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**CANADIAN GUIDANCE FRAMEWORK FOR THE MANAGEMENT OF
NUTRIENTS IN NEARSHORE MARINE SYSTEMS**

Scientific Supporting Document

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Summary

Over the last several decades nutrient over-enrichment of coastal marine waters has become a serious problem in many regions of the world. In response, many national and international agencies are in the process of developing guidelines for determining the amount of nutrients that nearshore systems can assimilate before they begin to exhibit the symptoms of nutrient over-enrichment. Within Canada, the Water Quality Task Group of CCME (Canadian Council of Ministers of the Environment) and Environment Canada have initiated a number of studies to assess the degree to which nutrient over-enrichment threatens our nearshore marine systems. CCME has determined that a nutrient guidance framework should be developed for nearshore Canadian marine systems. The objectives of this study were to develop a guidance framework that addresses nearshore marine eutrophication using a science-based approach to establish site-specific guidelines for managing nearshore marine eutrophication, and to carry out case studies to demonstrate the application of the guidance framework.

This document also provides an overview of environmental problems associated with nutrient over-enrichment, the causes of nutrient enrichment, and approaches to dealing with nutrient over-enrichment currently being developed in various jurisdictions world wide. Of the approaches reviewed, the USEPA approach is the most developed and documented in terms of providing science based stepwise procedures to establish nutrient criteria and it is recommended that this approach be adopted for establishing nutrient criteria for Canadian nearshore waters.

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1. Introduction

Within the last few decades eutrophication of coastal marine ecosystems has become recognized as a serious global problem, particularly along coastlines of the more industrialized nations. Nutrient enrichment is considered by many to be the greatest threat to the integrity of coastal systems (NRC 1994; Pelley 1998). It is also believed that this problem will become more severe as coastal watersheds are subjected to increased urbanization and industrialization. Cultural eutrophication of marine systems dramatically increased around the world beginning in the 1950s and 60s and has been related to an estimated eight-fold increase in consumption of chemical fertilizers, as well as to increases in burning of fossil fuels, land use changes, and wastewater discharges in coastal areas. During this period, the amount of nitrogen entering many of the coastal ecosystems in the northeastern U.S. has increased from 5 to 14 times above natural background values (Jaworski et al. 1997). In addition to ammonia-induced toxicity of aquatic organisms and nitrate stimulated growth of algae, increased N₂O generation and atmospheric emissions are also receiving attention, especially as they relate to potential increase of atmospheric greenhouse gases (Shiskowski 2006). Conley (2000) has estimated that phosphorus loading to estuaries has increased 2-6 fold. Within North America, it has been estimated that 60% of U.S. estuaries are moderately to heavily enriched with nutrients (Bricker et al. 1999, 2003). With some exceptions, coastal marine eutrophication does not currently appear to be as severe in Canada. Notable exceptions to this are estuaries located in agricultural areas within Prince Edward Island and New Brunswick.

The consequences of marine eutrophication can be quite severe. In extreme cases it can lead to finfish and shellfish mortality, death of benthic organisms, loss of important nursery habitat and biodiversity, and impairment of recreational value resulting in decrease in property values of coastal areas. As a result, the Canadian Council of Ministers of the Environment has determined that a guidance framework is necessary to aid in the management of nutrients responsible for causing nearshore marine eutrophication.

The prevention, control and management of eutrophication require an ability to determine the trophic status and assimilation capacity of aquatic systems. Only with this information can criteria be developed to serve as guidelines for the degree of nutrient enrichment permissible before the harmful effects of eutrophication become evident in coastal environments. These criteria can also serve as targets to restore water quality in degraded systems.

The major objectives of this document are to (i) develop a guidance framework that addresses nearshore marine eutrophication using a science-based approach to establish site-specific guidelines for managing nearshore marine eutrophication and (ii) perform case studies to demonstrate the utility of the guidance framework with respect to the data requirements and how the framework can be applied to set nutrient criteria.

2. Defining Nutrient Enrichment

The term **eutrophic** literally means ‘much feeding’ or ‘well nourished’. The term **eutrophication** refers to the processes by which aquatic systems become eutrophic. Nixon (1995) has defined eutrophication as ‘... an increase in the rate of supply of organic matter to a waterbody’, and indicated that this definition includes both autochthonous (i.e., produced within) and allochthonous (i.e., imported from outside) organic inputs. This definition does not imply that eutrophication is necessarily bad. Natural environments are often classified on a scale that ranges from oligotrophic (‘little feeding’) to hypertrophic (‘excessive feeding’). Some of the world’s most productive and valuable ecosystems, such as the upwelling areas found along the coasts of Peru and Nova Scotia, can be considered naturally eutrophic systems, and are among the world’s most commercially important marine areas in terms of the fisheries they support.

Another term, which is becoming more popular in use, is **nutrient over-enrichment**. The United States Environmental Protection Agency (USEPA) defines nutrient over-enrichment as “...the anthropogenic addition of nutrients, in addition to any natural processes, causing adverse effects or impairment to beneficial uses of a waterbody” (USEPA 2001a). The distinction between eutrophication and nutrient over-enrichment is that the former does not necessarily imply that an aquatic ecosystem is experiencing harmful effects as a result of nutrient inputs, while the latter term does.

An excess of nitrogen and, in some cases phosphorus, is usually responsible for causing eutrophication in coastal systems. Both are necessary components of aquatic ecosystems and, with the exception of ammonia, are not toxic by themselves. However, over-enrichment can produce conditions that result in harmful alterations of ecosystems, especially when they cause algal blooms that result in hypoxic or anoxic conditions and the creation of substances, such as hydrogen sulphide and methane, which are toxic to aerobic organisms.

3. Consequences and Symptoms of Nutrient Over-Enrichment

The consequences of nutrient over-enrichment are numerous and include changes in both structure (biological communities) and function (ecological processes) of aquatic ecosystems. These changes have been summarized in numerous documents (e.g., Bricker et al.1999; NRC 2000; USEPA 2001a).

The initial consequence of nutrient over-enrichment is increased plant growth. This is often accompanied by changes in species composition. Phytoplankton communities change from a primarily diatom based community to one in which smaller, flagellated forms assume dominance. This change may also be accompanied by an increase in the development of harmful algal blooms (HABs), such as red and brown tides, which can be

harmful to shellfish, fish, marine mammals and, in some cases, become a direct threat to humans.

In shallow environments and intertidal zones where sufficient light reaches the bottom, fast growing macroalgae species, such as *Ulva*, *Cladophora*, and *Enteromorpha*, may increase, eliminating slower growing macroalgae and sea grasses like *Zostera*. The latter are important habitats and nursery areas for many organisms. In deeper environments, the high phytoplankton concentrations may reduce bottom light levels to the point where sea grasses and other benthic plants are completely eliminated.

One of the most serious consequences of nutrient over-enrichment is the decreased level of dissolved oxygen it can cause within the water column. Increased plant growth results in increased sedimentation of organic particles and decomposition of this material may result in the depletion of dissolved oxygen causing either hypoxic or anoxic conditions, particularly in stratified systems where vertical mixing and exchange can be much reduced. In shallow coastal systems, excessive macroalgal growth can create anoxic conditions within the water column, especially during periods of warm water temperatures and during night time darkness. This can result in death and the subsequent decomposition of the macrophytes causing further depletion of dissolved oxygen that may result in the death and elimination of aerobic benthic organisms and, in severe cases, fish kills (Rabalais et al. 1996).

HABs are caused by a number of groups of microscopic algae and can have serious economic consequences. In addition to causing the death of zooplankton, fish, shellfish, seabirds, and marine mammals, they can cause serious illness in humans, such as paralytic, amnesic, and diarrhetic shellfish poisoning (Hallegraeff 1993).

The causes of HABs are not at all clearly understood. Although their incidence appears to be related to nutrient over-enrichment, this has not been unequivocally identified as the cause in all situations (Boesch et al. 1997). Many have suggested that nutrient over-enrichment alone is insufficient to cause a HAB, and that other physical, chemical, and biological factors are involved (Livingston 2001).

There is considerable evidence that the incidence of HABs is increasing worldwide (Anderson 1989; Smayda 1990), but some have suggested that this may be a result of more intensive monitoring (Hallegraeff 1993). There is, however, little agreement as to the reasons for this apparent increase. Possible causes include (ECOHAB 1993):

1. Species dispersal through, currents, storms, or other natural mechanisms.
2. Nutrient enrichment of coastal waters by human activities, leading to a selection for, and proliferation of, harmful algae.
3. Increased aquaculture operations which can enrich surrounding waters and stimulate algal growth.
4. Introduction of fisheries resources (e.g., through aquaculture development) which then reveal the presence of indigenous harmful alga.
5. Transport and dispersal of exotic HAB species via ship ballast water or shellfish seeding activities.

6. Long-term climatic trends in temperature, wind speed, or insolation.
7. Increased scientific and regulatory scrutiny of coastal waters and fisheries products.
8. Improved chemical analytical capabilities that lead to the discovery of new toxins and toxic events.

It has been suggested that the changes brought about in nutrient ratios as a result of anthropogenic nutrient over-enrichment may be largely responsible for HABs. A depletion of silicate relative to nitrogen and phosphorous is thought to change phytoplankton species composition from a diatom based community to one dominated by flagellates, some of which may be species responsible for HABs (Officer and Ryther 1980; Conley et al. 1993).

4. Nutrients of Concern

The nutrients most often implicated as limiting phytoplankton growth in marine systems are nitrogen, phosphorus, silicate, and iron. There is considerable evidence that the particular nutrient limiting productivity can vary seasonally; phosphorus during spring, nitrogen during summer, and silicate at the end of a spring diatom bloom (D'Elia et al. 1986; Taylor et al. 1995; Conley 2000). There is also evidence that reduction of phosphorus inputs to some estuaries directly influences the magnitude of the spring phytoplankton bloom, whereas reductions in nitrogen inputs reduce summer phytoplankton biomass (Conley 2000).

Although less intensively studied than phytoplankton, marine macroalgae growth in temperate systems also appears to be limited primarily by nitrogen, and nitrogen supply is thought to be the main determinate of peak seasonal rates of growth and net primary production (Valiela et al. 1997).

4.1 Nitrogen

In contrast to freshwater systems, where phosphorus is usually the limiting nutrient and, when present in excess, leads to eutrophication, temperate marine systems are thought to be most often limited by nitrogen. This is based on evidence from nutrient ratio data, bioassays and large-scale nutrient enrichment experiments (Ryther and Dunstan 1971; Nixon 1995; Nixon et al. 1996; Oviatt et al. 1995; Howarth and Marino 1998). The reason for this difference between freshwater and marine systems is not entirely clear, but it is often suggested to be due, at least in part, to the lower rates of nitrogen fixation in marine systems as a result of the lower availability of trace elements, such as iron and molybdenum, required for nitrogen fixation (Howarth and Marino 1998; Paerl and Zehr 2000), or to the apparent inhibition of nitrogen fixation by the salts contained in sea water (Marino et al. 2002). An additional factor that may be of importance is the fact that coastal marine systems receive water inputs from oceanic waters as well as upstream

terrestrial systems, and the former often have low N:P ratios, presumably as a result of denitrification on the continental shelves (Nixon et al. 1996).

The cycling of nitrogen in aquatic ecosystems is complex (Fig. 1). Nitrogen occurs in numerous forms. It is largely a gaseous cycle as opposed to a sedimentary cycle and the transformations between forms are mostly biologically mediated. Nitrogen fixation in aquatic systems is carried out mostly by cyanobacteria and other bacteria that transform the relatively inert N_2 gas into a biologically available form, initially as organic- NH_2 which eventually becomes incorporated into the food web. Death and decomposition releases nitrogen as NH_4^+ which is preferentially taken up by primary producers and once again incorporated into the aquatic food web, or used by chemosynthetic denitrifying bacteria as an energy source or terminal electron acceptor under anaerobic conditions.

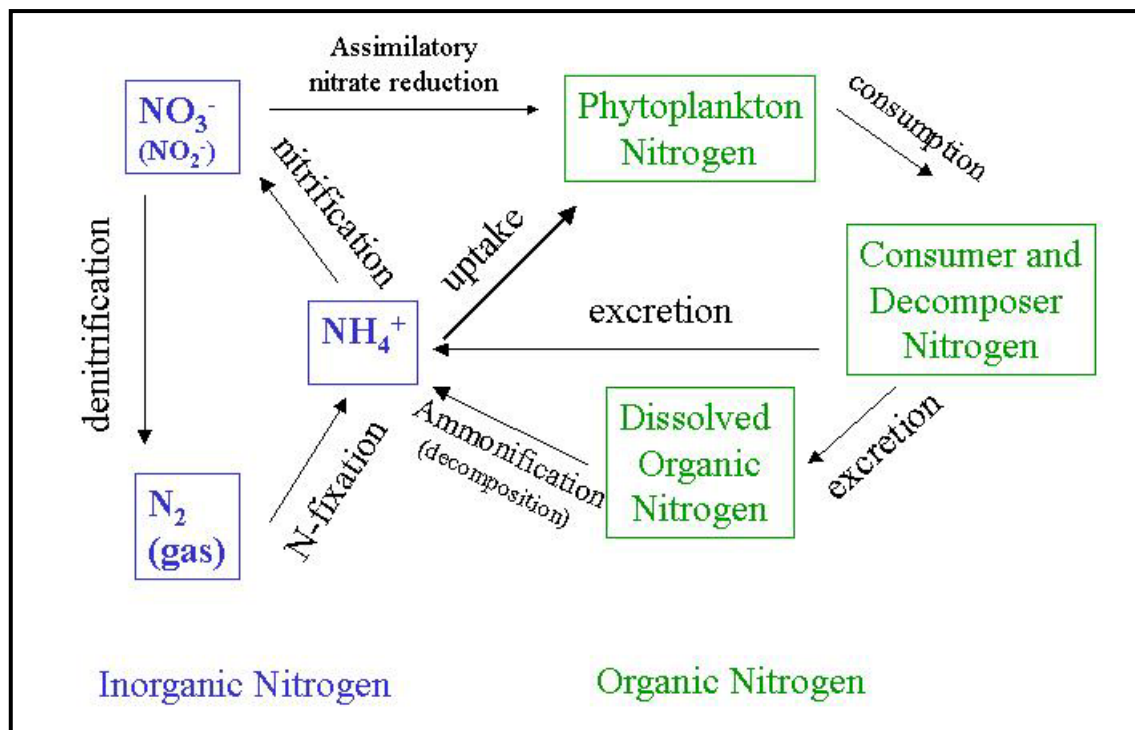


Figure 1. Aquatic nitrogen cycle.

The process of denitrification, which reduces NO_3 to NO_2 , NO , N_2O and N_2 , can be very active in coastal systems in areas where there is close proximity between aerobic and anoxic conditions and may be another reason why nitrogen is often the important limiting nutrient in marine systems (Seitzinger 1990; Law et al. 1991). The presence of anoxic conditions alone can accentuate eutrophication by inhibiting the denitrification process since it inhibits the transformation of NH_4^+ to NO_3 , the precursor to denitrification (Kemp et al. 1990).

4.2 Phosphorus

In contrast to nitrogen, the phosphorous cycle is a sedimentary cycle. The forms of phosphorus present in aquatic systems do not undergo large changes in oxidation-reduction states and the cycle is not mediated by biological processes to the same extent as nitrogen. The forms of phosphorus in seawater include that contained in particulate living and dead organisms and that present as dissolved inorganic and organic forms. Phosphorus entering the system is taken up by primary producers as dissolved phosphate and incorporated into the food chain where it is recycled upon death and decomposition, or becomes oxidized to an insoluble precipitate where it settles to the sediments and may remain sequestered until it is transformed into a soluble form under conditions of low oxygen concentration. The regeneration from sediments can be quite rapid (Fisher et al. 1982).

There is some evidence that phosphorus may be limiting in marine systems during periods of high freshwater inflows, such as may occur during the spring snow melt in temperate zones, but this is thought to be only temporary, and during summer and fall these systems become nitrogen limited (Fisher et al. 1982; D'Elia et al. 1986; Malone et al. 1996). Phosphorus may also be limiting in systems that have exceptionally high nitrogen inputs combined with stringent P input controls (Howarth 1998). This suggests that under some conditions, both phosphorus and nitrogen must be considered in plans designed to manage nutrient over-enrichment (Chapelle et al. 1994).

4.3 Silicate

Silicate is a major nutrient required by diatoms for construction of their outer cell walls. Nutrient over-enrichment with nitrogen and phosphorus is thought to result in a decrease in silicate within the water column as a result of excessive diatom growth followed by the settling of diatoms where silicate then becomes sequestered within the sediments (Officer and Ryther 1980; Conley et al. 1993). In coastal systems having high nutrient inputs, this decline in silicate is often followed by a shift from a diatom based phytoplankton community to one in which flagellates dominate (Officer and Ryther 1980; Conley et al. 1993).

4.4 Iron

Iron is an important trace element for primary producers and nitrogen fixing organisms and has been shown to be a major limiting nutrient in some oceanic marine systems (Martin et al. 1994). There is some evidence that it can be limiting in coastal systems having high turbidity because of its ability to form complexes with sediment particles (Zhang 2000). Its importance as a major limiting factor in estuaries, however, is generally considered to be minor in most instances (NRC 2000).

4.5 Nutrient Ratios

Redfield ratios¹ are often used to determine whether nitrogen or phosphorus is the nutrient likely to be limiting in a particular system and at a particular time. In freshwater studies N:P is traditionally computed on the basis of total nitrogen and total phosphorus, whereas in marine systems the ratio is typically computed based on the dissolved inorganic forms of nitrogen and phosphorus.

For phytoplankton in estuarine systems, Boynton et al. (1982) suggested atomic inorganic N:P values <10 indicate nitrogen limitation and values >20 indicate phosphorus limitation. Others have reported much wider ranges (Valiela 1995) and it has been suggested that some of this variation may be due to the form of nitrogen present, e.g., whether the inorganic nitrogen is in the form of ammonium- or nitrate-N (USEPA 2001a).

The ratio of nitrogen to silicate is also sometimes used in marine systems to determine which of the two is limiting for diatoms. Diatoms require silicate and nitrogen at a molar ratio of about 1:1 and when Si:N ratios fall below 1, conditions become more favourable for flagellates than for diatoms (Conley et al. 1993; Cloern 2001).

Benthic marine macrophytes have N:P values on the order of 30:1, and are much more depleted in phosphorus compared to the values for phytoplankton (Atkinson and Smith 1983).

5. Sources of Nutrients

Since pre-industrial times, the amount of biologically available nitrogen entering the biosphere each year has about doubled (Galloway et al. 1995; Howarth 1998), and there is a strong relationship between nitrogen inputs to a coastal system and the human population density within its watershed (Cole et al. 1993). The most important sources of anthropogenic nutrient inputs to coastal systems are wastewater discharges, fertilizers, and atmospheric deposition (Valiela 1995).

When scaled over the entire watershed, even small nutrient losses per unit area of the watershed can be quite large (NRC 2000; Sowels 2003). Of particular importance is fertilizer use since nitrogen, unlike phosphorus, is not retained to any large degree in soils. Wastewater discharges assume greater importance in heavily urbanized watersheds, especially if they contain high levels of organics that could increase the potential for development of decreased dissolved oxygen levels. In some cases it may be important to consider not the total loading rate, but rather the seasonal variation in loading rate and how this correlates with the time at which the system is most susceptible

¹ Redfield (1934; 1958) determined that the ratio of carbon:nitrogen:phosphorus in phytoplankton was on the order of 106:16:1 by moles and it is often assumed that phytoplankton take up these elements from seawater in the same proportion.

to eutrophication, for example, in the summer when freshwater inputs and flushing rates may be lowest (Vallino and Hopkins 1998).

Van Breemen et al. (2002) carried out a large-scale comprehensive study of nitrogen budgets for 16 large watersheds in the northeastern U.S. These watersheds were mainly forested but varied widely in other land use characteristics and human population densities. The items included in the budget were atmospheric deposition, amount of fertilizer applied, net feed and food inputs, biological nitrogen fixation, river discharge, accumulation and export of wood products, changes in soil nitrogen, and losses in land and river due to denitrification. Atmospheric deposition, nitrogen fixation by plants, and nitrogen imported in fertilizer, food and feed contributed about equal nitrogen inputs. About half of the losses were in gaseous form through denitrification, 20% were in riverine export, 6% were in food export, and 5% were in wood export. About 10% of the inputs were stored in soils indicating that the systems were not in steady state. This study provides some idea of potential nitrogen sources and their relative magnitudes in a particular set of watersheds, but it is important to appreciate that nitrogen budgets can vary widely from place to place.

Although surface runoff from the land is usually considered to be the major path by which non-point sources of nitrogen enter waterways, there is considerable evidence that groundwater inputs may be equally or more important in areas where aquifers are hydraulically connected to the sea through permeable soils. This has been found to be the case in a number of New England estuaries (Valiela et al. 1990; Paerl 1997).

