A NUTRIENT GUIDANCE FRAMEWORK FOR CANADIAN NEARSHORE WATERS

Prepared for

Environment Canada Water Quality Task Group

and

Canadian Council of Ministers of the Environment

By

M. Brylinsky
Acadia Centre for Estuarine Research
Acadia University
Wolfville, Nova Scotia

September 2006

Publication No. 84 of the Acadia Centre for Estuarine Research

Table of Contents

Page

1. Introduction 1
2. Defining Nutrient Enrichment 2
3. Consequences and Symptoms of Nutrient Over-enrichment 2
4. Nutrients of Concern 4
   4.1 Nitrogen 4
   4.2 Phosphorus 6
   4.3 Silicate 6
   4.4 Iron 6
   4.5 Nutrient Ratios 7
5. Sources of Nutrients 7
6. Nutrient Sinks 8
7. Indicators of Nutrient Over-enrichment 9
   7.1 Characteristics of Indicators 10
   7.2. Indicators Based on Organic Matter Supply 11
   7.3 Nutrient and Phytoplankton Based Indices 11
   7.4 Trophic Index for Marine Systems (TRIX) 12
   7.5 NOAA Index 13
   7.6 OPAR Comprehensive Procedure Index 13
   7.7. NEEA Overall Eutrophication Index (OEA) 14
   7.8 Assessment of Trophic Status (ASSETS) 16
   7.9 Estuarine Health Index (EHI) 16
   7.10 EQUATION 18
   7.11 DPSIR Index 19
   7.12 OAERRE Index 19
   7.13 PNCERS Index 20
   7.14 Indices Based on Phytoplankton Community Composition 20

8. Factors That Determine Susceptibility to Nutrient Over-enrichment 20
8.1 Physiographic Setting 21
8.2 Dilution 21
8.3 Water Residence Time 21
8.4 Water Column Stratification 21
8.5 Turbidity 22
8.6 Biological Communities 22
9. Indices of Susceptibility 23
9.1 Overall Level of Human Influence (OHI) 23
9.2 Eutrophication Risk Index (EUTRISK) 25
9.3 EQUATION 26
9.4. Other Indices 27
10. Approaches to Development of Nutrient Criteria 27
10.1 European Initiatives 28
10.2 U.S. Environmental Protection Agency 29
10.3 Australia and New Zealand 29
11. Role of Models 31
11.1 Regression models 31
11.2 Multivariate Models 32
11.3 Watershed Export-Coefficient Models 32
11.4 Simulation Models 33
12. Proposed Nutrient Guidance Framework for Canadian Estuaries
and Nearshore Waters 33
12.1 Introduction 33
12.2 Establish Regional Technical Assistance Group 34
12.3 Determine Scientific Basis for Criteria Development 36
12.4 Develop Coastal System Classification Scheme 36
12.5 Select Key Indicator Variables for Assessing System State 38
12.6. Determine Database Requirements and Availability 38
12.7 Determine Trophic Status and Establish Reference Conditions 39
12.8 Establish Nutrient Criteria 43

13. Acknowledgements 43

14. References 44

Case Study 1 Application of the nutrient guidance framework to nearshore
coastal systems along the Atlantic Nova Scotia coast 56

Case Study 2 Application of the nutrient guidance framework to the Boughton
River Estuary, Prince Edward Island 67

Appendices 56

Appendix I OSPAR Comprehensive Procedure 74

Appendix II Marine Ecological Classification System for Canada 79

List of Tables

Page

Table 7.1 Coastal system trophic status based on organic matter supply 11

Table 7.2. Criteria for evaluating the trophic status of marine systems 12

Table 7.3 Trophic status classification based on nutrient and chlorophyll 13

Table 7.4 OSPAR ecological quality elements and objectives for monitoring and assessing the biological response to nutrient enrichment based on the 2002 Bergen Declaration 14

Table 7.5 Eutrophication survey parameters 15

Table 7.6 Matrix used in the NEEA approach to determine overall level of eutrophic conditions 17

Table 7.7 EQUATION index components, objectives and data requirements ... 18

Table 9.1 Determination of estuary dilution potential 23

Table 9.2 Determination of estuary flushing potential 24

Table 9.3 Matrix for determining level of susceptibility of an estuary to nutrient over-enrichment based on dilution and flushing potential 24

Table 9.4 Matrix for determining level of overall human influence based on susceptibility and nutrient inputs 25

Table 9.5 Grading system for assessing coastal system vulnerability to nutrient over-enrichment 27

Table 9.6 Determining appropriate guideline trigger **v**alues 30

Table 12.1 Summary of approaches for reference condition determination 40

List of Figures

Page

Fig. 4.1 Nitrogen cycle ………………………………………………………… 5

Fig. 7.1 DPSIR assessment framework for eutrophication in coastal waters ….. 19

Fig. 9.1 Variables and their interrelationships used in the EUTRISK and PSA indices ……………………………………………………………………. 26

Fig 12.1 Elements of USEPA nutrient criteria development process …………. 35

Fig 12.2 Hypothetical frequency distribution of nutrient-related variables showing quantiles for reference high-quality data and mixed data …………… 41

Fig 12.3 Hypothetical example of load/concentration response of estuarine biota to increased enrichment ………………………………………………. 42

Fig 12.4 Illustration of the comparison of past and present nutrient data to establish a reference condition for intensively degraded estuaries …………….. 42

A NUTRIENT GUIDANCE FRAMEWORK FOR CANADIAN NEARSHORE WATERS

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 population growth 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, Conley (2000) has estimated that phosphorus loading to estuaries has increased 2-6 fold. Within North America, it has been estimated that 60% of US 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 a problem in Canada. A notable exception to this is some 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 and the resulting subsequent decrease in property values of coastal areas. As a result, the Canadian Council of Ministers of the Environment, in partnership with Environment Canada, has decided that a guidance framework be developed for managing the 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.

The major objectives of this document are to:

1. Develop a draft guidance framework that addresses nearshore marine eutrophication using a science-based approach to establish site-specific guidelines for managing nearshore marine eutrophication, and

2. Carry out tow case studies to demonstrate the utility of the guidance framework with respect to the data required and how the framework may 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 water body", 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 2001). 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 which can be present under conditions when dissolved oxygen is low, are not toxic themselves. Nutrient over-enrichment, however, 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, ammonia 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 1999; NRC 2000; USEPA 2001).

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 provide important habitat 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 that it can lead to the development of low levels of dissolved oxygen 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 may be related to excessive nutrient inputs in some instances. 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 (Hallegreaff et al. 1995).

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 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 2000).

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. 4.1). 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.

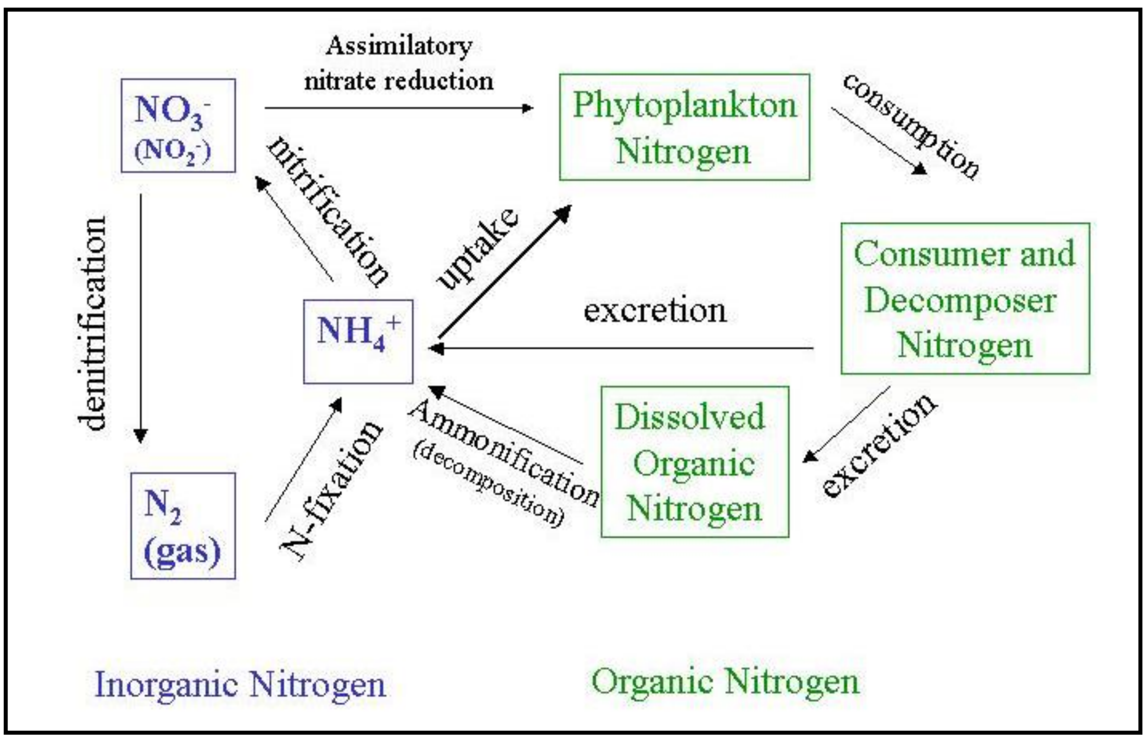


Fig. 4.1 Aquatic nitrogen cycle.

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 et al. 1996). 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 ratios[1] 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 P 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 2001).

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 1996), 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

[1] Redfield (1933; 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.

loading rate and how this correlates with the time at which the system is most susceptible to eutrophication, for example, in the summer when freshwater inputs and flushing rates may be lowest (Vallino and Hopkinson 1998).

Van Breemen et al. (2002) carried out a large-scale comprehensive study in which nitrogen budgets were determined 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.

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; 2000; 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 is 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).

In some coastal areas, the transport of nitrogen into coastal areas from offshore can be greater than land-based inputs (Howarth et al. 1996). 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 Straight/Straight of Georgia/Puget Sound estuarine system along the coast of British Columbia (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. The ability of flagellates to swim is also thought to lessen their chance of sinking to bottom sediments. 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. Setzinger et al. (1988) 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%, and that it can be related to the water residence time of an estuary.

The uptake of nitrogen by 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. Within 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 are currently in the process of developing indicators for a diversity of coastal systems (OSPAR 1997). 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 indictor 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 smoother 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.

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 have been proposed, the choice of which set is best 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 shown in Table 7.1.

Table 7.1 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 has been criticized for being 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 et al. 1992). Vollenweider et al. (1998) and others (Nixon 1992) 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 is used (Table 7.2).

Table 7.2. Criteria for evaluating the trophic status of marine systems (Vollenweider 1998).

Trophic Status	**TN ($mg\ m^{-3}$)**	**TP ($mg\ m^{-3}$)**	**Chlorophyll *a* ($\mu g\ L^{-1}$)**	**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:

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

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. These indices (Table 7.3) were included along with a total of 16 nutrient related parameters (see Table 7.5).

Table 7.3 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 *a* (ug 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 managing 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 7.4).

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 2004). Details of the step by step procedure employed in this approach are contained in Appendix I.

Table 7.4 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 *a*	Maximum and mean chlorophyll *a* 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 the U.S. National Oceanographic and Atmospheric Administration (NOAA) under the National Estuarine Eutrophication Assessment (NEEA), Bricker et al. (1999) developed an Overall Eutrophication Condition (OEC) tropic status index based on a subset of 16 parameters (Table 7.5) 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 7.5 Eutrophication survey parameters (from Bicker et al. 1999).

	PARAMETERS	EXISTING CONDITIONS (minimum values observed over a typical annual cycle)	TRENDS
ALGAL CONDITIONS	CHLOROPHYLL A	Surface concentrations: Hypereutrophic (>60 ug /l) High (>20, ≤60 ug/l), Medium (>5, ≤20 ug/l), Low (>0, ≤5 ug/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 factors5
	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 OXTGEN	**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]
ECOSYTEM/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 7.6) 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 subjective 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.

It should be noted that nutrient concentration is not included as part of these criteria since in their survey Bricker et al. (1999) were unable to find a clear relationship between nutrient concentration and the degree of eutrophication.

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).

Table 7.6 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	**1**	**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	**0.6**	**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	**0.3**	**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	**0.3**	**0.6**	**1**
		Low Secondary Symptoms	**Moderate Secondary Symptoms**	**High Secondary Symptoms**

7.10 EQUATION (<u>**E**</u>stuarine <u>**QU**</u>ality and Condi<u>**TION**</u>)

EQUATION is an integrated estuarine quality and condition index based on the four characteristics illustrated in Table 7.7 (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 is 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 available for download from 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 (1997) have reviewed the responses of benthic communities to hypoxic conditions.

Table 7.7 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 *a*, 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 icthyofaunal 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 (**D**riving forces, **P**ressures, **S**tate, **I**mpact, **R**esponses) Index

The European Environmental Agency is currently in the process of developing an eutrophication index based on an assessment of parameters that represent driving forces, pressures, system state, impact and responses (EEA 2001). 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 been 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.

