A REVIEW OF SCIENTIFIC APPROACHES SUPPORTING NNE ASSESSMENT FRAMEWORK DEVELOPMENT FOR SAN FRANCISCO BAY

DRAFT

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1 INTRODUCTION

1.1 BACKGROUND AND PURPOSE

The California State Water Resources Control Board (State Water Board) is developing nutrient water quality objectives for the State's surface waters, using an approach known as Nutrient Numeric Endpoints (NNE). The NNE is comprised of two components. First, it would establish a suite of numeric regulatory endpoints based on the ecological response of an aquatic waterbody to nutrient over-enrichment (eutrophication, e.g., algal biomass, dissolved oxygen). Second, nutrient-response models would be used to link the ecological response endpoints to site-specific nutrient targets and other potential management controls. The NNE, intended to serve as numeric guidance to translate narrative water quality objectives, is currently under development for all California estuaries (Sutula 2013).

San Francisco Bay represents California's largest estuary (70% by area of estuarine habitat statewide). Because of its size and complexity, State Water Board staff determined that it merits development of site-specific nutrient objectives. The State Water Board and the San Francisco (SF) Water Board have agreed to collaborate on the development of site-specific nutrient objectives for SF Bay and that the SF Water Board will lead on this effort. In 2012, the SF Water Board and its stakeholders jointly developed a strategy to development regulatory endpoints and nutrient-response model for San Francisco Bay.

The process to select NNE regulatory endpoints begins with synthesis of science and ends with policy decisions. In this document, we refer to the product of scientific synthesis as an "**NNE assessment framework**," defined as a structured set of decision rules that specify how to use monitoring data to categorize specific segments of SF Bay with respect to adverse effects on Bay beneficial uses due to nutrient-overenrichment. While the decision on regulatory endpoints should be informed by science, it is ultimately a policy decision. The intention is that the SF Water Board would propose regulatory endpoints for SF Bay, based on the synthesis of science represented in the NNE assessment framework and feedback from the SF Bay stakeholders.

The purpose of this document is to review approaches to developing an NNE assessment framework, based on existing work in the United States and other countries. This document would summarize existing literature for how those indicators have been used to assess ecological condition and recommend a suite of options to consider for further exploration. The intent is that this white paper would be used to initiate discussions via a kick-off meeting with a working group of experts in estuarine eutrophication to: 1) discuss possible approaches and 2) identify the types of analyses of existing data that would support their evaluation. The white paper would also be discussed with SF Bay stakeholders for feedback and comments on approaches as well as identification of additional data sources that could support the evaluation.

Conceptually, the assessment framework builds on work by McKee et al (2011), which reviewed candidate indicators indicative of eutrophication or other adverse effects to Bay beneficial uses, assessed status and trends in these indicators, identified data gaps and recommended next steps. This review served as a starting point for the development of a nutrient management program for San Francisco Bay, spearheaded by the San Francisco RWQCB. Since the publication of the McKee et al. (2011) report, this program has produced an overarching strategy or work plan to guide technical, outreach and policy elements (SFRWQCB 2012) and several technical work products related to addressing data gaps or building on recommendations in the McKee et al. (2011) report (e.g. Senn et al. 2013).

The review recommended developing regulatory endpoints for subtidal habitat based on indicators such as phytoplankton, nutrient concentrations, and dissolved oxygen. Work to review the science supporting dissolved oxygen objectives will be completed separately from this effort; thus assessment framework development will focus on indicators and metrics of phytoplankton and nutrient concentrations. A particular approach to developing this framework is not presumed at the outset; rather the intent is to select the appropriate approach with advice of experts and stakeholders as a part of the process. The assessment framework will also build on recent work, led by SFEI, to develop conceptual models of SF Bay ecological response to nutrient loads and linkage to Bay beneficial use (Senn et al. 2013).

1.2 Development of a Nutrient Assessment Framework for SF Bay: Process and Desirable Attributes

Process

To understand the context for this white paper, it is helpful to understand the process envisioned to develop the SF Bay Nutrient Assessment Framework. We envision this process to consists of 5 steps:

1. Review existing approaches to nutrient assessment framework development
2. Analyze existing data to test applicable approaches
3. Draft assessment framework
4. Test with existing or newly collected monitoring data
5. Refine assessment framework

Philosophically, each step requires the review and input of the stakeholder advisory group.

Review Existing Approaches. The first step in developing an assessment framework is to prepare a white paper summarizing potential approaches that have been used elsewhere in the United State or in other countries. This white paper will identify candidate indicators and metrics, summarize existing literature for how those indicators have been used to assess ecological condition and recommend a suite of options to consider for further exploration. This white paper would also be discussed with SF Bay stakeholders for feedback and comments on

approaches as well as identification of additional data sources that could support the evaluation. It will be used to initiate discussions via a kick-off meeting with a working group of experts in to: 1) discuss possible approaches and 2) identify the types of analyses of existing data that would support their evaluation.

Analyses of Existing Data. The next step is to analyze existing data from SF Bay estuary that would support the evaluation of possible approaches to nutrient assessment framework development. Analyses will focus on identifying how data on indicators or combinations of indicators can be used to identify alternative states and how decisions on data aggregation across temporal and spatial scales affects the results of the assessment.

Draft Assessment Framework. Results of the analysis of existing data will be used by the expert working group to draft an nutrient assessment framework for SF Bay. Workgroup participants will to develop the scientific foundation for the assessment framework, specifying to the degree possible: 1) indicators and specific metrics, 2) a number of categories representing "alternative states" from high to low ecological condition and/or beneficial use support and 3) decision rules for how data should be used to categorize the Bay or Bay segment being to the applicable "alternative state."

Test Assessment Framework With Monitoring Data and Refine (As Needed) Assessment Framework . The draft assessment framework will be tested with monitoring data, either existing or newly collected. This effort will be used as an opportunity to make any refinements to the assessment framework. Results of the assessment will be compiled into a Bay "report card" and communicated to the public.

Desirable Attributes of An Assessment Framework

Desirable attributes of an nutrient assessment framework for SF Bay are as follows:

- The assessment framework should employ indicator(s) that have a strong linkage to Bay beneficial uses. This linkage should be scientifically well supported and easily communicable to the public.
- One or more primary indicators of the assessment framework should have a predictive relationship with surface water nutrients and/or nutrient loads to the Bay.
- The assessment framework should employ the indicator(s) classify the Bay segments from very high ecological condition to very low ecological condition. It should be explicit how the magnitude, extent, and duration of the effects that cause the segment to be classified differently.
- The assessment framework should be spatially explicit for different segments of the Bay and different habitat types (deep versus shallow subtidal) as warranted by the ecological nature of response to nutrients.

- The assessment framework should specify what are the appropriate methods used to measure the indicator and the temporal and spatial density of data required to make that assessment.
- It should provide guidance on how the data should be analyzed to categorize the Bay segments.

1.3 IMPORTANT DEFINITIONS

For those outside the regulatory world, distinction between terms like "criteria," "standards", "objectives," and "endpoints" can be confusing. The purpose of this section is to provide definitions of the terms that are linked closely to how the NNE framework will be implemented.

Eutrophication: Eutrophication is defined as the acceleration of the delivery, in situ production of organic matter, and accumulation of organic matter (Nixon 1995). One main cause of eutrophication in estuaries is nutrient over enrichment (nitrogen, phosphorus and silica). However, other factors influence primary producer growth and the build-up of nutrient concentrations, and hence modify (or buffer) the response of a system to increased nutrient loads (hereto referred to as **co-factors**). These co-factors include hydrologic residence times, mixing characteristics, water temperature, light climate, grazing pressure and, in some cases, coastal upwelling.

***Indicator*:** A characteristic of an ecosystem that is related to, or derived from, a measure of biotic or abiotic variable, that can provide quantitative information on ecological condition, structure and/or function. With respect to the water quality objectives, indicators are the ecological parameters for which narrative or numeric objectives are developed.

Water Quality Standards: Water quality standards are the foundation of the water quality-based control program mandated by the Clean Water Act. Water Quality Standards define the goals for a waterbody by designating its uses, setting criteria to protect those uses, and establishing provisions to protect water quality from pollutants. A water quality standard consists of three basic elements:

- Designated uses of the water body (e.g., recreation, water supply, aquatic life, agriculture; Table 1.1),
- Water quality criteria to protect designated uses (numeric pollutant concentrations and narrative requirements), and
- Antidegradation policy to maintain and protect existing uses and high quality waters.

Water Quality Criteria: Section 303 of the Clean Water Act gives the States and authorized Tribes power to adopt water quality criteria with sufficient coverage of parameters and of adequate stringency to protect designated uses. In adopting criteria, States and Tribes may:

- Adopt the criteria that US EPA publishes under §304(a) of the Clean Water Act;
- Modify the §304(a) criteria to reflect site-specific conditions; or

- Adopt criteria based on other scientifically-defensible methods.

The State of California's water criteria are implemented as "water quality objectives," as defined in the Water Code (of the Porter Cologne Act; for further explanation, see below).
States and Tribes typically adopt both **numeric** and **narrative** criteria. **Numeric** criteria are quantitative. **Narrative** criteria lack specific numeric targets but define a targeted condition that must be achieved.

Section 303(c)(2)(B) of the Clean Water Act requires States and authorized Tribes to adopt numeric criteria for priority toxic pollutants for which the Agency has published §304(a) criteria. In addition to narrative and numeric (chemical-specific) criteria, other types of water quality criteria include:

- Biological criteria: a description of the desired biological condition of the aquatic community, for example, based on the numbers and kinds of organisms expected to be present in a water body.
- Nutrient criteria: a means to protect against nutrient over-enrichment and cultural eutrophication.
- Sediment criteria: a description of conditions that will avoid adverse effects of contaminated and uncontaminated sediments.

Water Quality Objectives: The Water Code (Porter-Cologne Act) provides that each Regional Water Quality Control Board shall establish water quality objectives for the waters of the state i.e., (ground and surface waters) which, in the Regional Board's judgment, are necessary for the reasonable protection of beneficial uses and for the prevention of nuisance. The State of California typically adopts both **numeric** and **narrative** objectives. **Numeric** objectives are quantitative. **Narrative** objectives present general descriptions of water quality that must be attained through pollutant control measures. Narrative objectives are also often a basis for the development of numerical objectives.

Numeric Endpoint: Within the context of the NNE framework, numeric endpoints are thresholds that define the magnitude of an indicator that is considered protective of ecological health. These numeric endpoints serve as guidance to Regional Boards in translating narrative nutrient or biostimulatory substance water quality objectives. They are called "numeric endpoints" rather than "numeric objectives" to distinguish the difference with respect to SWRCB policy. Objectives are promulgated through a public process and incorporated into basin plans. Numeric endpoints are guidance that presumably can evolve over time without the need to go through a formal standards development process.

Table 1.1. Definition of estuarine beneficial uses applicable to selection of E-NNE indicators.

Marine Habitat (MAR) - Uses of water that support marine ecosystems including, but not limited to, preservation or enhancement of marine habitats, vegetation such as kelp, fish, shellfish, or wildlife (e.g., marine mammals, shorebirds).

Estuarine Habitat (EST) - Uses of water that support estuarine ecosystems including, but not limited to, preservation or enhancement of estuarine habitats, vegetation, fish, shellfish, or wildlife (e.g., estuarine mammals, waterfowl, shorebirds).

Cold Freshwater Habitat (COLD) - Uses of water that support cold water ecosystems including, but not limited to, preservation or enhancement of aquatic habitats, vegetation, fish or wildlife, including invertebrates.

Warm Freshwater Habitat (WARM) - Uses of water that support warm water ecosystems including, but not limited to, preservation or enhancement of aquatic habitats, vegetation, fish or wildlife, including invertebrates.

Wildlife Habitat (WILD) - Uses of water that support terrestrial ecosystems including, but not limited to, preservation and enhancement of terrestrial habitats, vegetation, wildlife (e.g., mammals, birds, reptiles, amphibians, invertebrates), or wildlife water and food sources.

Rare, Threatened, or Endangered Species (RARE) - Uses of water that support habitats necessary, at least in part, for the survival and successful maintenance of plant or animal species established under state or federal law as rare, threatened or endangered.

Spawning, Reproduction, and/or Early Development (SPWN) - Uses of water that support high quality aquatic habitats suitable for reproduction and early development of fish. This use is applicable only for the protection of anadromous fish.

Migration of Aquatic Organisms (MIGR) - Uses of water that support habitats necessary for migration, acclimatization between fresh and salt water, or other temporary activities by aquatic organisms, such as anadromous fish

Commercial and Sport Fishing (COMM) - Uses of water for commercial or recreational collection of fish, shellfish, or other organisms including, but not limited to, uses involving organisms intended for human consumption or bait purposes.

Shellfish Harvesting (SHELL) - Uses of water that support habitats suitable for the collection of filter-feeding shellfish (e.g., clams, oysters and mussels) for human consumption, commercial, or sport purposes.

Aquaculture (AQUA) - Uses of water for aquaculture or mariculture operations including, but not limited to, propagation, cultivation, maintenance, or harvesting of aquatic plants and animals for human consumption or bait purposes.

Contact Water Recreation (REC-1) - Uses of water for recreational activities involving body contact with water, where ingestion of water is reasonably possible. These uses include, but are not limited to, swimming, wading, water-skiing, skin and SCUBA diving, surfing, white water activities, fishing, or use of natural hot springs.

Non-contact Water Recreation (REC-2) – Uses of water for recreational activities involving proximity to water, but not normally involving body contact with water, where ingestion of water is reasonably possible. These uses include, but are not limited to, picnicking, sunbathing, hiking, beachcombing, camping, boating, tidepool and marine life study, hunting, sightseeing, or aesthetic enjoyment in conjunction with the above activities.

2 DEVELOPMENT OF NUTRIENT NUMERIC ENDPOINTS (NNE) FRAMEWORK AND NUTRIENT-RESPONSE MODELS IN SAN FRANCISCO BAY: BASIC CONCEPTS

2.1 BACKGROUND FOR DEVELOPMENT OF NNES IN ESTUARIES

U.S. EPA initiated the National Nutrient Management Strategy in 1998 to begin addressing the pervasive impacts of excessive nutrient loading to both fresh and marine waters (Wayland 1998). A primary objective of the strategy was to develop numeric nutrient criteria to measure the progress of the management strategy. EPA issued a series of technical guidance manuals for the development of nutrient criteria.

The "Nutrient Criteria Technical guidance Manual: Estuarine and Coastal Waters" was released by EPA in October 2001. EPA Region IX had already convened the Regional Technical Advisory Group (RTAG) and the State Technical Advisory Group (STRTAG) to serve as a forum for collaboration among stakeholders, agencies, and all nine Regional Water Boards. RTAG and STRTAG focused on the development of nutrient criteria for fresh waters. In 2006 the STRTAG proposed the California Nutrient Numeric Endpoint framework as California's approach to nutrient objectives. The development of nutrient numeric endpoints for fresh waters is preceeding prior to estuaries with the caveat that endpoints for upstream waterbodies would consider potential downstream impacts on estuaries.