Aquaculture operations, especially finfish farms where food is added, can be important sources of nitrogen inputs in coastal areas where this activity occurs (Merceron et al. 2002). It has been estimated that only about 40% of the nitrogen contained in fish foods is incorporated into fish biomass, the rest being released to the environment as metabolic wastes, feces and uneaten food fragments (Strain and Hargrave 2005). Fish and seafood processing plants have also been considered as one of the potential sources of nutrients in coastal waters (Fisheries and Oceans Canada 2003).

In some coastal areas, the transport of nitrogen into coastal areas from offshore can be greater than land-based inputs (Howarth et al. 1998). This is true for many coastal areas along the Gulf of Maine (Townsend 1998; Mills 2003) where nutrient rich deep oceanic water upwells along the coast. Within Canada, similar situations exist within Halifax Harbour (Petrie and Yeats 1990) and the Juan de Fuca Strait/Straight of Georgia/Puget Sound estuarine system (Harrison et al. 1983).

6. Nutrient Sinks

Aside from being flushed out of coastal areas and transported to offshore environments, the major sinks for nitrogen entering coastal systems are burial in sediments and denitrification. The degree to which nitrogen becomes deposited and buried in sediments depends in part on the type of organisms present in the system. Nitrogen taken up by

diatoms, for example, is more likely to be deposited in sediments because of their high sinking rates, whereas nitrogen taken up by smaller flagellated organism is less likely to be deposited because of their lower sinking rates and ability to swim.

Physical characteristics, particularly those related to water column stratification, are also important in determining the degree to which materials are flushed out of as opposed to being entrained within the estuary. Stratified estuaries tend to entrain suspended materials and this property has lead to stratified estuaries being considered as 'nutrient and sediment traps' (Kennedy 1984).

Denitrification in sediments and wetlands is probably the dominant nitrogen sink. Seitzinger (1990) have shown that the rate of denitrification in coastal systems increases linearly with nitrogen loading rates, and that 40-50% of the nitrogen entering in the dissolved inorganic form is removed by denitrification. Dettmann (2001) has provided some evidence that this rate may be higher, on the order of 70%, as the water residence time of an estuary increases.

The uptake of nitrogen by some marine macrophytes and subsequent burial in sediments may also be an important sink for nitrogen. Valiela et al. (1997) has pointed out that coastal systems containing salt marshes often have lower water column nitrogen concentrations than those in which salt marshes are absent, but it is not clear if this is due to nitrogen uptake by the salt marsh plants or the fact that salt marshes create environmental conditions that favour denitrification.

The degree of bioturbation of sediments may also have an influence on denitrification rates as burrows are thought to contain the aerobic/anoxic interface required for this process to occur (Pelegri et al. 1994).

7. Indicators of Nutrient Over-enrichment

An important component of managing nutrient over-enrichment is the ability to determine the degree to which a system is impacted by excessive nutrient loading. Numerous approaches, based largely on sets of "indicators", have been developed in attempts to accomplish this. In the U.S., the National Oceanographic and Atmospheric Administration (NOAA) under the National Estuarine Eutrophication Assessment (NEEA) has developed an extensive set of indices of nutrient over-enrichment for estuaries (Bricker et al. 1999), and within Europe member states of the European Union have developed indicators for a diversity of coastal systems (OSPAR 1997, 2001). Australia and New Zealand have also been active in development of indicators of nutrient over-enrichment (ANZECC 2000).

The degree to which various indices give comparable results has not been fully assessed. In some cases when comparisons between various indices have been carried out, the results have been less than satisfactory. Newton et al. (2003), in a review of numerous European eutrophication criteria developed for marine systems, found that different

classification methodologies often result in different trophic categories. It is therefore important to fully understand the advantages and limitation of each of these approaches before they are applied to ensure that a fair evaluation is made. In addition, because the diversity among coastal systems, in terms of both structure and function, is great, it is important that the selection of indicators for any particular system be chosen carefully, and that they be specific and appropriate for the environmental conditions present within the system being evaluated.

7.1 Characteristics of Indicators

Indicators can take many forms, but all have the following essential characteristics (EC 2005):

Environmental indicators “... are selected key statistics which represent or summarize a significant aspect of the state of the environment ... They focus on trends in environmental changes, stresses causing them, how the ecosystem and its components are responding to these changes ... They are important tools for translating and delivering concise, scientifically credible information in a manner that can be readily understood and used by decision-makers at all levels of society.”

The International Council for the Exploration of the Sea (ICES) suggested the following criteria be used when developing environmental indicators useful for assessment and management of nutrient over-enrichment (ICES 2001):

1. Relatively easy to understand by non-scientists and other users.
2. Sensitive to a manageable human activity.
3. Relatively tightly linked in space and time to that activity.
4. Responsive primarily to a human activity, with low responsiveness to other causes of change.
5. Easily and accurately measured, with low error rate.
6. Measurable over a large proportion of the area over which the indicator is to apply.
7. Be based on an existing body or time series of data to allow a realistic setting of objectives.

Mills (2003) reviewed 30 existing programs worldwide that have proposed sets of indicators to assess nutrient over-enrichment in nearshore marine environments. The most common indicators used are the following:

1. **Chlorophyll *a*** - a measure of the concentration of phytoplankton present in a water body. The higher the concentration, the greater the degree of nutrient enrichment.
2. **Macroalgae abundance** – in shallow marine systems, blooms of macroalgae are a common result of nutrient over-enrichment. They can smother other benthic plants and result in loss of benthic habitat.

3. **Epiphyte abundance** - epiphytes are small plants that grow on the surfaces of other plants and substrates. One of the first signs of nutrient over-enrichment is the growth of epiphytes on benthic substrates and on slower growing plants such as seagrasses. At high concentrations they can result in the loss of submersed aquatic macrophytes by shading their leaves.
4. **Loss of submerged aquatic vegetation** - this may as a result of shading caused by high phytoplankton concentrations or heavy growth of epiphytes on the leaves.
5. **Low dissolved oxygen** - this occurs when phytoplankton die and sink to the bottom where they decompose using up large amounts of dissolved oxygen. This may also result from the growth, death, and decay of benthic macroalgae such as sea lettuce.
6. **Harmful algal blooms** - although it is still unclear as to the exact relationship, the incidence of harmful algal blooms is, in some cases, thought to be related to nutrient over-enrichment.

Although numerous sets of indicators or indices have been proposed, the choice of which approach is best under a given situation has not been resolved. The following presents a review of those that have been suggested and discussed most widely in the literature.

7.2 Indices Based on Organic Matter Supply

After reviewing numerous studies that reported on the amount organic matter delivered to coastal systems, Nixon (1995) proposed the trophic status classification (Table 1).

Table 1. Coastal system trophic status based on organic matter supply (Nixon 1995).

Trophic Status	Organic Carbon Supply (g C m⁻² yr⁻¹)
Oligotrophic	<100
Mesotrophic	100-300
Eutrophic	301-500
Hypereutrophic	> 500

Although this provides a very general guideline for evaluating trophic state, it is difficult to apply because it is based on only one variable, which may vary widely on an annual basis, in addition to being a variable that is not typically included in routine monitoring programmes (EEA 2001).

7.3 Nutrient and Phytoplankton Based Indices

The trophic classification of freshwater lakes has been well developed as a result of the extensive work of agencies such as of the Office of Economic Development and Cooperation (OECD) among others (Vollenweider and Kerekes 1982). The most widely accepted classification scheme is based on limiting nutrient (phosphorus in the case of freshwater systems) concentration, chlorophyll *a* concentration and Secchi disk depth. Although there is some evidence that the nutrient most often limiting in marine systems (nitrogen) can be correlated to marine productivity in coastal systems (Nixon 1995), others have found that nutrient concentration may not be a reliable indicator of eutrophication (Cloern 2001; Dettmann 2001) and most attempts to use this and similar approaches for coastal systems have not been particularly successful (Giovanardi and Tromellini 1992; Innamorati and Giovanardi 1992). Vollenweider et al. (1998) and others (Nixon 1995) suggest that this is largely due to the greater size and spatial gradients, as well as the more complex hydrodynamics, in coastal systems as compared to lakes. Others have argued that this approach should be considered further and point out that these same problems were considered insurmountable when the OECD approach for freshwater lakes was first attempted (Hakanson 1994).

7.4 Trophic Index for Marine Systems (TRIX)

A somewhat similar approach to the OECD trophic index for lakes is the Trophic Index for Marine Systems (TRIX) proposed for marine systems by Vollenweider et al. (1998). It is also based on nutrient concentrations, chlorophyll *a* levels and Secchi disk depth, but in this case total nitrogen as well as total phosphorus are used (Table 2).

Table 2. Criteria for evaluating trophic status of marine systems (Vollenweider 1998).

Trophic Status	TN (mg m⁻³)	TP (mg m⁻³)	Chlorophyll <i>a</i> (µg L⁻¹)	Secchi Depth (m)
Oligotrophic	<260	<10	<1	>6
Mesotrophic	≥260-350	≥10-30	≥1-3	3-≤6
Eutrophic	≥350-400	≥30-40	≥3-5	1.5-≤3
Hypereutrophic	>400	>40	>5	<1.5

More recently, this approach and index has been modified by eliminating Secchi depth, incorporating dissolved oxygen saturation, and developing a single quantitative index to express trophic state (Vollenweider et al. 1998; EEA 2001). The index is applied using a dataset that spans measurements of the four variables over space and time and is calculated as follows:

$$\text{TRIX} = (1/n) \sum_{i=1}^{i=n} [(M-L) / (U - L)] \quad \text{where,}$$

n = number of variables
M = measured value of the variable
L = lower limit
U = upper limit

This approach was originally developed for use in Mediterranean waters. Preliminary attempts to use this approach for evaluation of nutrient over-enrichment in a number of northern European coastal systems by the European Environmental Agency (EEA 2001) indicated some difficulties in its use, largely related to normalizing data in a way that allows valid comparisons within and between systems. However, they concluded that the general approach used has a high potential for assessing the degree of eutrophication. Recent attempts to apply this measure in Baltic waters (Vascetta et al. 2004) have met with some success.

7.5 NOAA Index

Bricker et al. (1999, 2003) developed a nutrient and chlorophyll based index as part of a program to evaluate the degree of eutrophication for U.S. estuaries (Table 3).

Table 3. Trophic status classification based on nutrient and chlorophyll (Bricker et al. 1999).

Degree of Eutrophication	Total Dissolved N (mg L ⁻¹)	Total Dissolved P (mg L ⁻¹)	Chl <i>a</i> (µg L ⁻¹)
Low	0 - ≤0.1	0 - ≤0.01	0 - ≤5
Medium	>0.1 - ≤1	>0.01 - ≤0.1	>5 - ≤20
High	>1	>0.1	>20 - ≤60
Hypereutrophic	-	-	>60

7.6 OSPAR Comprehensive Procedure Index

OSPAR is a consortium of European Union countries that have developed a comprehensive procedure for assessment and management coastal eutrophication (Bergen Declaration 2002). In this approach the degree of nutrient enrichment is assessed by comparing five basic ecological quality elements with ecological quality objectives (Table 4).

The ecological quality elements are essentially an expression of the state of the system and the ecological quality objectives are statements of the desired state relative to reference conditions indicative of values for states of the system when anthropogenic

nutrient inputs are minimal. This approach has been adopted as a model by COAST, a group representing European Community member states and other European Countries, having the mandate of implementing the European Union's Water Directorate (Vincent et al. 2004). Details of the procedure employed in this approach are in Appendix I.

Table 4. OSPAR ecological quality elements and objectives for monitoring and assessing the biological response to nutrient enrichment based on the 2002 Bergen Declaration (modified from Painting et al. 2004).

Ecological Quality Element	Ecological Quality Objective
Phytoplankton Chlorophyll <i>a</i>	Maximum and mean chlorophyll <i>a</i> concentrations during the growing season should remain below elevated levels, defined as concentrations >50% above the spatial (offshore) and/or historical background concentration.
Phytoplankton indicator species	Specific phytoplankton eutrophication indicator species should remain below respective nuisance and/or toxic elevated levels (and increased duration).
Winter nutrient (DIN and DIP) concentrations	Winter DIN and/or DIP should remain below elevated levels, defined as concentrations >50% above salinity related and/or region specific natural background concentrations.
Dissolved Oxygen	Oxygen concentration, decreased as an indirect effect of nutrient enrichment, should remain above region-specific oxygen deficient levels, ranging from 4-6 mg L ⁻¹ .
Changes/kills in zoobenthos in relation to eutrophication	There should be no kills in benthic animal species as a result of oxygen deficiency and/or toxic phytoplankton species.

7.7 NEEA Overall Eutrophication Index (OEA)

Based on an extensive survey of U.S. estuaries carried out by NOAA under the National Estuarine Eutrophication Assessment (NEEA), Bricker et al. (1999) developed an Overall Eutrophication Condition (OEC) trophic status index based on a subset of 16 parameters (Table 5) considered to be influenced by nutrient over-enrichment. These include three 'primary' indices and three 'secondary' indices. The primary indices are all measures of plant abundance and include phytoplankton chlorophyll *a* level, epiphyte abundance, and macroalgae abundance. The three secondary indices include loss of submersed aquatic vegetation (SAV), the presence of HABs, and levels of dissolved oxygen.

Table 5. Eutrophication survey parameters (from Bricker et al. 1999). Dark shaded parameters represent primary indices; light shaded parameters represent secondary indices.

ALGAL CONDITIONS	PARAMETERS	EXISTING CONDITIONS (minimum values observed over a typical annual cycle)	TRENDS	
	CHLOROPHYLL A	Surface concentrations: Hypereutrophic (>60 µg /l) High (>20, ≤60 µg/l), Medium (>5, ≤20 µg/l), Low (>0, ≤5 µg/l), Limiting Factors to algal biomass (N, P, light, other) Spatial coverage ¹ Months of occurrence Frequency of occurrence ²	Concentrations ^{3,4} Limiting factors Contributing factors ⁵	
	TURBIDITY	Secchi disk depths: High (<1m), Medium (≥1m, ≤3m), Low (>3m) Blackwater area	Concentrations ^{3,4} Contributing factors ⁵	
	SUSPENDED SOLIDS	Concentrations: Problem (significant impact upon biological resources) No Problem (No significant impact) Months of occurrence. Frequency of occurrence ²	(no trends information collected)	
	NUISANCE ALGAE TOXIC ALGAE	Occurrence: Problem (significant impact upon biological resources) No Problem (No significant impact) Dominant species Event duration (Hours, Days, Weeks, Seasonal, Other) Months of occurrence Frequency of occurrence ²	Event duration ^{3,4} Frequency of occurrence ^{3,4} Contributing factors ⁵	
	MACROALGAE EPIPHYTES	Abundance: Problem (significant impact upon biological resources) No Problem (No significant impact) Months of occurrence. Frequency of occurrence ²	Abundance ^{3,4} Contributing factors ⁵	
	NUTRIENTS	NITROGEN	Maximum dissolved surface concentration: High (≥1 mg/l); Medium (≥0.1, <1 mg/l); Low (≥0 <0.01 mg/l) Spatial coverage ¹ Months of occurrence	Concentrations ^{3,4} Contributing factors ⁵
PHOSPHORUS		Maximum dissolved surface concentration High (≥0.1 mg/l); Medium (≥0.01, <0.1 mg/l); Low (≥0 <0.01 mg/l) Spatial coverage ¹ Months of occurrence	Concentrations ^{3,4} Contributing factors ⁵	
DISSOLVED OXYGEN	ANOXIA (0 mg/l) HYPOXIA (>0 mg/l ≤2 mg/l) BIOLOGICAL STRESS (>2 mg/l ≤5 mg/l)	Dissolved oxygen condition: Observed No occurrence Stratification (degree of influence): (High, Medium, Low, Not a factor) Water column depth: (Surface , Bottom, Throughout water column) Spatial coverage ¹ , Months of occurrence, Frequency of occurrence ²	Min. avg. monthly bottom dissolved oxygen conc. ^{3,4} Frequency of occurrence ^{3,4} Event duration ^{3,4} Spatial coverage ^{3,4} Contributing factors ⁵	
	ECOSYSTEM/COMMUNITY RESPONSE	PRIMARY PRODUCTIVITY	Dominant primary producer: Pelagic, Benthic, Other	Temporal shift Contributing factors ⁵
		PLANKTONIC COMMUNITY	Dominant taxonomic group (number of cells): Diatoms, Flagellates, Blue-green algae, Diverse mixture, Other	Temporal shift Contributing factors ⁵
BENTHIC COMMUNITY		Dominant taxonomic group (number of organisms): Crustaceans, Molluscs, Annelids, Diverse mixture, Other	Temporal shift Contributing factors ⁵	
SUBMERGED AQUATIC VEGETATION INTERTIDAL WETLANDS		Spatial coverage ¹	Spatial coverage ^{3,4} Contributing factors ⁵	
NOTES: (1) SPATIAL COVERAGE: (% of salinity zone): High (>50, ≤100%), Medium (>25, ≤50%), Low (>10, ≤25%); Very Low (>0, ≤10%) (2) FREQUENCY OF OCCURRENCE: Episodic (conditions occur randomly), Periodic (conditions occur annually or predictably), Persistent (conditions occur continually throughout the year) (3) DIRECTION OF CHANGE: Increase, Decrease, No trend (4) MAGNITUDE OF CHANGE: High (>50%, ≤100%), Medium (>25%, ≤50%), Low (>0, ≤25%) (5) POINT SOURCE(S), NONPOINT SOURCE(S), OTHER				

The indices are arranged in a matrix (Table 6) that is used to establish a numerical score ranging from 0 to 1 which is calculated using a step-wise decision method that considers quantitative and/or qualitative measures of concentration, spatial coverage, and frequency of occurrence of each symptom. In the final step used to determine a score for an estuary, secondary symptoms are given more weight since they require a longer period of time to develop suggesting that the system is more likely to be subject to chronic nutrient over-enrichment.

7.8 Assessment of Trophic Status (ASSETS)

The above approach was developed further as part of the Assessment of Trophic Status (ASSETS) procedure (Bricker et al. 2003). Data was tabulated in a relational database and incorporated into a GIS system to provide better estimates of spatial differences within a particular system. This allows calculation of weighted values which provides a better quantitative description of the symptoms. ASSETS also incorporates a statistical procedure based on percentiles to establish quantitative values that lessens the influence of outliers in the database.

7.9 Estuarine Health Index (EHI)

Cooper et al. (1994) developed an Estuarine Health Index (EHI) that was designed to synthesize multidisciplinary information in an easily understandable form for a series of estuaries in Natal (RSA). The index is based on a three part classification of estuarine conditions that assesses the water quality and biological and physical characteristics of an estuary. Although the approach is comprehensive, it is quite complex, requires an extensive database and appears to work well only within a regional dataset (Ferreira 2000).

7.10 EQUATION (Estuarine QUALity and CondiTION)

EQUATION is an integrated estuarine quality and condition index based on the four characteristics illustrated in Table 7 (Ferreira 2000). Using a decision support system, a coastal system is evaluated on a scale of 1 (poor) to 5 (good).

This approach was tested using five coastal systems, two in the U.S and three in Europe, having a broad range of physical characteristics and was found to work quite well. Its major advantage over other indices is that it considers a broad range of factors in assessing trophic state. This however, requires a comprehensive data base and it will only be useful for coastal systems that have been well studied. The decision support system used in this approach is at <http://tejo.dcea.fct.unl.pt/bar-ca.htm>.

This is one of the few indices that propose to include an assessment of the state of the benthic community. Jorgensen (1996) has pointed out that benthic communities are one of the most sensitive parts of coastal systems experiencing nutrient over-enrichment and argues that it should be included in any eutrophication assessment program. Diaz and Rosenberg (1995) have reviewed the responses of benthic communities to hypoxic conditions.

Table 6. Matrix used in the NEEA approach to determine overall level of eutrophic conditions (from Bricker et al. 1999).




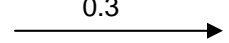
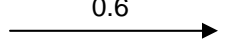
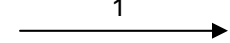
Overall Level of Expression of Eutrophic Conditions				
High Primary Symptoms		MODERATE Primary symptoms high with more serious secondary symptoms	MODERATE HIGH Primary symptoms high and substantial; secondary symptoms becoming more expressed, indicating potentially serious problems	HIGH High primary and secondary symptoms levels indicate serious eutrophication problems
Moderate Primary Symptoms		MODERATE LOW Primary symptoms beginning to indicate possible problems but still very few secondary symptoms expressed	MODERATE Level of expression of eutrophic conditions is substantial	HIGH Substantial levels of eutrophic conditions occurring with secondary symptoms indicating serious problems
Low Primary Symptoms		LOW Level of expression of eutrophic conditions is minimal	MODERATE LOW Moderate secondary symptoms indicate substantial eutrophic conditions, but low primary symptoms indicates other factors may be involved in causing the conditions	MODERATE HIGH High secondary symptoms indicate serious problems, but low primary symptoms indicates other factors may involved in causing conditions
	0 			
		Low Secondary Symptoms	Moderate Secondary Symptoms	High Secondary Symptoms

Table 7. EQUATION index components, objectives and data requirements (from Ferreira 2000).

