value of these waterbodies has been greatly diminished. The widespread changes in the biological communities of Cockburn Sound and Princess Royal Harbour occurred because of a lack of understanding of the ecological implications of chronic waste discharges into poorly flushed marine waters. Furthermore, at the time the critical planning decisions were made there was no effective statutory mechanism to control water pollution.

The Environmental Protection Authority of Western Australia (EPAWA) is now able to control pollution under the Environmental Protection Act (1986). Pollutant discharges to the environment are controlled through license conditions and if these conditions are contravened, the "polluter" can be prosecuted. Although the total load of pollutants entering the environment through licensed discharge points can now be effectively regulated, pollutant loads from diffuse sources such as contaminated groundwater, stormwater runoff and aerial inputs are much more difficult to control. If we are to maintain a healthy environment in the long-term, ecologically-based, pre-emptive control strategies must be developed and implemented. The goal of these strategies would be to ensure that cumulative pollutant loads are maintained below the capacity of the environment to satisfactorily assimilate them.

The cause of the changes in the biological communities of Cockburn Sound and Princess Royal Harbour, coupled with the predicted magnitude and nature of future waste discharge scenarios suggest that nutrient enrichment is likely to be the major threat to Perth's coastal waters. However, the level of total nutrient loading that can be safely assimilated (its *'nutrient assimilative capacity'*) is not known. Once quantified, however, the nutrient assimilative capacity becomes the ecological basis on which management plans are formulated and license conditions set.

This paper outlines the application of the assimilative capacity approach to the long-term management of pollutant inputs to the temperate nearshore waters of Western Australia, with particular reference to waters off metropolitan Perth. A more general discussion of this concept is outlined in EPA (1989).

2. Description of temperate Western Australian coastal ecosystems

Three major biotic assemblages are found in temperate Western Australian nearshore waters:

1. Seagrass meadows.
2. Algal dominated limestone reefs.
3. Bare sand communities.

Various combinations of these groups form the basic ecological units, or ecosystems, found along the Perth metropolitan coastline. Community structure within these ecosystems is largely determined by the bathymetry, exposure to wind and swell, and the circulation and flushing regimes.

The following discussion focuses on the nearshore marine environment of Perth, from the southern end of Warnbro Sound to the northern end of the Marmion Marine Park (Figure 1). It may, however, also be applied in general terms to other coastal marine areas of southern Western Australia.

The nearshore marine environment of metropolitan Perth is comprised predominantly of limestone reef with algal assemblages or sand with seagrass communities growing in the lee of the reefs or in sheltered embayments and coastal lagoons. There are areas of essentially bare sand with little macrophyte growth, and trenches over 20 m deep which are devoid of large primary producers but act as traps for organic matter and support some secondary production. The seagrass meadows are present because the lagoon is protected from wave action by the offshore reefs. These reefs are of aeolian origin and run parallel with the coast from a few hundred metres to ten kilometres from shore and from the intertidal to depths of 30 to 40 metres. The reefs extend along the west coast from Kalbarri to Cape Naturaliste (Figure 1) and, in waters less than 10 m deep, are characterised by overhangs, caves and bridges. As depths increase the reefs become less rugged.

A typical example of this reefal system is found in the Marmion Marine Park, described by Simpson and Ottaway (1987). In this area, the small kelp *Ecklonia radiata* is dominant on reefs to depths of about 20 metres, below which other large brown algae become more abundant eg. *Scytothalia dorycarpa*, *Platythalia angustifolia* and *Sargassum* spp. (Kirkman,1989). At depths of 30 m or more, the platform reefs are dominated by sponges, large hydroids and gorgonian corals. The seagrasses, *Amphibolis antarctica* and *Thalassodendron pachyrhizum*, are also found on reefs to this depth. The limestone reefs along this stretch of coastline provide the source of food and shelter for the Western Rock Lobster (*Panulirus cygnus*). These animals spend at least part of their lives in or near seagrass meadows.

The seagrass meadows are thought to be of great age, probably existing in their present form since the sea-level rose and stabilised about 5000 years ago (Semeniuk and Searle, 1986). The meadows are often monospecific, broken by blowouts or areas of bare sand where sometime, a catastrophic storm has removed entire seagrass plants including rhizomes (Kirkman and Kuo, 1990). These areas are thought to be originally crescentic but, after further damage to the weakened edges, become irregular in shape (Clarke and Kirkman, 1989). Small species of seagrass and green algae, particularly *Caulerpa*, colonise these blowouts and form dynamic communities which are often disturbed by storms but which also recover faster than the original species of *Posidonia* and *Amphibolis* (Kirkman and Walker, 1989; Kirkman and Kuo, 1990).

Because of the geomorphology of the southern Western Australian coastline there are a number of partially enclosed lagoons along the coast in which water movement is restricted and flushing times are prolonged by the presence of fringing reefs (Johannes and Hearn, 1985). The water circulation in these systems may be driven by local wind stress and regional pressure gradients (Hearn, 1983) and by water density gradients generated by factors such as differential heating, freshwater buoyancy fluxes and evaporation (D'Adamo, 1991; Hearn, 1991).