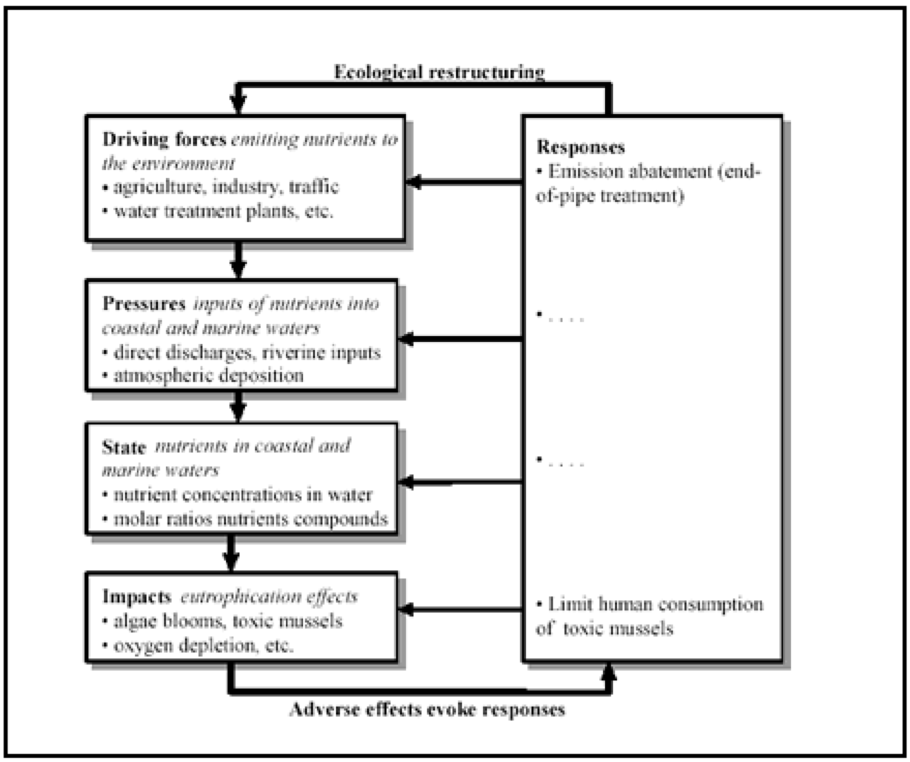


Figure 7.1. DPSIR assessment framework for eutrophication in coastal waters (from EEA 2001).

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). 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 K. 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. 1997; NRC 2000; USEPA 2001).

The assimilation capacity of a coastal system depends on a number of factors related to how it processes nutrients once it 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 water body that is replaced per unit of time. Both of these terms are commonly used to describe the flushing characterises 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 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 9.1 to 9.4 illustrate how dilution and export potential are determined.

Table 9.1 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 9.2 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 9.3.

Table 9.3 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 SUSCEPTABILITY				
FLUSHING POTENTIAL	HIGH	LOW SUSCEPTABILITY	LOW SUSCEPTABILITY	MODERATE SUSCEPTABILITY
FLUSHING POTENTIAL	MODERATE	LOW SUSCEPTABILITY	MODERATE SUSCEPTABILITY	HIGH SUSCEPTABILITY
FLUSHING POTENTIAL	LOW	MODERATE SUSCEPTABILITY	HIGH SUSCEPTABILITY	HIGH SUSCEPTABILITY
		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 9.4 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 (Duron 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. 9.1).

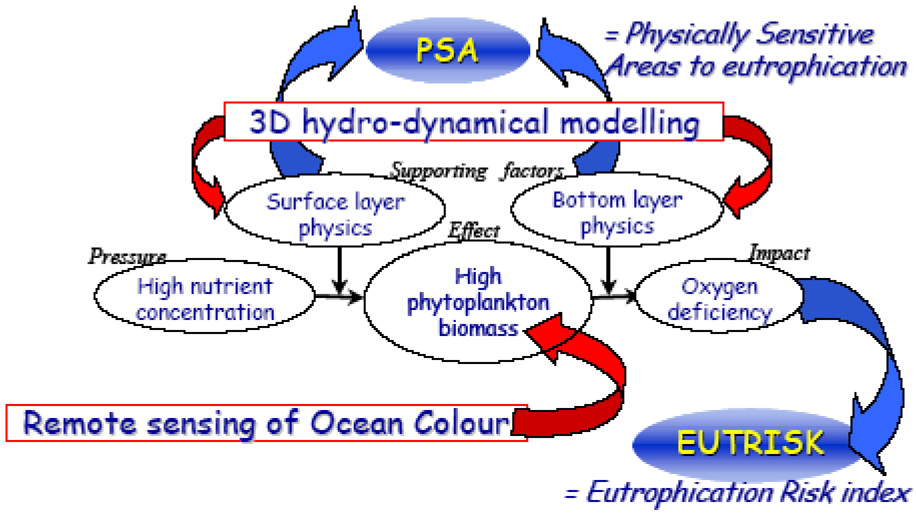


Figure 9.1 Variables and their interrelationships used in the EUTRISK and PSA indices (from Druon et al. 2002).

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 9.5 lists the scoring system for the first three components.

Table 9.5 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, large datasets and development of 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 procedures for arriving at nutrient criteria that have been developed by various agencies. In most cases these procedures 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 (1977; 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, 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 entirely 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 2001). 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 a 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 9.6 lists the basic steps recommended in developing the guidelines.

Table 9.6 Determining appropriate guideline trigger **v**alues (from ANZEC 2000).

I. Define Primary Management Aims • Define the water body (scientific information, monitoring data, classify ecosystem type) • Determine **environmental values** to be protected • Determine level of protection • Identify environmental concerns 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' - often defined in biological terms
II. Determine appropriate Guideline Trigger Values for selected indicators • 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
III. Apply the Trigger Values using (risk-based) Decision Trees or Guideline 'packages' • Water quality monitoring data • Site specific environmental information • Effects of ecosystem-specific modifying factors. - 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 and 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 1985; 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 turbidly 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 multinominal 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 indicted 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 some 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; USEPA 2001).

12. Proposed Nutrient Guidance Framework for Canadian Estuaries and Nearshore Waters

12.1 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, the complexity and our limited 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, which is essentially an empirically based approach to recognizing the differences between water bodies in terms of their response to nutrient inputs, appears to be the most practical approach at present.

The details and procedures on how to apply the reference condition approach have been most extensively developed and documented by the USEPA and for this reason it is recommended that this approach be adopted for Canadian nearshore waters. Other national and international agencies are largely still in the development stage of establishing nutrient guidelines for nearshore waters and have not yet put forward a comprehensive procedural framework comparable to that developed by the USEPA. Although the USEPA guidelines have been developed for U.S. estuaries, the general approach is generic enough to be applied to Canadian coastal waters without any severe restrictions, and it would be a major undertaking to develop similar guidelines specific to Canadian nearshore waters. Another advantage to adopting the USEPA guidelines is that it provides detailed alternative procedures for developing nutrient criteria in those instances where no pristine sites exist and the reference condition approach can not be adopted. In addition, although it is strongly recommended that the approach be carried out by a multidisciplinary team of expects to ensure that the best effort is made in establishing guidelines, in those instances where guidelines are required but time and financial resources are limited, it is possible to apply the approach on a more limited scale.

One major shortcoming, 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 guidelines.

The basic approach described in the USEPA guidelines is illustrated in Fig. 12.1. The following sections provide details of each step of the process. Additional details and a more comprehensive discussion of each step can be obtained from USEPA (2001).

12.2 Establish Regional Technical Assistance Groups

To be most effective, application of the reference condition approach requires 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 regional agencies. Potential sources of members include federal, provincial and municipal agencies, academic institutions, local community groups and private sector organizations.

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.

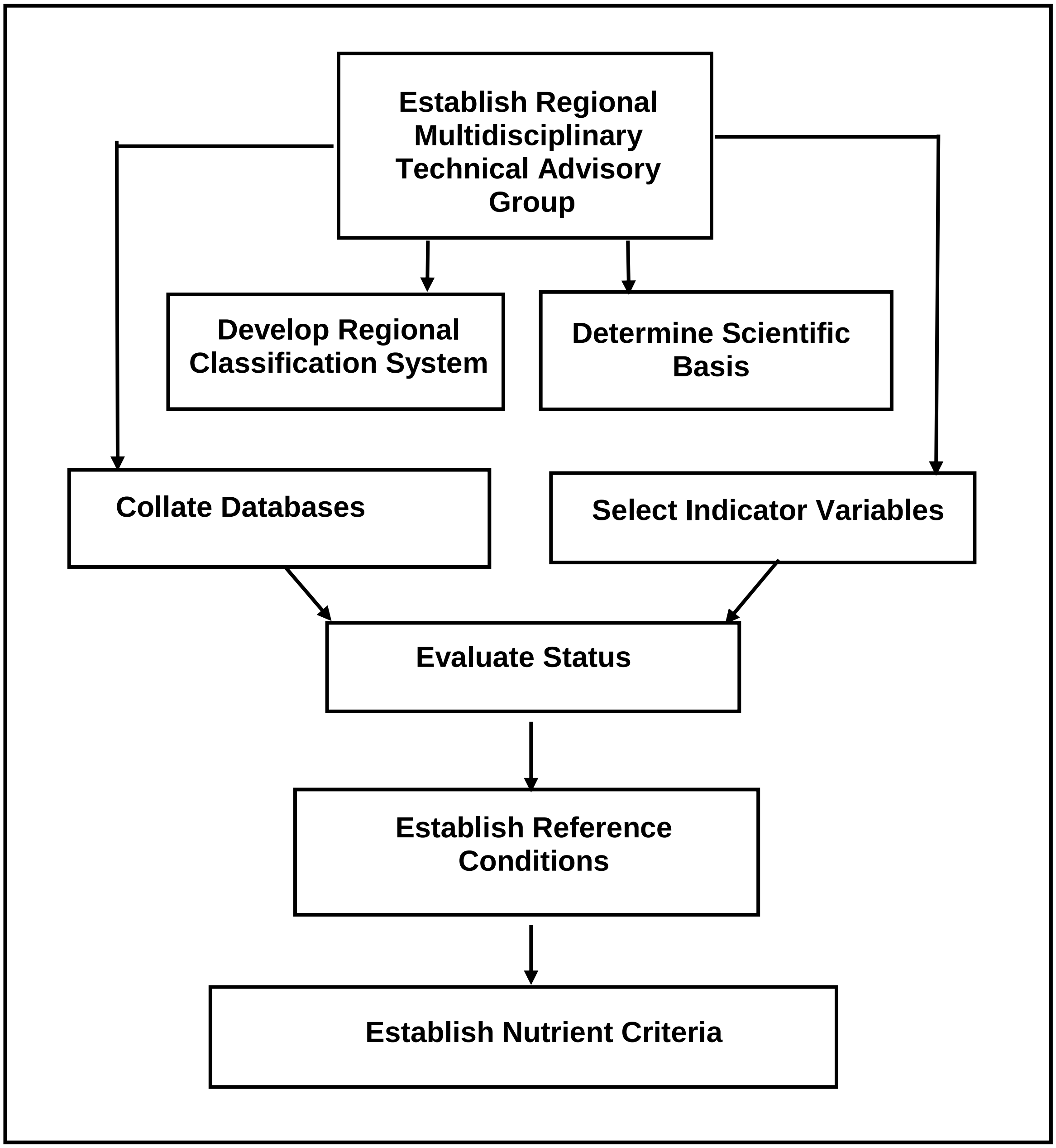


Fig. 12.1 Elements of USEPA nutrient criteria development process.

12.3 Determine Scientific Basis for Criteria Development

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 2001). 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 1992) 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 symptomatic 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 coastal system being addressed, other parameters may also be included.

12.4 Develop Coastal System 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 heir susceptibility to nutrient over-enrichment. This is perhaps one of the most difficult aspects of the approach and one 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 (Turner 2001; Livingston 2001; Jay et al. 2000). The USEPA (2001) 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 system based on their response to a variety of stressors, including nutrient over-enrichment (USEPA 2004).

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 recent scheme developed for Canadian marine systems (Harding 1997), which classifies Canada's marine areas according to ecozones, ecoprovinces, ecoregions, and ecodistrists (see Appendix II), is a good beginning, but additional classification focusing more directly on susceptibility to nutrient over-enrichment will have to also 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.

The general approach used in the NOAA Estuarine Eutrophication Survey (Bricker et al. 1991) for determining susceptibility to nutrient over-enrichment is an excellent framework for accomplishing this. Details of this approach are presented in Section 9 and it is suggested that this be adopted. The multivariate approach employed by Strain and Yeats (1999) for a number of Nova Scotian coastal system may also provide information useful 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.

12.5 Select Key Indicator Variables for Assessing System State

The parameters used to determine the current status of a coastal system 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.

12.6. 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 are made to obtain historical data that allows an evaluation of the degree to which the system has changed over time with respect to it 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 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 especially useful in those instances where no pristine sites are present in a particular region for establishing reference conditions.

12.7 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 category and are suitable for establishing reference conditions. In those cases where pristine sites for a particular category do not exist, it will be necessary to use hindcasting approaches based on historical or sediment core/paleoecological data or model simulations to determine appropriate reference conditions.

Five basic steps are required for development of nutrient criteria (USEPA 2001):

- Examination of any historical records that are available, including paleoecological evidence
- Determination of the reference condition using one of the several approaches described below and in Table 12.1
- Use of empirical modelling or proxy data sets in those cases where there is insufficient data on the historical characteristics of the system, or where no pristine watershed exists for a particular ecoregion/physical classification
- Interpretation of all available information by the regional experts
- Determination that the final criteria ensures that the designated use of the water body is protected and does not result in excessive nutrient input to downstream waters

In those instances where pristine sites can be identified, and there is data on more than one pristine site within a particular ecoregion/physical category, a frequency distribution approach is suggested to arrive at a reference condition and that the upper 75th percentile be used (Fig 12.2). When no data are available for a particular pristine system, it will be necessary to design a sampling program for collection of data on the causative and response indicator parameters previously decided to be most appropriate.

Near pristine systems are those in which there is some evidence of current or past activity within the watershed that is likely to be causing some anthropogenic nutrient input to the system. In this case the lower 25th percentile should be used (12.2).