Component	Objectives	Data requirements for descriptors
Vulnerability	Quantify system buffering capacity	Physiography (surface area, river inflow, tidal range, tidal regime, communication with ocean)
Water quality	Determine trophic balance based on nutrients, primary productivity and oxygen	Watershed information (population, watershed area); Primary production and nutrient indicators (mean chlorophyll <i>a</i> , yearly net primary production, mean nutrient concentration in river discharge); Reference parameters (mean salinity, temperature and dissolved oxygen)
Benthic Quality	Evaluate status of benthos in terms of biological communities, contamination, and bioaccumulation	Sediment contamination (estimated area affected); Bioaccumulation (excess over reference values); Benthic biomass and diversity, and equilibrium between epi/infauna (heuristic data, e.g., high/medium/low, present/absent)
Trophodynamics	Assess trophic web equilibrium based on ichthyofaunal data	Fishing and aquaculture activity; Quality of fish products; Fish diversity; Nursery areas (heuristic data, e.g., high/medium/low, present/absent)

7.11 DPSIR (Driving forces, Pressures, State, Impact, Responses) Index

The European Environmental Agency (EEA 2001) is developing an eutrophication index based on an assessment of parameters that represent driving forces, pressures, system state, impact, and responses (Fig. 2). The assessment of driving forces is based on the sources and levels of nutrient inputs to the system. The pressures are an assessment of the increases in human activities that may lead to eutrophication such as urbanization and agriculture. The state of the system involves an assessment of the degree to which the system has become nutrient over-enriched and includes an evaluation of nutrient concentrations, chlorophyll *a* levels, and dissolved oxygen levels. Assessment of impacts includes presence of HABs, fish mortality, and SAV loss. Responses involve determining what type of monitoring is required to better understand the system and the management activities required to alleviate the problem.

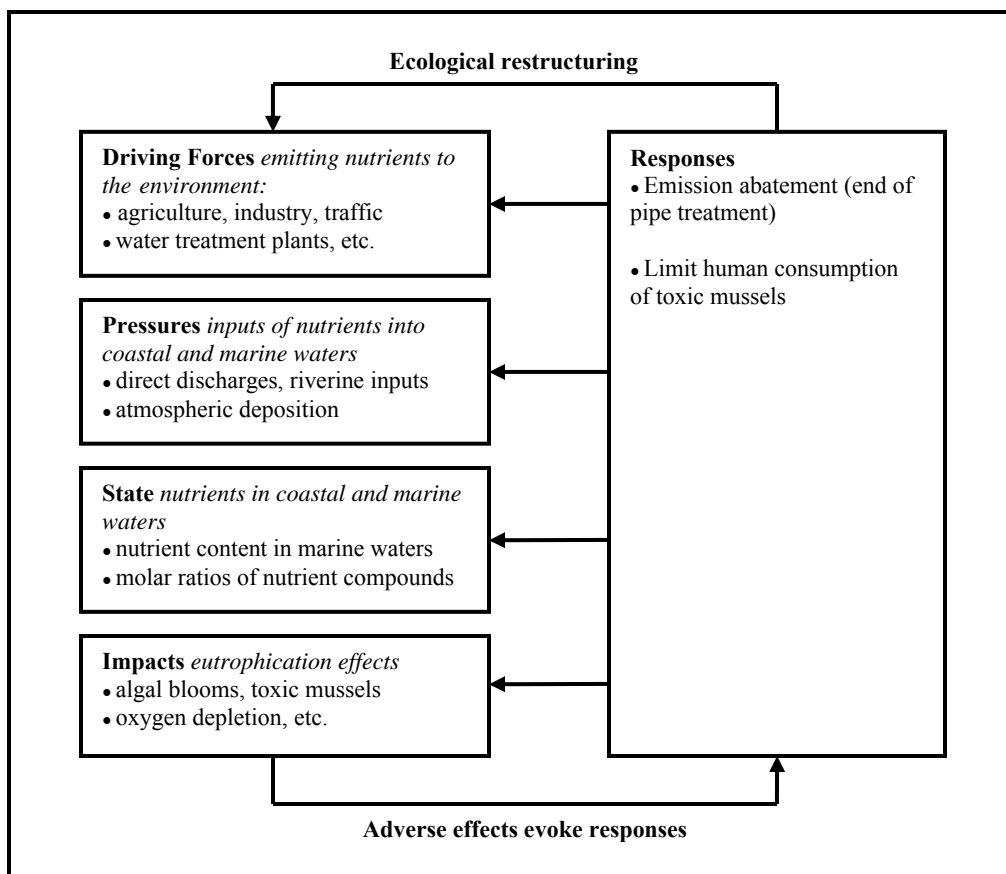


Figure 2. DPSIR framework for eutrophication in coastal waters (from EEA 2001).

7.12 OAERRE Index

The Oceanographic Applications to Eutrophication in Regions of Restricted Exchange (OAERRE) project is, in part, an attempt to develop a screening model for determining the trophic status of coastal systems in which hydrodynamic processes are restricted by the morphological characteristics of the system such as outer sills and sand bars (Tett et al. 2003). The model is based on one originally developed by the United Kingdom's Comprehensive Studies Task Team (CSTT 1997). It is a simple box model that requires information on nutrient input rates, chlorophyll yield per unit of dissolved available inorganic nitrogen, a number of light and nutrient related parameters that determine phytoplankton growth rate, and phytoplankton losses due to grazing and flushing. Its major shortcoming with respect to its general applicability is that it requires an estimate of phytoplankton yield relative to nitrogen concentration for the specific types of phytoplankton populations within the system, and this has to be determined based on laboratory mesocosm experiments (Gowen 1994).

7.13 PNCERS Index

The Pacific Northwest Coastal Ecosystems Regional Study (PNCERS) has begun development of a comprehensive set of indicators to assess the health of U.S. west coast estuaries (Parrish and Litle 2001). The indicators are grouped into three components: the physical environment, the biological system, and the socio-economic system. Its uniqueness relative to other proposed sets of indicators is in inclusion of socio-economic parameters and detailed consideration of the relevance of each indicator to particular types of estuaries based on a physical classification.

7.14 Indices Based on Phytoplankton Community Composition

Nutrient over-enrichment typically results in quantitative and qualitative changes in phytoplankton communities. Karydis and Tsirtsis (1996) examined 12 ecological indices commonly used to describe ecological communities in term of species diversity, abundance, evenness, dominance, and biomass and applied them to a data set on phytoplankton communities collected from coastal systems experiencing a range of degrees of eutrophication. Five of the indices were found to be effective in distinguishing oligotrophic, mesotrophic, and eutrophic systems. This approach, however, is quite narrow in scope and requires a substantial database on phytoplankton quantity and species composition.

8. Factors that Determine Susceptibility to Nutrient Over-enrichment

The level of nutrient input that can be assimilated by a coastal system before it begins to exhibit symptoms of nutrient over-enrichment is often referred to as its ‘assimilation capacity’. Management of nutrient over-enrichment requires at least an approximate estimate of what this capacity is for any particular system, and this topic has received considerable discussion in the literature (Bricker et al. 1999; NRC 2000; USEPA 2001a).

The assimilation capacity of a coastal system depends on a number of factors related to how it processes nutrients once they enter the system. In general, the primary factors include: the extent to which the nutrients become diluted; the amount of time the nutrients remain in the system; and the natural ability of the system to process the nutrients in terms of transforming them into forms that may or may not be available to primary producers (e.g., decomposition, recycling, denitrification, sequestration by settling) (USEPA 2004). These factors, in turn, depend on a number of coastal system characteristics of which the following have been suggested to be most important: physiographic setting, morphology, hydrodynamics, water column stratification characteristics, turbidity characteristics, and the types of biological communities present. Many of these characteristics are related to one another either directly or indirectly. The following is a brief description of the characteristics considered to be most important in determining assimilation capacity.

8.1 Physiographic Setting

The physiographic setting of the system includes its climatic conditions, type of coastal system (e.g., drowned river valley, embayment, coastal lagoon, fjord, etc.), the productivity base (i.e., freshwater wetlands, salt marshes, submerged macrophytes, phytoplankton, etc.), and morphology (watershed/waterbody surface area ratio, depth, surface area, shoreline indentation, presence of sills or other restrictions to offshore mixing, and exposure to wind and wave action). These factors are important in determining dilution of nutrient inputs, nutrient retention rates, flushing rates, vertical water column mixing, and biological nutrient uptake rates.

8.2 Dilution

Dilution refers to the capacity of the coastal system to dilute nutrient inputs and is a function of a number of factors such as morphology, volume of freshwater input, volume of the coastal system, and tidal amplitude.

8.3 Water Residence Time

Water residence time is a measure of the length of time water entering the system remains before it is flushed out of the system. Its reciprocal, turnover rate, or flushing rate, is the fraction of the water contained in a waterbody that is replaced per unit of time. Both of these terms are commonly used to describe the flushing characteristics of a system. The factors most important in determining residence time are basin morphology, freshwater inputs, tidal amplitude, and winds.

Estuaries having high flushing rates are generally less susceptible to nutrient over-enrichment because nutrients are retained in the estuary for relatively short periods of time making it less likely that algal blooms will occur (Bricker et al. 1999; NRC 2000). In addition, some forms of nutrients entering the system that are not readily available to phytoplankton are less likely to be retained in the system long enough to be transformed into an available form (Nixon et al. 1996).

8.4 Water Column Stratification

Stratified coastal systems are generally more susceptible to eutrophication for a number of reasons. The separation of freshwater and seawater into two masses that are relatively isolated inhibits the transfer of dissolved oxygen into bottom layers increasing the probability of the development of low dissolved oxygen concentrations in bottom waters. Stratification also increases the chances that a phytoplankton bloom will develop as a result of decreasing the depth of vertical mixing and lessening the chance that phytoplankton will sink below water depths where there is insufficient light for photosynthesis to occur. In systems where the major nutrient inputs are from offshore

waters, however, stratification may actually lessen the chance of a phytoplankton bloom occurring since the nutrient rich oceanic bottom waters may remain below the euphotic zone.

8.5 Turbidity

Some coastal systems are light limited as opposed to nutrient limited. The high turbidity can be caused either by extensive land erosion within the watershed or by resuspension of sediments in tidally energetic environments. The upper reaches of the Bay of Fundy is a good example of the latter. These systems are often unstratified and may have euphotic zone depths as shallow as a few centimetres making them relatively unresponsive to nutrient loading (Cloern 1987; LePape et al. 1996).

8.6 Biological Communities

The types of biological communities present in a system can greatly influence the degree to which eutrophic conditions develop by retaining nutrients within the system, grazing of phytoplankton and reducing the development of high phytoplankton concentrations, or transforming nutrients into forms that are more easily sequestered or become unavailable to primary producers.

Most coastal systems having low nutrient inputs are dominated by benthic plant communities while those having high nutrient loads are dominated by phytoplankton communities (Vollenweider et al. 1992) and/or ephemeral macroalgal blooms such as sea lettuce. The presence of filter feeding benthic communities, including those associated with shellfish aquaculture operations, can reduce the chance of development of phytoplankton blooms (Cloern 1982; Alpine and Cloern 1992; Meeuwig et al. 1998). Some aquaculture activities, such as those used in suspended mussel aquaculture, can potentially have a significant impact on nutrient cycling in coastal environments by diverting nutrients from the water column to the benthos through deposition of particulate fecal material and pseudofaeces.

In shallow systems where light does not become limiting for benthic plants, high nutrient loading rates may result in extensive development of fast growing benthic macrophytes such as sea lettuce (*Ulva* sp.). Excessive growth of sea lettuce, which is quite common in Prince Edward Island where most estuaries are relatively shallow, can greatly accentuate the problems associated with nutrient over-enrichment. Because sea lettuce is rooted within the substrate, unlike phytoplankton it and the nutrients it contains are not flushed out of the estuary and, even in systems with high flushing rates, this causes an increased level of nutrient entrainment within the estuary. In many cases, during summer months when the biomass of sea lettuce is greatest and respiration rates are highest due to the warm water, dissolved oxygen concentrations often decrease to zero during overnight hours resulting in massive die offs of the sea lettuce and prolonged anoxic conditions. This in turn can lead to the death of shellfish, crabs, and other bottom dwelling

organisms. Often this situation results in the death of the sea lettuce itself which then becomes easily uprooted and deposited along shorelines creating both unsightly and foul smelling conditions.

Biological communities that either create environmental conditions necessary for denitrification to occur, or contain organisms that carry out denitrification, can have a large influence on water column nitrogen concentrations (Sowles 2003). This becomes more important as water residence time increases (Seitzinger and Giblin 1996; Dettmann 2001). The presence of salt marshes and benthic communities that have high rates of bioturbation are thought to be especially important in this respect (Pelegri et al. 1994).

9. Indices of Susceptibility

In contrast to the numerous approaches that have been developed to assess the degree to which a coastal system exhibits symptoms of nutrient over-enrichment, comparatively few approaches have been developed to assess the assimilation capacity of a coastal system. Those indices that do exist focus largely on dilution and flushing characteristics. The following describes some of the approaches that have been developed to determine the susceptibility of a coastal system to nutrient over-enrichment.

9.1 Overall Level of Human Influence (OHI)

Bricker et al. (1999, 2003) have developed an index of susceptibility based on nutrient dilution and export potential, and the potential for increased nutrient inputs (nutrient pressures) as a result of human activity. The dilution potential is based on the volume and stratification characteristics of the estuary. The export potential is based on flushing rate and is determined by tidal amplitude and the ratio of freshwater inflow and estuary volume. The potential for increased nutrient input is assessed on demographic trends, expected changes in land use and plans for remediation. Tables 8 to 11 illustrate how dilution and export potential are determined.

Table 8. Determination of estuary dilution potential (modified from Bricker et al. 1999).

Stratification Type	Dilution Volume	Dilution Value	Dilution Potential
Vertically Homogenous	1/ Volume of Estuary	10^{-13} 10^{-12}	HIGH
Minor Vertical Stratification	1/Volume of Estuary	10^{-11}	MODERATE
Vertically Stratified	1/ Volume Freshwater Fraction	10^{-10} 10^{-09}	LOW

Table 9. Determination of estuary flushing potential (modified from Bricker et al. 1999).

Tidal Range (m)	Freshwater Inflow/Estuary Volume	Flushing Potential
Macro (>6)	Large or moderate (10^{00} to 10^{-2})	HIGH
Macro (>6)	Small (10^{-03} to 10^{-04})	MODERATE
Meso (>2.5)	Large (10^{00} to 10^{-1})	HIGH
Meso (>2.5)	Moderate (10^{-02})	MODERATE
Meso (>2.5)	Small (10^{-03} to 10^{-04})	LOW
Micro (<2.5)	Large (10^{00} to 10^{-01})	HIGH
Micro (<2.5)	Moderate (10^{-02})	MODERATE
Micro (<2.5)	Small (10^{-03} to 10^{-04})	LOW

Once the values for each index are determined, the final assessment of susceptibility is arrived at using the matrix in Table 10.

Table 10. Matrix for determining level of susceptibility of an estuary to nutrient over-enrichment based on dilution and flushing potential (modified from Bricker et al. 1999).

Level of Susceptibility				
Flushing Potential	High	Low Susceptibility	Low Susceptibility	Moderate Susceptibility
	Mod	Low Susceptibility	Moderate Susceptibility	High Susceptibility
	Low	Moderate Susceptibility	High Susceptibility	High Susceptibility
		High	Moderate	Low
		Dilution Potential		

The level of overall human influence is determined by evaluating the susceptibility based on flushing and dilution potential with the potential for the system to become subjected to increasing levels of nutrient inputs.

Table 11. Matrix for determining level of overall human influence based on susceptibility and nutrient inputs (modified from Bricker et al. 1999).

Overall Level of Human Influence			
High Susceptibility	MODERATE Even low nutrient additions may result in problem symptoms in these estuaries	MODERATE HIGH Symptoms are moderately to highly related to nutrient additions	HIGH Symptoms are probably closely related to nutrient additions
Moderate Susceptibility	MODERATE LOW Symptoms are minimally to moderately related to nutrient inputs	MODERATE Symptoms are moderately related to nutrient inputs	MODERATE HIGH Symptoms are moderately to highly related to nutrient inputs
Low Susceptibility	LOW Symptoms are likely predominantly naturally related or caused by human factors other than nutrient additions	LOW are predominantly naturally related or caused by factors other than nutrient additions	MODERATE LOW Symptoms may be naturally related or the high level of nutrient additions may cause problems despite low susceptibility
	Low Nutrient Input	Moderate Nutrient Input	High Nutrient Input

9.2 Eutrophication Risk Index (EUTRISK)

A Eutrophication Risk Index (EUTRISK) for European coastal areas is currently being developed by the Joint Research Centre of the European Union (Druon et al. 2002). This index is based on remote sensing measurements of chlorophyll in surface waters combined with an index of physically sensitive areas (PSA). The latter is evaluated on the basis of physical factors related to the dilution and flushing potential of the coastal system (Fig. 3).

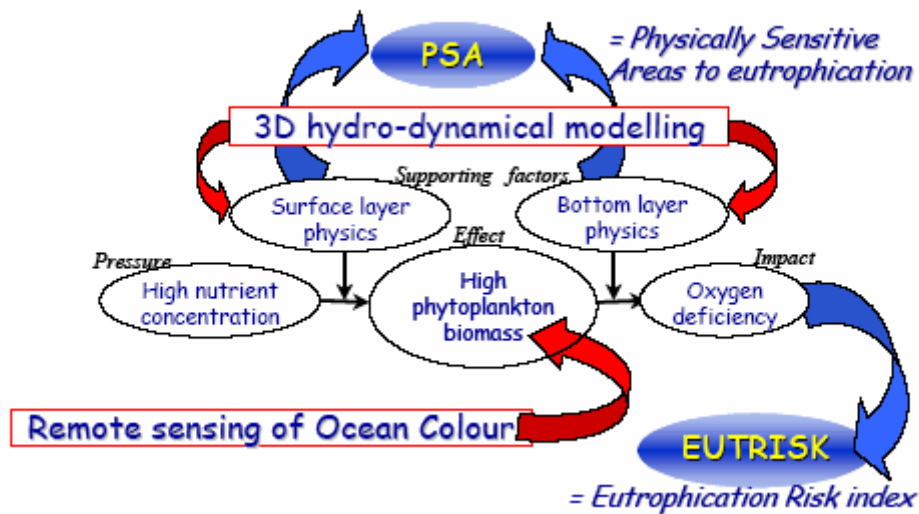


Figure 3. Variables and their interrelationships used in the EUTRISK and PSA indices (from Druon et al. 2002).

The EUTRISK and PSA indices separate coastal systems into the following categories:

- Hypertrophic and sensitive (recurrent anoxia)
- Hypertrophic and resistant (only exceptional severe hypoxia)
- Hypertrophic and hyper-resistant (light hypoxia)
- Eutrophic and sensitive (aperiodic anoxia or severe hypoxia)
- Mesotrophic and hypersensitive (recurrent anoxia)

9.3 EQUATION

The system vulnerability component of EQUATION (Ferreira 2000) is essentially an evaluation of a coastal system's ability to assimilate nutrients. It is calculated as a function of freshwater residence time, estuary number, coastal exchange, and the proportion of time the system is closed to the ocean.

Freshwater residence time is computed as the ratio of freshwater volume in the estuary (based on mean salinity) and freshwater inflow. The estuary number is an index of vertical water column stratification and is calculated as the ratio of freshwater inflow to tidal prism volume. Coastal exchange represents the degree to which water in the estuary mixes with offshore oceanic water and is calculated as the ratio of the tidal prism volume to the volume of the estuary. The proportion of time the system is closed to mixing with oceanic water is used to modify the previous vulnerability components using a heuristic matrix. Table 12 lists the scoring system for the first three components.

Table 12. Grading system for assessing coastal system vulnerability to nutrient over-enrichment (from Ferreira 2000).

Grade	Residence Time (days)	Estuary Number (%)	Coastal Exchange
5 (good)	< 10	≤ 1	≤ 1
4	< 20	≤ 10	≤ 10
3	< 30	≤ 25	≤ 35
2	< 40	≤ 100	≤ 70
1 (poor)	≤ 40	> 100	> 70

These scores are then incorporated into a classification matrix which calculates an average value that is used to determine the final index.

9.4 Other Indices

Most of the above indices of susceptibility deal with factors that influence the availability of nutrients. There exist, however, highly turbid systems and tidal and freshwater brackish areas in estuaries that are light, rather than nutrient, limited (Flemer 1970; Peterson et al. 1987; Sin et al. 1999). Cloern (1999) has developed a simple index of susceptibility that includes factors related to both nutrient and light availability. Although this index is useful in determining potential phytoplankton growth rates under varying light and nutrient regimes, it does not take into account other factors, such as flushing and dilution, which are also important in determining the concentration of phytoplankton.

