An Ecosystem Services Assessment Using Bioextraction Technologies for Removal of Nitrogen and Other Substances in Long Island Sound and the Great Bay/Piscataqua Region

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National Oceanic and Atmospheric Adminstration (NOAA)
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FOREWORD by Mr. Jeff Corbin and Dr. David Yoskowitz

More than forty years after the passage of the Clean Water Act, far too many of our coastal waters, bays, estuaries, rivers and lakes remain plagued by eutrophication – the result of excess inputs of nutrients from land-based sources. The problems manifest themselves in many forms such as hypoxia, fish kills, loss of seagrass and other critical bottom habitat, and algal blooms (sometimes in toxic forms). Exacerbating this problem is the fact that a large majority of our nation's population lives only a short distance from one of these waterbodies, thereby simplifying the pathway for nutrients to enter our waterways.

Lack of progress, however, is not due to lack of effort or interest. Regulatory tools have been used aggressively to reduce nutrient inputs from point sources, other Clean Water Act tools such as Total Maximum Daily Loads have been developed to quantify the needed pollution reductions, and incentive-based conservation programs to limit nutrient inputs from agriculture land have been widely adopted. But given all of these efforts, and facing an ever-growing coastal population (178 million by 2020 in our coastal watershed counties or 52% of the U.S. population), eliminating eutrophication remains an elusive goal.

So this stand-off begs the question – what do we need to do differently? The answer, in a word – innovate. In addition to accelerating efforts already underway, we need to incentivize the development of new approaches. Approaches that will not only reduce additional nutrient inputs or alleviate the symptoms of eutrophication, but could also have the added benefits of reducing costs and providing additional ecosystem services. Nutrient Trading, the concept of allowing different sources of a pollutant (e.g., a sewage treatment plant and a farm) to exchange a portion of their pollutant reduction requirements in order to achieve the same end result at a lower cost, called 'nutrient credit trading', has been explored for quite some time. In many areas of the country such programs are showing great promise.

The research in this report, however, takes the trading concept to a new innovative level by quantifying the potential for bioextraction – using shellfish to reduce nutrients from our waters. It has long been known that shellfish extract nutrients through their feeding and filtering process, but can such processes make a significant dent on the overall nutrient reductions needed to alleviate eutrophication? Can such an approach also reduce the economic burden on other sources of nutrient pollution? Will the increases in shellfish abundances provide additional ancillary water quality benefits as well as other ecosystem services that otherwise would not have been realized?

Clearly the utility and acceptance of any innovative approach will depend upon a few key factors, including, the magnitude of the nutrient reductions it can achieve, our ability to quantify and verify those reductions, the relative cost of the new approach compared to already existing approaches, and the added value of any co-benefits such as enhanced fish habitat and recreational fishing opportunities provided by oyster reefs to new aquaculture business. We must acknowledge that we face a stiff headwind in our goal to eliminate eutrophication and that expanding the number of tools in our tool bag can only enhance our chances of success. And we must also realize that environmental improvement and economic advancement can go hand-in-hand, and not one at the expense of the other.

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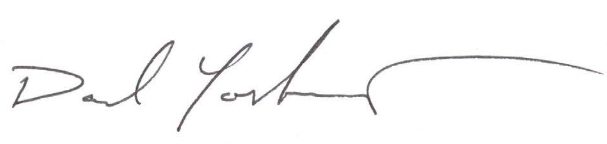

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EXECUTIVE SUMMARY

Nutrient related water quality degradation, called eutrophication, is an issue of concern in Long Island Sound (LIS) and Great Bay/Piscataqua Region Estuaries (GBP). This study evaluated the feasibility and potential of nutrient removal directly from the water through bioextraction through the filtration of shellfish, Eastern oyster (*Crassostrea virginica)* and quahog clam (*Mercenaria mercenaria*), as a complement to traditional land-based management measures.

Study goal and objectives

- Determine the mass of nitrogen removed from Long Island Sound and Great Bay Piscataqua Estuaries through shellfish (oyster, clam) aquaculture (bioextraction) at present and how significant it would be if aquaculture production areas were expanded.
- Assess how significant the removal through bioextractoin is in relation to the total nitrogen loading in these systems and how the systems compare to each other.
- Estimate the value of the 'ecosystem service', i.e. the water cleaning capacity, performed by the shellfish.
- Evaluate whether the removal is significant enough to support a role for shellfish growers in nutrient credit trading programs, taking into account the present situation and the potential expansion of shellfish aquaculture.

Models and methods

- An individual model to simulate growth and removal of nitrogen was developed in AquaShell for the Eastern oyster using data and literature values. The model was calibrated and validated showing results close to measured values. The model was integrated into the system scale EcoWin and local scale FARM models to simulate growth and nitrogen removal on the population level, the first time these models had included a component for Eastern oyster.
- In LIS, grid boxes (230) and vertical layers (10) of the high resolution Systemwide Eutrophication Model (SWEM) model were simplified into a coarser grid (42) with 2 vertical layers for the EcoWin ecosystem model. Water volumes, water quality conditions (e.g. nitrogen, carbon, salinity, etc.) and loading from various sources were passed from SWEM to EcoWin. Calibration and validation of EcoWin with measured data showed good correspondence of simulated results to measured values and thus was used with confidence in the simulation of shellfish bioextraction.
- Data and information about LIS culture practices were difficult to acquire, including the total area under cultivation. Two methods were used to develop upper, mid and lower estimates of cultivation area; 1) using harvest numbers and harvest densities, and 2) using total lease areas and interviews with growers to determine the area used. The middle estimate was used for the 'standard' EcoWin model simulation. Potential acres for an expansion scenario were determined from the maximum

area that might be used within the 40 ft depth contour. This provided a bracketing approach for estimates of current removal with an additional estimate for removal if production expanded.

- In both LIS and GBP, the local scale FARM model was used to determine removal locally and upscaled using caveats and assumptions to provide a system scale value for N removal by aquaculture.
- The ASSETS model was applied to determine the eutrophication condition within LIS and GBP.
- An economic analysis was conducted for LIS and GBP using a cost avoided approach to determine the value of the ecosystem service provided by the oyster aquaculture. The value represents the costs of providing a substitute or replacement for the nutrients removed through biotechnology, based on the costs associated with alternative nutrient removal methods that are avoided or replaced by the shellfish aquaculture.
- For LIS, a production estimate was also determined based on the costs of aquaculture production, which reflects the provision of ecosystems services as well as a food commodity. Because we were not able to collect data on revenues and costs from LIS shellfish growers, we used data from a survey of Rhode Island growers engaged in intensive oyster aquaculture.

Results and conclusions

- Bioextractive removal of nitrogen through shellfish aquaculture does not contribute significantly to the removal of nitrogen compared to total inputs.
- In LIS, the EcoWin simulation showed bioextraction of N by oyster aquaculture ranging from 0.11 to 1.88% of total N inputs, with an increase to almost 3% of total N inputs in an expansion scenario. Results that include oyster and clam cultivation are more promising, with a removal ranging from 0.58 – 10% under low and high values for acres currently in cultivation, and up to about 14% if cultivation is expanded. However, the quahog removal was not included in the model explicitly, only as a wild species with no natural mortalities and without a calibrated and validated growth model. Thus, we use only the oyster values for discussion.
- Nitrogen removal estimates based on the local scale FARM model applied to 4 areas in LIS (0.09 – 1.6% total input) compare well to the removal estimated from system scale EcoWin results (0.11 – 1.88% total N input). This suggests that in places where there is no system scale hydrodynamic model, such as GBP, upscaled local scale results can provide a reasonable estimate of removal, dependent on caveats and assumptions.
- Upscaled local scale N removal estimates from the local scale FARM model application at 2 sites in GBP show that oyster aquaculture removes 0.73 % of total inputs under current acres. If expanded to the maximum acreage allowable, 2.8% of total N inputs could be removed through oyster bioextraction.

- Expansion of oyster aquaculture would lead to removal of between 2 and 3% of the total N load in both systems.

- Based on a costs avoided method of estimation, the value of the N removed through oyster cultivation and harvest in LIS ranges from $8.5 to $230.3 million under the current acreage scenario. If LIS were to expand aquaculture production, the potential value could range between $17.4 and $469.3 million, depending on the alternative abatement approach considered.

- The value of estimated N removal based on aquaculture production costs, by comparison to avoided costs, for intensive aquaculture (using RI growers as a proxy for LIS growers) is $267.4 million for the current acreage scenario, and $545 million if production were expanded.

- The value of bioextraction estimated through the cost-avoided method for GBP is $1.1 and $1.3 million under the current scenario of 25 acres of shellfish lease area. If production acres were to expand to the potential 97 acres, the corresponding range of values is estimated to be $4.3 - $5.0 million.

- The use of shellfish biotechnology as a water quality management tool will require further verifications of actual production and revenues of shellfish harvesters, and modifications of existing public cost-share programs, or inclusion in economic nutrient trading programs. Regardless of whether shellfish farmers become eligible for nutrient credit trading, the valuation of the ecosystem services associated with shellfish culture will enhance public awareness of water quality issues, and could help shift attitudes to allow increased opportunities for shellfish aquaculture, and stimulate local economies.

Recommendations

- Improve governance and relationships among regulators, resource managers and the industry; this is a prerequisite for further development of bioextraction.

- Improve estimates of bioextraction using more accurate data on growers' aquaculture practices, costs and revenue. Actual harvest data and areas under cultivation are needed to estimate the N removal through aquaculture. This should include quahog in addition to oyster.

- Develop an individual model for quahog to add to the EcoWin and FARM models to improve bioextraction estimates in LIS, GBP and other estuaries where this approach will be applied.

- Acquire better data on grower costs and revenues for shellfish production in order to more accurately determine the value of the ecosystem service provided by shellfish aquaculture and to fully understand the incorporation of growers into a nutrient trading program.

- Increase sampling measurements to better understand local effects of global climate change and changes in all estuarine habitats, including salt marshes. This will reduce uncertainties in understanding of impacts of climate change on all habitats and provide insight about how those changes might change shellfish habitat.

- Undertake high resolution bathymetry and topography measurement to support predictions of coastal hazards and flooding from storms/sea level rise coupled with higher density sea level measurements to provide insight on impact to shellfish habitat and distribution.

- Determine the fate of primary production and linkages to hypoxia. There are uncertainties with respect to the understanding of sinking and horizontal export of primary production, and imbalances between sources and sinks of carbon in western LIS.

- Additional research on how climate and weather affect the amount and timing of nitrogen delivery to the estuary is needed. Climatic trends, including extreme rain and snow events, can affect the delivery of nitrogen loads to estuaries. More nitrogen is "flushed" from the landscape during wet periods. Also, rainfall patterns, in addition to temperature variations, are known to impact phytoplankton growth, e.g. extending the timing and duration of toxic blooms, and also Waste Water Treatment Plant (WWTP) processes; e.g. improving removal during warmer time periods. New England is experiencing more frequent higher intensity rain storms, and this trend is anticipated to continue.

- Conduct research to better understand trophic linkages between primary production and apex predators; food web dynamics are relatively poorly known.

- Conduct research to better understand the consequences of changes in climate on timing and fate of primary production.

- Support research to identify and optimize nitrogen load reduction actions to improve conditions in the estuary.

- Monitor macroalgae spatial distribution patterns consistently from year-to-year to better characterize long-term trends on an estuary wide basis.

- Fully explore policy options for including shellfish culture in N trading programs and other bioextraction opportunities, such as seaweed culture, ribbed mussel culture, or harvest of other N-containing noncommercial species, such as invasive green crabs (*Carcinus maenas*), Crepidula (limpets), or starfish.

INTRODUCTION AND BACKGROUND

Eutrophication is among the most serious threats worldwide to the function and services supported by coastal ecosystems (Bricker et al., 2007; Zaldivar et al., 2008; Diaz and Rosenberg, 2008; Bricker and Devlin, 2011 and papers therein). Many water bodies worldwide have experienced nutrient-related water-quality degradations with consequent impacts, such as hypoxic bottom waters and loss of valuable habitats, with cascading impacts on fisheries (e.g., Orth and Moore, 1984; Brietburg et al., 2009, 2009a; Lipton and Hicks 1999, 2003; Mistiaen et al., 2003; Beck et al. 2009, 2011). Concern about these conditions has led to legislation that requires monitoring, assessment and implementation of management measures to reverse coastal eutrophication in places where degradation is observed (e.g. US Clean Water Act, EU Water Framework Directive). Most of these management measures focus on reductions of land-based sources of nutrients, such as fertilizer application and wastewater treatment plant discharges, and in some cases these measures have been successful.

There is increasing recognition that returns on investment diminish with increased stringency in both point and non-point source controls, that additional management will result in incrementally smaller reductions in nutrient loads, and that at some point further reductions are not cost-effective (Stephenson et al., 2010). This is particularly the case for non-point sources, where control is difficult technically, but is also a problem for point sources once nutrients are reduced to concentrations at the limit of technological feasibility. Another argument for finding alternate management measures is highlighted by the different (and unpredictable) recovery paths that can occur, in contrast to a management model based on the presumption that the recovery path is the reverse of the impact path, and that both are linear (Duarte et al., 2009). Historical alterations in habitat quality, food webs, and community structure in coastal systems can alter nutrient processing, thus modifying the ecosystem response to reduced nutrient loads (Duarte et al., 2009). This modification may be due to a regime shift that results in an altered recovery trajectory, and/or a moving baseline, due to climate change for instance.

A State-Environmental Protection Agency (EPA) Nutrient Innovations Task Group concluded in a 2009 report (State – EPA Nutrient Innovations Task Group, 2009) that "continuing the status quo at the national, state, and local levels and relying upon our current practices and control strategies will not support a positive public health and environmental outcome." A systems approach that integrates watershed load reduction programs with enhanced nutrient processing in coastal systems may prove more effective at restoring ecosystem services at less cost than load reduction programs alone. Shellfish aquaculture, which has shown promise in reducing eutrophication impacts by taking up nutrients from the water, is an alternative method that might be used together with land-based methods.

Shellfish filter phytoplankton and detritus from the water, thereby reducing eutrophication by short-circuiting organic degradation and consequent effects on bottom-water dissolved oxygen, and remove nutrients through harvest, sequestration into shell and tissue, and through denitrification (Ferreira et al., 2007b, 2011a; Cerco and Noel, 2007; Kellogg et al., 2013; Rothschild et al., 1994; Carmichael et al., 2012; zu Ermgassen et al. 2012, 2013; STAC, 2013a, 2013b; Kellogg et al., 2013, 2014; Grabowski and Peterson, 2007; Hicks et al., 2004; Pollack et al., 2013). Recent research has shown that the removal efficiencies of nitrogen through shellfish aquaculture compare favorably to removal by existing agricultural and stormwater Best Management Practices (BMPs); the cost per unit removed also compares favorably to those BMPs (Rose et al., 2014a). Shellfish cultivation and harvest are presently being promoted as a means of sustainable domestic production by NOAA's 2009 Aquaculture Policy and National Shellfish Initiative, and locally through policies such as the 2009 Maryland Shellfish Aquaculture Plan; both national policy and local initiatives acknowledge the water-quality benefits provided in addition to seafood production.

The Regional Ecosystem Services Research Program Bioextraction Study

EPA's Office of Research and Development (ORD) has highlighted that research is needed to evaluate and model the nitrogen removal and ecosystem services provided by filter-feeding shellfish populations in estuaries (Compton et al., 2009). This project, funded by EPA under its Regional Ecosystem Services Research Program, researched the role of enhanced food web processing of nutrients by shellfish which has shown promise in other waterbodies through natural and engineered systems (e.g. Lindahl et al., 2005; Golen, 2009; Ferreira et al., 2011; Peterson et al., 2014; Burke and Menzies, 2010; STAC, 2013a; Bricker et al., 2014; Hoellein et al., 2014). This study evaluated the ecosystem service benefits through removal of nitrogen (N) by oyster and clam aquaculture, and the broader applicability of bioextraction management strategies in the specific context of two regional ecosystem restoration programs: Long Island Sound (Connecticut and New York; only Connecticut was included in the study) and the Great Bay/Piscataqua region (New Hampshire and Southern Maine).

The focus of the this study was the ecosystem service of nitrogen removal provided by shellfish bioextraction at present harvest rates and the potential increase in services under scenarios of expanded production. The water quality maintenance services of shellfish are a direct result of their "suspension-feeding" activity, which reduces concentrations of microscopic algae (phytoplankton) and suspended organic detritus and inorganic particles in surrounding waters (Shumway et al., 1985; Bayne, 1993; Cloern et al., 1982; Brumbaugh et al., 2006). In some coastal systems, shellfish, through their feeding activity and resultant deposition of organic material onto the bottom sediments, are abundant enough to influence or control the overall abundance of phytoplankton growing in the overlying waters. This control is accomplished both by direct removal of suspended material and by controlling the rate at which nutrients are exchanged between the sediments and overlying waters[1] (Ulanowicz and Tuttle, 1992; Newell and Koch, 2004; Kellogg et al., 2014). Encrusting, or fouling, organisms (e.g., tunicates, bryozoans, sponges, barnacles, and some polychaete worms) on oyster reefs and in the surrounding benthos are also suspension feeders and contribute to the overall filtering capacity of a reef (NOAA Fisheries Office of Habitat Conservation undated-a; Kellogg et al., 2014).

[1]*Field observations demonstrate that large populations of shellfish are capable of consuming a considerable fraction of the phytoplankton from overlying waters. For example, Haamer, and Rodhe (2000) showed that water passing a mussel bed in the sound between Sweden and Denmark was depleted of phytoplankton in the entire height of the fully mixed water column over a several kilometer long more or less continuous mussel bed. Similarly, field measurements by Grizzle et al. (2006) reveal a considerable fraction of suspended particulate matter is removed—up to 62 percent for the bivalve species* Mercenaria*—from waters overlying dense shellfish assemblages.*

Shellfish filtration can enhance water clarity allowing sufficient light penetration to support maintenance and expansion of seagrass habitat (Newell and Koch, 2004), an important estuarine nursery area, where juvenile invertebrates and fish are protected from predators. Ecosystem modeling suggests that restoring shellfish populations to even a modest fraction of their historic abundance could improve water quality and aid in the recovery of seagrasses, which can further reduce sediment resuspension and improve light conditions (Newell and Koch 2004; Kellogg et al., 2014). In addition, researchers suggest that robust populations of shellfish can suppress harmful blooms of phytoplankton, such as the ‘brown tides’ that have occurred along the mid-Atlantic coast (Cerrato et al., 2004), and help modulate blooms of other types of harmful plankton, including ‘red tides’ (Peabody and Griffin, 2008; NWFSC and WASG, 2002).