The task of establishing reference conditions becomes much more difficult in those situations where all of the coastal systems within a regional category are experiencing significant nutrient over-enrichment and no pristine sites exist. In this case, a diversity of approaches for establishing reference conditions are suggested and the choice of which approach is most appropriate depends on the kinds and extent of databases available.

Table 12.1 Summary of approaches for reference condition determination (from USEPA 2001).

Degree of Apparent Degradation	Method Recommended	Criterion Measure
A. In situ Observations as the Basis for Estuarine Reference Conditions		
1. Recognized unique excellent condition	Upper quartile or median ambient concentration (Fig 12.2).	Concentration of TP, TN, chl a, Secchi Depth
2. Some degradation, but reference sites exist	Lower quartile (Fig 12.2)	Same as above
3. Significantly degraded including all reference sites	Empirical hindcasting or statistical modelling of long term historical data sets (Fig. 12.3 and 12.4).	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

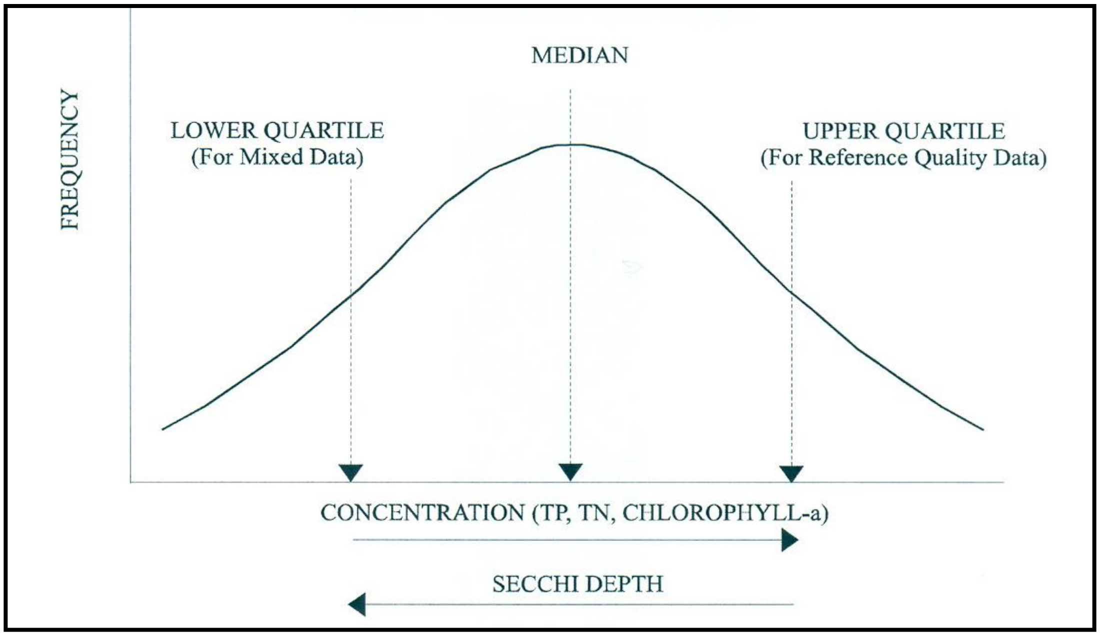


Fig 12.2 Hypothetical frequency distribution of nutrient-related variables showing quantiles for reference high-quality data and mixed data (from USEPA 2001).

If long-term historical data is available it may be possible to arrive at reference conditions based on an empirical analysis of the 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. Fig 12.3 illustrates how nutrient criteria can be arrived at using this 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 form 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.

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.

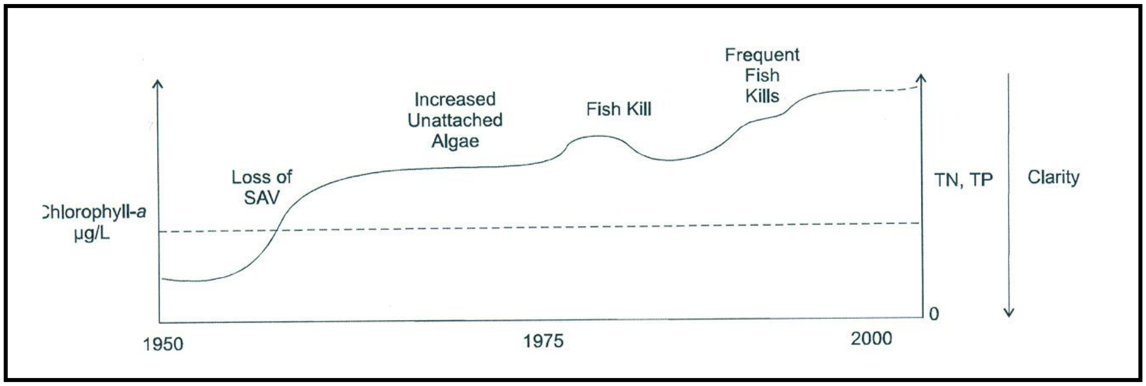


Fig 12.3 Hypothetical example of load/concentration response of estuarine biota to increased enrichment. Dashed line represents the selected reference condition level.

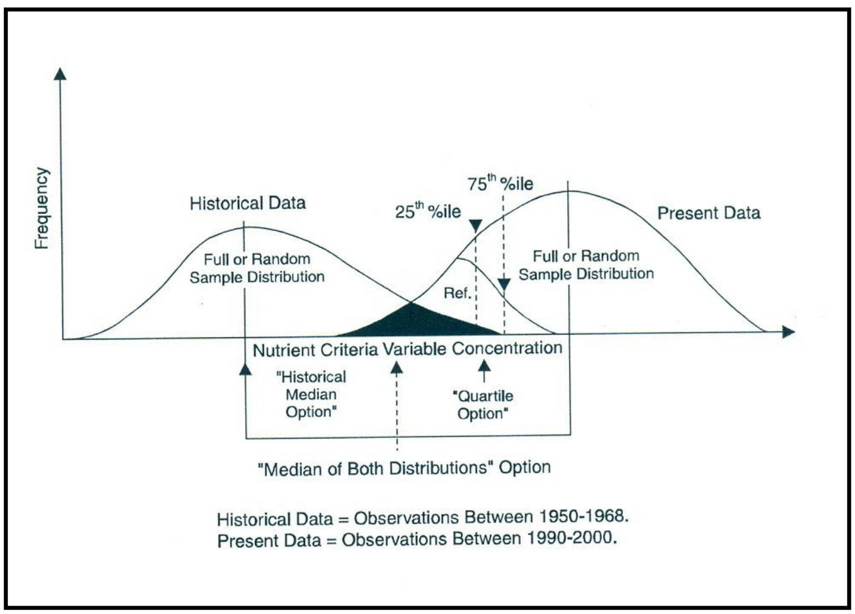


Fig 12.4 Illustration of the comparison of past and present nutrient data to establish a reference condition for intensively degraded estuaries. The option of selecting the distributions from both time periods is compared to an expected frequency distribution if the observations were available (from USEPA 2001).

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.

12.8 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 manages 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).

13. Acknowledgements

Peter Strain and Gary Bugden of the Bedford Institute of Oceanography reviewed early versions of the manuscript and provided many valuable comments. Elizabeth Roberts, Sushil Dixit, and Cindy Crane acted as project managers and reviewers. Other reviewers included Bruce Raymond, Darrell Taylor, Don Fox, Danielle Pelletier and Narender Nagpal.

14. References

Alpine, A.E. and J.E. Cloern. 1992. Trophic interactions and direct physical effects control phytoplankton biomass and production in an estuary. Limnol. Oceanogr. 37(5): 946-955.

Alvera-Azcarate, A., J.G. Ferreira and J.P. Nunes. 2003. Modelling eutrophication in mesotidal and macrotidal estuaries: The role of intertidal seaweeds. Est. Coast. Shelf Sci. 57:715-724.

Anderson, D.M. 1989. Toxic algal blooms and red tides: a global perspective. p. 11-16. *In*, T. Okaichi, D.M Anderson and T. Nemoto [eds.], Red Tides, Biology, Environmental Science and Toxicology. Elsevier, New York.

ANZECC. 2000. National Water Quality Management Strategy: Australian and New Zealand Guidelines for Fresh and Marine Water Quality. Australian and New Zealand Environment and Conservation Council and Agriculture and Resource Management Council of Australia and New Zealand.

Atkinson, M.J. and S.V. Smith. 1983. C:N:P ratios of benthic marine plants. Limnol. Oceanogr. 28(3):568-574.

Bergen Declaration. 2002. Fifth International Conference on the Protection of the North Sea, 20-21 March 2002. Bergen, Norway.

Billen, G. and J. Garnier. 1997. The Phison River plume: coastal eutrophication in response to changes in land use and water management in the watershed. Aquat. Microb. Ecol. 13:3-17.

Boesch, D.F., D.M. Anderson, R.A. Horner, S.E. Shumway, P.A. Tester and T.E. Whitledge. Harmful algal blooms in coastal water: Options for prevention, control and mitigation. NOAA Coastal Ocean Program Decision Analysis Series No. 10. Special Joint Report with the National Fish and Wildlife Foundation. U.S. Department of Commerce and U.S. Department of the Interior.

Bolam, S.G. and T.F. Fernandes. 2002. The effects of macroalgal cover on the spatial distribution of macrobenthic invertebrates: The effect of macroalgal morphology. Hydrobiologia. 475/476:437-448.

Borum, J. 1996. Shallow waters and land/sea boundaries, p. 179-203. *In*, American Geophysical Union [ed.], Eutrophication in Coastal Marine Systems. Coastal and Estuarine Studies. Volume 52:179-203.

Boynton, W.R. 1997. Potomac River integrated analysis project. Chesapeake Biological Laboratory. University of Maryland. Solomons, Maryland.

Boynton, W.R., J.D. Hagy, L. Murray, C. Stokes and W.M. Kemp. 1996. A comparative analysis of eutrophication patterns in a temperate coastal lagoon. Estuaries 19(2B):408-421.

Boynton, W.R. and W.M. Kemp. 2000. Influence of river flow and nutrient loads on selected ecosystem processes: a synthesis of Chesapeake Bay data, p. 269-298. *In*, J.E. Hobbie [ed.], Estuarine Science, a Synthetic Approach to Research and Practice. Island Press, Washington, D.C.

Boynton, W.R., W.M. Kemp and C.W. Keefe. 1982. A comparative analysis of nutrient and other factors influencing phytoplankton production, p. 69-90. *In*, V. Kennedy [ed.], Estuarine Comparisons. Academic Press, San Diego.

Bricker, S.B., C.G. Clement, D.E. Pirhalla, S.P. Orlando and D.R.G. Farrow. 1999. National estuarine eutrophication assessment effects of nutrient enrichment in the nation's estuaries. NOAA, National Ocean Service, Special Projects Office and the National Centers for Coastal Ocean Science. Silver Spring, MD: 71p.

Bricker, S.B., J.G. Ferreira and T. Simas. 2003. An integrated methodology for assessment of estuarine trophic status. Ecol. Mod. 169:39-60.

Chambers, P.A., Guy, M., E.S. Roberts, M.N. Charlton, R. Kent, C. Gagnon, G. Grove and N. Foster. 2001. Nutrients and their impact on the Canadian environment. Agriculture and Agri-Food Canada. Environment Canada, Fisheries and Oceans Canada, Health Canada and Natural Resources Canada. 241p

Chapelle, A., P Lazure, and M. Menesguen. 104. Modelling eutrophication events in a coastal ecosystem: Sensitivity analysis. Est. Coast. and Shelf. Sci. 39:529-548.

Cloern, J.E. 1982. Does benthos control phytoplankton biomass in South Francisco Bay? Mar. Ecol. Prog. Ser. 9:191-202.

Cloern, J.E. 1987. Turbidity as a control on phytoplankton biomass and productivity in estuaries. Cont. Shelf Res. 7(11/12):1367-1381.

Cloern, J.E. 1999. The relative importance of light and nutrient limitation of phytoplankton growth: a simple index of coastal ecosystem sensitivity to nutrient enrichment. Aquatic Ecology. 33:3-16.

Cloern, J.E. 2001. Our evolving conceptual model of the coastal eutrophication problem. Mar. Ecol. Prog. Ser. 210:223-253.

Coffaro, G. and A. Sfriso. 1997. Simulation model of *Ulva rigida* growth in shallow water of the Lagoon of Venice. Ecol. Mod. 102(1); 55-66.

Cognetti, G. 2001. Marine eutrophication: The need for a new indicator system. Mar. Poll. Bul. 42(3):163-164.

Cole, J.J, B.L. Peierls, N.F. Caraco and M.L. Pace. 1993. Nitrogen loading of rivers as a human-driven process, p. 141-157. *In* M.J. McDonell and S.T.A. Picket [eds.], Humans as components of ecosystems: The ecology of subtle human effects and populated areas. Springer-Verlag.

Conley, D.J. 2000. Biogeochemical nutrient cycles and nutrient management strategies. Hydrobiologia. 410:87-96.

Conley, D.J., C.L. Schelske and E.F. Stoermer. 1993. Modification of the biogeochemical cycle of silica with eutrophication. Mar. Ecol. Prog. Ser. 1001:179-192.

Cooper, J.A.G, A.E. Ramm and T.D. Harrison. 1994. The Estuarine Health Index: A new approach to scientific information transfer. Ocean and Coast Manag. 25:103-141.

CSTT. 1997. Comprehensive studies for the purpose of Article 6 and 8.5 of DIR 91/271 EEC, the Urban Waste Water Treatment Directive, 2nd ed. Published for the Comprehensive Studies Task Team of Group Coordinating Sea Disposal Monitoring by the Department of the Environment for Northern Ireland, the Environment Agency, the Scottish Environmental Protection Agency and the Water Services Association, Edinburgh.