10. Approaches to Development of Nutrient Criteria

Despite considerable effort, there has been little success in attempts to develop general nutrient criteria guidelines that indicate the absolute nutrient concentration a coastal system can have before it begins to exhibit symptoms of nutrient over-enrichment. The reasons for this have been discussed in Section 9 and are related to the complexity of factors that determine how coastal systems respond to nutrient inputs. The result is that most attempts to arrive at nutrient criteria depend on site specific evaluations which typically require extensive monitoring, the development of large datasets, and mathematical models specific to the site being studied.

Numerous agencies in the U.S., Europe, and Australia have begun the process of developing guidelines for establishing nutrient criteria for particular coastal systems. All of these efforts have common goals and many elements of the procedures are common to all. In most cases, the procedure involves categorizing regional coastal systems

according to their susceptibility to nutrient over-enrichment, and then establishing sets of reference conditions for each category based on information obtained from coastal systems that are considered to be relatively pristine and not impacted by nutrient over-enrichment. The analysis of data and decisions on appropriate values for reference nutrient levels for any particular system are typically made by teams of local experts. The following section reviews approaches for arriving at nutrient criteria that have been developed by various agencies. In most cases these approaches have only recently been developed and have not been subjected to the testing required to determine how well they will work in establishing nutrient criteria.

10.1 European Initiatives

A number of agencies within the European community have developed guidelines for assessing and managing eutrophication problems in European coastal waters. Major European conventions that address estuarine eutrophication are OSPAR (in the North East Atlantic), HELCOM (in the Baltic Sea), and BARCOM (in the Mediterranean Sea).

The European Union Member States, Norway, and the European Commission have jointly initiated the development of a Common Implementation Strategy in an attempt to establish a more uniform and harmonized set of criteria that would be adopted by all European agencies (Vincent et al. 2004). The approach being proposed is to develop a basic template of physical, chemical, and biological elements for assessment of nutrient over-enrichment and to link this to particular coastal system typologies. This would then be used by each member state to establish reference conditions for each typology based on the coastal systems specific to each country. The approach that is being proposed for adoption is essentially that previously developed by OSPAR (1997, 2001) and is described in Appendix I.

The initial step is to develop a typology of systems present in the member state's country. This forms the basis for development of a set of reference conditions, as well as programs for monitoring and reporting, for each specific typology identified. The parameters suggested for use in determining typologies are largely physical characteristics and include tidal range, wave exposure, depth, salinity, stratification characteristics, proportion of intertidal area, water residence time, substratum type, current velocity, and duration of ice coverage.

The reference conditions represent what the biological state of the system would be if there were no symptoms of nutrient over-enrichment present. It is suggested they be quantitative whenever possible, indicative of natural variability, and summarize the range of values for each reference condition parameter. The reference conditions are then used to evaluate the ecological quality of a particular system expressed as an Ecological Quality Ratio ranging from 0 to 1, with higher numbers representing better ecological quality.

The parameters used to determine ecological quality are largely indices of the status of the pelagic and benthic communities in the system. Suggested approaches to establishing reference conditions include analysis of data on existing undisturbed sites, the use of predictive or hindcasting models or expert judgement.

This approach is still in the development stage and, although a few pilot studies have been carried out by a number of member countries, the degree to which this procedure will work for all coastal systems within the participating countries has not yet been fully evaluated.

10.2 U.S. Environmental Protection Agency

The U.S. Environmental Protection Agency has developed a comprehensive technical guidance manual for establishing nutrient criteria in estuarine and coastal marine waters based on a reference condition approach (USEPA 2001a). The procedure employs a stepwise sequence of actions to arrive at nutrient criteria for a specific body of water. The nutrients emphasized are total nitrogen and total phosphorus. The establishment of reference conditions is carried out by a panel of regional specialists using information on existing pristine systems, knowledge of historical conditions, and models.

The basic steps in this procedure are as follows:

- Establish regional technical assistance groups
- Delineate nutrient ecoregions/coastal provinces appropriate to development of criteria
- Determine scientific basis for criteria development
- Define a physical classification system for the coastal systems of concern
- Select key indicator variables for assessing state of the coastal system
- Determine database requirements and assess availability
- Establish reference conditions
- Develop criteria
- Define management response

This is the most well developed and documented procedure available for establishing nutrient criteria. It was prepared to guide the efforts of State/Tribal and Federal agency personnel having the responsibility for developing nutrient criteria and, although the emphasis is on U.S. coastal systems, it is generic enough to have much wider applicability.

10.3 Australia and New Zealand

The Australian and New Zealand Environment and Conservation Council (ANZECC) has developed guidelines for nutrients in estuarine and marine waters that are also based on a reference condition approach using regional experts (ANZECC 2000). They recommend that the information used to establish reference conditions be based on at least bimonthly

monitoring over a minimum period of two years, but preferentially 5 to 10 years, and that for physical and chemical parameters, the 20th and 80th percentiles of the reference data should form the basis of the guidelines, and that the median value for the site should fall within this range. For systems that have not been previously studied, and for which some evaluation must be made, they have developed recommended default values of numerous indicators for different areas of the country. Table 13 lists the basic steps recommended in developing the guidelines.

Table 13. Determining appropriate guideline trigger values (from ANZECC 2000).

<p style="text-align: center;">I. Define Primary Management Aims</p> <ul style="list-style-type: none"> • Define the water body (scientific information, monitoring data, classify ecosystem type) • Determine environmental values to be protected • Determine level of protection • Identify environmental concerns <ul style="list-style-type: none"> e.g. - toxic effects - nuisance aquatic plant growth - maintenance of dissolved oxygen - effects due to changes in salinity • Determine major natural and anthropogenic factors affecting the ecosystem • Determine 'management goals' <ul style="list-style-type: none"> - often defined in biological terms
<p style="text-align: center;">II. Determine appropriate Guideline Trigger Values for selected indicators</p> <ul style="list-style-type: none"> • Determine a balance of indicator types (based upon level of protection and local constraints) • Select indicators relevant to concerns and goals • Determine appropriate guideline trigger values (low risk concentration of contaminants/stressors; may depend on level of protection) • Determine specific indicators to be applied
<p style="text-align: center;">III. Apply the Trigger Values using (risk-based) Decision Trees or Guideline 'packages'</p> <ul style="list-style-type: none"> • Water quality monitoring data • Site specific environmental information • Effects of ecosystem-specific modifying factors. <ul style="list-style-type: none"> - biological assessment - physical and chemical stressors - toxicants - sediments)

This approach is currently in the process of being tested in various regions of Australia to evaluate its practicality.

11. Role of Models

Mathematical models have been used to describe, understand, and forecast trends for many types of biological systems. The literature contains a wide diversity of models that focus on eutrophication of coastal systems. Existing models range from simple empirical statistical regression models to complex computer simulation models. Few of these models, however, are able to deal with all of the responses associated with nutrient over-enrichment, and none have been accepted for general use for all coastal systems. However, based on the successes achieved so far, it seems reasonable that sets of models, each specific to a particular type of coastal system, can be developed to provide useful management tools. Especially important in this respect are recent advances in the development of simple numeric hydrodynamic circulation models that require only readily available data such as bathymetry, tidal forcing, and freshwater discharge.

The variables most often predicted by eutrophication models are nutrient concentrations, phytoplankton biomass, and (in some cases) dissolved oxygen concentration. There has also been considerable effort and success in development of models capable of predicting changes in estuarine macroalgae as a result of nutrient over-enrichment (e.g., Coffaro and Sfriso 1997; Alvera-Azcarate et al. 2003)

11.1 Regression models

There have been numerous attempts to determine if the nutrient concentration in a coastal system can be correlated with symptoms of nutrient over-enrichment. The most common relationship sought after is that between nitrogen concentration and phytoplankton biomass, the latter typically being expressed as chlorophyll *a* concentration. There are many reports showing a good relationship between the two (e.g., Nixon and Pilson 1983; Gowen et al 1992; Monbet 1992). The range of response, however, is large and in some cases, particularly when the dissolved inorganic form of nitrogen is considered, the correlation can be negative as a result of dissolved nutrients being rapidly incorporated into the phytoplankton (Gowen et al. 1992; Edwards et al. 2003). Meeuwig et al. (1998) suggests that significant nutrient-chlorophyll correlations should only be expected for systems having water residence times longer than the specific growth rate of phytoplankton.

Some studies have found correlations between nitrogen loading as opposed to nitrogen concentration and phytoplankton biomass (Gowen et al. 1992; Janicki and Wade 1996). Borum (1996), however, found a poor relationship between these variables in an analysis that included 51 estuaries which lead Cloern (2001) to conclude that nitrogen loading alone is a poor predictor of phytoplankton production and abundance.

When assessing the use of these types of relationships, it is important to realize that not all coastal systems are limited by nutrients. Other factors, especially turbidity and its influence on light availability, may be more important than nutrients in limiting algal production. Monbet (1992), for example, in a study of 40 estuaries, found a good

relationship between nitrogen loading and algal biomass for microtidal systems, but a poor relationship for macrotidal system. He attributed this difference to the influence of turbidity on light availability generated by high tidal energy in the macrotidal systems.

11.2 Multivariate Models

There have been some attempts to apply multivariate approaches to predict variables representing symptoms of eutrophication. Lowery (1998), for example, had some success in using a multinomial logistic regression model based on nitrogen and phosphorus load, N:P ratio and salinity stratification, to predict the oxygen status of an estuary as normoxic, borderline hypoxic, hypoxic, and severely hypoxic.

Strain and Yeats (1999) carried out a multivariate principle components analysis to evaluate the relationships between nutrients, dissolved oxygen, and trace metals in 34 Nova Scotia coastal inlets. The results indicated that phosphate, ammonia, silicate, iron, manganese, and dissolved oxygen were all closely related to each other. Further analyses indicated that these factors were in turn correlated to the presence of sills, but not to anthropogenic nutrient inputs, tidal flushing times or other morphometric features of the inlets.

11.3 Watershed Export-Coefficient Models

Limnologists and lake managers have had considerable success in applying the Dillon-Rigler Export modelling approach (Dillon and Rigler 1975) to prediction of phosphorous concentration in lakes. These are regression based models that rely on export coefficients and simple empirical formulations to predict nutrient concentration and various other water quality parameters. This approach is based on the formulations developed by Vollenweider and Kerekes (1982) for predicting mean annual lake phosphorus concentration based on annual phosphorus loading, sedimentation rates and the flushing characteristics of the lake. Many believe that this approach can not be adopted for coastal systems because of the greater complexity of processes in coastal systems, especially those related to the numerous forms of nitrogen present, the transformations between these forms, and the more complex hydrodynamics influencing stratification and flushing rates (Billen and Garnier 1997; NRC 2000). Despite these complexities, there has been some success in using this approach and it appears to offer considerable promise. Recent attempts to apply the Dillon-Rigler approach to coastal systems have produced promising results. Dettmann (2001) developed a simple Vollenweider type model to estimate the proportion of total nitrogen loading either lost or exported from an estuary. A test of the model on 11 North American and European estuaries fit the data very well. Using a similar approach, Valiela et al. (2004) developed a model (NLOAD) to predict nitrogen concentration in estuarine systems based on loading rates and estimates of physical and biological transformation of nitrogen and flushing rates. It was applied to seven Cape Cod estuaries with good success. These examples appear to offer an excellent compromise between simple regression models and more complex, data intensive

simulation models. There is still, however, a requirement to quantitatively link the model predictions to variables indicative of nutrient over-enrichment.

Others studies that have had some success in using this approach include: Meeuwig (1999) who was able to predict chlorophyll *a* as a function of land use for 15 estuaries in Prince Edward Island using phosphorus concentration; Meeuwig et al. (2000) for 19 Finnish estuaries; and Johnes (1996), Whitehead et al. (1999), and Uncles et al. (2002) for a number of Great Britain estuaries.

11.4 Simulation Models

A large number of computer simulation models have been developed for coastal systems ranging from simple one and two dimensional models to complex three dimension models. Often these models are coupled hydrologic and biological models that include transport and stratification processes. They differ from mass balance models in having a greater emphasis on processes and some have been developed mainly to better understand how coastal systems operate in general. Few of these models deal well with prediction of changes in higher trophic levels and there appear to be no models that deal with toxic and nuisance algae. In recent years there have been numerous attempts to couple hydrologic-biological simulation models to watershed models with a focus on the anthropogenic activities within a watershed in an attempt to establishing management or remediation plans (Hopkinson and Vallino 1995).

Because simulation models typically require a great deal of time and resources to construct and validate, and tend to be site specific, this effort can not usually be justified except in those instances where a coastal system has a high value, is severely impacted and potential remediation costs are high.

Numerous surveys of the many existing simulation models available have been carried out comparing the resource requirements, strengths, weaknesses, and applicability for dealing with nutrient over-enrichment (e.g., Jorgensen 1994; USEPA 1996, 2001a).

12. Water Quality Guideline Derivation

The Protocol for the Derivation of Guidelines for the Protection of Aquatic Life (CCME 1991) is intended to deal specifically with toxic substances, and provide numerical limits or narrative statements based on the most current, scientifically defensible toxicological data. Nutrients do not fit this model because they are, in general, non-toxic to aquatic organisms at levels and forms present in the environment. However, secondary effects, such as eutrophication and oxygen depletion are serious concerns. Because aquatic communities are generally adapted to ambient conditions and the factors that determine the assimilation capacity of a coastal system and its response to nutrient inputs are not well understood, it is neither feasible nor desirable to establish generic nutrient criteria that are applicable to all types of coastal systems. Some of the effects of nutrients are

aesthetic and thus include an element of subjectivity. What is considered nuisance plant growth to some may be desirable to others. Based on these realities, a framework based approach that is consistent with CCME guideline principles is recommended.

13. Nutrient Guidance Framework for Canadian Nearshore Waters

13.1 Introduction

The similarities between the various organizations that have developed, or are in the process of developing, nutrient guidance criteria for coastal waters are greater than the differences. As previously discussed in Sections 9 and 10, our understanding of the factors that determine a coastal system's assimilation capacity and response to nutrient inputs currently restricts our ability to establish general nutrient criteria that would be applicable to all types of coastal systems. As a result, the reference condition approach appears to be the most practical and widespread approach at present. This approach is essentially an empirically based approach that compares the state of a number of similar systems, or historical differences in a single system, in order to recognise the differences between waterbodies, or over time, in terms of their response to nutrient inputs. The USEPA (2001a) defines a reference condition as "... the comprehensive representation of data from several similar minimally impacted, 'natural' sites on a waterbody or from within a similar class of waterbodies, i.e., median values of TN, TP, chlorophyll a, or Secchi depth."

The details and procedures for applying the reference condition approach have been most extensively developed and documented in a technical guidance manual prepared by the USEPA (2001a).^{*} Other national and international agencies are largely still in the development stage of establishing nutrient guidelines for nearshore water and have not yet put forward a comprehensive procedural framework comparable to that developed by the USEPA. The USEPA document contains a comprehensive guidance framework that progresses from data collection to nutrient criteria development. The procedures for developing nutrient criteria are presented in a comprehensive step-wise manner. Detailed guidelines and extensive background information is presented for each step of the process. Development of a similar guidance manual specific to Canadian nearshore systems providing comparable detail on the procedure and its application would be a major undertaking, and that it is recommended that this approach be adopted for Canadian nearshore waters. In addition, although the USEPA guidelines have been developed for U.S. estuaries, the approach is generic enough to be applied to Canadian coastal waters without any severe restrictions. Another important advantage to adopting the USEPA guidelines is that it presents alternative procedures for developing nutrient criteria in those instances where no pristine sites exist. In addition, although it is strongly recommended that the approach be carried out by a multidisciplinary team of experts to ensure that the best effort is made in establishing nutrient guidelines, in those instances

^{*} Although numerous other agencies have are in various stages of developing a procedure manual, only the USEPA has completed this process. The USEPA Technical Guidance Manual for establishing nutrient criteria is available at <http://www.epa.gov/waterscience/criteria/nutrient/guidance/marine/index.html>

where guidelines are required but time and financial resources are limited, it is possible to apply the approach on a more limited scale to develop interim guidelines subject to revision once adequate data become available.

The USEPA document also provides details on how one may go about developing estuarine classification systems, direction on how to establish monitoring systems in those instances where sufficient data are not available, and suitable field sampling designs and laboratory methodologies for development of the required databases. Also included are suggestions for the design of remediation programs for impacted systems.

One major shortcoming of adopting the USEPA framework for Canadian nearshore systems, however, is the greater proportion of fjord type systems present along Canadian coastlines and the lack of specific attention to these types of coastal systems in the USEPA document. The problems associated with development of nutrient criteria for fjords are discussed in Section 16.

The basic approach described in the USEPA guidelines is illustrated in Figure 4**. The following sections provide a general overview of each major step of the process, but do not provide all of the information necessary to successfully develop nutrient criteria. A much more detailed and comprehensive discussion of each step is contained in the USEPA (2001a) guideline document which should be consulted while establishing nutrient criteria for a particular nearshore system.

It is also important to realize that other, more novel, approaches, not specifically described in the USEPA guidelines, may be used to establish nutrient criteria when data limitations prevent the use of the more typical approaches. This may be the case for sites that are exceptionally unique and/or for which no pristine reference condition sites can be found. An example of a situation of this kind is presented in Section 15.

13.2 Establish Regional Multidisciplinary Technical Advisory Group*

To be most effective, application of the reference condition approach should rely on the combined effort of a multidisciplinary team of individuals. The disciplines to be represented on the task force depend on the objectives and resources available, but should at least include experts in physical, chemical, and biological oceanography. If the design of a remediation program is included in the objectives, it may also be necessary to include individuals with expertise in hydrology, water resource management, agriculture, and land-use planning. Whenever possible, these experts should be drawn from local and regional agencies. Potential sources of members include federal, provincial and municipal agencies, academic institutions, local community groups, and private sector organizations.

** See Section 1.4 and Chapter 7 of the USEPA (2001a) document.

*See Section 7.2 of the USEPA (2001a) document.

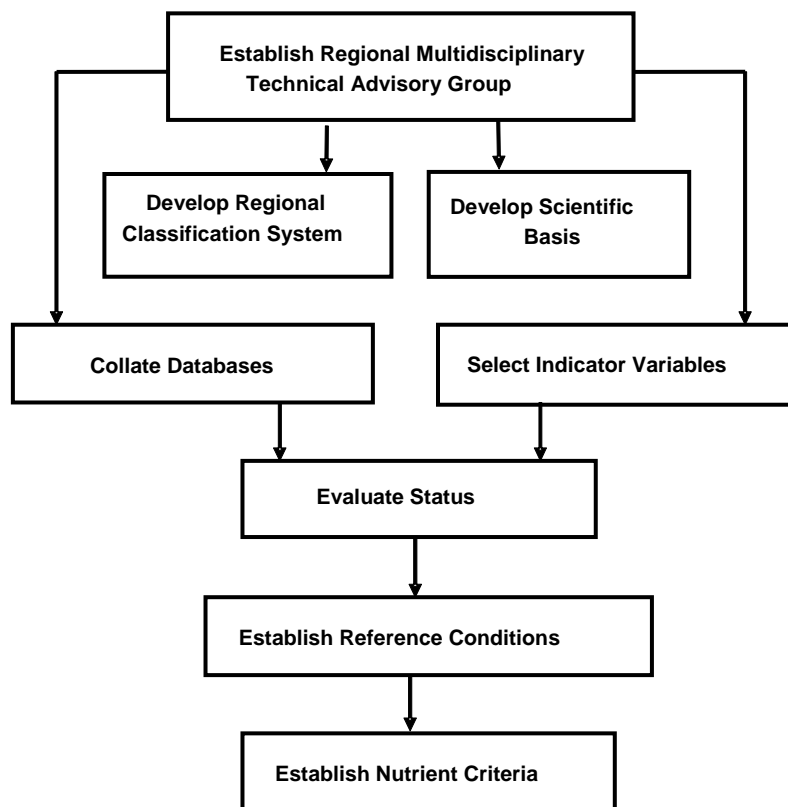


Figure 4. Elements of USEPA (2001a) nutrient criteria development process.

The technical assistance group should be well organized and coordinated and have a structure that allows for debate and resolution of differing viewpoints and any user conflicts that may arise.

13.3 Develop Regional Classification Scheme*

Coastal systems that are structurally and functionally similar may differ greatly in their response to nutrient inputs based solely on their geographic location as a result of differences in climate and geology. This requires that an appropriate ecoregion classification system be used. If a suitable system has not already been developed it will be necessary to develop one. It is also important to develop a classification system for coastal systems within the same ecoregion. This is typically done on the basis of the physical characteristics that are considered to be most important in determining the response to nutrient over-enrichment (see Section 8).

The reference condition approach is strongly based on the ability to classify coastal marine environments into categories based on their susceptibility to nutrient over-enrichment. This is perhaps one of the most difficult aspects of the approach and one

*See Chapter 3 and Section 7.3 of the USEPA (2001a) document.

which must be carried out with considerable care. Although a number of classification systems for coastal systems have been developed, most are based largely on morphological and stratification characteristics (Pritchard 1955; Dyer 1973; Biggs and Cronin 1981; Gregory et al. 1993; Gregory and Petrie 1994) with little focus on ecological processes, and there is a real need for a classification system focused more strongly on susceptibility to nutrient over-enrichment (Livingston 2001; Jay et al. 2000). The USEPA reviewed a number of classification systems for estuaries based on geomorphology, physical/hydrodynamic factors, and biological habitats and has recently developed an extensive system for classification of coastal systems based on their response to a variety of stressors, including nutrient over-enrichment (USEPA 2004). This document should be reviewed to provide guidance in developing a classification scheme in those instances where no suitable classification scheme currently exists.