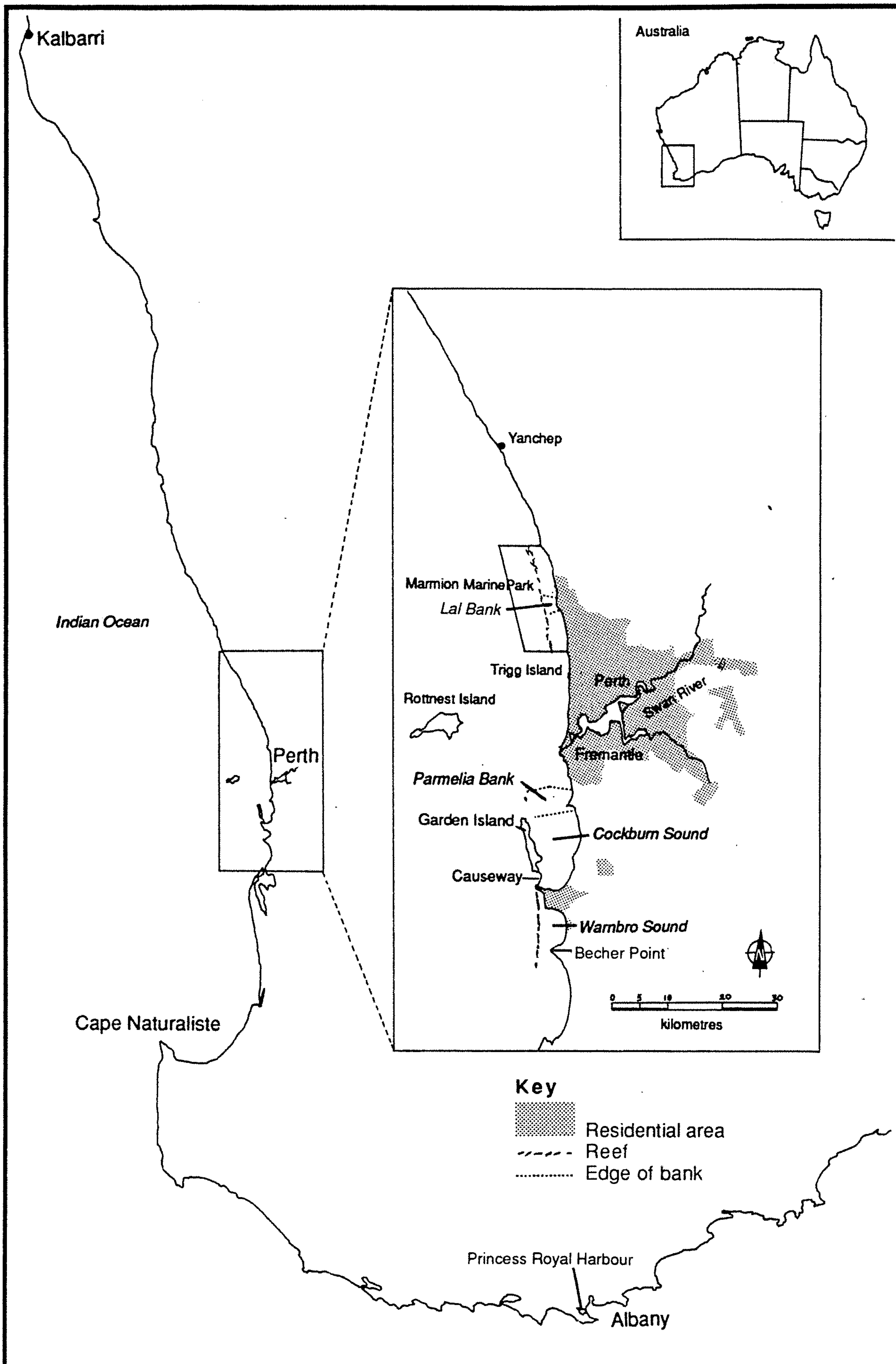

Figure 1. Southern Western Australian coastline.

Groundwater, seasonal river-flow and land runoff are significant natural sources of freshwater and nutrients for the nearshore marine environment. Johannes (1980) estimated that about 100 tonnes of nitrate-nitrogen enters the sea per year via groundwater inflow between Trigg Island and Lal Bank. Similarly Appleyard (1990) estimated that approximately 270 tonnes of total nitrogen entered the sea annually, between Yanchep and Becher Point (Figure 1) from this source. In undeveloped coastal areas, the nitrogen in groundwater is most likely derived from leguminous vegetation (Davidson and Jack, 1983). In developed areas the source is largely anthropogenic, originating from septic tank leachate, domestic waste disposal sites, industry, horticulture and from the application of garden fertilizers.

3. Defining the assimilative capacity concept

3.1 Theoretical definitions and assumptions

The *assimilative capacity* is a property of the environment and is a measure of its ability to accommodate the total input of contaminants, without unacceptable impact (modified from GESAMP, 1986; see EPA, 1989). The term *assimilative capacity* has been used synonymously with *environmental capacity* (GESAMP, 1986), *receiving capacity* (UNESCO, 1988), and *absorptive capacity* (UNESCO, 1988).

The concept of assimilative capacity as a strategy for environmental management is based on four fundamental assumptions:

- any input of a contaminant to the environment will cause change;
- there is a certain level of a natural contaminant which will not produce an unacceptable change in the marine environment at an ecosystem level;
- the marine environment consequently has a 'finite capacity' to accommodate natural wastes given a specified environmental quality objective; and
- this capacity can be quantified.

In this context the term *contaminant* is defined as a substance or energy introduced directly or indirectly into the environment by human activities. The term *pollutant* is defined as a contaminant which either affects living resources, is hazardous to human health, is a hindrance to marine activities, impairs the quality of sea water for different uses, or causes a reduction in amenity (after GESAMP, 1986).

The environmental behaviour and fate of each pollutant are central to determining the assimilative capacity of that pollutant. Some materials can be bio-magnified in the environment, others may simply accumulate, while some may have additive, synergistic or antagonistic effects on each other or with components of the natural environment. These interactions can be very complex and extremely difficult to predict.

One class of pollutant is synthetic materials. Synthetic materials are defined as those made directly or indirectly by humans and which are not found in nature except as the result of pollution. Specific examples of such synthetic materials are polychlorinated biphenyls (PCBs), organochlorines (eg. DDT), organotins (eg. TBT) and certain hormone analogues. Substances such as these may effect marine organisms directly or through bio-accumulation of these materials to toxic levels. In the past, the introduction of synthetic toxic materials such as DDT and TBT into the marine environment has caused widespread deleterious effects (Carson, 1962; IPCS, 1990). Given the lack of scientific knowledge regarding the short and long-term effects of most of these substances, the assimilative capacity for such materials is *a priori* set at zero and not considered further in this discussion. The only safe control approach for this class of pollutant is containment.

Natural pollutants can be subdivided into two broad groups: naturally occurring toxic substances such as heavy metals and hydrocarbons, and biostimulants, primarily the nutrients nitrogen and phosphorus (Table 1). The extent to which most toxic substances affect marine biota is primarily related to their concentration in water. Therefore their environmental impacts can be managed through the application of water quality criteria based on toxicological studies. Bioaccumulation of toxic substances in filter-feeding organisms and algae, and accumulation in the sediments must also be considered in managing the disposal of these substances.