D'Elia, C.F., J.G. sanders and W.R. Boynton. 1986. Nutrient enrichment studies in a coastal plain estuary; phytoplankton growth in large-scale continuous cultures. Ca. J. Fish. Aquat. Sci. 43:397-406.

Dettmann, E.H. 2001. Effect of water residence time on annual export of nitrogen in estuaries: a model analysis. 24:481-490.

Diaz, R.J and R. Rosenberg. 1995. Marine benthic hypoxia: a review of its ecological effects and the behavioural responses of benthic macrofauna. Oceanogr. Mar. Biol. Ann. Rev. 33:245-303.

Dillon, P.J. and F.H. Rigler. 1975. A simple method for predicting the capacity of a lake for development based on lake trophic status. J. Fish. Res. Bd. Canada. 31(11):1711-1778.

Druon, J.N., W. Schrimpf, S. Dobricic and S. Stips. 2002. The physical environment as a key factor in assessing the eutrophication status and vulnerability of shallow seas: PSA and EUTRISK (v1.0) Office for Official Publications of the European Communities. p. 40.

EC 2005. Canada's National Program of Action for the Protection of the Marine Environment from Land-Based Activities.(http://www.npa-pan.ca/issues/glossary/ index_e.htm#indicators).

ECOHAB 1993. The ecology and Oceanography of harmful algal blooms: A national research agenda. http://www.redtide.whoi.edu/hab/nationplan/ECOHAB/ECOHABhtml.html.

ECOSTAT. 2004. Review of eutrophication in European water policy and conceptual framework. ECOSTAT WG 2A, Version 2.4.

Edwards, V.R., P. Tett and K.J. Jones. Changes in the yield of chlorophyll *a* from dissolved available inorganic nitrogen after an enrichment event – applications for predicting eutrophication in coastal waters. Cont. Shelf Res. 23:1771-1785.

EEA. 1999. Nutrients in European Systems. Environmental Assessment Report No. 4. Office for Official Publications of the European Community. 40 p.

EEA. 2001. Eutrophication in Europe's coastal waters. European Environment Agency Topic Report 7/2001.

Ferreira, J.G. 2000. Development of an estuarine quality index based on key physical and biogeochemical features. Ocean. Coastal. Manage. 43(1):99-122.

Fisher T.R., P.R. Carlson ant R.T. Barker. 1982. Sediment nutrient regeneration in three North Carolina estuaries. Estuar. Coast. Shelf Sci. 14:101-116.

Fisher, T.R., E.R. Peele, J.W. Ammerman and L.W. Harding. 1992. Nutrient limitation of phytoplankton in Chesapeake Bay. Mar. Ecol. Prog. Ser. 82:51-63.

Flemer, D.A. 1970. Primary production in Chesapeake Bay. Chesapeake Sci. 11:117-129.

Fritzpatrick, J.J., J.C. Imhoff, E. Burgess and R. Brashear. 2001. Water quality models: A survey and assessment. Final Report for Project 99-WSM-5. Water Environment Research Foundation, Alexandria, Virginia.

Galloway, J.N., W.H. Schlesinger, C. Levy, A. Michaels and J.L. Schnoor. Nitrogen fixation: anthropogenic enhancement environmental response. Global Biogeochem. Cycles. 9:235-252.

Giovanardi, F. and E. Tromellini. 1992. Statistical assessment of trophic conditions: Application of the OECD methodology to the marine environment. Sci. Total. Environ. Suppl:211-233.

Gowen, R.J. 1994. Managing eutrophication associated with aquaculture development. J. Appl. Icthy. 10:242-257.

Gowen, R.J., P. Tett and P. Jones. 1992. Predicting marine eutrophication: the yield of chlorophyll from nitrogen in Scottish coastal waters, Mar. Ecol. Prog. Ser. 85:153-161.

Greening, H.S., G. Morrison, R.M. Eckenrod and M.J. Perry. 1997. The Tampa Bay resource-based management approach, p. 349-355. *In*, S.F. Treat [ed.], Proceedings, Tampa Bay Area Scientific Information Symposium 3: Applying Our Knowledge. Tampa Bay Estuary Program. St. Petersburg, Florida.

Gregory, D., B. Petrie, F. Jordan and P. Langille. 1993. Oceanographic, geographic and hydrological parameters of Scotia-Fundy and Southern Gulf of St. Lawrence Inlets, Can. Tech. Rpt. Hydro. and Ocean Sci. No. 143. 248p.

Gregory, D. and B. Petrie. 1994. A classification scheme for estuaries and inlets. Coastal Zone Canada 1994 Proceedings, p. 1884-1893.

Hakanson, L. 1994. A review of effect-dose-sensitivity models for aquatic ecosystems. Int. Revue ges. Hydrobiol. 79(4):621-667.

Hallegraeff, G.M. 1993. A review of harmful algal blooms and their apparent global increase. Phycologia. 32:79-99

Harding, L.E. 1997. A Marine Ecological Classification System for Canada. Report Prepared for the Marine Environmental Quality Advisory Group of Environment Canada. 57p.

Hinga, K.R., H. Jeon and N.F. Lewis. 1995. Marine Eutrophication Review. NOAA Coastal Ocean Program Decision Analysis Series No. 4. NOAA Coastal Ocean Office, Silver Springs, MD. 156 p.

Hopkinson, C.S. and J.J. Vallino. 1995. The relationships among man's activities in watersheds and estuaries: A model of runoff effects on patterns of community metabolism. Estuaries. 18(4):598-621.

Howarth, R.W. and R. Marino. 1998. A mechanistic approach to understanding why so many estuaries and brackish waters are nitrogen limited. *In*, T. Hellstrom [ed.][, Effects of nitrogen in the aquatic environment. KVA Report 1, Swedish Academy of the Sciences, Stockholm.

Howarth, R.W., D.P. Swaney, T.J. Butler and R. Marino. 2000. Climatic control on eutrophication of the Hudson River Estuary. Ecosystems. 3:210-215.

HydroGeologic. 1999. Selection of water quality components for eutrophication-related Total Maximum Daily Load Assessments. Task 4; Documentation of review and evaluation of eutrophication models and components. EPA Contact No. 68-C5-0020

Work Assignment No. 24. Prepared by HydroGeologic, Inc. Herndon, Va. and AQUA TERRA Consultants, Mountain View, Ca.

ICES. 2001. Report of the ICES Advisory Committee on Ecosystems. ICES Cooperative Research Report 249.

Innamorati, M. and F. Giovanardi. 1992. Interrelationship between phytoplankton biomass and nutrients in the eutrophicated areas of the north-western Adriatic Sea, p. 235-250. *In*, R.A. Vollenweider, R. Marchetti and R. Viviani [eds.], Marine Coastal Eutrophication, J. Science of the Total Environment, Suppl. 1992. Elseiver, Amsterdam.

Janicki, J. and D. Wade.1996. Estimating critical external nitrogen loads for the Tampa Bay estuary: An empirically based approach to setting management targets. Final Report. Tampa Bay Estuary Program, St. Petersburg, Florida.

Jaworski, N.A., R.W. Howarth and L.J. Hetling. 1997. Atmospheric deposition of nitrogen oxides onto the landscape contributes to coastal eutrophication in the northeastern U.S. Environ. Sci. Tech. 31:1995-2004.

Jay, D.A. 2000. An ecological perspective on estuarine classification, p.149-176. *In*, J.E. Hobbie [ed.], Estuarine science: A synthetic approach to research and practice. Island Press, Washington, D.C.

Johnes, P.J. 1996. Evaluation and management of the impact of land use change on the nitrogen and phosphorus load delivered to surface waters: the export coefficient modelling approach. J. Hydrol. 183:323-349.

Jorgensen, S.E. 1994. Fundamentals of Ecological Modelling. Elsevier, Amsterdam.

Jorgensen, B.B. 1996. Material flux in the sediment, p. 115-135. *In*, B.B. Jorgensen and K. Richardson [eds.], Eutrophication in coastal marine ecosystems. American Geophysical Union, Washington, D.C.

Karydis, M. and G. Tsirtsis. 1996. Ecological indices: A biometric approach for assessing eutrophication levels in the marine environment. Sci. Total Environ. 186:209-219.

Kemp, W.M., P. Sampow, J. Caffey, M. Maper, K. Henkunsion and W.R. Boynton. 1990. Ammonium recycling versus denitrification in Chesapeake Bay sediments. Limnol. Oceanogr. 35:1545-2563.

Kennedy, V.S. 1984. The Estuary as a Filter. Academic Press.

Law C.S., A.P. Rees and N.J.P. Owens. 1991. Temporal variability of denitrification in estuarine sediments. Estuar. Coast. Shelf Sci. 33:37-56.

LePape, O.L., Y. Del Amo, A. Menesguen, A. Aminot, B. Quequiner and P. Treguer. 1996. Resistance of a coastal ecosystem to increasing eutrophic conditions: the Bay of Brest (France), a semi-enclosed zone of Western Europe. Cont. Shelf Res. 16(15):1885-1907.

Lowery, T.A. 1998. Modelling estuarine eutrophication in the context of hypoxia, nitrogen loadings, stratification and nutrient ratios. J. Env. Manage. 52:289-305.

Harrison, P.J., T.R. Parsons, F.J.R. Taylor and J.D. Fulton. 1983. Review of the biological oceanography of the Strait of Georgia: Pelagic Environment. Can J Fish Aquat Sci 40: 1064-1094

Hopkinson, C.S. and J.J. Vallino. 1995. The relationships among man's activities in watersheds and estuaries: A model of runoff effects on patterns of estuarine community metabolism. Estuaries. 18(4):598-621.

Martin, J.H., R.M. Gordon and S.E. Fitzwater. 1991. The case for iron. Limnol. Oceanogr. 36:1793-1802.

Malone, T.C., D.J. Conley, T.F. Fisher, P.M. Gilbert, L.W. Harding and K.C. Sellner. 1996. Scales of nutrient-limited phytoplankton productivity in Chesapeake Bay. Estuaries. 19:371-385.

Marino, R., F. Chan, R.W. Howarth, M. Pace and G.E. Likens. 2002. Ecological and biochemical interactions constrain planktonic nitrogen fixation in estuaries. Ecosystems. 5:719-725.

Martin, J.H. and others. 1994. Testing the iron hypothesis in ecosystems of the equatorial Pacific Ocean. Nature. 371:319.328.

Merceron, M., M. Kempf, D. Bentley, J.D. Gaffet, J. LeGrand and L. Lamort-Datin. 2002. Environmental impact of a salmonid farm on a well flushed marine site; I. Current and water quality. J. Appl. Icthyol. 18:40-50.

Meeuwig, J.J. 1999. Predicting coastal eutrophication from land-use: an empirical approach to small non-stratified estuaries. Mar. Ecol. Prog. Ser. 176:231-241.

Meeuwig, J.J., J.B. Rasmussen and R.H. Peters. 1998. Turbid waters and clarifying mussels: their moderation of empirical chl:nutrient relations in estuaries in Prince Edward Island, Canada. Mar. Ecol. Prog. Ser. 171:139-150.

Meeuwig, J.J., P. Kauppilia and H. Pitkanen. 2000. Predicting coastal eutrophication in the Baltic: a Limnological approach. Can. J. Fish. Aquat. Sci. 57:844-855.

Mills, 2003. State of the Gulf Report: Nutrient Indicators. Report prepared for the Gulf of Maine Summit Planning Committee. NOAA Office of Ocean and Coastal Management.11p.

Monbet, Y. 1992. Control of phytoplankton biomass in estuaries: A comparative analysis of microtidal and macrotidal estuaries. Estuaries. 15:563-571.

Newton, A. J.D. Icely, M. Falcao, A. Nobre, J.P. Nunes, J.D. Ferreira and C. Vale. 2003. Evaluation of eutrophication in the Ria Formosa coastal lagoon, Portugal. Cont. Shelf Res. 23:1945-1961.

Nixon, S.W. 1995. Coastal marine eutrophication: A definition, social causes, and future concerns. Ophelia. 41:199-219.

Nixon, S.W. and M.E.Q. Pilson. 1983. Nitrogen in estuarine and coastal marine systems, p.565-648. In, E.J. Carpenter and D.G. Capone [eds.] Nitrogen in the marine Environment. Academic Press, New York.

Nixon, S.W., J.W. Ammerman, L.P. Atkinson, V.M Berounsky, G. Billen, W.C. Boicourt, W.R. Boyton, T.M. Church, D.M. DiToro, R. Elmgren, J.H. Garber, A.E. Giblin, R.A. Jahnke, N.J.P. Owens, M.E.Q. Pilson and S.P. Seitzinger. 1996. The fate of nitrogen and phosphorus at the land-sea margin of the North Atlantic Ocean. Biogeochemistry. 35:141-180.

NRC.1994. Priorities for Coastal Ecosystem Science. National research Council. National Academy Press, Washington, D.C.

NRC 2000. Clean coastal waters: Understanding and reducing the effects of nutrient pollution. National Academy Press. Washington, D.C.

Officer, C.B and J.H. Ryther 1980. The possible importance of silicon in marine eutrophication. Mar. Ecol. Prog. Ser. 3:83-91.

OSPAR. 1997. Comprehensive Procedure for the Identification of the Eutrophication Status of the Maritime Area. Agreement 1997-11.

OSPAR. 2001. Draft common assessment criteria and their application within the Comprehensive Procedure of the Common Procedure. *In*, OSPAR convention for the protection of the marine environment of the North-East Atlantic [ed.] Proceedings of the meeting of the Eutrophication Task Group (ETG), London , 9-11 October 2001.