The importance of having a well developed typology for a particular region can not be over-emphasized as this is essentially the basis for classifying systems according to their assimilation capacity. Classifications based on trophic state or level of human impact that focus on symptoms of nutrient over-enrichment should be avoided since the objective of the classification scheme is to assess susceptibility to nutrient inputs rather than the response to nutrient inputs. The scheme developed for Canadian marine systems (Harding 1997), which classifies Canada's marine areas according to ecozones, ecoprovinces, ecoregions, and ecodistricts (see Appendix II), is a good beginning, but additional classification focusing more directly on susceptibility to nutrient over-enrichment needs to be developed. The system characteristics suggested for use in this classification are morphology, water residence time, freshwater-saltwater exchanges, salinity, general water chemistry characteristics, depth, and grain size or bottom type. For estuaries, it may be important to sub-classify a system according to salinity zones. Some coastal systems may not fit well into any classification scheme due to unusually unique characteristics and this may require the development of criteria specific to that site and will require an historical database that includes data on pre-impacted conditions.

The general approach used in the NOAA Estuarine Eutrophication Survey (Bricker et al. 1999) for determining susceptibility to nutrient over-enrichment is an excellent framework for classifying coastal systems. Details of this approach are presented in Section 9 and it is suggested that this be adopted. The multivariate analysis approach employed by Strain and Yeats (1999) which employed common factor analysis procedures to classify a number of Nova Scotian coastal system may also provide useful guidance for developing a classification scheme.

Nearshore coastal systems not associated with freshwater inflows are generally quite different both structurally and functionally than coastal lagoons and estuaries. In addition to being deeper and more influenced by longshore water circulation patterns, the biological communities present are more characteristic of shelf ecosystems. Algal macrophytes such as rockweeds and kelps replace the seagrasses common to estuaries and the greater depths result in weaker benthic-pelagic couplings. With respect to nutrient inputs, they are more subject to offshore upwelling of nutrient rich water (Harrison et al. 1983; Townsend 1998). There have been very few studies that have

attempted to develop classification schemes for these systems based on their susceptibility to nutrient over-enrichment.

13.4 Develop Scientific Basis for Criteria Development*

Determining the scientific basis for criteria development involves establishing the particular approach (i.e., pristine sites, hindcasting, using historical data, paleoecological study, etc.) and the type of data (e.g., ambient nutrient concentration, nutrient loading rates, Secchi disk depth, light attenuation, etc.) that will be required to develop the nutrient criteria.

The major causal parameters recommended for development of nutrient criteria are total nitrogen and total phosphorus. Although nitrogen is considered to be the nutrient most often responsible for creating the adverse conditions associated with nutrient over-enrichment, a number of studies have shown phosphorus to be important in some systems, particularly in temperate zone estuaries during periods of high freshwater inflows (Fisher et al. 1992; Malone et al. 1996).

There is a great deal of controversy as to whether water column nutrient concentrations or nutrient loading rates are most appropriate for establishing nutrient criteria (USEPA 2001a). Some believe that total water column nutrient concentration provides the best indicator of short-term phytoplankton potential (Boynton and Kemp 2000), but that nutrient loading rates give a better indication of the long-term potential for development of conditions associated with nutrient over-enrichment. With a few exceptions (Nixon and Pilson 1983; Monbet 1992; Borum 1996), comparative studies that have attempted to find correlations between nutrient concentration and phytoplankton biomass among coastal systems have not been particularly successful (Hinga et al. 1995; Bricker et al. 1999). This is thought to be due to the multitude of other factors that control phytoplankton biomass (e.g., flushing rates, water residence times, light availability, grazing, relative importance of nitrogen vs. phosphorus, etc.) which results in a high degree of individuality in nutrient response among coastal systems.

A number of comparative studies (Boynton 1997) have reported strong correlations between nutrient loading and phytoplankton biomass, especially when loading is normalized for area (Nixon 1995) or water residence time (Dettmann 2001) of the coastal system. This is similar to the Vollenweider approach used to estimate permissible loadings for freshwater lakes (Vollenweider and Kerekes 1982).

The major response parameters of nutrient over-enrichment suggested for use in evaluating the status of coastal systems are phytoplankton chlorophyll *a* and water clarity. The latter is typically measured as Secchi disk depth or light attenuation. However, in some cases, particularly those in which evidence of symptoms of nutrient over-enrichment are suspected, it may also be important to include dissolved oxygen levels and macrophyte biomass as response variables. Depending on the particular

* See Chapter 2 of USEPA (2001a) document.

coastal system being addressed, other parameters may also be included (see Section 7.7 for additional parameters that may be appropriate).

13.5 Select Indicator Variables for Assessing System State*

The parameters used to assess the current status of the coastal system for which nutrient criteria are being sought need to be well defined, not only for assessing the need for remediation, but also to determine if they are suitable for establishing reference conditions.

Examples of indicators of causal variables include total nitrogen and total phosphorus. Although total nitrogen and total phosphorus are preferred, historical data on these forms of nitrogen and phosphorus may be limited making it necessary to use other forms, such as nitrate, ammonia, and phosphate.

Indicators of the response to nutrient inputs include phytoplankton chlorophyll *a*, water clarity, and dissolved oxygen. Other indicators of response may include loss of seagrasses/submerged aquatic vegetation and benthic macrofauna, increased growth of intertidal algae, and other changes within the intertidal community. Potential parameters for use as indicators are discussed in Section 7.

Howarth et al. (2000) point out that, because of both seasonal and annual variability in climatic conditions, which can have a large influence on factors related to the causes and responses of systems to nutrient over-enrichment, it is important to use multiyear datasets when applying indices in evaluating the status of a system.

13.6. Determine Database Requirements and Availability*

Existing databases are located and assessed with respect to the data requirements for determining the status of each coastal system. Efforts should be made to obtain historical data that allows an evaluation of the degree to which the system has changed over time with respect to its trophic status, as well as how much variability or stability it exhibits. This data is most commonly obtained from government agencies, but other sources, such as universities, monitoring programmes carried out by community based groups, anecdotal information and observations made by local residents, fisherpersons and aquaculturalists, and logs of scientific field crews, should also be considered. Regardless of the data source, it is important that some evaluation of its quality be made.

Historical data is especially valuable as it provides important information on how the system has changed over time and the degree of variability that may exist, particularly with respect to annual variations in climatic conditions. Historical data is essential in those instances where no pristine or near pristine sites exist for establishing reference conditions.

*See Chapter 4 of the USEPA (2001a) document.

*See Chapter 5 of the USEPA (2001a) document.

13.7 Determine Trophic Status and Establish Reference Conditions

Once the coastal systems of interest are classified into categories according to ecoregion and physical characteristics, the trophic status of each site is determined. This is accomplished using one, and preferably several, of the approaches described in Section 7. This information is then used to determine which sites fall within the pristine or near pristine category and are suitable for establishing reference conditions.

The various approaches for setting reference conditions are summarized in Table 14. Three of the approaches are based on in situ observations and vary depending on the degree of degradation associated with the reference sites being used to establish nutrient criteria. A fourth, watershed based, approach involves determining the nutrient loading rates to relatively pristine tributaries within an estuary and may be useful when no unimpacted reference sites exist within the lower part of an estuary. A fifth approach is applicable to situations where the area of concern lies in coastal waters offshore of the estuary.

13.7.1 Approaches Based on In Situ Observations

In those instances where the reference sites are determined to be in a pristine state (i.e., little evidence of any anthropogenic impact), it is recommended that the median be used to establish reference condition (Fig. 5). If the reference sites exhibit some evidence of anthropogenic impact, but exhibit little evidence of degradation, it is suggested that the value of the 75th percent quartile be used.

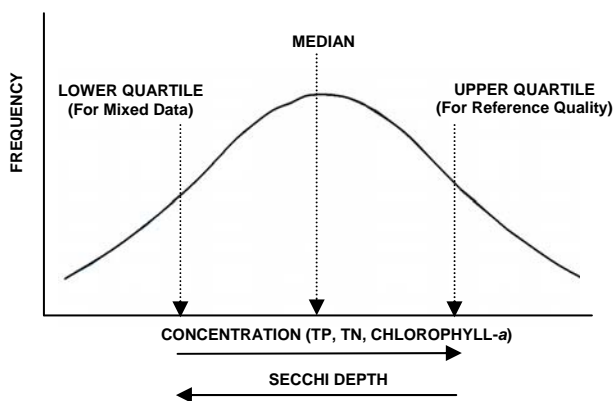


Figure 5. Hypothetical frequency distribution of nutrient-related variables showing quartiles for reference high-quality data and mixed data (from USEPA 2001a).

Table 14. Summary of approaches for reference condition determination (USEPA 2001a).

Degree of Apparent Degradation	Method Recommended	Criterion Measure
A. In situ Observations as the Basis for Estuarine Reference Conditions 1. Recognized unique excellent condition 2. Some degradation, but reference sites exist 3. Significantly degraded including all reference sites	Median ambient concentration (Fig.5). Upper quartile (Fig. 5). Intercept value on a regression or distribution curve as illustrated in Figs. 6 and 7 or by use of a comparable regression model.	Concentration of TP, TN, chl <i>a</i> , Secchi Depth Same as above Same as above
B. Watershed-Based Approaches for Estuarine Reference Conditions 4. Same as approach 3 above, but insufficient historical data	Reference sites along each tributary and calculate delivery. Summation is reference condition. Model required to back calculate load where all tributaries are degraded.	Load of TP and TN; model is required to convert load to estuarine concentration.
C. Coastal Reference Condition 5. Applicable to all coastal reaches - Estuarine plumes - Coastal areas	Index site approach; models may help distinguish anthropogenic contribution.	Concentrations

If it is determined that criteria should be based on a seasonal rather than annual basis it may be more appropriate to develop separate criteria for each season. For example, within Canada spring and summer are the most common growth periods and it may be determined that the criteria for chlorophyll *a* and Secchi depth are best based on these particular seasons. The seasonal approach has been adopted by the USEPA for streams and rivers (USEPA 2001b). The recommended procedure is to (1) calculate the median value of each reference site for each season of interest; (2) determine the value of the 75th percentile of the medians for each season; and (3) calculate the median value of the 75th percentile values for all of the seasons of interest. The latter then becomes the nutrient criteria. If the data also includes sites that may be only near-pristine, it may be appropriate to use the 25th percentile in place of the 75th.

In those cases where no pristine or near pristine sites exists, it may be possible to arrive at reference conditions based on either an historical or empirical analysis of the available data. The database would have to cover the period prior to evidence of symptoms associated with nutrient over-enrichment. One potentially serious problem in using this approach is that some symptoms of nutrient over-enrichment may be due to factors other

than nutrient inputs. An example is the loss of submersed aquatic vegetation as a result of increases in turbidity due to excessive erosion within a watershed, or increased nutrient retention as a result of barriers to flushing such as causeways. It is also important that the database used be extensive enough to represent the spatial and temporal variability, both seasonally and annually, among the causative and response parameters so that median or mean values can be determined along with an estimate of their variability.

Figure 6 illustrates one way of how nutrient criteria can be arrived at using this approach. The dashed line represents conditions prior to degradation and is then used to establish criteria for causal and response variables.

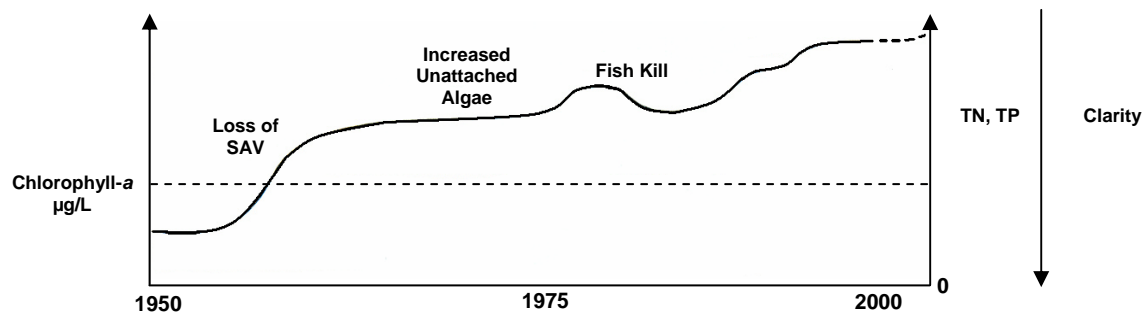


Figure 6. Hypothetical example of load/concentration response of estuarine biota to increased enrichment. Dashed line represents the selected reference condition (from USEPA 2001a).

It may also be possible to summarize the existing historical data as a frequency distribution for past and current data to produce a diagram as illustrated in Figure 7. In this case the reference condition can be established at the median between the historical median and the median for the present day data.

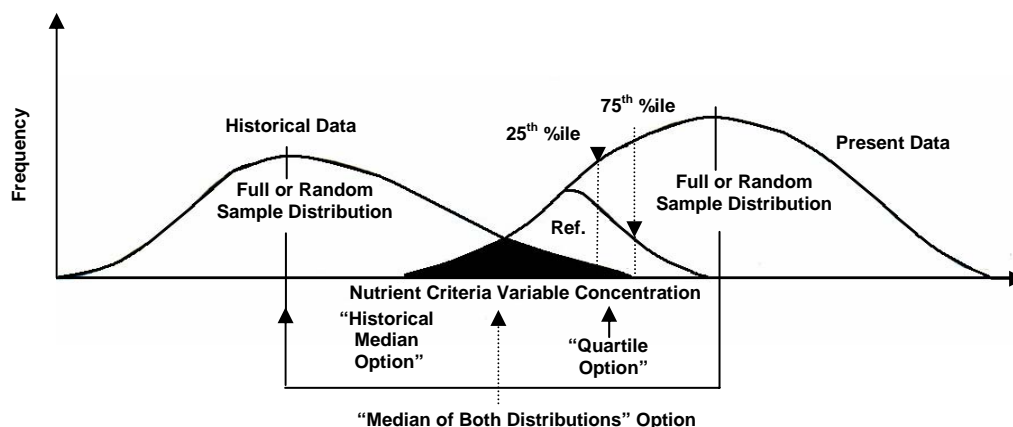


Figure 7. Illustration of the comparison of past and present nutrient data to establish a reference condition for intensively degraded estuaries (from USEPA 2001a).

13.7.2 Watershed-Based Approach

In the watershed approach an attempt is made to locate a tributary which is representative of the estuary and is not nutrient enriched. If the drainage basin characteristics of the tributary, other than those associated with anthropogenic activities, are similar to other tributaries within the estuary, the nutrient load from this tributary can be extrapolated to the rest of the estuary either empirically or with a model. The total nutrient load is then related empirically using appropriate models to the response parameters. In this case nutrient criteria may be based on nutrient load rather than nutrient concentration. This approach involves a number of assumptions.

In this approach the aerial nutrient load is determined for the unimpacted tributary and then extrapolated to the entire tributary system to give the 'reference load' for the estuary. There are a number of important assumptions for using this approach (see Table 6-2 of the USEPA (2001a) document). Especially important are the assumptions that all of the tributaries within the estuary are similar in terms of their geology, soil types and vegetation cover, nutrient loading is predominantly tributary based (no point sources and coastal nutrient input is small relative to estuarine inputs), and groundwater input is similar in all tributaries.

13.7.3 Coastal Reference Conditions

The coastal approach relies on knowledge of changes in the nutrient characteristics of estuarine plumes and offshore water. Considerable information is required on the mixing and dispersive capacity of the shelf area and this typically involves the development and use of hydrodynamic circulation models. This approach has not been particularly well developed, largely because nutrient over-enrichment problems are much less common in the more offshore coastal environments.

13.7.4 Reference Conditions Developed from Sediment Core Data

Another approach that has been suggested as potentially useful in establishing past conditions is by analyzing information obtained from sediment core data. Although the time scale resolution using this technique is relatively large, it can provide some information as to what conditions were like during periods when anthropogenic impacts are unlikely to have occurred. Examples include periods of hypoxic or anoxic conditions and, through analysis of phytoplankton remnants, an indirect indication of past nutrient conditions.

13.8 Establish Nutrient Criteria

Once reference conditions are established, setting nutrient criteria may also require consideration of what is acceptable in terms of the level of deviation from pristine conditions based on the anticipated human uses of the coastal system. These are essentially value judgements that must be made by the appropriate environmental managers and stakeholders.

It is also important that the initial criteria be calibrated and verified. This is done by applying the criteria to water bodies of known status. Failure of the criteria requires that they be re-evaluated and may involve consideration of factors not originally included in establishing the criteria (e.g., turbidity, water colour, toxins).

Once the criteria have been validated they can be used to determine the status of a particular system based on the causal variables (nitrogen and phosphorus) and the response variables (chlorophyll *a*, water clarity, dissolved oxygen and others) that may have been considered appropriate. Figure 8 provides an overview of the process used to establish nutrient criteria.

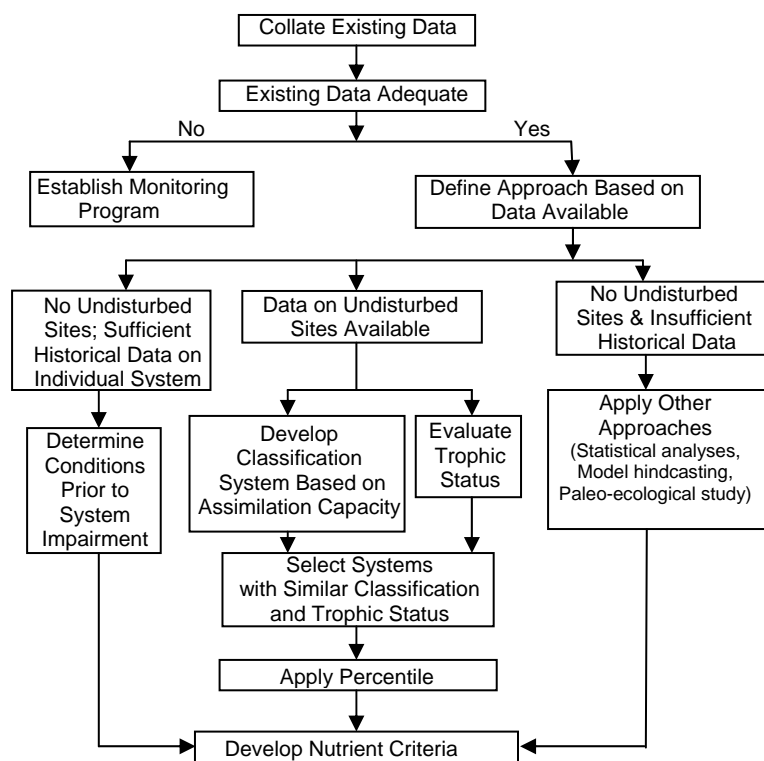


Figure 8. Overview of procedures for establishing nutrient criteria.

14. Nova Scotia Case Study

14.1 Introduction

The Atlantic shoreline of Nova Scotia contains a large number of coastal bays and inlets that are subject to various levels and types of anthropogenic impacts. The objective of this case study was to establish nutrient criteria for these systems using the reference condition approach based on an analysis of existing in situ data.

14.2 Study Sites

This case study is based largely on data collected as part of a study to evaluate the potential for using chemical factors to predict the eutrophication status of a number of coastal systems located along the shoreline of Nova Scotia (Strain and Yeats 1999). In that study data were collected on a total of 33 coastal systems, 20 of which were located along the Atlantic coastline of mainland Nova Scotia. The sites chosen for this case study are those located along the mainland Atlantic nearshore coastal region for which adequate data exist on the hydrographic and nutrient characteristics required to classify them according to their susceptibility to nutrient over-enrichment and their current trophic status. A total of 17 sites were identified as having the necessary data available (Fig. 9).

The area in which these sites are located can be considered as a single ecoregion. All watersheds in this region have similar geological characteristics and poorly developed soils. The entire coastline is rocky and relatively exposed, and all regions of the area experience similar meteorological conditions and tidal characteristics. In addition, most of the coastal inlets are relatively small and shallow with small drainage basins. A unique and important characteristic of these inlets, as well as others within the Gulf of Maine-Scotian Shelf system is that they are naturally productive systems as a result of offshore inputs of nutrient rich shelf waters (Petrie et al. 1999).

14.3 Approach

The reference condition approach (see Section 13.7) was used to establish nutrient criteria. This decision was based on the fact that all of the data required for classifying each system in terms of its susceptibility to nutrient over-enrichment and trophic status were available.

