Long Island Sound (LIS): Application of Technical Approach for Establishing Nitrogen Thresholds and Allowable Loads for Three LIS Watershed Groupings: Embayments, Large Riverine Systems and Western LIS Point Source Discharges to Open Water

Literature Review Memo
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Submitted to EPA Region 1
Submitted by Tetra Tech

Contents

1. Introduction

Project Background

Long Island Sound (LIS) is a semi-enclosed estuary that is approximately 58 mi long from east to west and has a width of 21 mi at its widest point (Figure 1) (Gobler et al. 2006). The Sound is 1,320 mi^2 in area with 600 miles of coastline and 18 trillion gallons of water (LISS 2017a). In contrast to most estuaries, LIS does not have a freshwater source at its head (LISS 1994). Instead, higher salinity Atlantic Ocean water enters at the east through Block Island Sound and lower salinity waters enter at the west through the East River (includes New York Harbor) and Harlem River tidal straits (LISS 1994). Additionally, most (90 percent) of the freshwater enters the Sound from three rivers—the Connecticut, Housatonic, and Thames rivers (LISS 2017a). According to Wolfe et al. (1991), the Connecticut River is the largest source of freshwater to LIS, contributing 70 percent of the freshwater load.

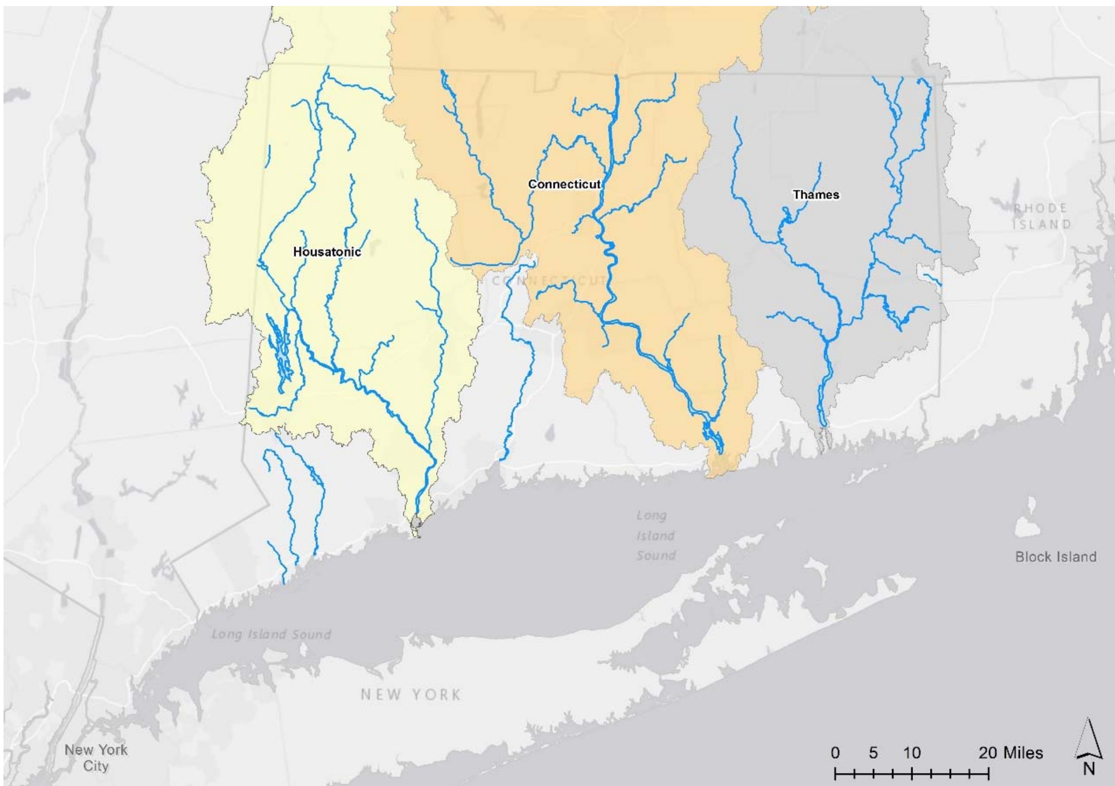

Figure 1. Map of LIS

The Sound has an average depth of 63 feet (LISS 2017a) and, according to Wolfe et al. (1991), a maximum depth of over 300 feet in the eastern area (LISS 2017a). According to Bricker et al. (2007), tidal amplitude ranges from less than 3.3 ft in the east to approximately 6.5 ft in the west and the Sound experiences only moderate flushing, with water having a mean residence time of 2–3 months. Riley (1967, cited in Gobler et al. 2006) documented a two-layered transport system within the LIS, where the fresher surface layer flows eastward toward the Atlantic Ocean and the deeper, more saline water flows westward. Salinity ranges from 23 parts per thousand (ppt) in the western end to 35 ppt in the eastern end (LISS 2017a).

The large LIS watershed encompasses 16,820 mi^2 and includes the states of Connecticut, Massachusetts, New Hampshire, New York, and Vermont (LISS 2017a). More than 23 million people live within 50 miles of LIS, and it provides an estimated $9.4 billion (2015 dollars) in value to the local economy (LISS 2017a).

The LIS is diverse ecologically. Over 120 finfish species live in the Sound and at least 50 species spawn there (LISS 2017a). LIS has shown evidence of seasonal hypoxia (defined as dissolved oxygen [DO] of less than 3 mg/L) since the 1970s (Parker and O'Reilly 1991). However, the severity of hypoxic events has moderated in the last 10 years (LISS 2015). Hypoxia is more severe in western LIS, near New York City, than in eastern LIS (CTDEEP 2015). Nitrogen pollution has been linked to hypoxia and other issues, such as loss of seagrass and growth of phytoplankton.

In 1985, EPA, along with the states of New York and Connecticut, formed the Long Island Sound Study (LISS), a partnership of state and federal agencies, user groups, concerned organizations, and individuals interested in restoring and protecting LIS (LISS 2017b). In 1994, the LISS developed the *Comprehensive Conservation and Management Plan* to protect and restore LIS (LISS 2017b). The Plan "identifies the specific commitments and recommendations for actions to improve water quality, protect habitat and living resources, educate and involve the public, improve the long-term understanding of how to manage the Sound, monitor progress, and redirect management efforts" (LISS 2017c). Throughout the years, the LISS has refined and made more specific its commitments and priorities through the *1996 Long Island Sound Agreement* and the *2003 Long Island Sound Agreement* (LISS 2017c).

The LISS has focused on understanding what is causing hypoxia and developing methods to control and manage nitrogen in LIS. In 1998, the LISS adopted *Phase III Actions for Hypoxia Management*. This plan identified sources and loads of nitrogen to LIS and recommended reduction targets for nitrogen. In 2000, the targets were incorporated into *A Total Maximum Daily Load Analysis to Achieve Water Quality Standards for Dissolved Oxygen in Long Island Sound*, also known as the LIS TMDL. This TMDL included a 58.5 percent nitrogen reduction to in-basin sources of enriched nitrogen (point sources: 60 percent; nonpoint sources: 10 percent). The LIS TMDL also identified actions and schedules for reducing nitrogen (NYSDEC/CTDEEP 2000).

In 2015, EPA developed *Evolving the Long Island Sound Nitrogen Reduction Strategy* to accelerate progress on achieving water quality standards and reducing nitrogen. As described in the Strategy, great progress has been made in reducing nutrient inputs to LIS. For example, by the end of 2015, upgrades to wastewater treatment facilities in Connecticut and New York have reduced annual nitrogen discharge by 41.8 million pounds, which is 99.8 percent of the LIS TMDL trade equalized wasteload allocation. However, the Strategy notes that EPA wants to conduct additional work to further reduce nitrogen in LIS and focus on additional adverse impacts to water quality from nutrients that the 2000 LIS TMDL does not address (e.g., loss of eelgrass). The Strategy is organized by three watershed groupings: embayments, large riverine systems, and priority western LIS point source discharges to open water. For each grouping, EPA is (1) developing nitrogen thresholds to translate the narrative water quality standard into a numeric target; (2) identifying where nitrogen watershed loading results in threshold exceedances; and (3) assessing options for load reductions from point and nonpoint sources needed to remain below thresholds. EPA will customize nitrogen loads for each watershed grouping and propose individual allocations for identified priority embayments/subwatersheds (LISS 2015).

The methodology for developing estimated load reductions and proposed allocations is described in greater detail in the memo, as described below.

- Section 2 describes the nitrogen loads to embayments and from tributaries, point source data, and water quality data gathered and processed for the analysis.
- Section 3 discusses how nitrogen thresholds are derived using stressor-response modeling. The section also includes a discussion of assessment endpoint analysis and selection.

Similar Large Scale Nutrient Reduction Efforts

Similar efforts are occurring for large coastal ecosystems in other countries. The Baltic Sea has been severely affected by eutrophication. As a result, several countries developed the HELCOM (Baltic Marine Environment Protection Commission—Helsinki Commission) Baltic Sea Action Plan (BSAP), which is a program to restore the ecological status of the Baltic marine environment by 2021. One of the program's four goals, eutrophication, seeks a "Baltic Sea unaffected by eutrophication." Specific goals include: clear water, natural levels of algal blooms, natural distribution and occurrence of plants and animals, and natural oxygen levels (HELCOM 2017). Download the original BSAP.

In 2014, the Baltic Marine Environment Protection Commission published *Eutrophication Status of the Baltic Sea 2007: A Concise Thematic Assessment*. This report provided a summary of progress made towards the goals and objectives of the BSAP. According to the report, progress is being made, but the Baltic Sea is still affected by eutrophication. Some waters were considered to be in good ecological status while others showed increasing nutrient levels. The report also cited the need for setting up a process to continually work on indicators and targets. For example, the expert group should regularly review agreed upon targets to incorporate new science and should also develop new core indicators where possible (Baltic Marine Environmental Protection Commission 2014).

Looking more closely at one country that is part of the HELCOM BSAP, Denmark has been incorporating mitigation measures for the past 25 years to reduce nutrients. Riemmann et al. (2016) looked at data and studied recovery of Danish coastal ecosystems after nutrient load reductions. They found that 25 years of mitigation measures have led to reduced nutrient inputs (50 percent or greater) to the system, which has led to reduced phytoplankton biomass, increased macroalgae cover in deeper waters, improved water clarity, and increased growth of eelgrass in deeper waters Riemmann et al. (2016).

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2. Database Development

Tetra Tech compiled and reviewed the data sources provided by EPA for this project. In addition, Tetra Tech conducted targeted literature searches for additional nitrogen loading data using searches on Google and Google Scholar for embayment name, major tributary name (section 2.3), and the community monitoring organizations described in Vaudrey et al. 2013. Descriptions of all of the data sources are provided in Table 1 through Table 4. Web links to the data sources are embedded as hyperlinks in the tables to make the text more readable.

Tetra Tech screened these literature and data sources for relevance to the study (i.e., for establishing nitrogen thresholds and allowable loads consistent with achieving desired water quality conditions and uses in embayments around LIS). Tetra Tech screened the data for representativeness using the following evaluation factors:

- Geographic scope: within the area of interest of the delineated LIS watershed.
- Temporal scope: 1/1/2006 to 12/31/2015 was the targeted period of interest. However, if data were limited within the period of interest or if limiting the scope to this time period would artificially truncate a dataset, the temporal scope was expanded. Data sources outside this temporal scope were not specifically targeted.
- Parameter scope: primarily nutrient (nitrogen and phosphorus species) and response (DO, chlorophyll *a*, Secchi depth, macroalgae, biotic indices) parameter data were compiled along with associated physical covariates (pH, temperature, salinity, turbidity). Data sources lacking nutrient or response data were not specifically targeted for inclusion.

Tetra Tech used data that were primarily available in a spreadsheet or relational database.
Tetra Tech also checked whether monitoring data were collected using appropriate QA/QC procedures (e.g., under an approved QAPP or standard operating procedures). In addition, Tetra Tech evaluated geospatial datasets for this project by mapping them to verify that the data fell within the correct waterbody.

As noted in Table 1 through Table 4, metadata and/or information on data quality were unavailable for several sources. If this information could not be easily located, Tetra Tech discussed with EPA the limitations in the use of these data. Data source reviews of EPA-provided data are in the same order below as subtasks A–D in the Performance Work Statement (PWS). The *Additional Sources Reviewed* sections are organized alphabetically first by those sources with data included in nitrogen load summaries and next by those sources reviewed but not currently being used in the qualitative analysis.

2.1 Nitrogen Loads

Tetra Tech extracted nitrogen loads, yields, and source contributions from the Vaudrey and Nature Conservancy datasets (see Review Notes in Table 1). These data were compiled by embayment into a spreadsheet. Units were standardized to kg/yr or kg/yr-km^2. Additional source data were considered, but were not on the same spatial or numeric scales as the data used in the compilation.

Table 1. Embayment Nitrogen Load Sources Reviewed

Source	***Review Notes***
EPA-provided Sources	
Vaudrey Research (Excel N Load Model attached as **appendix A from PWS**)	• EPA provided a Microsoft Excel nitrogen loading model developed by Dr. Jamie Vaudrey from the University of Connecticut. The model provides nitrogen loads and yields for 116 embayments to LIS using input data from 2010 to 2014. • Loads and yields are normalized by either area of open water embayment or watershed area. Nitrogen loads are allocated by source contributions (atmospheric deposition, fertilizer, sewer, combined sewer overflows [CSOs], septic systems and cesspools, and wastewater). Load scenarios are developed using different land use datasets (National Land Cover Dataset [NLCD] 2011 and Center for Land Use Education and Research [CLEAR] 2010 [a Connecticut statewide land use dataset]). • Tetra Tech contacted Dr. Vaudrey, who provided spatial coverage of the embayments as well as an explanation of the dataset. Dr. Vaudrey provided an updated Excel spreadsheet and corresponding final report in January 2017 (Vaudrey et al. 2016). Dr. Vaudrey explained that the CLEAR data are more accurate than the NLCD data except for three embayments (Pawcatuck River, East River, and Goldsmith's Inlet), for which the CLEAR data does not provide complete coverage.
The Nature Conservancy (TNC)	• EPA provided a link to a March 2016 TNC report describing the application of the Nitrogen Loading Model (NLM) to 13 embayments along the north shore of Nassau County and northwestern Suffolk County using input data from 2010 to 2015. • The model provides nitrogen loads and yields normalized by either area of open water embayment or watershed area and loads are broken down by source contributions (atmospheric deposition, fertilizer [lawns, recreation, agriculture], and wastewater [sewage treatment plants; septic systems and cesspools]).
LINAP	• EPA provided a Web URL to an overview page of the Long Island Nitrogen Action Plan (LINAP). • Work is underway with new loading estimates based on publicly agreed to process by early summer 2017. EPA and Tetra Tech have reached out to the LINAP group and set up a collaborative, technical transfer meeting. • Tetra Tech downloaded the action plan, Microsoft PowerPoint presentations, public comments, and project scope, and used them as background information.

Source	Review Notes
USGS	• EPA provided a general Web URL to USGS gauge historic monitoring data. • Tetra Tech reviewed available gauge data but did not find readily available nitrogen loading data within embayment watersheds that were not already represented in USGS reports (see section 2.3 for reviews of these reports).
Additional Sources Reviewed	
Fuss and O'Neill 2013	• A watershed plan was developed for the Rooster River using the Watershed Treatment Model (WTM) to calculate watershed pollutant loads based on both point and nonpoint sources using input data from 2000 to 2012. Input data to the model includes land cover/land use, CSOs, illicit discharges, septic systems, managed turf, and road sanding data. • Tetra Tech extracted modeled nitrogen load data by source type and by subwatershed; however, currently these data are used as background information as the spatial scale and methods were not similar enough to present with the Vaudrey and TNC data.
Fuss and O'Neill 2011	• A watershed and nitrogen reduction plan was developed for the Pequonnock River demonstrating use of the WTM to calculate existing nitrogen loads and pollutant load reductions. Model input data includes CSOs, illicit discharges, septic systems, and land cover. • Tetra Tech extracted existing nitrogen load data for the Pequonnock River watershed and 10 subwatersheds; however, currently these data are used as background information since the spatial scale and methods were not similar enough to present with the Vaudrey and TNC data.
Abdelrhman and Cicchetti 2012	• This journal article relates benthic activity to calculated nitrogen concentrations for 22 Northeast U.S. estuaries. Nitrogen loads are included for three embayments in LIS (Greenwich Cove, Niantic River, and Branford Harbor). • Reported nitrogen loads are from the 1990s and are outside the temporal scope of this project (derived from Latimer and Charpentier 2010—see section 2.3). This source was used for background information.

2.2. Point Source Dischargers

Tetra Tech summarized and compiled into a spreadsheet municipal and major industrial discharger loads, concentrations, and design flows from wastewater treatment plants (WWTPs) in the LIS watershed in Connecticut, Massachusetts, New Hampshire, New York, Rhode Island, and Vermont (Table 2). These data were either provided by EPA in several spreadsheets or summarized from discharge monitoring report data Tetra Tech queried from the Integrated Compliance Information System (ICIS). Point source discharge data were collected in accordance with EPA's NPDES regulations; specific requirements are listed below.

- NPDES permit monitoring requirements – 40 CFR 122.41(j), 122.44(i), and 122.48
- NPDES reporting requirements – 40 CFR 122.41(l)(4)(i) and 122.44(i)(2)
- NPDES recordkeeping requirements – 40 CFR 122.41(j)
- NPDES analytical methods requirements – 40 CFR 122.41(j), 122.44(i), and Part 136

Tetra Tech also summarized annual nitrogen loads from facilities in Connecticut from a USGS report (Mullaney and Schwarz 2013). See section 2.3 sources.

EPA provided contact information for Connecticut, New York, and Massachusetts for Tetra Tech to inquire if the states were aware of existing MS4 nitrogen loading estimates. State contacts did not identify existing nitrogen load estimates for Connecticut, Massachusetts, or Rhode Island. In addition, there are no MS4s within the LIS watershed in New Hampshire and Vermont.

New York State Department of Environmental Conservation (NYSDEC) provided a June 2015 report entitled *Determination of Regulated and Non-regulated Stormwater Loads to LIS from New York State* prepared under contract with the Long Island Sound Study (NYSDEC 2015). The purpose of the project was to assess the stormwater loads from the New York state portion of the LIS watershed, including quantifying the regulated stormwater load from the watershed load. NYSDEC established a methodology for determining the subwatershed loads, the load from the municipal boundaries, and the load from the actual MS4 systems. NYSDEC also completed estimates of the stormwater load from the sub-watershed, the municipal boundaries, and the MS4 systems for Port Jefferson Village and New York City. The DEC was not able to quantify the MS4 loads from all systems due to the lack of available data, specifically electronic availability of MS4 stormwater sewersheds and, for Nassau and Suffolk Counties where stormwater infiltration is a significant management practice within the MS4 system, data to be able to quantify the locations of these infiltration management practices within the MS4 system to remove that load from the surface water load. For Massachusetts, Tetra Tech estimated MS4 loads using a similar approach as NYSDEC that derives regional loading rates based on land use categories, only using the Opti-Tool regional loading rates. The Opti-Tool is a regional excel-based stormwater and nutrient BMP optimization tool developed by Tetra Tech and EPA Region 1[1].

The CT Department of Energy and Environmental Protection (CT DEEP) is in the process of beginning to develop load estimates using either (1) nitrogen concentration data collected for each town or (2) the WinSLAMM model. CT DEEP has collected stormwater monitoring data and provided these data to Tetra Tech; however, CT DEEP has not settled on an approach. Tetra Tech did not pursue further load estimations for Connecticut and Rhode Island at this stage. A small portion of the Pawcatuck embayment extends into Rhode Island, which overlaps with MS4s in Rhode Island. Due to the small overlap, Tetra Tech applied the same approach from Connecticut to Rhode Island.

Table 2. Discharger Flow, Nitrogen Load, and Concentration Sources Reviewed

Source	***Review Notes***
EPA-provided Sources	
Data compiled from NPDES ICIS database and states: **LIS CT-NY WWTP Summary.xlsx** (attached as **appendix B1 from PWS**)[1]	• EPA provided an Excel spreadsheet with CT and NY 2014 WWTP design flow, nitrogen load, and nitrogen concentration data for 109 facilities. • Tetra Tech summarized these data in a compiled discharger data table.

