Eelgrass and Water Quality: A Prospective Indicator for Long Island Nitrogen Pollution Management Planning

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Summary

Seagrass forms productive habitat in near shore coastal waters that supports an abundance of wildlife and ecosystem services. The environmental requirements needed for seagrass to be healthy have been incorporated into water quality management strategies for estuaries in other states. Eelgrass, *Zostera marina*, is the principal meadow forming seagrass in New York waters and its precipitous decline is primarily attributed to deteriorating water quality. Degraded water quality in the estuaries and bays of New York are frequently a result of eutrophication which is driven by excessive nutrient enrichment, a consequence of nitrogen pollution. The relationship of eelgrass health with nitrogen can be complicated by confounding impacts of eutrophic systems, such as low dissolved oxygen stress and sulfide toxicity from high organic matter sediments. There is a consistent relationship of seagrass health with the amount of light the plant receives that is empirically supported. Water clarity parameters and their major components, such as algal biomass (represented by chlorophyll a), have been utilized in the development of water quality criteria concerning nitrogen targets. This approach could be incorporated into nitrogen action planning for Long Island but continued management should be based on area specific data because light requirements for eelgrass can vary based on other environmental gradients such as temperature and sediment properties. The spatial and temporal resolution of eelgrass assessments across New York waters are unlikely to support comprehensive quantitative analysis of response to management actions. The capacity for adaptive management that incorporates eelgrass indicators will require improved data collection strategies. There are encouraging examples of water quality improvements and the natural capacity for seagrass to recover, fostering an ecosystem shift that is inspiring. After management actions are implemented and areas with improving water quality are suitable for eelgrass, recovery may still be deterred if there is no remnant eelgrass to foster expansion. This situation would require re-introductions and applications should consider genetic diversity and environmental tolerances of eelgrass founder sources to promote resilience for successful restoration. Ultimately, reducing the severity of eutrophication impacts in our cherished coastal water ways is expected to benefit society and help our desperate eelgrass meadows.

Introduction

Seagrass forms underwater meadows that are federally designated as 'Essential Fish Habitat' for many commercially and recreationally important fish and shellfish. Eelgrass, *Zostera marina* (Fig.1), is the perennial habitat forming seagrass in New York State coastal waters that provides ecosystem services, improves the seabed, and supports coastal resilience. Estimates from historic records suggest ~200,000 acres of seagrass were in NY waters during the 1930's, while as of 2009 only 21,803 acres currently remain (NYS STF 2009). In recent decades the decline in New York seagrass has been attributed to a multiple stressor response primarily from changes in water quality, water temperature, and physical disturbances (See Fig.2 for conceptual overview of impacts on seagrass habitat).

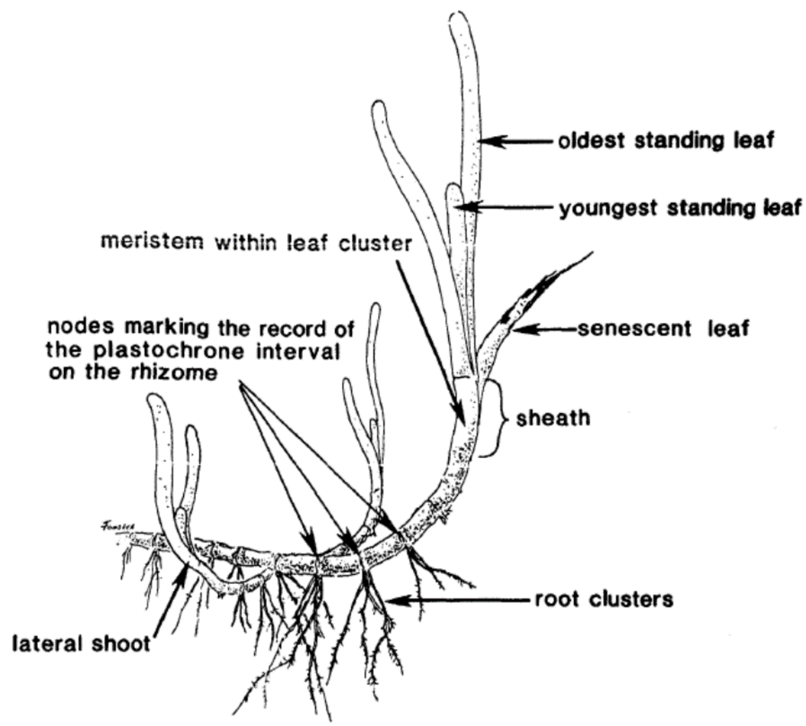

Figure 1 (from Thayer et al. 1984). Major morphological features of *Zostera marina.*

Poor water quality is considered the leading fundamental threat to eelgrass health and nutrient pollution degrades water quality. Nitrogen is a limiting nutrient in coastal marine and estuarine systems and the increase in the amount of nitrogen can lead to an overabundance of primary producers (such as algae) that dramatically changes ecosystem dynamics (Lee et al. 2004). This process of excessive nutrients driving excessive growth of algae is known as eutrophication.

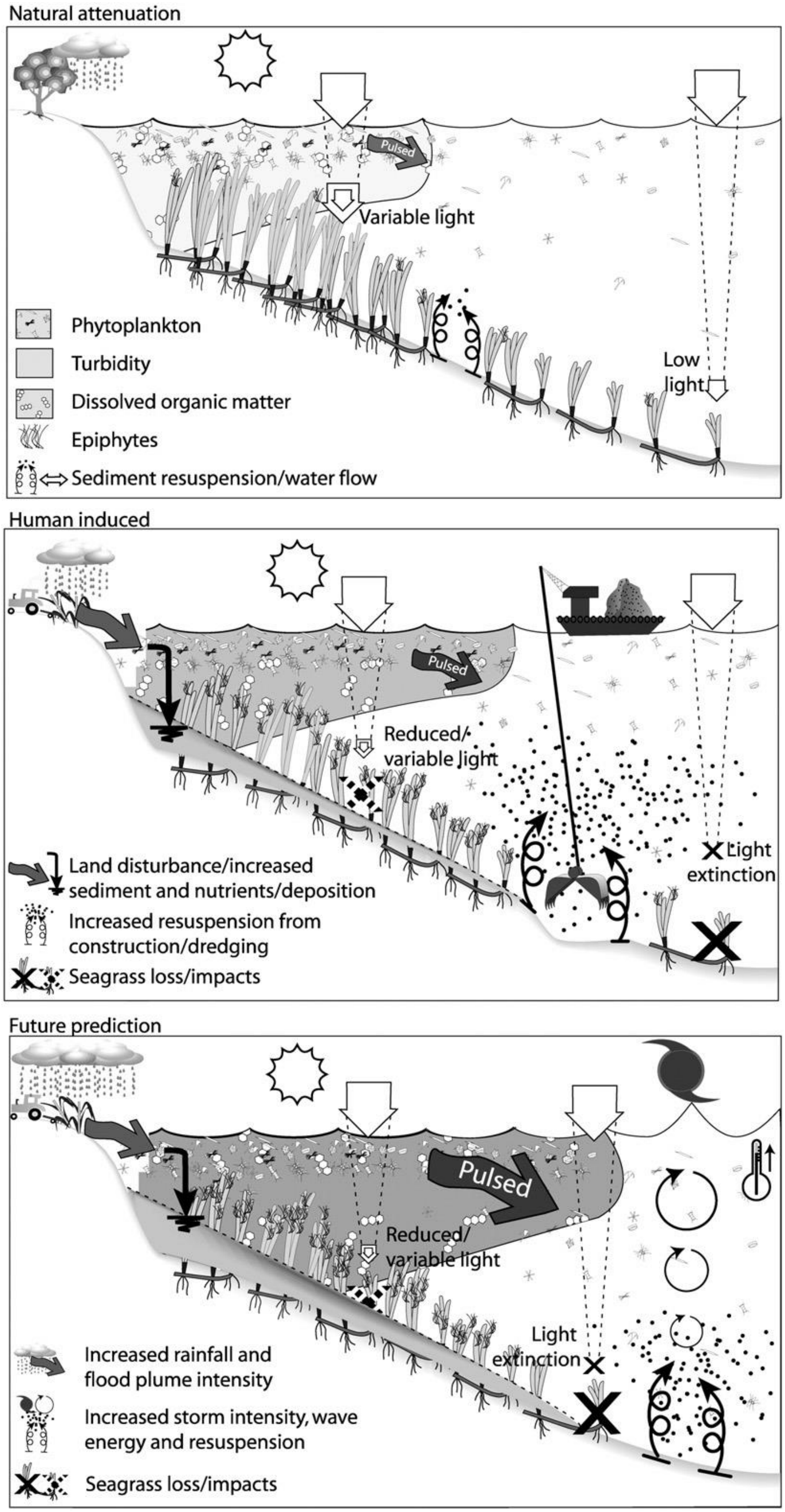

Figure 2 (From Ralph et al 2007). Conceptual model of light reduction and impact on seagrass with the following as the legend —

(A) Growth of seagrasses is strongly influenced by the optical quality of the water, including suspended sediments, dissolved organic matter and phytoplankton, which vary strongly according to run-off, as well as growth of microalgae on the leaves.

(B) Human activities, both on the land and in-water have led to changes in water quality that are reducing the light available to seagrasses.

(C) Future predictions are for a changing climate with increased sea surface temperature, more sporadic and intense run-off events and storms that will lead to periods of more sediment and nutrient loading and re-suspension.