Sutula et al. (2007) developed a conceptual framework for development of NNEs in estuaries based on the framework for streams (USEPA 2006). A work plan governing NNE development in estuaries was funded (McLauglin et al. 2009). Results of initial funding and an the work plan to continue NNE development has recently been updated (Sutula 2013). The work plan specifically identifies efforts by the San Francisco RWQCB and the Central Valley RWQCB to establish "site-specific" nutrient objectives for the San Francisco Bay (SFRWQCB 2012) and Delta.

2.2 APPROACHES TO SETTING NUTRIENT OBJECTIVES

Nutrient objectives are scientifically challenging. Nutrients are required to support life, but assessment of how much is "too much" is not straightforward. Typical paradigms used to set thresholds for toxic contaminants do not apply, in part because adverse effects of nutrient over enrichment are visible at orders of magnitude below recognized toxicity thresholds for ammonium and nitrate.

US EPA guidance on nutrient objective development generally recommends three means to set nutrient criteria (USEPA 2001): 1) reference approach, 2) empirical stress-response approach, and 3) cause-effect approach. The reference waterbody approach involves characterization of the distributions of nutrient in "minimally disturbed" waterbodies. Nutrient concentrations are chosen at some statistical percentile of those reference waterbodies. The empirical stress-response approach involves establishing statistical relationships between the causal or stressor

(in this case nutrient concentrations or loads) and the ecological response (changes in algal or aquatic plant biomass or community structure, changes in sediment or water chemistry (e.g., dissolved oxygen, pH). The cause-effect approach involves identifying the ecological responses of concern and mechanistically modeling the linkage back to nutrient loads and other co-factors controlling response (e.g., hydrology, grazers, denitrification, etc.).

SWRCB staff and USEPA Region 9 staff evaluated these three approaches for setting nutrient objectives in California waterbodies and determined that, while it may choose to ultimately incorporate some elements of all approaches into California's strategy for setting nutrient objectives, it would rely most heavily on the cause-effect approach. There were several reasons for this. First, the cause-effect approach has a more direct linkage with beneficial uses and is generally thought to lend itself to a more precise diagnosis of adverse effects. Second, the alternative approaches require a tremendous amount of data not currently available in such a large state. Third, the reference approach is particularly problematic because it automatically relegates a certain percentage of the reference sites to an "impaired" status. In addition, for many waterbody types, minimally disturbed reference sites are largely unavailable. Fourth, statistical stress-response relationships can be spurious, or have lots of unexplained variability (i.e., poor precision). This poor precision is translated to a larger margin of safety required (more conservative limits) for load allocations and permit limits. While waterbody typology, to some degree, can assist in explaining some of this variability, it cannot completely remove the concern. Thus, while simpler than the cause-effect approach, the empirical stress-response will result in more false negative and false positive determinations of adverse effects, and in the end will be more costly to the public.

For estuaries, reliance on the cause-effect approach is strongly suggested, because in the majority of circumstances, the reference or empirical stress-response approaches are simply untenable. Estuaries within California are highly variable in how they respond to nutrient loading due to differences in physiographic setting, salinity regime, frequency and timing of freshwater flows, magnitude of tidal forcing, sediment load, stratification, residence time, denitrification, etc. This combination of "co-factors" results in differences in the dominant primary producer communities (i.e., phytoplankton, macroalgae, benthic algae, submerged aquatic vegetation, emergent macrophytes). It also creates variability in the pathways that control how nutrients cycle within the estuary. At times, these co-factors can play a larger role in mitigating estuarine response to nutrient loads or concentrations, blurring or completely obscuring a simple prediction of primary productivity limited by nutrients (e.g., Figure 2.1). For example, in estuaries such as San Francisco Bay, synthesis of existing data by Cloern and Dugdale (2010) have clearly shown that surface water nutrient concentrations do not correlate with measures of primary productivity, in part because of important co-factors that override simple nutrient limitation of primary production.

2.3 KEY TENETS OF THE NNE APPROACH

The NNE framework for California waterbodies is basely largely on the cause-effect approach. The intent of the NNE framework is to control excess nutrient loads to levels such that the risk

or probability of impairing the designated uses is limited to a low level. If the nutrients present – regardless of actual magnitude – have a low probability of impairing uses, then water quality standards can be considered met.

The framework has three organizing principals (USEPA 2006):

1. *Ecological response indicators provide a more direct risk-based linkage to beneficial uses than nutrient concentrations or loads alone. Thus the NNE framework is based on the diagnosis of eutrophication or other adverse effects and its consequences rather than nutrient over enrichment per se.*

Except in some cases, such as unionized ammonium causing toxicity, nutrients themselves do not impair beneficial uses. Rather, ecological response to nutrient loading causes adverse effects that impair uses. Instead of setting objectives solely in terms of nutrient concentrations, it is preferable to use an analysis that takes into account the risk of impairment of these uses. The NNE framework needs to target information on ecological response indicators such as dissolved oxygen, surface water phytoplankton and harmful algal bloom (HAB) biomass (e.g., chlorophyll-a, water clarity), macroalgal biomass and percent cover, benthic algal biomass (sediment chlorophyll-a) and submerged aquatic vegetation (SAV) density and percent cover, and aesthetics (e.g., foul odors, unsightliness). These ecological response indicators provide a more direct risk-based linkage to beneficial uses than the ambient nutrient concentrations or nutrient loads. Given this approach, it is critical that tools be developed that link the response indicators back to nutrient loads and other co-factors and management controls (hydrology, etc.).

2. *A weight of evidence approach with multiple indicators will produce a more robust assessment of eutrophication.*

When possible, the use of multiple indicators in a "weight of evidence" approach provides a more robust means to assess ecological condition and determine impairment. This approach is similar to the multimetric index approach, which defines an array of metrics or measures that individually provide limited information on biological status, but when integrated, functions as an overall indicator of biological condition (Karr and Chu 1999).

3. *Use of "nutrient-response" models to convert response indicators to site-specific nutrient loads or concentrations.*

A key premise of the NNE framework is the use of models to convert numeric endpoints, based on ecological response indicators, to site- specific nutrient load goals appropriate for assessment, permitting, and TMDLs. A key feature of these models is that they account for site-specific co-factors, such as light availability, temperature, and hydrology that modify the ecological response of a system to nutrients.

2.4 REVIEW OF SCIENCE SUPPORTING NUTRIENT OBJECTIVE DEVELOPMENT IN SAN FRANCISCO BAY

McKee et al. (2011) reviewed literature and data relevant to the assessment of eutrophication and other adverse effects of nutrient overenrichment in San Francisco Bay, with the goal of providing information to formulate a work plan to develop NNEs for this estuary. The review had three objectives: 1) Evaluate indicators to assess eutrophication and other adverse effects of anthropogenic nutrient loading in San Francisco Bay, 2) Summarize existing literature in SF Bay using indicators and identify data gaps, and 3) Investigate what data and tools exist to evaluate the trends in nutrient loading to the Bay (McKee et al. 2011).

Recommended NNE Indicators for SF Bay

As noted previously, an NNE assessment framework is the structured set of decision rules that helps to classify the waterbody in categories from minimally to very disturbed, in order to determine if a waterbody is meeting beneficial uses. Development of an assessment framework begins by choosing response indicators, which were reviewed using four criteria: 1) strong linkage to beneficial uses, 2) well -vetted means of measurement, 3) can model the relationship between the indicator, nutrient loads and other management controls, and 4) has an acceptable signal: noise ratio to assess eutrophication.

For San Francisco Bay, indicators varied among four habitat types: 1) unvegetated subtidal, 2) seagrass and brackish SAV, 3) intertidal flats, and 4) tidally muted habitats (e.g. estuarine diked Baylands). Two types of indicators were designated. Primary indicators are those which met all evaluation criteria and would therefore be expected to be a primary line of evidence of the NNE assessment framework for SF Bay. Supporting indicators fell short of meeting evaluation criteria, but may be used as supporting lines of evidence. This terminology is used in order to provide a sense of level of confidence in how the indicators should be employed in a multiple lines of evidence context.

The review found four types of indicators met all evaluation criteria and are designated as primary: dissolved oxygen, phytoplankton biomass, productivity, and assemblage, and cyanobacterial abundance and toxin concentration (all subtidal habitats), macroalgal biomass and cover (intertidal habitat, tidally muted habitats, and seagrass habitats; Table 2.1). Other indicators evaluated met three or fewer of the review criteria and designated as supporting indicators: HAB cell counts and toxin concentration, urea and ammonium (all subtidal), light attenuation and epiphyte load (seagrass/brackish SAV). Ultimately, the real distinction between “primary” and “supporting” and how these classes of indicators would be used as multiple lines of evidence in an NNE assessment is entirely dependent on indicator group and particular applications to specific habitat types. Some primary indicators (e.g. dissolved oxygen) could be stand-alone, while for others such as phytoplankton biomass, productivity and assemblage, the SF Bay Technical Advisory Team recommended using them as multiple lines of evidence, as use of any one alone is likely to be insufficiently robust.

Table. 2.1 Data gaps and next steps for development of an SF Bay NNE assessment framework.

Type	Indicator	Designation	Data Gaps	Recommended Next Steps
Subtidal Habitat	Dissolved oxygen	Primary	Wealth of data exists. Technical Advisory Team does not have expertise to review adequacy of DO objectives. Review did not address dissolved oxygen data in the tidally muted habitats of SF Bay.	Consider update of science supporting Basin Plan dissolved oxygen objectives, if warranted by additional review by fisheries experts. Review could be for entire Bay or limited to the tidally muted areas of the Bay.
	Phytoplankton biomass , productivity, and assemblage	Primary	Need a review of science supporting selection of endpoints. Improved prediction of factors controlling assemblage	Recommend development of a white paper and a series of expert workshops to develop NNE assessment framework for phytoplankton biomass, productivity, taxonomic composition/assemblages, abundance and/or harmful algal bloom toxin concentrations. Recommend augmentation of current monitoring to include measurement of HAB toxin concentrations in water and faunal tissues.
	HAB species abundance and toxin conc.	Cyanobacteria = primary; Other HAB =supporting	Little data on HAB toxin concentrations in surface waters and faunal tissues.	
	Ammonium and urea	Supporting	Lack of understanding of importance of ammonia limitation of nitrate uptake in diatoms on Bay productivity vis-à-vis other factors. Lack of data on urea in SF Bay	Recommend formulation of a working group of SF Bay scientists to synthesize available data on factors known to control primary productivity in different regions in the Bay, and evaluate potential ammonium endpoints. Recommend collecting additional data on urea concentrations in SF Bay via USGS's water quality sampling over a two year period.
	Macrobenthos taxonomy, abundance and biomass	Co-factor	Lack of information on how to use combination of taxonomy, abundance, and biomass to assess eutrophication	Recommend utilization of IE-EMP dataset to explore use of macrobenthos to be used reliably to diagnose eutrophication distinctly from other stressors in oligohaline habitats. This may involve including biomass in the protocol to improve ability to diagnose eutrophication.
Seagrass Habitat	Phytoplankton biomass, epiphyte load and light attenuation	Phytoplankton biomass = primary, epiphyte load and light attenuation = secondary	Poor data availability of data on stressors to SF Bay seagrass beds. Studies needed to establish light requirements for seagrass and to assess effects of light attenuation	Recommend 1) Continued monitoring of aerial extent of seagrass every 3-5 years (currently no further system scale monitoring is planned beyond 2010), 2) studies to establish light requirements for SF Bay seagrass species, 3) development of a statewide workgroup to develop an assessment framework for seagrass based on phytoplankton biomass, macroalgae, and epiphyte load and 4) collection of

Type	Indicator	Designation	Data Gaps	Recommended Next Steps
	Macroalgae biomass and cover	Primary	Data gaps include studies to establish thresholds of macroalgal biomass, cover and duration that adversely affect seagrass habitat	baseline data to characterize prevalence of macroalgal blooms on seagrass beds. Studies characterizing thresholds of adverse effects of macroalgae on seagrass currently underway in other California estuaries should be evaluated for their applicability to SF Bay.
Intertidal Flat Habitat	Macroalgal biomass and cover	Primary	Lack of baseline data on frequency, magnitude (biomass and cover) and duration of macroalgal blooms in these intertidal flats	Recommend collection of baseline data on macroalgae, microphytobenthos and sediment bulk characteristics. Recommend inclusion of SF Bay scientists and stakeholders on statewide workgroup to develop an assessment framework for macroalgae on intertidal flats.
	Sediment nutrients	Supporting		
	MPB taxonomy and biomass	Supporting		
Muted Subtidal Habotat	Macroalgae	Primary	Lack of baseline data on biomass and cover in muted habitat types	Recommend collection of baseline data on macroalgae, dissolved oxygen, phytoplankton biomass, taxonomic composition and HAB species/toxin concentration in these habitat types. Recommendation to develop an assessment framework based on macroalgae, phytoplankton and dissolved oxygen in these habitat types. One component of this discussion should be a decision on beneficial uses that would be targeted for protection and to what extent the level of protection or expectation for this habitat type differ from adjacent subtidal habitat.
	Phytoplankton biomass, assemblage, HAB toxin conc.	Phytoplankton biomass, cyanobacteria = primary; assemblage and other HABs= supporting	Lack of baseline data on biomass and community composition, HAB toxin concentrations	
	Dissolved oxygen	Primary	Some data on dissolved oxygen exist. Unclear what levels of DO required to protect muted habitat beneficial uses	

The use of ammonium as an indicator received review, due to its hypothesized role in limiting phytoplankton primary production via nitrate uptake inhibition in Suisun Bay and the lower Sacramento River. The SF Bay Technical Advisory Team chose to include it as a supporting indicator because the importance of ammonium inhibition of diatom blooms relative to other factors controlling primary productivity Bay wide is not well understood. Additional review and synthesis were recommended, pending currently funded studies, to identify potential ammonium thresholds.

Table 2.1 summarizes data gaps and recommended next steps by McKee et al. (2011) for development of an SF Bay NNE assessment framework by habitat type. Data gaps and recommendations generally fall into four categories: 1) Monitoring to assess baseline levels of indicators of interest where data are currently lacking, 2) Analysis of existing data, 3) Field studies or experiments to collect data required for endpoint development, and 4) Formation of expert workgroups to recommend approach to assessment framework development and synthesize information to be used in setting numeric endpoints.

2.5 Indicators Under Further Consideration for the SF Bay NNE Assessment Framework

The SF Bay Water Board, with advice from stakeholders, chose to prioritize the development of NNE assessment framework for subtidal habitats in SF Bay. Seagrass, intertidal habitat, and diked Baylands are not included in this initial work. For subtidal habitat, McKee et al. (2011) review recommended developing regulatory endpoints for subtidal habitat based on indicators of phytoplankton, nutrient concentrations, and dissolved oxygen. Work to review the science supporting dissolved oxygen objectives will be completed separately from this effort; thus assessment framework development will focus on indicators and metrics of phytoplankton and nutrient concentrations.