The use of water quality criteria to protect ecosystems from the effects of biostimulants is, on the other hand, of little use as these materials can be removed rapidly from the water by marine plants. In contrast, the assimilative capacity approach is centred on quantifying the dominant factors controlling the conversion (or assimilation) of biostimulants into organic matter and incorporation into sediments and other internal sinks. It is from this information base, that the ecological consequences of a range of nutrient loadings can be predicted and the upper loading limits determined in relation to an acceptable level of ecological change.

Table 1. Control approaches for different pollutant types.

CLASS	TYPE	EFFECTS	CONTROL APPROACH
Natural	Biostimulants (nutrients)	Assimilated	Assimilative Capacity
Natural	Toxic substances (heavy metals)	Concentration related toxicity	Water Quality Criteria
Un-natural	Synthetic compounds (PCBs etc)	Largely unknown	Containment (AC = 0)

3.2 The concept of assimilative capacity

The concept of assimilative capacity is not new. For many years it held wide appeal to managers faced with the need to control, in some sensible manner, the impact of water pollution. Its failure to be widely adopted, despite a number of encouraging reports and assessments (GESAMP, 1986; UNESCO, 1988), can probably be best summarised by the weaknesses identified nearly 20 years ago (Westman, 1972). Ecologists correctly argued that any addition of a substance to a water body would change, in some manner, the receiving system. Therefore, in the strict sense, if the environmental quality objective is the maintenance of a pristine ecosystem (ie. there is no level of acceptable ecological change), it follows then that the assimilative capacity of this ecosystem is zero. On the other hand, managers argued that if the environmental quality objective is other than the maintenance of a pristine ecosystem, there would be some level of "no-discernable-effect" or "acceptable change" where materials would be assimilated and 'virtually disappear'.

The modern definitions of assimilative capacity incorporate subjective judgements about levels of ecological effects which are defined as acceptable or unacceptable; and the application of this approach requires that these changes are able to be predicted with some degree of confidence. Therein lies the fundamental weakness of the assimilative capacity concept in that it requires that ecological effects can be predicted, known and measured. Arguments against the assimilative capacity approach are largely centred around these fundamental points. Firstly, a perceived lack of scientific rigour in its determination (Greenpeace, 1990) and secondly, on the subjective nature of what comprises 'acceptable ecological change' (Campbell, 1986). It is commonly argued that the assimilative capacity approach is a 'license to pollute' and that it is diametrically opposed to the precautionary approach of zero or minimum inputs of pollutants. Largely, these criticisms are based on the use of assimilative capacity for toxic, persistent and bioaccumulatory substances. However they do not hold true in the Western Australian context as the assimilative capacity approach is being used here to manage the impacts of biostimulatory substances, primarily nutrients, which do not cause harm through direct toxicity (except in very high concentration), and are not bioaccumulants in the classical sense (that is they are not accumulated to high concentrations by filter feeding organisms or by passage through the food chain to higher trophic levels).

Environmental Capacity, as defined by GESAMP (GESAMP, 1986), does not appear to rely heavily on the need to be able to predict and measure effects, instead it utilises 'water quality criteria' in a key decision-making role. Reliance on water quality criteria as the primary management tool may be satisfactory, and in fact necessary, for some toxic substances, but this approach has many shortcomings. When applied to substances that either bioaccumulate to deleterious levels or to substances such as nutrients which can be rapidly assimilated, and stimulate excessive growth of aquatic plants, the water quality criteria approach is inadequate (see Section 4.5).

Biostimulants, as the name suggests, promote the growth of primary producers. The growth of fast-growing, opportunistic species such as phytoplankton and attached micro-algae, epiphytes and the attached and unattached macroalgae, is favoured over slower growing macrophytes such as seagrasses. The increased biomass of the opportunistic species can then reduce light availability and affect the growth and survival of the slower growing species. This effect will be most pronounced in plants at their lower depth (~light) limit.

Application of an ecosystem approach to managing the effects of nutrient enrichment overcomes the limitations of the water quality criteria approach. The assimilative capacity approach proposed here is aimed at understanding and predicting the short-, medium- and long-term environmental consequences of a discharge at the ecosystem level, at timeframes compatible with the processes of dispersion, dilution and biological assimilation. In contrast, the water quality criteria approach focuses on the short-term consequences of waste disposal, based on the concentration of individual elements or compounds in the aqueous phase.

It is only through understanding the interrelationships between nutrients, light and primary production in a particular ecosystem that the effects of nutrient loading can be predicted and appropriate management strategies formulated in advance. This requires a thorough understanding of ecosystem structure and function. The obvious problem however, is that the relationships between the key ecological processes and components cannot be determined precisely, that is, the approach is not deterministic, rather it is probabilistic (Campbell, 1986; Cullen, 1985). The probabilistic nature of these determinations requires that confidence or error terms be determined. Application of the lower error bound as the 'working assimilative capacity' (see section 3.5) acknowledges the probabilistic nature of assimilative capacity estimates.

3.3 Operational definition

Assimilative capacity is defined here as *'the capacity of the environment to assimilate waste whilst maintaining a specified level of environmental quality in the long-term'*. In its practical application, the concept of assimilative capacity requires that a specific use or set of uses be defined as acceptable for any specific water body, and hence any degradation of the system that alters or affects those uses beyond a certain point, can be defined as unacceptable. The use or uses of a water body can be defined on various criteria, including ecological judgements, commercial considerations and community perceptions. This leads to a requirement for a classification of waters for specified uses. The criteria and the classification may need to be interpreted by ecologists and other specialists, to ensure that community perceptions are accurately reflected in the classification categories.

This operational definition means that for the assimilative capacity concept to be applied, ecosystem boundaries must be set, pollutants be clearly identifiable and measurable, the area of potential effect determined, and the level of acceptable ecological change defined. Thus assimilative capacity can only be implemented on a strictly site and waste specific basis and for our purposes here the pollutant type is restricted to naturally occurring biostimulants.

Where effluent composition is constant, it may be possible to determine the assimilative capacity based on the whole effluent, without determining the assimilative capacity of the ecosystem for the components of the effluent.