High densities of microalgae (phytoplankton) and macroalgae (seaweed) decreases the amount of light penetrating through the water column to the seabed. The reduced water clarity and shading causes eelgrass to receive less light for photosynthesis which creates stress on the plant because it has less energy to support respiration. The reduced productivity of the seagrass makes it vulnerable to declines than can be lethal, especially when subject to concurrent stressors. Harmful algae blooms that reduce light to seagrass can have other impacts threaten animal and human health, e.g., creating toxins and driving hypoxia.

Nitrogen pollution on Long Island is not a new problem, farming practices after World War II contributed excess nutrients to the local waterways (e.g., duck farms near creeks and rivers) that promoted harmful algal blooms and hypoxic conditions. Much of that farming is now gone, but nitrogen loading into the coastal waters around Long Island are still high and this is impacting water quality. Several studies demonstrate that the largest contribution of nitrogen from land to our coastal waters is now primarily from wastewater (Gobler 2016, Kinney and Valiela 2011, Lloyd 2014, Lloyd et al. 2016). According to the population estimates for 2015 from the United States Census Bureau, Long Island NY has a population of approximately 7.8 million people which is almost 40% of the state's population. Waste disposal is a serious issue for the 18th most populous island in the World that has very high densities of people (average ~5,500 people/ sq mi). Such populated areas are often managed by sewers and wastewater treatment plants, which is more typical for western portions of the island. Heading east, many homes (estimated over 360,000: SC 2015) have septic systems, this creates a very different relationship with ground and surface water pollution.

Healthy coastal ecosystems provide goods and services that include support for recreation, tourism, and property values. Eelgrass habitat is integral to the environmental quality of coastal New York waters and its natural resources. Although there is still much to be learned concerning environmental interactions with seagrass biology, there other coastal States that have developed water quality management in relation to seagrass conservation. The preservation and potential recovery of eelgrass in New York water depends on multiple factors that will benefit from a holistic approach instead of a strategy driven by too few factors (Ganju et al. 2016).

This report aims to provide a summary of pertinent available information regarding seagrass and water quality with an emphasis on nitrogen relationships with eelgrass health. Rates of nitrogen loading, groundwater travel, and surface water flushing are important factors governing nitrogen levels but are variable and need to be quantified on a specific system basis which is not the intent of this report. A synthesis of other important consideration for eelgrass recovery is also provided. A broad first order view of indicators for water quality may provide logical common ground to develop initial management plans.

New York Eelgrass

Portions of three major estuarine systems around Long Island are currently known to accommodate eelgrass in New York waters. The 2,061 acres of eelgrass inventoried in Long Island Sound (LIS) are located toward the eastern end that benefits from tidal flushing of cooler and clearer open waters (Tiner et al. 2012). Only 421 acres of that eelgrass are in the New York portion of LIS and nearly all of it (96%) is found around Fishers Island (Tiner et al. 2012). Eelgrass in LIS normally occurs at a depth of less than 3 meters, but at Fishers Island it was found at depths up to 8 meters (NYS STF 2009).

Eelgrass was historically present throughout the Peconic Estuary between the north and south forks of Long Island, from Flanders Bay to Gardiner's Bay. Less than 1,600 acres of that habitat remained in 2000yr and was mostly in the eastern parts of the estuary where there is more influence from cooler and clearer open waters (Tiner et al. 2003). Another estuary wide eelgrass assessment was conducted in 2014yr and estimated coverage has declined to less than 1000 acres (PEP 2015).

The system of bays on Long Island's south shore, designated the South Shore Estuary Reserve (SSER), likely contain the most seagrass in New York waters. Approximately 20,015 acres occur from South Oyster Bay through Great South Bay and then into Moriches Bay and Shinnecock Bay; as estimated from an aerial survey conducted in 2002yr (NYS STF 2009). An updated system wide assessment of the SSER is needed and expected to happen soon. Widgeon grass, *Ruppia maritima*, is a shorter annual seagrass that also occurs in notable amounts in this system (Tinoco 2016). Field verification should help discern those beds from eelgrass meadows.

Nitrogen

Nitrogen is an essential nutrient for the growth and survival of living organisms. Primary productivity from photosynthesis by algae and plants is frequently controlled by the amount of nitrogen that is available to them. Most nitrogen is in the form of an inert (biologically unreactive) N_2 gas. Before the industrial and agricultural revolutions the supply of biologically available nitrogen on Earth was limited to the rate of bacterial nitrogen fixation, which is a slow and energy intensive process that resulted in many systems being nutrient limited and oligotrophic. Human activity has roughly doubled the world's average natural fixation rate (Howarth 2008) and the United States fluctuates anywhere from 5-14 times that rate (USEPA 2003).

Problems arise when there are large amounts of nitrogen in coastal waters. Algae can grow and reproduce at extremely high levels (bloom), getting dense enough to change the color of the water and restrict light to bottom dwelling organisms. Excessive blooms contribute their organic matter to respiration (decomposing) which uses oxygen, leading to hypoxia. Nitrogen enrichment is also attributed with increasing the toxins produced by harmful algal species (Hattenrath et al. 2010). Eelgrass is Submerged Aquatic Vegetation (SAV) that utilizes nitrogen, but nitrogen in large quantities can become detrimental to eelgrass and cause it to become more vulnerable to declines from other stressors.

There are three main categories of nitrogen cycle processes: nitrogen fixation, nitrification, and denitrification. Each process involves microbial mechanisms. The following is a simplified synopsis (see Stein and Klotz 2016 for more detail):

1. Nitrogen fixation is the conversion of N_2 (into ammonia/ammonium: NH_3/NH_4^+) that can be assimilated into biomass (becoming organic nitrogen) or respired further by microbes.
2. Nitrification is the oxidation of ammonia (NH_3) to nitrite (NO_2^-), or to nitrate (NO_3^-), and of nitrite (NO_2^-) to nitrate (NO_3^-). This process results in more biologically usable forms of fixed nitrogen (e.g., nitrate) becoming available for primary productivity.
3. Denitrification is the reverse anaerobic (without oxygen) pathway that ends up back at the inert N_2 gas.

Eelgrass can obtain nitrogen (ammonium and nitrate) from the sediment through their roots and from the water column through their leaves (Short and McRoy 1984). A shift from reliance on sediment to the overlying waters for nitrogen has been observed under water column nutrient enrichment (Touchette and Burkeholder 2000). As nutrient loading to the water column progresses, it can inhibit seagrass growth and survival as an indirect effect by stimulating algal overgrowth and causing reductions in light, but also as direct physiological effects which are exacerbated by elevated temperatures and light reductions. Ammonia toxicity has been reported in Z. marina (125 μM* water-column NH_{4+}, 5 weeks: Touchette and Burkeholder 2000). Ammonium/ammonia becomes enhanced in sheltered eutrophic estuaries due to the decomposition of organic matter from phytoplankton and seaweed, but also from anthropogenic wastewater sources (Van der Heide et al. 2008). Direct effects from elevated water column nitrate have also been reported in *Z. marina*, related to the high energy demands of sustained nitrate uptake and subsequent high internal carbon demands leading to a 'carbon drain' that weakens eelgrass tissue and causes the belowground biomass to deteriorate which leads to mortality (Burkholder et al. 1992, 1994). A water column concentration of nitrate at 5μM/day caused the percent carbon in seagrass to decrease and the meristem began to crumble, followed by death within 2 weeks (Burkholder 1994). Anthropogenic pollution from wastewater and fertilizers contributes nitrate to coastal waterways (Jadhao 2013). Ambient water nitrate concentrations in estuaries around Long Island range from 1-5μM year round, with locally higher concentrations near areas experiencing nutrient rich submarine groundwater discharge, high run-off or sewage outfalls (NYS STF 2009).

A study of eelgrass along nutrient gradients in three New England estuaries determined a negative relationship of leaf nitrogen content to area normalized leaf mass (Lee et al. 2004), which can provide an early indicator of over enrichment, a 'Nutrient Pollution Indicator' (NPI). Kennish and Fertig (2012) applied the 'NPI' in a New Jersey estuary and observed temporal variability but not spatially, which lead to the impression that the metric was unreliable for discerning a eutrophic gradient. A recent review of seagrass environmental indicators found that very few were specific to a single stressor and ratios, such as carbon to nitrogen, can provide useful information but are highly variable and influenced by other factors, such as available light, which complicates their application as criteria (Roca et al. 2016).

A study of 62 different estuaries in New England discovered that eelgrass health and survival were at considerable risk if nitrogen loading levels were over 50 Kg/ha/yr, and at lower levels it is likely controlled by other ecosystem factors unrelated to water quality (Latimer and Rego 2010). Benson et al. (2013) studied 70 different sites in 19 different Massachusetts estuaries through 4 growing seasons (2007-2009, 2011), found healthy eelgrass existed where tidally-averaged total nitrogen was less than 0.34 mg/L, "equivalent to a mid-ebb tide water-column total nitrogen of <0.37 mg/L". They also used eelgrass transplants as a bio-indicator of habitat quality and survival across sites decreased as total nitrogen levels increased. Sites that had >75% transplant success had average total nitrogen levels of 0.39mg/l. Recommended SAV habitat criteria

* Unit conversion (according to http://ocean.ices.dk/Tools/UnitConversion.aspx)
μM = μmol/l, μmol/l × MW = μg/l, MW of nitrogen NO_3 = 62.005010,
1 μM NO_3 = 62.005010 μg/l = 0.062 mg/l

derived from analysis of Peconic Estuary data was a mean summer (June-August) condition of 0.4 mg/L (PEP 1998).

See Appendix A: Table 1 for nutrient reference values.