Phytoplankton

Phytoplankton are unicellular organisms, which serve a critical ecosystem function of primary production, forming the base of pelagic foodwebs in many aquatic environments. Phytoplankton blooms are a natural phenomenon, typical of spring and summer periods of naturally high primary production which supplies energy to the ecosystem. However, phytoplankton respond rapidly to changes in nutrient concentrations and nutrient enrichment, which can lead to more frequent blooms, of greater intensity, and spatial and temporal extent [*Carstensen et al.*, 2011; *Cloern*, 2001]. Increased biomass is typically the first response to nutrient enrichment, often followed by species shifts, and accumulation of organic matter which results in oxygen depletion in the bottom water of stratified areas [*Cloern*, 1996; 2001; *W M Kemp et al.*, 2005]. Excessive blooms can also increase turbidity such that light penetration through the water column is significantly reduced, thus restricting growth of seagrasses [*Huntington and Boyer*, 2008]. Over production of harmful, toxin producing species can also result in ecosystem effects through poisonings of marine mammals and birds.

Because of their direct link and rapid response to nutrient additions, phytoplankton are considered a primary symptom of eutrophication and have been used extensively as a gauge of ecological condition and change [*Bricker et al.*, 2003; *Domingues et al.*, 2008]. Phytoplankton is used as an indicator or water quality element in various forms in a number of assessment frameworks and is typically considered one of the more robust in terms of establishment of thresholds [*Borja et al.*, 2011].

There are a number of considerations for using phytoplankton as an indicator of eutrophication [*Domingues et al.*, 2008]. Firstly, the establishment of reference condition for water quality may be difficult in systems for which there is no historical data. Secondly, there is a lack of guidance on sampling frequency, and for several water quality frameworks, the proposed frequency is insufficient to assess phytoplankton succession and may even preclude the detection of algal blooms. Finally, the use of chlorophyll-a as a proxy for biomass may overlook blooms of pico- and small nanoplankton, and overestimate the importance of large microphytoplankton because cellular chlorophyll-a content is often species-specific [*Domingues et al.*, 2008].

Phytoplankton Biomass (Chlorophyll-a Concentration, Bloom Intensity and Frequency

Chlorophyll-a is measured as a way to estimate the active phytoplankton biomass and is used extensively as an indicator of eutrophic condition for estuarine waters. Chlorophyll is the green pigment in all plants and Chlorophyll-a is the most common type of chlorophyll. Plants use chlorophyll to capture sunlight for photosynthesis. Chlorophyll-a concentrations are often highest just below the surface, not at the surface of the water.

Chlorophyll-a can be measured in several ways: discrete measures, continuous measurements via data sonde, and remote sensing. Discrete samples of chlorophyll-a are measured by filtering a known amount of sample water through a glass fiber filter. The filter paper itself is used for the analysis. The filter is ground up in an acetone solution and either a fluorometer or spectrophotometer is used to read the light transmission at a given wavelength, which in turn is used to calculate the concentration of chlorophyll-a. Continuous measurements in the field are made with a fluorometer probe mounted to a data sonde or similar logging device. The in situ water is exposed to light of a single wavelength. Some substances in the water sample, including chlorophyll-a, will give off light, or fluoresce, in response to the light. The amount of light emitted by the chlorophyll-a is measured and used to calculate the chlorophyll-a concentration. Field fluorometers must be calibrated routinely against discrete samples for accuracy. Chlorophyll-a is also measured remotely by satellite. Satellites measure the color of seawater to determine the amount of chlorophyll present. The ocean color is often blue, but the satellite can detect very small changes in the ocean color as a result of the chlorophyll in phytoplankton. Satellite measurements need to be compared to discrete measurements to calibrate the satellite measurements.

Phytoplankton blooms are expected to increase in frequency, duration and spatial extent as water bodies continue to experience nutrient over enrichment [*Bricker et al.*, 2003]. Bloom

duration can be directly quantified using continuous monitoring data. Frequency and spatial extent are typically assessed heuristically in the field and binned into groups (periodic versus episodic for frequency, and high, moderate, low and very low for spatial coverage) [*Bricker et al.*, 2003].

Phytoplankton Productivity

Primary production is the process by which autotrophic organisms "fix" inorganic carbon using solar energy to carry out metabolic processes and build cellular material. Production in marine waters is influenced by the supply of nutrients, light, temperature, flow regime, turbidity, zooplankton grazing and toxic substances. Low rates of annual primary production may indicate low susceptibility to enrichment while high rates of annual primary production represent higher susceptibility, possibly resulting in symptoms associated with undesirable disturbance [*Cloern*, 2001; *Devlin et al.*, 2007a; *S J Painting et al.*, 2007].

This productivity is typically measured using ^{14}C radiolabeling to measure the rate of carbon uptake over a defined area or volume. The method is based on the assumption that biological uptake of ^{14}C-labelled dissolved inorganic carbon (DIC) is proportional to the biological uptake of the more commonly found ^{12}C DIC. In order to determine uptake, one must know the concentration of DIC naturally occurring in the sample water, the amount of ^{14}C-DIC added, and the amount of ^{14}C retained in particulate matter (^{14}C-POC) at the end of the incubation experiment [*Steeman-Nielsen*, 1952].

Phytoplankton Taxonomic Composition or Assemblage

Changes in phytoplankton community composition are expected to occur as eutrophication develops in estuarine environments. Shifts may reflect a loss of biodiversity of organisms and a shift towards dominance of one or more species, but they often include increased abundance of opportunistic nuisance and toxic species that result from changing nutrient concentrations and ratios [*Borja et al.*, 2011]. Samples for phytoplankton taxonomy can be collected from whole water or can be collected using one or more phytoplankton nets of targeted mesh size. There are several methods for estimating phytoplankton community composition: identification and cell counts using microscopy, flow cytometry/particle counting, and pigment analysis by HPLC. Each has its own advantages and disadvantages , but all provide some measure of phytoplankton community structure [*R A Anderson*, 2005; *P E Kemp et al.*, 1993].

Harmful Algal Bloom Dominance and Toxin Concentrations

Some algal blooms may include a shift towards dominance of nuisance or toxic species which may have a detrimental impact to biological resources [*Bricker et al.*, 2003]. For example, excessive abundance of small phytoplankton species may clog the siphons of filer feeding bivalves and may cause respiratory irritation to fish. Excessive abundance of toxin producing organisms can result in poisonings of marine mammals and birds. Presence of nuisance and toxic species can be identified by the methods described above in phytoplankton community composition. Algal toxins can be measured on whole water samples using spectrophotometric and HPLC techniques.

Nutrient Concentrations and/or Ratios

Eutrophication is primarily caused by nutrient enrichment leading to increased production of organic matter [*Nixon*, 1995]. Primary producers need nutrients for growth and low concentrations of bioavailable nitrogen and phosphorus will limit primary production. Estuarine nutrient concentrations are highly dynamic and are rapidly transformed by biogeochemical processing. The concentrations of dissolved inorganic nutrients in the water column represents the instantaneous net "remainder" after processing by all other factors. Ambient nutrient concentrations are often correlated with nutrient loading into the systems [*Boynton and Kemp*, 2008; *Conley et al.*, 2000; *Hejzlar et al.*, 2009; *Smith et al.*, 2005]. Though empirical relationships between nutrient concentrations and biological response are dependent on a variety of site specific conditions and are highly variable among systems [*Carstensen et al.*, 2011; *Cloern*, 2001].

Both nitrogen and phosphorus can be limiting either exclusively or in combination (co-limitation). Ambient nutrient concentrations of dissolved inorganic nitrogen (DIN) or dissolved inorganic phosphorus (DIP) are used to determine nutrient limitation, usually with the suggestion that primary production is N-limited for DIN:DIP ratios below 10 and mainly P-limited for DIN:DIP ratios greater than 20 [*L A Anderson and Sarmiento*, 1994; *Klausmeler et al.*, 2004; *Redfield et al.*, 1963]. During blooms, ambient nutrient concentrations may become almost completely consumed, resulting in strong seasonal variability in nutrient concentrations. Changes in estuarine geomorphology also result in wide spatial variability in N- and P-limitation, due to variation in supply, removal, and biogeochemical transformations of nutrients [*Carstensen et al.*, 2011].

Relatively recent shifts in our conceptual understanding of eutrophication [*Cloern*, 2001; *Devlin et al.*, 2007a; *S J Painting et al.*, 2007] indicate that estuaries can have complex responses to nutrient inputs, including both direct and indirect responses, and the role additional factors that moderate ecosystem response. In estuarine systems, factors such as light climate and hydrology, affect the susceptibility of different waterbodies to nutrient enrichment [*S J Painting et al.*, 2007]. Consequently, the presence of high nutrient concentrations should be regarded as a potential cause for concern and may trigger further assessment of biological response indicators. Given the current understanding of the consequences of nutrient enrichment it is clear that, for any given aquatic situation, it is not possible to determine specific nutrient thresholds without reference to the biological response [*Devlin et al.*, 2007a].

3 REVIEW OF EXISTING ASSESSMENT METHODS/F RAMEWORKS

3.1 REGULATORY CRITERIA

A number of states and programs within the U.S. are in the process of developing nutrient criteria or biocriteria to protect waterbodies from nutrient overenrichment. Typically, these criteria are based on three types: 1) TN and TP, 2) water column chlorophyll a and 3) dissolved oxygen. Many programs have established narrative criteria for biological response indicators and are in the process of collecting monitoring data that would support the development of numeric values that are protective for specific estuaries (e.g. Maryland, Maine, and Chesapeake Bay for chlorophyll a). Florida has recently established site-specific TN and TP and chlorophyll a criteria for all the State's estuaries. Table 3.1 summarizes existing TN, TP and chlorophyll a criteria for estuaries and tidal rivers.

Of these states, the criteria promulgated for Florida estuaries and Chesapeake Bay represent the most scientifically well-documented approaches to establishing nutrient and chlorophyll a endpoints (USEPA 2007, USEPA 2010). In both cases, estuarine surface TN and TP criteria were established via modeling linkages with biological endpoints (maintenance of seagrass, maintenance of balanced algal population, dissolved oxygen). Although relevant for nutrient-response modeling of SF Bay, we choose not to include a synthesis of this work in our review. Establishment of chlorophyll a criteria based on maintenance of seagrass, which currently represent less than 3% of subtidal habitat in the Bay, is also not a relevant paradigm for SF Bay. Therefore we summarize the scientific paradigms and approaches used in Florida and for the Chesapeake Bay that relevant for the "maintenance of balanced algal populations."

Table 3.1 Summary of existing chl- a criteria by state for lakes and estuaries. Adapted from U.S. EPA. 2003. Survey of States, Tribes and Territories Nutrients Standards. Washington, DC

State	Chlorophyll a Numeric Criteria in Estuaries (all values in µg L-1 unless otherwise noted)
District of Columbia	Seasonal July 1–September 30 segment average chlorophyll *a* concentration of 25 applied to tidally influenced waters only.
Florida	In unvegetated subtidal habitats, chlorophyll a should not exceed 20 for greater than 10% of the time.
Hawaii	Chlorophyll *a* criteria applying to different locations within Lake Mead ranging from 5–45
North Carolina	Freshwater class C waters and tidal saltwaters: For lakes and reservoirs and other waters subject to growths of macroscopic and microscopic vegetation not designated as trout waters: <40. For lakes and reservoirs and other waters subject to growths of macroscopic and microscopic vegetation designated as trout waters: <15.
Oregon	Chlorophyll *a* criteria for: • Natural lakes which do not thermally stratify: <10 • Natural lakes which do not thermally stratify, reservoirs, rivers and estuaries: <15 (OAR340-041-0019)
Virginia	Site specific seasonal numerical chlorophyll *a* criteria applicable March 1–May 31 and July 1–September 30 for the tidal James River segments JMSTF2, JMSTF1, JMSOH, JMSMH, JMSPH (9 VAC 25-260-310), ranging from 10-23.

Florida

In Florida, the rationale or establishment of chlorophyll a criteria to protect a "balanced algal population" is based on the premise that nutrient-driven effects on algal growth and biomass accumulation can result in more frequent, short term blooms that decrease water clarity, adversely affect aesthetics, recreation, and aquatic life habitat. They specifically cite: 1) the increased harmful algal blooms, which can produce toxins that adversely affect both human health and aquatic life and 2) the effect of frequent algal blooms on the long-term balance of organic matter cycling within an estuary (Nixon 1995), leading to hypoxia or anoxia, which also can adversely affect habitat and aquatic life. Because toxic blooms are a frequent occurrence in Florida estuaries and coastal waters, EPA deemed appropriate the derivation of chlorophyll criteria on the basis of reducing the likelihood of nuisance algal blooms on recreation and recreational uses (Larkin and Adams 2007; Walker 1985).

Specific chl-a concentrations consistent with nuisance conditions were defined in that literature on the basis of trophic state boundaries, user perception studies, and observed impacts. While they acknowledge documentation supporting trophic state chl a thresholds is limited, they cite: 1) Assessment of Estuarine Trophic Status (ASSETS, Bricker et al. 2003), in which low algal bloom conditions were defined as maximum chl-a concentrations < 5 μg/L, medium bloom conditions as maximum chl-a concentrations 5–20 μg/L, high bloom conditions as maximum chl-a concentrations 20–60 μg/L, and hypereutrophic conditions as maximum chl-a concentrations above 60 μg/L and 2) the United Kingdom Comprehensive Studies Task Team maximum summer chl-a value of 10 μg/L as an estuarine eutrophic threshold (Painting et al. 2007. EPA maintained that frequently occurring, elevated chlorophyll a concentrations can be an expression of dominance by one or more phytoplankton species, potentially toxic or otherwise harmful or nuisance algae, citing cyanobacterial blooms in freshwater and brackish habitats (Chorus et al. 2000) and marine HABs (Anderson et al. 2008; Paerl et al. 2008; Glibert et al. 2010). They also utilized information on bloom frequencies typical of Florida estuaries and then identified concentrations typical of blooms of harmful or nuisance algae and indicative of imbalance of phytoplankton populations. One estimate for the range of observed monthly chl-a maxima was from 15 to 25 μg/L, depending on the type of estuary (coastal embayment, river-dominated, or lagoon) (Glibert et al. 2010). In a national survey, the average bloom chl-a concentrations were 20 μg/L or less for 7 of 10 large estuaries; concentrations were especially low for Florida Bay (8 μg/L) and Pensacola Bay (10 μg/L, Glibert et al. 2010) and higher for the St. Johns River Estuary (20 μg/L, Bricker et al. 2007). Based on this work, EPA selected a chl-a concentration target of 20 μg/L, with an allowable exceedance frequency of no more than 10 percent of monitoring data.