The index of susceptibility to nutrient over-enrichment was developed based on the parameters that reflect the system's capacity to dilute nutrients entering the system from land-based sources. The parameters used in the evaluation were tidal/freshwater volume ratio, flushing time, and degree of water column stratification. The trophic status of each site was based on water column nutrient and bottom water dissolved oxygen concentrations. Once evaluated, an appropriate subset of reference sites was selected from the 17 sites on the basis of relatively pristine trophic state and a similar index of susceptibility. The selected reference sites were then used to establish criteria for causal and response variables using a frequency distribution approach.

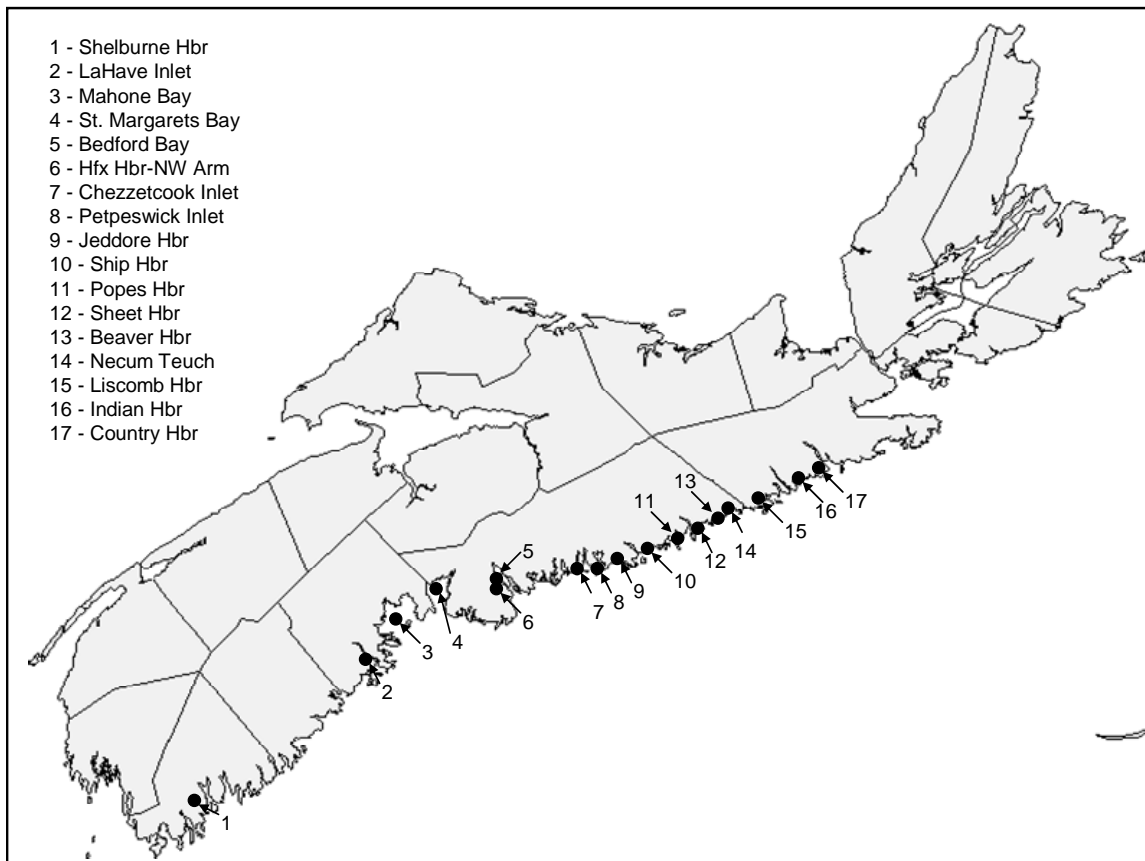


Figure 9. Location of sites selected for analysis in Nova Scotia.

14.4 Databases

Two databases were used in this study. Information on the morphological and hydrological characteristics of each site was obtained from Gregory et al. (1993). Data on temperature, salinity, and nutrient concentrations was obtained from the BIOCHEM* database currently being administered by the Department of Fisheries Oceans Canada. The data available through the BIOCHEM were largely collected by Strain and Yeats (1999). Summaries of the hydrological and chemical data are contained in Tables 15 and 16.

The hydrological data is quite extensive and contains all of the information required to classify the sites according to their physical characteristics. The nutrient and other chemical data is also quite extensive and is of a very high quality, having been collected by trained marine chemists using similar techniques and at a time when the coastal systems were most susceptible to nutrient over-enrichment, i.e., during late summer and early fall when water temperatures are high, and before fall turnover when stratification is likely to be strongest. The major shortcoming of the chemical database is that data were

* <http://www.meds-sdmm.dfo-mpo.gc.ca>

collected at only one time and one location for each coastal system and does not provide any indication of the degree of temporal or spatial variation within each site.

Table 15. Morphological and physical characteristics of each site.

Site	Watershed Area (sq km)	High Water Area (sq km)	Volume (10 ⁶ cu m)	Maximum Depth (m)	Mean Tidal Range (m)	Tidal Prism 10 ⁶ cu m	Tidal / Freshwater Volume Ratio	Flushing Time (hrs)
Beaver Hbr	13.5	13.5	121	21.3	1.4	18.4	723.1	88.1
Bedford Bay	16.8	16.8	510	71.0	1.5	24.8	109.4	261.4
Chezzetcook Inlet	14.4	14.4	9	5.5	1.4	12.8	165.2	14.0
Country Hbr	9.9	9.9	89	21.9	1.2	11.3	39.5	104.0
Hfx - NW Arm	93.1	93.1	1759	71.0	1.6	146.4	373.0	155.3
Indian Hbr	11.4	11.4	116	20.7	1.4	14.9	299.9	102.9
Jeddore Hbr	21.1	21.1	83	18.3	1.4	28.3	177.6	42.3
LaHave Inlet	19.6	19.6	77	27.4	1.6	29.1	27.1	38.7
Liscomb Hbr	19.8	19.8	106	14.3	1.4	27.3	53.5	54.2
Mahone Bay	216.5	216.5	2227	62.2	1.5	319.3	344.5	170.6
Necum Teuch Hbr	4.2	4.2	12	14.6	1.3	4.7	29.1	37.6
Petpeswick Inlet	14.0	14.0	30	11.0	1.4	14.5	186.3	31.5
Popes Hbr	11.3	11.3	57	21.9	1.4	15.2	716.5	52.5
Sheet Hbr	18.9	18.9	106	22.9	1.5	27.4	35.2	54.1
Shelburne Hbr	22.7	22.7	140	13.1	1.7	37.3	68.3	52.6
Ship Hbr	7.2	7.2	47	25.0	1.4	9.7	23.9	66.3
St. Margarets Bay	141.8	141.8	5191	91.4	1.6	223.8	416.3	294.2

14.5 Evaluation of Trophic Status

The trophic status of each site was evaluated on the basis of nutrient and dissolved oxygen concentrations. The nutrients used were dissolved inorganic nitrogen-N and phosphate-P. These represent the causal variables. The average nutrient concentration of surface (1 m) and bottom (ca. 1 m above maximum depth) water samples was used. For dissolved oxygen, the response variable, bottom water concentration was used*.

* In addition to dissolved oxygen concentration, commonly used response variables for assessing trophic status are phytoplankton chlorophyll *a* and Secchi disk depth. However, no data were available on the latter variables for these sites.

Table 16. Chemical characteristics of each site.

Site	Depth (m)	Salinity (psu)	Temp (°C)	Dissolved Oxygen (mg/L)	Dissolved Phosphorus (mg/L)	Dissolved Inorganic Nitrogen (mg/L)
Beaver Hbr	1	29.8	18.1	8.2	0.008	0.008
“	7	29.0	17.4	8.3	0.023	0.014
Bedford Bay	1	30.0	11.1	8.9	0.023	0.035
“	13	30.8	7.3	6.4	0.046	0.103
Chezzetcook Inlet	1	29.8	19.0	8.4	0.021	0.009
“	2	29.9	18.8	7.8	0.020	0.012
Country Hbr	1	29.0	18.0	8.1	0.013	0.011
“	19	30.3	14.7	6.5	0.042	0.058
Hfx-NW Arm	1	30.6	10.1	9.7	0.017	0.029
“	18	32.0	7.6	8.8	0.025	0.039
Indian Hbr	1	29.7	17.3	8.7	0.012	0.007
“	8	29.7	17.3	8.0	0.014	0.015
Jeddore Hbr	1	29.7	18.7	7.7	0.023	0.008
“	17	29.8	17.2	4.4	0.139	0.348
LaHave Inlet	1	27.8	19.6	7.8	0.037	0.010
“	8	29.1	18.1	5.2	0.076	0.078
Liscomb Hbr	1	29.4	18.7	8.0	0.035	0.052
“	14	29.6	17.4	7.0	0.035	0.037
Mahone Bay	1	30.4	17.6	8.1	0.012	0.006
“	13	30.5	12.1	8.4	0.019	0.010
Necum Teuch Inlet	1	29.6	18.2	8.2	0.014	0.006
“	3	29.7	12.1	7.8	0.013	0.005
Petpeswick Inlet	1	28.6	20.0	8.3	0.041	0.012
“	24	28.9	17.3	0.0	0.356	1.131
Popes Hbr	1	29.8	18.4	7.9	0.013	0.008
“	20	30.3	12.2	5.1	0.063	0.241
Sheet Hbr	1	17.3	19.0	8.6	0.002	0.011
“	14	30.1	13.9	5.3	0.053	0.204
Shelburne Hbr	1	30.5	17.6	8.1	0.012	0.010
“	13	30.7	17.0	7.0	0.032	0.062
Ship Hbr	1	26.9	18.9	8.2	0.011	0.008
“	23	30.6	7.3	0.4	0.109	0.866
St. Margarets Bay	1	30.2	17.9	7.8	0.012	0.005
“	14	30.5	13.2	8.4	0.016	0.014

Trophic state, in terms of low, medium, and high, was assigned for each concentration according to the guidelines developed by Bricker et al. (1999), and bottom water concentrations of dissolved oxygen. These are summarized in Table 17. An overall index, based on all three factors was calculated by first assigning a value of 1, 2 or 3 for

* Additional details of this classification system are provided in Section 7.7 of the main document.

low, medium, and high, respectively, and then summing these. The results are summarized in Table 18.

Table 17. Criteria for evaluating degree of nutrient over-enrichment (from Bricker et al. 1999).*

Parameter	Low	Medium	High
N (mg/L)	≤ 0.1	$>0.1 - <1.0$	≥ 1.0
P (mg/L)	<0.01	$>0.01 - <0.1$	≥ 0.1
DO (mg/L)	≥ 5	$>2 - \leq 5$	$0 - \leq 2$
*The guidelines for nutrients proposed by Bricker et al. (1999) are based on surface water concentrations. However, in this case study the average value of surface and bottom water concentrations was used because of the significant offshore nutrient input that enters the inlets as bottom water.			

Table 18. Ranking of nutrient over-enrichment status.

Site	Parameter			Overall
	N	P	DO	
Beaver Hbr	1	2	1	4
Bedford Bay	1	2	1	4
Chezzetcook Inlet	1	2	1	4
Country Hbr	1	2	1	4
Hfx - NW Arm	1	2	1	4
Indian Hbr	1	2	1	4
Jeddore Hbr	2	2	2	6
LaHave Inlet	1	2	1	4
Liscomb Hbr	1	2	1	4
Mahone Bay	1	2	1	4
Necum Teuch Hbr	1	2	1	4
Petpeswick Inlet	2	2	3	7
Popes Hbr	2	2	1	5
Sheet Hbr	2	2	1	5
Shelburne Hbr	1	2	1	4
Ship Hbr	2	2	3	7
St. Margarets Bay	1	2	1	4

The trophic ranking based on nitrogen and dissolved oxygen varied considerably among sites. In contrast, the trophic ranking based on phosphorus did not vary at all, presumably because of the naturally high phosphorus input from offshore waters. Based on the

overall ranking of the 17 sites evaluated, 12 (values <5) were considered to exhibit little evidence of anthropogenic nutrient over-enrichment, three (values ≥ 5 and <7) were considered to be moderately nutrient enriched, and the remaining two (values ≥ 7) were considered to be nutrient over-enriched.

14.6 Evaluation of Susceptibility to Nutrient Over-enrichment

Susceptibility to nutrient over-enrichment was based on an assessment of those factors that result in nutrients entering from upstream being either diluted or flushed out of the system. The parameters used in the assessment were tidal/freshwater volume ratio, flushing time, and degree of water column stratification* for each site. An index for each parameter (Table 19) similar to that proposed by Bricker et al. (1999) was employed to assess the susceptibility (see Section 9 of the main document). This was then quantified using a similar procedure to that used for quantifying trophic status. This resulted in seven sites being classified as having low susceptibility, nine sites being classified as having moderate susceptibility, and one site as having high susceptibility (Table 20).

Table 19. Criteria for determining susceptibility to nutrient over-enrichment.

Parameter	Low	Medium	High
Tidal Prism/FW Ratio	≥ 200	$\geq 100 - 200 <$	$100 <$
Flushing Time (days)	$3 <$	$\geq 3 - 10 <$	≥ 10
Stratification (sigma- <i>t</i>)	< 5	$> 5 - \leq 10$	≥ 10

* The degree of stratification was determined as the density difference between surface and bottom water (based on temperature and salinity) and is expressed as sigma- *t* values

Table 20. Susceptibility to nutrient over-enrichment.

SITE	TIDAL/FW		Flushing Time		Mixing		Overall Rating
	Ratio	Rating	Days	Rating	Sigma -t	Rating	
Beaver Hbr	723.1	1	3.7	2	0.4	1	4
Bedford Bay	109.4	2	10.9	3	1.1	1	6
Chezzetcook Inlet	165.2	2	0.6	1	0.0	1	4
Country Hbr	39.5	3	4.3	2	2.4	1	6
Hfx - NW Arm	109.8	2	6.5	2	1.0	1	5
Indian Hbr	299.9	1	4.3	2	0.1	1	4
Jeddore Hbr	177.6	2	1.8	1	0.3	1	4
LaHave Inlet	27.1	3	1.6	1	1.5	1	5
Liscomb Hbr	53.5	3	2.3	1	0.2	1	5
Mahone Bay	344.5	1	7.1	2	1.3	1	4
Necum Teuch Hbr	29.1	3	1.6	1	0.2	1	5
Petpeswick Inlet	186.3	2	1.3	1	0.1	1	4
Popes Hbr	716.5	1	2.2	1	1.9	1	3
Sheet Hbr	35.2	3	2.3	1	10.6	3	7
Shelburne Hbr	68.3	3	2.2	1	0.2	1	5
Ship Hbr	23.9	3	2.8	1	5.2	2	6
St. Margarets Bay	416.3	1	12.3	3	1.1	1	5

14.7 Selection of Reference Sites

The final selection of sites suitable to serve as reference condition sites was made by choosing those that showed little evidence of nutrient over-enrichment (low rating) and had similar indices of susceptibility to nutrient over-enrichment. Of the 17 sites, twelve were rated as having a low trophic status. Of these twelve sites, four had a low susceptibility to nutrient over-enrichment and eight had a moderate susceptibility. In order to maximize the number of reference used for establishing nutrient criteria, the eight sites having a low trophic status and a moderate susceptibility to nutrient over-enrichment were chosen as reference sites (Table 21).

14.8 Nutrient Criteria

The general procedure for establishing nutrient criteria using the reference condition approach based on data from coastal systems that show little evidence of nutrient over-enrichment involves an analysis of the frequency distribution of causal and response variables. Figures 10, 11 and 12 show the frequency distribution and cumulative frequency for dissolved inorganic nitrogen-N, dissolved inorganic phosphorus-P, and bottom water dissolved oxygen, respectively.

Table 21. Suitability of sites to serve as reference condition sites.

Site	Trophic Status	Susceptibility To Nutrient Over-enrichment	Reference Site
Beaver Hbr	LOW	LOW	N
Bedford Bay	LOW	MED	Y
Chezzetcook Inlet	LOW	LOW	N
Country Hbr	LOW	MED	Y
Hfx-NW Arm	LOW	MED	Y
Indian Hbr	LOW	LOW	N
Jeddore Hbr	MED	LOW	N
LaHave Inlet	LOW	MED	Y
Liscomb Hbr	LOW	MED	Y
Mahone Bay	LOW	LOW	N
Necum Teuch Inlet	LOW	MED	Y
Petpeswick Inlet	HIGH	LOW	N
Popes Hbr	MED	LOW	N
Sheet Hbr	MED	HIGH	N
Shelburne Hbr	LOW	MED	Y
Ship Hbr	HIGH	MED	N
St. Margarets Bay	LOW	MED	Y

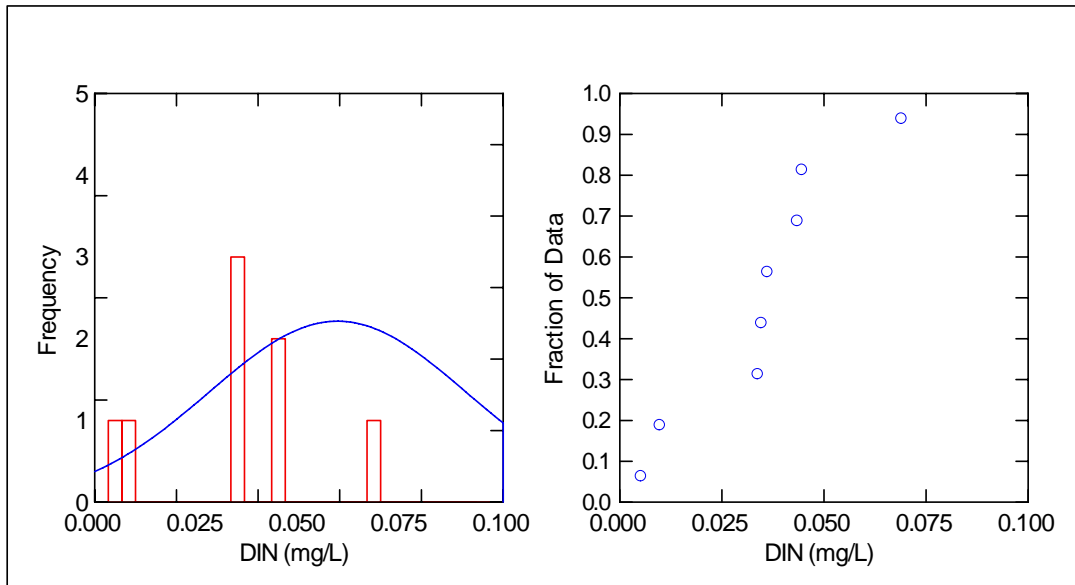


Figure 10. Frequency and cumulative frequency distribution for dissolved inorganic nitrogen-N.

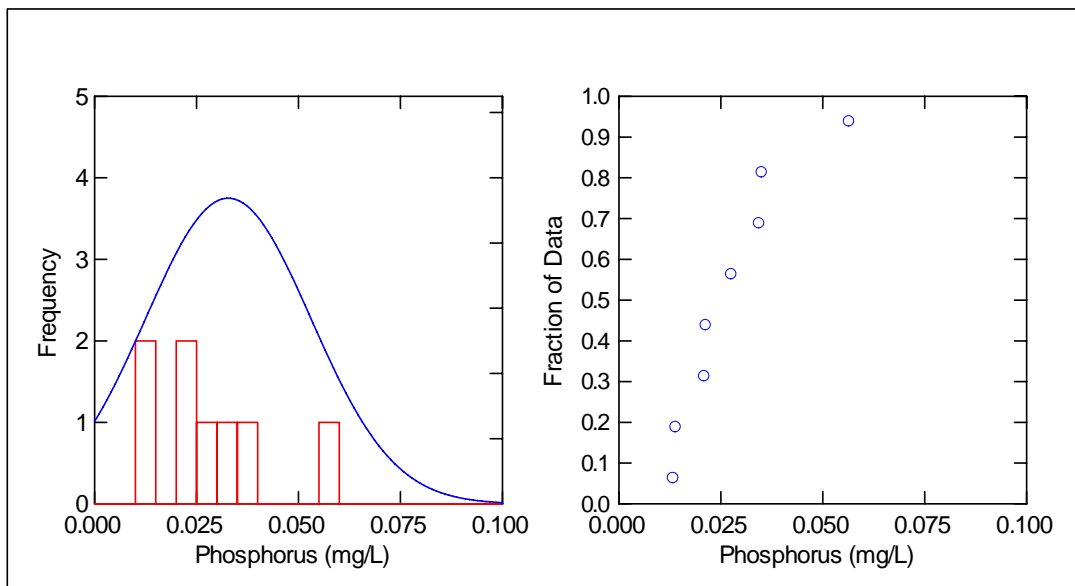


Figure 11. Frequency and cumulative frequency distribution for dissolved inorganic phosphorus-P.