As suspension feeders, shellfish also enhance water quality by concentrated deposition of feces and pseudofeces (particles collected on the gills that the shellfish do not use as food; Newell, 2004; Newell and Koch, 2004; Newell et al., 2007). This benthic-pelagic coupling removes particulate organic matter (POM) from the water column making it available to benthic detritivores, such as polychaetes and amphipods, which are a key element of the food chain for many estuarine fish. Increased biodeposition of organic matter in sediments can lead to increased bacterial denitrification that can help to remove N from estuarine systems (unless such deposition leads to hypoxic conditions which suppress nitrification; Childs et al., 2002). The associated bacteria in sediments of an oyster bed can remove 20 percent or more of the N in oyster wastes, using the same process that is used in modern waste-water treatment plants (Shumway et al., 2003). Further, filter-feeding shellfish not only remove N from the water column to the benthos; they also incorporate a high proportion of it into their tissues. Oyster tissue has a mean nitrogen content ranging from 5.64-9.72 percent, and shell content is 0.19-0.3 percent (Kellogg et al., 2014 and citations therein; Grizzle et al., 2016; Sebastino et al., 2015). When the shellfish are harvested, the N is removed from the system, thereby recycling nutrients from sea to land (Shumway et al., 2003; Lindahl et al., 2005). About half of this N is in the relatively large shell (Kellogg et al., 2013; STAC 2013a). In contrast,when species with lighter shells, such as blue mussels, are harvested, less N is removed (Newell and Koch, 2004).

By mediating water column phytoplankton dynamics and denitrification, shellfish are likely to reduce nutrients that stimulate excessive plant growth in coastal waters, which often leads to low dissolved oxygen levels (hypoxia) as a result of the decaying phytoplankton consuming most of the available oxygen, a serious environmental problem in many aquatic ecosystems worldwide (Atlantic States Marine Fisheries Commission, 2007; Diaz and Rosenberg, 2008). For example, 65 percent of U.S. estuaries and coastal bays are moderately to severely degraded by nutrient pollution from agricultural practices, urban runoff, septic systems, sewage discharges and eroding streambanks (Bricker et al., 2008).

Shellfish filtration rates are a function of several environmental factors, including temperature, salinity, and suspended particulate concentration (Cerco and Noel, 2005; Bayne, 1993, 1997; Barille et al., 1997)[2]. In addition, different shellfish species exhibit significant variation in filtration rates, with the rates generally increasing with species size (Powell et al. 1992; Cranford and Grant, 1990; Smaal and Zurburg, 1997; Prins et al, 1991).

[2]*Filtration activity is defined in terms of clearance rate (i.e., the volume of water cleared of particles per time unit).*

For large species, such as the Eastern or American oyster, *Crassostrea virginica,* filtration rates have been estimated at 163 liters per gram of oyster tissue per day (NOAA Fisheries Office of Habitat Conservation, undated-a). Newell (1988) calculated that the abundance of this species in Chesapeake Bay before 1870 was high enough that oysters could filter the entire volume of the bay in about three days. Nearly a century of exploitation and habitat destruction have reduced populations to 1% of previous numbers, and 325 days are now required to perform the same function. It must additionally be noted that since oyster populations are no longer present in some areas of the Chesapeake Bay, only a portion of the water mass could in any case be filtered.

Study Objectives and Linkages

The objectives of this research are to:

(1) Determine the mass of N removed from Long Island Sound and Great Bay/Piscataqua Region Estuaries through shellfish (oyster, clam) aquaculture (bioextraction);

(2) Assess how significant the removal is in relation to the total N loading in these systems and how the systems compare to each other;

(3) Evaluate whether the removal is significant enough to support a potential role for shellfish growers in nutrient credit trading programs, taking into account the present situation and a potential expansion of shellfish aquaculture.

The two locations, Long Island Sound (Connecticut [CT], New York [NY] - hereafter LIS), and the Great Bay/Piscataqua Region Estuaries (New Hampshire [hereafter GBP]), differ in nutrient loading, hydrodynamic forcing, and susceptibility to eutrophication, but are both areas of active ecosystem restoration efforts. This study has been executed in parallel with other work being conducted, and has benefited from prior research programs in the two study areas. The background and rationale for the two study sites is described below.

Long Island Sound (LIS): In 2001, EPA Regions 1 and 2 approved a Total Maximum Daily Load (TMDL) analysis for nitrogen discharges to attain water quality standards for dissolved oxygen in LIS (NYSDEC and CTDEP, 2000; NYCDEP, 2000). Over the past decade, investments of more than $1 billion have been made to upgrade wastewater treatment plants (WWTP) to remove nitrogen and to better manage non-point sources. Additional investments in watershed infrastructure and management practices are anticipated. However, recent waterbody (HydroQual, 2007) and watershed modeling (Evans, 2008) analyses conducted to support an evaluation of the TMDL show that fully attaining designated uses supported by water quality standards would require even greater reductions in watershed sources of nitrogen. Additional modeling analyses conducted as part of the LIS Study program have shown that nutrient bioextraction (defined here as removing nutrients from an aquatic ecosystem through the harvest of enhanced biological production, including but not limited to the aquaculture of suspension-feeding shellfish and/or algae), can potentially be effective in improving dissolved oxygen levels and in helping to attain water quality standards in a cost-effective manner (HydroQual, 2009). This work will be beneficial to the further evaluation of bioextraction needed as part of a “systems” approach that integrates watershed load reduction programs with enhanced nutrient processing to attain water quality standards and restore designated uses and ecosystem services.

Additionally, the LIS Study sponsored a December 3-4, 2009 workshop on the potential benefits of an expansion of shellfish and seaweed cultivation and harvest on concentrations of particulate and dissolved nitrogen in the Sound (details on presentations are available at http://longislandsoundstudy.net/issues-actions/water-quality/nutrient-bioextraction/**).** The workshop, funded by the LIS Study, invited international speakers and local regulatory, legal, and industry experts to discuss the potential application of these 'bioextractive technologies' within the Sound, highlighting the need for research to explore the feasibility of this technology. A key recommendation of the workshop was to develop a pilot study to evaluate the feasibility, benefits, and impacts of bioextraction systems in LIS.

Great Bay/Piscataqua Region Estuaries System (GBP): The Piscataqua Region Estuaries Partnership supported the development of nutrient criteria for the GBP (PREP, 2009). Overall, nitrogen loads to the estuary will need to be reduced on the order of 50 percent to achieve water quality goals (NHDES, 2009). While non-point sources are the dominant nitrogen load, point source loadings are also significant. The capital cost of upgrading the 18 wastewater treatment plants to remove nitrogen has been estimated to be more than $400 million. One of the concerns about investing in wastewater treatment facility upgrades is that the continued increase in non-point source nitrogen loads will nullify the impact of the upgrades. Preliminary work suggested that a large percentage of the non-point source nitrogen load can be attenuated through ecosystem services, such as losses in wetlands and through bioextraction by shellfish. Efforts to reduce nitrogen loads to the estuary will only be successful if these ecosystem services are quantified and at least maintained at present levels. Determining the value of these ecosystem services will provide decision-makers with information on the cost-effectiveness of maintaining ecosystem services for nitrogen removal relative to traditional measures such as wastewater treatment plants.

The Piscataqua Region Estuaries Partnership funded a study to determine the nutrient uptake and removal (bioextraction) potential for two size classes of oysters, particularly as a farmed species, in the GBP estuarine system (see Appendix 3). This field experiment was conducted in summer 2010 using field deployment methods known to be effective for shellfish (Eastern oyster, *Crassostrea virginica*) aquaculture. Data were also obtained based on historical production records of an active oyster farm in Little Bay (the northern end of the GBP). The study made preliminary estimates of the potential contribution of oyster aquaculture to nutrient (particularly nitrogen) management goals for the state's estuarine waters.

The objective of this study is to capitalize and expand on the two separately funded bioextraction field projects described to answer the questions posed above. This study aims to compare the potential role of bioextraction technologies (the cultivation and harvest of shellfish only, i.e. Eastern oyster, northern quahog) in maintaining and enhancing ecosystem function and associated services, such as water quality improvements (reductions of nutrients, suspended sediment, chlorophyll [CHL]) through the filtration of water by bivalves, which allows seagrasses, and thus fish habitat, to re-establish in high turbidity systems in these systems. The results of this study will support regional water quality management programs, provide tools for broader application nationally, and contribute to the ongoing domestic and international discussions regarding the use of bioextraction for nutrient management and the inclusion of shellfish growers in nutrient trading programs nationally.

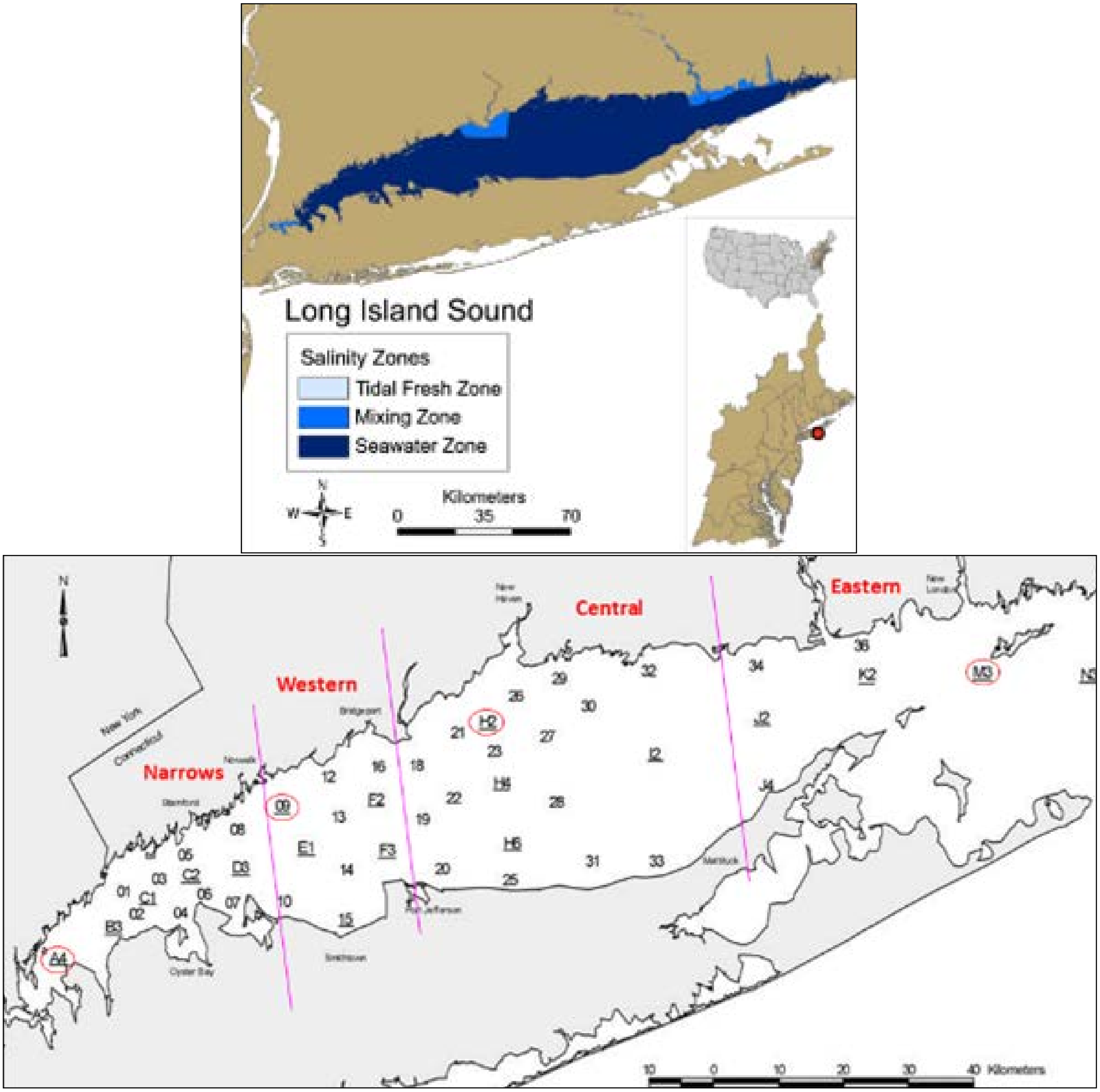
DESCRIPTION OF STUDY SITES

Long Island Sound

Long Island Sound is a large estuary (3,259 km^2), located between Connecticut to the north and Long Island to the south (Figure 1, Table 1). The three major tributaries, Connecticut, Thames and Housatonic Rivers, enter from the north, contributing 90% of the total freshwater to the Sound, with the Connecticut River alone accounting for about 70% of the total (Wolfe et al., 1991). The Sound is connected to the Atlantic Ocean at its eastern end via Block Island Sound and connects to the East River and New York Harbor to the west. The average salinity in the western end is 23 psu, suggesting significant longitudinal mixing, though the East River discharge promotes stratification, particularly during the spring runoff period (Bricker et al., 1997). Salinity at the eastern end of the Sound is 35 psu, matching that of seawater, with an overall Sound average of 28 psu. Average depth is about 20 m, with an average tidal range of about 2 m in the west and 1 m in the east. Residence time is 2-3 months. The watershed area is 12,773 km^2 and includes parts of Vermont, New Hampshire, Massachusetts, Connecticut, Rhode Island and New York. The watershed is highly developed, particularly in the New York metropolitan area, as well as around Bridgeport and New Haven, two of Connecticut's largest cities. The watershed population in 2010 was about 8.93 million people, with nearly half the population living near the coast in New York and Connecticut (longislandsoundstudy.net). While activities in the watershed contribute to state and local economies, they have taken a toll on its environmental health. Severe water quality degradation and critical habitat loss have resulted from a multitude of anthropogenic activities along its coastline (Latimer et al., 2014).

Table 1: Characteristics of Long Island Sound

Estuary Characteristic	Value	Source
Watershed area (km^2)	12,773	Smith, 2003; Bricker et al., 2007
Estuarine area (km^2)	3,259	Smith, 2003; Bricker et al., 2007
Watershed population(10^3)	4,900	Smith, 2003; Bricker et al., 2007
Nitrogen load (t N/y) for 1990	50,000	LIS Study
Average tidal range (m)	1.85	Smith, 2003; Bricker et al., 2007
Estuary volume (10^6 m^3)	65.5	Smith, 2003; Bricker et al., 2007
Freshwater inflow (m^3/s)	57	Wolfe et al., 1991; Smith, 2003; Bricker et al., 2007
Average depth (m)	19.5	Smith, 2003; Bricker et al., 2007
Average salinity	28	Smith, 2003; Bricker et al., 2007



Flushing time (d)	60 - 90	Smith, 2003; Bricker et al., 1997, 2007
Aquaculture area (acres) CT only (Approved + Conditional)	45,000	T. Getchis, CT SeaGrant, pers. communication; LIS Study

Figure 1: Location map of Long Island Sound (upper panel) and zone delineation (lower panel) for use in application of ASSETS eutrophication assessment method. The Sound is considered one salinity zone (seawater; >25 psu) but was divided into zones (Narrows, Western, Central, Eastern) for consistency and comparability of results with previous studies. Circled sampling stations (A4, 09, H2, M3) are used in subsequent figures to illustrate data patterns across the Sound.

Nutrient loading and eutrophication status history

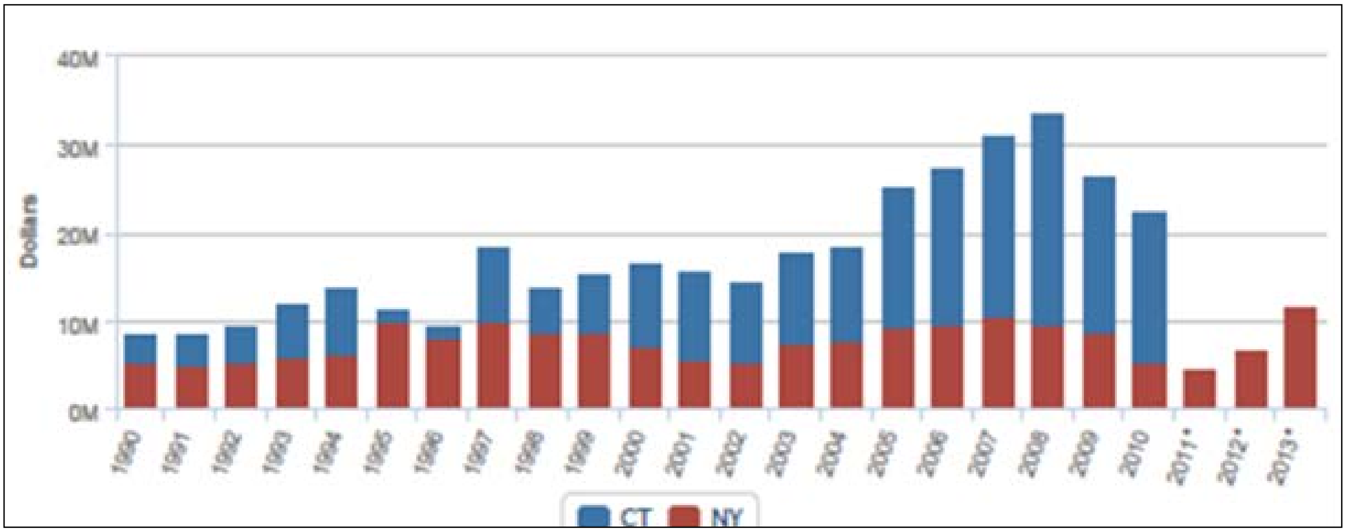
Long Island Sound's large and highly developed watershed has resulted in large amounts of nutrients being discharged into the waterbody, primarily through atmospheric deposition, sewage treatment plant effluent, and non–point source runoff. Nitrogen loads increased dramatically over the last two centuries as a result of human habitation of the watershed (Varekamp et al., 2014 and citations therein). Those loads, combined with strong summer thermal stratification in the western part of the Sound, make LIS susceptible to seasonal low dissolved oxygen levels (hypoxia), which have been identified as one of the top water quality concerns in the Sound and is one of the main indicators of eutrophication impacts. Low oxygen levels can occur naturally in estuaries during the summer, when calm weather conditions prevent the mixing of the water column that replenishes bottom water oxygen during the rest of the year. However, studies of the limited historical data base for the Sound suggest that summer oxygen depletion in western LIS has grown worse since the 1950s (CT DEEP, 2013). Since 1987, the causes and effects of hypoxia (dissolved oxygen levels below 3 mg L^{-1}) have been the subject of intensive monitoring, modeling and research through the Long Island Sound Study (LISS). The areal extent and duration have been tracked, with records showing the maximum area of hypoxia was 393 square miles in 1994, and the longest hypoxic event lasted 79 days in 2008 (CT DEEP, 2013). While there are additional eutrophication impacts that have been documented in LIS, for example the complete loss of seagrasses in the western Sound (Lopez et al., 2014), hypoxia was the subject of an analysis to try to determine the nitrogen load reductions necessary to meet water quality objectives.