Chesapeake Bay

In the Chesapeake Bay, multiple lines of evidence were used to derived chlorophyll a criteria (EPA 2007), based on adverse effects associated with high chl-a in Chesapeake Bay include seasonal hypoxia or anoxia (Smith et al. 1992, Hagy et al. 2004, Bricker et al. 2008), decreased water clarity affecting submerged aquatic vegetation (SAV) (Dennison et al. 1993, Kemp et al. 2004), and blooms of potentially harmful algal taxa (HABs) (Cloern 2001, Marshall et al. 2005, 2009, Mulholland et al. 2009, Morse et al. 2011).These lines of evidence included (1) analysis of

historical and recent data to establish baseline chl-a for the mainstem Bay; (2) detection of long-term trends of chl-a; (3) quantification of climatic forcing of chl-a; (4) identification of a relationship between DO and chl-a; (5) quantification of the effects of chl-a on water clarity and habitat suitability for SAV; (6) establishment of linkages between chl-a and cyanobacteria toxin concentrations.

Thresholds for the historical reference periods (1960-1980) ranged from 15 to 35 µg L^{-1} in spring, and from 7 to 54 µg L^{-1}, with the 1970s having higher thresholds than the 1960s (EPA 2007,). The oligohaline region had the highest surface chl-a thresholds, declining to the lowest thresholds for the polyhaline portion of Chesapeake Bay. The lowest thresholds were ~ 4-7 µg L^{-1}in the polyhaline region for the 1960s ranging up to the highest thresholds were ~ 40-55 µg L^{-1} in the oligohaline region for the 1970s historical reference period. The mesohaline and polyhaline regions had higher thresholds for surface chl-a in high-flow conditions than in mid- or low-flow conditions while the oligohaline region had higher thresholds for surface chl-a in low-flow than in high-flow conditions. The lowest thresholds were ~ 4-7 µg L^{-1}in the polyhaline region for the 1960s ranging up to the highest thresholds were ~ 40-55 µg L^{-1} in the oligohaline region for the 1970s historical reference period.

Low summer bottom-water DO occurred at high chl-a, with no observations of DO > 3 mg L-1 (the deep-water 30-d mean DO criterion) when May-Aug chl-a was > 16 µg L^{-1}, or of DO > 1.7 mg L-1 (the minimum DO criterion for fish; USEPA 2003) when May-Aug chl-a was > 22 µg L^{-1}.

Diatoms usually dominate the floral composition of Chesapeake Bay, with seasonally variable contributions by other algal taxa including dinoflagellates, cryptophytes, and cyanobacteria whose abundance varied seasonally. Exceptional occurrences of dinoflagellates blooms were not sufficient to support chl-a criteria on regional and seasonal bases. However, in tidal fresh and oligohaline regions, toxic blooms of the cyanobacteria, *Microcystis aeruginosa*, can reach high chl-a in summer. Simple linear regression showed significant relationships ($p < 0.05$) between surface chl-a and cell counts of *M. aeruginosa* for the upper Bay and four of seven tidal tributaries. Chl-a thresholds separating high-risk from middle- and low-risk for surface and above-pycnocline chl-a and were 29.2 and 29.0 µg L^{-1}, respectively. A threshold of 27.5 µg L^{-1} was established as protective against toxic Microcystis in the Bay (U.S. EPA 2007).

Based on these analyses, a set of reference criteria were developed for Chesapeake Bay (summarized in Table 3.2). These reference concentrations should only be applied to mainstem Chesapeake Bay surface, open-water habitats only during the spring (March 1 through May 31) and summer (July 1 through September 30) seasons, the most critical seasons for addressing algal-related impairments.

Although community composition was not directly incorporated into the EPA 2007 analysis, Buchanan et al. (2005) quantified the habitat conditions supporting phytoplankton reference communities in Chesapeake Bay. They reported maximum spring and summer chlorophyll a concentrations (in µg·liter-1), respectively, for tidal fresh (13.5, 15.9), oligohaline (24.6, 24.4), mesohaline (23.8, 13.5), and polyhaline (6.4, 9.2).

Table 3.2 Chesapeake Bay chlorophyll a reference concentrations (from EPA 2007).

Salinity Regime[2]/ Water Column Location	Season[3]	Water Clarity Criteria Application Depth[4] (m)	Chlorophyll *a* Reference Concentration (µg·liter^{-1})
Historical Chlorophyll *a* Reference Concentrations[5]			
Oligohaline	Spring	–[6]	18
Mesohaline	Spring	–	8
Polyhaline	Spring	–	4
Oligohaline	Summer	–	46
Mesohaline	Summer	–	23
Polyhaline	Summer	–	5
Dissolved Oxygen Impairment-Based Chlorophyll *a* Reference Concentrations			
Deeper Waters Which Stratify	Annual	–	10–15
Shallow Waters	Annual	–	30
Water Clarity Impairment-Based Chlorophyll *a* Reference Concentrations			
Tidal Fresh/Oligohaline	SAV	0.5	43
Tidal Fresh/Oligohaline	SAV	1.0	11
Mesohaline/Polyhaline	SAV	0.5	39
Mesohaline/Polyhaline	SAV	1	16
Mesohaline/Polyhaline	SAV	2	3

[1]All chlorophyll *a* reference concentrations apply as µg·liter^{-1} across the surface waters of open-water designated-use segments for the applicable salinity regime and season.

[2]Tidal Fresh = 0 – <0.5 ppt salinity; oligohaline = 0.5– <5 ppt salinity; mesohaline = 5–18 ppt salinity; polyhaline = >18 ppt salinity.

[3]Spring = March 1–May 31; Summer = June 1–September 30; SAV or SAV growing season: for tidal-fresh, oligohaline, and mesohaline habitats = April 1–October 31; for polyhaline habitats = March 1–November 30 (U.S. EPA 2003a).

[4]Water clarity criteria application depth for each Chesapeake Bay Program segment as published in U.S. EPA 2003b and as adopted into Delaware, Maryland, Virginia and the District of Columbia's water quality standards regulations.

[5]Reference concentrations only apply to mainstem Chesapeake Bay segments.

[6]Not applicable.

3.2 NON-REGULATORY ASSESSMENT FRAMEWORKS

Over the past decade, much work has been done to establish standardized methodologies to assess ecological quality in estuaries, with several methods developed specifically for eutrophication [*Andersen et al.*, 2011; *Bricker et al.*, 2003; *Devlin et al.*, 2011; *Domingues et al.*, 2008; *Zaldivar et al.*, 2008] and conduct surveys to evaluate the magnitude and extent of eutrophication [*Andersen et al.*, 2011; *Borja et al.*, 2009; *Bricker et al.*, 1999; *Devlin et al.*, 2011; *Garmendia et al.*, 2012].

In Europe, there has been a vast expansion in methods, due to the adoption of the European Union Water Framework Directive (WFD). The aim of the WFD is to achieve good ecological status in all EU member state waterbodies, where good status represents a no more than 50% deviation from reference conditions. Assessments are carried out at a waterbody level, and reference conditions are defined for each waterbody type based on characteristics including tidal range, mixing, exposure and salinity [*Devlin et al.*, 2011]. Each EU member state is required to adopt the WFD process though the selection of waterbody types, reference conditions, specific indicator variables and assessment methods can vary among member states [*VIncent et al.*, 2002]. Birk et al. (2012) document over 300 methods developed for compliance with the WFD alone, as many countries preferred developing country-specific methods instead of a handful of methods applicable Europe-wide (e.g. Birk and Schmedtje, 2005; Borja et al., 2009).

Assessment Framework Utilizing Multiple Categories of Indicators

Several indicator-based assessment frameworks have been developed to assess eutrophic condition of estuaries with respect to eutrophication utilizing multiple indicators. The most representative assessment frameworks have been found to incorporate annual data with sampling throughout the year, to capture frequency of occurrence and spatial extent in indicator metrics, and use of a combination of indicators into an overall condition rating [*Devlin et al.*, 2011].

Tables 3.3-3.4. provides a brief summary of integrated assessment frameworks that utilize multiple groups of indicators (Ferreira et al. 2011). Studies comparing eutrophication status results generated for the same system using different assessment frameworks have indicated that results can vary slightly depending on which framework is applied (Table 3.5) [*Devlin et al.*, 2011; *Garmendia et al.*, 2012]. Different frameworks apply similar indicators, but differences in timeframes of data analysis (seasonal versus annual), characteristics included in the indicator metrics (concentration, spatial coverage, frequency of occurrence), and how to combine indicators into multiple lines of evidence, had an effect on the overall outcome of the assessment [*Devlin et al.*, 2011].

Table 3.3 Methods of eutrophication assessment and examples of biological and physico-chemical indicators used and integration capabilities (pressure-state and overall; modified from Borja et al. 2012). From Ferreira et al. 2012.

Method Name	Biological indicators	Physico-chemical indicators	Nutrient load related to impairments	Integrated final rating
TRIX[b]	Chl	DO, DIN, TP	no	yes
EPA NCA Water Quality Index[a]	Chl	Water clarity, DO, DIN, DIP	no	yes
ASSETS[e]	Chl, macroalgae, seagrass, HAB	DO	yes	yes
TWQI/LWQI[c]	Chl, macroalgae, seagrass	DO, DIN, DIP	no	yes
OSPAR COMPP[g]	Chl, macroalgae, seagrass, phytoplankton indicator species	DO, TP, TN, DIN, DIP	yes	yes
WFD[f]	phytoplankton, Chl, macroalgae, benthic invertebrates, seagrass,	DO, TP, TN, DIN, DIP, water clarity	no	yes
HEAT[d]	Chl, primary production, seagrass, benthic invertebrates, HAB, macroalgae	DIN, DIP, TN, TP, DO, water clarity	no	yes
IFREMER[h]	Chl, seagrass, macrobenthos, HAB	DO water clarity, SRP, TP, TN, DIN, sediment organic matter, sediment TN, TP	no	yes
STI[i]	Chl, Primary Production	DIN, DIP	no	no

[a] USEPA, 2005, 2008.
[b] Vollenweider *et al.*, 1998.
[c] Giordani *et al.*, 2009.
[d] HELCOM, 2009.
[e] Bricker *et al.*, 1999, 2003, 2007.
[f] Devlin, pers.Com.
[g] OSPAR, 2002, 2008.
[h] Souchu *et al.*, 2000.
[i] Ignatiades, 2005.

Table 3.4. Summary of approaches used for assessment of eutrophication applicable to shallow and deepwater unvegetated subtidal habitat. Adapted from Devlin et al. 2011.

		UK WFD	OSPAR	TRIX	ASSETS	EPA NCA	TWQI/LWQF	HEAT
Grouping of Variables	Causative Factors	Nutrient Load	DIN and DIP concentration, ratios, and loads	DIN and TP concentration	DIN and DIP loads	DIN, DIP conc	TN, TP, DIN and DIP conc.	DIN and DIP
	1^{ary} effects	Chl-a, PP indicator species, seasonal changes in cell abundance of diatoms/dinoflagellates, SAV, macroalgae	Chl-a, PP indicator species, macroalgae, microphytobenthos, SAV	Chl-A	Chl-a macroalgae	water clarity, chl-a	Chl a, SAV, macroalgae	Chl a, water clarity, SAV,
	2^{ary} effects	DO	DO, zoobenthos and/or fish kills, organic carbon	DO	Nuisance/toxic blooms	DO	DO	Benthic invertebrates
	Other effects		Algal toxins					
Temporal sampling framework		Annual chla and DO, winter DIN, monthly PP groups	Growing season chl-a (Mar-Sept), Winter DIN, summer DO	Annual	Annual	One sample per year (per station) within summer index period	Results can be derived based on one time period, multiple periods recommended	Growing season chl-a (Mar-Sept), Winter DIN, summer DO
Spatial sampling framework		Sampling in estuaries and nearshore defined by salinity, reported by waterbody	Sampling defined by salinity in estuaries, nearshore	Sampling mostly in larger offshore systems; results reported by region	Sampling in salinity zones, synthesized to waterbody, region, then national, with reporting at all levels	Sampling is regional, synthesized to national level, reported at regional and national level	For shallow, benthic PP dominated. Can be applied to single stations or groups of stations.	Sampling defined by salinity in Baltic Sea
Assessment of indicators		Deviation from reference conditions	Deviation from reference conditions	Placement on scale from 1-10 TRIX units	Deviation from reference conditions	Deviation from reference conditions	Deviation from reference condition	Deviation from reference condition
Comb-ination Method		Indicator scores are averaged within in indicator group. Final score gives classification status	One out, all out for individual categories and overall classification	Linear combo of logarithm of variables modified by scaling coeff.	Scores of ave. primary and secondary indicators combined in a matrix	Indicators assessed individually. WQI based on % of samples in 4 categories.	TWQI scores combined as the sum of weighted quality values for individual indicators.	One out, all out for individual categories and overall classification

Table 3.5 Summary of procedures used for evaluating the eutrophic status of estuarine and coastal waters and categories used for final classification. From Devlin et al. 2011.