Oviatt, C.A., P. Doering, B. Nowicki, L. Reed, J. Cole and J. Frithsen. 1995. An ecosystem level experiment on nutrient limitation in temperate coastal marine environments. Mar. Ecol. Prog. Ser. 116:171-179.

Paerl, H.W. 1997. Coastal eutrophication and harmful algal blooms: Importance of atmospheric deposition and groundwater as "new" nitrogen and other nutrient sources. Limnol. Oceanogr. 42(5, part 2):1154-1165.

Paerl, H.W. and J.P. Zehr. 2000. Nitrogen fixation. *In*, D.F. Kirchman [ed.], Marine Microbial Ecology. Academic Press, San Diego.

Painting, S.J., S.I. Rogers, D.K. Mills, E.R. Parker, M.J. Devlin, H.L. Rees, S.P. Milligan and P. Larcombe. 2004. Assessing the suitability of OSPAR EcoQOs vs. ICES Criteria. A U.K. Working Paper to the ICES Study Group to Review Ecological Quality Objectives for Eutrophication (SGEUT). Centre for Environment, Fisheries and Aquaculture Science, Great Britain.

Parrish, J.K. and K. Litle. 2001. PNCERS 2000 Annual Report. Coastal Ocean Programs, NOAA.

Pelegri, S.P., L.P. Nielsen and T.H. Blackburn. 1994. Denitrification in estuarine sediment stimulated by irrigation activity of the amphipod *Corophium volutator*. Mar. Ecol. Prog. Ser. 105:285-290.

Pelley, J. 1998. Is coastal eutrophication out of control? Environ. Sci. Tech. / News:462-466.

Peterson, D.H., L.E. Schemel, R.E. Smith, D.D. Harmon and S.W. Hager. 1987.The flux of particulate organic carbon in estuaries: phytoplankton productivity and oxygen consumption. U.S. Geological Survey Water Supply Series, Selected Papers in the Hydrologic Sciences, p. 41-49.

Petrie, B. and P. Yeats. 1990. Simple models of the circulation, dissolved metals, suspended solids and nutrients in Halifax harbour. Water Poll. Res. J. Can. 25:325-349.

Rabalais, N.N., R.E. Turner, D. Justic, Q. Dortch, W.J. Wiseman and B.K. Gupta. 1996. Nutrient changes in the Mississippi River and system responses on the adjacent continental shelf. Estuaries. 19:386-407.

Redfield, A.C. 1934. On the proportions of organic derivatives in sea water and their relationship to the composition of plankton James Johnstone Memorial Volume (Liverpool), p.176.

Redfield, A.C. 1958. The biological control of chemical factors in the environment. Am. Scientist. 46:205-222.

Ross, A.H., W.S.C. Gurney. A, M.R. Heath, S.J. Hay and E.W. Henderson. 1993. A strategic simulation model of a fjord ecosystem. Limnol. Oceanogr. 38:128-153.

Ryther, J.H. and W.M. Dunstan. 1971. Nitrogen, phosphorus and eutrophication in the coastal marine environment. Science. 171:1008:1013.

Seitzinger, S. P. 1990. Denitrification in aquatic sediments, p. 301-322. *In*, N.P. Revsbech and J Sorensen [eds.], Denitrification in soil and sediment. Plenum Press.

Seitzinger, S.P. and A.E. Giblin. 1996. Estimating denitrification in North Atlantic continental shelf sediments. Biogeochemistry. 35:235-260.

Sin, Y., R.W. Wetzel and I.C. Anderson. 1999. Spatial and temporal characteristics of nutrient and phytoplankton dynamics in the York River estuary Virginia: analysis of long-term data. Estuaries. 22(2A):260-275.

Smayda, T.J. 1990. Novel and nuisance phytoplankton blooms in the sea: evidence for a global epidemic, p. 29-40. In, E. Graneli, B. Sundstrom, L. Edler and D.M. Anderson [eds.], Toxic Marine Phytoplankton. Elsevier, New York.

Sowles, J. 2003. Nitrogen in the Gulf of Maine: Sources, susceptibility and trends. White Paper No 1. A Workshop on Nutrient Management in the Gulf of Maine. NOAA/UNH Cooperative Institute for Coastal and Estuarine Environmental Technology, the Gulf of Maine Council of the Marine Environment and NOAA's National Estuarine Ocean Service.

Stigebrandt, A. 2001. FJORDENV – a water quality model for fjords and other inshore waters... Earth Sciences Centre, Goteborg University, Goteborg, C40 2001. 41 p.

Strain, P.M. and B.T. Hargrave. 2005. Salmon aquaculture, nutrient fluxes and ecosystem processes in southwestern, New Brunswick. Chapter 2 in: The Handbook of Environmental Chemistry, Vol. 5: Water Pollution. Environmental effects of marine finfish aquaculture. Springer, Berlin Heidelburg.

Strain, P.M. and P.A. Yeats. 1999. The relationships between chemical measures and potential predictors of the eutrophication status of inlets. Mar. Poll. Bull. 38(12):1163-1170.

Taylor, D., S. Nixon, S. Granger and B. Buckley. 1995. Nutrient limitation and the eutrophication of coastal lagoons. Mar. Ecol. Prog. Ser. 127:235-244.

Tett, P. and 13 others. Eutrophication and some European waters of restricted exchange. Cont. Shelf Res. 23:1653-1671.

Townsend, D.W. 1998. Sources and cycling of nitrogen in the Gulf of Maine. J. Marine Systems. 16:283-295.

Uncles, R.J., A.I. Fraser, D. Butterfield, P. Johnes and T.R. Harrod. 2002. The prediction of nutrients into estuaries and their subsequent behaviour: application to the Tamar and comparison with the Tweed, U.K. Hydrobiologia. 475/476:239-250.

USEPA. 1996. Proceedings of the National Nutrient Assessment Workshop. EPA 822-R-96-004. Office of Water, United States Environmental Agency.

USEPA. 2001. Nutrient Criteria Technical Guidance Manual: Estuarine and Coastal Marine Waters. EPA-822-F-01-003. Office of Water. 4304.

USEPA. 2004. Classification Framework for Coastal Systems. U.S. environmental protection Agency, Office of Research and Development, National Health and Environmental Effects Research Laboratory, Research Triangle Park, N.C. Report No. EPA 600/R-04/061. 66p.

Valiela, I. 1995. Marine ecological processes. 2nd ed. Springer-Verlag.

Valiela, I, J. Costa, K. Foreman, J.M. Teal, B.L. Howes and D. Aubry. 1990. Transport of groundwater-borne nutrients form watersheds and their effects on coastal waters. Biogeochemistry. 10:177-197.

Valiela, I., J. McClelland, J. Hauxwell, P.J. Behr, D. Hersh and K. Foreman. 1997. Macroalgal blooms in shallow estuaries: Controls and ecophysiological and ecosystem consequences. Limnol. Oceanogr. 42(5, part 2):1105-1118.

Valiela, I., S. Mazzilli, J.L. Bowen, K.D. Kroeger, M.L. Cole, G. Tomasky and T. Isaji. 2004. ELM, an estuarine nitrogen loading model: Formulation and verification of predicted concentrations of dissolved inorganic nitrogen. Water, Air and Soil. Poll. 157:365-391.

Vallino, J.J. and C.S. Hopkinson. 1998. Estimation of dispersion and characteristic mixing times in Plum Island Sound Estuary. Estuar. Coast. Shelf. Sci. 46:333-350.

Vascetta, M., P. Kauppila and E. Furman. 2004. Indicating eutrophication for sustainability considerations by the trophic index TRIX: Does our Baltic case reveal its usability outside Italian waters? Peer Conference, 17 November 2004.

Van Breemen, E.W. Boyer, C.L. Goodale, N.A. Jaworski, K. Paustian, S.P. Seitzinger, K. Lajtha, B. Mayer, D. Van Dam, R.W. Howarth, K.J. Nadelhoffer, M. Eve and G. Billen. 2002. Where did all the nitrogen go? Fate of nitrogen inputs to large watersheds in the northeastern U.S.A.

Vincent, C., H. Heinrich, A. Edwards, K. Nygaard and J. Haythornthwaite. 2004. Guidance on typology, reference conditions and classification systems for transitional and coastal water. COAST 2.4HOL6 Final Draft. CIS Working Group.

Vollenweider, R.A. and J.J. Kerekes. 1982. Eutrophication of waters: Monitoring, assessment and control. Organization for Economic Co-operation and Development, Paris. 156 p.

Vollenweider, R.A., A. Rinaldi, and G. Montanari. 1992. Eutrophication, structure and dynamics of a marine coastal system: results of ten-year monitoring along the Emilia-Romagna coast (Northwest Adriatic Sea), p. 35-62. *In*, R.A. Vollenweider, R. Marchetti and R. Viviani [eds.], Marine coastal eutrophication. Elseiver, Amsterdam...

Vollenweider, R.A., F. Giovanardi, G. Montanari and A. Rinaldi. 1998. Characterization of the trophic conditions of marine coastal waters with special reference to the NW Adriatic Sea: Proposal for a trophic scale, turbidity and generalized water quality index. Environmetrics. 9:329-357.

Werme, C. 2001. Assessing, monitoring, and controlling nitrogen pollution in the Gulf of Maine. White Paper No. 2. Managing nitrogen impacts in the Gulf of Maine. A Workshop on Nutrient Management in the Gulf of Maine. NOAA/UNH Cooperative Institute for Coastal and Estuarine Environmental Technology, the Gulf of Maine Council of the Marine Environment and NOAA's National Estuarine Ocean Service.

Whitehead, P.G., E. Wilson and D. Butterfield. 1999. Water quality processes in catchments: an integrated modelling approach for scenario analysis, p. 85-100. In, S.T. Trudgill, D.E. Walling and B.W. Webb [eds.], Water quality: Processes and Policy. John Wiley and Sons.

Zhang, J. 2000. Evidence of trace metal limited photosynthesis in eutrophic estuarine and coastal water. Limnol. Oceanogr. 45(8):1871-1878.

Case Study 1

Application of the Nutrient Guidance Framework to Nearshore Coastal Systems Along the Atlantic Nova Scotia Coast

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 is to establish nutrient criteria for these systems using the reference condition approach based on an analysis of existing in situ data.

2. Study Sites

This case study is based largely on data previously 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 was 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 exists on the hydrographic and nutrient characteristics required to classify them according to their susceptibly to nutrient over-enrichment and their current tropic status. A total of 17 sites were identified as having the necessary data available (Figure 1).

The area in which these sites are located can be considered as a single ecoregion. The watersheds in this region all 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 that they are naturally productive systems as a result of offshore inputs of nutrient rich shelf waters (Petrie et al. 1999).

3. Approach

To apply the reference condition approach based on an analysis of the available in situ data, each site was evaluated with respect to its susceptibility to nutrient over-enrichment and its current trophic status. An index of susceptibility to nutrient over-enrichment was based on 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.

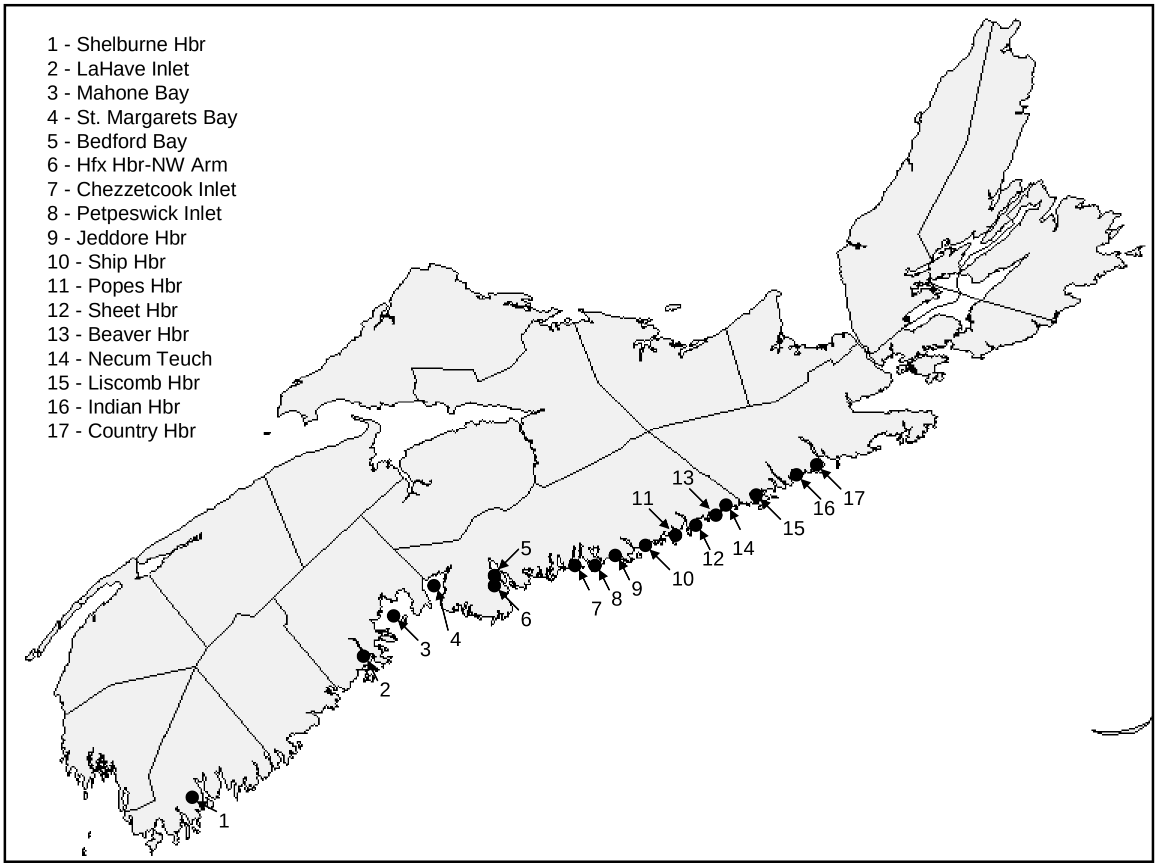


Figure 1. Location of sites selected for analysis.