A Total Maximum Daily Load (TMDL) analysis, approved in 2001, led to an agreement by the EPA, CT and NY to reduce nitrogen loads by 58.5 percent by 2017 from early 1990s baseline levels to improve DO levels and meet water quality standards (NYSDEC and CTDEP, 2000). These efforts have resulted in greater than 40% reductions in combined nitrogen loads from CT and NY and the expectation is for continued reductions. While dissolved oxygen improvements have been documented, they have been slow and masked by weather-driven variability (CT DEEP, 2013; LISS, 2013). Additionally, the continued population growth within the already densely-populated LIS basin is of concern for both point and non-point source loads in the future. However, full implementation of the 2000 TMDL (HydroQual, 2007), in addition to the growing success of LIS's nutrient trading program (CT DEP, 2010), is expected to result in future water quality improvements.

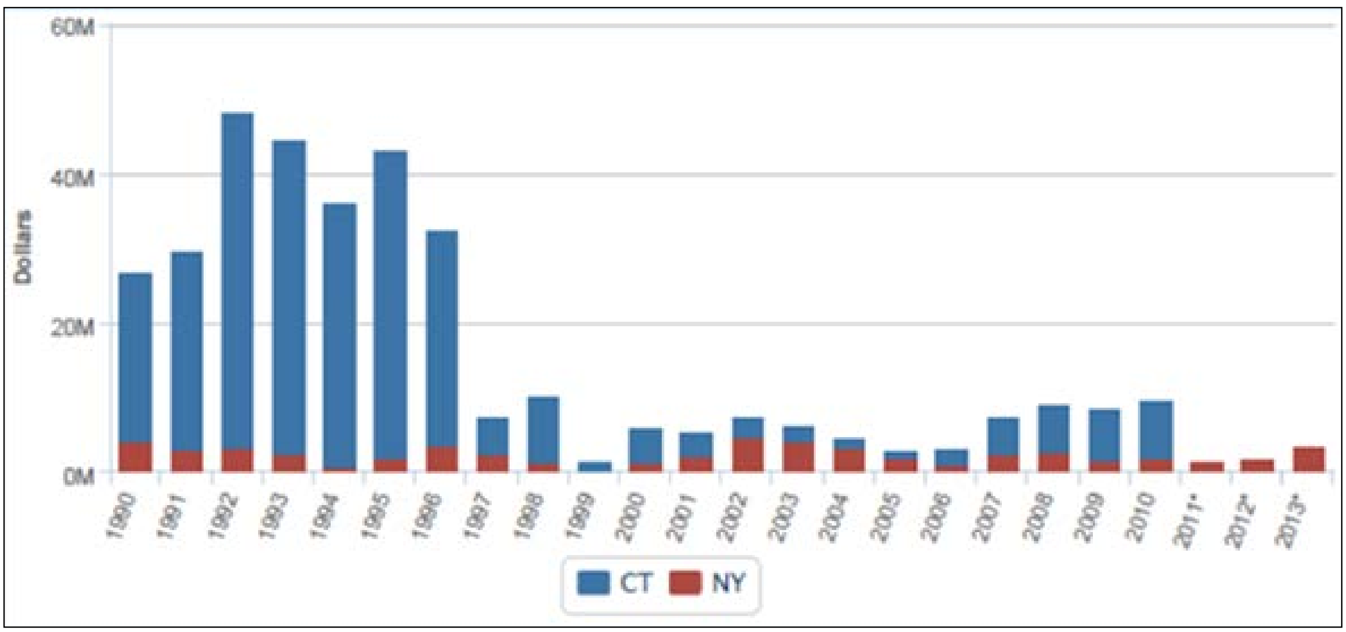
Shellfish cultivation history

The shellfish industry in LIS is long-standing and well established. The industry includes harvest of both Eastern oyster (*Crassostrea virginica*) and hard clam (quahog) (*Mercenaria mercenaria*) from shellfish grounds in the states of CT and NY, which share the Sound. Long Island's history, culture, and traditions are closely linked to notable clam and oyster harvests from the Sound. The NY 'Blue Point' oyster from LIS, first marketed as such in 1813, is still known as a high quality product (Kurlansky, 2006; Churchill, 1920; State Shellfish Commission reports dating back to 1880s), and CT oyster production (4×10^6 bushels per year) was third only to Chesapeake Bay in the early 1900s (both Virginia and Maryland produced 5×10^6 bushels per year; Churchill, 1920). For many years NY was the leading producer of hard clams in the United States (NY seafood council, ww.nyseafood.org), mainly from bay waters south of Long Island. The dockside value of hard clams landed in NY alone exceeded that of any other fish or shellfish species landed in the state for every year from 1972 through 1977 (New York Seafood Council, ww.nyseafood.

org). Recent cultured and wild caught hard clam and oyster harvests in LIS have provided over 300 jobs and $30 million in revenue annually (CT Department of Agriculture, 2011; Goldberg et al., 2012; http://longislandsoundstudy.net/). Figure 2 shows that the value of clam harvest has increased recently, while the value of oysters from CT and NY peaked in the early 1990s.



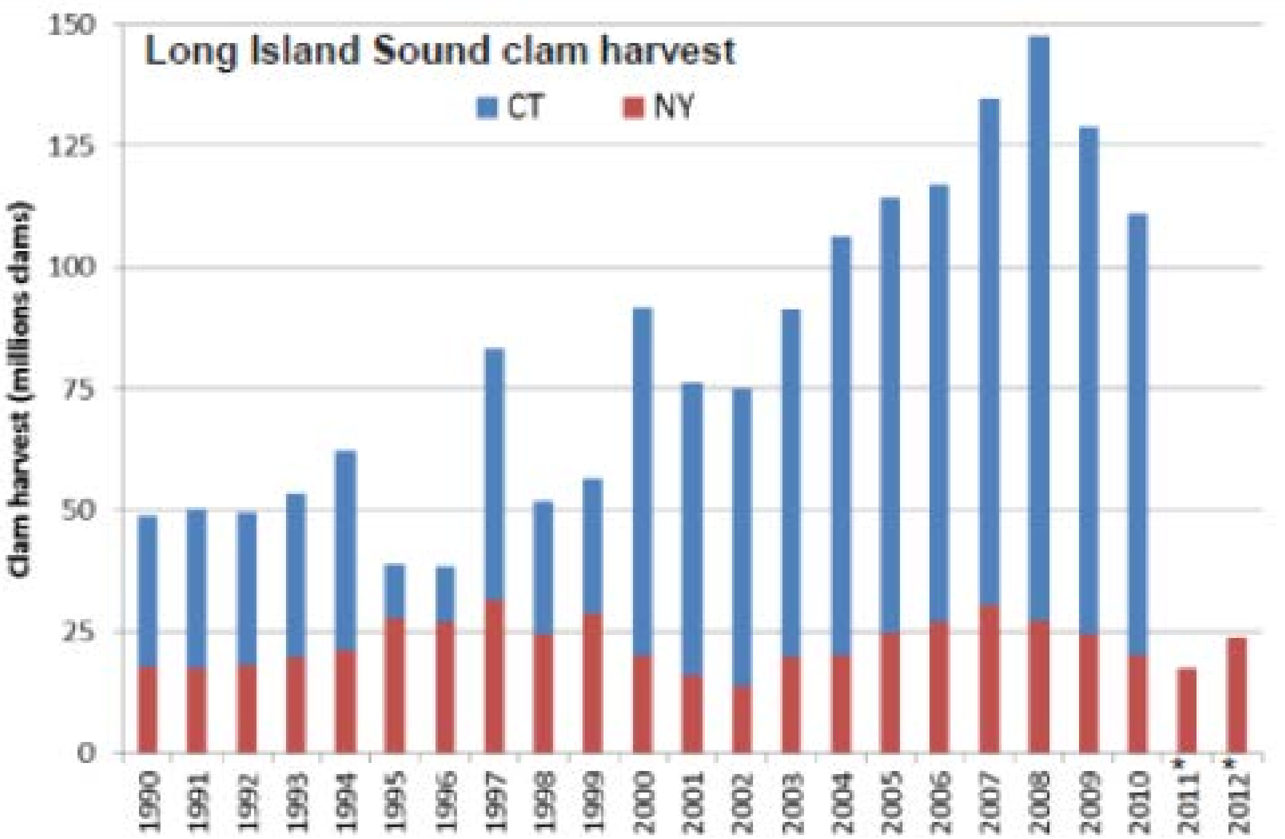
*The largest cultivated acreage producer failed to report harvest statistics from 2008 to 2010. As a result, the overall average harvest growth rate was factored into the reported figures by the company to obtain an estimate for 2009 and 2010 harvest numbers. However, no growth rate was factored for 2008 harvest numbers



**Since 2008, CT oyster harvest has not been available but resource managers believe that harvest continues to rise.*

Figure 2: Value of clam (upper panel) and oyster (lower panel) harvest in Long Island Sound, 1990 – 2012 (aquaculture + wild caught).

The hard clam harvest has more than tripled in the past decade (Figure 3), in part because some lobster fishermen have turned to clamming as lobster harvests declined and then closed completely. Recent CT landings data are not available, but NY clam harvest declined from 2007 to 2011. This is thought to be due in part to NY clam growers harvesting other Long Island waters, including the Peconics and the South Shore since, unlike CT where harvest is only allowed from leased beds, the majority of NY oystermen and clam growers go where they find the greatest supply.



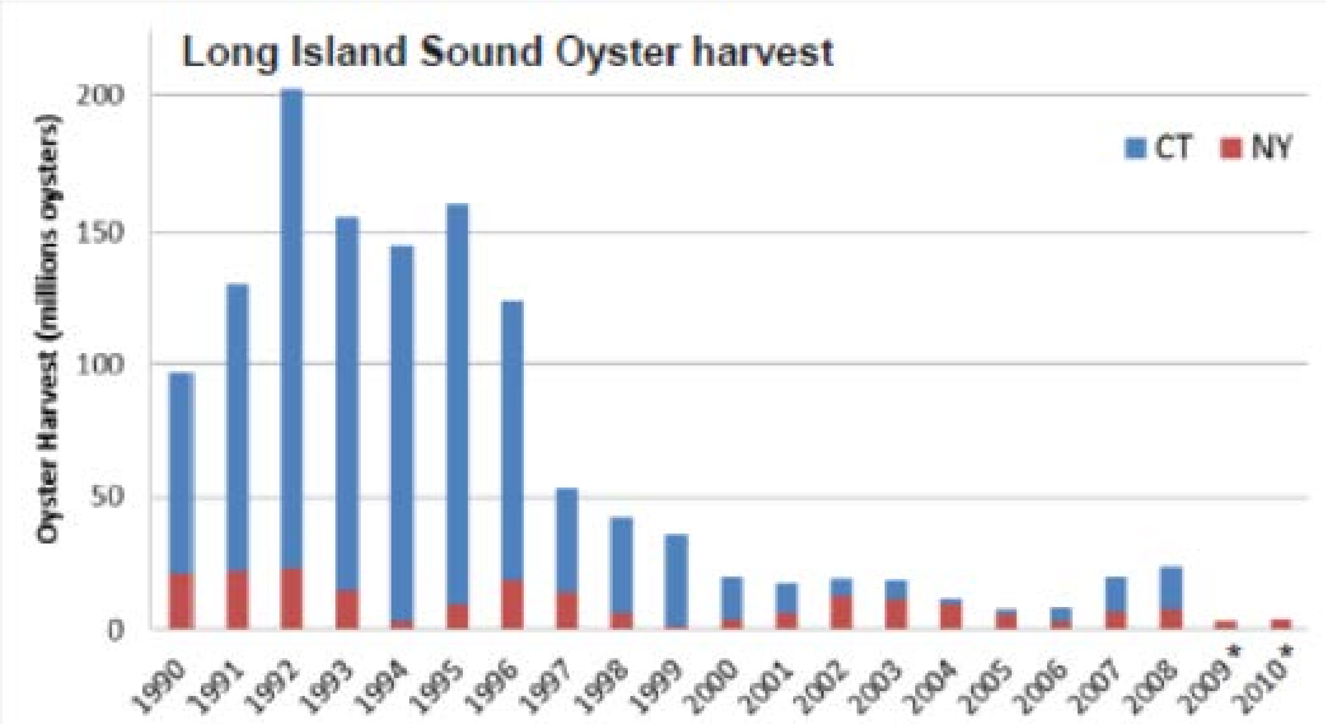
Clam harvest numbers:

CT - converted to numbers from reported bushels using average (213 clams / bag) based on Little Neck: 400 count bags, Top Neck: 200 count bags, Cherrystone:150 count bags. Chowders: 100 count bags and assuming that they are harvested in equal (bushel) amounts.

NY – converted to numbers from reported bushels using average (263 clams / bushel) based on Little Neck: 600 count bushels, Top Neck: 200 count bushels, Cherrystone:150 count bushels. Chowders: 100 count bushels and assuming that they are harvested in equal (bushel) amounts.

*Recent CT harvest counts not available. The largest cultivated acreage producer failed to report harvest statistics from 2008 to 2010. As a result, the overall average harvest growth rate was factored into the reported figures by the company to obtain an estimate for 2009 and 2010 harvest numbers. However, no growth rate was factored for 2008 harvest numbers. (from: longislandsoundstudy.net/)

Figure 3: Harvest of clams and oysters (next page) in Long Island Sound, 1990 – 2012.



Oyster harvest numbers:

CT - 1990-2003 converted to numbers from bushels using 200 oysters per bushel, 2004-2008 converted from 100 count bags.

NY – converted to numbers from reported 200 count bushels.

*Since 2008, CT counts have not been available, but resource managers believe that harvests continue to rise. (from: www.longislandsoundstudy.net/)

Figure 3 (contd.): Harvest of clams (previous page) and oysters (above) in Long Island Sound, 1990 – 2012.

The Sound was also well known for its oyster trade, especially during the 19th and early 20th centuries. Oyster production grew in the 1980s and 1990s due to successful oyster culture practices which typically grow oysters and clams on the bottom with no gear. A typical harvest size oyster is about 90g, and the legal length for harvest is 3 inches (7.63 cm). The large commercial oyster industry peaked in 1992 and declined mainly due to MSX, a parasitic disease. Oyster harvests began to rebound in 2006 (Figure 3). In CT this was due in part to efforts to restore and protect oyster habitats. Since 2011, CT harvest data have not been available (Figure 3), but resource managers believe that harvests continue to rise (http://longislandsoundstudy.net/).

In 2012, there were 127,884 acres of approved oyster harvest area in CT, 99,132 acres conditionally approved, 138,849 acres restricted and 23,384 acres classified as prohibited from shellfish harvest (longislandsoundstudy.net/). In NY, there were 412,018 certified acres (equivalent to approved acres in CT), 1,613 seasonal (harvest allowed only during specific months of the year – similar to conditionally approved in CT), and 75,499 acres where shellfishing was prohibited (http://longislandsoundstudy.net/). Within the approved and conditional acres in the CT side of LIS, there were a total of 66,042 acres of leased shellfish growing area; 49,463 state leased and 16,579 town leased acres (K. DeRosia-Banick, CT Department of Agriculture, Bureau of Aquaculture, pers. comm.)

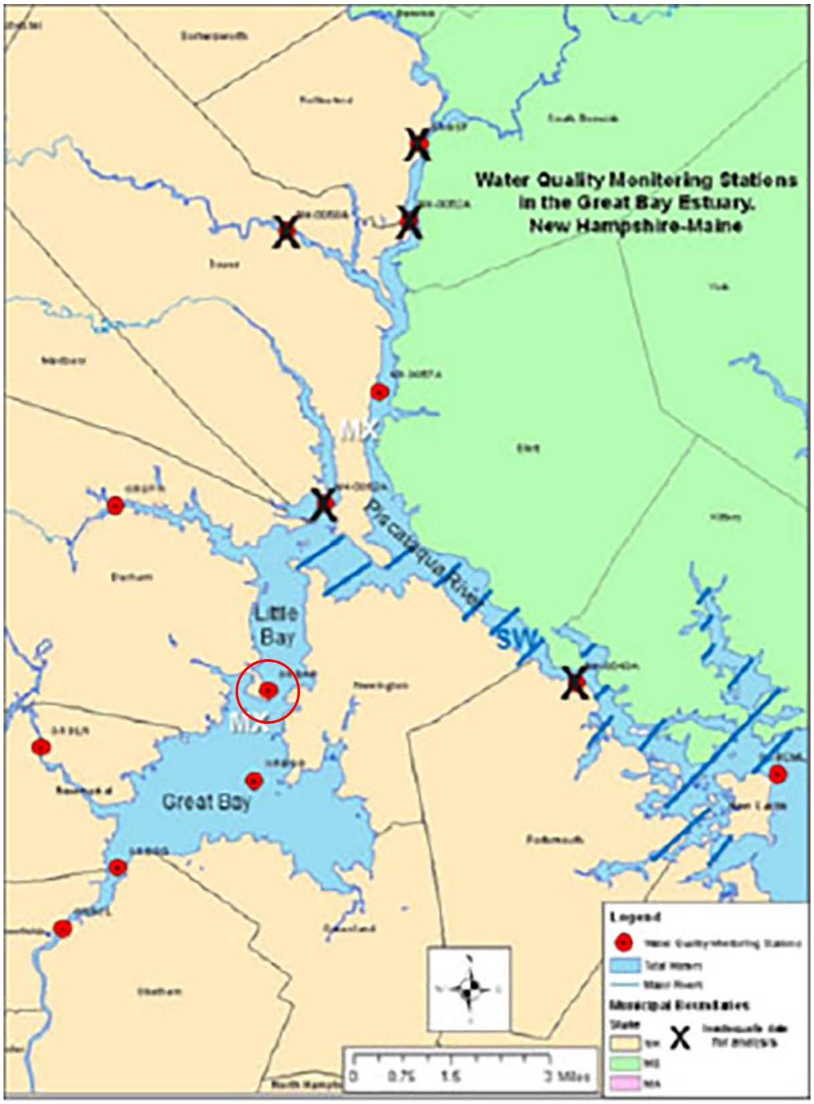


Figure 4: Map of Great Bay/Piscataqua Region Estuaries (upper panel). Sampling stations marked with X had inadequate data for analysis and were not included in the assessment. Data from the Adam's Point sampling station (circled) is used to illustrate the statistical and other analyses. The seawater zone (>25 psu) is marked with blue hatchmarks, the rest is considered mixing zone (0.5 – 25 psu). Sampling stations are indicated as red dots. Data from 7 stations were used to represent mixing zone (NH-0057A, GRBOR, GRBAP, GRBLR, GRBCL, GRBSQ, GRBGB) and one station was used to represent seawater zone (GRBCML); Location map (lower panel) of Great Bay / Piscataqua Estuary within Northeast region.