3.4 Acceptable ecological change

The concept of assimilative capacity is predicated on the assumption that some level of anthropogenic long-term ecological change to an ecosystem is acceptable and that this change (referred to hereafter as acceptable change) can be defined in measurable biological terms. Any definition of acceptable change should take into account the maintenance of ecosystem functioning, the preservation of genetic diversity and effects on species abundance. Assuming that ecosystem dysfunction (loss of key ecological processes) constitutes unacceptable ecological change, the range of acceptable change can therefore vary from no change (eg. for the maintenance of a pristine ecosystem) up to a point just before ecosystem dysfunction occurs. In the former case the assimilative capacity is, by definition, zero. In reality, the level of acceptable change will vary from detectable changes in water quality (but not exceeding beneficial use criteria, see Section 4.1) with no detectable changes in species diversity and abundance or loss of key ecological processes, to detectable changes in species diversity and abundance but with maintenance of key ecological processes.

The level of acceptable change is determined after all existing and predicted future uses of a waterbody have been considered from a total community perspective. In a conservation reserve such as a marine park the level of acceptable change would generally be less than in waters used for other purposes such as commercial and industrial activities. Only after the level of acceptable change has been determined is it appropriate for scientists to attempt to determine the assimilative capacity.

3.5 Error terms in the assimilative capacity context

Understanding the relationships between all components of an ecosystem, even for relatively 'simple' ecosystems, is effectively an impossible task. As a result, any estimate of assimilative capacity will contain a degree of uncertainty (error), the size of which depends upon the level of understanding of these relationships. This probabilistic nature of assimilative capacity determinations is one of the major criticisms of this approach.

The accuracy of assimilative capacity estimates and, therefore the magnitude of the error term, is related primarily to the number of intermediary processes operating between the pollutant of concern and the response shown by the components of the biological system that are used as 'health indicators', in other words the pathway complexity. It follows then that the magnitude of the error term is inversely proportional to the level of understanding of how the ecosystem functions (key ecological processes) in relation to the pollutant in question (Figure 2). At some point, however, additional resources allocated to increasing understanding further will not result in significant reductions in the error term and a balance must be achieved between the management objectives and the degree of uncertainty that is acceptable to the managers.

As a further precautionary measure, the lower error bound (Figure 2) should be used as the 'working assimilative capacity' for that system (ie. the maximum acceptable pollutant loading to the system). Thus, a large difference between the assimilative capacity and the working assimilative capacity reflects a high degree of uncertainty in the initial assimilative capacity determination. As our understanding of the ecological system in question increases, the degree of uncertainty decreases thereby reducing the difference between the assimilative capacity and the 'working assimilative capacity'.

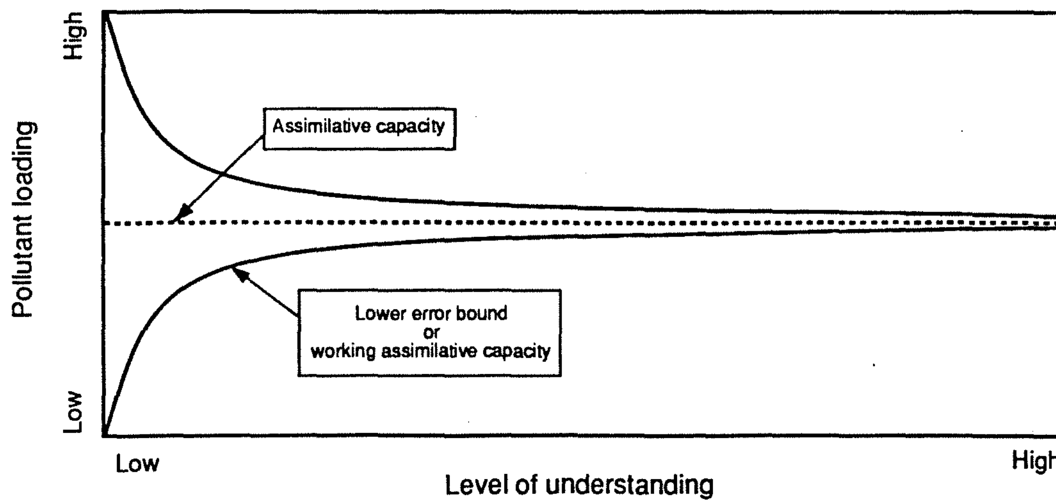

Figure 2. Conceptual relationship between the error associated with the assimilative capacity determination and the level of ecological understanding of the system in question.

If the management objective for an ecosystem includes the protection of rare, endangered, or unique species and habitats, any error in prediction may be considered too great and effectively preclude the setting of an assimilative capacity.

3.6 Partitioning assimilative capacity

The process of apportioning the assimilative capacity of an ecosystem can be described by analogy to the distribution of a cake. Essentially the size of the cake (the assimilative capacity) is fixed, however, the relative sizes of the slices (the portions of the assimilative capacity) may vary in relation to various factors. In the allocation process, consideration may be given to dividing all the available assimilative capacity to support the existing uses of the ecosystem; to allocate some to the proposed uses of the ecosystem; or a portion of the assimilative capacity may be retained to facilitate unforeseen uses of the ecosystem in future years.

In a similar manner to the primary decision of defining 'acceptable ecological change' for an ecosystem, the process by which the assimilative capacity of that system is apportioned will be decided socio/politically. However, the assimilative capacity (given an acceptable level of ecosystem alteration) must always be determined from an ecological basis.

4. Determining assimilative capacity

In order to determine whether certain types of waste can be discharged to a particular environment, it is necessary to designate the environmental quality objective (the level of acceptable ecological change) first and then to determine the current ecological status of the receiving environment in relation to this objective. This can be determined by comparing the results of quantitative surveys in the area of potential impact to 'control' sites. If the receiving environment has already been altered beyond the designated acceptable level, then, by definition, the assimilative capacity has been exceeded and pollutant loads should be reduced. Alternatively, if the existing status of the receiving environment is within the level of acceptable ecological change then again, by definition, the assimilative capacity has not been exceeded. In this case some increase in certain pollutant loads can be considered.