[1] https://cfpub.epa.gov/si/si_public_record_report.cfm?dirEntryId=289305

Source	***Review Notes***
LIS CT-NY 2015 WWTP Summary 11 28 16.xlsx[1]	• As an addition to the appendix B1 dataset, EPA provided an Excel spreadsheet with CT and NY 2015 WWTP design flow, nitrogen load, and nitrogen concentration data. • Tetra Tech summarized these data in a compiled discharger data table.
Data submitted by state to EPA for progress under TMDL: **LIS TE WLA Master File.xlsx** (attached as **appendix B2 from PWS**) [1]	• EPA provided an Excel spreadsheet with CT and NY WWTP nitrogen loads for the years 1994–2015. • EPA also provided an updated version of these data with 2015 updated loading (TE WLA 2015 Master File v2.xlsx). • The same 109 facilities were included as in the appendix B1 dataset. • Tetra Tech summarized these data in a compiled discharger data table.
Large, Direct Discharging Wastewater Treatment Facilities (a.k.a. Municipal Waste Water Treatment Plants [MWWTP]; Pollution Control Facilities [WPCF]) discharging to the open waters of LIS. These WPCFs are identified on the EPA Region 1 GIS map, along with other pertinent information, including per-MWWTP design flow and TMDL (lbs/day). (attached as **appendix B3 from PWS**) [1]	• EPA provided a PDF map of CT and NY MWWTPs. • Tetra Tech contacted the EPA Region 1 data contacts provided and received spatial data for most of the facilities included. • This source was used for background information to obtain a general understanding of dischargers in portions of the watershed.
LIS2014NH_VT_MAPointSourceNitrogenLoads.xlsx[1]	• EPA provided an Excel spreadsheet with MA, NH, and VT 2007, 2010, and 2014 WWTP design flow and average flow, nitrogen load, and nitrogen concentration data. • Tetra Tech summarized these data in a compiled discharger data table.
LISNitrogenfactsheetExhibitA.xls[1]	• EPA provided an Excel spreadsheet with MA, NH, and VT 2004–2005 WWTP design flow, nitrogen load, and nitrogen concentration data (dischargers sorted by the 3 major watersheds). • Tetra Tech summarized these data in a compiled discharger data table. • Tetra Tech only used the 2004–2005 data for facilities where no more recent data was available. The 2004–2005 data was carried forward for 15 MA facilities that did not have more recent loading data.

Source	Review Notes
MS4 sources	• EPA provided a general URL to EPA MS4 information. • EPA also provided contact information for CT, MA, and NY MS4 contacts. • Tetra Tech contacted each coordinator and will use state MS4 nitrogen loading data as available.
Industrial discharge information	• EPA provided a general URL for an ICIS/PCS search. • Tetra Tech conducted an ICIS search for dischargers and summarized annual loads from industrial facilities within the LIS watershed.
Additional Sources Reviewed	
No additional sources were reviewed directly to obtain discharger nitrogen loading.	

[1] Metadata and/or information on the quality of data from this source is currently unavailable.

2.3 Tributary Nitrogen Loads

Tetra Tech summarized and compiled nitrogen load data for each major tributary watershed (Connecticut, Thames, and Housatonic). These data came from EPA-provided sources and additional sources found during a literature review. Tetra Tech identified limited watershed nitrogen load estimates that occurred from 1988 to 2009, but focused on available data from 1999 forward to characterize the most recent watershed loading conditions available. Including a range of recent data rather than the most recent year helps account for significant interannual variability during especially wet or dry years. Sources summarized with major tributary watershed loads include:

- USGS annual nitrogen load estimates from 1999 to 2009 for nitrogen management zones that correspond closely with the Connecticut River watershed (Mullaney and Schwarz 2013)
- ArcView GIS Generalized Watershed Loading Function (AVGWLF) model data representing 1999–2005 (Evans 2008)
- NOAA National Estuarine Eutrophication Assessment (NEEA) representing 1994–2004 (Bricker et al. 2007)
- 2002 Northeastern and Mid-Atlantic regional SPARROW model (Moore et al. 2011)
- Load estimates from USGS reports including data from 1988 to 1998 (Mullaney et al. 2002)
- A Hydrological Simulation Program–Fortran (HSPF) deterministic model for Connecticut representing 1991–1995 (AQUA TERRA and HydroQual 2001)
- 1992–1993 New England SPARROW model (Moore et al. 2004)
- LIS TMDL nitrogen load estimates using input data from 1988 to 1990 (NYSDEC and CTDEEP 2000)

Tetra Tech also compiled annual nitrogen load (kg/yr) and yield (kg/yr-km^2) data from USGS reports (described in Table 3) calculated at specific USGS gauges throughout these watersheds.

Table 3. Tributary Nitrogen Loads Sources Reviewed

Source	***Review Notes***
EPA-provided Sources	
USGS data • Cited in Mullaney 2016a and at http://ct.water.usgs.gov/	• EPA provided a citation to a 2016 USGS report (Mullaney 2016a) with appendix data in Excel spreadsheets as well as a general URL to the New England Water Science Center, Connecticut Office. • Tetra Tech reviewed materials posted at the URL provided. Tetra Tech contacted John Mullaney on 11/10/2016 and obtained appropriate versions of his recent reports with nitrogen loading data (these additional sources are included in the *Additional Sources Reviewed* section of this table). • The Mullaney 2016a report *(USGS-SIR-2015-5189)* cited by EPA is a USGS report with annual nitrogen loads and yields from 1974 to 2013 using weighted regressions on time, discharge, and season (WRTDS) for 14 sites. • Tetra Tech summarized discharges, concentrations, yields, and loads by USGS site.
• USGS monitoring estimates for the CT River	• EPA provided a report that examines total nitrogen concentrations and loads at 13 sites in the upper CT River Basin between December 2002 and September 2005. • Data from the 13 sites in this report were carried forward to the USGS-SIR-2013-5171 report. Nitrogen load data are outside of the temporal scope of data collection efforts. This source was used for background information.
• USGS maintains two monitoring stations (Essex and Old Lyme) in the brackish waters of the lower CT River and LIS to collect water quality data, including temperature, salinity, specific conductance, pH, and dissolved oxygen	• EPA provided URLs to USGS gauge data for two stations. • Tetra Tech downloaded these data, but no nutrient data were available. This source was used for background information.
Systemwide Eutrophication Model (SWEM) outputs for nearshore waters and embayments	• EPA provided a general URL for the SWEM model. • Tetra Tech and EPA are pursuing output from the current managers of SWEM at UCONN (O' Donnell and McCardell). • Tetra Tech is also pursuing use of the existing NYHOPS hydrodynamic model (Georgas) and perhaps the UCONN-based hydrodynamic model ROMS (Whitney).

Source	*Review Notes*
NE SPARROW Model	• EPA provided a URL for the New England region-calibrated SPARROW model. • Tetra Tech accessed 1992–1993 New England hydrologic network GIS files and model prediction tables and selected data based on the SPARROW catchments within the LIS watershed. • Tetra Tech identified national and regional SPARROW models in addition to the New England model. These models included two National SPARROW models (1987 and 2002) and the Northeastern and Mid-Atlantic SPARROW model (2002), as well as other related models described in the associated USGS reports (Moore et al. 2004, 2011).
ArcView GIS Generalized Watershed Loading Function (AVGWLF) Model • AVGWLF Tool directly • Dr. Evans AVGWLF work	• EPA provided a series of URLs to an overview page about the AVGWLF ArcGIS add-on model and a 2008 report by Dr. Barry Evans (Evans 2008). • Tetra Tech downloaded the NY and New England Regional data (GIS data and weather data) and section 5-specific data (streams and raster data). These data were used as background information. • Tetra Tech reviewed the Evans 2008 report and summarized data. CT River watershed nitrogen loads were summarized at the watershed level. WWTPs lack coordinates and subbasins within the CT River watershed have nitrogen loading data available.
Additional Sources Reviewed	
NYSDEC/CTDEEP 2000 (Long Island Sound Nitrogen TMDL)	• 2000 LIS NYSDEC and CTDEEP TMDL estimated nitrogen loads for an average flow year based on point source monitoring data from 1988 to 1990 and nonpoint source estimation from surface land runoff, groundwater transport, CSOs, WWTP discharges, and atmospheric deposition from 1988 to 1989. • Tetra Tech extracted nitrogen load estimates delivered to LIS by major tributary watersheds. Estimates for the Housatonic and Thames river watersheds were significantly less than estimates derived from other methods.
Mullaney 2016b (USGS-OFR-2016-1007)	• This USGS report contains annual nitrogen loads and yields from 2006 to 2013 for 14 sites using the USGS load estimator, LOADEST. • Tetra Tech extracted and summarized loads and yields from this data by USGS site.

Source	*Review Notes*
Mullaney and Schwarz 2013 (USGS-SIR-2013-5171)	• This USGS report contains annual nitrogen loads and yields from 1999 to 2009 for 37 sites, 5 unmonitored sites, 26 areas, and 82 CT wastewater treatment facilities. Site loads and yields were calculated using LOADEST, while unmonitored sites and area loads and yields were calculated via regression. • Tetra Tech extracted and summarized data by USGS site, watershed, and discharger.
Moore et al. 2011	• This journal article describes the nutrient estimates generated using the Northeastern and Mid-Atlantic SPARROW model. • Table 4 includes comparisons by major river basins (including the CT River) for the 1988–1990 LIS TMDL, 2002 National SPARROW model, and 1994–2004 NEEA. • Tetra Tech summarized nitrogen load estimates by major basin.
Moore et al. 2004 (USGS-SIR-2004-5012)	• This USGS report describes the nutrient estimates generated using the New England SPARROW model. • Table 7 includes comparisons by major river basins (including all three major basins) for the 1987 National SPARROW model, the 1992–1995 LISS, 1991–1995 CT HSPF, and 1988–1998 USGS Mullaney estimates. • Tetra Tech summarized nitrogen load estimates by major basin.
Alexander et al. 2007	• This journal article used SPARROW to model headwater nitrogen contribution to stream networks in the Northeast United States. • No nitrogen load data were included in this article. This source was used for background information.
Coalition to Save Hempstead Harbor and Fuss and O'Neill 2015	• This is the 2015 annual report of the Coalition to Save Hempstead Harbor (CSHH) water monitoring program. • No nitrogen load data are included in this report. This source was used for background information. Ambient water quality data have been requested from CSHH directly.
Collins et al. 2013	• This research paper compares rates of primary production and nitrogen loading to develop an eutrophication index. • No nitrogen load data are included in this paper. This source was used for background information.

Source	*Review Notes*
Deacon et al. 2006 (USGS-SIR-2006-5144)	• This USGS report contains total nitrogen concentrations and loads calculated for 13 river sites in the upper CT River basin from 2002–2005. • Nitrogen load data are outside of the temporal scope of data collection efforts and not on the same spatial scale as the three major riverine watersheds. This source was used for background information.
Driscoll et al. 2003	• This journal article describes the evaluation of management strategies to reduce nitrogen inputs in the Northeast United States. • No nitrogen load data are included in this article. This source was used for background information.
Farley and Rangaranjan 2008	• This report describes the use of the AVGWLF model to estimate point and nonpoint sources of pollution to LIS. Annualized nitrogen load data reported prior to 2005. • Basin load estimates in this report are not at a comparable spatial scale for the CT River Basin ("out of basin" contributions from MA, NH, and VT not included). In addition, attenuation was not considered for basinwide estimates and nitrogen load data are outside of the temporal scope of data collection efforts. This source was used for background information.
Fuss and O'Neill 2015	• This is the 2013/2014 annual water quality report of Friends of the Bay (Oyster Bay). • No nitrogen load data are included in this report. This source was used for background information. Ambient water quality data have been requested from Friends of the Bay directly.
Georgas et al. 2009	• This journal article compares AVGWLF and HSPF models for estimation of total nitrogen nonpoint source loadings in CT and NY watersheds. • No nitrogen load data are included in this article. This source was used for background information.
HEP 2010	• Nutrient loading tables developed by the NJ and NY Harbor and Estuary Program (HEP) for drafting the 1996 Comprehensive Conservation and Management Plan. Includes total nitrogen loads for 1988–1989 and hydrodynamic conditions to NJ and NY estuaries and bay systems. • Nitrogen load data are outside of the geographic and temporal scope of data collection efforts. This source was used for background information.

Source	*Review Notes*
Latimer and Charpentier 2010	• This journal article describes a watershed loading model used to estimate 1990s total nitrogen loading rates to 74 estuaries in New England. Watershed scales included are at a much different spatial scale than the three major riverine watersheds of interest. • Nitrogen load data are assembled at a different spatial scale and are outside of the temporal scope of data collection efforts. This source was used for background information.
Latimer and Rego 2010	• This journal article describes eelgrass extent as a function of watershed-derived nitrogen loading for shallow estuaries in New England. Nitrogen loads are based on the Latimer and Charpentier 2010 paper and are generalized over 62 estuaries. • Nitrogen load data are assembled at a different spatial scale and are outside of the temporal scope of data collection efforts. This source was used for background information.
Lee and Lwiza 2008	• The journal article examines bottom DO variability in LIS, including evaluation of total nitrogen loading effects on bottom DO. • No nitrogen load data are included in this article. This source was used for background information.
Lloyd 2014	• This TNC and Peconic Estuary Program report describes the use of the Nitrogen Loading Model (NLM) to compare nitrogen loads from various sources to 43 subwatersheds to the Peconic Estuary. • Peconic Estuary is out of geographic scope of project. This source was used for background information.
Mullaney et al. 2002 (USGS-WRI-2002-4044)	• USGS report with total nitrogen loads estimated from monitoring data from 1988 to 1989 at 28 monitoring sites. • Nitrogen load data are outside of the temporal scope of data collection efforts (more recent data were available in later Mullaney reports). This source was used for background information.
Mullaney 2013 (USGS- SIR-2013-5008)	• This USGS report provides nutrient and *E. coli* loads for the Niantic River Estuary in 2005 and 2008–2011. • The report includes nitrogen load data for three USGS stations that are documented in the more recent USGS report, Mullaney 2016b. This source was used for background information.

Source	**_Review Notes_**
Scorca and Monti 2001 (USGS-WRI-00-4196)	• This USGS report contains calculated nitrogen loads discharged from four streams (Cold Spring Brook, Glen Cove Creek, Mill Neck Creek, and Nissequogue River) into LIS during 1985–1996. • Nitrogen load data are assembled at a different spatial scale and are outside of the temporal scope of data collection efforts. This source was used for background information.
Stearns and Wheler and CDM 2008	• This is a nitrogen reduction feasibility plan from selected MA POTWs prepared for MA DEP. • Tetra Tech plans to use EPA-provided nitrogen loads for facilities in MA (see section 2.2). This source was used for background information.
Smith et al. 2008	• This journal article contains measurements of in-stream nitrogen loss in reaches of the CT River during two studies in 2005. • Nitrogen loads were calculated at sampling sites in the CT River watershed for 2 days (kg/hr). It is not advisable to extrapolate load estimates from 2 days of data and compare them to other sources of loading data, which were computed on an annual basis. Thus, this source was used for background information.
Trench et al. 2012 (USGS-SIR-2011-5114)	• This USGS report contains nutrient loads and yields estimated for 47 USGS stations in the Northeast United States from 1975 to 2003. • Nitrogen load data are outside of the temporal scope of data collection efforts. More recent USGS gauge load data were available in other USGS reports. This source was used for background information.
Whitall et al. 2006	• This USDA Forest Service Proceedings document contains anthropogenic nitrogen inputs to estuaries, including LIS, modeled using the Watershed Assessment Tool for Evaluating Reduction Strategies for Nitrogen (WATERSN) model. • Includes nitrogen flux (kg N/ha/yr) for the entire LIS, not on a similar spatial scale as the three major riverine watersheds. The model is representative of conditions prior to 2005, outside of temporal and geographic scope of data collection efforts. This source was used for background information.
Wilson et al. 2008	• This journal article contains an examination of wind-induced effects on vertical mixing, stratification, and eutrophication in western LIS. • No nitrogen load data are included in this article. This source was used for background information.

Source	*Review Notes*
Yang et al. 2015	• This journal article describes a process-based model compared with LOADEST model results for anthropogenic effects on pollution flux in Northeast U.S. rivers. • The model is representative of conditions prior to 2005 and is scaled for the entire eastern United States, outside of the temporal and geographic scope of data collection efforts. This source was used for background information.
Zimmerman et al. 1995 (USGS-WRI-95-4203)	• This USGS report examines water quality data from 1972 to 1992 for the Connecticut, Housatonic, and Thames rivers. • No nitrogen load data are included in this report. Nitrogen concentration data are summarized from inland USGS gauges but are also outside of the temporal scope of data collection efforts. This source was used for background information.

2.4 Water Quality Data

Tetra Tech contacted the EPA-provided water quality monitoring organizations, local monitoring organizations with established QAPPs (Vaudrey et al. 2013), and other water quality monitoring organizations recommended by local stakeholders. Tetra Tech also queried the Water Quality Portal and identified water quality data from CT DEEP, the Interstate Environmental Commission (IEC), and the EPA Environmental Monitoring and Assessment (EMAP) program. Dataset time periods were 2006 through 2013 from CT DEEP and 2006 through 2010 from IEC. After discussion with data owners, Tetra Tech determined that the CT DEEP- and IEC-provided datasets would be more comprehensive as the primary data source than the data currently available in the Water Quality Portal. The EPA EMAP data loaded into the Water Quality Portal were the 2006 National Coastal Condition Assessment (NCCA) data, which were not available in full format on EPA's website. These data were compiled and organized with the 2010 NCCA data, which were available only on EPA's website. Water quality monitoring data that met project QAPP requirements were reviewed and formatted in a relational database. Water quality data sources reviewed are described in Table 4. Tetra Tech and EPA are working together to determine which data sources to include, based on availability, applicability, and cost. A final summary will be provided in the deliverable for Task D.

Table 4. Water Quality Data Sources Reviewed

Source	*Review Notes*
EPA-provided Sources	
CT Department of Energy and Environmental Protection (CT DEEP)	• EPA provided a URL to CT DEEP LIS water quality monitoring program maps. • EPA also provided contact information for CT DEEP water quality staff. • Tetra Tech contacted CT DEEP water quality staff and received physical, nutrient, and phytoplankton water quality data from 2006 to 2015 in relational databases and a series of Excel spreadsheets.
EPA National Coastal Condition Assessment (NCCA)	• EPA provided a URL to the overview page about EPA's NCCA. • Tetra Tech downloaded 2010 NCCA site information and benthos, fish, hydrographic profile, and water chemistry data. Data were selected for stations within LIS. • In addition, 2006 NCCA data were available only as a summary from EPA's website, so Tetra Tech downloaded 2006 NCCA data through the Water Quality Portal.
Long Island Sound Integrated Coastal Observing System (LISICOS)	• EPA provided a URL to the LISICOS map viewer page. • EPA also provided contact information for staff at LISICOS. • Tetra Tech contacted LISICOS staff, who are reviewing Tetra Tech's data request. Data were not provided in time to summarize in this memo.
Suffolk County	• EPA provided a URL to Suffolk County's marine water quality monitoring program website. • EPA also provided contact information for Suffolk County. • Tetra Tech contacted Suffolk County staff, who provided water quality data from 2006–2015 in one spreadsheet.
NY City Department of Environmental Protection (NYC DEP)	• EPA provided a URL to NYC DEP's annual harbor water quality reports. • EPA also provided contact information for NYC DEP. • Tetra Tech contacted NYC DEP staff, who provided water quality data for around 40 stations from 2006–2015 in a single geodatabase.

Source	***Review Notes***
Interstate Environmental Commission (IEC)	• EPA provided a URL to the IEC overview page. • EPA also provided contact information for IEC staff. • Tetra Tech contacted IEC staff, who have provided water quality data of interest from 2006 to 2015, including physical parameters and more limited nutrient and chlorophyll *a* data.
Dr. Gobler, Stony Brook University	• EPA provided a URL to the Long Island Water Quality Index website updated by Dr. Gobler. • EPA also provided contact information for Dr. Gobler. • Tetra Tech contacted Dr. Gobler, who planned to provide water quality data of interest, but data were not provided in time to summarize in this memo.
Systemwide Eutrophication Model (SWEM) model outputs for nearshore waters and embayments	• EPA provided a URL with the history of the Systemwide Eutrophication Model (SWEM). • Tetra Tech and EPA are pursuing output from the current managers of SWEM at UCONN (O' Donnell and McCardell). • Tetra Tech is also pursuing use of the existing NYHOPS hydrodynamic model (Georgas) and perhaps the UCONN-based hydrodynamic model ROMS (Whitney).