In these areas, oysters and clams are harvested commercially (http://longislandsoundstudy.net/) and shellfishermen can only harvest on their own leased beds. In CT there are ~43 individuals and businesses that hold leases on shellfish beds (D. Carey, Director CT Department of Agriculture Bureau of Aquaculture, pers. comm.). In NY, with the exception of one lease in Oyster Bay, 'baymen' can harvest shellfish in any approved waters with the proper permits, including state waters outside the Long Island Sound watershed. One major NY lease holder accounts for the majority (~80%) of the cultured shellfish landings, with several smaller companies contributing to the total (R.Rheault, Northeast Shellfish Growers Association, pers. comm.).

Great Bay/Piscataqua Region Estuaries

The Great Bay/Piscataqua Region Estuaries (GBP) is a relatively small estuarine system of 54.7 km^2, located between New Hampshire and Maine. It is tidally dominated and is composed of the Piscataqua River, Little Bay and Great Bay areas (Table 2). Eight major rivers, as well as several small creeks and their tributaries, drain into the Estuary. GBP is a well-mixed estuary due to tidal height and velocity as well as estuary geometry, though there can be a moderate level of stratification in some areas during high flow periods. Average depth is about 4 m and average salinity is 21 psu. The Great Bay National Estuarine Research Reserve (NERR) is part of GBP and includes 21.4 km^2 of tidal waters and mudflats, and 77.2 km of shoreline (GBNERR, 2011). The GBP watershed area covers 2,651 km^2 and includes parts of 57 towns in Maine and New Hampshire. Approximately, 287,700 people lived in the GBP watershed in 2010 (P. Trowbridge, N.H. Dept. of Environmental Services, pers. comm.). The population of the Piscataqua Region (which includes the GBP watershed and the watershed for the Hampton-Seabrook Estuary in New Hampshire) grew from 109,861 in 1930 to 377,427 in 2010. Noted for its valuable water resources, cultural resources, and business and industry, the Piscataqua Region is very important to state and local economies.

Table 2: Characteristics of the Great Bay / Piscataqua Estuary.

Great Bay/Piscataqua Region Estuaries Characteristics	Value	Source
Watershed area (km^2)	2,651	Table 6, NHEP (2007)
Estuarine area (km^2)	54.7 (20.3 Seawater zone, 34.4 Mixing zone)	Table 6, NHEP (2007)
Watershed population (10^3)	287.7	P. Trowbridge (N.H. Dept. of Environmental Services, pers. comm.)
N load (metric ton N/y) 2009-2011 (32% WWTP, 68% non-point)	1,113 (as TN) 542 (as DIN)	PREP (2013)
Tidal range (m)	2.0 – 2.6 (average for estuary) (Spring tide 2.5 m, Neap tide 1.5 m)	Table 6, NHEP (2007)
Estuary volume (10^6 m^3)	196	Table 6, NHEP (2007)
FW inflow (m^3/s)	49.3	Table 6, NHEP (2007)
Average depth (m)	3.2	Table 6, NHEP (2007)
Average Sal.	26.0	Table 6, NHEP (2007)
Flushing time (d)	28.5	Table 6, NHEP (2007)
Aquaculture area (acres) 5 leases at present	25.5	R. Grizzle (University of New Hampshire, pers. comm.)

Nutrient loading and eutrophication status history

In 2009 – 2011, total nitrogen load to Great Bay/Piscataqua was 1,225 tons, less than the 1200 tons discharged to the bay in 2005-2006, the highest loads measured since the first measures in 2003 (PREP, 2013). The higher loads occurred during years of higher rainfall and the decreases are concurrent with drier years of less rainfall. Due to the variability in these data, no long or short-term trends can be determined. Non-point sources are the largest portion of the discharge of nitrogen to GBP, accounting for 68% of total inputs. Nitrogen from fertilizers from lawns and farms, septic systems, animal wastes, and air deposition onto the watershed is carried into the bay through rain and snowmelt runoff, river flow, and groundwater flow. The balance of nitrogen is discharged to the bay or into tributary rivers from 18 municipal sewage treatment plants (PREP, 2013). The major contributors of nitrogen to the bay are related to population growth and associated building and development patterns within the watershed. Presently, GBP exhibits many of the classic symptoms of eutrophication, including low dissolved oxygen in tidal rivers, increased macroalgae growth, and increasing occurrences of nuisance and invasive macroalgal species (PREP, 2013). Of major concern is the loss of eel grass which has declined by 35% since 1996 (Short, 2011).

The observed water quality issues led to the development of nutrient criteria to provide guidance for meeting water quality goals (NH DES, 2009). Additionally, the Piscataqua Region Comprehensive Conservation and Management Plan (CCMP) was updated in 2010 to help address issues impacting the water quality and environmental health of estuaries in the Piscataqua Region (PREP, 2010). Water quality conditions are expected to improve in the future as a result of planned management measures. Restoration of reefs and expansion of oyster aquaculture has been studied as a potential boost to traditional land based nitrogen management measures (Nash and Elliott, 2012; Konisky et al., 2014).

Shellfish cultivation history

The shellfish aquaculture industry in GBP is relatively small, but growing, and primarily focuses on oysters (R. Grizzle, pers. comm.). In 2011 there were five growers with a total of 14.5 licensed acres which grants growers permission (but not exclusive rights) to use public lands (land below the mean high tide line is public trust land in NH). In 2012, there were six growers and 17.5 licensed acres. In early 2013, there were 8 growers and 24.5 acres licensed, with a total of 25.5 acres by the end of 2013 when one of the existing growers was granted a new 1 acre site. Currently, growers are in the third year of culture with ~$8,000,000 in oyster sales for the year (2014). No farm is greater than 4.5 acres, and typically oysters are grown in cages or racks and bags. Cage culture produces oysters with thinner shells than bottom grown oysters; a typical harvest size GBP oyster is smaller (~70g) than the ~90g harvestable weight observed in LIS. For every oyster sold, the grower sends $0.015 to the state which supports the general funds of the New Hampshire Department of Fish and Game. Additionally, aquaculturists pay annual fees for their licenses ($100 license renewal fee) and certifications, as well as a per acre fee for their use of public land ($200 acre $^{-1}$). Presently most operations use rack and bag systems which are mesh bags inserted into cages, commonly referred to as 'oyster condominiums' (R. Grizzle, UNH, pers. comm.). While all present operations are subtidal, there is interest in expanding to intertidal areas. There are plans to expand aquaculture operations in the future, most likely bottom culture due to concerns (described as 'social carrying capacity,' Angel and Freeman, 2009) by waterfront landowners and boaters who do not want to see floating gear or risk boats getting entangled in culture gear.

METHODS AND SUPPORTING EXPERIMENTS AND DATA

Overview of Project Approach

This bioextraction study uses a combination of local and system scale circulation and aquaculture models, watershed models, field data and experiments and laboratory experiments (Figure 5) to evaluate the impacts of watershed discharges on water quality and the interaction of shellfish aquaculture and water quality. Changes in land-use and related changes in nutrient inputs can be analyzed by means of existing watershed models and the effects of the modified load can be simulated via system-scale ecological models, such as EcoWin.NET. Such models have been applied to the LIS Watershed (e.g. HydroQual, 2007). Results of those models can be combined with finer scale models (HydroQual, 2009), in order to capture small-scale variability relevant for long-term system simulation, and to allow multiple-year (decadal) simulations which are relevant to socio-economic timescales. Outputs of these models can then be processed in the Assessment of Estuarine Trophic Status (ASSETS; Bricker et al., 2003) eutrophication model to provide an aggregated image of nutrient related consequences to the system. The results from application to recent data were compared to previous assessments to evaluate trends. This screening model, together with socio-economic models, is a management-level tool to aid in decision-making, and provides a digest of the detailed outputs of the various research models.

The Farm Aquaculture Resource Management (FARM) model combines physical and biogeochemical models, bivalve growth models and screening models for determining shellfish production and for eutrophication assessment (Ferreira et al, 2007). The model simulates processes at the farm scale (about 100-1000 m in length), considering advective water flow and transport of relevant water properties. These include total particulate matter (TPM), phytoplankton and organic detritus, ammonia, and dissolved oxygen. The model is applicable to suspended culture from rafts or longlines, as well as to bottom or near-bottom culture. Horizontal water transport is simulated using a one-dimensional model, to which vertical transport is added for suspended culture. The Eastern oyster (*Crassostrea virginica*) was incorporated into the FARM model.

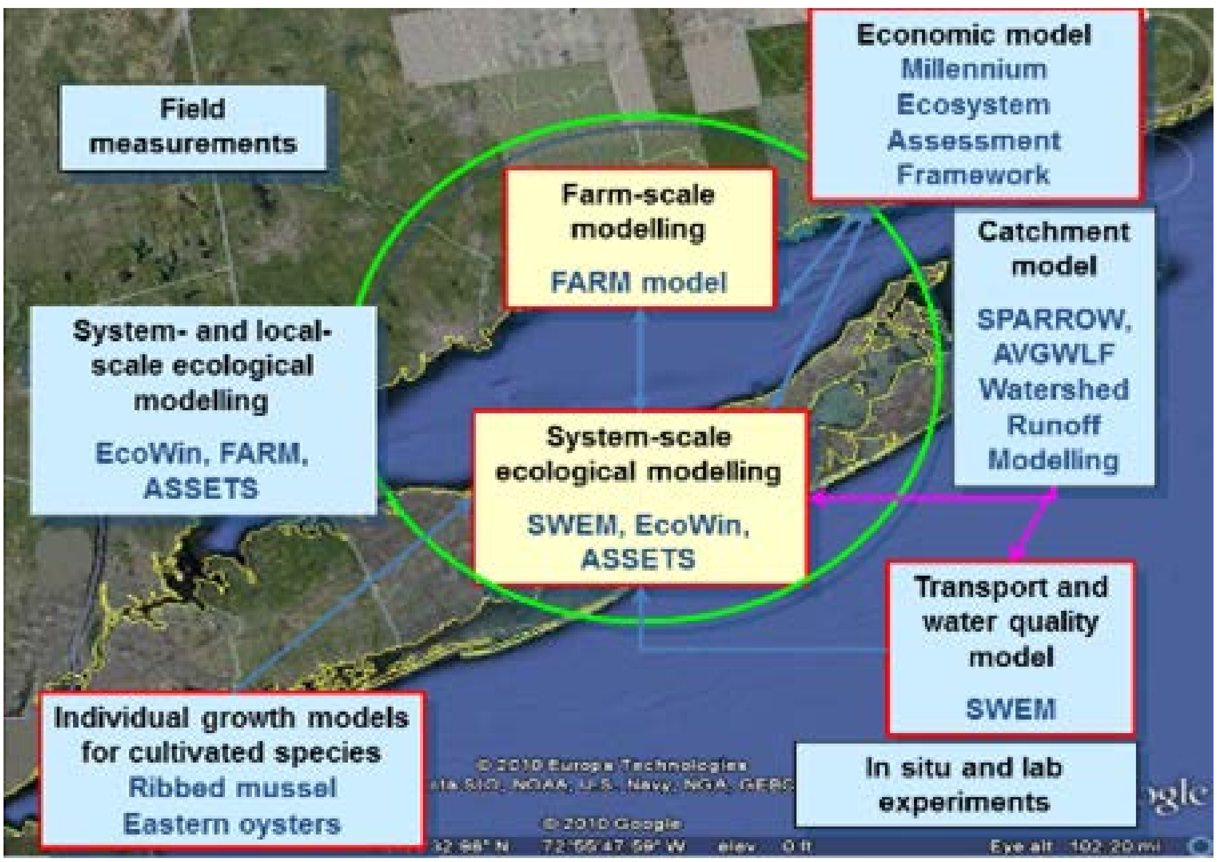


Figure 5: The Regional Ecosystem Services Program Bioextraction Framework.

Supporting Experiments and Data

Long Island Sound

Experiments

Experiments were performed to provide the basis for development of an individual growth model for *Crassostrea virginica*. It was necessary to incorporate the individual model into the FARM local scale model and the EcoWin ecosystem model in order to accurately simulate the population growth and harvest, as well as the removal of nitrogen from LIS at farm and system scales. While these models were able to simulate growth and other characteristics of several species of shellfish, until this study, they did not include *Crassostrea virginica.* These experiments helped to build out the models for fulfillment of project goals.

Filtration and feeding activities of young, adult oysters (36.8±9.8 mm mean shell height) were quantified using the biodeposition method (Galimany et al., 2011) at the dock of Hillard Bloom Shellfish Inc. in South Norwalk, CT on 18 and 26 October, 2012. Water temperature ranged from 15.2 to 17.3°C, dissolved oxygen was high (5.2-6.5 mg L^{-1}), and salinity was 20-23. On sampling days, chlorophyll a (hereafter CHL) levels were relatively modest (2.0-3.3 µg L^{-1}), but percentage of the seston accounted for by organic matter was reasonably high (23-31); therefore, representative filtration and feeding data were obtained.

Data from sixteen oysters on each of the two sampling days were very similar, providing confidence that data could be combined for subsequent use. Oysters cleared 3.27-4.26 L hr^{-1} g dry $weight^{-1}$, and had assimilation efficiencies of 53 and 55% on the two days, respectively. The data were considered to be appropriate for use by collaborators to parameterize the FARM model for CT coastal waters where oyster culture occurs (see section ‘AquaShell Individual Model’).

Data for application of AquaShell, FARM and ASSETS

Monitoring data for 2008-2012, obtained from Long Island Sound Integrated Coastal Observing System (LISICOS; http://lisicos.uconn.edu; M. Lyman, CT DEEP, pers. comm.) were used for the application of the individual oyster model, which was incorporated into EcoWin system-scale and local-scale FARM models. Additionally the data were used to evaluate eutrophic condition through application of the ASSETS model.

The data are a combination of monthly grab sample results (January – December) for CHL, temperature, salinity, and dissolved oxygen (hereafter DO) for 17 stations across the Sound (Figure 1). Other data and information needed to complete these analyses were acquired from the Long Island Sound Study (www.longislandsoundstudy.net/) and from M. Tedesco (LIS Study, pers. comm.). Data for years 2008-2012 were used in an attempt to use a robust dataset to provide an evaluation of the most recent conditions in the estuary and to reduce potential bias due to anomalous weather years. Data results from monthly temperature and salinity samples at 4 stations in LIS are shown for years 2008-2012 in Figure 6. Result of monthly CHL samples are shown in Figure 7, and DO concentrations in Figure 8 for the same stations and time-frame.

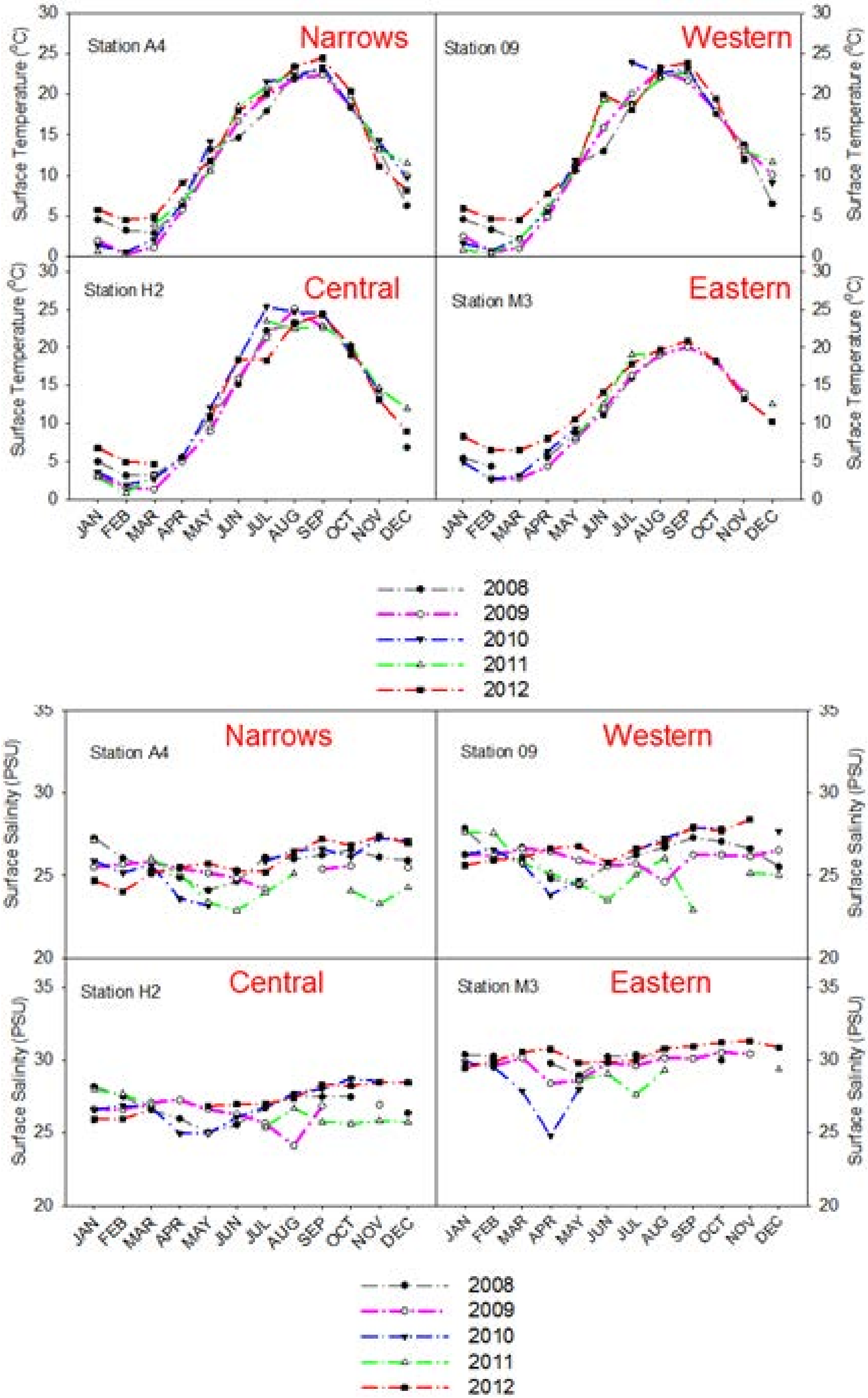


Figure 6: Monthly temperature (upper panel) and salinity (lower panel) at 4 LIS stations January - December, 2008 – 2012. (see Figure 1 for locations).