Light and Clarity

Seagrasses can require more light to sustain growth than terrestrial plants and other photosynthesizing marine autotrophs (Dennison et al. 1993). This is related to the large respiratory demand of root structures that are often subject to anoxic or hypoxic sediments (Goodman et al. 1995). Eelgrass in particular can require much more light, over 20% to even 37% surface irradiance to survive (see Appendix A: Table 2 for light and clarity reference criteria). Studies have shown that *Z. marina* may be less efficient with its use of absorbed light as compared to other species (Ralph et al. 2007).

Low light levels can affect the ability of seagrass to fix carbon and eelgrass needs high light levels, especially in the spring, to generate the carbohydrate reserves that it relies upon most of the year (Burke et al. 1996). Low light or shading causes reliance on stored carbohydrates to maintain metabolic processes, this leads to depletion when respiratory burdens persist (Ralph et al. 2007). High nitrogen and reduced light conditions both create respiratory demands of eelgrass that deplete it of life sustaining forms of carbon that is synergistically dangerous. Additional stress for respiratory demands can contribute to the risk of failure, e.g.: high temperature and sediment sulfide. “Eutrophication can both decrease the light available to seagrasses, while concurrently increasing their effective light requirements” (Goodman et al. 1995).

Models that relate secchi depths to maximum depth limits for eelgrass growth frequently have a minimum light requirement of 11% surface irradiance but Ochieng et al. (2010) provided evidence that 11% is insufficient for long term eelgrass survival and at 34% surface irradiance or less, eelgrass had significant reductions in mass. Benson et al. (2013) reported eelgrass loss when <20% surface light irradiance reaches the bed, but eelgrass persisted at >25% surface light levels. Chesapeake Bay water clarity criteria for shallow-water bay grass, which includes eelgrass, is ~22% of surface light through the water column and the secchi depth for this criteria has a large range from 0.2-1.9 meters (USEPA 2003). Secchi depth is a basic and useful measure that is easy to obtain but work by Carstensen et al. (2016) illustrates the need to verify model assumptions of this relationship with surface irradiance through actual measurements of the light attenuation coefficient.

Light attenuation coefficient (Kd) is a parameter used to characterize the amount of light that travels through the water to depth. Most marine SAV habitats have a Kd range of 10-30% (Kemp et al. 2004). Kd is mainly determined by two components, chlorophyll *a* (proxy measure for phytoplankton biomass) and Total Suspended Solids (TSS) (USEPA 2003). Protective chlorophyll *a* (Chl-a) concentrations considered to be conducive of good water clarity were developed for different salinity regimes in the Chesapeake Bay, e.g., 8 μg/L for mesohaline and polyhaline (USEPA 2003). A factor often overlooked when evaluating the amount of light that the plant receives is losses due to material directly on the leaves of plants, such as: algae, bacteria, sediment, and detritus. This factor can be expressed by the epiphytic light attenuation coefficient (Ke), which increases exponentially with epiphyte biomass (USEPA 2003). Kemp et al. (2004) developed formulas to calculate the percent light through the water to SAV depth and percent light at leaf (see Fig. 3) from the attenuation coefficients that require area specific water column and epiphytic community monitoring data. Measures of epiphyte load have been evaluated for use as an indicator of coastal nutrient impact and threshold ranges were able to be derived from seagrass response metrics (Nelson 2017).

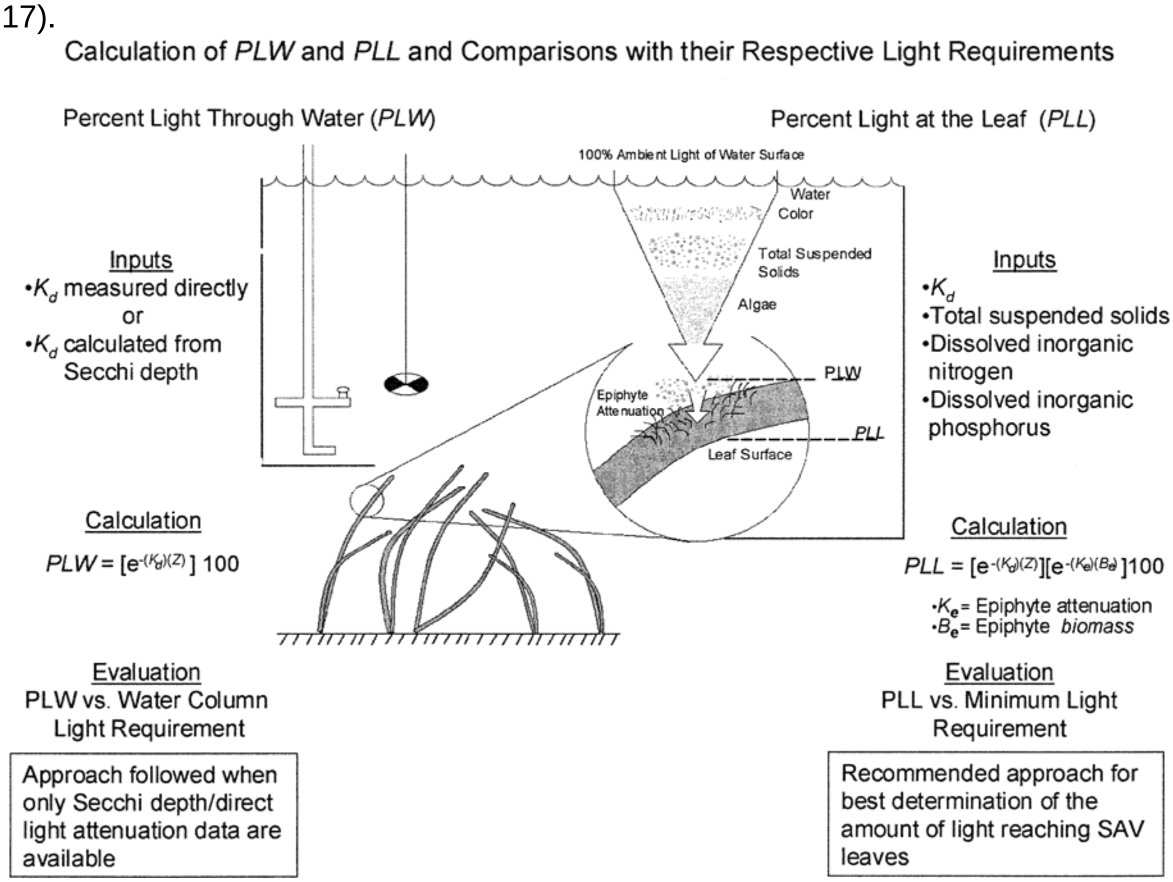

Figure 3. (From Kemp et al. 2004) Conceptual representation of how percent surface light through the water (PLW) and percent surface light at the leaf (PLL) are calculated and how these parameters are used to evaluate a site as a potential habitat for SAV.

See Appendix A: Table 2 for light and clarity reference values.

Algae

Under low to moderate nutrient conditions, eelgrass stays competitive with other photoautotrophs because they can uptake nutrients from above and below ground biomass. Algae generally have a higher affinity for nitrogen, so as water column nutrients become more abundant, phytoplankton and macroalgae can out-compete seagrass and their blooms diminish the available light to the beds (Nelson 2009). Hauxwell et al. (2001) found an exponential decrease in aboveground production of eelgrass as the macroalgal canopy height increased, identifying an approximate 9-12 cm critical height of the macroalgal canopy at which eelgrass declines. Dense phytoplankton blooms can severely reduce light in the water column and Brown Tide (*Aureococcus anophagefferens*) events have had major impacts on the health of New York eelgrass beds (NYS STF 2009). Light limitations are particularly detrimental to eelgrass seedlings and new shoots since they will only germinate or grow in an optimal range of conditions. Spring is the time of year that eelgrass in the northeast builds carbohydrate reserves that it relies on during other times of the year, shading from algal blooms can limit the ability of eelgrass to produce those critical reserves (Burke et al. 1996).

Nelson and Lee (2001) suggest setting seagrass protective water quality criteria to standards that minimize the response of algae, since the primary response to anthropogenic nutrient loading is mediated through algal blooms, which in turn impacts seagrass. Other than shading, algae blooms add organic matter to the sediment as they sink and decompose, promoting the decline of oxygen which subsequently increases the risk of the grass being subject to sulfide toxicity (Hoffle et al. 2011).

Chlorophyll *a* (chl-a) is the primary type of photosynthetic pigment in algae, therefore water column measurements of chl-a are used as a representative proxy for the amount of phytoplankton present. In the Chesapeake Bay, chl-a concentrations of 8 μg/L (mesohaline > 5-18 ppt and polyhaline >18 ppt) were considered protective against negative water clarity effects, given the range of total suspended solids (TSS) concentrations of 10 to 15 mg/l which were identified as habitat requirements for bay grasses at the 1-meter water depth application (USEPA 2003).The Chesapeake Bay program cautioned that assumptions and assignment of criteria should be made on a bay segment basis because values will vary on temporal and spatial scales. In some areas of the estuary the chl-a/algal biomass component of the total light attenuation was found to be minor as compared with non-algal solids (USEPA 2003).

Investigations across southeastern Massachusetts estuaries by Benson et al. (2013) found a clear positive relationship observed between total nitrogen, chl-a/phytoplankton biomass, particulate organic matter, and light attenuation in the water column. Light levels at the bottom were directly related to losses in eelgrass coverage and lower survival of transplanted eelgrass shoots. The long-term (2000-10yr) average chl-a concentration was 5.1µg/L in eelgrass areas that were categorized as currently supporting healthy/stable beds, versus degraded or lost beds. Reviews of nutrient input impacts on aquatic systems found that coastal marine waters with average chl-a > 5µg/L displayed hypertrophic conditions, indicating greatly excessive nutrient inputs (Håkanson 1994, Smith et al. 1999). Benson et al. (2013) point out that unlike conditions that can occur in the large riverine estuaries like the Chesapeake Bay, the shallow estuaries in their study tend to be groundwater dominated with little inorganic sediment loading and TSS, resulting in light attenuation primarily from in situ phytoplankton.