Source	*Review Notes*
Datasets from local watershed groups[2]	• EPA provided a citation for a Vaudrey et al. (2013) community survey of watershed monitoring groups (cited as Alonzo in footnote provided by EPA). • EPA also provided contact information for Peter Linderoth of Save the Sound, who interfaces with many community monitoring groups. • Tetra Tech identified 10 monitoring organizations with established QAPPs to contact by reviewing appendix B of the Vaudrey et al. (2013) community survey. Tetra Tech contacted Mr. Linderoth with this list and verified the list was accurate and not missing major monitoring organizations. Mr. Linderoth provided contact information and background. • Tetra Tech contacted the monitoring organizations as described below. In addition, in some cases, contacts referred Tetra Tech to other organizations that Tetra Tech contacted as described in the *Additional Sources Reviewed* section.
Clean Up Sound and Harbor (CUSH)	• Tetra Tech received historic CUSH water quality data of interest in a series of Excel spreadsheets. CUSH monitors for physical, nutrient, and response parameters in Stonington and Mystic harbors from 2008 to present. • Historic data from CUSH is not well organized; however, data were submitted to University of Rhode Island Watershed Watch (URI WW) and archived. Tetra Tech compiled the archived data from URI WW supplemented with other monitoring data from CUSH.
Coalition to Save Hempstead Harbor and the Hempstead Harbor Protection Committee	• Coalition to Save Hempstead Harbor provided monitoring data from 2004–2016 in annual spreadsheets with physical, DO, and SD, and limited chl *a* sampling in 2016. The Coalition coordinates monitoring for the Hempstead Harbor Protection Committee.
Friends of the Bay	• Tetra Tech received water quality monitoring data of interest from Friends of the Bay. • Friends of the Bay has monitored Oyster Bay and Cold Spring Harbor for over 13 years. Data were provided from 2006 to 2015 for physical, nutrient, and response parameters.

[2] Alonzo, J. et al. 2013. *Evaluation of Current Community-Based Monitoring Efforts and Recommendations for Developing a Cohesive Network of Support for Monitoring Long Island Sound Embayments*. Accessed January 2017. http://digitalcommons.uconn.edu/cgi/viewcontent.cgi?article=1001&context=marine_sci.

Source	*Review Notes*
Cedar Island Marina	• Cedar Island Marina plans to send Tetra Tech water quality monitoring data of interest, but data were not provided in time to summarize in this memo. Cedar Island Marine Research Lab has over 20 years of physical, DO, and benthic monitoring data for Clinton Harbor. It is possible that these data are focused on stormwater outfall monitoring and might not be of interest as an ambient water quality monitoring dataset.
The Harborwatch Water Quality Monitoring Program of Earthplace	• Harborwatch provided data from 2006–2015 at around 30 stations in a 6 tab spreadsheet. No nutrient monitoring data were included but DO, temperature, salinity, and Secchi depth were included.
The Maritime Aquarium at Norwalk	• The Maritime Aquarium provided cruise water quality monitoring data from 2006–2015. • The Maritime Aquarium has over 12 years of physical data for Norwalk Harbor.
Millstone Environmental Lab	• Tetra Tech received an annual water quality monitoring PDF report from the Millstone Environmental Lab. No tabular or relational data were available. The Millstone Environmental Lab has over 35 years of monitoring data for Niantic Bay; however, these data are primarily focused on studying the thermal impacts of the facility on the surrounding wildlife. These data will be used for background information.
Rocking the Boat	• Recent monitoring data from Rocking the Boat is stored with the Bronx River Alliance. Tetra Tech downloaded ambient data of interest for one station at the mouth of the Bronx River within the geographic scope of interest from the Bronx River Alliance data access portal. • Data do not include qualified, censored, or quality assurance results and, according to staff at both organizations, an accessible record of these data is not currently available. Tetra Tech does not plan to process these data further or include them in any analysis.
Bridgeport Regional Aquaculture Science and Technology Center	• Data are stored with the Maritime Aquarium at Norwalk. The Bridgeport Regional Aquaculture Science and Technology Center has over 19 years of monitoring data. These data were requested from the Maritime Aquarium at Norwalk.

Source	Review Notes
Additional Sources Reviewed	
EPA Office of Research and Development (ORD)	• EPA ORD provided data that includes compiled data from USEPA, UCONN researchers, and Cedar Island Marina Research Laboratory. • Some duplicative data were provided that Tetra Tech obtained from the primary source. Primary source data were used when available.
University of Rhode Island Watershed Watch (URI WW)	• URI WW works with and organizes community monitoring groups in RI and ensures that they operate under an established QAPP. • Tetra Tech received water quality data of interest from URI WW from CUSH, Save the Bay, and Watch Hill Conservancy.
NOAA (Hunts Point)	• NOAA provided monitoring data from a federal research project for one sampling season at Hunts Point in 2012.
University of Connecticut (Vaudrey)	• University of Connecticut monitoring data were provided from 2011–2014 embayment research by Dr. Jamie Vaudrey.
University of Connecticut (Yarish)	• University of Connecticut monitoring data were provided for three stations from research by Dr. Charlie Yarish.
Northport Harbor Water Quality Protection Committee	• This group does not perform water quality sampling and referred Tetra Tech to Dr. Gobler.
Oyster Bay Cold Spring Harbor Protection Committee	• This group does not perform water quality sampling; monitoring is performed by Friends of the Bay.
Setauket Harbor Taskforce	• This group plans to start water quality monitoring in the summer of 2017.
Manhassett Bay Protection Committee	• This group had no data of interest (primarily *E. coli* and fecal coliform monitoring). In addition, data were not collected under an established QAPP.

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3. Deriving Nitrogen Thresholds

3.1 Background on Developing Nitrogen Thresholds

An objective of this project is developing and updating nitrogen thresholds for LIS waterbodies to protect valued uses, including aquatic life and recreation. This will be accomplished using a multiple lines of evidence approach that relies on scientific literature, distributional statistics on nitrogen concentrations, and finally on empirical stressor-response models linking nitrogen (concentrations and/or loads) to response variables in the different waterbodies. The protective nitrogen thresholds will then be translated into nitrogen reduction targets.

The use of scientific literature refers to review and extraction of nitrogen targets developed to protect similar resources in estuarine and coastal settings comparable to LIS. The use of thresholds from scientific literature for developing targets for use in a variety of applications similar to this one has been applied by EPA several times and substantial literature exists on the effects of nitrogen on estuaries (USEPA 2000, 2001, 2010c, 2012, 2015).

The distributional statistics line of evidence refers to evaluating distributions of nitrogen concentrations in different waterbodies and using those values to inform protective nitrogen thresholds. EPA has used the distribution of nutrient concentrations from reference waterbodies for setting nutrient thresholds for a number of applications, including TMDLs and permitting (USEPA 1999, 2001, 2015, 2016). The same concept can be extended to distributions from other nutrient concentration populations as well, including those from time periods known to be supporting uses (i.e., temporal reference) (USEPA 2010a) and from populations known to be supporting their designated uses (e.g., USEPA 2015). In this way, identifying distributions of nitrogen concentrations or loads for embayments known to be in attaining condition can provide a line of evidence for those values that protect valued uses and, thus, provide a line of evidence for developing nitrogen thresholds.

Stressor-response relationship modeling refers to the application of empirical statistical modeling to nutrient and response data to identify either (1) thresholds of response that represent unacceptable changes or (2) concentrations consistent with protection of an already existing, known desired reference condition (e.g., DO standard, light levels necessary for seagrass growth). EPA advocates the use of, and have themselves used, the stressor-response relationship approach for establishing relationships and their impacts to waterbodies (USEPA 1999, 2000, 2007, 2013). EPA has also developed national guidance on the approach (USEPA 2010b).

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3.2 Stressor-Response Modeling

Literature-based and distributional-based analyses are well established approaches and their application widely described in existing documentation (e.g., USEPA 1999, 2001, 2012, 2015). Here we provide greater detail on stressor-response modeling, which will be used as an additional line of evidence in support of literature based and distributional nitrogen thresholds. Empirical stressor-response modeling is used to estimate a relationship between nitrogen concentrations and a response measure that is either a direct assessment endpoint or relatable to assessment endpoints indicative of use protection (e.g., an existing water quality criterion such as DO, biological indicator values used to measure aquatic life use support, or chlorophyll *a* concentrations known to affect clarity or recreation) (USEPA 2010). Then, nitrogen and phosphorus concentrations that are protective of designated uses can be derived from the estimated relationship (USEPA 2010).

For the LIS project, empirical stressor-response relationship modeling will be applied, as data allow, as a line of evidence to support derivation of nitrogen thresholds using two general approaches: linear/nonlinear modeling with interpolation and use of change point analysis.

Stressor-response approaches encompass a suite of analytical techniques for exploring and identifying thresholds in the relationships between response variables and nutrient loads concentrations. Typical response variables for this context include water chemical aquatic life use indicators (e.g., DO, pH), algal biomass and/or algal assemblage metrics (e.g., chlorophyll, percent nutrient sensitive diatoms, percent cyanobacteria), harmful algal indicators (e.g., cyanotoxin concentration, percent nuisance taxa), and aquatic life use indicators or biocriteria indicators (e.g., seagrass light levels, trophic state indices, algal multimetric indices or individual metrics scores, invertebrate multimetric indices or individual metrics). The value of these indicators is their direct linkage to designated uses. They, therefore, provide a way to connect nutrient loads or concentrations to use protection. Any stressor-response relationship that is modeled will be based on a sound and defensible conceptual model of the linkage between nutrients and response endpoints in the LIS ecosystem.

Distributional statistics, spearman correlation analysis, and visual plotting are first used to explore and identify potential relationships between nutrient stressors and response variables. Locally weighted scatterplot smoothing (LOWESS) is frequently used to explore stressor-response relationships. The LOWESS technique is a useful tool for modeling nonlinear relationships (Cleveland 1979). LOWESS fits simple models to localized subsets of the data to construct a function that describes, essentially, the central tendency of the relationship. LOWESS fits segments of the data to the model. LOWESS output will include scatterplots with LOWESS curves as well as a description of the bandwidth used and any weighting function applied. Relationships of interest are selected and used for guiding subsequent stressor-response analyses. As mentioned, a variety of analytical techniques exist, but two approaches are generally applied: regression analysis and change point analysis to identify thresholds in response variables to increasing nutrient concentrations.

For regression modeling, traditional simple linear regression approaches and nonlinear models (e.g., generalized additive models) will be applied to paired stressor and response data. When existing desired conditions for response variables are known (e.g., chlorophyll *a* concentrations that protect light levels for seagrass growth), nitrogen loads or concentrations consistent with support of that target response condition will be modeled from the stressor-response relationship (e.g., interpolated), nitrogen thresholds will be identified, and uncertainty associated with those thresholds will be estimated.

Hierarchical regression models might be used if applicable. For example, if there is a large disparity in sample density among embayments, then hierarchical models can be used to build a global embayment model that is weighted for individual embayments using embayment-specific data. Similarly, hierarchical models might be used if there are major drivers influencing response among waterbodies (e.g., residence time, morphology).

There may be responses for which known response target conditions do not exist. In these cases, an option would be to identify changes in the condition of the response with increasing nutrients. Known as change points, such conditions can reflect ecosystem state shifts that are frequently adverse. Such change points could be used to identify concentrations or loads of concern. For change point analysis, nonparametric deviance reduction is used to identify thresholds (King and Richardson 2003; Qian et al. 2003). This technique is similar to regression tree models, which are used to generate predictive models of responses for one or more predictors. The change point in this application is the first split of a tree model with a single predictor variable (nutrient concentration).

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3.3 Assessment Endpoint Targets

3.3.1 Review of Scientific Basis and Selection of Potential Assessment Endpoints and Indicator Variables

To determine appropriate assessment endpoints for LIS[3], a literature review was conducted to evaluate biological, chemical, and physical aspects of the endpoints so that designated uses would be protected. True assessment endpoints are or reflect valued ecosystem characteristics that are to be protected. Designated uses and their associated narrative criteria can be considered assessment endpoints. These assessment endpoints (e.g., shellfish propagation and harvesting) are often difficult to predict or measure directly. Therefore, simpler endpoints (referred to as *indicators* or *measures*) are usually evaluated. These simpler endpoints are measurable and predictable, and serve as surrogate measures to link stressors and outcomes. The simpler endpoints should have clear linkages to both nutrients and the ecosystem characteristics to be protected.

Measures include measures of effects, defined as “measurable changes in an attribute of an assessment endpoint or its surrogate in response to a stressor to which it is exposed”; measures of exposure, defined as “measures of stressor existence and movement in the environment and their contact or co-occurrence with the assessment endpoint”; and measures of ecosystem and receptor characteristics (USEPA 1998). The Watershed Approach literature tends to refer to these measures as *indicators*.

A target is simply a value of a measure that is consistent with attaining the assessment endpoint or management objective. In other words, a target is equivalent to a value for protecting a specific use at a given site. For this effort, the target will be the numeric value of an indicator variable that supports a balanced natural population of aquatic flora and fauna in LIS.

LIS is split in half between New York and Connecticut and thus follows the water quality standards of both states. Narrative water quality standards language related to nutrients and aquatic life is provided below.

[3] Assessment endpoints are ecosystem characteristics that are to be protected.

Relevant Narrative Water Quality Standards for LIS

New York

Nutrients: None in amounts that will result in growths of algae, weeds and slimes that will impair the waters for their best usages (6 NYCRR 703.2).

Aquatic Life: These waters shall be suitable for fish, shellfish and wildlife propagation and survival (6 NYCRR 701.10, 701.11, 701.12, and 701.13).

Source: https://www.epa.gov/sites/production/files/2014-12/documents/nywqs-section1.pdf

Connecticut

Nutrients: The loading of nutrients, principally phosphorus and nitrogen, to any surface water body shall not exceed that which supports maintenance or attainment of designated uses (Section 22a-426-9).

Aquatic Life: It is the state's goal to restore or maintain the chemical, physical and biological integrity of surface waters. Where attainable, the level of water quality that provides for the protection and propagation of fish, shellfish, and wildlife and recreation in and on the water shall be achieved (Section 22a-426-4).

Biological integrity (i.e., biological condition) is further defined as: Sustainable, diverse biological communities of indigenous taxa shall be present. Moderate changes, from natural conditions, in the structure of the biological communities, and minimal changes in ecosystem function may be evident; however, water quality shall be sufficient to sustain a biological condition within the range of Connecticut Biological Condition Gradient Tiers 1-4 as assessed along a 6 tier stressor gradient of Biological Condition Gradient (See section 22a-426-5 of the Regulations of Connecticut State Agencies) (Section 22a-426-9).

Source: https://www.epa.gov/sites/production/files/2014-12/documents/ctwqs.pdf

Designated uses in LIS include primary and secondary contact recreation, fish propagation and survival, wildlife habitat, and shellfish harvesting for human consumption. Specific designated uses for LIS are described in the callout box below. Taking the designated uses into consideration, Tetra Tech examined studies that looked at responses to nutrients of seagrass, macroalgae, DO, phytoplankton, harmful algal blooms (HABs), and oysters. Tetra Tech will select endpoints that are present in the waters/embayments being evaluated, clearly link changes in nutrients to measured changes in endpoint values, and are currently being measured.

Based on information gathered during the literature review, Tetra Tech assessed each endpoint using several major factors, including how ecologically important they are to LIS, how sensitive they are to nutrient inputs, how directly related they are to designated uses, and what their potential for recovery is.

Designated Uses in LIS

New York—Open waters of the East River and LIS; approximately 700 square miles.

- Class SA: The best usages are shellfishing for market purposes, primary and secondary contact recreation and fishing. These waters shall be suitable for fish, shellfish and wildlife propagation and survival. Chronic DO shall not be less than a daily average of 4.8 mg/L (may fall below 4.8 mg/L for a limited number of days, as defined in the formula in the water quality standards); acute DO shall not be less than 3.0 mg/L at any time.
- Class SB: The best usages are primary and secondary contact recreation and fishing. These waters shall be suitable for fish, shellfish and wildlife propagation and survival. Chronic DO shall not be less than a daily average of 4.8 mg/L (may fall below 4.8 mg/L for a limited number of days, as defined in the formula in the water quality standards); acute DO shall not be less than 3.0 mg/L at any time.
- Class SC: The best usage is fishing. These waters shall be suitable for fish, shellfish and wildlife propagation and survival. The water quality shall be suitable for primary and secondary contact recreation, although other factors may limit the use for these purposes. Chronic DO shall not be less than a daily average of 4.8 mg/L (may fall below 4.8 mg/L for a limited number of days, as defined in the formula in the water quality standards); acute DO shall not be less than 3.0 mg/L at any time.
- Class I: The best usages are secondary contact recreation and fishing. These waters shall be suitable for fish, shellfish, and wildlife propagation and survival. In addition, the water quality shall be suitable for primary contact recreation, although other factors may limit the use for this purpose. DO shall not be less than 4.0 mg/L at any time.

Connecticut—Connecticut portion of the LIS; approximately 613 square miles.

- Class SA: Habitat for marine fish, other aquatic life and wildlife; shellfish harvesting for direct human consumption; recreation; industrial water supply; and navigation. Chronic DO not less than 4.8 mg/L with cumulative periods of DO in the 3.0–4.8 mg/L range as detailed in footnote in Table 1 of the water quality standards; acute DO not less than 3.0 mg/L at any time.
- Class SB: Habitat for marine fish, other aquatic life and wildlife; commercial shellfish harvesting; recreation; industrial water supply; and navigation. Chronic DO not less than 4.8 mg/L with cumulative periods of DO in the 3.0–4.8 mg/L range as detailed in footnote in Table 1 of the water quality standards; acute DO not less than 3.0 mg/L at any time.

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3.3.2 Selecting Assessment Endpoints and Water Quality Indicator Variables

Selecting assessment endpoints to protect the designated uses in LIS represents a balance among environmental sensitivity to nutrient pollution and available data. To develop estimated reduction levels and allocations, it is important to select assessment endpoints that are sensitive to nitrogen/phosphorus pollution, so that one can infer that the reductions and allocations will protect less sensitive receptors from such pollution. Additionally, it is important to choose endpoints with sufficient data to allow quantitative thresholds to be developed through stressor-response relationship modeling (e.g., empirical or regression models).

There are numerous endpoints that can, at a minimum, be qualitatively related to nutrient enrichment (Bricker et al. 2008). Tetra Tech searched scientific databases (e.g., Google Scholar and state research and agency reports) to investigate the current science on responses of assessment endpoints to nutrients and other likely stressors in LIS. The assessment endpoints examined included seagrass, macroalgae, DO, phytoplankton, HABs, and oysters. Table 5 provides a brief summary table of key factors considered in determining the most appropriate endpoint(s) for LIS. Additional information about each assessment endpoint analyzed is available following Table 5.