Method	Summary of assessment procedure	Categories for final classification of status, and colour coding (Fig. 4)
WFD[a] used in the UK	Considers physico-chemical and biological QEs. For each element, one or more indicators are used. EQRs are calculated for each indicator using the ratio between measurements and reference condition. An EQR is calculated for each quality element by averaging the indicator scores for all components within each QE. The final BQE score, a number between 0 (worst) and 1 (best) relates to the five equidistant boundary classes. The worst classification status for the BQE's and the physico-chemical status are taken as the overall water body status. In this one-out-all-out approach, if any element has a state less than or equal to Moderate, then the water body is considered to be impacted	Classification status is calculated from the worst BQE BQE's are integrated by normalising outputs to 0–1 score (EQR score): High (best, blue) Good (green) Moderate (yellow) Poor (orange) Bad (worst, red)
OSPAR COMPP[b]	Thresholds are set for each indicator for estuarine, coastal and offshore waters. The full OSPAR COMPP procedure is applied only to areas indicated as Potential Problem Area or Problem Area by the screening procedure. Indicators are used in Categories I to IV to determine levels of nutrient enrichment and impacts, and combined within each category in a one-out-all-out process. A final assessment as Problem Area (+), Potential Problem Area or Non Problem Area (−) is made after evaluating scores for Categories I–IV with the one-out-all-out approach	Problem area (+) Potential problem area Non problem area (−)
TRIX[c]	The annual average of all parameters are combined without assigning individual parameter ratings, to give a ranking between 0 and 10, where: $TRIX = [\log10(\text{Chl } a) + \log10(\text{aD\%O}) + \log10(\text{DIN}) + \log10(\text{TP}) - k]/m$ The coefficients $k = -1.5$ and $m = 1.2$ are fixed to establish the lower and upper limits of the index	Index is scaled from 0 to 10, ranging from oligotrophy (scarcely productive—open sea) to eutrophy (highly productive)
ASSETS[d]	Three components are included: Influencing Factors (IF), Eutrophic condition (EC) and Future Outlook (FO) and the three components are combined into one ASSETS rating. Only the EC component is included in this study Five indicators are evaluated: primary symptoms (Chl-*a*, macroalgae) and secondary symptoms (DO, SAV, nuisance/toxic blooms). Area-weighted values are calculated from salinity zone assessments and area-weighted system wide ratings are made for each variable. Primary (average) and secondary (worst) symptom scores are combined in a matrix to determine an overall score or rating for eutrophic condition of the waterbody	The final scores for eutrophication status are: High (worst, red) Moderate high (orange) Moderate (yellow) Moderate low (green) Low (best, blue) Colour codes are consistent with WFD scales

Table. 3.5 continued

Method	Summary of assessment procedure	Categories for final classification of status, and colour coding (Fig. 4)
EPA NCA[e]	The Water Quality Index (WQI) uses a combination of DIN, DIP, DO, Chl-*a* and Water Clarity for assessment. Indicator scores are determined using the percentage of samples that are Good, Fair or Poor/No Data, e.g. DIN: Good: <10% of samples are Poor and >50% are Good Fair: 10–25% of samples are Poor and/or >50% are Poor or Fair Poor: >25% of samples are Poor All indicators are combined in a similar fashion to determine the rating for a site: Good = A maximum of one indicator is Fair and no indicators are Poor Fair = One of the indicators is rated Poor or two or more indicators are Fair Poor = Two or more of the five indicators are rated Poor To determine the WQI by region and nation, results from each area are used to determine a final assessment score where: Good: <10% of areas are in Poor condition and >50% are Poor or Fair Fair: 10–20% of areas are in Poor condition or >50% are Fair or Poor Poor: >20% of areas are in Poor condition	A final WQI assessment is determined by site and then by region and nation: Good (green) Fair (yellow) Poor (red)

Pressures (nutrient loading) and Future Outlook are not included here, as they are not used by all approaches

EQR ecological quality ratio, *DIN* dissolved inorganic nitrogen, *DIP* dissolved inorganic phosphorus, *TP* total phosphorus, *Chl-a* chlorophyll *a*, *DO* dissolved oxygen, *SAV* submerged aquatic vegetation, *spp.* species, *BQE* biological quality element, *PP* phytoplankton, *QE* quality element

[a] CEC (1991a, b)

[b] OSPAR (2002, 2005, 2008)

[c] Vollenweider et al. (1998)

[d] Bricker et al. (1999, 2003, 2007), Ferreira et al. (2007), www.eutro.org and www.eutro.org/register

[e] USEPA (2001a, b, 2005, 2008)

UK WFD Framework for Eutrophication

Here we review the United Kingdom (UK) assessment protocol for eutrophication. The WFD classifies waterbodies into one of five ecological condition categories: High, Good, Moderate, Poor or Bad. Initial risk of eutrophication is assessed based on nutrient load, turbidity, flushing time, and tidal range. The ecological condition category is assessed using three biological quality elements: phytoplankton, macroalgae, and angiosperms. The final assessment also includes a measure of physico-chemical status including dissolved inorganic nitrogen and dissolved oxygen.

Each biological quality element consists of one or more indicators that measure different aspects of the biological community (phytoplankton includes CHL-a and cell counts of abundance and composition, macroalgae includes biomass and areal coverage, angiosperms include biomass and area coverage) [*Devlin et al.*, 2011]. For each indicator, final measurements are converted into a normalized ecological quality ratio by first converting the data into a numerical scale between zero and one (where status class boundaries are not necessarily equidistant) and then averaging the scores for all indicators and related to one of the five assessment classes. Classification of overall ecological condition status is determined using a one-out-all-out approach: where the overall status reflects the worst category from results for any biological quality element or physico-chemical element [*Devlin et al.*, 2011]. In this review we focus specifically on the phytoplankton biological quality element and the nutrient physico-chemical element. Here we review the nutrient physico-chemical element and the phytoplankton biological quality element. The sampling period for all elements is a minimum of six years, with sampling frequency no less than 12 times per year, collected monthly [*Devlin et al.*, 2007b].

UK WFD Nutrients Water Quality Element. Nutrient thresholds for the UK WFD assessment framework are generated using a tool based on a cause and effect model that relates elevated nutrients indices of ecosystem response [*Devlin et al.*, 2007a]. The tool specifically looks at three indices: (1) Evidence of nutrient enrichment based on the calculation of an annual winter nitrogen concentration; (2) Modeling of potential primary production based on a waterbody characteristics and light availability; (3) Evidence of undesirable disturbance as measured by dissolved oxygen levels. A stepwise analysis scheme is employed to determine overall eutrophic condition. Initial classification of the water bodies is based on comparison of mean winter dissolved inorganic nitrogen concentration against predetermined nutrient thresholds. Winter is defined as the period when algal activity is lowest and when dissolved nutrients should show conservative behavior [*Devlin et al.*, 2007a]. Nutrient thresholds are also normalized to a salinity gradient, allowing for dilution of nutrients with increasing salinity. If estuaries exceed the initial thresholds for "Good" water quality, potential primary productivity is estimated from a simple screening model that uses equilibrium nutrient concentrations and light limited growth rates to calculate production [*Devlin et al.*, 2007a; *S Painting et al.*, 2006]. If the potential primary production is greater than 300 g C m^{-2} y^{-1}, a level defined by Nixon [1995] as representing eutrophic status, and winter dissolved inorganic nitrogen concentration is

greater than 30 µM, than the estuary is considered to have moderate or worse eutrophic condition. The final metric, used to determine the severity of adverse impacts, is dissolved oxygen concentration. Dissolved oxygen concentration is reported as either a growing season mean (March to September). Thresholds for dissolved oxygen that mark the boundaries between Moderate and Poor and Poor and Bad are derived from criteria set for fish in transitional waters which supports conditions for juvenile fish in the freshwater reaches of estuaries [*Best et al.*, 2007]. Dissolved oxygen concentrations less than 5 mg L^{-1} negatively affect sensitive species of fish and invertebrates and is, thus, the boundary between moderate and poor. Dissolved oxygen levels below 2.5 mg L^{-1} negative impact most fish species and is thus the boundary between poor and bad condition. Overall condition is based on the combination of the three indices and is summarized in Table 3.6.

Table 3.6. UK WFD classification based on deviation from reference conditions. Classification is assessed via progression through the three indices [Devlin et al., 2007a]. Bold line indicators management action point.

		Index 1: Nutrient Concentration	**Index 2: Production**	**Index 3: Undesirable Disturbance**
Statistic for Index		**Mean Winter DIN (µM)**	**Growing Season Potential Primary Productivity**	**Growing Season Mean Dissolved Oxygen Concentration**
Units		µM	g C m^{-2} y^{-1}	mg L^{-1}
Index		I_{DIN}	I_{PP}	I_{DO}
Classification	**High**	$I_{DIN} \leq 12$	n/a	n/a
	Good	$I_{DIN} \leq 18$	n/a	n/a
	Good		$I_{DIN} \geq 30$ **µM** $I_{PP} < 300$	$I_{DO} > 5$
	Moderate		$I_{DIN} \geq 30$ **µM** $I_{PP} \geq 300$	$I_{DO} > 5$
	Poor		$I_{DIN} \geq 30$ **µM** $I_{PP} \geq 300$	$I_{DO} \leq 5$
			$I_{DIN} \geq 30$ **µM** $I_{PP} \geq 300$	$I_{DO} \leq 2$

UK WFD Phytoplankton Biological Quality Element . There are three indicators proposed for the phytoplankton biological quality element of the UK WFD for coastal waters: 1) phytoplankton biomass measure as CHL-a, 2) the frequency of elevated phytoplankton counts measuring individual species and total cell counts, and 3) seasonal progression of phytoplankton functional groups through the year [*Devlin et al.*, 2007b]. The first index, phytoplankton biomass as CHL-a (I_{CHL}), is defined as the 90th percentile of chlorophyll concentrations during the growing season (March to September). The boundary conditions are different by salinity strata. For marine waters, the reference value is proposed as 10 µg L^{-1} (implying 50% elevation of the background value of 6.7 µg L^{-1} and a reasonable C:Chl factor of 0.012). For low salinity waters, where the level of production may be expected to be higher, a reference value of 15 µg L^{-1} is proposed (implying a background value of 10 µg L^{-1} chlorophyll and a C:Chl factor of 0.02; Table. 3.X)[*Devlin et al.*, 2007b].

Table 3.7 Thresholds for concentrations of chl a, dissolved oxygen and dissolved inorganic nitrogen for the UK WFD assessment method. From Devlin et al. 2011.

<table>
<tr><th>Method</th><th>Chl-a reference thresholds (µg l^{-1})</th><th>DO reference thresholds (mg l^{-1})</th><th>DIN reference thresholds</th><th>Source and criteria</th></tr>
<tr>
<td>UK WFD</td>
<td>
BIOMASS INDICATOR:
In the UK, five statistical measures of Chl-a biomass are made over two salinity bands using combined data for a six year reporting cycle. Compliance with the threshold is given a score of 1 for each statistical measurement, with an optimum score of 10. The final score (a value between 0 [bad] to 10 [best]) is normalized to an equidistant EQR score (0 – 1).

SALINITY RANGE – LOW (0 – 25ppt)
<table>
<tr><th>No</th><th>Chl-a measurement</th><th>Threshold</th></tr>
<tr><td>1:</td><td>Average Chl-a conc,</td><td>≤15µg l^{-1}</td></tr>
<tr><td>2:</td><td>Median Chl-a conc,</td><td>≤12µg l^{-1}</td></tr>
<tr><td>3:</td><td>% Chl-a less than 10 µg l^{-1}</td><td>>70%</td></tr>
<tr><td>4:</td><td>% Chl-a less than 20 µg l^{-1}</td><td>>80%</td></tr>
<tr><td>5:</td><td>% Chl-a greater than 50 µg l^{-1}</td><td><5%</td></tr>
</table>
SALINITY RANGE – HIGH (>25 ppt)
<table>
<tr><th>No</th><th>Chl-a measurement</th><th>Threshold</th></tr>
<tr><td>6:</td><td>Average chl-a conc,</td><td>≤10µg l^{-1}</td></tr>
<tr><td>7:</td><td>Median chl-a conc,</td><td>≤8µg l^{-1}</td></tr>
<tr><td>8:</td><td>% Chl-a less than 10 µg l^{-1}</td><td>>75%</td></tr>
<tr><td>9:</td><td>% Chl-a less than 20 µg l^{-1}</td><td>>85%</td></tr>
<tr><td>10:</td><td>% Chl-a greater than 50 µg l^{-1}</td><td><5%</td></tr>
</table>
<table>
<tr><th>TOTAL SCORE</th><th>EQR</th><th>STATUS CLASS</th></tr>
<tr><td>0 - 2</td><td>0.000 - 0.133</td><td>Bad</td></tr>
<tr><td>3 - 4</td><td>0.200 - 0.300</td><td>Poor</td></tr>
<tr><td>5 - 6</td><td>0.400 - 0.500</td><td>Moderate</td></tr>
<tr><td>7 - 8</td><td>0.600 - 0.700</td><td>Good</td></tr>
<tr><td>9 - 10</td><td>0.800 - 1.000</td><td>High</td></tr>
</table>
TAXA ABUNDANCE INDICATOR:
Cell counts for each sampling period are used to calculate the number of times the threshold is exceeded (as %) when:
1.Any Single taxon (species) >500,000 cell l^{-1}
2.Total Abundance > 10^{6} cells l^{-1}

<table>
<tr><th>% exceedances</th><th>Normalised score (= ref/value)</th><th>Final EQR</th></tr>
<tr><td>0-10</td><td>1.0--0.5</td><td>0.8 - 1.0</td></tr>
<tr><td>10–20</td><td>0.5--0.25</td><td>0.6 - 0.8</td></tr>
<tr><td>20–40</td><td>0.25--0.13</td><td>0.4 - 0.6</td></tr>
<tr><td>40–60</td><td>0.13--0.08</td><td>0.2-0.4</td></tr>
<tr><td>60–100</td><td>0.08--0.0</td><td>0 - 0.2</td></tr>
</table>
Normalised score calculated by reference condition (5%) divided by value. Final EQR normalised to equidistant boundaries (0 to 1)
</td>
<td>
Annual 5th percentiles:
>5.7 mg l^{-1} = High
4.0 <5.7 mg l^{-1} = Good
2.4<4.0 mg l^{-1} = Mod
1.6 <2.4 mg l^{-1} = Poor
<1.6 mg l^{-1} = Bad

EQR
High: 0.8 – 1.0
Good: 0.6 – 0.8
Mod: 0.4 – 0.6
Poor: 0.2 – 0.4
Bad: 0.0 – 0.2

For DO, 5th percentile of all data (collected monthly). Reporting period is typically over 6 years.
</td>
<td>
Winter DIN thresholds for UK estuaries:
(clear estuaries)
as µM: as mg l^{-1}:
<20 = High <0.28 = High
<30 = Good <0.42 = Good
<45 = Mod <0.63 = Mod
<67 = Poor <0.94 = Poor
>67 = Bad >0.94 = Bad

These thresholds are for clear waters only, defined by mean annual SPM (SPM < 10 mg l^{-1})
In turbid waters (>10 mg l^{-1} SPM), a secondary threshold (70 µM, 0.98 mg l^{-1}) may be applied, where
<70 µM (0.98 mg l^{-1}) = Good
>70 µM (0.98 mg l^{-1}) = Moderate

If the secondary threshold is applied, then 99th percentiles are calculated from the data and compared to the threshold.
</td>
<td>
Chl-a
Devlin et al., 2007a, 2007b, www.ukwfd.org