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 Federal Department of Fisheries Ocean. The data available through the BIOCHEM database is largely that collected by Stain and Yeats (1999). Summaries of the hydrological data and chemical data are contained in Tables 4.1 and 4.2.

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

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

Table 4.1. Morphological and physical characteristics of each site.

Site	Watershed Area (sq km)	High Water Area (sq km)	Volume (10^9 cu m)	Maximum Depth (m)	Mean Tidal Range (m)	Tidal Prism	Tidal / Freshwater Volume Ratio	Flushing Time (hrs)
Beaver Hbr	13.5	13.5	0.121	21.3	1.4	18.4	723.1	88.1
Bedford Bay	16.8	16.8	0.510	71.0	1.5	24.8	109.4	261.4
Chezzetcook Inlet	14.4	14.4	0.009	5.5	1.4	12.8	165.2	14.0
Country Hbr	9.9	9.9	0.089	21.9	1.2	11.3	39.5	104.0
Hfx - NW Arm	93.1	93.1	1.759	71.0	1.6	146.4	373.0	155.3
Indian Hbr	11.4	11.4	0.116	20.7	1.4	14.9	299.9	102.9
Jeddore Hbr	21.1	21.1	0.083	18.3	1.4	28.3	177.6	42.3
LaHave Inlet	19.6	19.6	0.077	27.4	1.6	29.1	27.1	38.7
Liscomb Hbr	19.8	19.8	0.106	14.3	1.4	27.3	53.5	54.2
Mahone Bay	216.5	216.5	4.227	62.2	1.5	319.3	344.5	170.6
Necum Teuch Hbr	4.2	4.2	0.012	14.6	1.3	4.7	29.1	37.6
Petpeswick Inlet	14.0	14.0	0.030	11.0	1.4	14.5	186.3	31.5
Popes Hbr	11.3	11.3	0.057	21.9	1.4	15.2	716.5	52.5
Sheet Hbr	18.9	18.9	0.106	22.9	1.5	27.4	35.2	54.1
Shelburne Hbr	22.7	22.7	0.140	13.1	1.7	37.3	68.3	52.6
Ship Hbr	7.2	7.2	0.047	25.0	1.4	9.7	23.9	66.3
St. Margarets Bay	141.8	141.8	5.191	91.4	1.6	223.8	416.3	294.2

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*.

Table 4.2. 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

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

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 5.1. An overall index, based on all three factors was calculated by first assigning a value of 1, 2 or 3 for low, medium and high, respectively, and then summing these. The results are summarized in Table 5.2.

Table 5.1. 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 - ≤ 5	0 - ≤ 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 5.2. 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

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

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.

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 6.1) similar to that proposed by Bricker et al. (1999) was employed to assess the susceptibility (see Section 9). This was then quantified using a similar procedure to that used for quantifying trophic status. This resulted in two sites being classified as having very low susceptibility and three sites being classified as having relatively high susceptibility (Table 6.2).

Table 6.1. Criteria for determining susceptibility to nutrient over-enrichment.

Parameter	Low	Medium	High
Tidal Prism/FW	≥200	≥100 - 200<	100<
Flushing Time	3<	≥3 - 10<	≥10
Stratification	<5	>5 - ≤10	≥ 10

Table 6.2. 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	1	0.4	1	3
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	5
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

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 and had similar indices of susceptibility to nutrient over-enrichment. A total of 9 sites fulfilled both criteria (Table 7.1).

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 8.1, 8.2 and 8.3 show the frequency distribution and cumulative frequency for dissolved inorganic nitrogen-N, dissolved inorganic phosphorus-P and bottom water dissolved oxygen, respectively.

Table 7.1. 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	HIGH	N
Chezzetcook Inlet	LOW	MED	Y
Country Hbr	LOW	MED	Y
Hfx-NW Arm	LOW	MED	Y
Indian Hbr	LOW	MED	Y
Jeddore Hbr	HIGH	MED	N
LaHave Inlet	LOW	MED	Y
Liscomb Hbr	LOW	MED	Y
Mahone Bay	LOW	MED	Y
Necum Teuch Inlet	LOW	MED	Y
Petpeswick Inlet	HIGH	MED	N
Popes Hbr	MED	LOW	N
Sheet Hbr	MED	HIGH	N
Shelburne Hbr	LOW	MED	Y
Ship Hbr	HIGH	HIGH	N
St. Margarets Bay	LOW	HIGH	N

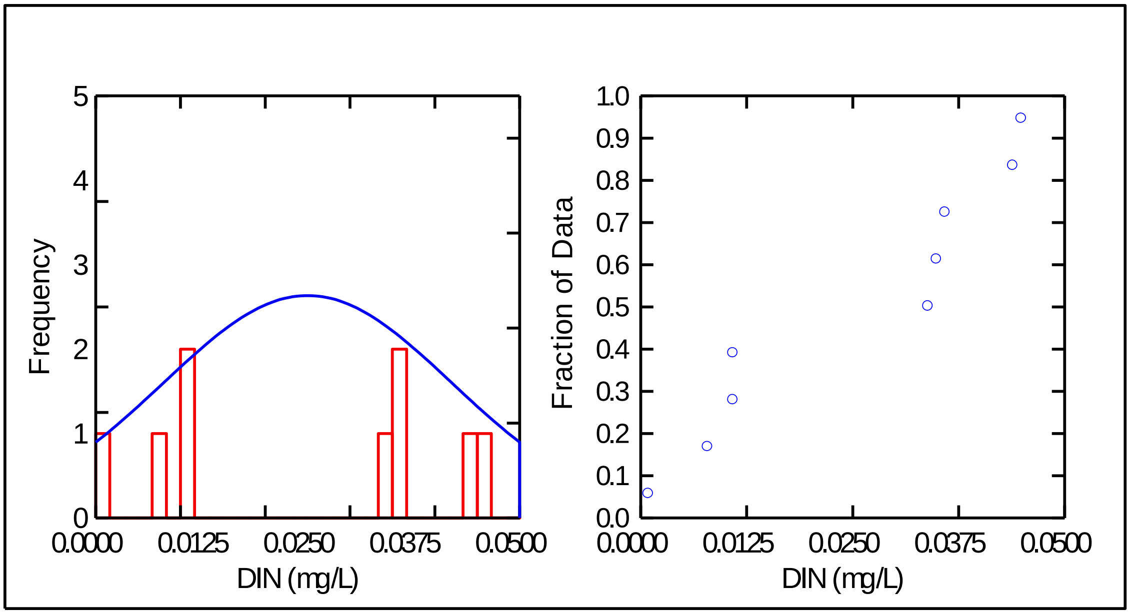


Figure 8.1. Frequency and cumulative frequency distribution for dissolved inorganic nitrogen-N.

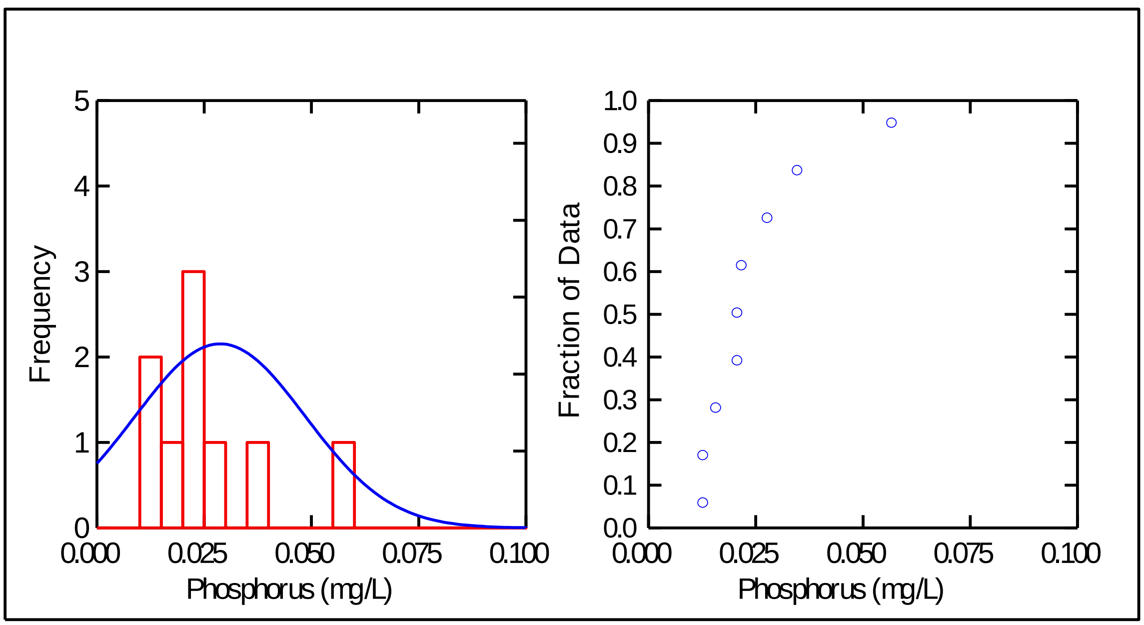


Figure 8.2. Frequency and cumulative frequency distribution for dissolved inorganic phosphorus-P.

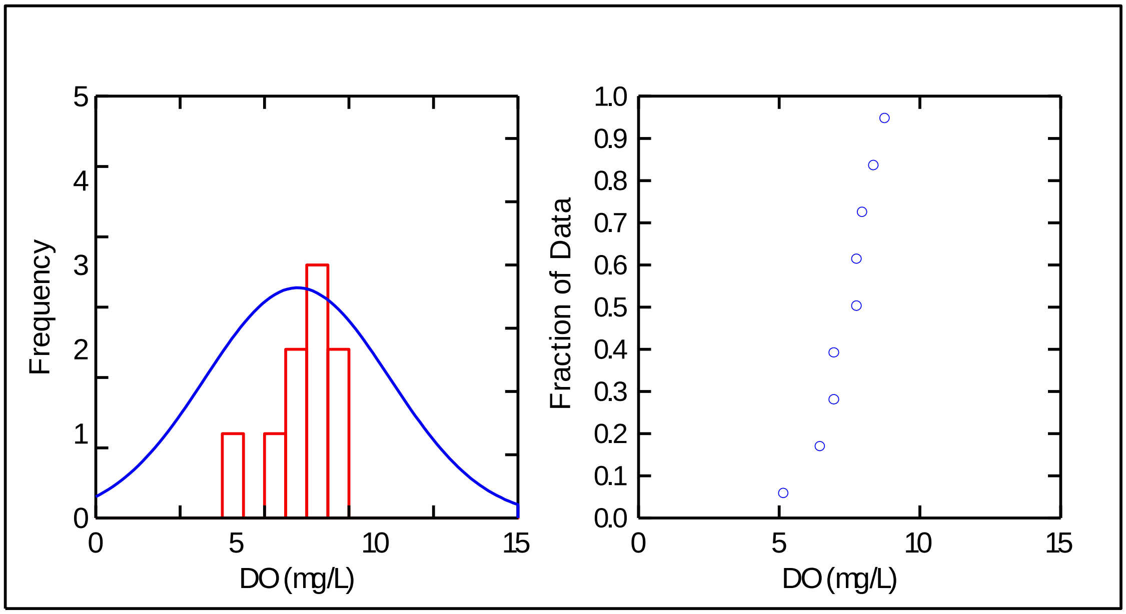


Figure 8.3. Frequency and cumulative frequency distribution for bottom water dissolved oxygen.

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 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. The final decision for which criteria to adopt, however, depends on the level of water quality considered to be acceptable, and the practicality and economics of reducing nutrient inputs in situations where nutrient over-enrichment is a problem.

Table 8.1 lists the values of the 25th, median and 75th percentile along with other basic statistics for each of the causal and response variables.

Table 8.1. Summary statistics for reference sites.

Statistic	DIN	P	DO
Minimum	0.001	0.013	5.2
Maximum	0.045	0.057	8.8
Mean	0.025	0.025	7.4
Standard Deviation	0.017	0.014	1.1
25th Percentile	0.010	0.015	6.8
Median	0.034	0.021	7.8
75th Percentile	0.040	0.032	8.2

9. References

Gregory, D., B.Petrie, F. Jordan and P. Langille. 1993. Oceanographic, geographic and hydrological parameters of Scotia-Fundy and southern Gulf of St. Lawrence inlets. Can. Tech. Rep. Hydrogr. Ocean Sci. No. 143.

Petrie, B., P. Yeats and P. Strain. 1999. Nitrate, phosphate and silicate atlas for the Scotian Shelf and the Gulf of Maine. Can. Tech. Rep. Hydrogr. Ocean Sci. No 203:

Strain, P.M. and P.A. Yeats. 1999. The relationship between chemical measures and potential predictors of the eutrophication status of inlets. Mar. Poll. Bull. 38(12): 1163-1170.

Case Study 2
Application of the Nutrient Guidance Framework to the Boughton River Estuary, Prince Edward Island.

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 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 (P. Lane and Associates Ltd. 1991). The objective of this case study is to establish nutrient criteria that would reduce the incidence of these events.

2. Site Description

P. 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 km^2 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 x to x m and the estuarine surface area is 7.6 and 9.9 sq km^2 at low and high tide, respectively. The low tide volume of the estuary is 29.1 x 10^6 m^3 and the residence time is 2.1 days (Lane and Associates Ltd. 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).