Table 5. Assessment Endpoints for Evaluating the Magnitude and Effects of Nutrients, including Advantages and Disadvantages

Endpoint	Importance	Linkages to, or Effects of, Nutrients	Advantages	Disadvantages
Seagrass	• Valuable marine habitat • Primary food source for many organisms	• Spatial extent, density, and growth rates decline with decreased light transmittance • Light requirement usually 20–25% surface irradiance • Light transmittance decreases with decreased clarity in part due to excess phytoplankton or epiphytic biomass from increased nutrients	• Mechanism of nutrient impact mostly well-understood • Colonization depth (Z_c) useful indicator • Once Z_c goal is established, can use light requirements to infer water clarity requirement and water column chlorophyll *a* criteria • Historical depth of colonization could be used to infer reference water clarity	• Cofactors exist: salinity stress, food web change, dredging, propeller scarring, sediment loading, disease • Response to nutrients can be slow (especially recovery) • Affected by fetch and water temperature

Endpoint	Importance	Linkages to, or Effects of, Nutrients	Advantages	Disadvantages
Macroalgae	• Can displace native eelgrass • Can shade or displace native benthic algae • Provides habitat and food source for some benthic macroinvertebrates and fish, but varies in palatability and utility as food source versus natural sources	• Eutrophication of a system often leads to excessive growth of macroalgae • Excessive macroalgae often causes die-off of other aquatic plants/eelgrass	• Clear linkages to eutrophication • Can handle decreased water quality better than other aquatic plants (i.e., eelgrass) • Relatively easy to identify and quantify species growth and transfer from eelgrass/seagrass to macroalgae habitat • Can use δ15N of macroalgae to verify source of nutrients (e.g., through wastewater or fertilizer)	• Growth is affected by many factors: nutrients, light transmittance, N:P ratio, herbivory, depth, temperature
DO	• Hypoxia kills fish and invertebrates • Hypoxic or low DO areas nullified as suitable habitat	• Nutrients affect organic loading through algal growth, depleting oxygen • Nutrients accelerate decomposition rates by microbial stimulation, consuming oxygen	• Existing criteria • Well-established basis for protection of aquatic life • Clear linkages to nutrient enrichment • Extensive database of monitored conditions • Collected LIS DO data come from multiple datasets (10 different reporting organizations) between 2000 and 2015, and represent primary, secondary, open water, and other embayments; DO data total 32,704 data points across 511 monitoring stations in LIS	• DO can be affected by many other factors than just nutrient inputs

Endpoint	Importance	Linkages to, or Effects of, Nutrients	Advantages	Disadvantages
Phytoplankton	• Primary producers and important component of marine food web • Excess growth affects clarity, DO, habitat, aesthetics	• Nutrients are key limiting factors for growth rate	• Responsive to nutrients, well-established basis for use as indicator • Biomass data in estuarine waters are routinely monitored and data are generally abundant • Satellite-derived chlorophyll data readily available for many coastal waters • Collected chlorophyll a data in LIS include data from 34 surrounding embayments between 2000 and 2015, including chlorophyll a data from 180 sites and corrected chlorophyll a data from 188 sites in primary, secondary, and other embayments, and in open water (21 sites have both chlorophyll a and corrected chlorophyll a data)	• Other factors can interfere with evaluating stressor-response relationships • Species composition data limited; differences in field sample and taxonomic methods may increase uncertainty • Most estuaries lack species composition models developed for nutrient response
HABs	• HABs can impact human health and marine organisms • Often associated with toxins leading to faunal kills, shellfish contamination, economic impacts, decline in aesthetic value, and environmental and ecological damage	• Nutrients are key limiting factors for algal growth rate • Nutrient pollution can lead to more severe blooms that occur more often	• Foul odor and reduced aesthetics can lead to public awareness • Responsive to excess nutrients	• Other factors can interfere with evaluating stressor-response relationships • Not enough data on HAB occurrence in the Sound

Endpoint	Importance	Linkages to, or Effects of, Nutrients	Advantages	Disadvantages
Oysters	• Food source for the public • Ability to filter water • Oyster beds can serve as a home for small fish • Historic and economic significance	• Low levels of oxygen associated with nutrient pollution can impact oyster survival	• Ease of determining oyster population	• No direct linkages found • Changes in oyster catch linked to a number of factors other than nutrients

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3.3.2.1 Seagrass

A. Background/Introduction

Submerged aquatic vegetation (SAV) is a term used to describe rooted, vascular plants that grow completely underwater except for periods of brief exposure at low tides. SAV can grow in marine, estuarine, and freshwater environments. SAV that grow in ocean environments are also known generally as *seagrasses* (NOAA 2012). Seagrasses provide habitat for many fish, birds, and invertebrates, and they are a source of food for some aquatic organisms. Additionally, seagrasses stabilize sediments at the bottom of waterbodies and play an important part in carbon and nutrient cycles. Water quality benefits associated with seagrasses include increased water clarity, as seagrasses stabilize sediment, filter polluted runoff, and absorb nutrients (NOAA 2012).

Both Connecticut and New York have adopted provisions related to seagrasses, including eelgrass (*Zostera* spp.). Through the Coastal Management Act (1980), Connecticut recognizes eelgrass as a coastal resource "…to protect, enhance, and allow for the natural restoration of eelgrass flats…" (CGS 22a-92(c)(2)(A)) (Latimer et al. 2014). As a result, CTDEEP, which administers the Act, considers eelgrass protection when reviewing and approving activities such as construction and dredging of piers and docks (Latimer et al. 2014). New York is working to protect seagrass through their Seagrass, Research, Monitoring, and Restoration Task Force, created in 2006 (Latimer et al. 2014). The Task Force is charged with "developing recommendations on elements of a seagrass management plan with the goal of preserving, restoring, and mapping the native seagrass populations on Long Island" (Latimer et al. 2014). Additionally, in 2010, the New York State Senate and Assembly passed the Seagrass Protection Act to protect seagrass in state waters (Latimer et al. 2014). Some of the provisions include restrictions on (1) fishing gear used in seagrass meadows, (2) use of harmful chemicals and pesticides near seagrass, and (3) fertilizer application during certain times of the year (Latimer et al. 2014). The 2010 act was never signed by the governor because of concerns over included language, but it was later revised and signed in 2012 (Latimer et al. 2014). The new law includes similar provisions, such as restricting mechanical fishing gear, but it omits seasonal restrictions that were included in the 2010 act on the use of phosphorus-containing fertilizer in coastal communities (Seagrass Protection Act of 2010; Seagrass Protection Act of 2012; Stolorow 2012).

Both Connecticut and New York have reported that seagrasses, particularly the eelgrass *Zostera marina*, provide protection, spawning and nursery areas, and food sources for a number of species with commercial, recreational, and ecological importance such as the winter flounder, bay scallop, and hard clam (New York State Seagrasses Task Force 2009; State of Connecticut 2007). Constanza et al. (2014) valued seagrasses at $28,916 ha/yr globally in 2011, which is an increase from $26,226 ha/yr, when it was assessed in 1997. Losses of seagrasses or fragmentation of seagrass beds can result in losses of aquatic organisms that rely on seagrasses for habitat or food (McCloskey and Unsworth 2015; Reed and Hovel 2006).

The presence and distribution of seagrass is dependent on a number of factors, including light penetration, nutrients, substrate, temperature, current velocity, wave energy, and salinity (LISS 2004), with light as the main factor limiting seagrass growth and distribution (Cornell 2012; Koch and Beer 1996; UCONN 2010; Vaudrey 2008a). It is important to note, however, that these factors are interrelated. As an example, Koch and Beer (1996) attributed the disappearance of *Z. marina* in the western end of LIS to eutrophication-induced declines in water quality, accentuated by large tidal ranges (3 m) that affected light availability, compared to a smaller range (1 m) in the eastern portion. Researchers have pointed to the importance of anthropogenic influences on SAV health (Blake et al. 2014; Burkholder et al. 2007; Orth et al. 2006). In many aquatic systems such as LIS, SAV has been used as an indicator of overall water quality and ecosystem health.

In LIS, eelgrass is the predominant species of SAV growing along the coast, although widgeon grass (*Ruppia maritima*) has been present in some brackish waters and 17 other species of SAV have been identified in the tidal waters of the Connecticut River (Latimer et al. 2014; LISS 2004; New York State Seagrasses Task Force 2009). Eelgrass has also been the most widely studied SAV in LIS.

Globally, water quality degradation and nutrient enrichment have played a role in the decline of SAV (Burkholder et al. 2007). A 2009 global assessment reported that between 1980 and 2006, seagrasses were lost at an average rate of 110 km^2 per year and that 29 percent of the known areal extent of seagrass had disappeared since seagrass was first recorded in 1879 (Waycott et al. 2009). The authors also reported that seagrass rates of decline (median) increased from 0.9 percent per year before 1940 to 7 percent per year between 1990 and 2006 (Waycott et al. 2009). As in the global study, there has been a historic decline of seagrasses in LIS. An analysis of historical information indicates that although eelgrasses were abundant prior to the early 1900s, eelgrass is only currently found in eastern LIS. In the 1930s, eelgrass was nearly wiped out by a fungal disease, with an estimated loss of 90 percent of eelgrass along the Atlantic coast (LISS 2013). As reported by the State of Connecticut (2007) and Latimer et al. (2014), populations in eastern LIS were reported to have recovered by the 1950s and continued to thrive into the 1970s, while populations in western LIS did not recover (Knight and Lawton 1974; McGill 1974, both cited in State of Connecticut 2007; Latimer et al. 2014). According to Latimer et al. 2014, “since the 1950s, eelgrass populations along the Connecticut coast have suffered additional losses thought to be linked to the effect of N loading on the coastal ecosystem.” In the 1990s, there have been reports of declines in seagrass abundance in coves and embayments. This more modern eelgrass decline has been attributed to the effects of anthropogenic activity, particularly nitrogen pollution from sewage discharges and stormwater runoff (LISS 2017; NOAA 2012). Latimer et al. (2014) cites anthropogenic nitrogen loading from watersheds as the most detrimental factor affecting seagrass.

The U.S. Fish and Wildlife Service’s National Wetlands Inventory Program (NWI) has conducted eelgrass inventories in the eastern end of LIS since 2002 (LISS 2017). These inventories have provided a baseline of seagrass extent (estimated through aerial photography) and density (visually estimated via boat or underwater camera) and have also shown some increase in abundance overall, from approximately 1,600 acres in 2002 to 2,061 acres in 2012 (Tiner et al. 2013). See Table 6 for more detailed information about the different surveys conducted, specific waterbodies studied, and acreage changes observed throughout the study years.

Table 6. Differences in Eelgrass Survey Results from 2002 to 2012 (Tiner et al. 2013)

Waterbody	2002–2006 Acreage Change	2006–2009 Acreage Change	2009–2012 Acreage Change	2002–2012 Acreage Change
Mystic Harbor[1]	+61.9	+20.9	+3.1	+85.9
Niantic Bay[1]	+130.2	-57.0	+31.7	+104.9
Stonington Harbor[1]	+28.0	-15.1	+2.0	+14.9
Duck Island Roads	+5.3	-0-	+1.3	+6.6
Poquonnock River[2]	-2.9	-3.1	+1.1	-4.9
Little Narragansett Bay	-2.8	+60.0	-15.9	+41.3
Quiambog Cove	+70.7	-20.6	-19.1	+31.0
Palmer-West Cove	+0.1	-12.2	+0.3	-11.8
Mumford Cove	-11.0	+7.0	+8.9	+4.9
New London Harbor	+3.9	-0.9	+8.2	+11.2
Goshen Cove	-4.9	-27.7	-3.2	-35.8
Jordan Cove	-6.5	+1.7	+5.6	+0.8
Rocky Neck State Park	+7.7	-7.7	+2.1	+2.1
Connecticut River Area	-0-	+2.1	[4]	+2.1
Fishers Island, NY	+7.8	+22.5[3]	+56.9	+87.2
North Shore, NY	+9.2	-14.4	+0.3	-4.9
Plum Island, NY	+9.5	-1.9	-0.1	+7.5
Total	**+306.2**	**-46.4**	**+83.2**	**+343.0**

Notes:
[1] Priority embayments from the Task Order.
[2] Fixed spelling from original study (Paquonock) to correct spelling (Poquonnock).
[3] Two large beds totaling 122.1 acres on the south side of Fishers Island could be seen on the 2009 imagery, but they were not visible on 2006 imagery due to environmental conditions. Field inspections in 2006 had located robust beds in this area and recorded their occurrence as points since the beds could not be accurately delineated on the imagery. Consequently, for the 2009 report, this acreage was not treated as a gain because robust beds were noted in this area in 2006 and their boundaries could not be established.
[4] Not assessed due to lack of field review; could not be verified on imagery as either a loss or gain, since site was only recorded in past through field observation.

In the *Long Island Sound Comprehensive Conservation and Management Plan 2015*, the LISS lists a goal of restoring and maintaining an additional 2,000 acres of eelgrass by 2035. This is in addition to the 2012 baseline of 2,061 acres (LISS 2015). According to a GIS-based eelgrass habitat suitability index model funded by the New England Interstate Water Pollution Control Commission and the LISS, approximately 161,000 acres in LIS are within the depth range appropriate for eelgrass (Vaudrey et al. 2013).

B. Sensitivity to Nutrients

Nutrient pollution, particularly nitrogen, has been shown to affect seagrass health by (1) stimulating primary producers that compete with seagrass for available light and (2) exacerbating the effects of other factors such as turbidity (Vaudrey 2008a). Studies have highlighted the importance of water clarity (i.e., light availability) on the health of seagrass communities and the relationship with nutrient enrichment (Dennison 1987; Duarte 1991; Lee et al. 2007). Vaudrey (2008a) states that light availability is the main factor controlling *Z. marina* growth and distribution in LIS, while nutrients and temperature are secondary factors affecting it.

Increases in nutrient concentrations play a substantial role in the decline of seagrass, but the effect is indirect via the effect on the growth of primary producers, which affects the availability of light in the water column. For instance, increases in nutrients might increase epiphytes, which grow on seagrasses (Dennison et al. 1993), or there might be an increase in phytoplankton production, which causes a decrease in light availability (and an increase in light attenuation, or K_d). Vaudrey (2008a) states that if nitrogen is limiting, it can directly affect *Z. marina* by stimulating productivity. When nutrient concentrations increase in a waterbody, there can also be an increase in other primary producers, thus leading to increased turbidity in the water column. Phytoplankton, epiphytes, and macroalgae are similar because they prevent light from reaching the seagrass (shading), which decreases the ability of seagrass to grow.

A locally relevant example in Mumford Cove, Connecticut illustrates how seagrass is affected by nitrogen loading. According to Vaudrey et al. (2010), *Z. marina* was absent from Mumford Cove during the time that Groton WWTP discharged to the cove. In 1987, discharge from the WWTP was relocated after pressure from local residents to address permit violations and water quality issues (LISRC 2010). Within 5 years of the WWTP diversion, green macroalgae *Ulva lactuca* biomass and area cover was drastically reduced, which allowed *Z. marina* and *Ruppia maritima* to recolonize the southern half of Mumford Cove (Vaudrey et al. 2010). Areal coverage of seagrass continued to increase in the cove, increasing by 50 percent by 2002 (15 years after the discharge relocation) (LISRC 2010). Additionally, by 2002 *Z. marina* was considered to be the dominant seagrass in the cove (LISRC 2010; Vaudrey et al. 2010) and was found as "patches or moderately dense beds over an area approximately 24 ha" (Vaudrey et al. 2010).

A nitrogen loading study conducted in the Waquoit Bay system in Massachusetts found that *Z. marina* is particularly sensitive to eutrophication, even at low loading rates, and that light limitation resulting from nutrient-limiting primary producers affected the growth and health of seagrasses (Hauxwell et al. 2003). Some other factors that add to light attenuation are color and suspended sediments. The decay of organic matter also consumes oxygen, which deprives seagrass from the oxygen needed to survive during periods of respiration.

There are published literature reviews that summarize *Z. marina* responses to nutrient inputs (Burkholder et al. 2007; McLaughlin and Sutula 2007; Vaudrey 2008a). Some other specific studies that are geographically relevant include the following:

Latimer et al. (2014) reported that minimum light requirements for east coast seagrass range from 15 to 35 percent of surface irradiance. This corresponds with LIS- and Massachusetts-specific studies (Dennison and Alberte 1985; Koch and Beer 1996; Moore 1991 all cited by Latimer et al. 2014), which report values within this range. Latimer at al. (2014) also calculated the maximum depth limit for seagrass in LIS using the Lambert-Beer equation, a K_d of 0.7 m^{-1}, and a minimum light requirement of 22 percent of surface irradiance, which resulted in a maximum depth limit calculation of 2.16 m (Latimer et al. 2014). Latimer et al. (2014) further analyzed maximum depths by using K_d values reported by Koch and Beer (1996) as typical for eastern LIS (K_d = 0.5 m^{-1}) and western LIS (K_d = 1.0 m^{-1}). Resulting calculations by Latimer et al. (2014) indicated a maximum depth limit for seagrass in eastern LIS of 3.1 m (using a K_d of 0.5 m^{-1}) and 1.5 m for western LIS (using a K_d of 1.0 m^{-1}).

Z. marina has a tolerance for each stressor. Yarish et al. (2006) and Vaudrey (2008a) recommended limits of *Z. marina* habitat parameters and compared them to the same factors derived from past research in the Chesapeake Bay conducted by Batiuk et al. (2000) (see Table 7). Vaudrey (2008a) states that, similar to the Chesapeake Bay, LIS would "encounter similar problems with the use of K_d as a management criterion," which is that the light extinction coefficient cannot be adjusted to "accommodate different tidal ranges or restoration depths."

Table 7. Comparison of Recommended Habitat Requirements for Growth and Survival of *Z. marina* (adapted from Vaudrey 2008a)

Habitat Requirements	Recommended Guidelines for Chesapeake Bay reported in Batiuk et al. (2000)	Recommended Guidelines for LIS reported in Yarish et al. (2006)	Recommended Guidelines for LIS reported in Vaudrey (2008b)	Guideline Type
Minimum Light Requirement at the Leaf Surface (%)	> 15		> 15	Primary requirement (must estimate epiphyte biomass)
Water Column Light Requirement (%)	> 22		> 22	Substitute for minimum light requirement at the leaf surface
K_d (1/m)	< 1.5	< 0.7	< 0.7	Provided for reference, use minimum light as the standard
Chlorophyll *a* (µg/L)	< 15	< 5.5	< 5.5	Secondary requirement (diagnostic tool)
Dissolved Inorganic Nitrogen (mg/L)	< 0.15 (mesohaline and polyhaline)	< 0.05[1]	< 0.03	Secondary requirement (diagnostic tool)
Dissolved Inorganic Phosphorus (mg/L)	< 0.02 (tidal fresh, oligohaline, and polyhaline); < 0.01 (mesohaline)	< 0.02[2]	< 0.02	Secondary requirement (diagnostic tool)
Total Suspended Solids (mg/L)	< 15	< 30		Secondary requirement (diagnostic tool)
Sediment Organics (%)	0.4–12	< 3	10	Habitat constraint
Vertical Distribution (m)	$Z_{max} = 0.5m + Z_{min}$		$Z_{max} = 1m + Z_{min}$	Habitat constraint
Sediment Sulfide Concentration (µM)	< 1000			Habitat constraint
Current Velocity (cm/s)	10 < X < 100			Habitat constraint

[1] Yarish et al. (2006) provided dissolved inorganic nitrogen values in µM, which was converted to mg/L. The dissolved inorganic value for Yarish et al. (2006), which Vaudrey (2008a) cited, is based only on NO_3 and not all three dissolved inorganic nitrogen species.
[2] Yarish et al. (2006) provided dissolved inorganic phosphorus values in µM, which was converted to mg/L.

Similarly, Howes et al. (2003) compared nitrogen thresholds and water quality classifications based on site-specific biological and chemical indicators, including eelgrass, that were developed independently for three Cape Cod embayments (see Table 8). Based on water quality classifications in the state of Massachusetts, total nitrogen concentrations associated with levels of eelgrass health fit in the following qualitative and quantitative classifications (Howes et al. 2003).

- *Excellent*: total nitrogen concentrations below 0.30 mg/L (corresponds to supporting dense eelgrass beds).
- *Excellent/good*: total nitrogen concentrations 0.30–0.39 mg/L (corresponds to eelgrass being present).
- *Good/fair*: total nitrogen concentrations 0.39–0.50 mg/L (corresponds to eelgrass not being present).
- *Moderate impairment*: total nitrogen concentrations 0.50–0.70 mg/L (corresponds to unsustainable conditions to support eelgrass).
- *Significant impairment*: total nitrogen concentrations 0.70–0.80 mg/L (corresponds to eelgrass being absent).
- *Severe degradation*: total nitrogen concentrations over 0.80 mg/L (corresponds to eelgrass being absent).

Table 8. Nitrogen Thresholds and Coastal Water Classifications for Great, Green, and Bournes Ponds in the Town of Falmouth (Howes et al. 2003)

Classification of Nitrogen-based Water Quality	Trophic Classification	Source[1]		
		Howes et al. (2003) Citing School for Marine Science and Technology (SMAST)[2]	Howes et al. (2003) Citing Cape Cod Commission (Eichner et al. 1998)	Howes et al. (2003) Citing Buzzards Bay Project/Massachusetts Coastal Zone Management (Costa et al. [1992] and in press)
		Total Nitrogen (mg/L)	Total Nitrogen (mg/L)	Total Nitrogen (mg/L)
Excellent	Oligotrophic	< 0.30	ND	ND
Excellent/Good	Oligo to Mesotrophic	0.30–0.39	< 0.34	< 0.39
Good/Fair	Mesotrophic	0.39–0.50	0.34–0.39	0.39–0.44
Moderate Impairment	Mesotrophic to Eutrophic	0.50–0.70	ND	ND
Significant Impairment	Eutrophic	0.70–0.80	ND	ND
Severe Degradation	Hyper-Eutrophic	> 0.80	ND	ND

Note:
ND = not determined
Values are long-term (> 3 year) average mid-ebb tide concentrations of total nitrogen (mg/L) in the water column.
[1] Howes et al. (2003) did not provide full citations for SMAST, Eichner et al. (1998), or Costa et al. (1992) so these citations were not included in Sources Cited at the end of this section.