Temperature

Temperature directly influences the metabolism of eelgrass, respiration increases as temperatures increases, which has a large impact on the plant's physiology. The ratio of photosynthesis to respiration decreases dramatically for eelgrass at higher temperatures, which can lead to a carbon loss for the plant instead of a carbon gain (Marsh et al. 1986). Respiration in the leaves can increase as much as 10-20x the normal rate if temperatures exceed normal seasonal maximums (Marsh et al. 1986). A study of eelgrass beds over eight years in the Chesapeake Bay by Moore et al. (2014) found exposure to rapidly increasing temperatures during the summer months, 4–5 °C above normal (25-30℃) over two weeks, can result in widespread diebacks (e.g., 97% decline in 2010). Above 30℃ has been shown to be detrimental to eelgrass by impairing enzyme activities (Lambers 1985, Marsh et al. 1986, Zimmerman et al, 1989).

Elevated temperatures can cause eelgrass to have other negative interactions, including with nutrient pollution as observed in the exacerbated effects with water column nitrate (Burkholder et al. 1992). The Pacific Northwest coast of the U.S. has relatively cool waters (10℃ annual mean) and *Zostera marina* is able to survive in many places where nitrogen is 3-10x levels shown to have direct nutrient toxicity in areas on the east coast that can reach up to 30℃(Nelson 2009).

Bacterial metabolism also increases with rising temperatures, in sediment it can be 2-3x for each 10℃ increment (Thamdrup et al. 1998, Sand-Jensen et al. 2007). Hydrogen sulfide can become toxic to eelgrass as it accumulates from an end product of anaerobic (without oxygen) microbial respiration in the sediment (Goodman et al. 1995). Sulfur content was significantly higher (up to 34x) in eelgrass rhizomes at 27°C as compared to 18°C or 21°C (Hoffle et al. 2011). Warm coastal waters during the summer around Long Island are subject to depleted oxygen (hypoxia), resulting in the combined threats of respiratory stress and sulfur toxicity. Experimental studies by Pulido and Borum (2010) induced eelgrass mortalities that demonstrated a large negative impact of high temperature with depleted oxygen.

Sediment

Seagrasses are rooted plants and therefore sediment is a critical part of its environment. Seagrasses can acquire nutrients through the roo15ts and leaves; relative uptake from either route appears dependent on availability and environmental conditions (Nelson 2009). The majority of nitrogen uptake can occur through the leaves (Hemminga et al. 1994) but algae have a higher affinity for water column nutrients, so the ability to access sediment nutrients allows seagrass to persist and sometimes provides an advantage since marine sediments frequently have higher nutrient levels than the water column (Nelson 2009).

Coarser sediments tend to be more oxygenated with lower amounts of organic matter and seagrass are noted to be more abundant in sediments with less than 20%-30% silts and clays by weight, and generally absent in sediments with organic content >5% or 2mM of porewater sulfide (Kemp et al. 2004). Around Long Island, seagrass meadows are generally not observed in sediment with >2% organic matter (Peterson 2016, Pickerell 2016) and restoration projects are recommended more conservative threshold values of <0.5% organic matter content (Vaudrey et al. 2013).

Increased organic matter loading, and the subsequent decomposition in the sediment creating sulfur compounds, can increase the sulfur concentration in seagrass roots and rhizomes (Roca et al. 2016). Sulfur occurs naturally in sediment but high sulfide concentrations become toxic to plants (a phytotoxin) and inhibits enzymes, disrupting adenosine triphosphate (ATP) synthesis which is essential for cellular energy (Raven and Scrimgeour 1997, Pedersen et al. 2004). Respiration requires oxygen, which gets depleted as more organic matter is supplied to the sea bed (Diaz and Rosenburg 2008). Anaerobic reactions (without oxygen) contribute to the conversion of sulfate to the more toxic sulfides (Goodman et al. 1995). High sulfide levels can reduce seagrass growth and lead to mortality (Hoffle et al. 2011). Experiments have shown toxic effects to eelgrass at sulfide concentrations >400µM (Goodman et al. 1995).

Seagrass provides oxygen from photosynthesis to its root structure which alleviates anaerobic conditions and can mitigate sulfide toxicity (Caffrey and Kemp 1991). This remedy may be ineffective when photosynthesis is insufficient due to shading. Reduced water clarity is often accompanied with the eutrophic conditions that promote sediment sulfide; organic matter loading and anaerobic respiration. Research in Great South Bay revealed compounding effects of shading from algal blooms that can lead to seagrass death at sulfide concentrations around 300µM (NYS STF 2009). At night, when photosynthesis has ceased, the health of the plant relies on oxygen concentrations from the water column which is consequently when hypoxic or anoxic conditions are most likely to occur (Hoffle et al. 2011).

See Appendix A: Table 3 for sediment reference values.

Protective and Restorative Criteria

The success rate for seagrass restoration is low (<40%) across the globe (Katwijk et al. 2016). Many projects have used conditions of existing seagrass beds to guide selection of restoration sites. Leschen et al. (2010) found this approach was severely insufficient and their successful sites were selected only after additional high resolution ground-truthing and test transplanting. Well established eelgrass beds have positive feedback mechanisms from below ground biomass (roots and rhizomes) that provide physical resistance to disturbance and erosional forces, and from above ground biomass that promotes particle settling which reduces turbidity and increases clarity, and through photosynthetic productivity that builds their energy reserves and supplies oxygen to the sediment. The density of shoots can be an important factor in eelgrass meadow resiliency as demonstrated in the greater ability of higher density beds to take up nitrogen and avoid any related toxic impacts, than beds with lower shoot densities (Van der Heide et al. 2008). There are other biotic factors of seagrass ecosystems that support meadow stability, e.g.; invertebrate grazers can play an important role in controlling algae growth in shallow benthic systems (Heck and Valentine 2007), which can alleviate the burden of epiphytes on eelgrass and increase the amount of light received by the leaves.

Peconic Estuary data analysis derived a value of 0.4 mg/L total nitrogen as a threshold for eelgrass preservation (PEP 1998) and an extensive survey of Massachusetts estuaries (Benson et al. 2013) found that eelgrass was healthier in locations with a threshold of total nitrogen <0.34mg/L (tidally averaged). Since established beds are more tolerant, these values may not be conducive for natural recovery and restoration efforts as even better conditions are likely needed for re-colonization.

A technical synthesis of water quality and habitat requirements of SAV in the Chesapeake Bay derived threshold values (e.g., <0.15 mg/l of dissolved inorganic nitrogen) for restoration targets within salinities appropriate for *Zostera marina* (Batiuk et al. 1992; Dennison et al. 1993). After a few years of implementation, the habitat requirements were considered to have some limitations and a subsequent technical synthesis was conducted (Batiuk et al. 2000). A primary deficiency was not considering light attenuation at the leaf surface, which can be considerable when there is high epiphytic growth and sediment accumulating on leaf. A new algorithm was developed as a quantitative tool to apply water quality data to define site specific SAV habitat suitability (see Fig. 3).

The Long Island Sound Habitat Restoration Initiative has suggested Water Quality Criteria for eelgrass (LISS 2003), which includes less than .03 mg/L dissolved inorganic nitrogen, based upon a study of three seagrass sites in LIS by Koch et al. (1994). Conservative values were chosen based on work that emphasized regenerating eelgrass beds require better conditions than for simply maintaining existing beds (Okubo and Slater 1989).

Studies in the Chesapeake show some areas where light was considered adequate but SAV was still lacking, may have additional constraints resulting from physical (e.g., wave energy, currents) and geo-chemical (e.g., sediment organic matter) conditions (Koch 2001). Figure 4 shows a conceptual model of interactive seagrass habitat requirements. An eelgrass habitat suitability model developed for LIS used criteria to account for light, sediment, dissolved oxygen and temperature conditions (Vaudrey et al. 2013). A seagrass habitat model developed for Narragensett Bay, RI, used strategies to incorporate interactive effects, including sediment organic carbon with light requirements (Detenbeck and Rego 2015).