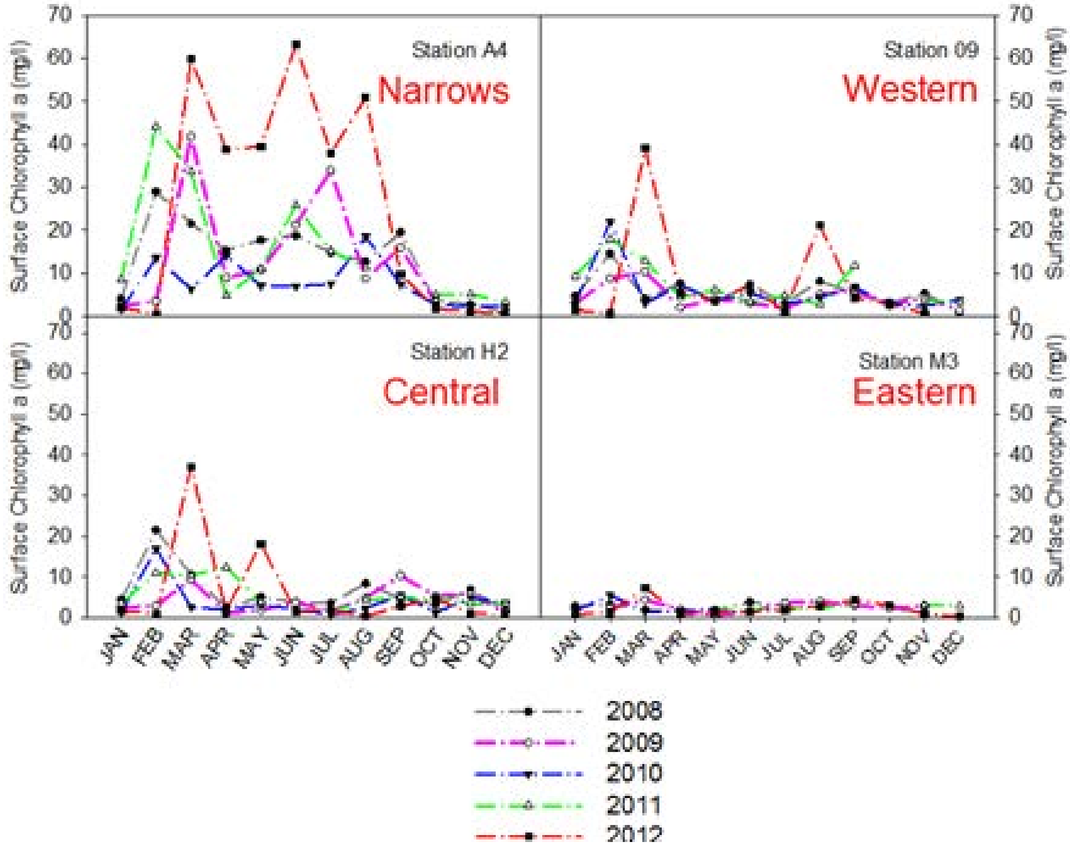


Figure 7: Monthly chlorophyll concentrations, January – December, 2008 – 2012 at 4 Long Island Sound stations (see Figure 1 for locations).

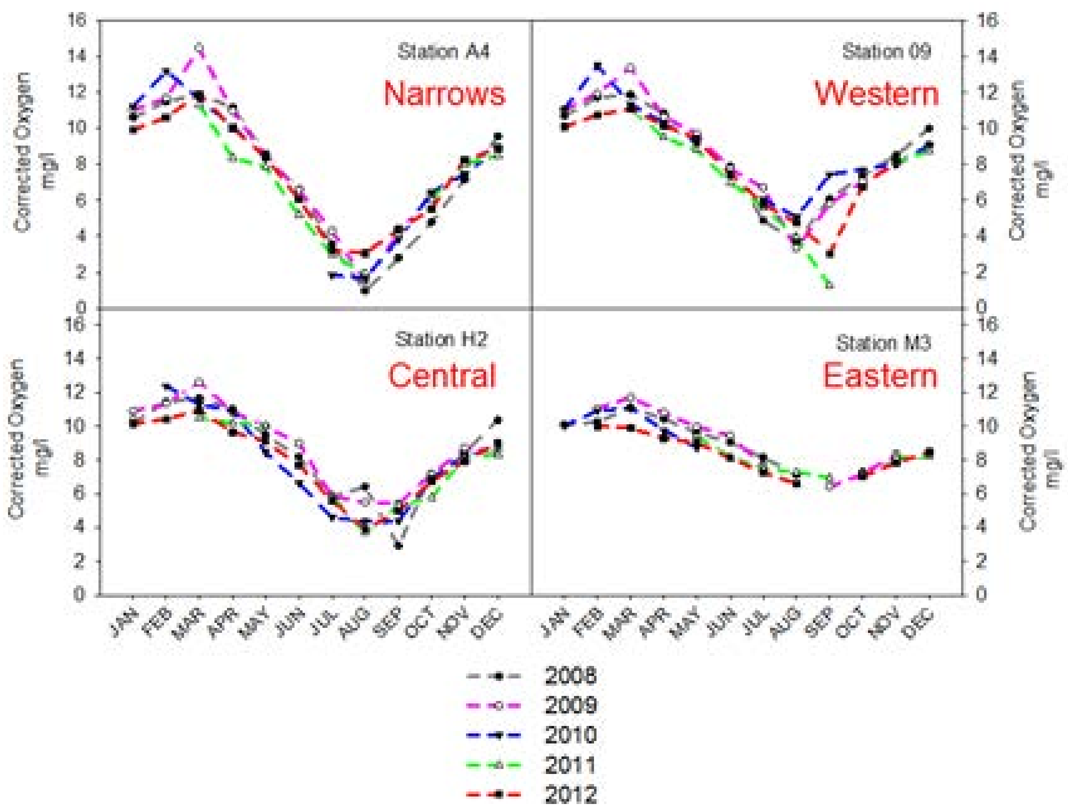


Figure 8: Monthly bottom water dissolved oxygen concentrations, January – December, 2008 – 2012 at 4 Long Island Sound stations (see Figure 1 for locations).

Statistical tests of Long Island Sound Data

A statistical analysis was performed on the data to rule out any trends in the dataset that might bias results. Jonckheere-Terpstra (JT) tests (Zar, 1999) were performed to detect trends through time for water quality data (temperature, salinity, CHL and DO) at each station. All data for each station were grouped by year. The null hypothesis for the JT test was that there was no trend through time. A standard α- level of 0.05 was used to define the threshold for acceptance or rejection of the null hypothesis. P-values greater than α- level of 0.05 indicated that there was no trend.

Great Bay/Piscataqua Region Estuaries

Experiments

Experiments were performed to provide validation of the growth and nutrient removal (nitrogen [N] and carbon [C]) capabilities determined for *Crassostrea virginica* in GBP by application of the FARM model. Hatchery-reared oysters were deployed for three years (2010 – 2012) at six sites in the Great Bay/Piscataqua Region Estuaries, New Hampshire, to determine nutrient removal through bioextraction (Appendix 3). Sites were chosen to represent a range of ambient nutrient concentrations, water flow conditions, and location within the estuary. Oysters were deployed in polyethylene-mesh bags typically used on oyster farms in New England. At the beginning of the study, approximately 1,000 "seed" size 0.3 year old (19-42 mm shell height) and 200, 1.3 year old (28-79 mm shell height) oysters were placed into two separate polyethylene bags which were suspended 10 cm off the bottom attached to plastic coated wire cages at each of the study sites. Samples of oysters from the bags were taken on seven occasions over the course of the 2.3-year study, and analyzed for N and C content and size.

A quadratic polynomial fit to the combined (all sites, both size classes) datasets for shell height indicated that on average a 'cocktail' size oyster (63 mm shell height) would be reached in the third growing season, and 'regular' size (76 mm) would require a full 3 years. However, there was wide variability in growth, with some individuals reaching regular size in the second growing season. There were significant differences in growth rates and %N and %C in shell and soft tissue among the study sites. Growth variations were positively correlated (but not significantly) with CHL concentrations (2.9 to 6.7 µg/L), and negatively correlated (but not significantly) with mean water flow speeds (14 to 35 cm/s). Percent N in soft tissue and shell were highest at the two sites with highest mean dissolved inorganic N (NO2/NO3 + NH4) concentrations in the water column, with means among the six sites varying from 6.9 to 8.4 %N in soft tissue, and 0.07 to 0.18 %N in shell. Grand means (all sites, seasons and years combined) of soft tissue N and C for regular size oysters were 7.3% and 38.5%, respectively; and for shell N and C were 0.13% and 12.0%, respectively. Although the overall means were similar to previous research in other areas, these data also indicate that oyster size, shell thickness, seasonality, and nutrient concentration in ambient water can strongly affect %N and %C content of oysters. These results provided data needed for validation of local scale FARM model simulation results.

Data for application of AquaShell, FARM and ASSETS

Monitoring data from stations in GBP for the years 2005-2010 were used for the application of the individual oyster model, the FARM model and to evaluate eutrophic condition through application of the ASSETS model. The data are a combination of monthly grab sample results for CHL, temperature, salinity, and DO from monitoring programs run by the National Estuarine Research Reserve (NERR), the University of New Hampshire (UNH) and the US Environmental Protection Agency (EPA). Of the 13 sampling stations in the Great Bay/Piscataqua Region Estuaries (GBP;), there were adequate data for stations: NH-0057A, GRBSQ, GRBCL, GRBAP, GRBOR, GRBCML, GRBLR, GRBGB. There were datasonde observations for the same time period for DO only at stations GRBLR, GRBOR, GRBSQ and GRBGB and GRBCML (Figure 4). For most of these stations data were collected monthly during April – December, though at station GRBAP, adjacent to the Jackson Estuarine Laboratory, data were collected monthly throughout the year. We use GRBAP for illustration of annual cycles below since it is the only station with data for all months. Data for years 2005-2010 were used for the eutrophication assessment in an attempt to use a robust dataset to provide an evaluation of the most recent conditions in the estuary and to reduce potential bias due to anomalous weather years. Data from monthly samples of surface temperature and salinity are shown in Figure 9 for 2005 – 2010 at Station GRBAP. Figure 10 shows monthly surface CHL data sampled at Station GRBAP for years 2005 – 2010. Figure 11 shows data from DO samples at the same station for the same time frame.

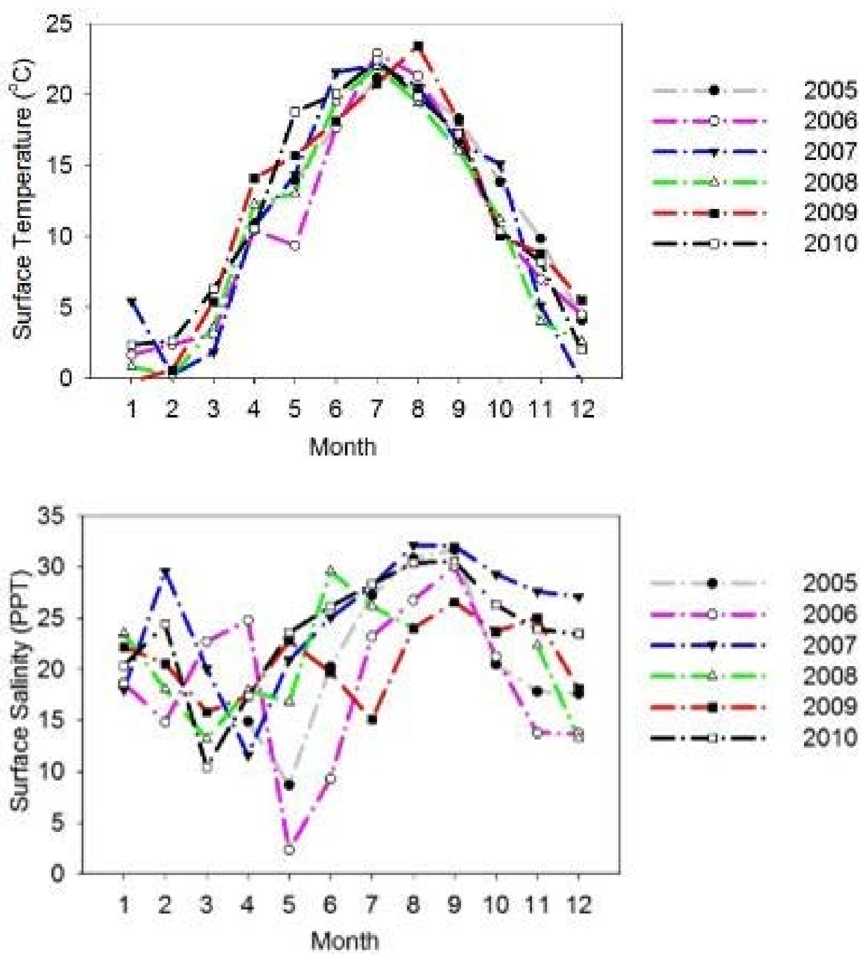


Figure 9: Monthly temperature (upper panel) and salinity (lower panel) in Great Bay/Piscataqua Region Estuaries at station BRBAP, located near Adam's Point the only station for which there are data for January - December, 2005 – 2010 (where 1- 12 = Jan - Dec).

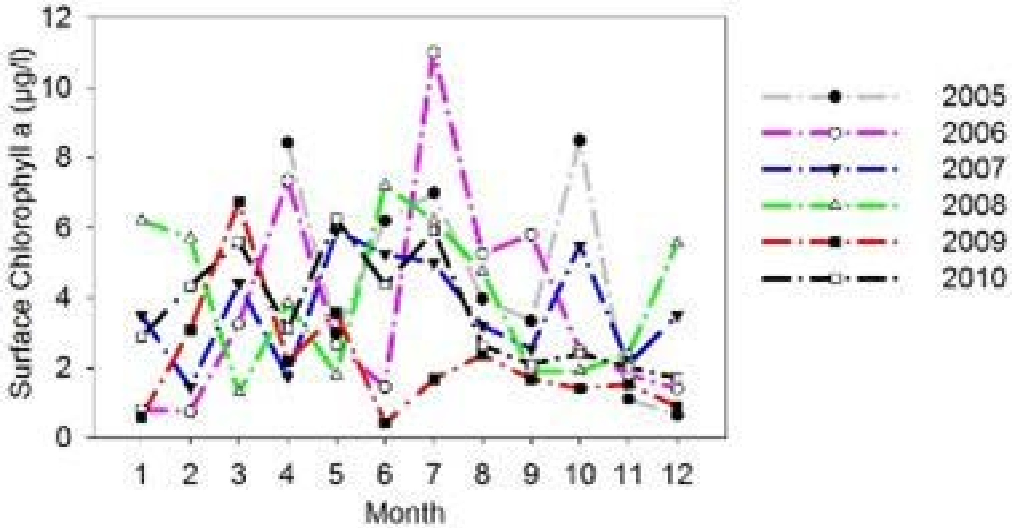


Figure 10: Great Bay/Piscataqua Region Estuaries station GRBAP monthly chlorophyll concentrations, January – December, 2005 – 2010.

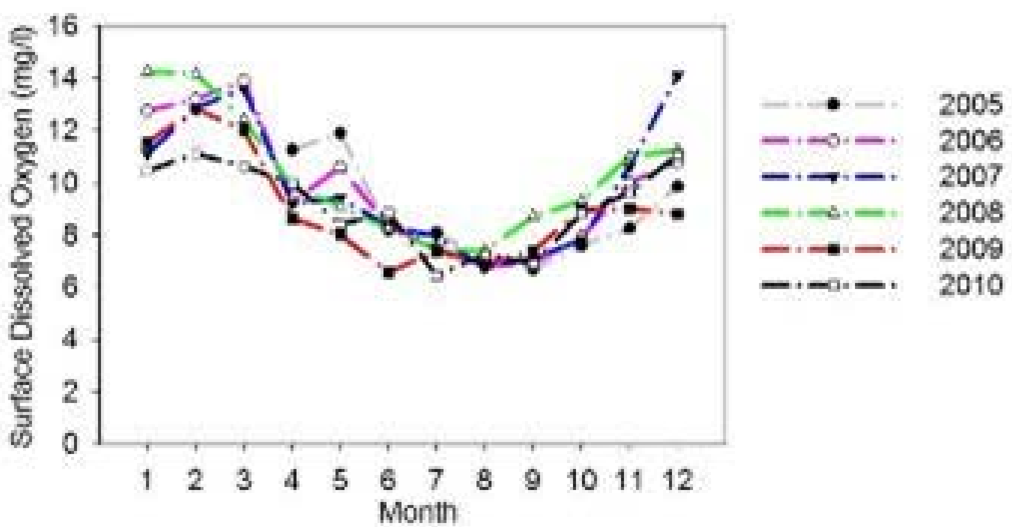


Figure 11: Great Bay/Piscataqua Region Estuaries station GRBAP Adam's Point dissolved oxygen concentration monthly, April – December 2005- 2010.

Stastical tests of Great Bay Piscataqua Data

A statistical analysis was performed on the data to rule out any trends in the dataset that might bias results. Jonckheere-Terpstra (JT) tests (Zar, 1999) were performed to detect trends through time for water quality data (temperature, salinity, CHL, and DO) at each station. All data for each station were grouped by year. The null hypothesis for the JT test was that there was no trend through time. A standard α- level of 0.05 was used to define the threshold for acceptance or rejection of the null hypothesis. P-values greater than α- level of 0.05 indicated that there was no trend.

Shellfish Aquaculture Practices and Cultivation Areas

The models used in this study to simulate current and potential oyster and clam production require data inputs to accurately represent the growth, harvest and the removal of nutrients from the water column. Environmental factors such as water temperature, current speed, salinity, and concentrations of food (i.e. CHL, particulate organic matter, total particulate matter) are measured regularly by the CT Department of Energy and Environmental Protection (DEEP) and the data are reliable and accessible. Information and data for shellfish cultivation practices are also required and are as important to the modeling as the environmental data. Information on shellfish culture practices, however, is not always accessible and is more complicated than the environmental data. Here the culture practices are described in detail. One of the more important model inputs is the area of active cultivation of shellfish and the density of oysters and clams that are planted on cultivated lease or licensed areas. Together, these are used to determine the total shellfish population that is filtering the water, helping to remove nutrients. Since this information was not available from the LIS growers, alternate methods of determination were required and are detailed below.