DO
Best et al., 2007

DIN
Devlin et al., 2007a
www.ukwfd.org.au
</td>
</tr>
</table>

Table 3.7 continued

Method	Chl-*a* reference thresholds ($\mu g\ l^{-1}$)	DO reference thresholds ($mg\ l^{-1}$)	DIN reference thresholds	Source and criteria
ASSETS	*Annual 90th percentile*: 0-5 $\mu g\ l^{-1}$ = Low 5-20 $\mu g\ l^{-1}$ = Moderate >20 $\mu g\ l^{-1}$ = High >60 $\mu g\ l^{-1}$ = Hypereutrophic *Spatial coverage of worst case conditions:* 0-10% = Very Low 10 – 25% = Low 25% - 50% = Moderate >50% = High *Frequency of occurrence of worst case conditions*: Persistent Periodic Episodic	*Annual 10th percentile*: 0 $mg\ l^{-1}$ = Anoxia 0-2 $mg\ l^{-1}$ = Hypoxia 2-5 $mg\ l^{-1}$ = Biologically Stressful *Spatial coverage of worst case conditions:* 0-10% = Very Low 10 – 25% = Low 25% - 50% = Moderate >50% = High *Frequency of occurrence of worst case conditions*: Persistent Periodic Episodic	ASSETS does not use nutrient concentrations in the assessment formulation, only nutrient loads	Bricker et al. 1999, 2003, 2007 **Chl-*a*** 90th percentile of annual data is used. These thresholds are used for all systems except Florida Bay for which thresholds are lower (i.e. High [worst] for Florida waters = 2 – 5 $\mu g\ l^{-1}$) **DO** 10th percentile of annual data is used
EPA NCA	*Summer value*: 0-5 $\mu g\ l^{-1}$ = Good 5-20 $\mu g\ l^{-1}$ = Fair >20 $\mu g\ l^{-1}$ = Poor	*Summer value*: >5 $mg\ l^{-1}$ = Good 2-5 $mg\ l^{-1}$ = Fair <2 $mg\ l^{-1}$ = Poor	*Summer value:* as µM: as mg l^{-1}: <7 = Good <0.1 = Good 7-36 = Fair 0.1-0.5 = Fair >36 = Poor >0.5 = Poor	EPA 2001, 2005, 2008 Data from the summer index period are used for determination of Chl-*a* and DIN condition at individual sites of US East, Gulf and West coast systems. Reference conditions are different for sensitive waterbodies
OSPAR COMPP	*Growing season 90th percentile:* Threshold = 15 $\mu g\ l^{-1}$ >15 $\mu g\ l^{-1}$ = threshold exceeded indicating a Problem Area Maximum and mean concentrations may also be compared to this threshold.	*5th percentile of growing season data:* Threshold = 4 $mg\ l^{-1}$ <4 $mg\ l^{-1}$ = threshold exceeded indicating a Problem Area	*Winter DIN for UK estuaries*: Threshold = 30 µM (0.42 $mg\ l^{-1}$) >30 µM = threshold exceeded indicating a Problem Area N:P Ratio Threshold: 24:1 where >24:1 is indicative of a Problem Area	OSPAR 2005, 2008 A one-out-all-out procedure is used to determine the classification of each of the four Categories and for the Overall Assessment

Thresholds are not used in the TRIX approach. For DIN, the WFD (as applied within the UK) and OSPAR use µM units, while EPA NCA uses mg l^{-1}. Thresholds are given here in both units to enable comparisons among methods

Mod moderate, *SPM* suspended particulate matter

The second index, elevated phytoplankton abundance (I_E), assesses the presence, abundance and frequency of occurrence of elevated counts of algal species relative to undisturbed conditions. This index is based on three attributes, one which is a measure of the frequency that elevated biomass (CHL) exceeds a reference threshold and three of which focus on counts of algae that may result in the decline of ecosystem health in an undesirable disturbance (Table 3.8) [*Devlin et al.*, 2007b]. Each attribute is calculated from the number of times it exceeds the threshold as a proportion of the total number of sampling times per year, and is recorded as a six year mean. The proposed thresholds are for three groups of phytoplankton and for counts of chlorophyll exceeding a threshold. The first phytoplankton threshold identifies any species of phytoplankton, excluding *Phaeocystis* species, that exceed counts of 10^6 cells L^{-1} [S], the second phytoplankton threshold identifies *Phaeocystis sp*. that exceed counts of 10^6 cells L^{-1} [P], and the third threshold identifies where the total taxa counts exceeds counts of 10^7 cells L^{-1} [T]. The chlorophyll count within this index identifies any chlorophyll measurement that exceeds 10 µg L^{-1}. The final index is calculated as the sum of these attributes: $I_E = \Sigma$ (CHL + S + P + T).

Table 3.8 Proposed boundary conditions for phytoplankton abundance relating to occurrences of elevated taxa counts over a six year period. From Devlin et al. 2007b.

Normative definition	Index	Equation – {sum $[T]+[P]+[S]+[\text{chl}]../4\} * 100$	Classification boundaries	
Phytoplankton abundance	I_E	I_E: Sum of the occurrence of any single species ($>10^6$) plus *Phaeocystis* sp. ($>10^6$), plus total cell counts ($>10^7$) and counts of chlorophyll >10 μg l^{-1} over a six year period	High Good Moderate Poor Bad	<15% <30% <40% <50% >50%

This index is composed of counts of four attributes within the tool. Samples are taken in growing season between April and September.

The third index, seasonal succession of functional groups (I_F), represents the deviation of the natural progression of dominant functional groups throughout the seasonal cycle relative to undisturbed conditions. Counts of four major functional groups, including diatoms, dinoflagellates, microflagellates (excluding *Phaeocystis*) and *Phaeocystis sp*. are averaged for each month over a sampling year, and are normalized and reported as a monthly Z score. Monthly Z scores for each functional group are compared to a specific reference curve for different classes of waterbodies. A final score is based on the number of data points from the test waterbody which fell within the standard deviation range set for each monthly point of the reference growth curve [*Devlin et al.*, 2007b].

Trophic Index (TRIX)

TRIX integrates oxygen saturation, phytoplankton chlorophyll-a, nitrogen and phosphorus concentrations to assess the trophic state of coastal marine waters and lagoons [*Giovanardi and Vollenweider*, 2004; *Vollenweider et al.*, 1998]. TRIX is based on the assumption that eutrophication processes are mainly reflected by changes in the phytoplankton community, which is typically only true for coastal waters and estuaries dominated by deep subtidal habitat. It was developed for use in Italian coastal waters and lagoons. The index is given by equation 1:

Equation 1 $$TRIX= [\log_{10}(CHLa * \%DO * N * P) + 1.5] / 1.2$$

where CHLa is the chlorophyll-a concentration (μg L^{-1}), %DO is dissolved oxygen represented as the absolute percent deviation from saturation (%), N is the concentration of dissolved inorganic nitrogen (ammonia + nitrate + nitrite) in μg-at L^{-1}, P is the concentration of dissolved inorganic phosphorus as phosphate (μg-at L^{-1}). The TRIX score is scaled from 0 to 10, covering a range of four trophic states (0-4 high quality and low trophic level; 4-5 good quality and moderate trophic level; 5-6 moderate quality and high trophic level and 6-10 degraded and very high trophic level).

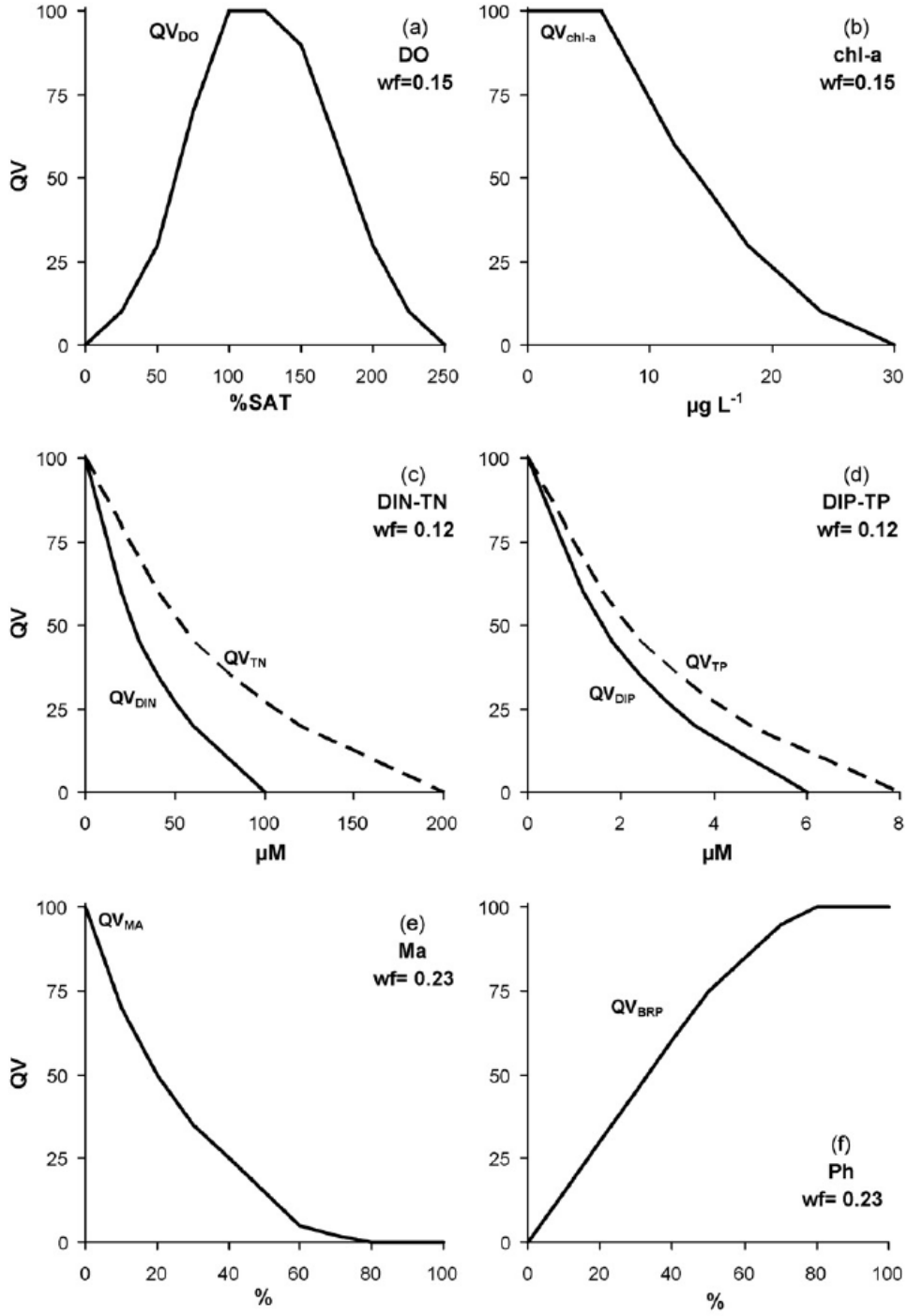


Figure 3.1. Relationships among analytical measurements of (a) dissolved oxygen saturation (DO), (b) chlorophyll-a (Chl-a), (c) dissolved inorganic and total nitrogen (DIN-TN), (d) dissolved inorganic and total phosphorus (DIP-TP), (e) macroalgal coverage (Ma), (f) phanerogam coverage (Ph) and respective Q values (QV). wf: weighting factors used in TWQI calculation[Giordani et al., 2009].

Assessment of Estuarine Trophic Status (ASSETS)

ASSETS is an integrated methodology used to comparatively rank the eutrophication status of estuaries and coastal areas. It was developed for use in the U.S. National Estuarine Eutrophication Assessment (NEEA), but has been extended and refined for use in other estuarine systems around the world. The methodology is described in detail elsewhere [*Bricker et al.*, 2003; *Bricker et al.*, 1999].

The ASSETS assessment includes three diagnostic tools: an assessment of pressure (influencing factors [IF]), an evaluation of state (eutrophic condition [EC]), and the expected response (future outlook [FO])[*Bricker et al.*, 2003; *Bricker et al.*, 1999; *Devlin et al.*, 2011; *Garmendia et al.*, 2012]. The IF assessment is based on two factors: the nutrient loading (input) from the watershed and/ or ocean and the susceptibility of the system (capability of the system to dilute or flush the nutrient inputs). The overall IF falls into one of five categories (low, moderate-low, moderate, moderate-high, and high) that are determined by a matrix that combines susceptibility and load factors. The EC is evaluated based on a combination of primary and secondary symptoms of eutrophication sampled monthly. The two primary symptoms are phytoplankton (evaluated as CHL-a concentration, frequency, and spatial coverage) and macroalgae (magnitude and frequency of "problem status," where "problem" indicates a detrimental impact on any biological resource). The three secondary symptoms are bottom water dissolved oxygen (concentration, spatial coverage, and frequency of low events), nuisance and toxic blooms (duration and frequency of "problem status"), and submerged aquatic vegetation (SAV) ("problem status or change in spatial coverage" and the magnitude of the change)[*Bricker et al.*, 1999; *Garmendia et al.*, 2012]. The EC rating is determined by a matrix that combines the average score of the primary symptoms (chlorophyll "a" and macroalgae) and the highest score (worst impact) of the secondary symptoms (dissolved oxygen, nuisance and toxic blooms and SAV) and categorizes estuaries into one of five categories (low, moderate-low, moderate, moderate-high, and high). The FO rating, is determined by a matrix that combines the susceptibility and expected change in loading factors and classifies estuaries into one of the five categories (worsen-high, worsen-low, no change, improve-low, and improve-high). The assessment then combines results of the three components into a single overall rating of bad, poor, moderate, good, and high trophic status using a matrix approach [*Bricker et al.*, 2003; *Bricker et al.*, 1999; *Devlin et al.*, 2011; *Garmendia et al.*, 2012]. Thresholds for each indicator are given in Table 3.3.

Table 3.9. Indicators and thresholds applied in the ASSETS framework [Bricker et al., 2003].

	Index	Indicator	Statistic for Index	Thresholds and Ranges
Primary Symptoms	**Phytoplankton**	CHL-a	90th percentile of monthly data	Hypereutrophic: > 60 □g L^{-1} High: > 20 □g L^{-1} but ≤ 60 □g L^{-1} Medium: > 5 □g L^{-1} but ≤ 20 □g L^{-1} Low: ≤ 5 □g L^{-1}
		Spatial Coverage	Heuristic of Monthly Data	High, Moderate, Low, or Very Low
		Frequency		Periodic, Episodic, or Persistent
	Macroalgae or Epiphytes	Biomass and Cover	Heuristic of Monthly Data	Problem: detrimental impact to biological resources No Problem: no apparent impact on biological resources
		Spatial Coverage		High, Moderate, Low, or Very Low
		Frequency		Periodic, Episodic, or Persistent
Secondary Symptoms	**Dissolved Oxygen**	Bottom water Concentration	10th percentile of monthly data	Anoxia: 0 mg L^{-1} Hypoxia: > 0 mg L^{-1} but ≤ 2 mg L^{-1} Biologically Stressful: > 2 mg L^{-1} but ≤ 5 mg L^{-1}
		Spatial Coverage	Heuristic of Monthly Data	High, Moderate, Low, or Very Low
		Frequency		Periodic, Episodic, or Persistent
	SAV Loss	Magnitude of Loss	Analysis of Monthly Data	High Loss: ≥ 50 but ≤ 100 % of estuarine surface water area Medium Loss: ≥ 25 but > 50% of estuarine surface water area Low: ≥ 10 but > 25% of estuarine surface water area Very Low: ≥ 0 but > 10% of estuarine surface water area
	Nuisance and Toxic Blooms	Observed Occurrence	Cell Counts of Dominant Species	Problem: detrimental impact to biological resources No Problem: no apparent impact on biological resources
		Duration	Monthly Data	Hours, Days, Weeks, Seasonal, Other
		Frequency	Heuristic of Monthly Data	Periodic, Episodic, or Persistent

OSPAR

OSPAR is the mechanism by which fifteen Governments of the western coasts and catchments of Europe, together with the European Community, cooperate to protect the marine environment of the North-East Atlantic. The OSPAR Eutrophication Strategy sets the objective to combat eutrophication in the OSPAR maritime area. The OSPAR Common Procedure is used to identify the eutrophication status and assess compliance with the Ecological Quality Objectives (EcoQO) for eutrophication for the North Sea (www.OSPAR.org).