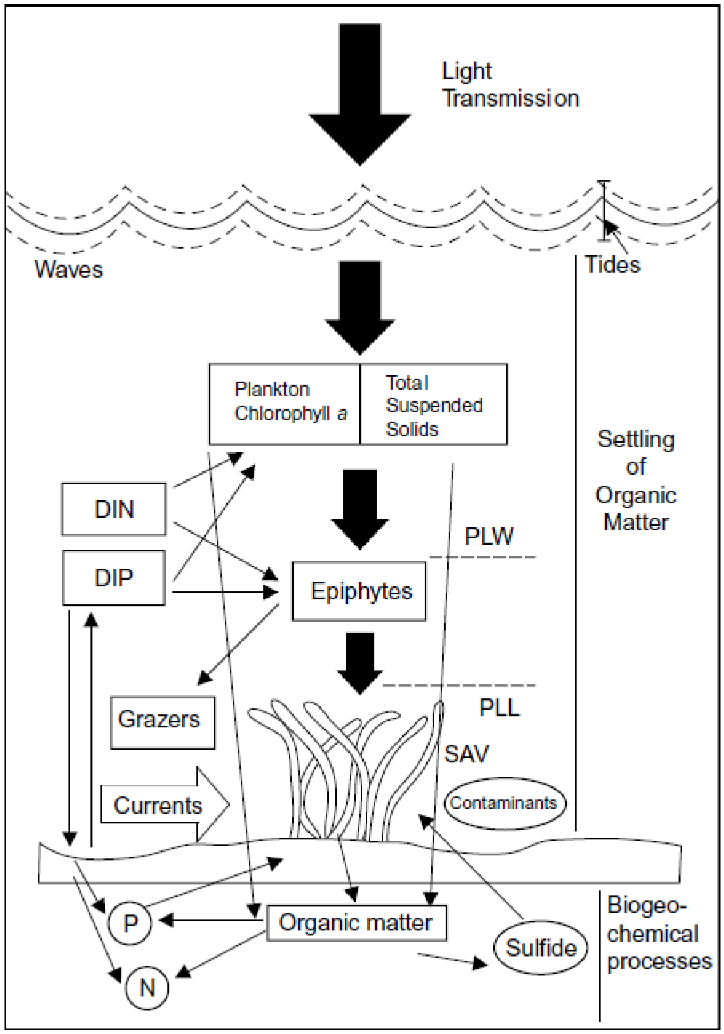

Figure 4. (From Batiuk et al. 2000) **Interaction between Light-Based, Physical, Geological and Chemical SAV Habitat Requirements.** Interaction between previously established SAV habitat requirements, such as light attenuation, dissolved inorganic nitrogen (DIN), dissolved inorganic phosphorus (DIP), chlorophyll a, total suspended solids (TSS) and other physical/chemical parameters discussed: waves, currents, tides, sediment organic matter, biogeochemical processes. P = phosphorus; N = nitrogen; PLW = percent light through water; PLL = percent light at the leaf.

See Appendix A: Table 4 for physical reference values.

Discussion

Seagrass meadows are declining globally at an estimated 7% per year (Waycott et al. 2009). Water quality is critically important to the health of seagrass; both have deteriorated under wide spread anthropogenic eutrophication. Nearly two decades ago Bricker et al. (1999) reported that nearly all US estuarine waters exhibited some eutrophic conditions. Excessive algae growth under eutrophic conditions reduces light that seagrass receives, which limits the plants productivity and its resilience to other environmental stressors. The amount of light required to maintain healthy seagrass varies in response to other environmental conditions, e.g., eelgrass has been shown to have higher light requirements corresponding with higher sediment organic matter content (Kenworthy et al. 2014). Excessive nitrogen, primarily through groundwater, drives eutrophication that impairs coastal water quality around Long Island which impacts habitat, resiliency and the economy (LINAP 2016). Reducing nitrogen inputs to our coastal water bodies should provide relief from eutrophic driven impacts.

There are encouraging stories of nutrient reductions and eelgrass recovery. The diversion of municipal treated wastewater discharge out of Mumford Cove (48ha embayment in CT) allowed for a natural recovery from a macroalgae (*Ulva*) dominated system back to patchy and moderately dense beds of *Z. marina* across nearly half of the cove (Vaudrey et al. 2010). It should be noted that this is not a very large bay and it took nearly fifteen years to observe this type of system change, but systems with long term exposure to extremely eutrophic conditions may need extensive time for recovery, especially if an ecological tipping point had been surpassed (Nelson 2009). Water quality impacts the benthos, e.g., delivery of particulate organic matter, but the response expected following management actions is likely to be much faster in the water column relative to changes exhibited in the benthos (Vaudrey et al. 2013).

A large recovery of seagrass has been achieved after nutrient loading reductions were implemented in Tampa Bay FL. Their technical approach included empirical regressions of chl-a concentrations and light attenuation determined for sustainable seagrass growth, followed by empirical relationships developed between nitrogen loading and chl-a concentrations. This approach does have potential to be applied to other systems and a similar strategy was utilized in the Chesapeake Bay (Batiuk et al. 1992, 2000, USEPA 2003). Clarity - chlorophyll *a* relationships represent a first order response to nutrient enrichment that has fundamental influence on light in the water column essential for seagrass health. Excessive algae blooms also have impacts that degrade our coastal waters and discourage public enjoyment from noticeable changes in color (e.g.; brown tide, rust tide), to driving hypoxia and fish kills, or more seriously from harmful toxin production. Although there is knowledge to be learned from the process Tampa Bay employed, there are a few fundamental differences that impede realistic expectations of reproducing the same dynamics in New York, e.g.; they have several species of seagrass and none of them are *Zostera marina. See Appendix B for a more detailed case study of the progress that was made in Tampa Bay*.

Habitat requirements and environmental parameters for healthy seagrass have been used as a tool to develop criteria for water quality management (Greening et al. 2014, Howes et al. 2003, Trowbridge 2009, USEPA 2003). Published values regarding these relationships can inform initial estimates to start developing new water quality planning focused on nitrogen pollution management (examples in App. A). Conservative thresholds are recommended when considering that stable eelgrass beds have positive feedback mechanisms and are more tolerant of environmental conditions than eelgrass in need of recovery. Maintaining a long term water quality management strategy strictly on published values is not recommended. There are interacting parameters that influence seagrass health, for example, light requirements can vary dependent on sediment condition and water temperature. Factors that control the amount of light seagrass receives can vary, in the water column (components of turbidity beside phytoplankton), in the benthos (macroalgae canopy), and directly on the grass leaves (epiphytes). A review of numeric nutrient criteria development by Bierman et al. (2014) attests to the multiple factors that should be considered to properly characterize relationships with eelgrass across an estuary. Considerable subject reviews of technical literature regarding seagrass environmental relationships are provided by Nelson (2009) and Vaudrey (2008).

Climate change can already be observed in New York and the predicted changes in coastal water temperatures are expected to increase the vulnerability of its marine ecosystems to other stressors (Rosenzweig et al. 2011). Recent temperature trends in the Peconic Estuary are considered to be a significant barrier to eelgrass restoration efforts (Pickerell 2016). Decades of monitoring in the Chesapeake Bay shows gradually reduced eelgrass cover with declining water clarity and more recent episodes of eelgrass loss are attributed to an emergent interaction with warming temperatures (Lefcheck et al. 2017).

The relationships between eutrophication related stressors and ecosystem response are complex, but there is an empirically supported and consistently documented connection between clarity and seagrass (Krause-Jensen et al. 2008). Accounting for other factors (e.g., temperature and sediment) and quantifying their effects is encouraged to improve the predictive power of the empirical relationships (Krause-Jensen et al. 2008). Another confounding factor that is not quantified are impacts from herbicides and pesticides to eelgrass and the animals that graze epiphytes (NYS STF 2009).

Monitoring is fundamental to assess responses to initial management actions and to provide support for adaptive management that includes refining criteria and metrics appropriate to target areas (Batiuk et al. 2000, Lefcheck et al. 2014). Seagrass monitoring has mostly been insufficient across New York waters to support quantitative analysis of eelgrass dynamics and trends. Assessments consist primarily of estuary wide surveys through interpretation of aerial photography. Surveys conducted for the Long Island Sound Study have occurred with some regularity (3-4yr intervals: Tiner et al. 2012) and there are plans to conduct another survey this year. The Peconic Estuary went 14yrs in between aerial surveys (2000 to 2014). There are plans to survey the South Shore Estuary Reserve this year, but it has not been mapped since 2000. Seagrass meadows are dynamic and this deficiency in estuary survey frequency was cited in the 2009 New York Seagrass Task Force Report. Maps from these aerial image based surveys provide estimates of coverage with some rough determinations of relative seagrass density. The Peconic Estuary Program has supported sentinel bed monitoring based on shoots counts by divers (Pickerell and Schott 2016), which is a higher resolution determination of eelgrass density that serves as a primary metric for the plant's health (Roca et al. 2016). Their bed monitoring has been done on an annual basis since 1997, representing the best eelgrass data set reported for NY waters. Ideally, eelgrass and water quality metrics should be coupled spatially and temporally to provide confident determinations of their relationships on an area specific basis. Extensive long term monitoring of seagrass and water quality is cited as one of the primary elements for programmatic success in Tampa Bay (Greening et al. 2014). Eelgrass monitoring and assessment that has occurred across coastal NY waters can be examined for water quality analysis but in many cases it is unlikely to have the resolution needed for any substantial rigor. There will need to be a more resolute data collection effort to allow for applied analytic capacity needed to support continued planning decisions and management actions.

Seagrasses have shown natural capability for recovery after water quality improvements are obtained, but meadow resurgence may not be possible if areas are devoid of seagrass, with no adjacent vegetative shoots or sufficient influx of seeds. The situation of improving environment but no seagrass sources will require restorative actions to supply a new foundation of vegetative shoots, and/or seeds. Restoration strategies should consider genetics of the seagrass used as a source for planting activities in order to promote resilience of a recovering meadow through diversity (Unsworth et al. 2015). Research showing differences in the tolerance of regional eelgrass populations to low light and warm temperatures could help guide selection of donor sources (Short et al. 2012). Planting of clonal vegetative shoots may be needed to jump start a new bed form but use of seeds has greater potential for genetic diversity and possibly for adaptive (phenotypic) response (Reynolds et al. 2012). Successful large scale recovery of eelgrass was initiated in coastal Virginia with seed based strategies (Orth et al. 2012). Approaches and techniques for restoration should be chosen based on site specific considerations and a combination of applications may provide the most comprehensive benefits.