Long Island Sound

The shellfish aquaculture industry in Long Island Sound consists primarily of a traditional practice that has not changed significantly over the last century. Wild larvae are collected on shell that is spread in areas where larvae are known to set well (typically in shallow beds around the mouths of rivers where salinity is slightly lower). A significant portion of the CT industry relies on wild sets of oyster seed in waters that are restricted from harvest for direct human consumption because the waters do not meet the sanitation standards set by the Interstate Shellfish Sanitation Conference (ISSC). These oysters must be ‘relayed’ in clean waters for a period of weeks to allow them to cleanse any bacterial and viral contaminants before they can be sold. A smaller percentage (about 10%) of LIS oyster producers use hatchery-reared seed and ‘intensive’ culture methods (i.e. using gear) involving floating upweller systems (FLUPSY) and cages. These different approaches involve different planting densities and harvest practices, and can yield very different products targeting different markets.

Traditional ‘extensive’ oyster culture starts by placing ’cultch‘ shell in shallow seed beds that are known to attract good sets of larvae. Producers and state managers place millions of bushels[3] of dried shell in these beds in late spring as substrate for larval settlement in July of each year. In late fall or early spring the seeded cultch will be harvested and sold to growers who will plant the seed (now approximately an inch in length) on deeper water leases typically in open waters. For two to three years the oysters are tended, periodically moved from bed to bed, sorted and fattened until they reach an average legal harvest size of about 90 grams and at least 3 inches length (Figure 12). Leaseholders will sometimes drag starfish mops over the beds to tangle these oyster predators so they can be brought to the surface and killed in vats of boiling water.

A substantial fraction (50-70%) of the CT oyster harvest depends on the harvest and replanting of oysters from restricted waters to clean waters for final grow-out and subsequent sale. These oysters have been growing in waters that are contaminated with bacteria and viruses so they are not fit for direct human consumption until they have been relayed in clean waters for a period of weeks to allow them to purge their contaminants. Typically, oysters harvested from these beds are a collection of various sizes ranging from 1 to 3 inches, or occasionally larger. Larger animals are planted at high densities in clean water leases and can be harvested just a few weeks later. Smaller seed are planted at lower densities and replanted for 1-2 additional years of growth (Figure 12).

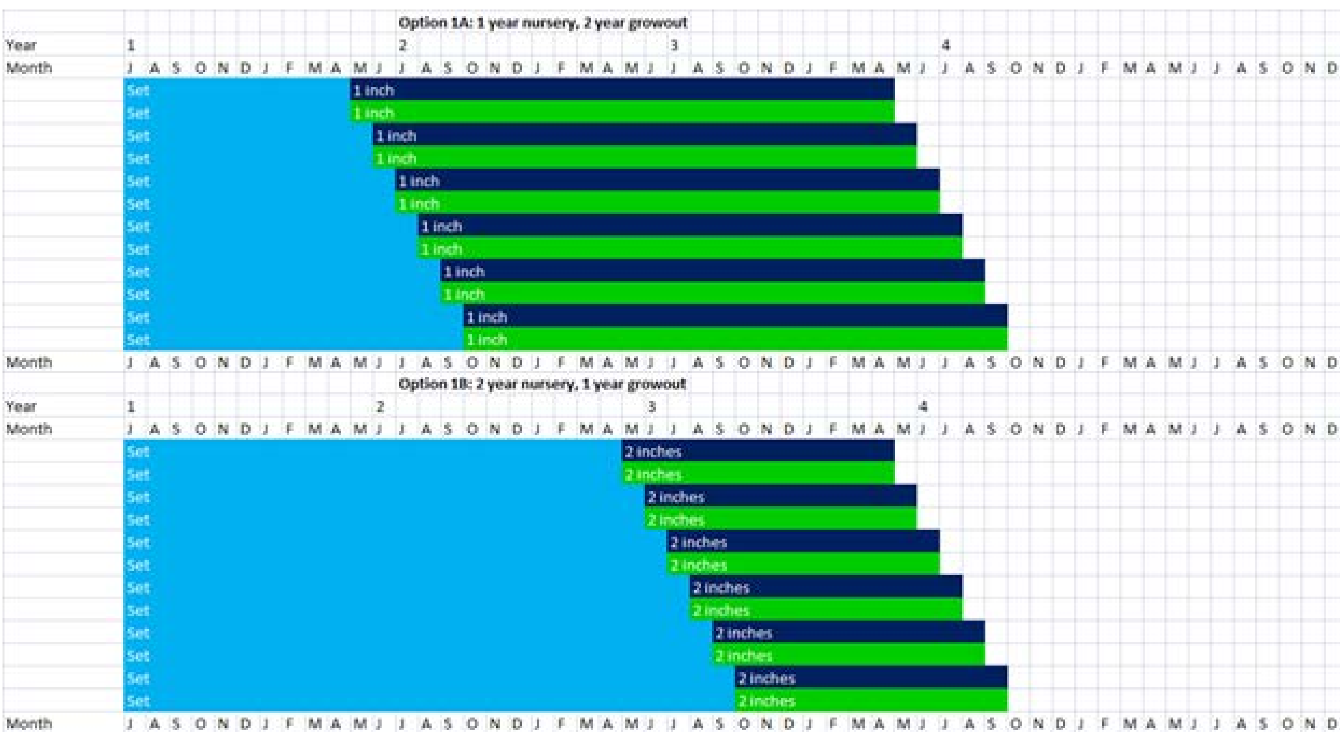


Figure 12: Bottom culture (extensive), using no gear: one inch seed with two year growout (upper panel), two inch seed with one year growout (lower panel).

[3]The number of fresh oysters in one bushel varies dependent on size and type of oyster. A bushel will hold ~250 wild caught three inch (76 mm) oysters while there are ~300 per bushel of more uniform size cultured oysters and sometimes more if oysters are flat (Terry Witt, Chesapeake Bay grower, pers. comm.; Karl Roscher, Aquaculture Division, MD Department of Natural Resources, pers. comm.; Don Webster, Aquaculture Specialist, MD SeaGrant Extension Program, pers. comm.). There are about 650 shells to a bushel of cultch used for spreading on beds for substrate (Terry Witt, Chesapeake Bay grower, pers. comm.)

Several small-scale growers in CT have adopted intensive culture methods that are increasingly com mon in New England. These growers purchase hatchery-reared seed at a size of 1-8mm often placing these seed into powered nursery systems called upwellers. These consist of baskets that hold the small seed while water is pumped up through the bed of seed delivering food-rich waters. Once the seed reach 12-30 mm they are transferred to plastic mesh-bags or cages for further growout (Figure 13). Many of these growers transfer the seed to the bottom for final growout once they are greater than 30-60mm, a size at which they are more resistant to predation by crabs and starfish. Some growers leave them in the cages until they reach market size.

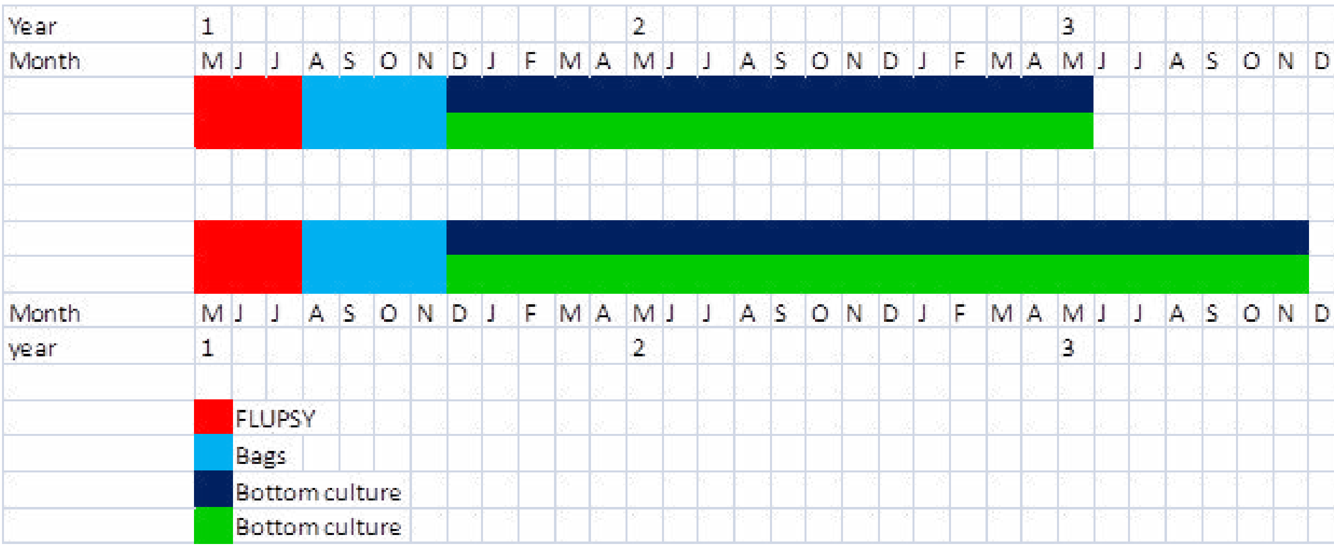


Figure 13: Bottom culture (intensive), using gear in CT waters.

These different culture methods yield oysters with different shape and weight and often end up in different markets with substantially different prices. Traditional bottom-grown oysters are slower growing, less uniform in shape, and have a thicker shell. These oysters are typically harvested at a slightly larger size (4.5 inches, ~90 g) after 2-4 years on the bottom, fetch a slightly lower price ($0.40) and are often used in cooking or shucking.

Intensive growers have perfected the production of uniform, single oysters that are more suitable to the high-end rawbar trade. Using intensive culture and selected genetic lines that are bred for fast growth and disease resistance, these growers can produce a 3.5 inch oyster in 18-30 months. When growers hold the oysters in cages right up to market size the oysters will typically have a much thinner shell than a bottom grown oyster. Many growers elect to finish their oysters by placing them on the bottom for the last 6-12 months of growth before harvest to produce a product with a thicker, harder shell and a fuller meat. These oysters are typically harvested at a smaller size (~70 g) and fetch a higher market price ($0.50 or more).

The traditional bottom grown oyster culture dwarfs the production of the intensive culture practice. Intensive growers have much higher up-front costs including seed, cages, and the labor to keep the gear clean. With lower capital costs and lease acreage many times greater than the small-scale intensive farmers, the traditional growers typically produce 10-20 times the oyster harvest of intensive growers.

The traditional harvest, however, is highly dependent on the vagaries of the natural sets of wild oyster larvae. Following strong sets the traditional growers do well, but if sets are poor for several consecutive years their harvests are very lean.

Determination of cultivation area in CT waters of Long Island Sound

As noted above, to assure that we are modeling nutrient removal through shellfish aquaculture as accurately as possible, the acres of lease areas under cultivation must be known with confidence. Although the amount and location of lease acres are known, the lease areas actually cultivated are not, thus an alternate method of determination of cultivation area must be employed. Harvest amounts and size of a harvestable oyster and clam is also important, but since 2007 harvest data have not been reported by any CT growers. Furthermore, since there were no data on lease acres or harvest available from NY we consider only the CT side of LIS in the further modeling effort and in the determination of cultivated areas.

Two methods were used for determination of the lease area that is cultivated for oysters and clams in CT; 1) a reverse calculation using the reported value of harvested oysters and typical density of oysters in cultivated areas, and 2) a bracketing approach using typical percentage of lease area reported by growers where an upper, mid and lower range was used to try to address the uncertainty in cultivated area. The percent of total harvest within the different parts of the Sound (D. Carey, CT Department Aquaculture, Bureau of Aquaculture, pers. comm.) was used to area weight the results of these two methods. The areas determined to be cultivated for each modeling box were used in the EcoWin ecosystem model simulations (see section below: EcoWin.NET model) to determine current harvest and N removal by oysters and clams.

Method 1: Determination of cultivated oyster acres using harvest estimates

Harvest estimates for the CT oyster and clam industries were determined by extrapolating reported harvest numbers from the CT Department of Agriculture (http://www.ct.gov/doag/cwp/view.asp?a=1369&q=271358).

Oyster Harvest

Oyster landings collapsed in 1997 after a disease epizootic. Growers and managers observed several years of strong natural sets between 2004 and 2010 (Figure 14) which was anticipated to result in strong seed planting and increased landings approximately three years later. Growers who responded to a survey (Appendix 2) reported doubling of landings over the same period. Based on their reports, in 2012 an oyster harvest valued at \$21 x10^6 is estimated. Considering an average price of \$0.40 per oyster, 52 million oysters were harvested.

Further, based on a typical density of 30 oysters m^{-2} in CT oyster leases (D. Carey, CT Department of Agriculture, Bureau of Aquaculture, pers. comm.), the estimated area for oyster cultivation is 440 acres planted. This seems low given that there are approximately 70,000 acres of shellfish lease area. However, it is likely that many acres are planted at greater densities, especially for relay oysters that are collected in restricted waters at a large size and grown out in approved areas at 100-200 m^{-2} for short durations (1-2 months), while a much larger area is planted at 20-40 m^{-2}, harvested two years later at only 10 oysters m^{-2}. By comparison, resource managers believe that there are 400 to 1400 acres being planted with oysters.

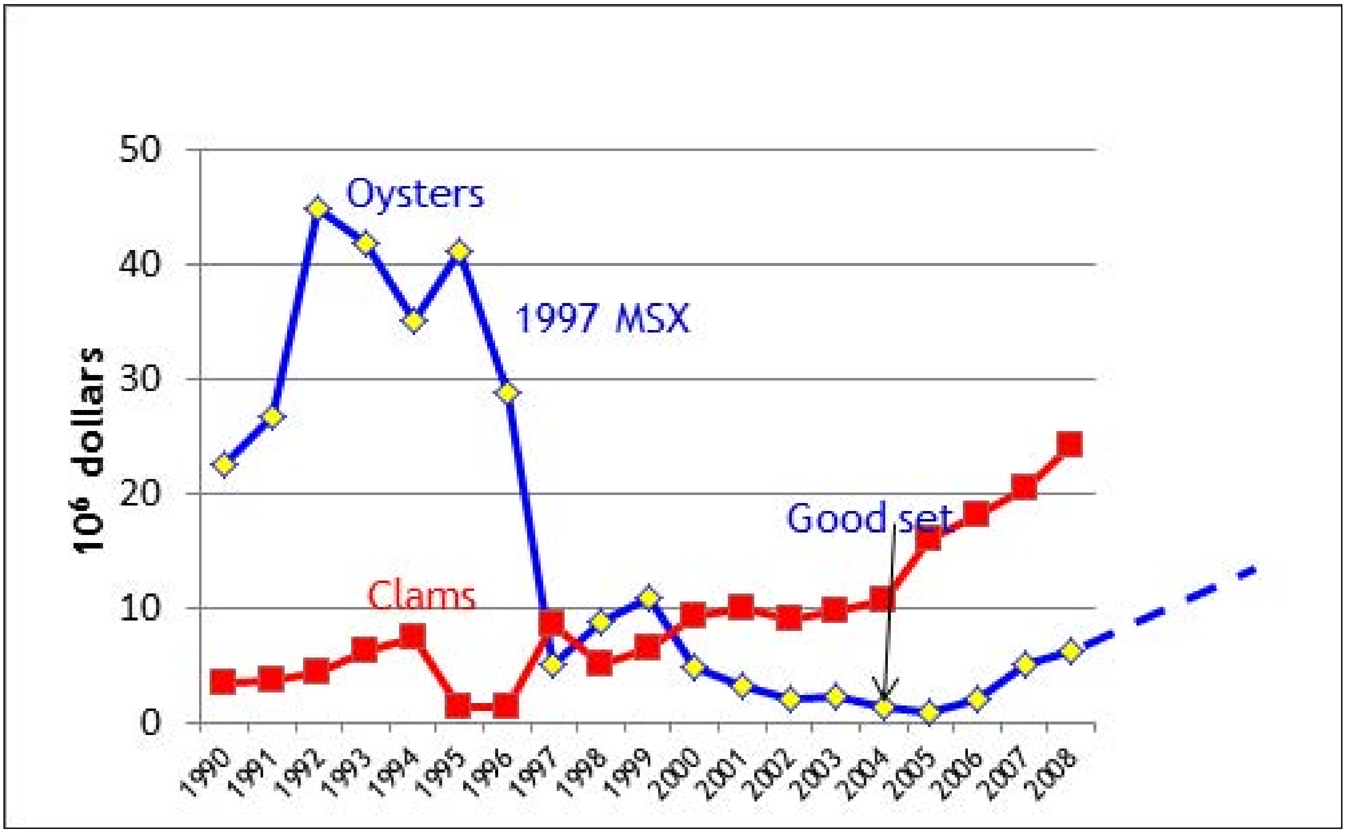


Figure 14: Value of Connecticut Clam and Oyster Landings 1990-2008.

Our model also needed the spatial distribution of total annual harvest within LIS. Resource managers describe the percentage distribution of total harvest among areas of the Sound based on knowledge of harvest (i.e. landings) in each town. These percentages were used to allocate the percentage of total oyster harvest to EcoWin model boxes that include lease areas (Table 3; note: the grid of final harvest may not be the same as the grid where the oysters were initially harvested for restricted relay):

Table 3: Distribution of total oyster harvest in EcoWin model grid boxes

EcoWin model grid, box	Percentage of total oyster harvest produced in EcoWin grid box
Grid 2, 23	2
Grid 4, 25	77
Grid 6, 27	1
Grid 9, 30	15
Grid 12, 33	1
Grid 20, 41	4

The average market weight of a harvestable size oyster was determined by several dealers who note that although the seasonal weight of a hundred-count bag varies, an industry year-round average estimate for bottom-grown CT oysters is ~20lb (9.1 kg). This yields an average legal harvest size oyster (at least 3 inches = 7.62 cm) that weighs 91 grams (0.02 lb).

Clam Harvest

Clam harvest estimates were projected forward from the most recent data collected assuming that landings are dependent on factors including: openings of restricted relay grounds, extended closures due to significant rain events (i.e. Hurricane Irene) and that clam landings drop when oyster harvests are strong. The extrapolated value[4] for 2012 clam landings was $18 x 10^6. Based on the value, the typical ratios of different clam sizes (29% necks, 51% tops, 16% cherries, 4% chowders), and prices of clams, the total clam harvest was estimated at 109 x 10^6 clams. Growers in 2013 noted that a large percentage of harvested clams (50-70% of total landings) are taken from restricted relay waters and are typically planted in open waters at very high densities (100-200 m^{-2}) to make them easier to retrieve at harvest. There is a very small area of relayed clams planted at very high densities and a much larger area being harvested at very low densities. For the majority of CT beds that are dependent on catching a wild set of clams, typical densities are probably similar to the regional average of 3-10 m^{-2} (R. Rheault, East Coast Shellfish Growers Association, pers. comm.). The percentage distribution of total clam harvest among areas of the Sound was determined by resource managers in the same manner as for oysters. The percentage distribution of the total clam harvest into the EcoWin modeling boxes to use for determination of the acres in cultivation based on the total areas available is shown in Table 4.