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 nutrient concentration data 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 reference conditions derived from historical data. As a result, the approach employed was to use an empirical analysis of in situ data on causal and response variables during periods when the symptoms of nutrient over-enrichment were minimal. The three response variables used were Secchi Disk depth, phytoplankton chlorophyll *a* concentration and per cent dissolved oxygen saturation. The potential causal variables examined were nitrogen and phosphorus concentrations. If significant relationships could be shown to exist between these variables, it would then be possible to determine the nutrient concentrations below which the response variables would be at acceptable levels.

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 P. Lane and Associates Ltd. (1991) during 1988-89. Physical, chemical, and biological data was 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 (Figure 4.1). Of the estuarine sites, three were located within the inner section of the estuary and the remaining two were located in 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 (Figure 4.2), five in the inner section and three in the outer section.

5. Frequency and Times of Hypoxic Events

Analysis of the data bases indicates that hypoxic[*] conditions occur mainly in bottom waters during the late summer-early fall period. Table 5.1 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.

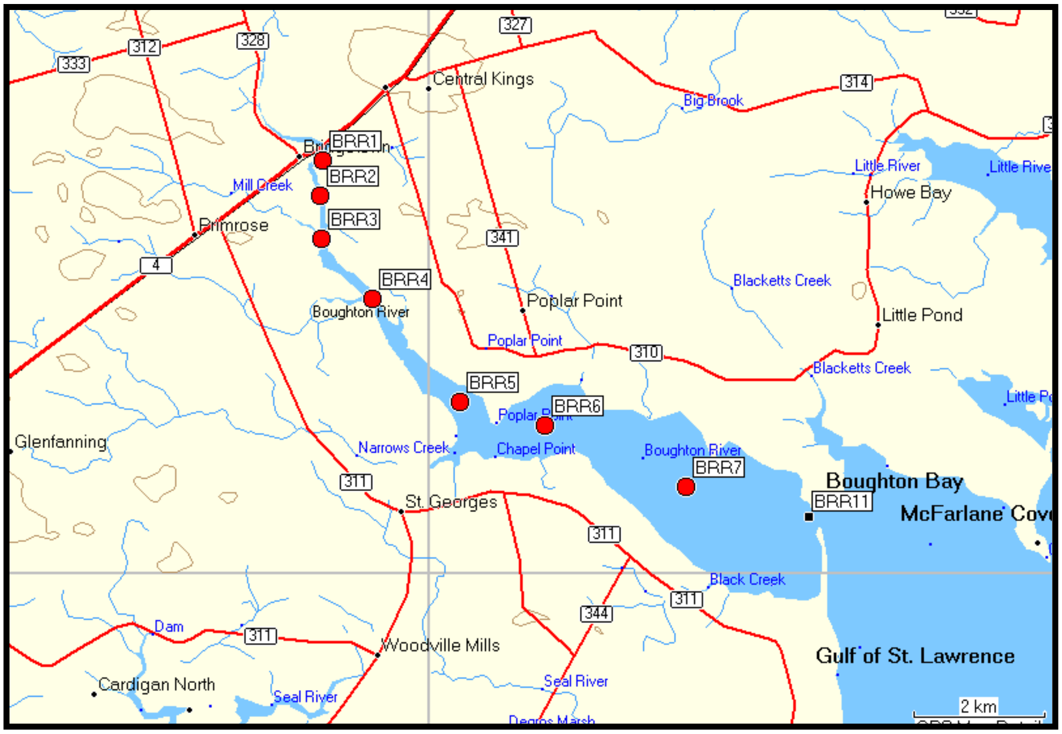


Figure 4.1. Location of sites sampled during 1988-89.

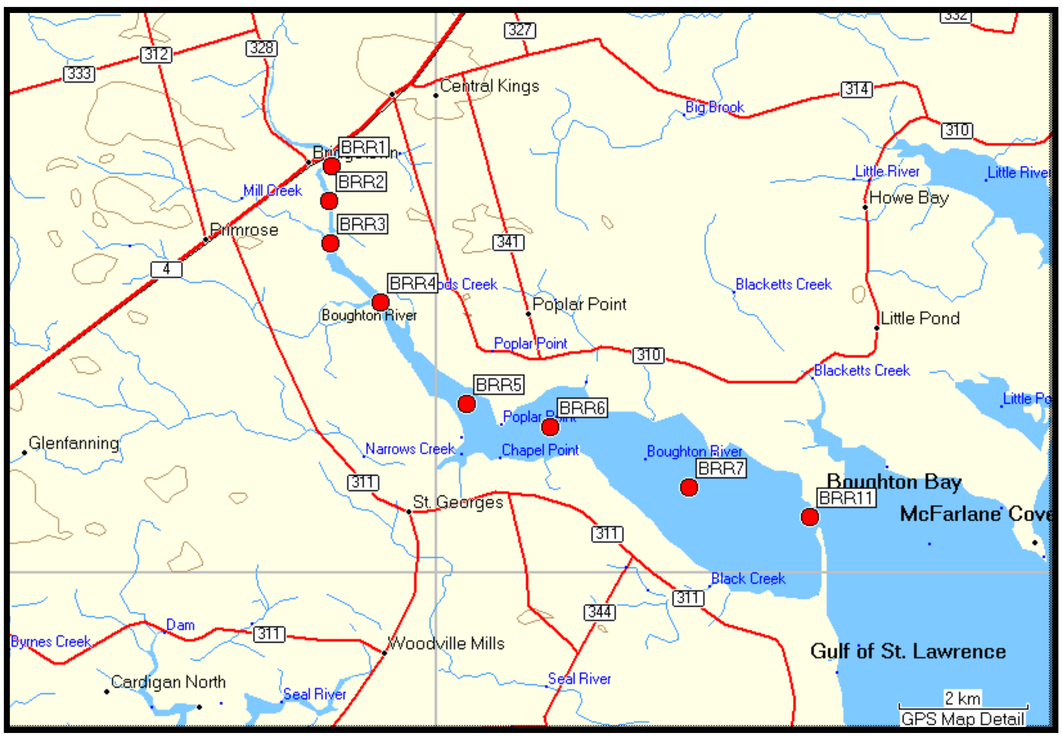


Figure 4.2. Location of sites sampled between 1998 and 2005.

Table 5.1. Number of times and dates when dissolved oxygen saturation was ≤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	

6. Relationships Between Response and Causal Variables

Figure 6.1 illustrates that, for data collected within the estuarine sampling sites between 1998 and 2005 (see Figure 2), 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 phosphorous, 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.

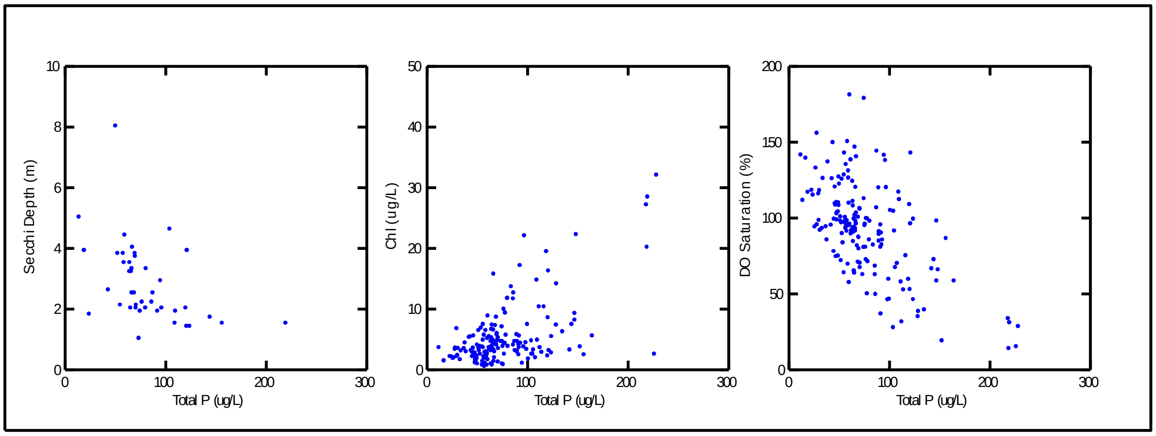


Figure 6.1. Relationship between response variables and total phosphorous concentration.

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 7.1 along with the corresponding values of the response variables. Figure 7.1 is a frequency plot of total phosphorus concentration for times when dissolved oxygen saturation levels are ≤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 µg/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

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

phosphorus levels are maintained below this level during the most critical periods, it should significantly reduce the occurrence of hypoxic events.

Table 7.1. 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 *a* (µg/L)	Total Phosphorus (µg/L)	DO Saturation (%)	Secchi Depth (m)	Chl *a* (µg/L)	Total Phosphorus (µg/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
25th Percentile	-	4	100	28	1.9	2	50	75
75th Percentile	-	20	190	42	3.3	5	80	110

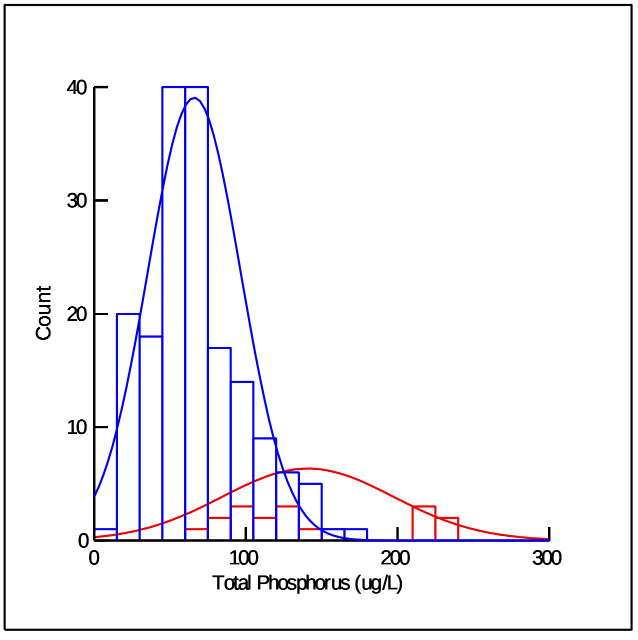


Figure 7.1. Frequency plot of total phosphorus concentration for times when dissolved oxygen saturation levels are ≤50 % (red) and >50% (blue).

References

P. Lane and Associates. 1991. Prince Edward Island Estuaries Study. Water quality in the Cardigan River, Boughton River and St. Peter's Bay. Report prepared for Environment Canada and Price Edward Island Department of the Environment.

APPENDIX I.

The OSPAR COMPRENSIVE 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

Category I Degree of Nutrient Enrichment

- **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 concentrations[1]**
  Elevated level(s) (defined as concentration > 50 % above[2] salinity related and/or region specific natural background concentration)
- **3 Increased winter N/P ratio (Redfield N/P = 16)**
  Elevated cf. Redfield (> 25)

Category II Direct Effects of Nutrient Enrichment (during growing season)

- **1 Maximum and mean Chlorophyll a concentration**
  Elevated level (defined as concentration > 50 % above[2] 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. *Ulva*)

Category III Indirect Effects of Nutrient Enrichment (during growing season)

- **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)

Category IV Other Possible Effects of Nutrient Enrichment (during growing season)

- **1 Algal toxins (DSP/PSP mussel infection events)**
  Incidence (related to II.2)

2 Other values less than 50 % can be used if justified

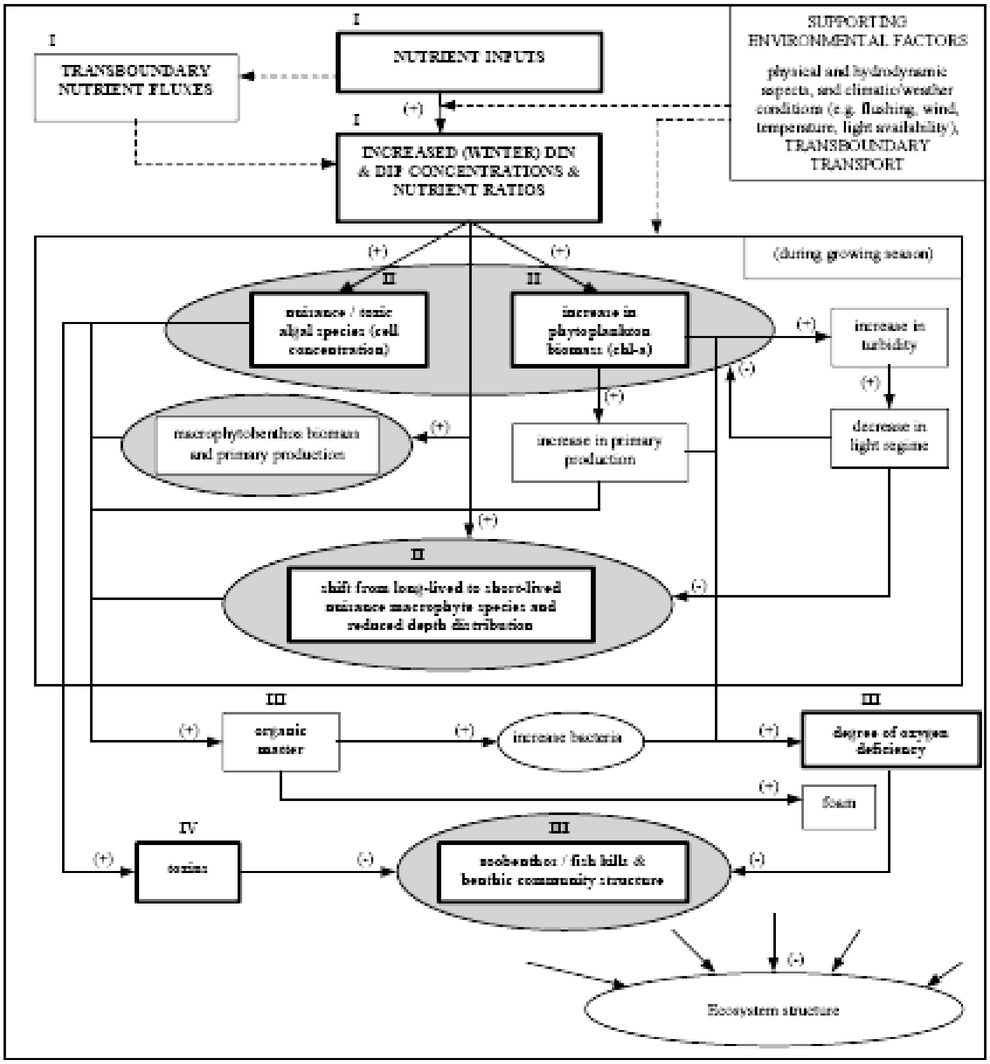


Figure 6.3. Main Interrelationships between the Assessment Parameters (in bold) of the OSPAR Comprehensive Procedure (COMPP).