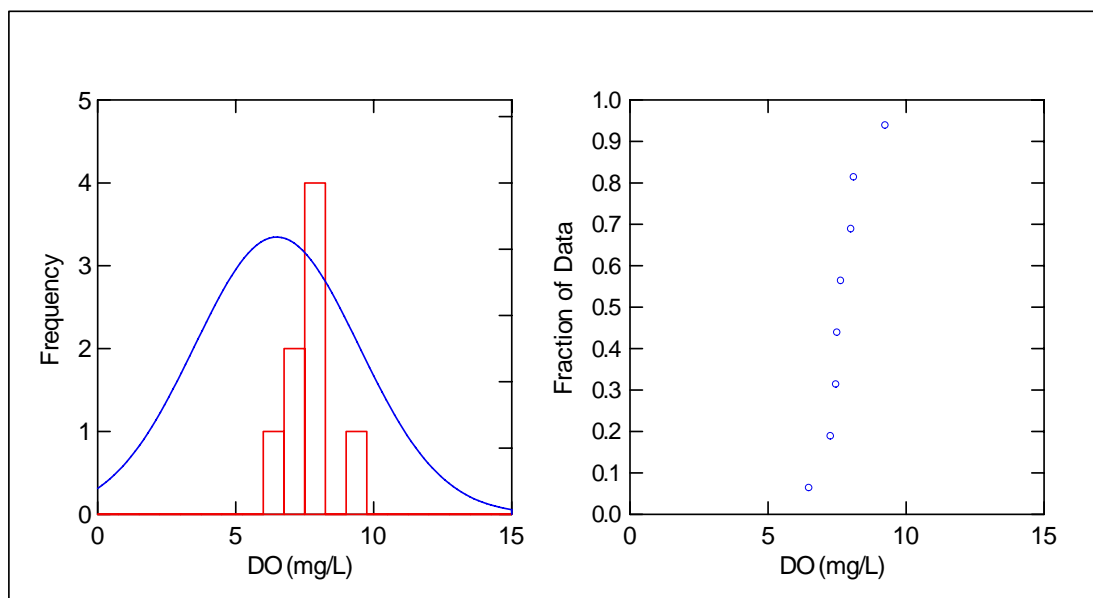


Figure 12. Frequency and cumulative frequency distribution for bottom water dissolved oxygen.

In establishing nutrient criteria, either the median, the 25th or 75th quartile concentration is typically used*. The median is recommended when the reference sites are considered to

* See Section 13.7 of the main document and Chapter 6 of the USEPA (2001a) document.

be relatively pristine and unimpacted by anthropogenic nutrient inputs. The 25th percentile is recommended when at least some of the reference sites are considered to be likely to exhibit some, but not serious, degradation as a result of anthropogenic nutrient inputs. The 75th percentile is recommended when there is strong evidence that all of the reference sites are relatively pristine with minimal anthropogenic nutrient inputs. For dissolved oxygen, the 25th and 75th percentiles are reversed since high concentrations are better than low levels.

Since all of the sites used to establish reference conditions in this study have human populations in their watersheds, it is suggested that the 25th percentile be adopted for nutrients and the 75th percentile for dissolved oxygen. For DO, the 25th and 75th percentiles are reversed since high concentrations (oxia) are better than low levels (anoxia). The final decision for which criteria to adopt, however, depends on management goals (i.e., level of water quality considered to be acceptable) for reducing nutrient inputs in situations where nutrient over-enrichment is a problem.

Table 22 lists the values of the 25th, median, and 75th percentile along with other basic statistics for each of the causal and response variables.

Table 22. Summary statistics for reference sites.

Statistic	Dissolved Inorganic- N (mg/L)	Dissolved Phosphorus (mg/L)	Dissolved Oxygen (mg/L)
Minimum	0.005	0.013	6/5
Maximum	0.069	0.057	9.3
Mean	0.035	0.028	7.7
Standard Deviation	0.020	0.020	0.8
25 th Percentile	0.020	0.015	6.8
Median	0.035	0.025	7.6
75 th Percentile	0.045	0.035	7.8

15. Boughton River Estuary (Prince Edward Island) Case Study

15.1 Introduction

The Boughton River estuary is located along the eastern shoreline of Prince Edward Island. It contains a number of aquaculture sites for blue mussels and oysters and supports commercial harvests of soft shell clams, quahogs, American oysters, eels, and smelt. It is also used for recreational boating and fishing.

Over the last few decades the results of a number of water quality monitoring programs have revealed that portions of the estuary experience periods of hypoxia in which dissolved oxygen concentrations decrease to levels below 50% saturation. These events appear to be associated with phytoplankton blooms that originate in the lower, freshwater portion of the river (Lane and Associates 1991). The objective of this case study is to establish nutrient criteria that would reduce the incidence of these events.

15.2 Site Description

Lane and Associates (1991) provide a general description of the Boughton River and its watershed. The Boughton River drains an area of about 95 sq km² and discharges into Boughton Bay which in turn discharges into the Gulf of St. Lawrence. Numerous smaller rivers and brooks also drain into the estuary. The head of the estuary extends to about 1.5 km above Bridgetown. The estuarine portion of the River is about 11 km long and contains a narrow (ca. 6.5 km long) inner section that widens at Poplar Point. Beyond Poplar Point the outer portion of the estuary widens into a Bay. A longshore sandspit at the mouth of the Bay restricts seaward entrance into the Bay from the Gulf of St. Lawrence to a 200 m wide channel.

The estuarine portion of the River is relatively shallow, especially within the inner section where average low tide depths in some areas may be less than 1 m. The wider, outer section of the estuary is much deeper, with maximum low tide depths ranging between 7-12 m. The tidal amplitude ranges from 1.1 to 1.7 m and the estuarine surface area is 7.6 and 9.9 sq km² at low and high tide, respectively. The low tide volume of the estuary is $29.1 \times 10^6 \text{ m}^3$ and the residence time is 2.1 days (Lane and Associates. 1991). Water column stratification within the estuary varies greatly, from strongly stratified to vertically homogenous, depending on variations in tidal amplitude and freshwater discharge.

More than half (about 55%) of the Boughton River watershed is forested. The remainder is mostly cleared land (40%) and wetlands (5%). Most of the cleared land is used for agricultural activities related to livestock operations (pasture and forage crops).

15.3 Approach

Because no pristine sites similar to the Boughton River estuary exist within Prince Edward Island, it was not possible to use the reference condition approach to establish nutrient criteria. There are also no data on nutrient concentrations available prior to the times when algal blooms and hypoxic incidents are known to have occurred, so nutrient criteria could not be established based on historical data. As a result, the approach employed was to use a statistical analysis of in situ data on causal and response variables to determine the conditions when the symptoms of nutrient over-enrichment were minimal. The rationale for this approach was that if nutrient levels could be kept below

the levels experienced during times when bottom waters became hypoxic, it would lead to the elimination of hypoxic events.

The three response variables used were Secchi disk depth, phytoplankton chlorophyll *a* concentration, and percent dissolved oxygen saturation. The potential causal variables examined were nitrogen and phosphorus concentrations. If significant relationships could be shown to exist between the causal and response variables, it would then be possible to determine the nutrient concentrations below which the response variables would be at acceptable levels.

15.4 Databases

There is considerable data available on water quality for the Boughton River and estuary. An intensive study of the Boughton River system was carried out by Lane and Associates (1991) during 1988-89. Physical, chemical, and biological data were collected at weekly intervals between June and November in 1988, and three times per month between May and November in 1989. Six sites were sampled; five within the estuary and one in the freshwater reach of the River just above the head of the estuary (Fig. 13). Of the estuarine sites, three were located within the inner section of the estuary and the remaining two were located within the outer section.

More recent, but less intensive, monitoring of the Boughton River and estuary has been carried out during the period between 1998 and 2005 by various Provincial and Federal agencies. Within the estuary, eight sites have been monitored (Fig. 14), five in the inner section and three in the outer section.

15.5 Frequency and Times of Hypoxic Events

Data analysis indicated that hypoxic* conditions occur mainly in bottom waters during the late summer-early fall period. Table 23 lists the number of times and dates of occurrences of hypoxic events recorded for bottom waters in the available databases. It is noteworthy that no hypoxic events were observed during the spring and that the number of hypoxic events each year appears to be increasing. In addition, most hypoxic events occur within the inner section of the estuary.

* The term hypoxic as used here refers to instances when dissolved oxygen levels were ≤ 50 % saturation.

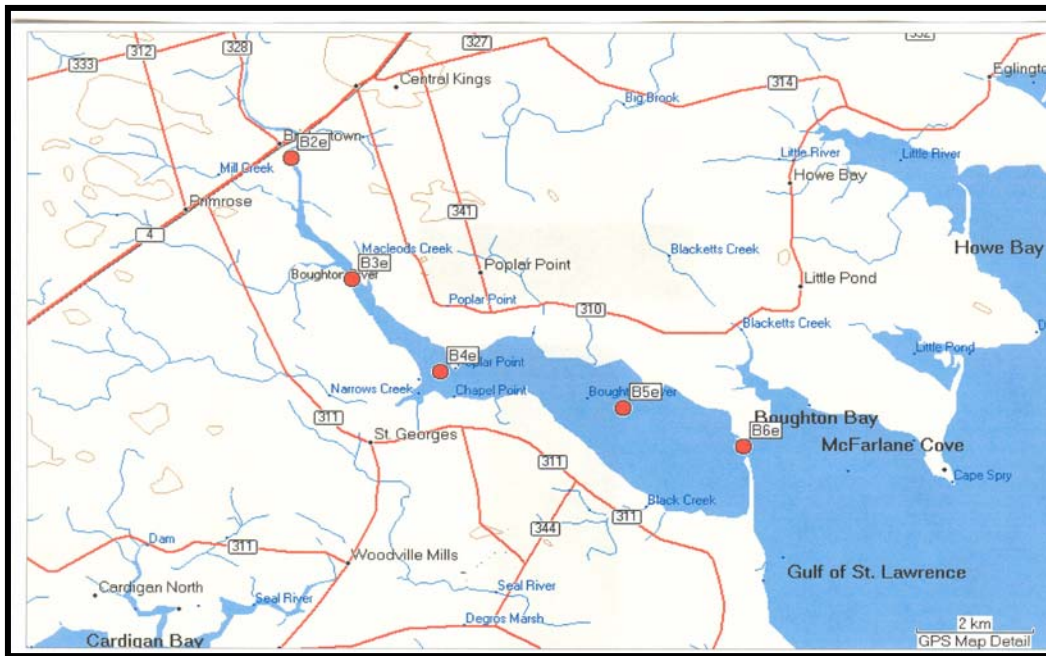


Figure 13. Location of sites sampled during 1988-89.



Figure 14. Location of sites sampled between 1998 and 2005.

Table 23. Number of times and dates when dissolved oxygen saturation was $\leq 50\%$ within bottom waters at each sample site.

Site	Year	N	N ≤ 50	Dates
B2e	1988	18	0	
B2e	1989	18	0	
B3e	1988	19	0	
B3e	1989	19	1	09/26
B4e	1988	21	0	
B4e	1989	17	1	09/12
B5e	1988	20	2	07/20; 07/27
B5e	1989	18	1	09/12
B6e	1988	4	0	
B6e	1989	7	0	
BRR1	1998	2	0	
BRR1	2000	2	1	08/08
BRR1	2004	16	3	07/21; 08/23; 09/07
BRR1	2005	12	0	
BRR2	1998	2	0	
BRR3	1998	2	0	
BRR3	2004	16	2	07/21; 08/23
BRR3	2005	12	2	08/17; 08/29
BRR4	1998	2	0	
BRR4	1999	2	1	08/04
BRR4	2001	2	0	
BRR4	2002	2	1	08/07
BRR4	2003	2	1	08/04
BRR4	2004	16	2	07/21; 08/23
BRR4	2005	14	3	08/17; 08/29; 09/21
BRR5	1998	2	0	
BRR5	2004	16	1	08/23
BRR5	2005	12	2	08/17; 08/29
BRR6	1998	2	0	
BRR6	1999	2	1	08/04
BRR6	2001	2	0	
BRR6	2002	2	0	
BRR6	2004	15	0	
BRR6	2005	14	0	
BRR7	1998	2	0	
BRR7	1999	2	1	08/04
BRR7	2000	2	0	
BRR7	2001	2	0	
BRR7	2002	2	0	
BRR7	2003	2	0	
BRR11	2002	2	0	
BRR11	2003	2	00	
BRR11	2004	16	0	
BRR11	2005	14	0	

15.6 Relationships between Response and Causal Variables

Figure 15 illustrates that, for data collected within the estuarine sampling sites between 1998 and 2005 (see Figure 14), there are statistically significant ($p < 0.05$) relationships between the three response variables and total phosphorus concentration*. Chlorophyll *a* concentration shows a positive correlation with total phosphorus, and both Secchi disk depth, and dissolved oxygen saturation exhibit a negative correlation with total phosphorus. In contrast, no significant correlations were observed between the response variables and inorganic nitrate or total nitrogen concentration. As a result, it was determined that the nutrient response criteria should be based on total phosphorus concentration.

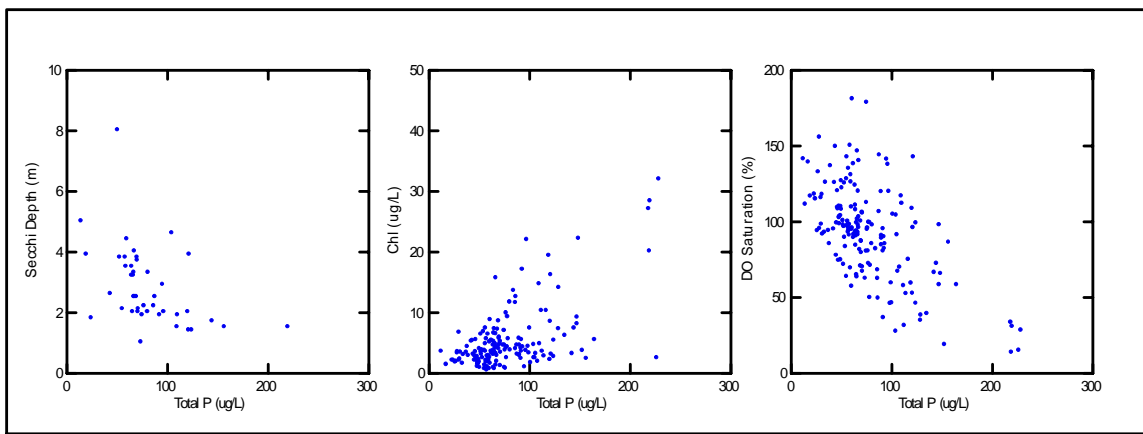


Figure 15. Relationship between response variables and total phosphorous concentration.

15.7 Establishing Nutrient Criteria

Nutrient criteria for total phosphorus were established by examining the values of total phosphorus concentration during times when hypoxic events occurred within the estuary. These values, as well as those when no hypoxic events occurred, are listed in Table 24 along with the corresponding values of the response variables. Figure 16 is a frequency plot of total phosphorus concentration for times when dissolved oxygen saturation levels are $\leq 50\%$ and $> 50\%$.

In order to minimize the occurrence of hypoxic events, it is suggested that total phosphorus concentrations within the estuary not exceed $50 \mu\text{g/L}$ during the late summer-early fall period, a value that approximates the concentration representing the 25th percentile for conditions when dissolved oxygen saturation values are $> 50\%$. If total phosphorus levels are maintained below this level during the most critical periods, it should significantly reduce the occurrence of hypoxic events.

* These figures do not include the data collected by Lane and Associates (1991) since only inorganic phosphate, which showed no significant correlation to any of the response variables, was measured in that survey.

Table 24. Statistics for response and causal variables during times when dissolved oxygen saturation was ≤ 50 % and >50 % in bottom waters within the inner section of the estuary.

Statistic	DO Saturation ≤ 50 %				DO Saturation >50 %			
	Secchi Depth (m)	Chl <i>a</i> ($\mu\text{g/L}$)	TP ($\mu\text{g/L}$)	DO Saturation (%)	Secchi Depth (m)	Chl <i>a</i> ($\mu\text{g/L}$)	TP ($\mu\text{g/L}$)	DO Saturation (%)
N	2	21	18	21	48	183	181	196
Minimum	1.5	2	68	1.3	1.0	1	13	50.2
Maximum	2.2	45	230	49.4	8.0	22	165	180.6
Mean	1.9	13	141	31.7	2.9	4	73	101.1
Median	1.9	8	127	34.4	2.9	5	66	98.7
25 th Percentile	-	4	100	28	1.9	2	50	75
75 th Percentile	-	20	190	42	3.3	5	80	110

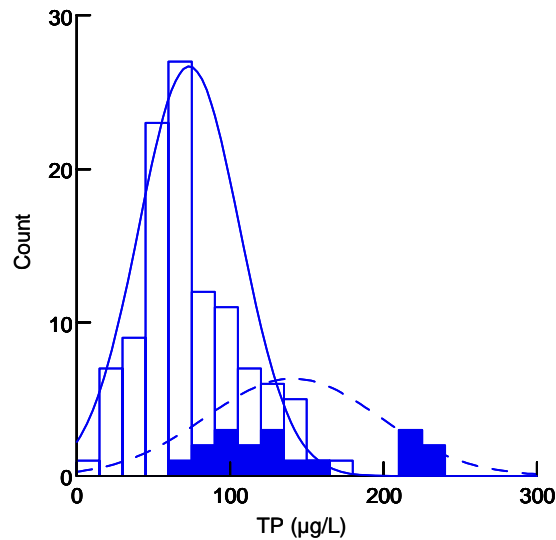


Figure 16. Frequency plot of total phosphorus concentration for times when dissolved oxygen saturation levels are ≤ 50 % (solid bars) and $>50\%$ (open bars).

This case study illustrates the application of an alternative approach to developing nutrient criteria in those situations where the existing database is inadequate for applying the more typical reference condition approaches.

16. The Special Case of Fjords

Fjords are coastal embayments characterized by having a sill at their seaward extension. They are usually formed as a result of glacial activity that scours the mouth of the embayment and, during retreat, deposits glacial debris that forms a sill. Fjords are most common in areas of recent glacial history such as Canada and northern Europe.

The presence of an outer sill results in a water column that typically becomes strongly stratified into two distinct layers: a surface layer that exchanges freely with offshore water, and a deeper layer landward of the sill that has a much more restricted exchange with offshore water. Whereas the surface waters of a fjord may have flushing times on the order of days, depending on the depth of the sill the deeper basin water may have exchange rates on the order of weeks, months and, in some cases, years. As a consequence of the restricted flushing of the deeper basin water, the bottom waters of fjords are much more susceptible to the accumulation of nutrients and organic materials which results in a greater likelihood of the development of hypoxic conditions.

There is no reason to believe that the same approaches used to develop nutrient criteria for other nearshore systems can not be applied to fjords. The difficulty, however, arises when attempting to apply the reference condition approach based on percentiles because of the limited number of fjords that may be present in a particular ecoregion, and the greater variability in assimilation capacity among fjords as a result the effect that sill depth has on both stratification and flushing rates. For this reason, it is likely that the most appropriate method for setting nutrient criteria for a fjord would be to use the historical data approach which does not require that a classification system be developed.

17. Conclusions

It is important that anyone attempting to establish nutrient criteria for nearshore marine waters realize that the procedures described above and in greater detail in the USEPA document (2001a), are not all inclusive, and that as we learn more about how coastal systems respond to nutrient inputs, other approaches for developing nutrient guidelines are likely to be developed. This is a new and rapidly growing area of study and we are far from having reached the point where any particular approach can be considered as being the best. It is especially important to realize that, because nearshore marine waters have been much less extensively studied than offshore waters in most regions of Canada, in many instances the amount and quality of data available may not be adequate to employ the approaches described above, but that other approaches, aside from that of establishing a monitoring program to collect the required data, may prove to be reasonable given the data limitations. An example of this situation is clearly presented in the PEI case study.

The USEPA approach to setting nutrient criteria for nearshore marine waters is a relatively new development, however, it is the most extensively documented approach when compared to those being proposed or developed by other jurisdictions. The

approach has been developed over a long period of time by a large number of experts and, perhaps most importantly, is the only approach for setting nutrient criteria in marine nearshore waters which has reached the point of being documented in a detailed stepwise manner. The requirement of pristine or near-pristine sites for establishing nutrient criteria can be challenging and would require that more emphasis be placed on finding and using sites that, although not considered pristine, have what may be considered to be 'acceptable' water quality. Because of the heterogeneity among nearshore waters, classifying sites into 'similar' categories based on structural and functional characteristics may also prove challenging. To meet these challenges, the importance of the Regional Technical Advisory Groups (see Fig. 4) in providing expertise and direction can not be over emphasized.

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APPENDIX I.

The OSPAR COMPREHENSIVE PROCEDURE (OSPAR 1997)

Assessment criteria and their assessment levels within the Comprehensive Procedure

In order to enable Contracting Parties to undertake a harmonised assessment of their waters subject to the Comprehensive Procedure it was necessary to develop a number of the qualitative assessment criteria into quantitative criteria that could be applied in a harmonised way. On the basis of common denominators within a wide range of qualitative and quantitative information provided by Contracting Parties on the criteria and assessment levels already used, a set of assessment criteria were selected and further developed into quantitative criteria for use in a harmonised assessment. It should also be noted that, although the levels against which assessment is made may be region-specific, the methodology for applying these assessment criteria is based on a common approach.