Table 4: Distribution of total clam harvest in EcoWin model grid boxes.

EcoWin model grid, box	Percentage of total clam harvest produced in EcoWin grid box
Grid 2, 23	10
Grid 4, 25	25
Grid 6, 27	28
Grid 9, 30	32
Grid 12, 33	4
Grid 20, 41	1

It is also key for model simulations to know the planting density of clams. It is estimated that 60% of the harvest (833.4 million animals) is planted at a density of 100 m^{-2} on 2,059 acres, i.e. 8,333,788 m^2, mostly located in grid boxes 23, 25, 27 and 32. Some 40% (554 x 10^6 animals) are harvested at a density of less than 10 m^{-2} suggesting a cultivation area of approximately 13,691 acres. Using a more conservative

[4] *Landings were extrapolated by CT Bureau of Aquaculture & Laboratory Services, NYSDEC Shellfish Division because no CT clam data were reported since 2010.*

estimate for wild clam density of 3 m^{-2}, the average in Narragansett Bay (R. Rheault, East Coast Shellfish Growers Association, pers. comm.), would suggest a greater acreage being harvested, about 40,000 acres.

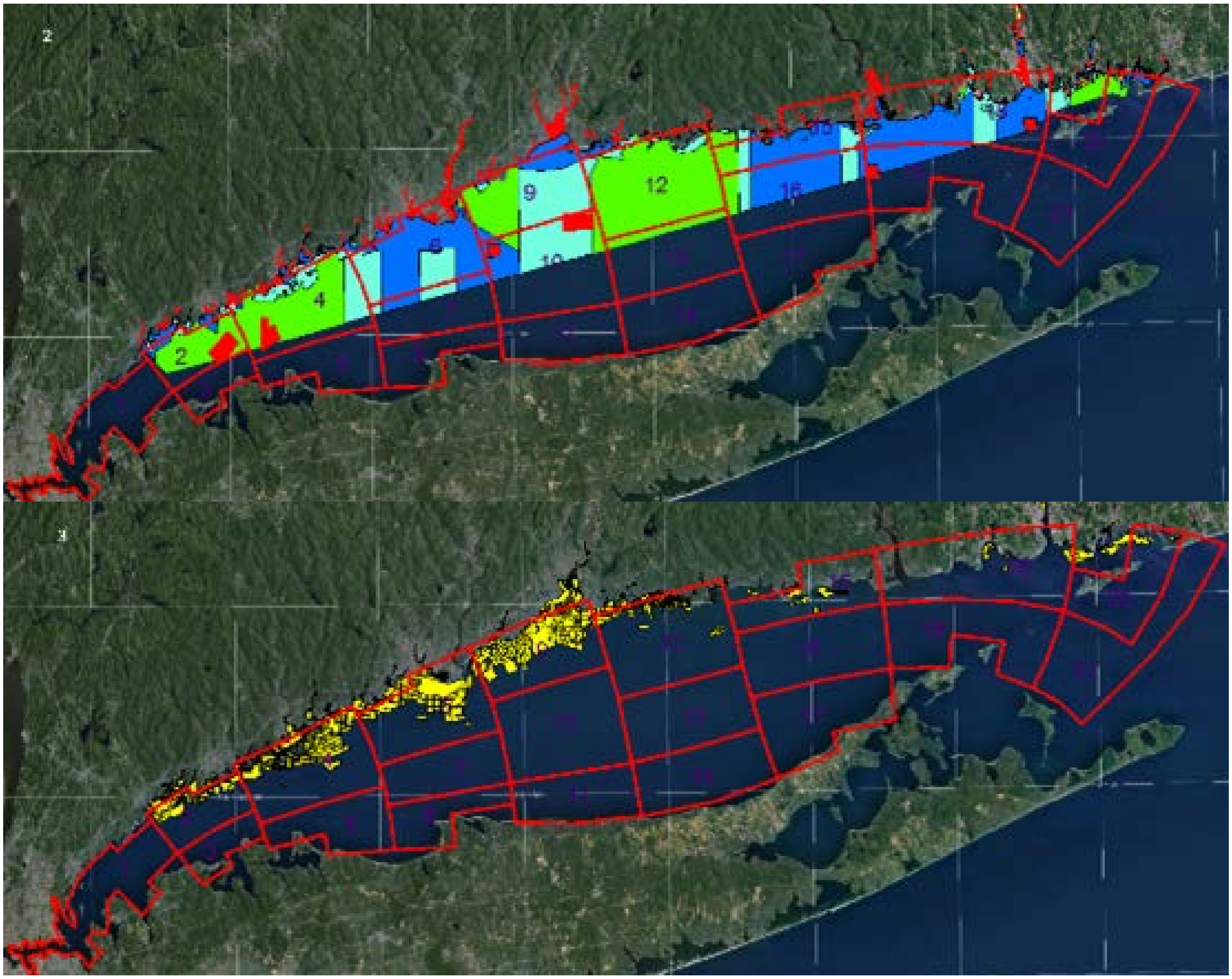


Figure 15: (upper panel) Total classified shellfish area, 351,728 acres (includes all classified waters: approved (green),conditional (dark blue), restricted (light blue), prohibited (red)) (managed shellfish beds data for 2011 from CT Department of Energy and Environmental Protection); lower panel) lease areas in yellow = 61,185 acres (includes all classified lease areas: approved, restricted, conditional, prohibited) (classified shellfish bed data for 2011 from CT Department of Energy and Environmental Protection).

This is still a small fraction of the acres under lease, for oysters it is less than 1% of the known lease areas (61,185 acres including all classified areas: approved, conditional, restricted, prohibited) based on maps of managed shellfish beds from CT Department of Energy and Environmental Protection (Figure 15). These results lead to the conclusion that many of these acres are rarely worked. It seems the majority of clam leases in CT are used to harvest wild clams at relatively low densities. The majority of these leases would be softer bottom leases, thus probably not suitable for conversion to oyster planting.

The results of the calculations from estimated harvest and income provide the probable acres under cultivation for oysters (440) and for clams (600-1200) with the distributions of total harvest noted above.

Method 2: Determination of area of cultivation from grower interviews and total known lease area

The second method used to determine cultivated lease acreage was from the total known lease areas and the typical percentage cultivated based on phone interviews with a subset of CT shellfish growers (Table 5). Much of the information provided by the growers confirmed the culture practices identified in the first method.

Table 5: Questions and responses from phone interviews with CT shellfish growers. (contact list provided by K. DeRosia-Banick, Department. Agriculture, Bureau of Aquaculture).

Questions	Answers
1. How many acres do you lease?	<30 - >10,000
2. What percentage of your lease acres do you actively use?	80% - 100% Most said that they used all the lease, all the time but the larger growers were using parts of the lease for shell, relay, holding product in places where they would grow slowly during bad market times. The smaller growers were using most of the lease most of the time for active cultivation and harvest.
3. What is the split between oysters and clams on your lease acres?	Highly variable, but no grower used more than 50% for oyster: Oyster 5% - 50% Clam 50% - 95%
4. Do you harvest and sell all year?	All growers harvest and sell all year
5. What is the 'high time' for sales?	Holidays, October - March
6. Do you 'rest' any part of your leases on an annual basis?	Not typically. Most growers said they only rest parts of leases when they do not produce, they use as much as possible all the time (see Q2)

Based on these conversations a bracketing approach was used to provide an upper and lower limit of probable acres of lease that would also serve to highlight the uncertainty in these estimates. For the lower side of the bracket we used 440 acres for oysters and 600-1200 acres for clams as determined in Method 1. For the upper limit of lease area the 61,185 acres of total lease areas were used, and for a middle estimate 42,000 acres was used. Note that these are total lease areas, not the area being cultivated (see Table 6).

For the area actively being cultivated, half of the total lease areas were considered, i.e. 30,593 acres for the upper and 21,000 acres for the middle estimate, since there are many reasons that the entire area is not cultivated, including unsuitable bottom area, use for shell storage or other activity, etc. The final three proposed areas of lease that are under cultivation for use in the EcoWin model simulations are; 440 (lower), 21,000 (mid), 30,593 (upper; Table 6). The mid lease area (i.e. 21,000) is proposed as the standard for the EcoWin model to calibrate and validate against. The growers used anywhere from 5% to 50% of their lease acres for oyster cultivation, the rest (i.e. 50% - 95%) is used for growing clams. To streamline the modeling but keep the same approximate ratio, 25% of the area was designated for oyster and 75% for clam cultivation (Table 7, Table 8).

Table 6: Lease areas, cultivated areas for use in EcoWin modeling.

Lease area bracket	Total lease area	Area in cultivation	Area in oyster cultivation = 25% (acres)	Area in clam cultivation =75% (acres)
Low		1040 - 1640	440	600-1200
Mid	42,000	21,000	5,250	15,750
High	61,185	30,593	7,648	22,945

Information on seeding density was not available thus the numbers identified in Method 1 (i.e. 62 individuals m^{-2} with 30 oysters m^{-2} at harvest, 55% mortality) were used with the lease acres shown in Table 6. Mortality of cultivated oysters in LIS is highly variable, depending for example on hurricanes and disease outbreaks. A value of 55% was used as an average, based on a reported range varying between 30-80% over the culture cycle. It was not possible to obtain any further information in grower interviews. The distribution of lease acres among the EcoWin model boxes was allocated according to Method 1 with resultant cultivated acres within each box shown for oysters in Table 7 and for clams in Table 8. Note that EcoWin does not explicitly address clam growth—the role of quahogs in removal of particulate organic matter (including phytoplankton) is, however, explicitly simulated in the model through inclusion as a wild species. For clam seeding densities we used a weighted average of 62 individuals m^{-2} based on the observations in Method 1 that 60% of clams are planted at 100 individuals m^{-2} and 40% at 10 individuals m^{-2}.

Table 7: Distribution of acres in oyster cultivation (based on percent total harvest within each grid box) in EcoWin grid boxes and areas of each used in calculations of the low, standard, high and potential cultivation scenarios.

OYSTERS EcoWin grid box	% total acres	low (440 acres)	standard (5250 acres)	high (7648 acres)	potential (11,116 acres)
2, 23	2	8.80	105	153	222
4, 25	77	339	4043	5889	8559
6, 27	1	4.40	53	76	111
9, 30	15	66	788	1147	1667
12, 33	1	4.40	53	76	111
20, 41	4	17.60	210	306	445

Table 8: Distribution of acres in clam cultivation (based on percent total harvest within each grid box) in EcoWin grid boxes and areas of each used in calculations of the low, standard, high and potential cultivation scenarios.

CLAMS EcoWin grid box	% total acres	low (440 acres)	standard (5,250 acres)	high (7,648 acres)	potential (11,116 acres)
2, 23	10	44	525	765	1112
4, 25	25	110	1313	1912	2779
6, 27	28	123	1470	2141	3112
9, 30	32	141	1680	2447	3557
12, 33	4	18	210	306	445
20, 41	1	4.4	53	76	111

Determination of potential/expanded lease area, cultivated area

One of the questions this study addressed was the potential removal of nitrogen in the case where lease areas were to be expanded. A scenario of what might be possible was evaluated using expansion of potential lease acres based on the 40 foot contour within the Sound, CT side only, to calculate the number potentially cultivatable acres (from K. DeRosia-Banick, CT Department of Aquaculture, Bureau of Aquaculture, pers. comm.). The calculation is based on all beds within the contour excluding all current beds, natural and managed (note that no one can lease designated natural beds) for an estimated additional 88,926 acres that could potentially be cultivated. No analysis was done of the suitability of bottom, and therefore the same logic used for current lease areas was used to determine potential areas, i.e. that only half of the area is suitable for expansion, or 44,463 acres. Of those acres, the same proportion for oysters (25%) and clams (75%) as for current leases was used to calculate a potential additional 11,116 acres for oyster culture and 33,347 acres for clam culture. For consistency, the same seeding densities and mortalities used in the current scenarios are proposed for the expansion n scenario.

Based on the results of this analysis, the following were used for simulations within EcoWin for LIS:

- Bracketing of cultivated lease acres to represent low, mid, high cultivation area for EcoWin model for oysters and clams (Table 8).
- The distribution of production in EcoWin boxes for oysters and clams determined in Method 1 (Table 3, Table 4).
- Use 30 oysters m^{-2} oyster density at harvest (starting density 62 oysters m^{-2}, 55% mortality).
- Use weighted average 64 clams m^{-2} at seeding (55% mortality).
- Use the 40ft contour estimate of potential acres as suitable area for expansion scenario (see Table 7, Table 8) for final oyster and clam cultivation areas to use in EcoWin simulations).

Great Bay/Piscataqua Region Estuaries

In New Hampshire a few small-scale oyster farmers are experimenting with intensive culture methods. Since wild sets of oyster larvae are sporadic at best, these growers are reliant on hatchery-reared seed which allows these growers to work with lines of oysters that have been selectively bred for fast growth and disease resistance. By utilizing the intensive culture methods described above for CT growers (i.e. upwellers, cages, plastic mesh bags), NH growers are able to get a small 3.5 inch thin-shelled 70 g oyster to market in 18-30 months. These growers ask $0.55-$0.65 wholesale for their oysters which are typically served at high-end restaurants and raw bars.

Intensive growers are able to plant at relatively high densities as long as algal concentrations and current speeds are both high enough to support good growth. Planting densities of 100-200 m^{-2} can be attained under optimal conditions.

The oyster aquaculture industry is just getting started in GBP, thus there are few growers who have brought a crop to market. However, the industry is growing and production is expected to continue to grow at double-digit rates as was observed in Massachusetts and Rhode Island once regulatory obstacles were cleared.

Modeling Tools

AquaShell Individual Model

An individual growth model based on the AquaShell™ generic framework (Silva et al., 2011) was calibrated and validated for Eastern oyster, *Crassostrea virginica* using environmental drivers from this study (Table 9) together with experimental (see section 'Supporting Experiments and Data') and literature growth data (i.e. shell length, total fresh weight and tissue dry weight; Figure 16 - Figure 18) from Long Island Sound (Loosanoff, 1947; Loosanoff and Nomejko, 1949; Bricelj et al.. 1992; Shaw and McCann, 1963; Zarnoch and Schreibman, 2012). This model uses a net energy balance approach (see in Silva et al., 2011) with functions published in the literature (Dame, 1972; Hoffman 1992; Kobayashi et al, 1997; Riisgård, 1988) and new morphometric formulations adjusted to the studied site. The individual model for *Crassostrea virginica* (Figure 19) was included in the WinShell software (www.longline.co.uk/winshell).

Table 9. Drivers and set up used to run the individual growth model in WinShell (Start day for growth = 50, seed size as total fresh weight = 1.75 g) Data from M. Lyman, CTDEEP for long term monitoring Station 09 for 2008. Note that there has been no significant change in water quality from 2008-2012, thus this is representative of current conditions (see section 'Supporting Experiments and Data' where POM is particulate organic matter and TPM is total particulate matter).

Day	Temperature	Salinity	Chlorophyll a	POM	TPM
	(ºC)	(psu)	(μg L^{-1})	(mg L^{-1})	(mg L^{-1})
10	4.61	28.1	5.5	0.87	4.6
36	3.15	27.9	18.1	2.03	8.0
63	2.61	27.0	9.6	1.91	6.0
98	5.67	24.9	7.9	1.77	7.0
127	10.68	24.7	4.0	1.46	8.0
153	11.67	26.2	5.5	1.69	14.0
183	15.19	26.6	4.6	1.90	31.0
215	21.04	26.9	4.8	1.37	6.0
243	22.77	27.3	5.8	1.39	6.0
277	17.63	27.1	3.7	0.95	6.0
307	13.89	26.7	9.0	1.02	6.0
347	6.57	25.6	0.9	0.83	8.0

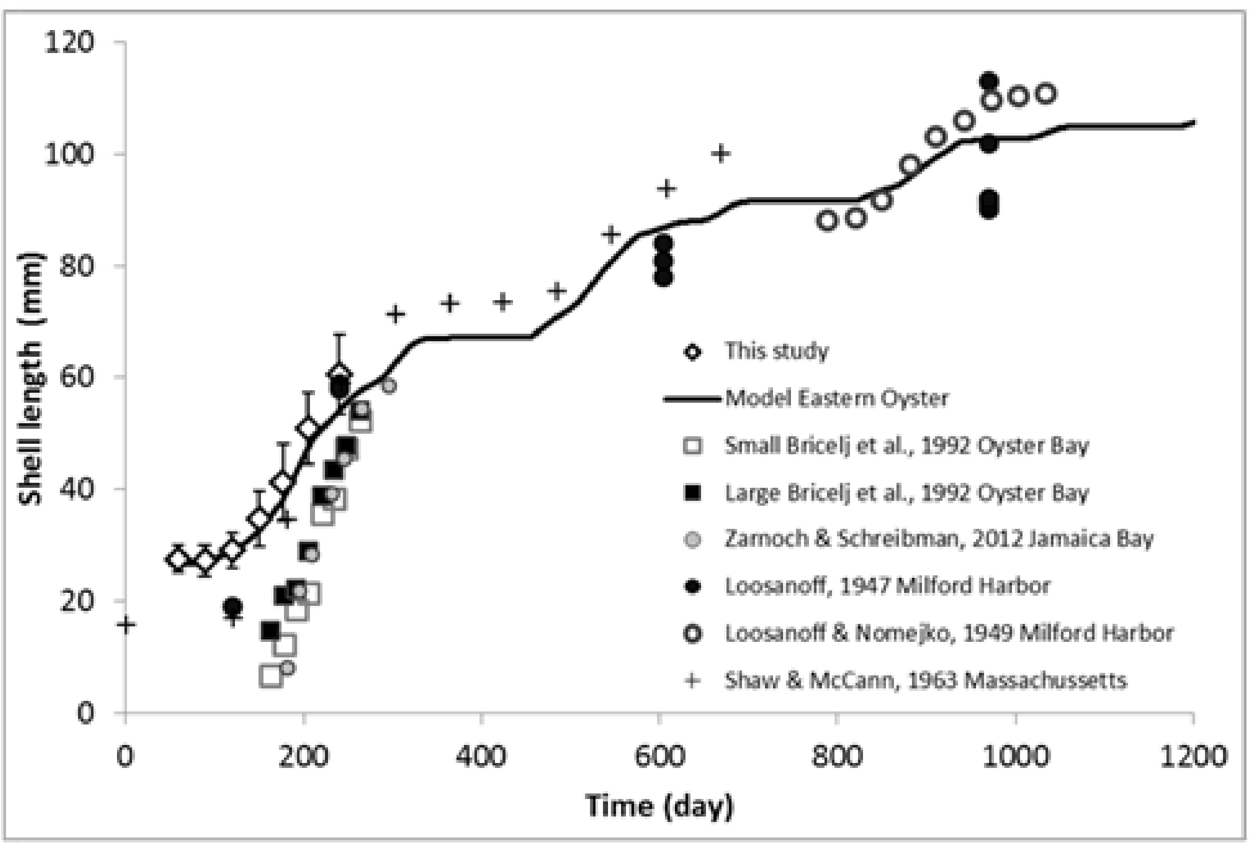


Figure 16: Simulation of Eastern oyster growth (shell length) in Long Island Sound.