The goal of healthy seagrass is a dignified endpoint considering the ecosystem services, such as wildlife habitat and near shore stability, it provides to coastal communities. In order to get there, we need to manage points along the way, i.e.; reductions leading to first order responses and subsequent longer term system changes. This will require improved monitoring and assessment to support the adaptive management needed to arrive at the restored functioning coastal ecosystems that healthy seagrass meadows represent. Ultimately, reducing the severity of eutrophication will improve habitat, environmental quality, and the public’s ability to enjoy our cherished coastal water ways.

References

- Batiuk, R.A., P. Bergstrom, M. Kemp, E. Koch, L. Murray, J.C. Stevenson, R. Bartleson, V. Carter, N. Rybicki, J. Landwehr, C. Gallegos, L. Karrh, M. Naylor, D. Wilcox, K. Moore, S. Ailstock, and M. Teichberg. 2000. Chesapeake Bay submerged aquatic vegetation water quality and habitat-based requirements and restoration targets: a second technical synthesis, 205. Annapolis: United States Environmental Protection Agency.

- Batiuk, R. A., R. Orth, K. Moore, J. C. Stevenson, W. Dennison, L. Staver, V. Carter, N. Rybicki, R. Hickman, S. Kollar and S. Bieber. 1992. Chesapeake Bay Submerged Aquatic Vegetation Habitat Requirements and Restoration Targets: A Technical Synthesis. CBP/TRS 83/92. U.S. EPA Chesapeake Bay Program, Annapolis, Maryland.

- Benson, J. L., Schlezinger, D., & Howes, B. L. 2013. Relationship between nitrogen concentration, light, and Zostera marina habitat quality and survival in southeastern Massachusetts estuaries. *Journal Of Environmental Management*, *131*129-137. doi:10.1016/j.jenvman.2013.09.033

- Bierman, Victor J.; Diaz, Robert J.; Kenworthy, W. Judson; and Reckhow, Kenneth H. 2014. "Joint Report of Peer Review Panel for Numeric Nutrient Criteria for the Great Bay Estuary New Hampshire Department of Environmental Services June, 2009". Additional Regional Technical Reports & Resources. Paper 1. http://scholars.unh.edu/rtr/1

- Bricker, S.B., Clement, C.G., Pirhalla, D.E., Orlando, S.P. and Farrow, D.R., 1999. *National estuarine eutrophication assessment: effects of nutrient enrichment in the nation's estuaries*. US National Oceanographic and Atmospheric Administration, National Ocean Service, Special Projects Office and the National Center for Coastal Ocean Science.

- Burke K. Marianne, Dennison C. William, Moore A. Kenneth (1996) Non-structural carbohydrate reserves of eelgrass *Zostera marina*. *Marine Ecology Progress Series* Vol. 137: 195-201

- Burkholder JoAnn M., Glasgow Howard B. Jr., Cooke Jacob E., (1994) Comparative effects of water-column nitrate enrichment on eelgrass *Zostera marina*, shoalgrass *Halodule wrightii*, and widgeongrass *Ruppia maritime*. *Marine Ecology Progress Series* Vol. 105: 121-138

- Burkholder JoAnn M., Mason Katherine M., Glasgow Howard B. Jr. (1992) Water-Column nitrate enrichment promotes decline of eelgrass *Zostera marina*: evidence from seasonal mesocosm experiments. *Marine Ecology Progress Series* Volume 81:163-178

- Caffrey, J.M., Kemp, W.M., (1991) Seasonal and spatial patterns of oxygen production, respiration and root-rhizome release in *Potamogeton perfliatus* L. and *Zostera marina* L..

Aquat. Bot. 40, 109–128.

- Carstensen, J., Krause-Jensen, D., & Balsby, T. 2016. Biomass-Cover Relationship for Eelgrass Meadows. *Estuaries & Coasts, 39*(2), 440-450. doi:10.1007/s12237-015-9995-6

- Dennison, W.C., Orth, R.J., Moore, K.A., Stevenson, J.C., Carter, V., Kollar, S., Bergstrom, P.W. and Batiuk, R.A. 1993. Assessing water quality with submersed aquatic vegetation: habitat requirements as barometers of Chesapeake Bay health. *BioScience*, 43: 86-94.

- Detenbeck, N. and S. Rego. 2015. Predictive Seagrass Habitat Model. U.S. Environmental Protection Agency, Washington, DC. EPA/600/R-15/003.

- Diaz, R.J. and Rosenberg, R., (2008) Spreading dead zones and consequences for marine ecosystems. *Science*, *321*(5891), pp.926-929.

- Duarte, C.M., 1991. Seagrass depth limits. Aquat. Bot. 40 (4), 363-377.

- Ferguson R, Wood L, Graham D. (1993) Monitoring spatial change in seagrass habitat with aerial photography. *Photogrammetric Engineering & Remote Sensing*. 59:1033-1038.

- Ganju, N. K., Brush, M. J., Rashleigh, B., Aretxabaleta, A. L., Barrio, P., Grear, J. S., & ... Vaudrey, J. P. 2016. Progress and Challenges in Coupled Hydrodynamic-Ecological Estuarine Modeling. *Estuaries And Coasts*, (2), 311. doi:10.1007/s12237-015-0011-y

- Gobler, C. 2016. Long Island South Shore Estuary Reserve Eastern Bays Project: Nitrogen Loading, Sources and Management Options. *Prepared by Stony Book University for New York State Department of State.*

- Goodman L. Jill, Moore A. Kenneth, Dennison C. William. 1995. Photosynthetic responses of eelgrass (*Zostera marina L.*) to light and sediment sulfide in a shallow barrier island lagoon. *Aquatic Botany 50 37-47* 0304-3770/95/809.50 SSDI 0304-3770(94)00444-7

- Greening, H. and Janicki, A., 2006. Toward reversal of eutrophic conditions in a subtropical estuary: Water quality and seagrass response to nitrogen loading reductions in Tampa Bay, Florida, USA. *Environmental management*, *38*(2), pp.163-178.

- Greening, H., Janicki, A., Sherwood, E.T., Pribble, R. and Johansson, J.O.R., 2014. Ecosystem responses to long-term nutrient management in an urban estuary: Tampa Bay, Florida, USA. *Estuarine, Coastal and Shelf Science*, *151*, pp.A1-A16.

- Håkanson, L., 1994. A Review on Effect-Dose-Sensitivity Models for Aquatic Ecosystems. *Internationale Revue der Gesamten Hydrobiologie und Hydrographie*, *79*(4), pp.621-667.

- Hattenrath, T.K., Anderson, D.M. and Gobler, C.J., 2010. The influence of anthropogenic nitrogen loading and meteorological conditions on the dynamics and toxicity of Alexandrium fundyense blooms in a New York (USA) estuary. Harmful Algae, 9(4), pp.402-412.

- Hauxwell, J., Cebrian, J., Furlong, C., Valiela, I., 2001. Macroalgal canopies contribute to eelgrass (Zostera marina) decline in temperate estuarine ecosystems. Ecology 82, 1007–1022.

- Heck, K.L., and J.F. Valentine. 2007. The primacy of top-down effects in shallow benthic ecosystems. Estuaries and Coasts 30: 371–381.

- Hemminga, M., Koutstaal, B., Van Soelen, J., & Merks, A. 1994. The nitrogen supply to intertidal eelgrass (Zostera marina).

- Höffle, H., Thomsen, M.S. and Holmer, M., 2011. High mortality of Zostera marina under high temperature regimes but minor effects of the invasive macroalgae Gracilaria vermiculophylla. Estuarine, Coastal and Shelf Science, 92(1), pp.35-46.

- Howarth, R. W. 2008. Review: Coastal nitrogen pollution: A review of sources and trends globally and regionally. *Harmful Algae*, *8*(HABs and Eutrophication), 14-20. doi:10.1016/j.hal.2008.08.015

- Howes, B.L., Samimy, R. and Dudley, B., 2003. Site-Specific Nitrogen Thresholds for Southeastern Massachusetts Embayments: Critical Indicators. *Massachusetts Estuaries Project. MA DEP. Revised December*, *22*, p.2003.

- Jadhao, Rajesh Bhagwat. 2013. Contamination Of Water By Nitrate: A General Discussion Problems Associated, Causes, Prevention And Effects. International Journal of Innovative Research & Development ISSN: 2278/0322 Volume 2 Issue 7 http://www.ijird.com/index.php/ijird/article/view/36124/29266

- Katwijk, M.M., Thorhaug, A., Marbà, N., Orth, R.J., Duarte, C.M., Kendrick, G.A., Althuizen, I.H., Balestri, E., Bernard, G., Cambridge, M.L. and Cunha, A., 2016. Global analysis of seagrass restoration: the importance of large-scale planting. Journal of Applied Ecology, 53(2), pp.567-578.

- Kemp, W. M., Batiuk, R., Bartleson, R., Bergstrom, P., Carter, V., Gallegos, C. L., & ... Wilcox, D. J. (2004). Habitat Requirements for Submerged Aquatic Vegetation in Chesapeake Bay: Water Quality, Light Regime, and Physical-Chemical Factors. *Estuaries*, (3). 363.

- Kennish, M. J. & Fertig, B. 2012. Application and assessment of a nutrient pollution indicator using eelgrass (Zostera marina L.) in Barnegat Bay–Little Egg Harbor estuary, New Jersey. *Aquatic Botany*, *96*(1), 23-30. doi:10.1016/j.aquabot.2011.09.005

- Kenworthy, W. & Fonseca, M. 1996. Light requirements of seagrasses Halodule wrightii and Syringodium filiforme derived from the relationship between diffuse light attenuation and maximum depth distributions. *Estuaries* 19, 740-750.