To be able to predict the assimilative capacity for any unique pollutant/site combination it is necessary to be able to predict the impact of the specified pollutants at the levels encompassing the likely range of discharge concentrations and amounts, both in the short and long-term. This can only be done on a unique site and pollutant basis, although emphasis should be placed on the component of any discharge suspected to be the most detrimental since this will set the assimilative capacity for the effluent. In the case of nutrients, this would be the nutrient that is most limiting to plant growth. In determining the assimilative capacity of an ecosystem for a specific pollutant, it is essential to consider all sources of that pollutant to ensure that when the assimilative capacity is determined it is not exceeded by inputs from sources beyond management control.

4.1 Interaction with beneficial use

The marine environment is a highly valued resource in the broadest sense, and its utilisation must be determined only after a thorough evaluation has been made on both socio-economic and scientific grounds. Only when a suitable use is determined can the assimilative capacity concept be investigated and possibly applied. Valid uses of marine waters include maintenance and preservation of aquatic ecosystems, passive recreation, direct contact recreation, navigation, commercial and recreational fishing and aquaculture.

In Western Australia the usage classification of waters is known as 'Beneficial Use' (Anon.,1981), a term synonymous with 'Protected Environmental Value' used in the Australian and New Zealand Environment and Conservation Council draft national water quality guidelines. A **beneficial use** of the environment is defined as "... **a designated use of a specified part of the environment for the overall benefit of the community.**" For example a particular part of Cockburn Sound might be designated for direct contact recreation such as swimming or wind-surfing. The *beneficial use* of another area might be for industrial purposes. *Beneficial uses* can also be applied as a planning tool to partition use and minimise conflict when proposed uses of the receiving environment are incompatible. For example the discharge of untreated abattoir effluent into a body of water precludes direct contact recreation such as swimming.

Each *beneficial use* has a unique set of environmental quality criteria that must be met in order to preserve that beneficial use. The environment quality criteria are largely derived from toxicity tests and public health considerations. For example the water quality criteria for direct contact recreation specifies a very low level of faecal bacteria contamination for human health reasons. That same level would be unnecessary if the waters were for industrial purposes only. However, if more than one *beneficial use* is applied to the same water body, the most stringent criteria must apply. Due emphasis and consideration should be given to assigning the *beneficial use* of a system as this is a crucial step in determining the acceptable level of ecological change, and hence the assimilative capacity of that system.

Thus *beneficial use* is predominantly concerned with maintaining a designated environmental quality and depends on the most appropriate perceived use for that part of the environment at that point in time.

To quantify an acceptable level of ecological change that is compatible with a specified *beneficial use* it is essential to establish a working ecological knowledge of the water body in question, together with any adjacent waters with ecological or hydrodynamic linkages. This knowledge should include all the habitats and ecological processes linked to the *beneficial use*. Investigations should be undertaken to produce habitat inventories, baseline (spatio-temporal) descriptions of key habitats and species, and identification of key processes that maintain the system. It is within the framework of this description of the water body that the community's perception of what is considered to be the acceptable level of ecological change can be accurately defined through an iterative procedure involving managers and scientists.

4.2 Boundary determination

A marine ecosystem rarely has a well-defined physical boundary, however, to determine its assimilative capacity, it is necessary to impose a 'boundary'. Boundaries must be chosen to encompass the area of potential impact of the pollutant in question. Dispersion and dilution of pollutants are controlled to a large degree by the hydrodynamics and bathymetry of the area in question. In the first instance, the boundary of a nutrient effects model for point source inputs could be set as the point where a conservative tracer in the contaminant stream is diluted to background levels. For practical reasons, where the system boundary intersects major biotic assemblages such as seagrass meadows or limestone reefs, the boundary should be altered to include these features.

Once boundaries have been set, the assimilative capacity concept requires that the effects of different loadings of the pollutant be predicted, and using the designated level of environmental quality (level of acceptable ecological change), maximum pollutant loads (ie. the assimilative capacity) can be determined. To estimate assimilative capacity in a real situation it is necessary to evaluate the likely effects of the contaminants on many aspects of the receiving environment, within the context of the proposed *beneficial uses*. To do this the most cost effective approach is to use simulation models.

4.3 Model determination

To estimate assimilative capacity using a modelling approach it is important that attention be focussed onto habitats, species, and processes which are critical to the proposed *beneficial use* and the functioning of the ecosystem as a whole.

The starting point in this process is to develop a conceptual model of the receiving system in question. In the first instance, the model should be elementary with the specific purpose to identify the most important components and processes of the system. Initially, this could be a simple model, limited to say 20 or 30 components. Such a simple conceptual model can be based on existing published literature, and on the expertise of local ecologists and oceanographers. A conceptual model of this type is shown in Figure 3.

The next step is to determine, or estimate, the effect that each known pollutant in the effluent will have on important habitats, species and processes. This step in the model building process is dependant on the collective wisdom and experience of biologists and oceanographers in identifying potential effects, and their magnitude, by examination of all the information available for the area in question.

These estimates need to be in the form of functional relationships including measured or estimated rates or fluxes. For example, to relate algae to nutrients, the relationship between algal growth rate and nutrient concentration must be established. This information can then be used to make estimates of the relationship between nutrient load to a system and the accumulation of algal biomass. The estimates of effects must include direct and indirect effects, at least in the first instance, but some may be excluded at a later stage. Each simple-effects model can be linked together to form a larger model of the effect of certain pollutants.