The specific Ecological Quality Objectives for eutrophication agreed at the 5th North Sea Conference (Bergen Declaration 2002) are (OSPAR 2005):

- Winter DIN and/or DIP should remain below elevated levels, defined as concentration > 50% above salinity related and/or region-specific natural background concentrations;
- Maximum and mean region-specific chlorophyll a concentrations during the growing season should remain below region-specific elevated levels, defined as concentrations > 50% above the spatial (offshore) and/or historical background concentration;
- Region/area-specific phytoplankton eutrophication indicator species should remain below respective nuisance and/or toxic elevated levels (and increased duration);
- Oxygen concentration, decreased as an indirect effect of nutrient enrichment, should remain above region specific oxygen deficiency levels, ranging from 4-6 mg oxygen per litre;
- There should be no kills in benthic animal species as a result of oxygen deficiency and/or nuisance/toxic phytoplankton indicator species for eutrophication.

Under OSPAR (2005), nutrient concentrations are assessed by plotting the winter nutrient concentrations of each year in relation to the respective measured salinity values ("mixing diagrams"). In winter, defined as period when algal activity is lowest, DIN and DIP show a conservative behavior and, therefore, a good linear relationship with salinity (decreasing concentration with increasing salinity from coast to offshore). The salinity normalized nutrient concentration (with 95% confidence interval) is plotted in relation to the respective year in order to establish trends in the winter nutrient concentrations and the level of elevation (compared with background concentration).

In determining the maximum and mean chlorophyll a levels in estuaries, chlorophyll a concentrations are averaged over the salinity range during the growing season. Table 3.10 gives the area-specific natural background and elevated concentrations of chl-a.

Table 3.10 Area specific background concentrations and elevated nutrient concentrations of chlorophyll a during growing season in relation to salinity. From OSPAR 2005.

	Region	Salinity	Background concentration	Background concentration	Elevated levels
			Chlorophyll *a* µg/l, means	Chlorophyll *a* µg/l, maxima	Chlorophyll *a* µg/l, means
North Sea	Belgium		10	15	
	Coast		10		>15
	Denmark	>34.5	2-4		>4.5
	Coast	<34.5	2-10		
	Germany	>34.5	2	10-13	3
	Coast	<34.5	2-4	13-18	3-6
	Netherlands	>34.5	2-4		>4.5
	Coast	<34.5	10	10	>15
	Norway		2-4		>4.5
	Coast		2-10		
	UK	>34.5	5-10	10	>10
	Coast	<34.5	8-12	15	>20
Channel	France	>34	2	10	> 4
Wadden Sea	Denmark	<30			>22-24 (needs verification)
	Germany	29-32	2-4	12-20	3-6
	Netherlands	<30		16	>22-24 (needs verification)
Skagerrak	Denmark	32-34	<1.25		
	Norway	33			
	Sweden		1.5		>2
Kattegat	Denmark		1.5		>2
	Sweden		1.5	1.5	>2
Atlantic	France		2	10	>4
	Ireland coast	>34.5	<7		>10
	Norway				
	Portugal				
	Spain	>34.5			>12
	coast			8	
	UK/Scotland	>34.5	5	10	>10
	coast	<34.5	10	15	>15
Southern Irish Sea and Eastern Celtic Sea	Ireland Offshore	>34.8			
Atlantic to Irish Sea	Coast	>34.5	<7		>10
Estuaries	Belgium				

Table. 3.10 Continued

	Region	Salinity	Background concentration	Background concentration	Elevated levels
			Chlorophyll *a* µg/l, means	Chlorophyll *a* µg/l, maxima	Chlorophyll *a* µg/l, means
	Denmark				
	France			13 (variable at sal <30)	>18-20(variable at sal < 30)
	Germany	0-30	5-8	12-40	7.5-12
	Ireland				
	Netherlands	<30		2-6	
	Western Scheldt				>9-10
	Ems Dollar				>18-20
	Norway				
	Portugal:				
	Sado		10?		>9
	Tagus				>14
	Mondego		10?		>9
	Spain				
	Sweden				
	UK				

OSPAR distinguishes two types of phytoplankton indicator species: nuisance species (forming dense "blooms") and toxic species (already toxic at low cell concentrations). Examples of levels considered as elevated levels and their effects are provided in Table 3.11. Use of nuisance and toxic blooms has not seen wide-spread use because of uncertainty in linkage to anthropogenic nutrients.

Table 3.11 Elevated levels of area-specific nuisance and toxic phytoplankton indicator species and the types of their effects. From OSPAR 2005.

Phytoplankton indicator species	Elevated levels	Effects
Nuisance species		
Phaeocystis spp. (colony form)	> 10^6 cells/l (and >30 days duration)	nuisance, foam, oxygen deficiency
Noctiluca scintillans	> 10^4 cells/l (area coverage > 5 km^2)	nuisance, oxygen deficiency
Toxic (toxin producing) species		
Chrysochromulina polylepis	> 10^6 cells/l	toxic; fish and benthos kills
Gymnodinium mikimotoi	> 10^5 cells/l	toxic; fish kills, PSP mussel infection
Alexandrium spp.	> 10^2 cells/l	toxic; PSP mussel infection
Dinophysis spp.	> 10^2 cells/l	toxic; DSP mussel infection
Prorocentrum spp.	> 10^4 cells/l	toxic; DSP mussel infection

HELCOM Eutrophication Assessment Tool (HEAT)

HEAT is a multi-metric indicator-based tool for assessment of eutrophication status [*HELCOM*, 2009]. HEAT has been developed specifically for the HELCOM Integrated Thematic Assessment of Eutrophication in the Baltic Sea. Ecological objectives related to eutrophication were adopted in the HELCOM Baltic Sea Action Plan. They are: concentrations of nutrients close to natural levels, clear water, natural level of algal blooms, natural distribution and occurrence of plants and animals, and natural oxygen levels [*HELCOM*, 2009]. HEAT is an indicator based assessment framework which groups indicators as follows: (1) physical- chemical features (PC), (2) phytoplankton (PP), (3) submerged aquatic vegetation (SAV), and (4) benthic invertebrate communities (BIC). Groups 1 and 2 (PC and PP) are considered 'primary signals' of eutrophication, while groups 3 and 4 (SAV and BIC) are considered 'secondary signals' [*HELCOM*, 2009]. For each indicator a eutrophication quality objective (EutroQO) or target is calculated from the reference condition (RefCon) and the acceptable deviation (AcDev) from reference condition. When the actual status (AcStat) exceed the EutroQO, the area in question is regarded as 'affected by eutrophication'' or falling below the "good-moderate" threshold [*Andersen et al.*, 2011].

Reference Conditions (RefCon), are the biological quality elements that exist, or would exist, with no or very minor disturbance from human activities. They should represent the continuum that is naturally present and must reflect variability. The HEAT tool uses three principles for setting RefCons: (1) reference sites, (2) historical data, and (3) modeling. Expert judgment can also be used as a supplement. RefCons as applied in the Baltic sea were typically basin specific and varied by an order of magnitude over the salinity gradient of the sea.

The acceptable deviation (AcDev) values are basin specific. Two different principles were used for setting the AcDev, according to whether indicators show a positive response (increasing in value) to increases in nutrient inputs or a negative response (decreasing in value). For an indicator showing positive response (e.g. nutrient concentrations and chlorophyll-a), AcDev has an upper limit of +50% deviation from RefCon [*HELCOM*, 2009]. Setting AcDev to 50% implies that low levels of disturbance (defined as less than +50% deviation) resulting from human activity are considered acceptable while moderate (greater than +50%) deviations are unacceptable (boundary between good and moderate in the WFD) [*Andersen et al.*, 2011]. For indicators responding negatively to increases in nutrient input (e.g. Secchi depth and depth limit of SAV) the AcDev's have in principle a limit of -25% [*HELCOM*, 2009], although AcDev's used for benthic invertebrates are slightly greater in magnitude, ranging from -27 to -40% [*HELCOM*, 2009]. Whereas an indicator with positive response can theoretically show unlimited deviation, indicators showing negative response have a maximum deviation of -100% and a deviation of -25% is, in most cases, interpreted as the boundary between good and moderate in the WFD [*Andersen et al.*, 2011].

Each site is assigned an ecological condition category as set up by the WFD: high (best condition), good, moderate, poor, and bad (worst condition) [*HELCOM*, 2009]. To assign a category, an Ecological Quality Ratio (EQR) is calculated for each site based on the RefCon and

AcStat. The boundary between good and moderate status is where the deviation from RefCon is equal to the AcDev. All other categories are assigned based on a defined deviation of the AcStat from RefCon [*Andersen et al.*, 2011]. An EQR value and a set of class boundaries are calculated for each indicator, but the overall status classification depends on a combination of indicators. First, indicator EQR values are combined to give an EQR value for a specific Quality Element (QE), and similarly the indicator class boundaries are combined to give the class boundaries for the QE. In the simplest case, where all indicators within a QE have equal weights, the EQR for the QE is the average of the indicators' EQRs within the QE and each QE class boundary (e.g. Moderate/Good boundary) is found as the average of the class boundary values for all indicators representing that specific QE. Within a QE, it is also possible to assign weighting factors to indicators according to expert judgment. The classification of the QE is then given by comparison of the weighted averages of the EQRs with the weighted averages of the individual class boundaries. Thus, the same weighting is applied both in calculation of the EQR for the specific QE as well as QE class boundary values. The lowest rated of the QEs will because of the 'One out—all out' principle determine to final status classification [*Andersen et al.*, 2011].

Transitional Water Quality Index (TWQI)

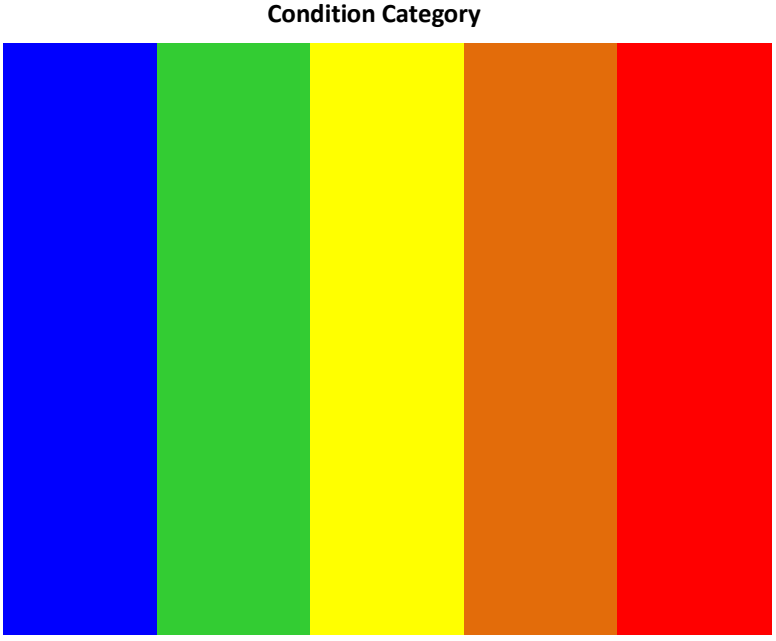
The TWQI was developed to assess trophic status and water quality in transitional (i.e. estuarine) aquatic ecosystems of Southern Europe [*Giordani et al.*, 2009]. It was developed specifically for shallower estuarine systems, where benthic vegetation controls primary productivity, making phytoplankton only indices unsuitable. The index was based on the water quality index of the U.S. National Sanitation Foundation and integrates the main causal factors (inorganic nutrients), key biological elements (primary producers) and indicator effects (dissolved oxygen). The TWQI utilizes six main variables: relative coverage of seagrass and opportunistic macroalgae species, concentration of dissolved oxygen, phytoplankton chlorophyll-a, dissolved inorganic nitrogen and phosphorus. Non-linear functions are used to transform each measured variable into a Quality Value (QV) (Figure 3.1.) [*Giordani et al.*, 2009]. Each quantity is then multiplied by a weighting factor to account for the relative contribution of each variable to the overall water quality (adding up to a total percentage of 100): dissolved oxygen = 15%, CHL-a = 15%, DIN-TN = 12%, DIP-TP = 12%, macroalgal coverage = 23%, seagrass coverage = 23%. The QV_{DO} for dissolved oxygen follows a bell shaped curve where the QV increases from 0 to 100 from dissolved oxygen levels of 0 percent saturation to 125 % saturation and decreases again from 100 to 0 as DO saturation increases from 125% to 250% (saturations over 125% are often associated with blooms in primary producer groups). The QV_{CHLa} is zero (worst condition) when concentrations of CHL-a are greater than 30 mg m^{-3} and 100 (best condition) when CHL-a concentrations are less than 6 mg m^{-3}. The QV_{DIN} is inversely related to DIN concentrations where QV_{DIN} is 100 when DIN is 0 µM and QV_{DIN} is 0 when DIN is greater than 100 µM. The most significant decrease in QV_{DIN} is imposed at the 0-20 µM range because the main transformation in primary production was found to occur in this range [*Viaroli et al.*, 2008], and it was found to be a critical threshold for other lagoons (see Souchu et al. 2000). The QV_{DIP} was set up similar to QV_{DIN} where QV_{DIP} is 100 when DIP is 0 µM and QV_{DIP} is 0 when DIP is greater than 6 µM. The QV_{Ph} and QV_{Ma} are based on the percent of estuarine surface area colonized. The QV_{Ma} is zero (worst condition) when macroalgae percent cover

exceeded 80% of estuarine surface area and 100 (best condition) when macroalgae percent cover was less than 10%. The utility function for seagrass was opposite to macroalgae such that QV_{Ph} is zero (worst condition) when seagrass percent cover was less than 10% of estuarine surface area and 100 (best condition) when seagrass percent cover was greater than 80%. An index value is calculated as the sum of the weighted quality values, ranging from 0 (poorest) to 100 (best condition). The index has been tested and validated in several estuarine systems that differ in anthropogenic pressures and eutrophication levels.

The French Research Institute for the Exploration of the Sea (IFREMER) Classification for Mediterranean Lagoons

The IFREMER developed a classification scheme for benthically-dominated French Mediterranean lagoons [*Souchu et al.*, 2000; *Zaldivar et al.*, 2008], which is based on several physical, chemical and biological potential indicators of eutrophication in the various components of the lagoon ecosystem: benthic, phytoplankton, macrophytes, macrofauna, sediments and water. It allows for the classification of a lagoon into five eutrophication levels formalized by five different colors from blue (no eutrophication), green, yellow, orange, and red (high eutrophication), similar to the color scheme used by the Water Framework Directive (WFD). Overall classification is based on the worst partial value of the elements listed above. Each component of the ecosystem is assessed independently allowing for identification of which component is experience degradation. Indicators are scored against thresholds based on an annual average of the data. Elements and thresholds used to assess the water column are presented in Table 3.12. Thresholds are based on an annual average of data collected.