Parameters for which Assessment Criteria and their assessment levels are identified are shown in boxes with bold lines. Biological elements are shaded. Continuous arrow lines with (+) and (-) indicate 'having stimulating effect upon', and 'having inhibiting effect upon', respectively. Dashed arrow lines indicate 'having influence upon'.

Key:	I	=	Category I	Degree of Nutrient Enrichment (Causative factors)
	II	=	Category II	Direct Effects of Nutrient Enrichment
	III	=	Category III	Indirect Effects of Nutrient Enrichment
	IV	=	Category IV	Other Possible Effects of Nutrient Enrichment

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

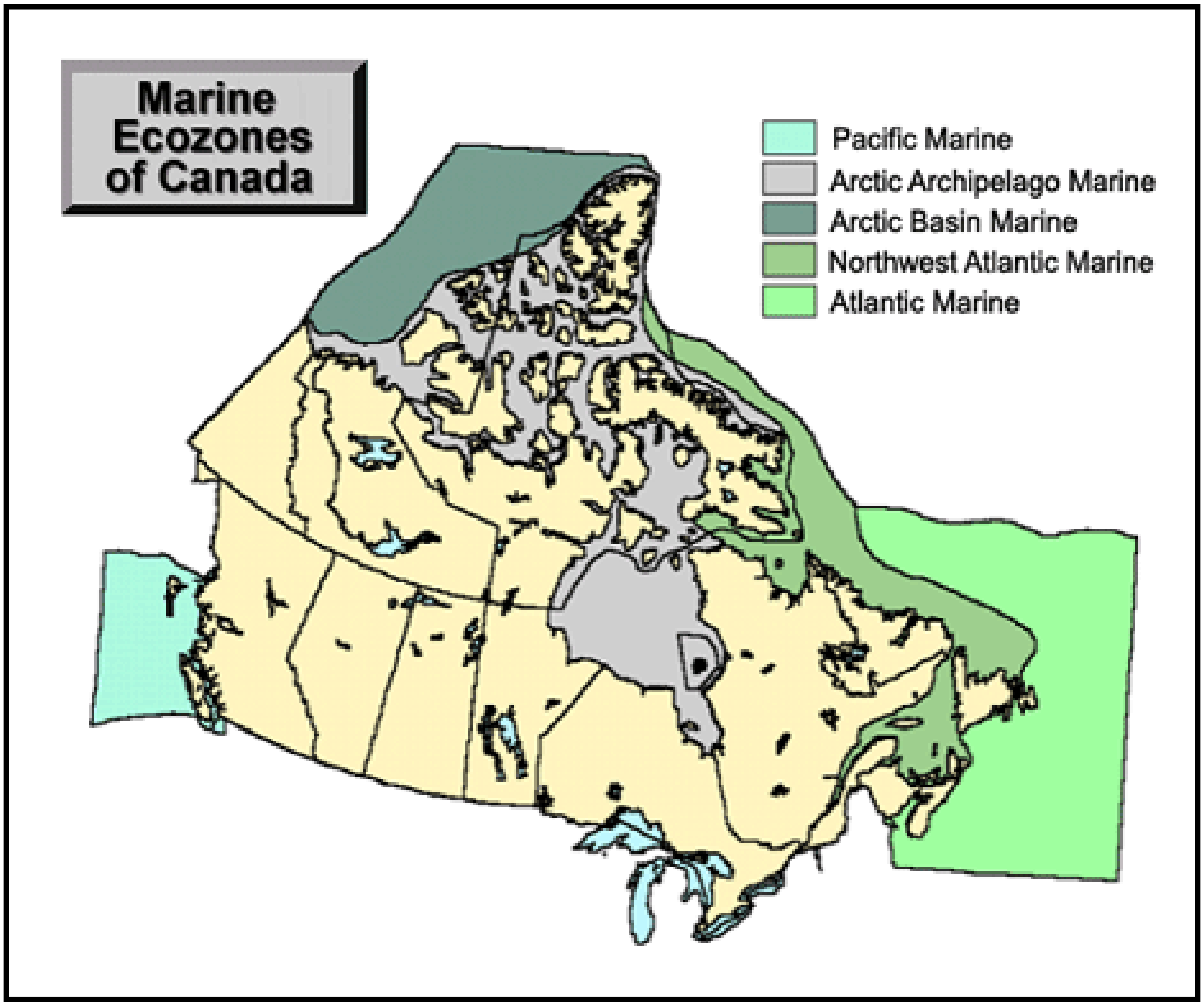
Region specific characteristics should be taken into account, such as physical and hydrodynamical aspects, and weather/climate conditions. These region specific characteristics may play a role in explaining the results of the classification.

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

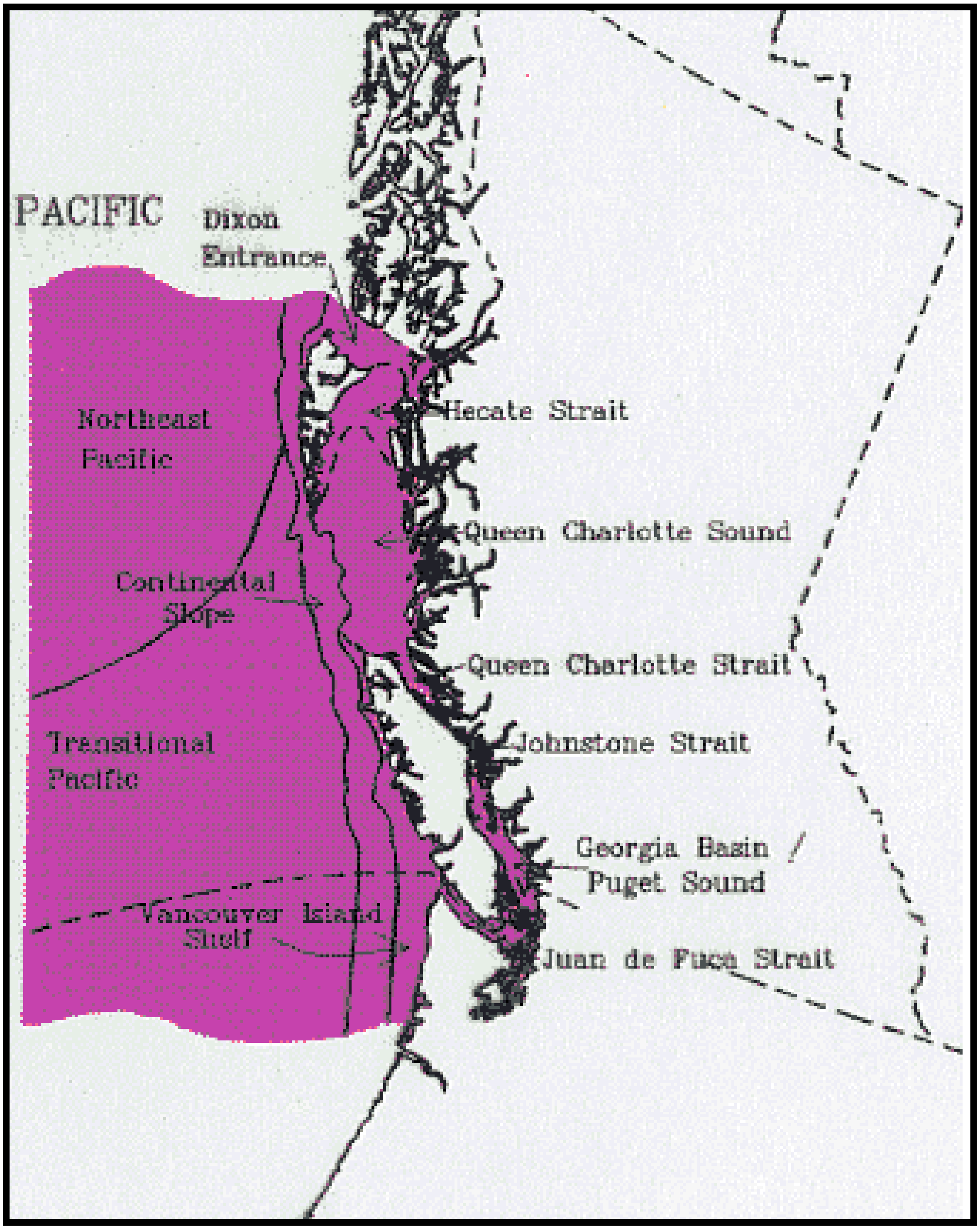
APPENDIX II

Marine Ecological Classification System for Canada

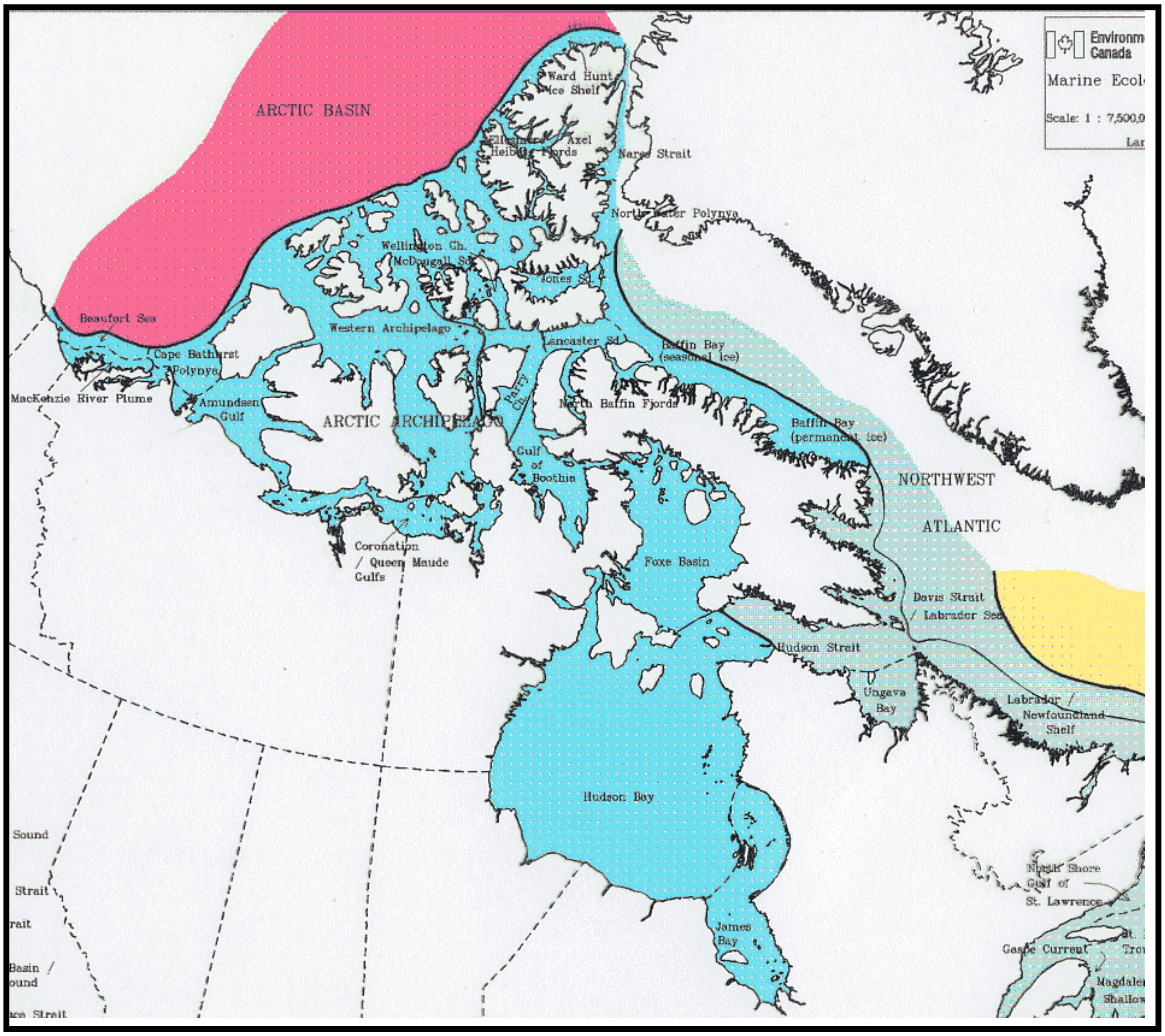
Marine Ecozones of Canada



The Pacific Marine Ecozones



The Artic Basin and Artic Circle Ecozones



The Atlantic and Northwest Atlantic Ecozones