[2] The nitrogen values presented were developed as part of the Ashumet Valley Plume Nitrogen Management Project for the Town of Falmouth and the Air Force Center on Environmental Excellence by MEP Tech Team members B.L. Howes and J.R. Ramsey. These values are preliminary and need refinement by the MEP. Note that classification is by sampling location not full estuary, since each system shows a nitrogen gradient from headwaters to inlet.

The decline of *Z. marina* in Waquoit Bay, Massachusetts, was attributed to nitrogen loading pollution (Hauxwell et al. 2003). Significant seagrass loss of 80–96 percent bed area lost in 10 years was found at loads of approximately 30 kg N/ha/yr, and near-total disappearance at loads greater than or equal to 60 kg N/ha/yr (Hauxwell et al. 2003).

Bowen and Valiela (2001) found diminished eelgrass due to increases in phytoplankton and macroalgal biomass in northeastern U.S. estuaries. The authors used the Waquoit Bay Nitrogen Loading Model to determine changes in watershed nitrogen loading to Waquoit Bay since the 1930s. Areal cover of eelgrass was sharply reduced at nitrogen loads > 20 kg/ha/yr; meadows disappeared completely by the time nitrogen loads exceeded 100 kg/ha/yr (Bowen and Valiela 2001). In Cape Cod waters, the authors found that nitrogen loads 15–30 kg/ha/yr corresponded to near-complete destruction of eelgrass meadows (Bowen and Valiela 2001).

Bintz et al. (2003) conducted a mesocosm experiment in Narragansett Bay and found that negative effects of increased nutrient inputs (nutrient treatments included additions of 6 mmol N/m^2/d and 0.5 mmol P/m^2/d) were exacerbated by increasing the water temperatures by 4 degrees Celsius. The researchers found that nutrient treatments (along with the warm water temperatures) decreased eelgrass density and belowground-production, as well as increasing the time-interval between the initiation of new leaves (Bintz et al. 2003).

In a study that evaluated *Z. marina*, water quality parameters, bottom light, light attenuation, and percent surface transmittance at 70 sites in 19 Massachusetts estuaries, the authors found that healthy seagrass sites had the lowest concentrations of total nitrogen (0.42 mg/L) and total chlorophyll *a* (5.1 µg/L) (Benson et al. 2013). Sites with healthy seagrass had tidally averaged total nitrogen concentrations of less than 0.34 mg/L and ebb-tide total nitrogen of less than 0.37 mg/L, and eelgrass survival required bottom light of greater than or equal to 100 µE/m^2/s. They also found that there was a positive relationship between total nitrogen and total chlorophyll *a* concentrations (Benson et al. 2013). Healthy beds had the highest percent light penetration (23.7 percent), while degraded/declining sites had percentages approximately 21.0 percent or less (Benson et al. 2013). The authors also found that percent eelgrass transplant survival was related to total nitrogen concentrations (i.e., the lower the nitrogen concentration, the higher the transplant success). Sites with greater than 75 percent transplant success had an average total nitrogen concentration of 0.39 mg/L (Benson et al. 2013).

Table 9 shows the relationship between seagrass transplant survival and total nitrogen concentrations during the study (2007–2009 and 2011) (Benson et al. 2013). The authors concluded that using *Z. marina* alone to set nutrient targets “will result in the most restrictive and inclusive target concentration since eelgrass is such a sensitive indicator of water quality.” However, they went on to state that the combination of nitrogen and bottom light would provide a more “robust” approach since total nitrogen does not account for water depth (Benson et al. 2013).

Table 9. Relationship between *Z. marina* Transplant Survival and Total Nitrogen Concentrations (Benson et al. 2013)

Z. marina Transplant Survival (%)	Total Nitrogen (mg/L)
< 25	0.68 ± 0.11
25–50	0.67 ± 0.11
50–75	0.49 ± 0.12
> 75	0.39 ± 0.03

The Massachusetts Estuaries Project (MEP) has been working on developing a linked watershed/estuary model to determine nitrogen thresholds for 89 estuaries in southeastern Massachusetts to protect the health of each estuary using indicators, including eelgrass. Based on available linked watershed/estuary models available for 33 estuaries, the total nitrogen target concentrations to restore eelgrass habitat range from 0.34 mg/L to 0.55 mg/L (MEP n.d.).

Wazniak et al. (2007) analyzed existing monitoring data on water quality and seagrasses in coastal bays of Maryland and Virginia to determine trends in eutrophication. The authors determined biologically relevant threshold values for nutrients and chlorophyll *a* in the Maryland coastal bays (Wazniak et al. 2007). Based on monitoring data collected between 2001 and 2003 in the Maryland coastal bays, researchers determined that in order to maintain seagrass health, total nitrogen and total phosphorus concentrations should be 0.65 mg/L and 0.037 mg/L, respectively (Wazniak et al. 2004). Thresholds for *eutrophic* conditions were set at 1.0 mg/L and 0.1 mg/L for total nitrogen and total phosphorus, respectively (Wazniak et al. 2004).

Burkholder et al. (2007) noted that considering nutrient loads rather than water column nutrient concentrations might be important in nutrient-enriched conditions (Burkholder et al. 2007). Often, plants tend to rapidly take up nutrients or sediments adsorb nutrients in the early stages of nutrient-enrichment (Burkholder et al. 2007; Suttle and Harrison 1988; Suttle et al. 1990), and the rate at which nutrients are recycled in the water column might play an important role in the concentration of nutrients present in the water column (e.g., nutrients are taken up by seagrasses more quickly in the spring/summer than in the fall/winter when there are fewer leaves) (Burkholder et al. 2007; Howarth 1988).

Studies have identified impacts on seagrass health due to increasing waterbody nutrient concentrations, but seagrasses also can impact the overall water column nutrient concentrations. Vaudrey (2008a) indicated that nitrogen assimilation by *Z. marina* is more important than denitrification for removal of nitrogen and that, in spring and early summer, *Z. marina* beds act as a nitrogen sink and release nutrients to the environment in the fall and winter periods when leaves die.

C. Strengths and Weaknesses of Using This Endpoint

Seagrasses are considered a “robust ecological indicator” because of their susceptibility to human disturbances and in addition to the many examples above, the European Water Framework Directive also uses seagrasses as an indicator of eutrophication (Borja et al. 2012). They are a valuable marine habitat and often considered “foundational or keystone species” in aquatic systems as mentioned earlier in this section (Hughes et al. 2009). Changes in seagrass characteristics are often symptoms of changes in the environment, as they are sensitive to nutrient inputs (e.g., Hauxwell et al. 2003). They are also considered a long-term indicator, since they are rooted and cannot move (Sutula 2011).

The advantage to using seagrass as an endpoint is that the mechanisms of nutrient impacts are mostly well-understood—the spatial extent, density, and growth rates decline with decreased light availability and light availability is affected by epiphytic and phytoplanktonic algal biomass. The amount of light needed for seagrass is site-specific and needs to be tailored to the location, but these light requirements can be derived using current and historical colonization depths.

However, some disadvantages to using seagrass as an endpoint include the fact that seagrass response to nutrients is indirect and thus might not be immediate. Light availability can also be influenced by a variety of factors, including colored dissolved organic matter and inorganic sediment. Other factors also play a part in the health of seagrasses, including bed sediment quality, waves, tides, salinity, food web changes, dredging, channeling, and scarring. The overall interaction of factors contribute variability to the stressor-response relationship. The review also identified this endpoint's susceptibility to local embayment interaction/effects.

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3.3.2.2 Macroalgae

A. Background/Introduction

Macroalgae are large aquatic multicellular algae that can be seen with the naked eye (macroscopic) and occur in many colors (e.g., green, red, brown, blue) and forms (e.g., tall, mat-like). Macroalgae are eukaryotic, photosynthesizing organisms (Schmidt 2003a). Green (Chlorophyta), Red (Rhodophyta), and Brown (Phaeophyta) are the most common groups of macroalgae (Hauter and Hauter 2010). Green algae, whose coloring comes predominantly from chlorophyll, include many unicellular species, but some species are multicellular/colonial including the common sea lettuce, *Ulva lactuca*. Chlorophytes are common in freshwater but also inhabit marine bays and estuaries (Schmidt 2003b). Red algae contain a pigment called phycobilins and inhabit shallow marine water. They also help to create coral reefs from calcium carbonate (Schmidt 2003d). Brown algae contain pigments like fucoxanthin and are most common near rocky coastlines. The most common brown algae are fucoids and kelps, the latter of which grow very large and form offshore masses, also known as *kelp forests* (Schmidt 2003c).

Nutrient enrichment resulting in hypoxia and a shift from eelgrass- to macroalgae-dominated systems has been a problem for many coastal estuaries. Experts believe that this shift from slower growing vascular macrophytes to faster growing macroalgae and phytoplankton is an indication of eutrophication (McGinty et al. 2004). Waquoit Bay, Narragansett Bay, and LIS are just a few examples of systems that have been adversely affected (Bowen and Valiela 2001; Deacutis 2007; Fox et al. 2008; Keser et al. 2003; Lyons et al. 1995). The shift from healthy eelgrass beds to macroalgae can be measured simply by noting the change in species composition or by a measurement of area (e.g., 74 percent of the bottom in Mumford Cove was *Ulva lactuca* under nutrient enriched conditions) (Vaudrey et al. 2010). However, with nutrient management programs, there is some evidence that this shift can be reversed and eelgrass can recover (Vaudrey et al. 2010).

B. Sensitivity to Nutrients

Anthropogenic activities that result in nutrient discharges add to the growth of macroalgal blooms and eutrophication (LaMontagne et al. 2002; Lapointe 1997). Macroalgal blooms can outgrow or kill seagrasses by altering the competitive balance from slow-growing to fast-growing primary producers (Collado-Vides et al. 2007; Hauxwell et al. 2001; McGlathery 2001; Taylor et al. 1995).

Macroalgal blooms can attract grazing organisms and alter community structure directly by changing competition among algal species for nutrients (Howarth et al. 2000; McGlathery 1995). Macroalgal blooms can destroy the habitats of indigenous species, where the macroalgae use up oxygen and alter biogeochemical cycles (ECOHAB 1995). This leads to hypoxia/anoxia, resulting in decreased biological diversity (Lapointe 1997; NRC 2000). It can also change benthic habitat structure, further affecting herbivore control of macroalgal biomass (Fox et al. 2012).

Many macroalgal bloom studies have been undertaken in relation to nutrient inputs and eelgrass decline:

- Nahant Bay, Massachusetts: Blooms of *Pilayella littoralis* were observed, resulting from high ammonia levels in the 1980s (Pregnall and Miller 1988).

- Waquoit Bay, Massachusetts: Following an increase in nutrients in the late 1980s and early 1990s, *Cladophora* and *Gracilaria* blooms formed, replacing *Zostera* (Peckol et al. 1994; Valiela et al. 1992).

- Waquoit Bay, Massachusetts: Demonstrated that increasing canopy heights of macroalgae reduced *Zostera* density, recruitment, growth, and production rates (Hauxwell et al. 2001).

- LIS near Millstone Point, Connecticut: Short-term fouling and overgrowth of eelgrass and blue mussels resulted from a *Cladophora* bloom, presumably related to nutrient enrichment (Keser et al. 2003).

- Mumford Cove in LIS, Connecticut: Fifteen years after cessation of nutrient enrichment, this cove recovered from 40 years of point source anthropogenic nutrient input, returning from an *Ulva*-dominated system to a *Zostera*-dominated one (Vaudrey et al. 2010).

There is limited evidence to provide a linkage to nitrogen/phosphorus pollution loadings (i.e., point and nonpoint sources) and the development of macroalgal HABs (Lapointe et al. 2005). For example, McClelland and Valiela (1998) and Bowen and Valiela (2001) used stable N isotope ratios (δ^{15} N; ‰) of macroalgae as evidence that nutrient inputs from inland waters are the primary source of nitrogen to macroalgal blooms.

Three indirect methods have mainly been used to examine whether nitrogen or phosphorus or both limit macroalgal productivity: (1) examining the N:P ratios of dissolved inorganic nutrients in the water column, (2) examining N:P ratios of the algal tissue, and (3) conducting N and P enrichment experiments on macroalgae. Each method has limitations such as not knowing at what point one should sample the water column (nutrient increases can be in pulses or in a steady stream) and the different uptake and storage capacities of nutrients in each species of macroalgae (Fong et al. 2003). Examples of relevant studies examining macroalgal productivity are provided below:

- *Gracilaria edulis* was exposed to nutrient pulses, which resulted in increased tissue nitrogen, chlorophyll *a*, and amino acids (Costanzo et al. 2000).

- Fong et al. (1993) conducted a microcosm nutrient enhancement experiment in shallow coastal lagoons on different algal groups (green macrophytes, phytoplankton, and benthic cyanobacterial mats) under five N:P treatments. They found that nitrogen directly controlled macroalgal biomass and, when the nitrogen supply exceeded macroalgal demand, nitrogen was then available to the other algal groups.

- LaMontagne et al. (2002) found that decreased denitrification due to increasing macroalgal cover in Childs River, Cape Cod, Massachusetts, could create a positive feedback loop as decreasing denitrification would increase nitrogen availability and could, therefore, increase macroalgal cover.

- Lee and Olsen (1985) found that nitrogen, rather than phosphorus, controlled the growth of green algae, specifically *Enteromorpha* spp. and *Ulva lactuca* in Rhode Island salt ponds. They also found that nitrogen-enriched environments allowed for green algae to outcompete widgeon grass (*Ruppia maritima*), a favored food source for waterfowl and potentially preferred benthic habitat for young flounder.

The role herbivory plays in the dynamics of macroalgae is unsettled. In some areas, grazers may control macroalgal blooms. However, if the grazers are overwhelmed by blooms or become prey for other benthic invertebrates, they will not provide sufficient control (Fox et al. 2012). Morgan et al. (2003) studied the relative influence of grazing and nutrient supply on growth of *Ulva lactuca* in estuaries of Waquoit Bay and found that bottom-up effects (enrichment) can overwhelm top-down forces (consumption) in nutrient-enriched estuaries.

Studies have shown that light intensity (irradiance), temperature, water depth, presence of grazers, water movement, and desiccation can also all affect macroalgal growth (Dring 1981; Fox et al. 2008; Lapointe 1987; Mann 1973; Nielsen et al. 2002; Valiela et al. 1997).

Light and nutrients are critical factors that control macroalgal growth and productivity. When light is plentiful, nutrients become the limiting factor (Lapointe and Tenore 1981), and when nutrients are present in abundance, irradiance plays a more important role in determining productivity (Lapointe and O'Connell 1989). Light is usually limiting during the winter and early spring, while nutrients limit growth during the summer. However, macroalgae can undergo biological changes to maximize photosynthesis and growth by obtaining nutrients from the water column or sediments (Krause-Jensen et al. 1996). Macroalgae have the ability to optimize productivity under different irradiance and nutrient limitations (both N- and P-) (Lapointe 1997). A study conducted by Lapointe and Tenore (1981) examined the effects of nitrogen additions and different light conditions on the macroalga *Ulva fasciata*. They found that nitrogen additions enhanced growth under high light and that chlorophyll content increased more with increased nitrogen loading under both high and low light (Lapointe and Tenore 1981). Macroalgal blooms are known to terminate when phytoplankton prevents light from reaching the macroalgae, a condition that could be more common in areas with longer water residence times (Valiela et al. 1997). Fox et al. (2008) measured macrophyte biomass monthly for 6 years in three estuaries subject to different nitrogen loads. While watershed land use largely influenced seasonal and interannual differences in standing stock, irradiance could have also been a secondary limiting factor controlling biomass in the higher loaded estuaries by limiting the depth of the macroalgal canopy (Fox et al. 2008).

Studies conducted in Waquoit Bay and LIS found similar thresholds for total nitrogen loads (Hauxwell et al. 2003; Latimer and Rego 2010). Hauxwell et al. (2003) found substantial eelgrass loss (and shift to a macroalgal-dominated community) in Waquoit Bay at loads of ~30 kg N/ha/yr, with total eelgrass disappearance at loads ≥ 60 kg N/ha/yr. In their analysis of 62 watershed-estuary systems in New England, Latimer and Rego (2010) found that nitrogen loading rates greater than 50 kg N/ha/yr would decrease the ability of eelgrass to thrive, while eelgrass was absent at loading rates of over 100 kg N/ha/yr. However, these empirical relationships developed for small coastal watersheds may not be valid for larger river systems (LISS 2015).

Howes et al. (2003) compared nitrogen thresholds and water quality classifications based on site-specific biological and chemical indicators, that were developed independently, by three different entities for three Cape Cod embayments (see Table 8 in seagrass section). Based on water quality classifications in the state of Massachusetts, total nitrogen and macroalgae presence fit in the following qualitative and quantitative classifications (Howes et al. 2003):

- *Excellent*: total nitrogen concentrations below 0.30 mg/L (corresponds to macroalgae generally not being present).
- *Excellent/good*: total nitrogen concentrations 0.30–0.39 mg/L (corresponds to macroalgae generally not present but might be present).
- *Good/fair*: total nitrogen concentrations 0.39–0.50 mg/L (corresponds to macroalgae not present or present in limited amounts, even though a good healthy aquatic community still exists).
- *Moderate impairment*: total nitrogen concentrations 0.50–0.70 mg/L (corresponds to macroalgal accumulations occurring in some regions of the embayments).
- *Significant impairment*: total nitrogen concentrations 0.70–0.80 mg/L (corresponds to macroalgal accumulations being present).
- *Severe degradation*: total nitrogen concentrations over 0.80 mg/L (corresponds to large and pervasive macroalgal accumulations).

The LISS monitoring program has a multidecadal time series on total nitrogen concentrations in LIS (LISS 2015). These data can be used in conjunction with the literature to create models of the system that could derive an LIS-specific threshold (LISS 2015). Table 10 provides information from various studies about macroalgae relevant to developing numeric thresholds.

Table 10. Information on Macroalgae Relevant to Developing Numeric Thresholds

Characteristic	Study Location and Response	Citation
Decline in eelgrass	LIS, CT Short term declines were directly associated with fouling and overgrowth (2 occasions), once by blue mussels and once by a bloom of green algae (*Cladophora* spp.).	Keser et al. 2003
Eelgrass recovery after nutrient enrichment reversal	Mumford Cove, CT WWTP discharged into cove for over 40 years, allowing *Ulva lactuca* to cover 74% of bottom. After WWTP diversion, *Zostera marina* was present within 5 years and firmly established (> 50% of cove) within 15 years.	Vaudrey et al. 2010
Nitrogen or phosphorus limitation	Narragansett Bay, RI *Ulva lactuca, Gracilaria tikvahiae, Cladophora sp.* For both types of systems, the extent of limitation will likely depend on the loading of both nitrogen and phosphorus, not just one or the other.	Taylor et al. 1995
Macroalgal biomass	Greenwich & Narragansett Bays, RI Dense patches of *Ulva lactuca* and *Gracilaria takvahiae* peaking in the summer months, corresponding to periods of hypoxia that seem to be related to the tides.	Deacutis 2007
Macroalgal biomass	Coastal Lagoons, RI Nitrogen input increased growth of *Enteromorpha* spp. and *Ulva lactuca*, outcompeting widgeon grass (*Ruppia maritima*).	Lee and Olsen 1985
Macroalgal blooms	Waquoit Bay (MA): Childs River, Quashnet River, Sage Lot Pond *Cladophopra vagabunda, Gracilaria tikvahiae, Zostera marina* Higher land-derived nitrogen loads lead to more algal biomass and larger seasonal differences and a consistently larger crop of biomass during the seasonal low. Study suggests that while peak macroalgal accumulation may be driven by increased nitrogen supply, the canopy is eventually limited by light availability.	Fox et al. 2008
Mitigating macroalgal blooms	Waquoit Bay (MA): Childs River, Quashnet River, Sage Lot Pond *Cladophora vagabunda, Gracilaria tikvahiae, Zostera marina* Macrophytes were eaten by various grazers: shrimp, amphipods, crabs, isopods, worms, snails, and fish. Response of macroalgae to nutrient and herbivore controls may be species-specific—responses of three species studies were different. There is difficulty in explaining the controls on the blooms in bottom-up or top-down terms because the systems are more complex.	Fox et al. 2012
Effect of nitrogen supply and grazing on algal growth (*Ulva lactuca*)	Different nitrogen loads in estuaries of Waquoit Bay (MA): Sage Lot Pond (SLP) = 14 kg/ha/yr Quashnet River (QR) = 350 kg/ha/yr Childs River (CR) = 601 kg/ha/yr Mean percent net growth of *U. lactuca* fronds: 69.2 ± 4.4 (SLP) 104.8 ± 7.5 (QR) 251.9 ± 18.2 (CR)	Morgan et al. 2003

Characteristic	Study Location and Response	Citation
Macroalgal biomass	Waquoit Bay, MA $\delta^{15}N$ of groundwater nitrate increase from - 0.9‰ to +14.9‰ as wastewater nitrogen increases from 4 to 86% of the total N pool; resulting in receiving waters average $\delta^{15}N$ of DIN increasing from +0.5‰ to +9.5‰, correlated with increases in $\delta^{15}N$ of eelgrass, macroalgae, cordgrass, and suspended particulate matter. • Magnitude of change in $\delta^{15}N$ of eelgrass is similar to the change in $\delta^{15}N$ of DIN in groundwater among estuaries. • Magnitude of change in $\delta^{15}N$ of cordgrass, macroalgae, and suspended particulate organic matter is approximately one third the size of the change in eelgrass $\delta^{15}N$ among estuaries.	McClelland and Valiela 1998
Macroalgal blooms	Waquoit Bay, MA Decreased denitrification due to increasing macroalgal blooms could create a positive feedback loop where increasing cover decreases sediment denitrification and increases nutrient availability.	LaMontagne et al. 2002
Increases in macroalgal biomass and phytoplankton	Benthic macroalgae and phytoplankton biomass in Waquoit Bay increased when receiving increasingly higher nitrogen loads.	Bowen and Valiela 2001

C. Strengths and Weaknesses of Using This Endpoint

Macroalgae have a known and distinct response to nutrient enrichment and are considered a common response variable where nutrient enrichment occurs. As a result, macroalgal growth (and associated eelgrass decline) can be used to support recommended thresholds for nitrogen loads. Macroalgae, as their name implies, are relatively easy to see and identify, especially the shift from eelgrass to macroalgae. As a result, this is not only noticeable to the public, but facilitates assessment.