- Kenworthy, W., Gallegos, C., Costello, C., Field, D., & di Carlo, G. 2014. Dependence of eelgrass (Zostera marina) light requirements on sediment organic matter in Massachusetts coastal bays: Implications for remediation and restoration. *Marine Pollution Bulletin*, *83*(Seagrass meadows in a globally changing environment), 446-457. doi:10.1016/j.marpolbul.2013.11.006

- Kinney, E.L. and Valiela, I. 2011. Nitrogen Loading to Great South Bay: Land Use, Sources, Retention, and Transport from Land to Bay. *Journal of Coastal Research*. 27 (4): 672 – 686.

- Koch, E.W., 2001. Beyond light: physical, geological, and geochemical parameters as possible submersed aquatic vegetation habitat requirements. Estuaries, 24(1), pp.1-17.

- Koch, E. W., C. Yarish, S. Beer, R. Troy, G. Capriullo, R. Linden, and J. Rehnberg. 1994. Environmental monitoring, seagrass mapping and biotechnology as means of fisheries habitat enhancement along the Connecticut Coast. Report to Connecticut DEP, Office of Long Island Sound Programs. 42pp.

- Krause-Jensen, D., Sagert, S., Schubert, H., Boström, C. 2008. Empirical relationships linking distribution and abundance of marine vegetation to eutrophication. Ecological Indicators 8, 515-529.

- Lambers, H. (1985). Respiration in intact plant tissues: its regulation and dependence on environmental factors, metabolism and invaded organisms. In: Douce, R., Day, D, A, (eds.) *Encyclopedia of plant physiology*. New series. Higher plant cell respiration. Springer-Verlag, Berlin p. 418-474

- Latimer, J. S., & Rego, S. A. 2010. Empirical relationship between eelgrass extent and predicted watershed-derived nitrogen loading for shallow New England estuaries. *Estuarine, Coastal And Shelf Science*, *90*231-240. doi:10.1016/j.ecss.2010.09.004

- Lefcheck, J. S., Wilcox, D. J., Murphy, R. R., Marion, S. R. and Orth, R. J. (2017), Multiple stressors threaten the imperiled coastal foundation species eelgrass (Zostera marina) in Chesapeake Bay, USA. Glob Change Biol. doi:10.1111/gcb.13623

- Lee, K., Short, F. T., & Burdick, D. M. 2004. Development of a nutrient pollution indicator using the seagrass, Zostera marina, along nutrient gradients in three New England estuaries. *Aquatic Botany*, *78*197-216. doi:10.1016/j.aquabot.2003.09.010

- Leschen, A. S., Ford, K. H., & Evans, N. T. 2010. Successful Eelgrass (Zostera marina) Restoration in a Formerly Eutrophic Estuary (Boston Harbor) Supports the Use of a Multifaceted Watershed Approach to Mitigating Eelgrass Loss. *Estuaries and Coasts*, (6). 1340.

- LINAP. 2016. Long Island Nitrogen Action Plan Scope: New York State Department of Environmental Conservation and the Long Island Regional Planning Council. http://www.dec.ny.gov/docs/water_pdf/linapscope.pdf

- LISS (Long Island Sound Study). 2003. Long Island Sound Habitat Restoration Initiative, Technical Support for Coastal Habitat Restoration. Section 3: Submerged Aquatic Vegetation. EPA Long Island Sound Office, Stamford CT.

- Lloyd, S. 2014. Nitrogen load modeling to forty-three subwatersheds of the Peconic Estuary. *The Nature Conservancy in partnership with the Peconic Estuary Program.*

- Lloyd, S., Mollod G., LoBue C., Lindberg M. 2016. Modeling nitrogen source loads on the north shore of Long Island. *The Nature Conservancy.*

- Marsh, J. A., Dennison, W. C., Alberte, R. S. 1986. Effects of temperature on photosynthesis and respiration in eelgrass (*Zostera marina L.*). J. exp. *Mar. Biol. Ecol.* 101: 257-267

- Moore, K.A., Shields, E.C. and Parrish, D.B., 2014. Impacts of varying estuarine temperature and light conditions on Zostera marina (eelgrass) and its interactions with Ruppia maritima (widgeongrass). *Estuaries and coasts*, *37*(1), pp.20-30.

- Nelson, Walter G. (ed.), 2009. Seagrasses and Protective Criteria: A Review and Assessment of Research Status. Office of Research and Development, National Health and Environmental Effects Research Laboratory, EPA/600/R-09/050.

- Nelson, Walt. 2017. Development of an epiphyte indicator of nutrient enrichment: Threshold values for seagrass epiphyte load. ECOLOGICAL INDICATORS. 74:343-356.

- Nelson, T.A. and Lee, A., 2001. A manipulative experiment demonstrates that blooms of the macroalga Ulvaria obscura can reduce eelgrass shoot density. Aquatic Botany, 71(2), pp.149-154.

- NYS STF (New York State Seagrass Task Force). 2009. Final Report: Recommendations to the New York State Governor and Legislature. http://www.dec.ny.gov/docs/fish_marine_pdf/finalseagrassreport.pdf

- Ochieng, C. A., Short, F. T., & Walker, D. I. 2010. Photosynthetic and morphological responses of eelgrass (*Zostera marina L.*) to a gradient of light conditions. *Journal Of*

Experimental Marine Biology And Ecology, 382117-124. doi:10.1016/j.jembe.2009.11.007

- Okubo, A. and J. Slater. 1989. Simple model of eelgrass growth and water quality: another catastrophe theory? In: Cosper, E.M., V.M. Bricelj and E. J. Carpenter (eds.), Coastal and Estuarine Studies, p. 693-712.

- Olesen, B. & Sand-Jensen, K. 1993. Seasonal acclimatization of eelgrass Zostera marina growth to light. *Mar. Ecol. Prog. Ser.* 94, 91e99.

- Orth, R. J., K. A. Moore S. R. Marion, D. J. Wilcox, and D.B. Parrish. 2012. Seed addition facilitates eelgrass recovery in a coastal bay system. Marine Ecology Progress Series 448:177-195. doi: 10.3354/meps09522

- Pedersen, O., Binzer, T. and Borum, J., 2004. Sulphide intrusion in eelgrass (Zostera marina L.). Plant, Cell & Environment, 27(5), pp.595-602.

- PEP (Peconic Estuary Program). 1998. Peconic Estuary Surface Water Quality: Nitrogen, Dissolved Oxygen, and Submerged Aquatic Vegetation Habitat. *Suffolk County Dept. of Health Services. Office of Ecology.*

- PEP (Peconic Estuary Program). 2015. Peconic Estuary Program Ecosystem Status Report. Yaphank, NY. 74 pp.

- Peterson, Bradley J. 2016. “Evaluating the impact of multiple stressors on eelgrass populations in the south shore estuaries of Long Island”. New York Seagrass Symposium, 4/5/16, Riverhead, NY.

- Pickerell, Christopher H. 2016. “Eelgrass (*Zostera marina L.*) Restoration Efforts on Long Island, NY: Past, Present, and Future”. New York Seagrass Symposium, 4/5/16, Riverhead, NY.

- Pickerell, C. and Schott, S. 2016. Peconic Estuary Program 2015 Long Term Eelgrass (*Zostera marina*) Monitoring Program: Progress Report 16. Prepared for the Peconic Estuary Program by Cornell Cooperative Extension Marine Program.

- Pulido, C., Borum, J., 2010. Eelgrass (Zostera marina) tolerance to anoxia. Journal of Experimental Marine Biology and Ecology 385, 8e13.

- Ralph, P., Durako, M., Enríquez, S., Collier, C., & Doblin, M. (2007). Impact of light limitation on seagrasses. *Journal Of Experimental Marine Biology And Ecology*, *350*(The Biology and Ecology of Seagrasses), 176-193. doi:10.1016/j.jembe.2007.06.01

- Raven, J.A. and Scrimgeour, C.M., 1997. The influence of anoxia on plants of saline habitats with special reference to the sulphur cycle. Annals of Botany, 79 (suppl 1), pp.79-86., 79-86.

- Reynolds, LK, M. Waycott, K. J. McGlathery, R. J. Orth, and J. C. Zieman. 2012. Eelgrass restoration by seed maintains genetic diversity: case study from a coastal bay system. Marine Ecology Progress Series 448:223-233. doi: 10.3354/meps09386.

- Roca, G., Alcoverro, T., Krause-Jensen, D., Balsby, T., van Katwijk, M., Marbà, N., & ... Romero, J. 2016. Response of seagrass indicators to shifts in environmental stressors: A global review and management synthesis. *Ecological Indicators*, *63*310-323. doi:10.1016/j.ecolind.2015.12.007

- Rosenzweig, C., W. Solecki, A. DeGaetano, M. O'Grady, S. Hassol, P. Grabhorn (Eds.). 2011. *Responding to Climate Change in New York State: The ClimAID Integrated Assessment for Effective Climate Change Adaptation.* Technical Report. New York State Energy Research and Development Authority (NYSERDA), Albany, New York. www.nyserda.ny.gov

- Sand-Jensen, K.A.J., Pedersen, N.L. and Søndergaard, M., 2007. Bacterial metabolism in small temperate streams under contemporary and future climates. *Freshwater Biology*, *52*(12), pp.2340-2353.

- Short, F.T., Klein, A.S., Burdick, D.M., Pickerell, C., Vaudrey, J., Bayley, H. and Evans, T., 2012. The Eelgrass Resource of Southern New England and New York: Science in Support of Management and Restoration Success. Final Report to The Nature Conservancy.

- Short F. T., McRoy C. P. 1984. Nitrogen Uptake by Leaves and Roots of the Seagrass *Zostera marina L. Botanica Marina* Vol. XXVII, pp. 547-555

- Smith, V.H., Tilman, G.D. and Nekola, J.C., 1999. Eutrophication: impacts of excess nutrient inputs on freshwater, marine, and terrestrial ecosystems. *Environmental pollution*, *100*(1), pp.179-196.