The assessment criteria selected for further development fall into the following categories:

- *Category I Degree of nutrient enrichment*
- *Category II Direct effects of nutrient enrichment*
- *Category III Indirect effects of nutrient enrichment*
- *Category IV Other possible effects of nutrient enrichment*

The main interrelationships between the assessment parameters and their categories are shown in Figure I.

Agreed harmonised assessment criteria and their assessment levels

For each criterion an assessment level has been derived (based on a level of elevation) with the exception of nutrient inputs for which there should also be an examination of trends. The level of elevation is defined, in general terms, as a certain percentage above a background concentration. The background concentration is, in general terms, defined as a salinity related and/or region specific derived spatial (offshore) and/or historical background concentration.

In order to allow for natural variability in the assessment, the level of elevation is generally defined as the concentration of more than 50 % above the salinity related and/or region specific background level (e.g. DIN and DIP concentrations).

The agreed harmonised assessment criteria and their respective assessment levels of the Comprehensive Procedure.
Assessment parameters
<p>Category I Degree of Nutrient Enrichment</p> <ul style="list-style-type: none"> • 1 Riverine total N and total P inputs and direct discharges (RID) Elevated inputs and/or increased trends (compared with previous years) • 2 Winter DIN- and/or DIP concentrations¹ Elevated level(s) (defined as concentration > 50 % above² salinity related and/or region specific natural background concentration) • 3 Increased winter N/P ratio (Redfield N/P = 16) Elevated cf. Redfield (> 25)
<p>Category II Direct Effects of Nutrient Enrichment (during growing season)</p> <ul style="list-style-type: none"> • 1 Maximum and mean Chlorophyll a concentration Elevated level (defined as concentration > 50 % above² spatial (offshore) /historical background concentrations) • 2 Region/area specific phytoplankton indicator species Elevated levels (and increased duration) • 3 Macrophytes including macroalgae (region specific) Shift from long-lived to short-lived nuisance species (e.g. <i>Ulva</i>)
<p>Category III Indirect Effects of Nutrient Enrichment (during growing season)</p> <ul style="list-style-type: none"> • 1 Degree of oxygen deficiency Decreased levels (< 2 mg/l: acute toxicity; 2 - 6 mg/l: deficiency) • 2 Changes/kills in Zoobenthos and fish kills Kills (in relation to oxygen deficiency and/or toxic algae). Long term changes in zoobenthos biomass and species composition • 3 Organic Carbon/Organic Matter Elevated levels (in relation to III.1) (relevant in sedimentation areas)
<p>Category IV Other Possible Effects of Nutrient Enrichment (during growing season)</p> <ul style="list-style-type: none"> • 1 Algal toxins (DSP/PSP mussel infection events) Incidence (related to II.2) 2 Other values less than 50 % can be used if justified

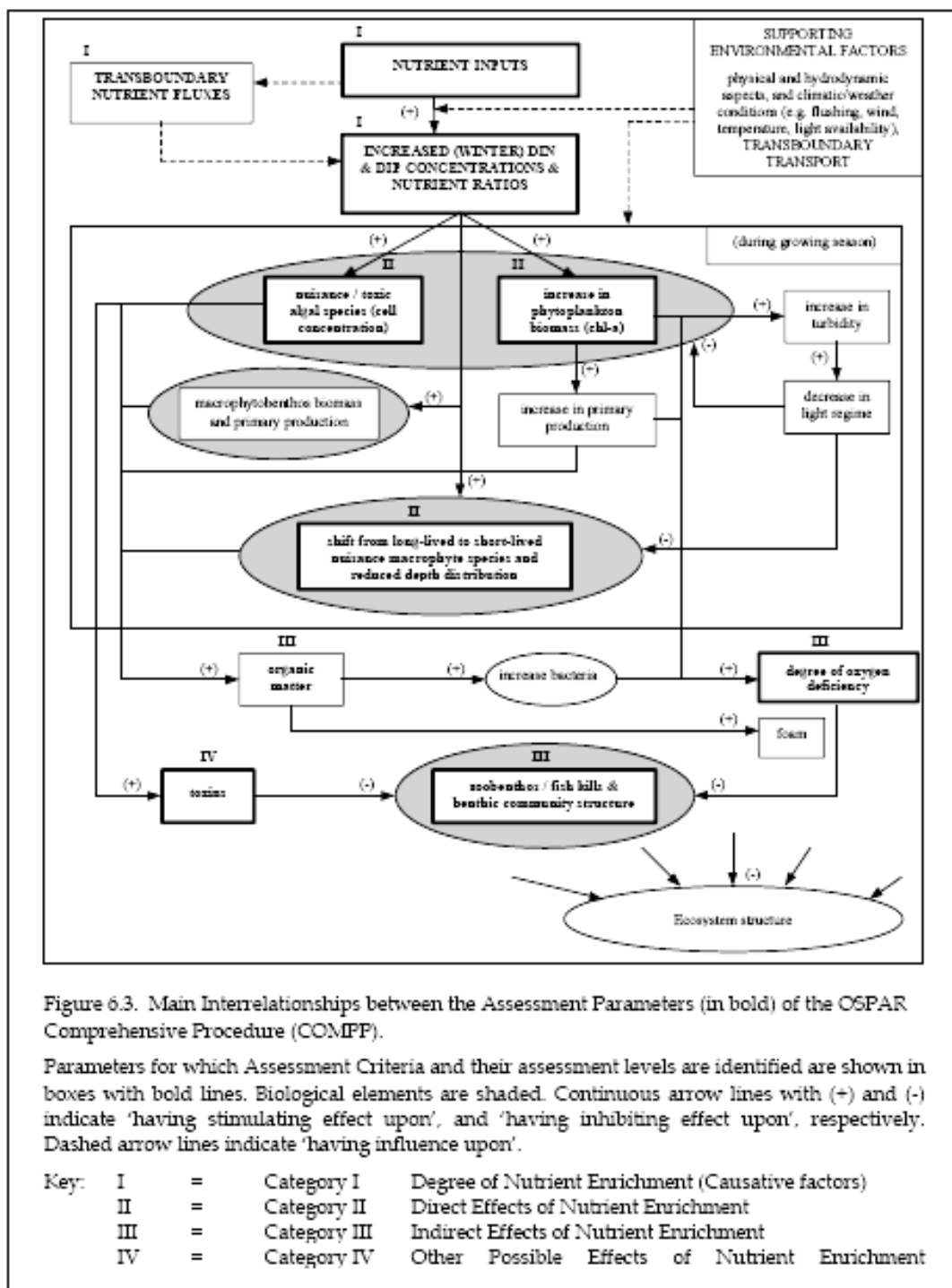


Fig. A1. Main Interrelationships Between the Assessment Parameters (in bold) of the OSPAR Comprehensive Procedure.

Classification on the basis of the harmonised assessment criteria and their respective assessment levels

For a harmonised holistic assessment of eutrophication status of an area one needs at least to address the common assessment parameters listed in the four categories of the assessment procedure.

To carry out the classification of the eutrophication status of areas of the maritime region each Contracting Party should undertake a number of steps, which are outlined below.

The first step is to provide a score for each of the harmonised assessment criteria being applied according to Table 1.

The second step will bring these scores together according to Table II to provide a classification of the area.

The third step is to make an appraisal of all relevant information (concerning the harmonised assessment criteria their respective assessment levels and the supporting environmental factors), to provide a transparent and sound account of the reasons for establishing a particular status for the area.

Finally this process should enable the classification of the maritime area in terms of problem areas, potential problem areas, and non-problem areas.

Integration of Categorised Assessment Parameters for Classification

The assessment levels of the agreed harmonised assessment criteria form the basis of the first step of the classification.

The next step is the integration of the categorised assessment parameters mentioned in Table I to obtain a more coherent classification. For each assessment parameter of Categories I, II, III and IV mentioned in Table I it can be indicated whether its measured concentration relates to a problem area, a potential problem area or a non-problem area as defined in the OSPAR Strategy to Combat Eutrophication. The results of this step are summarised in Table I and explained below:

- a. Areas showing an increased degree of nutrient enrichment accompanied by direct and/or indirect/other possible effects are regarded as **‘problem areas’**;
- b. Areas may show direct effects and/or indirect or other possible effects when there is no evident increased nutrient enrichment, e.g. as a result of transboundary transport of (toxic) algae and/or organic matter arising from adjacent/remote areas. These areas could be classified as **‘problem areas’**;
- c. Areas with an increased degree of nutrient enrichment, but without showing direct, indirect/ other possible effects, are initially classified as **‘potential problem areas’**;
- d. Areas without nutrient enrichment and related (in) direct/other possible effects are considered to be **‘non-problem areas’**.

Integration of Categorized Assessment Parameters for Classification				
	Category I (Degree of nutrient enrichment)	Category II (Direct Effects)	Category III and IV (Indirect effects/other possible effects)	Classification
A	+	+	and/or +	problem area
B	-	+	and/or +	problem area*
C	+	-	-	potential problem area
D	-	-	-	non-problem area

(+) = Increased trends, elevated levels, shifts or changes in the respective assessment parameters in Table I.
 (-) = Neither increased trends nor elevated levels nor shifts nor changes in the respective assessment parameters in Table I.

Note: Categories I, II and/or III/IV are scored ‘+’ in cases where one or more of its respective assessment parameters is showing an increased trend, elevated level, shift or change.

Supporting Environmental Factors

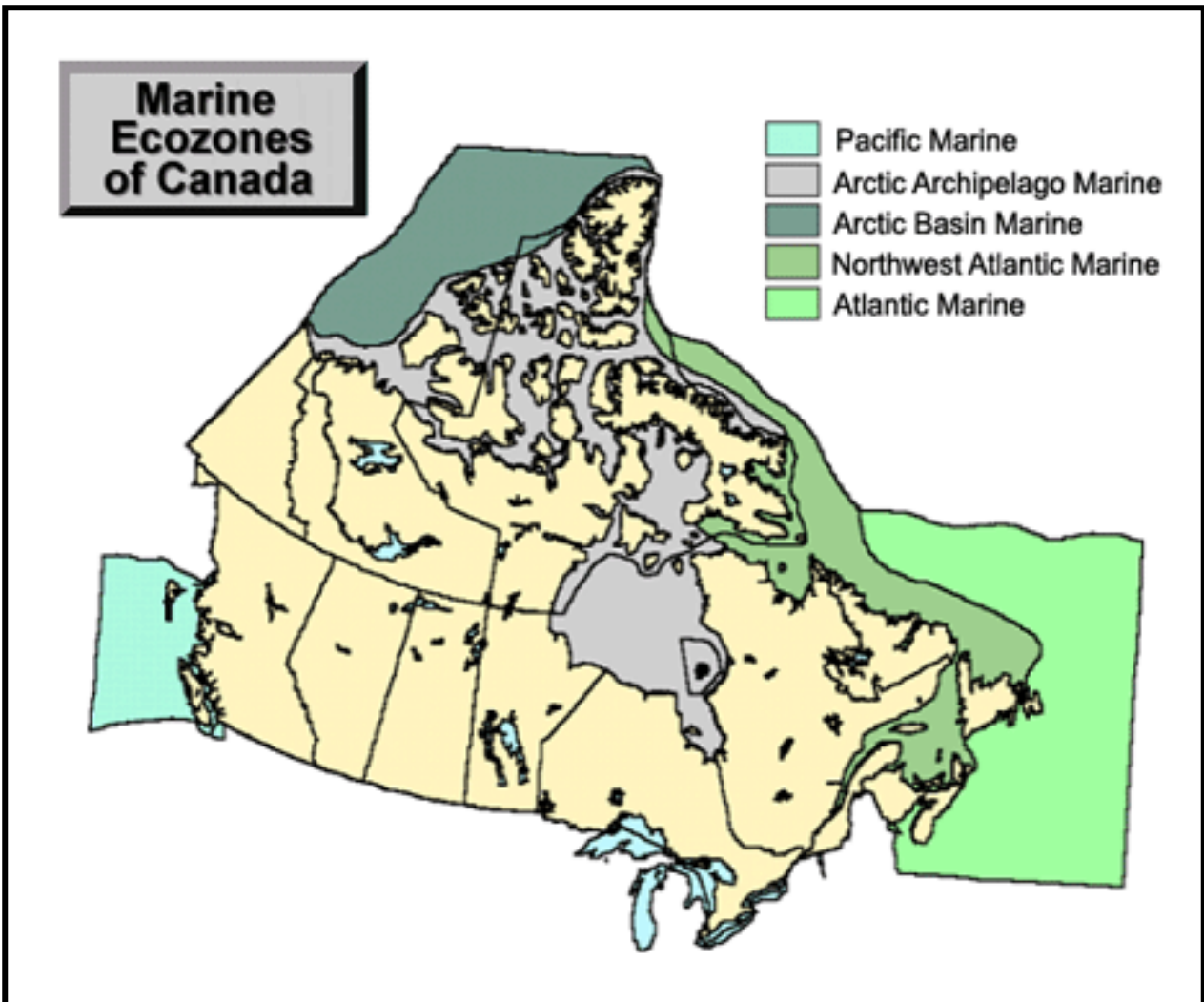
Region specific characteristics should be taken into account, such as physical and hydrodynamical aspects, and weather/climate conditions. These region specific characteristics may play a role in explaining the results of the classification.

*Caused by transport from other parts of the maritime area.

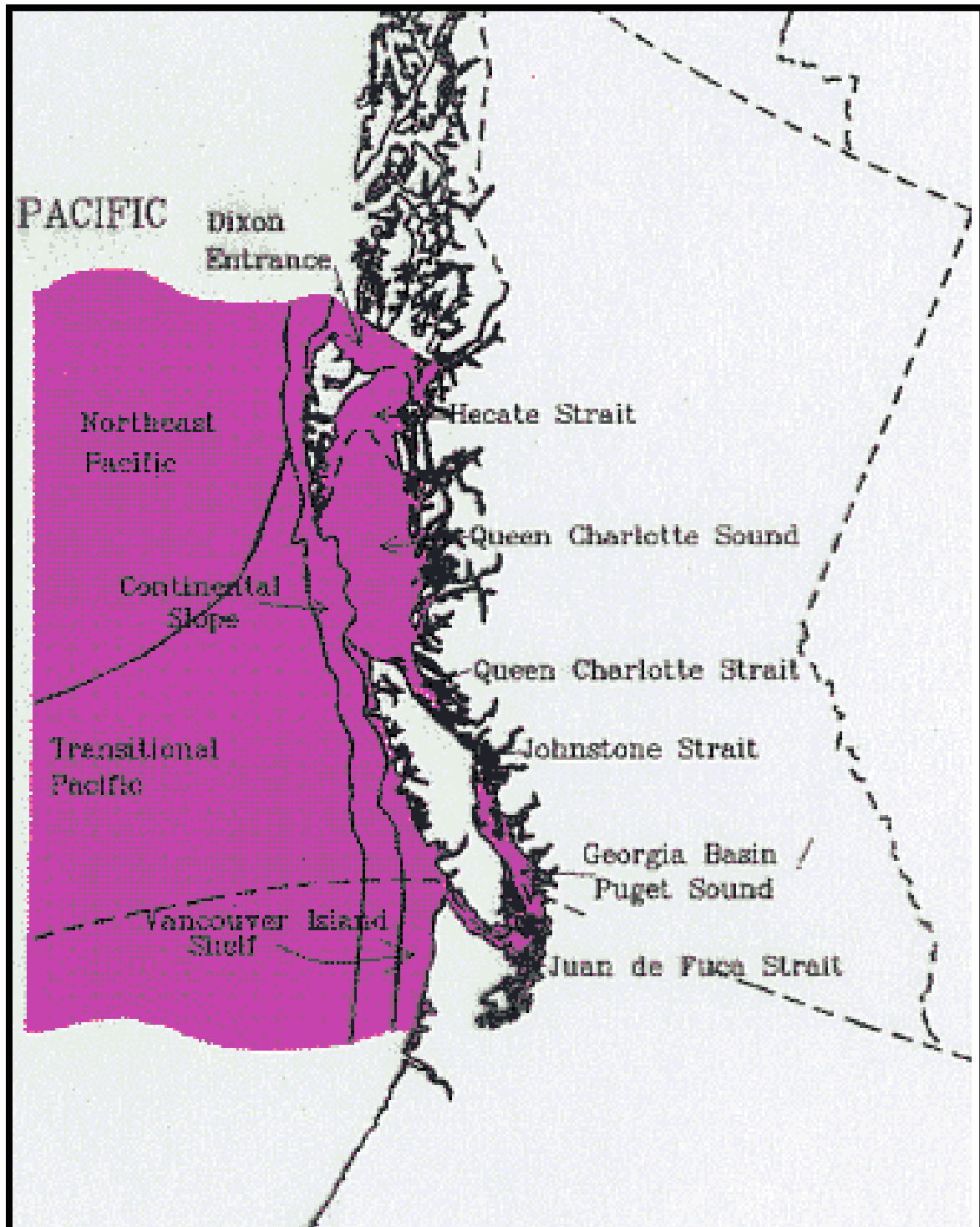
APPENDIX II

Marine Ecological Classification System for Canada

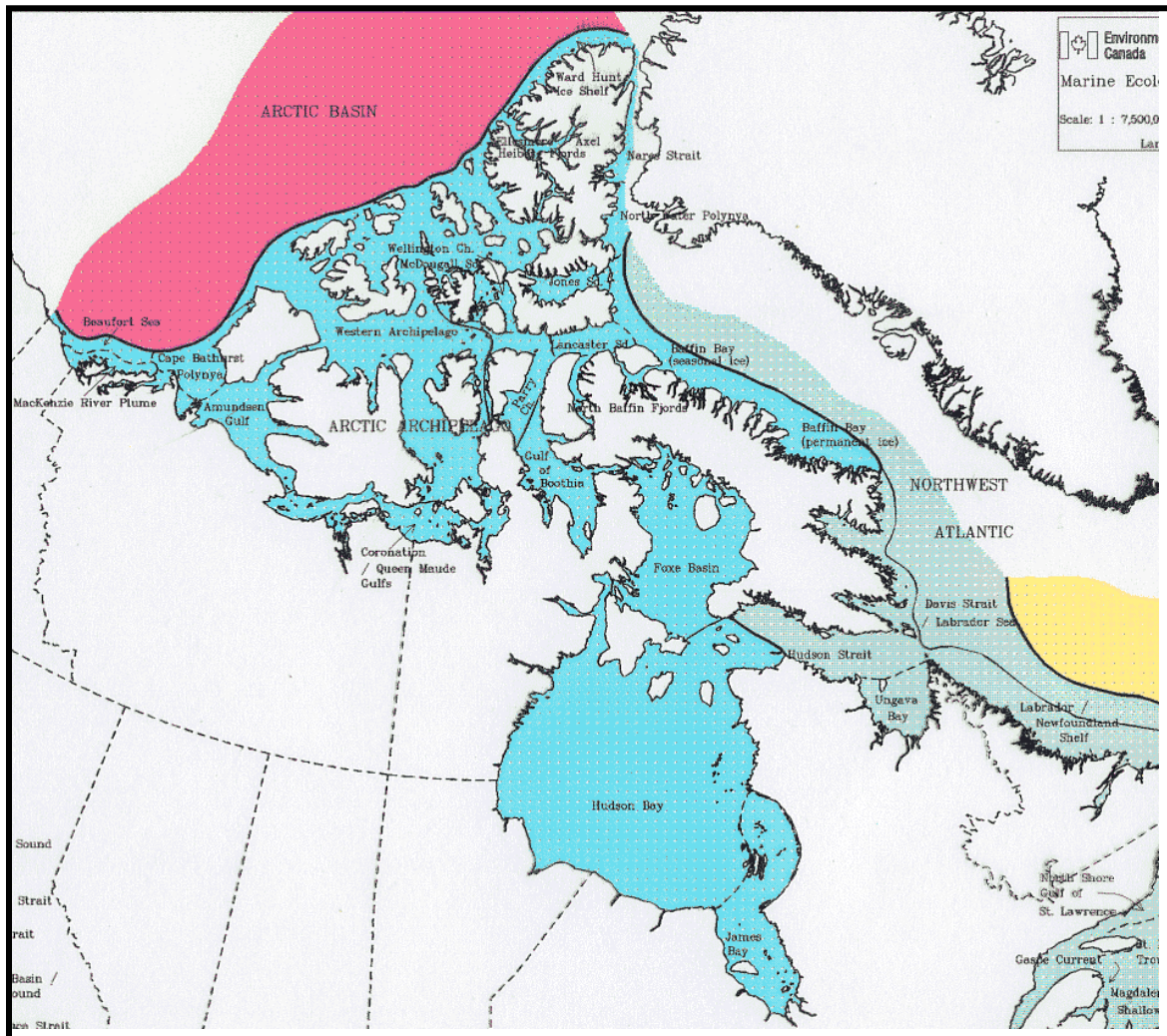
Marine Ecozones of Canada



The Pacific Marine Ecozones



The Arctic Basin and Arctic Circle Ecozones



The Atlantic and Northwest Atlantic Ecozones