Macroalgae respond to a variety of factors. In addition to nutrients, light, temperature, depth, and grazing all play a role in the growth and proliferation of macroalgae. These factors would introduce uncertainty into nutrient-macroalgae relationship modeling.

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3.3.2.3. Dissolved Oxygen (DO)

A. Background/Introduction

DO is the concentration of oxygen gas contained in water. It dissolves directly from the atmosphere and is also produced during photosynthesis in aquatic plants or algae. Aquatic organisms require oxygen for respiration. As DO concentration decreases, the ability of organisms to take in adequate oxygen also decreases. The effects of low DO can be measured through water and sediment chemistry or by observing biological endpoints. Biological endpoints indicative of low DO conditions include mortality (i.e., fish kills), decreased growth, avoidance, and changes in species composition. DO is typically measured with a discrete grab sample or with continuous monitoring sondes deployed at a field site, and the measurements are reported as mg/L or as percent saturation.

The amount of DO in water is influenced by temperature, salinity, movement of water, nutrient pollution, and general oxygen demand in the system. Increasing temperatures and salinity both decrease the solubility of oxygen in water. In estuaries, salinity gradients are the most common cause of stratification (Diaz 2001; Hagy et al. 2004), where lighter, fresh water flows on top of denser and more saline water. Temperature can be another contributing factor to stratification (Stanley and Nixon 1992), where warm, lighter surface water (heated by sunlight) overlies denser, cooler water. Moving or turbulent water tends to increase DO concentration through introduction of atmospheric oxygen through reaeration. Circulation is increased by wind and tide and decreased by calm conditions and stratification. Nitrogen and phosphorus pollution is linked with increased plant and algae growth, which potentially results in increases of oxygen due to photosynthesis and decreases due to respiration. Other factors that increase the demand for DO include decomposition of organic matter by bacteria, fungi, and other decomposer organisms. DO can fluctuate widely within hours, owing to wind-induced mixing, tides, wind-induced seiches, and diel patterns of photosynthesis and respiration. Tides and seiches can move low-DO bottom water into nearshore zones (Breitburg 1990); daytime photosynthesis increases DO and nighttime respiration decreases DO (Breitburg 1990); and onset of wind can mix an unstratified but stagnant body of water. If consumption exceeds production, DO can decline to hypoxic levels (e.g., < 3 mg/L in LIS) or anoxic levels (< 0.1 mg/L).

Hypoxia is an issue affecting estuaries and coastal regions throughout the United States and globally. The amount of information about LIS is growing (e.g., Howell and Simpson 1994), along with information from the northern Gulf of Mexico (Rabalais et al. 2001; Rabalais et al. 2002), Chesapeake Bay (Batiuk et al. 2009; Breitburg 1990; Hagy et al. 2004), Pamlico Sound (Stanley and Nixon 1992), and many other areas. LIS has shown evidence of seasonal hypoxia for decades with western LIS experiencing episodic hypoxia beginning in the 1970s and severe recurring hypoxia reported in the East River and bottom

waters in the western Narrows (Parker and O'Reilly 1991). The severity of hypoxic events has moderated over the past 10 years, with the summer of 2015 showing the second smallest area of hypoxia recorded during the 28-year record of available data (LISS 2015). Figure 2 shows late summer DO concentrations measured in LIS in 2015. Other summer hypoxia maps from the Connecticut Department of Energy and Environmental Protection can be accessed at
http://www.ct.gov/deep/cwp/view.asp?a=2719&q=325532&depNav_GID=1654.

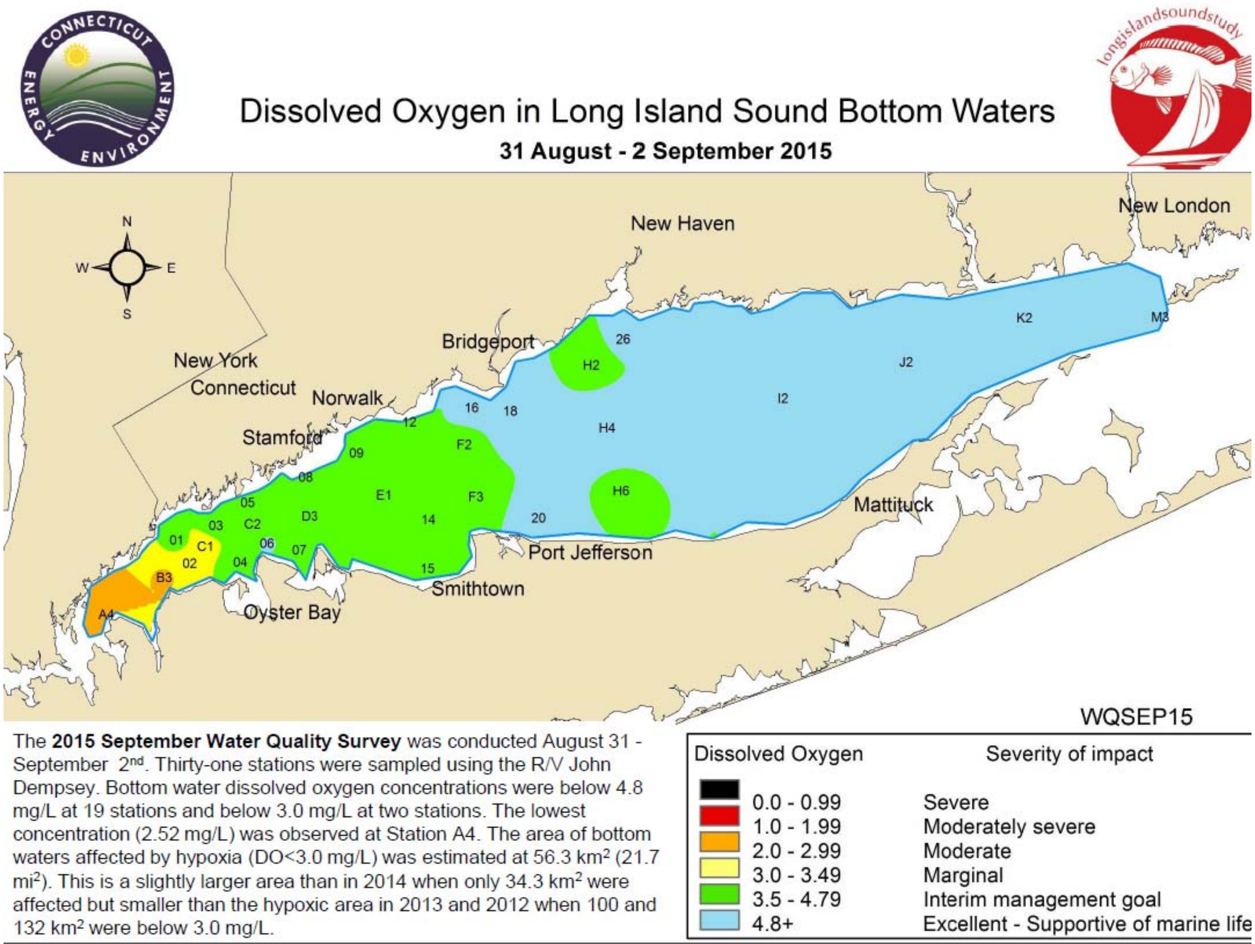

Figure 2. Concentration of DO Measured in LIS in Late Summer 2015 (CTDEEP 2015)

A large hypoxic zone in an estuary can lead to substantial changes in fish, benthic, and plankton communities. The lack of oxygen forces fish and mobile benthic invertebrates to migrate out of an area and represents a habitat loss. In extreme cases, anoxia can lead to fish kills (Howell and Simpson 1994; Kidwell et al. 2009). Benthic organisms that cannot escape are variably affected as the oxygen levels decline. Chronic hypoxia and short-term but recurring anoxia cause marked changes in the benthic invertebrate community of estuaries (Baker and Mann 1992; Baker and Mann 1994a; Baker and Mann 1994b; Baustian and Rabalais 2009; Breitburg 2002). In addition to mortality, motile organisms may migrate from the hypoxic zone (Diaz 2001). Even intermittent hypoxia can cause benthic assemblage composition to shift to resistant or tolerant organisms and to reduce fish predation pressure in the hypoxic zone due to avoidance (Kidwell et al. 2009). Hypoxia has been implicated in recent increases and late-summer dominance of gelatinous zooplankton (jellyfish and ctenophores) in Chesapeake Bay and other eastern estuaries, because the gelatinous zooplankton tend to be more tolerant of hypoxia (Grove and Breitburg 2005). If the hypoxia extends into shallow waters, it could affect spawning and nursery areas of important fish species. In LIS, hypoxia is known to decrease growth of winter flounder and abundance of lobster, squid, bluefish, and butterfish (Howell and Simpson 1994).

The DO concentrations typically observed in LIS during the late summer are thought to reduce the abundance and diversity of fish; reduce organism growth rates; cause mortality in sensitive, slow moving or larval lifestages; reduce disease resistance in exposed organisms; and cause degradation of habitat (LISS 2016). Species thought to be the most sensitive to hypoxia include squid, bluefish, butterfish, winter flounder, and lobster (Howell and Simpson 1994).

B. Sensitivity to Nutrients

The cause and effect relationship between nitrogen/phosphorus pollution and marine and estuarine hypoxia is clear and unequivocal on a global scale (Conley et al. 2009a; Conley et al. 2009b; Conley et al. 2009c; Diaz 2001; Diaz and Rosenberg 2008; Dodds 2006). In LIS, nitrogen is the nutrient primarily responsible for algal growth (LISS 2015). Table 11 summarizes literature findings on trends, causes, and effects of hypoxia in estuaries and coastal areas. The pathway of eutrophication leading to hypoxia is well known. Increases in the concentrations of nitrogen or phosphorous can trigger excess algal growth. The organic algal biomass then decomposes, which consumes oxygen, depleting the water column of DO. Organic loading by itself (such as raw sewage or untreated pulp mill effluent) can also cause hypoxia. Although oxygen is produced by algal growth, both respiration by the algae and decomposition (bacterial respiration) and respiration by other organisms deplete the oxygen. The combined respiration rates can use up the oxygen at night and in deep waters, where there is insufficient light to support photosynthesis.

Hypoxia and anoxia in bottom waters results in surface sediment anoxia, sometimes setting up severe reducing zones. The reducing environment of the sediment has biogeochemical consequences, including release of soluble reactive phosphorus, ammonia (NH_3), and toxic hydrogen sulfide (H_2S) (Diaz and Rosenberg 2008; Kemp et al. 2005; McCarthy et al. 2008). The sediment of hypoxic zones, thus, becomes a potential source of internal nutrient loading, which can further exacerbate eutrophication.

Systems that have had persistent and chronic hypoxia often fail to recover even after pollution loadings have been reduced, possibly because of internal loading (Conley et al. 2009a; Conley et al. 2007; Diaz and Rosenberg 2008). Hypoxia creates a carbon/energy sink in deep water. The carbon is removed from the food chain and is not available to top predators (many commercial and sport fish) or filter feeders (oysters and clams). Reduced fishery production of hypoxic zones has been documented worldwide (Diaz and Rosenberg 2008), although it is can be offset to some extent by increased fisheries production at the margins of the hypoxic zone.

Table 11. Summary of Literature Findings on Trends, Causes, and Effects of Hypoxia in Estuaries and Coastal Areas

Response	Cause/ Predictor	Location	Equation Describing Relationship?	Synopsis	Reference
DO	Eutrophication	LIS	No	Long-term trend evaluation of eutrophication and low DO.	O'Shea and Bronsnan 2000
DO, hypoxia	Abundance and growth of organisms	LIS	No	Effect of low DO on resident species.	Howell and Simpson 1994
DO, hypoxia	Nitrogen loading	Chesapeake Bay	Yes	A simple model for forecasting the effects of nitrogen loads on Chesapeake Bay hypoxia. Target 35% nitrogen loading reduction will reduce hypoxic volumes by 36–68%, roughly half of loadings reported between 1980 and 1990.	Scavia et al. 2006
Eutrophication	Sediment records	Chesapeake Bay	No	Reconstruction of the progression of eutrophication and anoxia/hypoxia over the past 5 centuries.	Zimmerman and Canuel 2002
Hypoxia	Benthic foraminifers	Chesapeake Bay	No	Benthic foraminifera (protists) were used as bioindicators to estimate the timing and degree of changes in DO over the past five centuries. Low DO correlated with nitrogen loading, also modified by river flow.	Karlsen et al. 2000
Hypoxia	Nitrogen	Chesapeake Bay	Yes	Long-term pattern of hypoxia and anoxia in Chesapeake Bay and its relationship to NO_3- loading. Requires 40% reduction of nitrogen loading, to a total nitrogen loading of 50 x 10^6 kg/yr.	Hagy et al. 2004
Phytoplankton, clarity, DO, and SAV	Nutrient loading	Choptank and Patuxent Rivers	No	Nutrients and response over time.	Fisher et al. 2006

Howes et al. (2003) compared nitrogen thresholds and water quality classifications based on site-specific biological and chemical indicators, that were developed independently, by three different entities for three Cape Cod embayments (see Table 8 in seagrass section). Based on water quality classifications in the state of Massachusetts, total nitrogen and dissolved oxygen concentrations fit in the following qualitative and quantitative classifications (Howes et al. 2003):

- *Excellent*: total nitrogen concentrations below 0.30 mg/L (corresponds to dissolved oxygen concentrations greater than 6.0 mg/L and only small oxygen depletions, generally not less than 90% of air equilibrium).

- *Excellent/good*: total nitrogen concentrations 0.30–0.39 mg/L (corresponds to dissolved oxygen concentrations not less than 6.0 mg/L with occasional depletions being rare).

- *Good/fair*: total nitrogen concentrations 0.39–0.50 mg/L (corresponds to dissolved oxygen concentrations generally not less than 5.0 mg/L with depletions to below 4.0 mg/L being infrequent).

- *Moderate impairment*: total nitrogen concentrations 0.50–0.70 mg/L (corresponds to dissolved oxygen concentrations not less than 4.0 mg/L).

- *Significant impairment*: total nitrogen concentrations 0.70–0.80 mg/L (corresponds to stressful dissolved oxygen concentrations).

- *Severe degradation*: total nitrogen concentrations over 0.80 mg/L (corresponds to periodic complete and/or near complete loss of oxygen in bottom waters).

The Massachusetts Estuaries Project (MEP) has been working with a linked watershed/estuary model to determine nitrogen thresholds for 89 estuaries in southeastern Massachusetts to protect the health of each estuary using indicators, including benthic communities and infaunal habitats. Based on available linked watershed/estuary models available for 33 estuaries, the total nitrogen target concentrations range from 0.4 mg/L to 0.6 mg/L to maintain healthy infaunal habitat, and 0.45 mg/L to 0.910 mg/L to sustain benthic communities (MEP n.d.).

In LIS, the efforts to reduce nitrogen input in the watershed have been successful in moderating hypoxic events generally in the main body of the Sound (LISS 2015). Thus, DO will be an important endpoint to consider in this study. Other coastal areas and estuaries in the Northeast are challenged with nutrient pollution and assess their influence by measuring nutrients, chlorophyll *a*, DO, and water clarity.

C. Strengths and Weaknesses of Using This Endpoint

DO is frequently monitored in aquatic systems. An abundance of DO monitoring data gathered over multiple years is available for LIS and trends can be evaluated using that information (Figure 3). DO is directly linked to survival, growth, and other endpoints in aquatic biota; thus, there is a distinct cause-and-effect relationship between DO and survival and fitness in fish and aquatic invertebrates. However, DO dynamics are complex, fluctuating significantly across seasons and even within a single 24-hour period. The use of DO monitoring data will be evaluated carefully to understand the implications of measured concentrations. The correlation between nutrients and resulting DO will be analyzed to evaluate the relationship between a range of nutrient concentrations and effects in individual embayment ecosystems, which can be significantly different.