- SFWMD (Southwest Florida Water Management District). 2017. "District Reports Consistent Seagrass Increases in Tampa Bay". News Release, February 1, 2017. http://www.swfwmd.state.fl.us/news/article/2201/

- Stein, L. Y., & Klotz, M. G. 2016. Primer: The nitrogen cycle. *Current Biology*, *26*R94-R98. doi:10.1016/j.cub.2015.12.021

- SC (Suffolk County) 2015. Suffolk County Comprehensive Water Resources Management Plan. http://www.suffolkcountyny.gov/Departments/HealthServices/EnvironmentalQuality/WaterResources/ComprehensiveWaterResourcesManagementPlan.aspx

- Thamdrup, B., Hansen, J.W. and Jørgensen, B.B., 1998. Temperature dependence of aerobic respiration in a coastal sediment. FEMS Microbiology Ecology, 25(2), pp.189-200.

- Thayer, G.W., Kenworthy, W.J. and Fonseca, M.S., 1984. *Ecology of Eelgrass Meadows of the Atlantic Coast: a community profile* (No. FWS/OBS-84/02). National Marine Fisheries Service, Beaufort, NC (USA). Beaufort Lab.; Virginia Univ., Charlottesville (USA). Dept. of Environmental Sciences.

- Tiner, R.W., H.C. Bergquist, D. Siraco, and B.J. McClain. 2003. An Inventory of Submerged Aquatic Vegetation and Hardened Shorelines for the Peconic Estuary, New York. U.S. Fish and Wildlife Service, Northeast Region, Hadley, MA. Prepared for the Peconic Estuary Program of the Suffolk County Department of Health Services, Office of Ecology, Riverhead, NY. 47 pp.

- Tiner, Ralph, T., McGuckin, Kevin, MacLachlan, Andrew. (2012). 2012 Eelgrass Survey for Eastern Long Island Sound, Connecticut and New York. . U.S. Fish and Wildlife Service, National Wetlands Inventory Program, Northeast Region, Hadley, MA. National Wetlands Inventory report. 20 pp. including Appendix.

- Tinoco, Amanda. 2016. "Effects of Hurricane Sandy on Fire Island National Seashore, NY: Assessing water quality, seagrass and nekton communities in Great South Bay." New York Seagrass Symposium, 4/5/16, Riverhead, NY.

- Touchette B. W., Burkholder J. M. 2000. Review of nitrogen and phosphorus metabolism in seagrasses. *J exp Mar Bio Ecol 250*(1-2):133-167

- Trowbridge, P., 2009. Numeric nutrient criteria for the Great Bay Estuary. New Hampshire Department of Environmental Services, 1120.

- Unsworth, R.K., Collier, C.J., Waycott, M., Mckenzie, L.J. and Cullen-Unsworth, L.C., 2015. A framework for the resilience of seagrass ecosystems. *Marine pollution bulletin*, *100*(1), pp.34-46.

- USEPA (United States Environmental Protection Agency). 2003. *Ambient Water Quality Criteria for Dissolved Oxygen, Water Clarity and Chlorophyll a for the Chesapeake Bay and Its Tidal Tributaries (Regional Criteria Guidance). April 2003*. EPA 903-R-03-002. Region III Chesapeake Bay Program Office, Annapolis, MD.

- Van der Heide, T., Smolders, A. P., Rijkens, B. A., van Nes, E. H., van Katwijk, M. M., & Roelofs, J. M. 2008. Toxicity of reduced nitrogen in eelgrass (Zostera marina) is highly dependent on shoot density and pH. *Oecologia*, *158*(3), 411-419.

- Vaudrey, Jamie M. P. 2008. Establishing Restoration Objectives for Eelgrass in Long Island Sound, Part 1: Review of the Seagrass Literature Relevant to Long Island Sound. *Department of Marine Sciences, University of Connecticut.*

- Vaudrey, J.M.P., J. Eddings, C. Pickerell, L. Brousseau., C. Yarish. 2013. Development and application of a GIS-based Long Island Sound Eelgrass Habitat Suitability Index

Model. Final report submitted to the New England Interstate Water Pollution Control Commission and the Long Island Sound Study. 171 p. + appendices.

- Vaudrey J, Kremer J, Branco B, Short F. 2010. Eelgrass recovery after nutrient enrichment reversal. *Aquatic Botany*. 93(4):237-243.

- Waycott, M., Duarte, C.M., Carruthers, T.J.B., Orth, R.J., Dennison, W.C., Olyarnik, S.,Calladine, A., Fourqurean, J.W., Heck Jr., K.L., Hughes, A.R., Kendrick, G.A., Ken-worthy, W.J., Short, F.T., Williams, S.L., 2009. Accelerating loss of seagrassesacross the globe threatens coastal ecosystems. Proc. Natl. Acad. Sci. U. S. A. 106,12377–12381.

- 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. report number CWF 314-R. University of Connecticut. 105 pp.

- Zimmerman, R. C., Smith, R. D., Alberte, R. S. (1989). Thermal acclimation and whole-plant carbon balance in *Zostera marina L*. [eelgrass]. J. exp. *Mar. Biol. Ecol.* 130: 93-109

Appendix A

Tables with values from literature that pertain to environmental thresholds and criteria to support seagrass habitat.

Table 1. Nutrients

Citation	DIN mg/l	Total Nitrogen mg/l	Total Phosphorous mg/l	DIP mg/l
Batiuk et al. 2000	<0.15			<0.02
Benson et al. 2013		<0.34		
PEP 1998	0.009 +/- 0.001	<0.4		
Vaudrey 2008	<0.03			<0.02
Vaudrey et al. 2013		<0.029 optimal	<0.071 optimal	
Yarish et al. 2006	<0.03			<0.02

Table 2. Light and Clarity

Citation	Chlorophyll *a* µg/l	Surface Irradiance, water column light	Kd m^{-1}	Secchi Depth m	TSS mg/l	Light at leaf surface
Batiuk et al. 2000	<15	22%	1.5		<15	>15%
Benson et al. 2013	5.1	>25%				
Duarte 1991,Olesen & Sand-Jensen 1993		11%				
Ferguson et al. 1993				0.3-2		
Greening et al. 2014	4.6-13.2	20.5%				
Kenworthy & Fonseca 1996		24-37%				
Ochieng et al. 2010		>34%				
Pickerell 2016			< 0.6			
USEPA 2003	8	22%		0.2-1.9		
Vaudrey 2008	<5.5	22%	<0.7			>15%
Vaudrey et al. 2013		>25% min	<0.46 optimal			
Yarish et al. 2006	<5.5		<0.7		<30	

Table 3. Physical

Citation	**Clay/Silt** %	**Organic content** %	**H2S**	Internal O2
Kemp et al. 2004	<20-30	<5	<2 mM	
Koch 2001	2-56	0.4-16	<1 mM	
Leschen et al. 2010	<35			
Pedersen et al. 2004				35% air saturation
Vaudrey 2008		0.4-10		
Vaudrey et al. 2013	<2 optimal	<0.5 optimal		
Yarish et al. 2006	<20	3-5	<400µM	

Table 4. Sediment

Citation	**Temperature** ℃	**Water movement Min** cm/s	**Water movement Max** cm/s	**Wave Tolerance**
Hoffle et al. 2011	<27			
Koch 2001		3-16	50-180	<2 m
Vaudrey et al. 2013	<21 optimal			
Yarish et al. 2006		5	100	

Appendix B: Case Study - Tampa Bay FL

Impacts from eutrophication were readily evident in Tampa bay FL by the 1980s, such as noxious algae blooms, reduced water clarity, and episodes of stressfully low dissolved oxygen (Greening et al. 2014). Seagrasses were reduced to less than half of their estimated1950s levels (Greening and Janicki 2006). In recent years this estuary has received a lot of attention because their seagrass beds have made a remarkable recovery, now over 41,000 acres in 2016, surpassing the recovery goal of 38,000 bay wide acres set 24 years ago by the Tampa Bay Estuary Program (SFWMD 2017). The estuary was able to shift from a turbid algae dominated system back to a seagrass dominated system.

Greening et al. (2014) cite four key elements for their programmatic success:

1. Active community involvement and agreement.
2. Regulatory and voluntary reductions in nutrient loading.
3. Extensive long term monitoring of seagrass and water quality.
4. Public and private sector commitment to attain goals.

Targets for nutrient load reductions were developed in conjunction with seagrass restoration goals. Empirical regressions of chlorophyll *a* concentrations and light attenuation were determined for sustainable seagrass growth. Additional empirical relationships were developed between nitrogen loading and chlorophyll *a* concentrations, which were used to determine how Tampa Bay responds to changes in loading. When nitrogen load reductions and chlorophyll *a* targets were met, seagrass cover increased. (Greening and Janicki 2006)

The success achieved in Tampa Bay is inspiring and there is knowledge in their story that can be applied to other seagrass systems, especially in the technical approach to criteria and targets. Expectations of the same recovery dynamics in other systems is unrealistic because there are multiple variables involved that will not be the same. Even though the Tampa Bay estuary had lost a lot of seagrass, a considerable amount remained which has significant implications for recovery as compared to bays that have lost seagrass and lack remnant sources. There are several different species of seagrass in Tampa Bay which allows for greater ranges in temperature and salinity tolerance. A large portion of the nitrogen loading reductions were achieved through upgrades in waste water treatment and this may not be achievable in other loading scenarios.