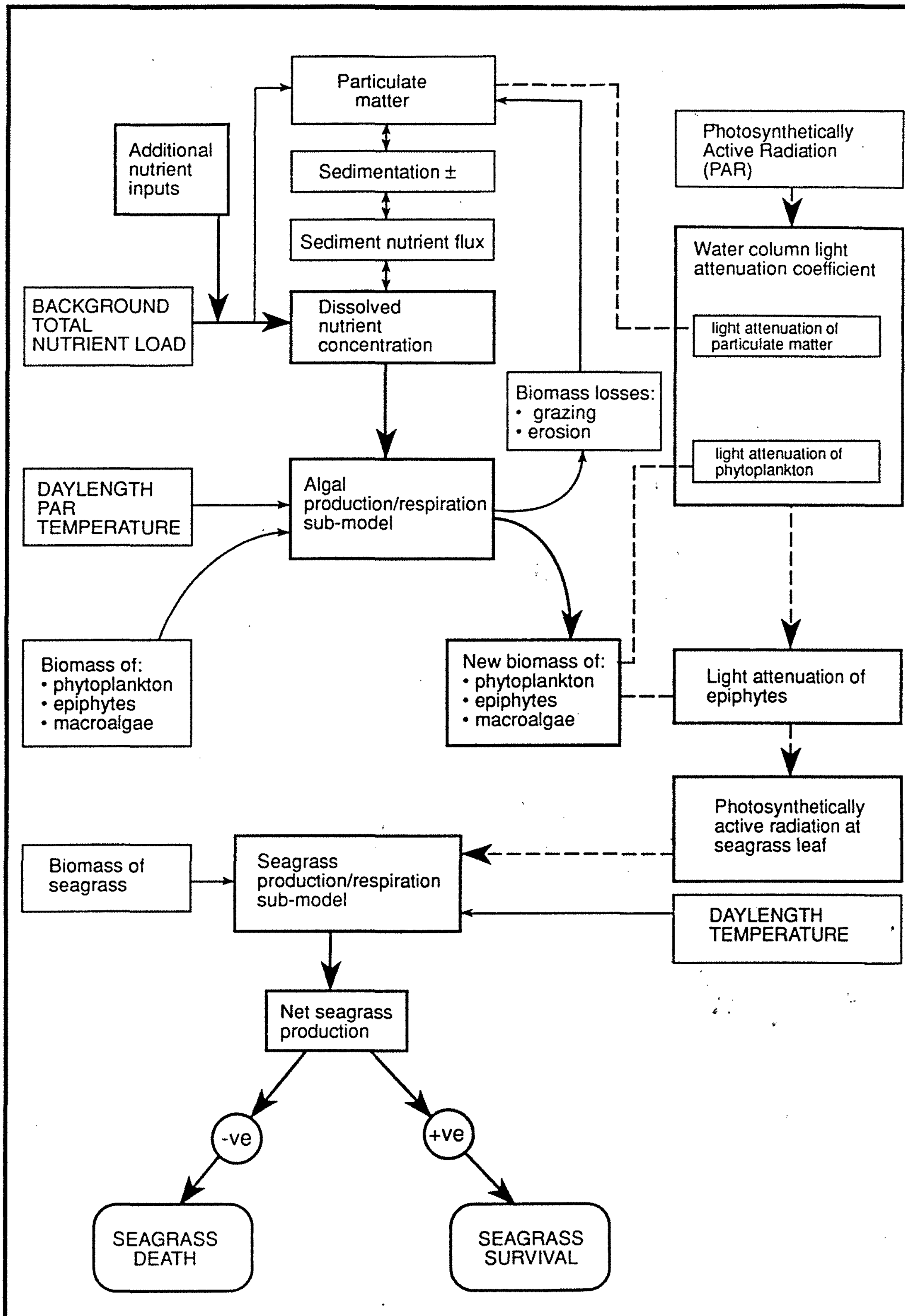

Figure 3. A simple conceptual ecological model for a seagrass dominated ecosystem.

This step of estimating the effects of pollutants is integral to the system model-building procedure, and a critical part of the overall outcome of the model-building approach. The simulation model should be constructed to examine the effect of a range of nutrient concentrations and loadings over various timescales, and their subsequent effect on the key variables chosen for the model. This will provide the link between a specific load of nutrients and the ecological consequences of this loading; the major goal of nutrient assimilative capacity determinations.

An analysis can then be undertaken to determine the key processes operating in the ecosystem and those most sensitive to various input scenarios. Decisions can then be made as to the applicability of the existing process data and whether more information is required to refine the model. The focus of the field and laboratory studies can be determined in this way to make the most efficient use of available resources.

4.4 Ecological criteria

The development of ecological criteria on which to measure the effectiveness of management measures and the reliability of assimilative capacity estimates is largely determined by the dominant *beneficial use* chosen for the ecosystem. Although adjacent ecosystems may have very similar or identical components, each could have a different set of ecological health indicators or key species depending on its designated level of environmental quality. In the case of maintenance of aquatic systems these criteria will be system-based, and can only be derived from a synthesis of the ecological data, the oceanographic information, and the sensitivity of flora and fauna to the discharge. Such a synthesis is best accomplished using a modelling approach.

The ecological criteria most useful will be those selected from the the population level of organisation or higher. Thus, at the population level the criteria would probably include such measures as abundance, distribution, biomass or production of a species, and at the system level such variables as primary production, species composition, total number of species, total abundance, and total biomass.

Ecological indicators have been used to determine stress. For example, log-normal transformations of species data (Gray and Pearson, 1982) and abundance/biomass comparisons (Warwick *et al.*, 1987) have refined the ability of managers to detect stress at an early stage. Gray (1989) defines three clear changes in community structure in response to stresses. These are a) a reduction in diversity, b) retrogression to dominance by opportunistic species and c) reduction in mean size of the dominating species.

Ecological indicators have a number of desirable qualities. They should be likely to be affected by changes in the pollutants of concern, be of importance in the system, relatively abundant and easily identifiable and measurable. In most cases it will also be necessary to include variables not likely to be affected by the pollutants of concern, particularly from amongst those deemed to be key variables in the system. This is to optimise the probability that unpredicted changes will not pass unnoticed. They should also be more sensitive and indicative of change than the dominant component of the criteria defining acceptable ecological change. For instance, if a level of acceptable ecological change due to nutrient enrichment was set as some percentage loss of seagrass biomass from the lower end of its depth range, the ecological health indices should include parameters such as standing crops of epiphytes or light availability, rather than simply presence or absence of seagrass. In this way, monitoring results would validate the predictive model from which the assimilative capacity was derived, also ensuring early detection of undesirable trends.

4.5 Measuring assimilative capacity

Because of the inherent complexities involved in predicting responses of the natural environment to anthropocentric perturbation, it is necessary to validate any predictions of assimilative capacity by direct measurement in either the field, mesocosm or laboratory. Alternatively, it may be possible to generate empirical assimilative capacity models and validate theoretical assimilative capacity models where the biological effects of a range of pollutant loads to an ecosystem are known.