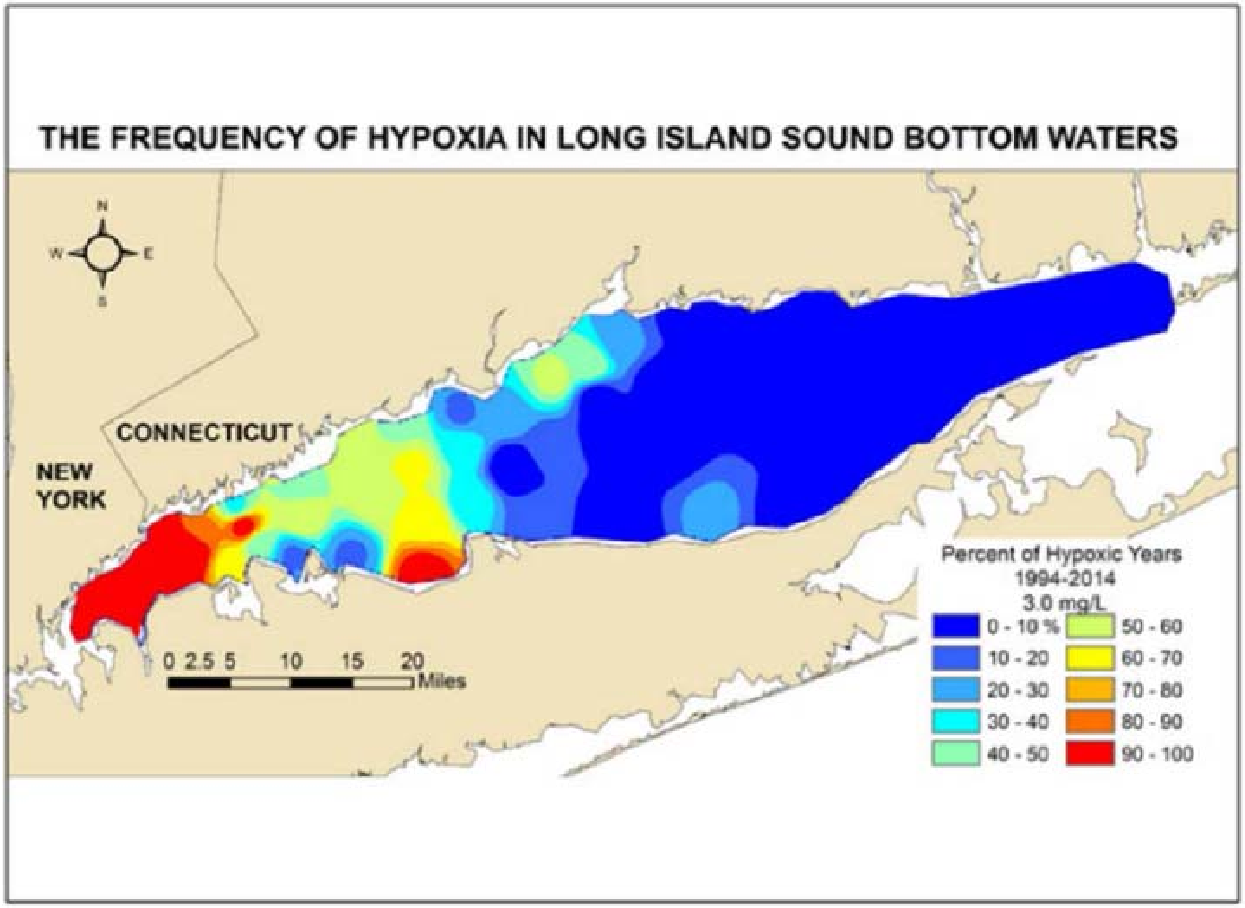

Figure 3. Frequency of Hypoxia in LIS from 1994 to 2014 (LISS 2015)

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3.3.2.4. Phytoplankton

A. Background/Introduction

Phytoplankton are microscopic, free-floating autotrophic organisms that inhabit aquatic ecosystems and consist of several taxonomic groups such as chlorophytes, cryptophytes, cyanobacteria, chrysophytes, diatoms, and dinoflagellates. They exist in varying sizes (e.g., < 2 µm [picoplankton], 2–20 µm [nanoplankton], > 20 µm [microplankton]) (Paerl et al. 2003; Paerl and Justić 2011). While phytoplankton can include mixotrophic species (taxa that acquire energy through autotrophic and heterotrophic pathways, typified by some dinoflagellates), they are typically dominated by autotrophic taxa (Boyer et al. 2009). Phytoplankton represent an important component of estuarine food webs through photosynthesis and primary production, as well as playing a central role in oxygen, nutrient, and carbon cycling (Paerl and Justić 2011). Their primary production rates are extremely variable and can range from near zero to several grams of carbon per square meter per day (Paerl and Justić 2011). Phytoplankton account for at least half of estuarine and coastal primary production, having fast growth rates and some having the ability to grow and proliferate in *blooms* (e.g., dinoflagellates, cryptophytes, cyanobacteria) (Paerl and Justić 2011). Blooms can be defined as "events of rapid production and accumulation of phytoplankton biomass that are usually responses to changing physical forcings" (Cloern 1996). Oftentimes, changes in phytoplankton communities precede changes in the ecosystem such as changes in oxygen balance, fisheries, benthic communities, and plant life (Paerl and Justić 2011); thus phytoplankton are often used as an indicator of change in nutrient inputs to the waterbody as well as in evaluating response to other stressors (Domingues et al. 2008).

LIS is an estuary known to have high levels of primary production, with substantial phytoplankton growth (Riley 1956; Sun et al. 1994).[4] Based on national estuarine studies conducted by Bricker et al.

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[4] The most comprehensive study of the characteristics of LIS and phytoplankton was conducted by Riley and his colleagues in the 1950s (e.g., Conover 1956; Riley 1956), in which they analyzed physical, chemical (e.g., nutrient dynamics), and biological (e.g., phytoplankton, zooplankton, pelagic fish eggs and larvae) features (Capriulo et al. 2002). Many independent studies have been conducted since then but not all have been published and their study durations and/or the geographic regions have been limited (Capriulo et al. 2002).

(1999, 2007), LIS has been considered eutrophic for several decades, as indicated by increased abundances of phytoplankton. A study that took place between 1992 and 1995 in all three regions of LIS (western, central, and eastern regions) showed a wide range of observed chlorophyll *a* concentrations between 0.4 and 67 µg/L (Capriulo et al. 2002). Although LIS is generally considered eutrophic, Capriulo et al. (2002) indicated that there appears to be a wide range of spatial and temporal variation, which is consistent with other studies. For example, coastal phytoplankton communities in estuaries show strong seasonal and spatial distributions, as seen in the Chesapeake Bay, where chlorophyll is strongly influenced by freshwater inputs (Paerl and Justić 2011). The impact of freshwater inputs into LIS are similar to those of the Chesapeake Bay, which has low flushing rates (e.g., residence time of 2–3 months in LIS) (Bricker et al. 2007). The relatively long residence time is attributed to LIS's physical features. In addition, researchers have found that during low flows, the East River was the main source of inorganic nutrients (nitrates and orthophosphates), and during high flows, the main source of inorganic nutrients was the Connecticut River (Buck et al. 2005), which indicates that both seasonal and geographical factors should be taken into account when evaluating levels of phytoplankton biomass to protect designated uses in LIS.

In addition to temporal and spatial variation of phytoplankton biomass within LIS, phytoplankton species composition differs among locations within the Sound (Capriulo et al. 2002). However, phytoplankton composition is generally dominated by diatoms, except during the summer, when dinoflagellates and other small phytoflagellates are more abundant (Sun et al. 1994; Suter et al. 2014). The majority of phytoplankton consists of diatoms in the winter through spring. In contrast, dinoflagellates increase in abundance in the summer while diatom biomass decreases (Conover 1956). In general, larger sized phytoplankton, such as diatoms, survive better in variable nutrient environments and can exist in higher concentrations because of their ability to store nutrients when available (Sutula 2011). Based on which phytoplankton species outcompete others under stressful conditions (e.g., increased nutrient inputs), phytoplankton composition can also be considered in protecting designated uses. In recent years, Suter et al. (2014) found that diatoms decreased in biomass more than in past years compared to other species, which increased. Increases in non-diatom species were attributed to dinoflagellates such as Prymnesiophyceae, Cryptophyceae, Raphidophyceae, and Euglenophyceae (Suter et al. 2014). Despite these declines, diatoms remained the most abundant taxa during the 8 years of analysis from 2002 to 2009 (Suter et al. 2014).

Seasonal Blooms in LIS

LIS continues to experience two distinct blooms annually, as it did in the 1950s (Conover 1956). This is similar to the incidence rate for northeastern coasts of the United States along the Gulf of Maine, which experiences a winter-spring bloom and a summer-fall bloom (Conover 1956; George et al. 2015; Sun et al. 1994). During the winter, cold temperatures and strong mixing caused by turbulence keep the water column well-mixed and more turbid (Conover 1956). Because the water is cold and there is not enough light for phytoplankton, blooms generally do not occur in the winter. In LIS, there is a late winter-spring bloom and a late summer bloom, which both occur when there is a breakdown of stratification within the water column. Factors such as nutrients, water temperature, salinity, density, hydrography, phytoplankton and zooplankton species composition, and available light affect the intensity of the blooms (Capriulo et al. 2002; George et al. 2015). In LIS, algal blooms are more intense in the western end (Sun et al. 1994), which is attributable to physical and geomorphological features of LIS. In addition, studies have found that temperature-stimulated grazing can influence the magnitude of the winter-spring diatom bloom (Keller et al. 2001; Oviatt et al. 2007). The size and length of the seasonal blooms in LIS could be affected by any of these factors.

The winter-spring bloom (mostly comprised of diatoms in LIS) occurs toward the end of winter (any time between January and March)—when light increases (i.e., phytoplankton are light-limited in the winter) and rapid stratification due to temperature and salinity occurs—and it ends when nutrients are depleted, self-shading or grazing becomes too intense, or there is inclement weather (Behrenfeld 2010; Capriulo et al. 2002; Chen et al. 1988; Conover 1956; George et al. 2015; Riley 1956; Thomas et al. 2000). In a study observing copepod abundance in LIS in 1982 and 1983, the highest chlorophyll concentrations were found during a mid-winter bloom in February (approximately 25 µg/L) (Peterson 1985). George et al. (2015) conducted a study in 2010 and 2011, when the winter-spring bloom was not initiated because of stratification (i.e., the water temperatures were at an annual minimum), but rather because the rate of phytoplankton growth exceeded zooplankton grazing (George et al. 2015). Chlorophyll *a* concentrations averaged 3 µg/L before the bloom, reaching 11 µg/L during the bloom and dropped to less than 2 µg/L after the bloom, at the monitoring station in central LIS (George et al. 2015).

Towards the end of summer, vertical mixing of the water column occurs because of the cooling of water temperatures. This mixes the nutrients from the bottom of the water column toward the surface. Because phytoplankton are nutrient-limited during the summer, they respond to sudden increases in the availability of nutrients, which can result in the summer-fall bloom.

Rice and Stewart (2013) found that there appears to be a shift in the seasonality of phytoplankton blooms compared to historical trends. Historical data (i.e., 1939 and 1952–1958) showed that there used to be a larger bloom (roughly double that of fall blooms) in the winter-spring (between late January and early March), but in recent years (1995–2010), blooms have appeared similar in size in the spring and the fall—mean chlorophyll *a* concentration peaked at 8.9 µg/L in the early spring and at 8.4 µg/L in the fall (Rice and Stewart 2013).

The regular occurrence of seasonal blooms in the winter-spring and summer-fall combined with the varying degrees of the size and time when the blooms have occurred in recent years indicate that the frequency and size of nonseasonal blooms in LIS might be a potential factor to consider when looking at nutrient impacts on phytoplankton.

Measuring Phytoplankton

Phytoplankton have many characteristics that make them useful indicators of eutrophication in estuaries, including rapid growth rates in varying nutrient concentrations and many standard methods that can be used in measuring them (Boyer et al. 2009; Domingues et al. 2008; Paerl et al. 2007). Phytoplankton can also be characterized by biomass and composition. *Biomass* refers to the mass of phytoplankton in the water column (i.e., the amount present), and *composition* refers to the taxonomic makeup of the phytoplankton assemblage (i.e., the diversity of species present).

Biomass is typically measured using extraction and analysis of photopigments with a variety of methods, the most common being the spectrophotometric or fluorescence analysis of chlorophyll *a*. Chlorophyll *a* has a long history of being used as a surrogate measure of phytoplankton abundance (or biomass) in the water column, because of its abundance in most species of phytoplankton (Boyer et al. 2009; Cullen 1982; Day et al. 1989; Paerl et al. 2003; Steele 1962; Sun et al. 1994). However, a weakness of the use of chlorophyll *a* as a measure of phytoplankton is the variability of cellular chlorophyll content among species (Boyer et al. 2009). In LIS, diatoms were the most correlated with chlorophyll *a* of any phytoplankton taxa (Suter et al. 2014). Photopigment-based analyses are an estimate of biomass only, as photopigment composition varies across different algal taxa and even within taxa across a variety of environmental conditions (Paerl et al. 2003; Sutula 2011). Biovolume estimation based on microscopy or cytometry and imaging is another approach for biomass estimation, as well as gravimetry to estimate dry and ash-free dry mass.[5]

Composition is primarily measured with collection and microscopic identification to the lowest practical taxonomic level, and other methods are evolving as gene-based techniques improve. Chromatographic separation of photopigments has allowed biomass and gross assemblage composition to be analyzed simultaneously and is being used more often (Mackey et al. 1996; Millie et al. 2004; Paerl et al. 2006; Paerl et al. 2007; Pinckney et al. 2001). As a result of competitive differences among multiple species, as nutrient concentrations increase, species composition, diversity, and abundance of different floral species are changed from what is expected of natural habitats in the region (Hutchinson 1959; Tilman 1977, 1981, 1985).

Chlorophyll *a* also serves as an index of the productivity and trophic condition of waters. Higher concentrations of chlorophyll *a* are indicative of overproduction of algae, which might be related to nitrogen/phosphorus pollution. Excess phytoplankton biomass (i.e., higher chlorophyll *a* concentrations) is associated with nuisance algae blooms, reduced light and shading seagrasses, low dissolved oxygen, and shifts in aquatic species composition.

Phytoplankton species generally exhibit increases in growth rates with a rise in the availability of limiting resources such as nutrients (Wetzel 2001). There is also a large collection of scientific literature on the correlative effect of nutrients on both the alteration of species composition and increased biomass across the range of aquatic ecosystems (Kalff 2002; Schindler 1990; Wetzel 2001). As a result, one would expect a concurrent shift in species composition as well as an increase in the total abundance of primary producers (Elser et al. 2007; Kalff 2002; Schindler 1990; Smith and Schindler 2009; Wetzel 2001). Because the effect of nutrients on species composition is correlated with overall system productivity or biomass, chlorophyll *a* can be considered an indicator of a harmful increase in nutrient concentrations.

B. Sensitivity to Nutrients

Increased nutrient loads have been documented to cause shifts in the composition of phytoplankton and zooplankton assemblages in estuarine and freshwater ecosystems (Arhonditsis et al. 2007; Armitage and Fong 2004; Cloern 2001, 1996). Some species shifts occur in response to a change in the relative abundance of different nutrients. Increased abundance of nitrogen and phosphorus sometimes results in silica limitation, which favors non-diatom species because they do not require silica (Arhonditsis et al. 2007; Armitage and Fong 2004; Cloern 2001, 1996).

In LIS, there is a eutrophication gradient from east to west (LISS 2008). Capriulo et al. (2002) concluded that excess nutrient loading into the Sound appears to be resulting in elevated phytoplankton and zooplankton biomass (and elevated chlorophyll) in the western end. Capriulo et al. (2002) found that phytoplankton productivity and taxonomic composition in LIS were attributed to the ratio of various inorganic nutrients such as nitrogen, phosphorus, and silicate (which helps diatoms sustain rapid growth rates) (Capriulo et al. 2002). The ratio of nitrogen to phosphorus availability in LIS could vary depending on location because of the higher sewage-derived nutrient inputs in the western end (Capriulo et al. 2002). Gobler et al. (2006) found that nitrogen inputs had more impact on phytoplankton communities

[5] http://nutrients.tetratech.com/library/NutrientandResponseVariableOverviews.html

in central LIS than in the western or eastern LIS, based on data collected in July 2000 and April 2001. Another study conducted by Goebel et al. (2006) concluded that high levels of production and phytoplankton biomass are consistent with nitrogen loads into western and central LIS. The authors also noted that the “ranges and temporal variability in daily and annual rates compare favorably to those found through other nearby, eutrophic estuarine systems, such as Narragansett and Chesapeake Bays, despite variations in spatial distributions of production” (Goebel et al. 2006).

Research Studies Providing Relevant Information about Phytoplankton as an Endpoint

The capacity for nutrients to cause increased primary productivity and biomass, change in algal species composition, and changes in chlorophyll concentration with nutrient concentrations has been observed in numerous estuarine studies (see Table 12).

Table 12. Phytoplankton Information Relevant to Nutrient Impacts

Response	Citation
In the Chesapeake Bay, this study found the following: To prevent low DO[1]: • 7.2–11 µg/L (mean chlorophyll *a* from May through August) in the deeper bay • 9.0–14 µg/L (annual mean chlorophyll *a*) in the tidal tributaries To provide sufficient light conditions for seagrass: • 7.9–12 µg/L (mean chlorophyll *a*) at 2 m depths • 19–28 µg/L (thresholds during the growing season) at 1 m depths	Harding et al. 2014
Mid- and lower Choptank River (part of the Chesapeake Bay) where seagrass was able to survive: 1985–1988 (May through October) • Mean DIN < 10 µM • Mean dissolved phosphate < 0.35 µM • Mean suspended sediment < 20 mg/L • Mean chlorophyll *a* < 15 µg/L • Mean light attenuation coefficient (K_d) < 2 m^{-1}	Stevenson et al. 1993
Chlorophyll correlated with total nitrogen load and DO changes in Chesapeake Bay	Boynton et al. 1996
In Patuxent River Estuary, Chesapeake Bay: • Decrease in total nitrogen and total phosphorus (40–60% decrease) from point source controls led to a decrease in dissolved inorganic nitrogen, total nitrogen, dissolved inorganic phosphorus, and total phosphorus concentrations. No decline in chlorophyll noted. • Total nitrogen in the river decreased from 1.75–3.50 mg/L (125–250 µM) (pre-biological nitrogen removal [BNR] system put into place in 1991) to < 1.75 mg/L (< 125 µM) (post-BNR). Dissolved inorganic nitrogen in the upper estuary declined from 0.84 mg/L (60 µM) to 0.63 mg/L (45 µM). • Total phosphorus in the river decreased from 0.12–0.37 mg/L (4–12 µM) (pre-phosphorus ban in 1984) to < 0.12 mg/L (< 4 µM) (post-phosphorus ban). Dissolved inorganic phosphorus in the upper estuary decreased from 0.046 mg/L (1.5 µM) to 0.039 mg/L (1.25 µM).	Testa et al. 2008
Increased summer cyanobacteria abundance and biomass in impaired waters because of decreased light and increased nutrients in Chesapeake Bay	Marshall et al. 2006
Phytoplankton Index of Biotic Integrity developed for Chesapeake Bay: • Based on chlorophyll, composition, and carbon content • Discriminates degraded and reference sites	Lacouture et al. 2006

Response	Citation
Patuxent River, MD: • Increase in P of 1.5-10x • Increase in N of 1.4-2.6x • Resulted in nearly 7-fold increase in chlorophyll *a* in experimental mesocosms	Bundy et al. 2003

[1] Ecosystem impairments by low DO were defined by values of 3.0 and 2.0 mg/L (2.0 mg/L is considered the minimum requirement for fish to survive).

C. Strengths and Weaknesses of Using This Endpoint

Advantages of using phytoplankton include their known sensitivity to nutrient enrichment, their importance in aquatic food webs, and their key intermediate role in driving impacts on light availability and dissolved oxygen along with other impacts (e.g., nuisance blooms, cyanoHAB toxin production). Also, chlorophyll is commonly monitored due to the existence of easy to use and well established techniques as well as emerging technologies (e.g., remote sensing).

Disadvantages include the various other factors that can influence phytoplankton composition and biomass and introduce variability into nutrient-phytoplankton response relationships, as well as limitations in some types of phytoplankton data, especially species composition, due to expertise and resource requirements.

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3.3.2.5. Harmful Algal Blooms (HABs)

Although no general trends in HAB occurrences in LIS were determined from research conducted, reports do provide evidence of some HABs occurring periodically in LIS. Researchers have reported that blooms occur in LIS, especially blooms caused by *Alexandrium*, which includes many species that produce toxins leading to paralytic shellfish poisoning (PSP) (Anglès et al. 2012; Dam et al. 2011; Gobler and Hattenrath-Lehmann 2011). Toxic *Alexandrium* spp. were first detected in LIS in the early 1980s (Dam et al. 2011) and the first HAB for this species was identified in 2006 in Northport and Huntington harbors, causing shellfishing closures (NYSDEC n.d.). One of the most severe *Alexandrium* blooms occurred near Northport Harbor in 2008—the bloom continued from April to June and led to the closing of more than 7,000 acres of shellfish beds (Dam et al. 2011). According to Hattenrath et al. (2010), the 2008 bloom caused high toxicity in blue mussels (*Mytilus edulis*) and native soft shell clams (*Mya arenaria*). Additionally, Hattenrath et al. (2010) suggested that a wastewater source, such as the plant that discharges to Northport Harbor, could promote growth of *Alexandrium*. However, more research is needed to definitively link anthropogenic nitrogen loading to *Alexandrium* blooms.

In 2012, the acreage of HAB closures from Northport Bay to Huntington Bay was approximately 10,000 acres (LISS 2012). NYSDEC maintained a list of all HAB activity by waterbody and county from 2012 to 2016. Although NYSDEC did not specifically report a HAB in any of the priority embayments, one news article from *News 12* in April 2011 reported a HAB occurrence in Northport, New York, which is a priority embayment (NYSDEC 2016; Researchers find levels 2011). Documented HAB occurrences in Connecticut were not available, but the Connecticut Department of Energy and Environmental Protection (DEEP) provides advisories on HABs within the state.

Blooms of *Cochlodinium polykrikoides* (a red tide producing dinoflagellate species) also were reported in Peconic Estuary in the late summers of 2002 through 2009, and were the presumed cause of shellfish and fish mortality (Branca and Focazio 2009). According to Gobler et al. (2012), blooms of *Cochlodinium polykrikoides*, a harmful dinoflagellate, occur elsewhere in LIS and are common along the east coast of the United States. Gobler et al. (2012) found that this dinoflagellate is nutritionally flexible and responds favorably to different forms of nitrogen, depending on the environment (e.g., eutrophic, mesotrophic). Although HABs are known to be caused by increases in nutrients, the process is complex and numerous other factors influence their occurrence and toxin production, including growth of macroalgae and phytoplankton and hydrodynamics. HABs are not being proposed as an endpoint for LIS because insufficient data were available to link specific incidences and trends in HAB occurrences to nutrient increases in the Sound.