Table 3.12. Water quality elements and thresholds measured in the IFREMER assessment framework for French Mediterranean lagoons. Eutrophication is scored from blue (no eutrophication) to red (high eutrophication) [Souchu et al., 2000; Zaldivar et al., 2008].



U.S. EPA's National Coastal Assessment

The US EPA's National Coastal Assessment (NCA) is implemented through a federal—state partnership, and is designed to answer questions on environmental conditions in coastal waterbodies at a regional – national scale. The results supplement the US Clean Water Act (CWA) where waterbodies identified as not meeting state water quality criteria for designated uses require actions to correct pollution caused impairments [*USEPA*, 2001; 2005; 2008]. Of the five EPA NCA indices of condition in coastal waterbodies, the Water Quality Index (WQI) is the indicator describing nutrient related conditions and will be the only one reviewed here. This method uses five indicators: dissolved inorganic nitrogen (DIN), dissolved inorganic phosphorus (DIP), Chl-a, water clarity (by Secchi depth and by comparison of light reaching the water surface and at 1 m depth) and dissolved oxygen. The WQI uses the EPA Environmental Monitoring and Assessment Program's (EMAP) probabilistic randomly selected sampling framework where samples are taken once per year (per station) by region during a summer index period (June through September; [*USEPA*, 2001]). An evaluation is made for each of the five indicators at each site by comparison with regionally defined reference conditions and a combined water quality index rating is calculated for each site, then for the region and the nation based on the ratio of individual indicators that are rated as Good, Fair or Poor [*Devlin et al.*, 2011]. Thresholds for each indicator are based on assumed reference conditions, are given in Table 3.13.

An indicator is considered Good if less than 10% of samples are Poor and 50% are Good; condition is fair if 10–25% of samples are Poor and/or 50% are Poor or Fair; and condition is Poor if more than 25% of samples are Poor. All indicators are combined in a similar fashion to determine the rating for a site: where Good is a maximum of one indicator is Fair and no indicators are Poor; Fair is one of the indicators is rated Poor or two or more indicators are Fair; and Poor is two or more of the five indicators are rated Poor. To determine the WQI by region and nation, results from each area are used to determine a final assessment score where: Good is less than 10% of areas are in Poor condition and more than 50% are Poor or Fair; Fair is 10–20% of areas are in Poor condition or greater than 50% are Fair or Poor; and Poor if greater than 20% of areas are in Poor condition.

Table 3.13. Thresholds for each indicator used in the US EPA NCA [Devlin et al., 2011].

	Classification

Indicator Specific Assessment Frameworks-Phytoplankton Index of Biotic Integrity

One use of phytoplankton community structure data is to combine it into an index of biological integrity (IBI). IBIs are becoming more common for assessment of estuarine ecological condition and management focus in the face of physical and chemical transformation, habitat destruction, and changes in biodiversity (Borja et al. 2008). An IBI describes the biological condition of an assemblage of plants or animals, typically based on the diversity and relative abundance of species or the presence or absence of pollution tolerant species. A key element of developing an IBI is the ability to describe the community response of the assemblage (e.g., benthic invertebrates, phytoplankton, etc.) along gradient of physical or chemical stress from minimally disturbed or "reference state" to highly disturbed.

IBIs developed and used in Chesapeake Bay present an example of how phytoplankton community structure data can be synthesized to provide information about the ecological health of the Estuary and about the ability to support specific beneficial uses. A Phytoplankton Index of Biotic Integrity (P-IBI) was developed in Chesapeake Bay using an 18 year data set (Lacouture et al. 2006). The P-IBI combined the scores of pollution-sensitive, biologically important metrics of the phytoplankton community into a single index. Like other multi-metric indexes, the P-IBI is more sensitive to habitat conditions than its component metrics, which include chlorophyll-a, the abundances of several potentially harmful species, and various indicators of cell function and species composition (Lacouture et al. 2006).

Thirty-eight phytoplankton metrics were used to quantify the status of phytoplankton communities relative to water quality conditions (Table 3.12). Least-impaired (reference) habitat conditions have low dissolved inorganic nitrogen (DIN) and orthophosphate (P04) concentrations and large Secchi depths. Impaired (degraded) habitat conditions have high DIN and P04 concentrations and small Secchi depths. The phytoplankton communities of these contrasting habitat conditions showed many significant differences (Table. 3.14, Buchanan et al. 2005). Twelve discriminatory metrics were chosen, and different combinations of these twelve metrics were scored and used to create phytoplankton community indexes for spring and summer in the four salinity regimes in Chesapeake Bay.

Table 3.14 Phytoplankton metrics examined in the development of the Chesapeake Bay Index of Biotic Integrity. From Lacouture et al. 2006.

TABLE 2. Phytoplankton metrics examined for discriminatory ability and their Kruskal-Wallis (χ^2) test results for significant difference between reference and degraded communities (* p = 0.05–0.1, ** p = 0.01–0.05, *** p < 0.01, ns = not significant). All metric values are for the above-pycnocline layer, except surface chlorophyll *a*. Carbon:chlorophyll *a* is the ratio of total nano-micro phytoplankton (2–200 µm) biomass to chlorophyll *a* in the above-pycnocline layer. Since picophytoplankton biomass is not included in the numerator, carbon:chlorophyll *a* values are somewhat underestimated in summer when picophytoplankton are most abundant. Average cell size is total biomass divided by total abundance of the nano-micro phytoplankton size fractions. See Table 1 for season and salinity zone definitions.

	Spring				Summer			
Metric	F	O	M	P	F	O	M	P
Chlorophyll *a*	**	***	**	***	***	***	***	***
Chlorophyll *a* surface	**	***	***	***	***	***	***	***
Pheophytin	***	***	***	***	***	***	***	***
Total biomass nano-micro phytoplankton	ns	**	***	ns	***	***	ns	***
Total abundance nano-micro phytoplankton	ns	***	ns	*	***	***	**	ns
Carbon:chlorophyll *a*	*	ns	***	***	ns	ns	***	***
Average cell size nano-micro phytoplankton	ns	*	***	**	ns	*	ns	***
Chlorophyte abundance	ns	ns	***	ns	***	***	***	ns
Chlorophyte biomass	ns	ns	***	ns	***	***	***	ns
Chrysophyte abundance	ns	**	ns	ns	**	**	ns	***
Chrysophyte biomass	ns	**	ns	ns	**	**	***	***
Cryptophyte abundance	ns	***	ns	***	ns	ns	*	*
Cryptophyte biomass	ns	**	ns	***	ns	ns	ns	*
% total biomass composed of cryptophytes	ns	ns	***	**	***	***	ns	***
Cyanophyte abundance	ns	*	**	ns	***	***	ns	***
Cyanophyte biomass	*	ns	**	ns	***	***	ns	***
% total biomass composed of cyanophytes	ns	ns	***	ns	ns	ns	ns	***
Diatom abundance	*	ns	ns	*	***	***	***	ns
Diatom biomass	ns	ns	***	ns	***	***	ns	***
% total biomass composed of diatoms	ns	***	***	ns	ns	***	ns	ns
Dinoflagellate abundance	ns	***	ns	*	ns	***	***	ns
Dinoflagellate biomass	ns	***	*	ns	ns	***	ns	***
% total biomass composed of dinoflagellates	ns	***	ns	ns	ns	***	ns	ns
Prasinophyte abundance	ns	***	ns	ns	ns	**	ns	**
Prasinophyte biomass	ns	***	ns	ns	ns	**	ns	**
Picoplankton abundance (Virginia only)	**	ns	*	ns	**	*	***	ns
Picoplankton biomass (Virginia only)	**	ns	*	ns	**	*	***	ns
Cochlodinium heterolobatum abundance			ns				ns	ns
Cochlodinium heterolobatum biomass			ns				ns	ns
Microcystis aeruginosa abundance	ns	ns	ns	ns	***	**	ns	ns
Microcystis aeruginosa biomass	ns	ns	ns	ns	***	**	ns	ns
Prorocentrum minimum abundance	**	***	***	***	ns	*	ns	ns
Prorocentrum minimum biomass	**	***	***	***	ns	*	ns	ns
Dissolved oxygen	ns	***	***	***	ns	**	***	***
Dissolved organic carbon	***	**	***	ns	***	***	***	ns
Particulate carbon	**	ns	***	***	***	***	ns	***
Total organic carbon	***	*	***	**	***	***	***	ns
Total suspended solids	***	***	***	***	***	***	***	***

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APPENDIX I – CATALOGUE OF SF BAY DATA AVAILABLE FOR ANALYSIS OF EXISTING DATA

The existing data available to test out assessment approaches generally falls into two categories: 1) USGS water quality sampling and 2) IEP monitoring data.

The parameters sampled and the time periods for which these data are available are summarized in this appendix.

USGS

USGS consists of a long term data set collected from 1975-2011, with the exact coverage varying by station (Figure A1.1, Table A1.1). Nutrients were sampled regularly beginning in 2004 at a subset of all stations. Parameters consist of Chl-a, DO, SPM, salinity, temp, depth, and nutrients (NO2, NO3, NH3, PO4, Si). During the period of 1992-2001, USGS also collected phytoplankton composition data. These data were analyzed by Cloern and Dulford (2005).

Figure A1.1 USGS water quality sampling stations in SF Bay.

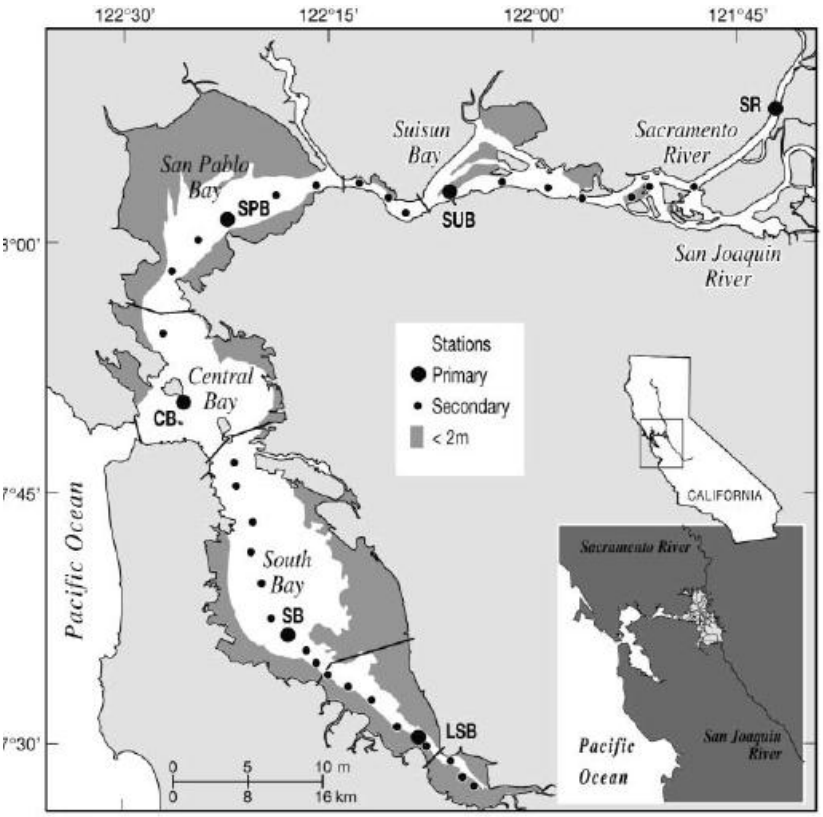

Figure A1.2 Station at which phytoplankton taxonomic composition data were collected (primary stations) during 1992-2001.

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DWR-IEP

The Department of Water Resources (DWR) and the Interagency Ecological Program (IEP) have been collecting data from 1975-2011, with exact coverage varying by station (Figure A1.3, Table A1.2). Parameters collected include Chl-a, BOD, SPM, TDS, VSS, salinity, depth, pH, DO, turbidity, temp, pheophytin-a, DOC, TOC, nutrients (NH3, TKN, NO3, NO2, DON, TON, PO4, TP, Si), and taxonomic assemblage. For the latter, 16 phytoplankton species were enumerated prior to 2008 while 21 species were enumerated from 2008-2010.

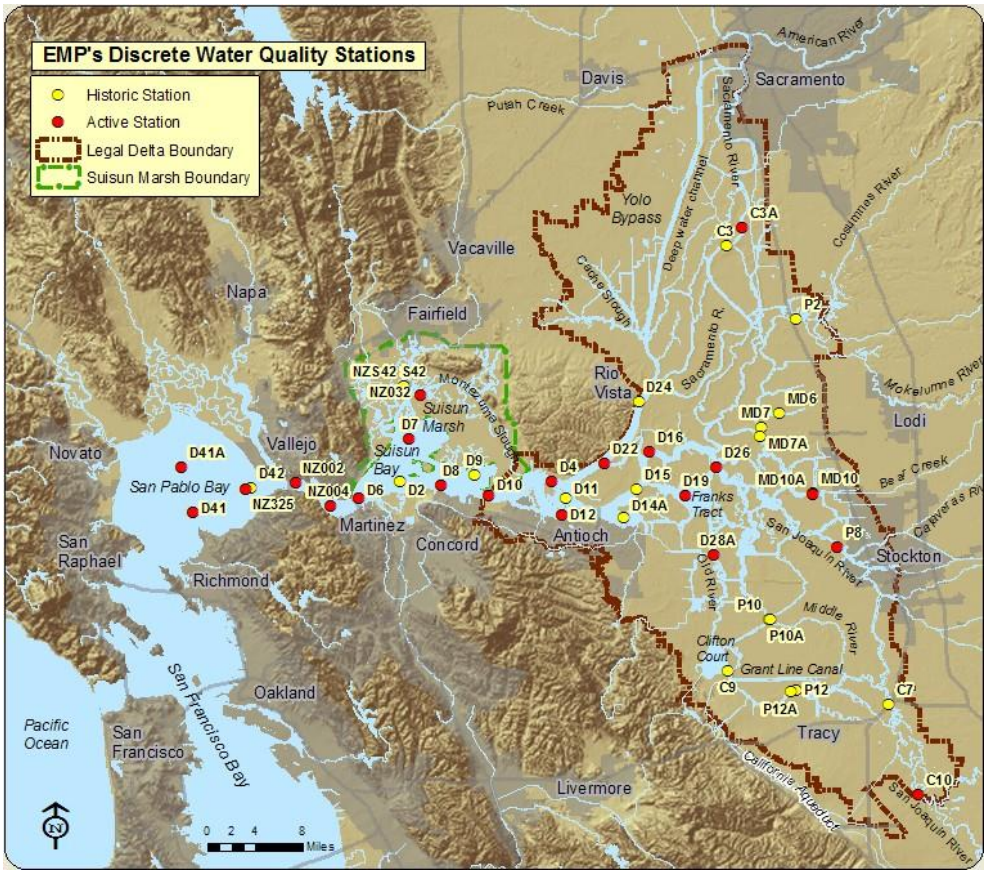

Figure A1.3 Stations sampled under the DWR-IEP monitoring program.