Long-term data from Cockburn Sound (Figure 1) provides a simple example of how the nutrient assimilative capacity of a marine embayment can be determined using historical data. Cockburn Sound is a 'nitrogen limited' embayment to the south of Perth bounded by Parmelia Bank to the north, Garden Island to the west and a rock-fill causeway to the south and has been used extensively as a port facility, and as receiving waters for industrial and domestic waste since the 1950s. Prior to industrialisation, seagrass meadows (*Posidonia* spp.) grew on the fringing sand banks to a depth of about 10 m. By the early 1980s most of these meadows had died (Anon., 1979).

In this case-study, system averaged data are derived from a long-term Cockburn Sound monitoring programme undertaken during summer over the past 10 years. Seagrass mortality in the deeper areas of Cockburn Sound was caused by light starvation as a result of increased light attenuation caused by phytoplankton blooms which, in turn, have been shown to be related to nitrogen inputs (Chiffings, 1979; Cambridge *et al.*, 1986). To rehabilitate this ecosystem, the management goal is to reduce phytoplankton standing crops, thus improving light penetration to the extent that seagrass growth is not light limited. This is based on the assumption that relationship between seagrass survival and light is the key ecological process for this ecosystem.

A relationship has been derived linking light penetration to chlorophyll *a* concentration (a measure of phytoplankton standing crop) in the water column (Pearce, 1990; Figure 4).

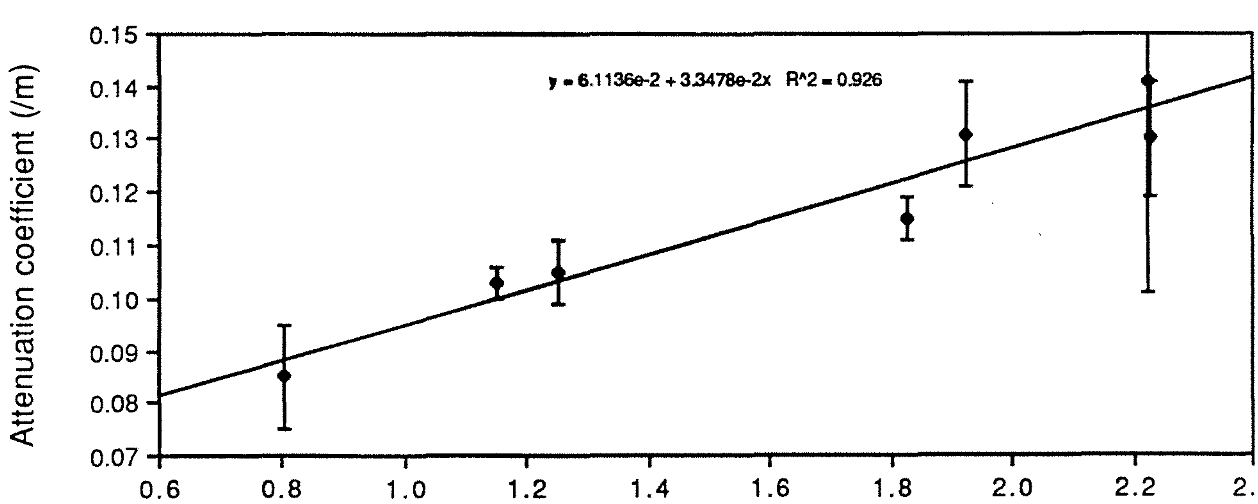

Figure 4. Relationship between mean chlorophyll a and light attenuation coefficient derived from long-term averages of system-wide data collected in Cockburn Sound during summer.

Chlorophyll *a* concentration in the water column is positively related to mean daily nitrogen load to Cockburn Sound (Pearce, 1990; Cary *et al.*, 1991; Figure 5a). If the environmental quality objective is set such that "...sufficient light be available to a depth of 9 m for long-term survival of *Posidonia* spp." and having established the above relationships (Figure 4), coupled with knowledge of the light requirements of seagrasses (Masini *et al.*, 1990), it is possible to draw a direct link between average daily nitrogen load and the depth of seagrass survival (Pearce, 1990; Figure 5b). Therefore, the assimilative capacity of Cockburn Sound is the load of nitrogen, which can be added to the Sound that will not cause the key seagrass species to be adversely affected at a depth of less than 9 m in the long-term; that is about 500 kg/day. Thus the relationship between the key pollutant (ie. nitrogen) to the system and a key component of the ecosystem (ie. seagrass) can be established and is directly applicable to management by providing an ecological basis to set maximum pollutant loadings.

The predominantly negative relationships between water column nutrient and chlorophyll *a* concentrations (Figure 6a and b) imply that a management programme aimed at increasing water column nutrient concentration would cause a reduction in chlorophyll *a* concentration and therefore an increase in water clarity. Clearly, in this case study, a narrow approach based solely on water quality criteria, would be inappropriate and misdirected. Effective management requires a broad, holistic, ecosystem-based approach utilising quantifiable, predictive relationships between causes (in this case nutrient inputs to the system) and effects (seagrass survival; Figure 5b).

4.6 Monitoring and review of assimilative capacity

Once the assimilative capacity has been determined and discharge commenced, regular monitoring should be undertaken to ensure that the designated environmental quality is maintained. The monitoring programme should be temporally compatible with the key processes in question and focus on components of the environment that are sufficiently sensitive to the pollutant/s being discharged such that any undesirable trends can be readily identified and intercepted through the implementation of appropriate management strategies.