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3.3.2.6. Oysters

Since the 1950s, there has been no general trend in the amount of oysters caught in Connecticut and New York. In 1997, there was a drop in the harvest of oysters due to a disease outbreak. The decline in the economic value of the oyster industry from reduced harvests has yet to fully recover (LISS 2017). Oysters from the Sound are harvested by either Connecticut or New York. There are several sources of oyster data for the area, which report the data differently. NOAA reports the pounds of Eastern oysters caught each year from New York (1950–2015) and Connecticut (1950–2007) and the value of the amount caught in dollars. The Connecticut Department of Agriculture reports the number of bags caught each year (1990–2010) and the value of the amount caught in dollars. The LISS has overlaid the New York and Connecticut oyster harvest since 1990, which shows that Connecticut far exceeds the economic value of oysters in New York for most of the years that data were collected from both states (LISS 2017). There are no data on the amount of oysters harvested in Connecticut since 2011.

When comparing the NOAA data, it is interesting to note that the increases and decreases in the amount of oysters from each state do not correlate with each other. For example, Connecticut had a peak at well over 4 million pounds between 1991 and 1994 while New York was on the low end ranging from 126,000 to 886,000 pounds within the same time period.

While there has not been any data reported from Connecticut since 2011, “resource managers believe that harvests continue to rise” (Aquaspace 2016). Brooks (2015) describes a thriving oyster industry throughout Connecticut the past few years and the local farmers who are confident about the oyster industry in LIS.

MacKenzie (2007) examines the historic decline of oyster landing on the east coast (from South Carolina to Rhode Island). The author reports a general decline from 1890 to 2004, including in LIS—a decline in Connecticut-New York-Rhode Island oyster landings from 3.8 million bushels in 1890 to 1.5 million bushels in 1940 (MacKenzie 2007). The article includes several main causes of the decline in landings along the east coast: (1) falling demand for oysters, (2) economic depressions, and (3) biological and physical damage to oysters and oyster beds from predation, siltation, storms, and dredging. This research, combined with more current estimates of oyster landings in Connecticut and New York, do not point to a compelling linkage between declines in oyster landings and nutrient enrichment.

Although oyster catch is an indirect reflection of water quality, oysters are not being proposed as an endpoint for LIS because changes in oyster populations were reported to be associated with more than nutrients (e.g., damage from storms, decline in demand for oysters), so it would be difficult to apportion the effects of nutrients on oyster populations in LIS.

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3.3.2.7. Summary of Assessment Endpoints

Table 13 presents a summary of numeric values related to ecosystem response (including seagrass, macroalgae, dissolved oxygen, and phytoplankton) to nitrogen and phosphorus inputs relevant to LIS. These values were previously provided in sections 3.3.2.1 to 3.3.2.4.

Table 13. Summary of Endpoints

<table>
<tr><th>Study Location</th><th>Response</th><th>Nitrogen</th><th>Phosphorus</th><th>Other</th><th>Other Explanation</th><th>Citation</th></tr>
<tr><td rowspan="4">Guidelines for Long Island Sound</td><td rowspan="4">Recommendation for limits of critical Z. marina habitat parameters</td><td rowspan="4">DIN < 0.05 mg/L</td><td rowspan="4">DIP < 0.02 mg/L</td><td>K_d (1/m) < 0.7</td><td rowspan="4">Guidelines for Long Island Sound. N and P values in Yarish et al. (2006) reported in µM.</td><td rowspan="4">Yarish, C., R. E. Linden, G. Capriulo, E. W. Koch, S. Beer, J. Rehnberg, R. Troy, E. A. Morales, F. R. Trainor, M. DiGiacomo-Cohen, and R. Lewis. 2006. Environmental Monitoring, Seagrass Mapping and Biotechnology as Means of Fisheries Habitat Enhancement along the Connecticut Coast. Final Grant Report to CT DEP Long Island Sound Research Fund, CWF-314-R.</td></tr>
<tr><td>Chlorophyll a < 5.5 µg/L</td></tr>
<tr><td>TSS < 30 mg/L</td></tr>
<tr><td>Sediment Organic matter (%): < 3</td></tr>
<tr><td rowspan="6">Guidelines for Long Island Sound (Case Study Sites)</td><td rowspan="6">Restoration guidelines for SAV based on water quality and habitat-based requirements</td><td rowspan="6">DIN < 0.03 mg/L (secondary requirement - diagnostic tool)</td><td rowspan="6">DIP < 0.02 mg/L (secondary requirement - diagnostic tool)</td><td>Primary requirement: Minimum Light Requirement at the Leaf Surface (%) > 15 %</td><td rowspan="6">Guidelines for Long Island Sound (Case Study Sites)</td><td rowspan="6">Vaudrey, J.M.P. 2008b. Establishing Restoration Objectives for Eelgrass in Long Island Sound. Part II: Case Studies. Final Grant Report to the Connecticut Department of Environmental Protection, Bureau of Water Protection and Land Reuse and the U.S. Environmental Protection Agency.</td></tr>
<tr><td>Substitute for minimum light requirement at leaf surface: Water Column Light Requirement (%) > 22 %</td></tr>
<tr><td>K_d (1/m) < 0.7 (for reference, use minimum light as standard)</td></tr>
<tr><td>Chlorophyll a < 5.5 µg/L (secondary requirement - diagnostic tool)</td></tr>
<tr><td>Sediment Organics (%): <10 (habitat constraint)</td></tr>
<tr><td>Vertical distribution (m): Zmax = 1 m + Zmin (habitat constraint)</td></tr>
</table>

Study Location	Response	Nitrogen	Phosphorus	Other	Other Explanation	Citation
Long Island Sound	Decrease in the ability of eelgrass to thrive	Watershed-derived nitrogen loading > 50 kg N/ha/yr				Latimer, J.S., and S.A. Rego. 2010. Empirical relationship between eelgrass extent and predicted watershed-derived nitrogen loading for shallow New England estuaries. *Estuarine, Coastal and Shelf Science* 90(4):231–240. doi: http://dx.doi.org/10.1016/j.ecss.2010.09.004.
	Eelgrass absent	Watershed-derived nitrogen loading > 100 kg N/ha/yr				
Narragansett Bay, Rhode Island - Mesocosm experiment	Negative effects on *Z. marina* (decrease in density and below-ground production and an increase in time-interval between initiation of new leaves)	Nutrient additions of 6 mmol N/m^2/d (dissolved form from stock solutions of $NaNO_3$)	Nutrient additions of 0.5 mmol P/m^2/d (dissolved form from stock solutions of KH_2PO_4)	Increased temperatures by 4 degrees Celsius		Bintz, J.C., S.W. Nixon, B.A. Buckley, and S.L. Granger. 2003. Impacts of temperature and nutrients on coastal lagoon plant communities. Estuaries 26(3):765–776.
Waquoit Bay, Massachusetts	Significant seagrass loss of 80–96 % bed area	Contributing watershed nitrogen loads of approx. 30 kg N/ha/ yr				Hauxwell, J., J. Cebrian, I. Valiela. 2003. Eelgrass Zostera marina loss in temperate estuaries: relationship to land-derived nitrogen loads and effect of light limitation imposed by algae. *Marine Ecology Progress Series* 247:59–73.
	Near-total disappearance of seagrass	Contributing watershed nitrogen loads ≥ 60 kg N/ha/yr				

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Massachusetts Estuaries (19 estuaries)	Healthy seagrass	TN 0.42 mg/L		Total chlorophyll a = 5.1 µg/L; Light penetration = 23.7 %		Benson, J.L., D. Schlezinger, and B.L. Howes. 2013. Relationship between nitrogen concentration, light, and Zostera marina habitat quality and survival in southeastern Massachusetts estuaries. *Journal of Environmental Management* 131:129–137.
	Degraded/ declining seagrass sites			Light penetration < 21.0 %		
	Percent eelgrass transplant survival < 25 %	TN: 0.68 ± 0.11 mg/L				
	Percent eelgrass transplant survival: 25–50 %	TN: 0.67 ± 0.11 mg/L				
	Percent eelgrass transplant survival: 50–75 %	TN: 0.49 ± 0.12 mg/L				
	Percent eelgrass transplant survival > 75 %	TN: 0.39 ± 0.03 mg/L				
Southeastern Massachusetts estuaries	Eelgrass survival	Tidally averaged TN < 0.34 mg/L; Ebb-tide TN < 0.37 mg/L		Bottom light ≥ 100 µE/m^2/s		
Waquoit Bay, Massachusetts: Sage Lot Pond (SLP)	Mean percent net growth of *U. Lactuca* fronds	Watershed N load: 14 kg/ha/yr		% net growth: 69.2 ± 4.4 (SLP)		Morgan, J.A., A.B. Aguiar, S. Fox, M. Teichberg, and I. Valiela. 2003. Relative influence of grazing and nutrient supply on growth of the green macroalga Ulva lactuca in estuaries of Waquoit Bay, Massachusetts. *Biological Bulletin* 205:252–253.
Waquoit Bay, Massachusetts: Quashnet River (QR)	Mean percent net growth of *U. Lactuca* fronds	Watershed N load: 350 kg/ha/yr		% net growth: 104.8 ± 7.5 (QR)		
Waquoit Bay, Massachusetts: Childs River (CR)	Mean percent net growth of *U. Lactuca* fronds	Watershed N load: 601 kg/ha/yr		% net growth: 251.9 ± 18.2 (CR)		

Study Location	Response	Nitrogen	Phosphorus	Other	Other Explanation	Citation
Waquoit Bay, Massachusetts	Increases in δ^{15}N of eelgrass, macroalgae, cordgrass, and suspended particulate matter.	δ^{15}N of groundwater nitrate increase from -0.9 ‰ to +14.9 ‰ as wastewater nitrogen increase from 4% to 86 % of the total N pool			Receiving waters δ15N of DIN increasing from +0.5 ‰ to +9.5 ‰, correlated with increases in δ15N of eelgrass, macroalgae, cordgrass, and suspended particulate matter. • Magnitude of change in δ15N of eelgrass is similar to change in δ15N of DIN in groundwater among estuaries. • Magnitude of change in δ15N of cordgrass, macroalgae, and suspended particulate organic matter is approximately one third the size of the change in eelgrass δ15N among estuaries.	McClelland, J.W., and I. Valiela. 1998. Linking nitrogen in estuarine producers to land-derived sources. *Limnology and Oceanography* 43(4):577-585. doi: 10.4319/lo.1998.43.4.0577.
Southeastern Massachusetts estuaries	Restoration of eelgrass	TN: 0.34–0.55 mg/L				MEP. n.d. *The Massachusetts Estuaries Project: Reports Available to Download*. Accessed February 2017. http://www.oceanscience.net/estuaries/reports.htm. Download individual reports for the 33 embayment systems here.
	Restore benthic habitat	TN: 0.45–0.910 mg/L				
	Support healthy infaunal habitat	TN: 0.4–0.6 mg/L				

Study Location	Response	Nitrogen	Phosphorus	Other	Other Explanation	Citation
Cape Cod, MA embayments (Great, Green, and Bournes ponds)	Dense eelgrass beds	TN: < 0.30 mg/L			SMAST values	Howes, B.L., R. Samimy, and B. Dudley. 2003. *Site-Specific Nitrogen Thresholds for Southeastern Massachusetts Embayments: Critical Indicators—Interim Report*. Prepared by Massachusetts Estuaries Project for the Massachusetts Department of Environmental Protection. Accessed February 2017. http://yosemite.epa.gov/OA/EAB_WEB_Docket.nsf/Verity%20View/DE93FF445FFADF1285257527005AD4A9/$File/Memorandum%20in%20Opposition%20...89.pdf
	Eelgrass present	TN: 0.30–0.39 mg/L				
	Eelgrass not present	TN: 0.39–0.50 mg/L				
	Eelgrass not sustainable	TN: 0.50–0.70 mg/L				
	Eelgrass absent	TN: > 0.70 mg/L				
Cape Cod, MA embayments (Great, Green, and Bournes ponds)	Macroalgae generally not present	TN: < 0.30 mg/L			SMAST values	Howes, B.L., R. Samimy, and B. Dudley. 2003. *Site-Specific Nitrogen Thresholds for Southeastern Massachusetts Embayments: Critical Indicators—Interim Report*. Prepared by Massachusetts Estuaries Project for the Massachusetts Department of Environmental Protection. Accessed February 2017. http://yosemite.epa.gov/OA/EAB_WEB_Docket.nsf/Verity%20View/DE93FF445FFADF1285257527005AD4A9/$File/Memorandum%20in%20Opposition%20...89.pdf
	Macroalgae generally non-existent but might be present	TN: 0.30–0.39 mg/L				
	Macroalgae not present/present in limited amounts	TN: 0.39–0.50 mg/L				
	Macroalgal accumulations occur in some regions	TN: 0.50–0.70 mg/L				
	Macroalgal accumulations observed	TN: 0.70–0.80 mg/L				
	Macroalgal accumulations are large and pervasive	TN: > 0.80 mg/L				

Study Location	Response	Nitrogen	Phosphorus	Other	Other Explanation	Citation
Cape Cod, MA embayments (Great, Green, and Bournes ponds)	DO > 6.0 mg/L	TN: < 0.30 mg/L			SMAST values	Howes, B.L., R. Samimy, and B. Dudley. 2003. *Site-Specific Nitrogen Thresholds for Southeastern Massachusetts Embayments: Critical Indicators—Interim Report*. Prepared by Massachusetts Estuaries Project for the Massachusetts Department of Environmental Protection. Accessed February 2017. http://yosemite.epa.gov/OA/EAB_WEB_Docket.nsf/Verity%20View/DE93FF445FFADF1285257527005AD4A9/$File/Memorandum%20in%20Opposition%20...89.pdf
	DO not less than 6.0 mg/L with occasional depletions being rare	TN: 0.30–0.39 mg/L				
	DO generally not less than 5.0 mg/L with depletions < 4.0 mg/L being infrequent	TN: 0.39–0.50 mg/L				
	DO generally does not fall below 4.0 mg/L	TN: 0.50–0.70 mg/L				
	Stressful DO conditions	TN: 0.70–0.80 mg/L				
	Periodic complete/near complete loss of oxygen	TN: > 0.80 mg/L				
Cape Cod, MA	Near complete destruction of eelgrass meadows	Watershed N loads: 15–30 kg/ha/yr				Bowen, J.L., and I. Valiela. 2001. The ecological effects of urbanization of coastal watersheds: Historical increases in nitrogen loads and eutrophication of Waquoit Bay estuaries. *Canadian Journal of Fisheries and Aquatic Sciences* 58:1489–1500.

Study Location	Response	Nitrogen	Phosphorus	Other	Other Explanation	Citation
Northeastern U.S. estuaries	Sharply reduced eelgrass areal cover	Watershed N loads: > 20 kg/ha/yr			(reported based on other studies but without citation)	Bowen, J.L., and I. Valiela. 2001. The ecological effects of urbanization of coastal watersheds: Historical increases in nitrogen loads and eutrophication of Waquoit Bay estuaries. *Canadian Journal of Fisheries and Aquatic Sciences* 58:1489–1500.
	Completely disappeared eelgrass meadows	Watershed N loads: > 100 kg/ha/yr			(reported based on other studies but without citation)	
Chesapeake Bay	Prevent low DO			Chlorophyll *a*: • 7.2–11 µg/L (mean chlorophyll *a* from May through August) in the deeper bay • 9.0–14 µg/L (annual mean chlorophyll *a*) in the tidal tributaries		Harding, L.W., Jr., R.A. Batiuk, T.R. Fisher, C.L. Gallegos, T.C. Malone, W.D. Miller, M.R. Mulholland, H.W. Paerl, E.S. Perry, and P. Tango. 2014. Scientific bases for numerical chlorophyll criteria in Chesapeake Bay. *Estuaries and Coasts* 37:134–148.
	Provide sufficient light conditions for SAV			Chlorophyll *a*: • 7.9–12 µg/L (mean chlorophyll *a*) at 2 m depths • 19–28 µg/L (thresholds during the growing season) at 1 m depths		

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Patuxent River Estuary (Chesapeake Bay tributary)		TN in river decreased from 1.75–3.50 mg/L (125–250 µM) (pre-biological nitrogen removal [BNR] system put into place in 1991) to < 1.75 mg/L (< 125 µM) (post-BNR). DIN in upper estuary declined from 0.84 mg/L (60 µM) to 0.63 mg/L (45 µM).	TP in river decreased from 0.12 to 0.37 mg/L (4–12 µM) (pre-P ban in 1984) to < 0.12 mg/L (< 4 µM) (post-P ban). DIP in upper estuary decreased from 0.046 mg/L (1.5 µM) to 0.039 mg/L (1.25 µM).		Decrease in TN and TP (40–60 % decrease) from point source controls led to a decrease in DIN, TN, DIP, and TP concentrations. No decline in chlorophyll noted.	Testa, J.M., W.M. Kemp, W.R. Boynton, and J.D Hagy, III. 2008. Long-term changes in water quality and productivity in the Patuxent River Estuary: 1985 to 2003. *Estuaries and Coasts* 31:1021–1037.
Chesapeake Bay	Eliminate anoxia	Reduce TN loading to 50×10^6 kg/yr, or 40 % reduction from recent levels				Hagy, J.D., Boynton, W.R., Keefe, C.W., and K.V. Wood. 2004. Hypoxia in Chesapeake Bay, 1950-2001: Long-term change in relation to nutrient loading and river flow. *Estuaries* 27(4):634–658.
Mid- and lower Choptank River (part of the Chesapeake Bay)	Regrowth of SAV during the growing season (May–October)	Mean DIN < 0.140 mg/L (< 10 µM)	-F12	• Chlorophyll a < 15 µg/L • Mean light attenuation coefficient (K_d) < 2 m^{-1}		Stevenson, J.C., L.W. Staver, and K.W. Staver. 1993. Water quality associated with survival of submersed aquatic vegetation along an estuarine gradient. *Estuaries* 16(2):346–361.

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Chesapeake Bay Guidelines	Growth and survival of SAV in the Chesapeake Bay and its tributaries	DIN < 0.15 mg/L (mesohaline and polyhaline) (secondary requirement - diagnostic tool)	DIP < 0.02 mg/L (tidal fresh, oligo-, and polyhaline); < 0.01 mg/L (mesohaline) (secondary requirement - diagnostic tool)	Primary requirement: Minimum Light Requirement at the Leaf Surface (%) > 15 % Water Column Light Requirement (%) > 22 % (Secondary requirement: substitute for minimum light requirement at leaf surface) K_d (1/m) < 1.5 (for reference, use minimum light as standard) Chlorophyll a < 15 µg/L (secondary requirement - diagnostic tool) TSS < 15 mg/L (secondary requirement - diagnostic tool) Sediment Organics (%): 0.4–12 (habitat constraint) Vertical distribution (m): Zmax = 0.5 m + Zmin (habitat constraint) Sediment sulfide concentration (µM) < 1000 (habitat constraint) Current Velocity (cm/s): 10 < X < 100 (habitat constraint)	Chesapeake Bay Guidelines	Batiuk, R., P. Bergstrom, M. Kemp, and M. Teichberg. 2000. *Chesapeake Bay Submerged Aquatic Vegetation Water Quality and Habitat-Based Requirements and Restoration Targets: A Second Technical Synthesis*. Printed by the United States Environmental Protection Agency for the Chesapeake Bay Program, Annapolis, MD.

Study Location	Response	Nitrogen	Phosphorus	Other	Other Explanation	Citation
Maryland Coastal Bays	Maintain seagrass health	TN: 0.65 mg/L	TP: 0.037 mg/L			Wazniak, C., B. Sturgis, M. Hall, and W. Romano. 2004. Nutrient Status and Trends in the Maryland Coastal Bays. Chapter 4.1 in *Maryland's Coastal Bays: Ecosystem Health Assessment*. Maryland Department of Natural Resources. DNR-12-1202-0009. Maryland Coastal Bays Program, Annapolis, MD. Accessed January 2017. http://dnr.maryland.gov/waters/coastalbays/Documents/entire_publication.pdf.
	Eutrophic condition thresholds	TN: 1.0 mg/L	TP: 0.1 mg/L			