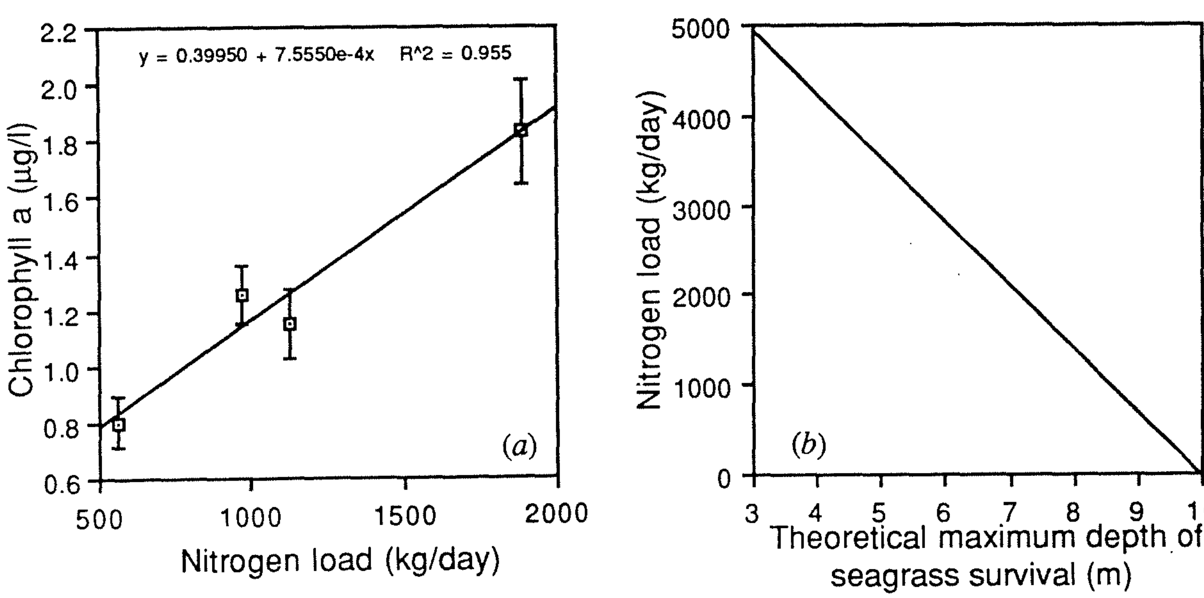

Figure 5. Relationships derived from long-term averages of system-wide data collected in Cockburn Sound during summer.

(a) Relationship between average daily industrial nitrogen load and chlorophyll a concentration

(b) Theoretical relationship between average daily industrial nitrogen load and the maximum depth of seagrass survival.

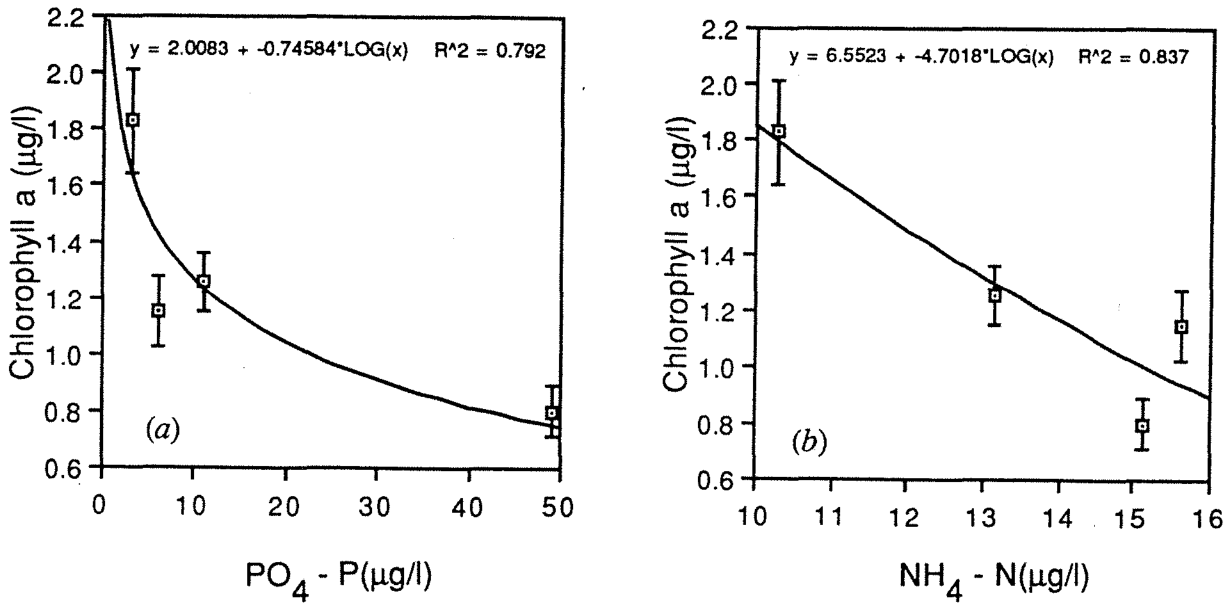

Figure 6. Relationship between mean chlorophyll a and water column nutrient concentration derived from long-term averages of system-wide data collected in Cockburn Sound during summer. (a) phosphate-phosphorus and (b) ammonium-nitrogen.

The monitoring programme may indicate that a revision of the initial estimate of the assimilative capacity is required to maintain the designated environmental quality of the ecosystem. The frequency of review will depend on the proportion of the assimilative capacity being utilised and the level of uncertainty (ie. magnitude of the error term) in the initial determination. It must also take into consideration both the critical ecological time-frames (eg. timescale of seagrass decline) and the time-frame required to effect changes in pollutant loadings if required (eg. commissioning new pollution control infrastructure).

5. Summary and conclusions

Biostimulants have been identified as constituting the most significant threat to the ecological health of coastal marine ecosystems in Western Australia (Pearce, 1990). It is clear that water quality criteria are of little use in developing management strategies for biostimulants given their rapid assimilation into the environment. The assimilative capacity approach examines the cumulative effects of pollutants on the ecosystem as a whole and in this approach the independent variable (the level of acceptable ecological change) is defined, and from that the dependent variable (the biostimulant load) is determined. This provides an ecological basis on which to limit waste discharge. The assimilative capacity approach allows for the maintenance of pristine ecosystems in that an assimilative capacity for a particular pollutant can be set as zero.

The merits of employing the assimilative capacity approach as a management tool for Western Australian marine ecosystems are summarized below:

- assimilative capacity is measurable and provides an ecological basis on which to limit waste discharges;
- assimilative capacity is dynamic, and can vary with community perceptions and scientific knowledge;

- the error term in assimilative capacity can be determined and allows for appropriate safeguards against unpredicted effects; and
- assimilative capacity can be continually refined from analysis of monitoring programs.

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