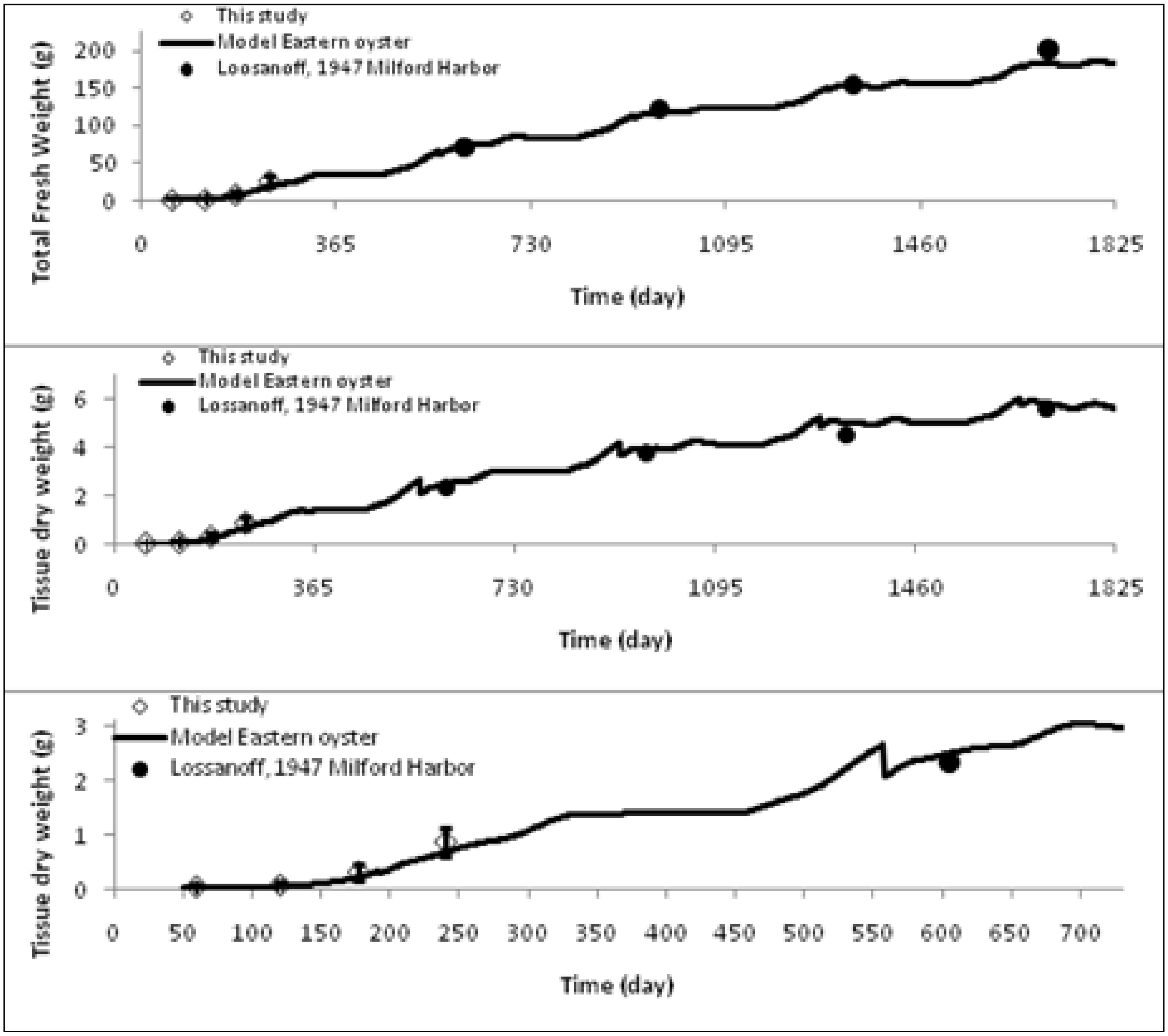


Figure 17: Simulation of Eastern oyster growth (total fresh weight and tissue dry weight) in Long Island Sound.

Figure 16 and Figure 17 show a good fit between the predicted shell length (mm), total fresh weight (g) and tissue dry weight (g) with the observed data from this study and the literature. In Figure 18A, the shell length observed in this study was used to calibrate the Eastern oyster model.

Note that the oyster size at the start day of the growth was much lower in Bricelj et al. (1992) and Zarnoch and Schreibman (2012) than in this study. Consequently, the data were not used in the validation plot (Figure 18B).

The calibration plot shows that although the predictions are overfitted in relation to the line of identity, the latter fits within the 95% confidence interval of the mean of the observed data in this study (Figure 18A). The oysters originating from LIS in Shaw and McCann (1963) were seeded in Massachusetts and seemed to grow faster than in this study.

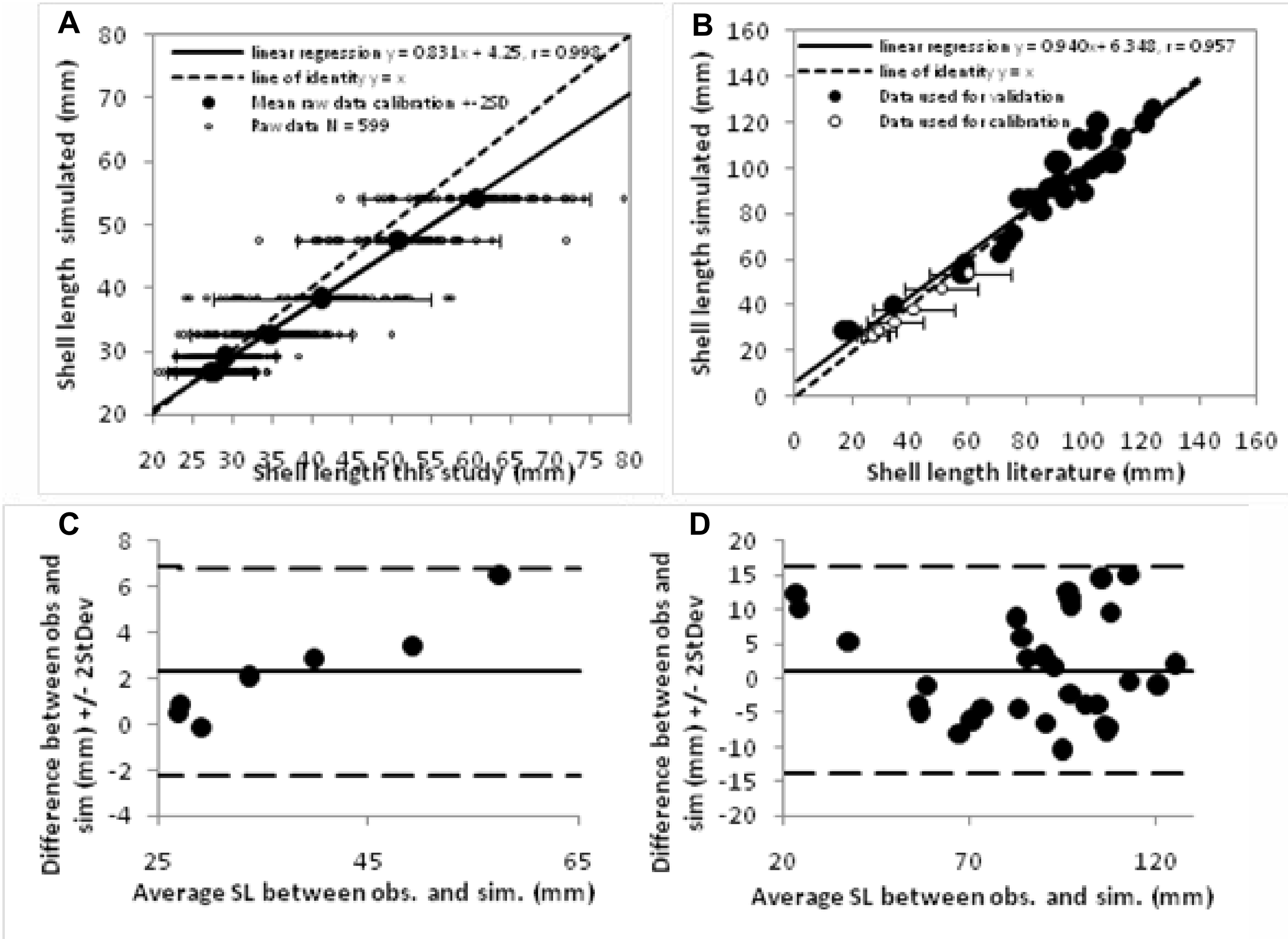


Figure 18: Shell length (height) (mm) observed in this study (A-Calibration) and from literature (B-Validation) vs Eastern model predicted values, together with the line of identity (dotted) and the fitted regression line (solid). (C) and (D) bias analysis, SL = shell length; sim = simulated; obs. = Observed. Error bar = 2xstandard deviations. Data from Loosanoff 1947, Loosanoff & Nomejko 1949, Shaw & McCann 1963.

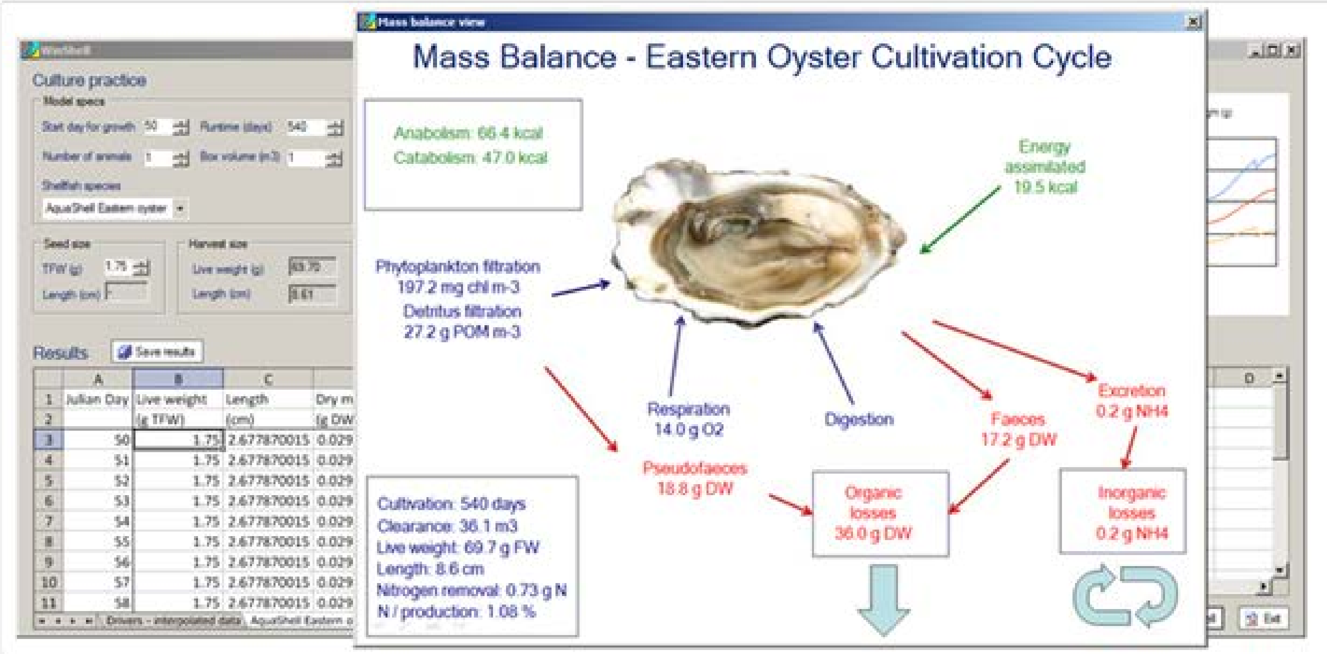


Figure 19: Simulation of mass balance for Eastern oyster, showing the interface of the WinShell program in the background.

The validation of the model on data from literature was conducted with and without Shaw and McCann data and both gave similar results of an almost perfect match between the line of identity and the fitted linear regression (Figure 18B).

System Wide Eutrophication Model (SWEM)

The System Wide Eutrophication Model (SWEM) is a previously calibrated and validated high resolution 3-D coupled hydrodynamic-eutrophication-sediment nutrient flux model that operates on the timeframe of one to two years (compared to the decadal timeframe of the EcoWin model). The computational grid for the study area included in this study (the Harlem/East River and LIS) is comprised of 230 longitudinal and lateral elements and ten vertical water column layers (2300 grid cells) while the complete SWEM grid includes 16,000 cells. The hydrodynamic transport model (Blumberg et al., 1999) applied in SWEM is based on the Estuarine, Coastal, and Ocean Model (ECOM; Blumberg and Mellor, 1987) source code. The model is driven by measured water level, meteorological forcing, spatially and temporally varying surface heat flux and freshwater fluxes from the numerous rivers, wastewater treatment plants, combined sewer overflows, and runoff from the land that enter the NY/NJ Harbor Estuary, LIS, and the New York Bight. The hydrodynamic model solves a coupled system of differential, prognostic equations describing conservation of mass, momentum, heat and salt. Detailed reporting on the hydrodynamic model calibration and validation is available in HydroQual (1999) and Blumberg et al. (1999). The model has been further validated for additional years by the Contamination Assessment and Reduction Project (CARP) and a report on hydrodynamic model skill assessment results (i.e. comparison of model outputs to measured data) for the additional conditions, years 1998-2002, is available at www.carpweb.org.

The water quality model source code underlying both of the SWEM Eutrophication and Sediment Nutrient Flux models is Row Column Aesop (RCA) which originates from the Water Analysis Simulation Program (WASP) developed by Hydroscience (HydroQual's predecessor firm) in the 1970's (DiToro et al., 1981). The RCA source code is a general purpose code used to evaluate and address a number of water quality issues that are specific to a given water body. It formulates mass balance equations for each model segment for each water quality constituent or state-variable of interest. These mass balance equations include all horizontal, lateral and vertical components of advection and diffusive/dispersive mixing between model segments; physical, chemical and biological transformations between the water quality variables within a model segment; and point, non-point, fall-line, and atmospheric inputs of the various water quality variables of interest. The partial differential equations, which together with boundary conditions make up the water quality model are solved using several mass conserving finite difference techniques.

The SWEM eutrophication and sediment nutrient flux modeling kinetics developed in RCA include several key features. These include kinetics representing: available light, algal growth, nutrient and organic carbon cycling, sediment dynamics, and the dissolved oxygen balance that are described in Landeck Miller and St. John (2006) and HydroQual (1999). Essentially the same framework as SWEM has been applied to other systems such as Chesapeake Bay and mesocosms of Narragansett Bay with consistent and similar parameters (DiToro, 2001; DiToro et al., 2001; Lowe and DiToro, 2001).

The calibration and validation of SWEM required a synoptic database including hydrodynamic and water quality measurements for both the water column and sediment bed. Model calibration involved the comparison of model calculations to measured values. Model validation involved the further comparison of model calculations to measured values, but for a set of conditions different from calibration conditions. SWEM was calibrated to data collected during 1994-95 and validated for data collected in 1988-89 as described in HydroQual (1999, 2002, 2005); Blumberg et al. (1999) and Landeck Miller and St. John (2006).

The SWEM model was used in this study to calculate LIS water volume fluxes and to provide external nutrient loadings to drive the EcoWin ecosystem model. This involved aligning the higher resolution computational grid of SWEM with a lower resolution computational grid developed for EcoWin. Water volume fluxes calculated by SWEM were passed to EcoWin at the boundaries of EcoWin; at the modeled locations of heads-of-tide, WWTPs, Combined Sewer Overflows (CSOs), and storm water discharges; and at each EcoWin computational grid interface. The SWEM calculated water volume fluxes were passed to EcoWin hourly in a spreadsheet for 24 months of hydrodynamic and hydrologic conditions. The external nutrient loadings developed for SWEM for four loading conditions were also passed to EcoWin in a spreadsheet on a time varying (i.e., hourly to monthly) basis for heads-of-tide, WWTPs, CSOs, storm water, and atmospheric deposition. While the EcoWin model itself calculates concentrations of nutrients, algal biomass, and dissolved oxygen throughout LIS, values for temperature, salinity, algal biomass, and nutrient and DO concentrations calculated by SWEM were provided in a spreadsheet as time-weighted averages to further inform the EcoWin work and for use in the application of the FARM and ASSETS models.

SWEM Flow and Nutrient Loading Conditions Provided to EcoWin (SWEM)

The hydrologic and hydrodynamic conditions data passed from SWEM to EcoWin are for 24 months of hydrologic and hydrodynamic conditions that occurred in LIS during calendar years 1988 and 1989. Conditions in LIS vary from year to year and it is not possible to know what the conditions will be in upcoming years thus, for modeling evaluation purposes, it is a common practice to select previous hydrologic and hydrodynamic conditions to use for future planning purposes. The 1988 and 1989 calendar years were selected for a variety of reasons:

- Dedicated sampling programs in LIS were conducted in 1988 and 1989 for initial Total Maximum Daily Load (TMDL) development purposes. The summer of 1989 was known to have more rainfall than average years and also had pronounced hypoxia occurring in the Sound. Hypoxia in 1988 was more critical than in 1989, and rainfall patterns in 1988 were more typical than those in 1989.
- The DO TMDL for nitrogen that was prepared in 2000 and is currently being implemented was based on eighteen months of hydrologic and hydrodynamic conditions that occurred between April 1988 and September 1989.
- State and federal agencies continuing to work on the phased TMDL for LIS extended the planning hydrologic and hydrodynamic conditions to be inclusive of the 24 months of conditions that occurred in LIS during calendar years 1988 and 1989 to overlap with New York – New Jersey Harbor TMDL planning activities.
- Through the leadership of several state and federal agencies continuing to work on nutrient management in LIS, several different nutrient loading scenarios beyond the TMDL reductions were developed and evaluated based on the 1988 and 1989 conditions.
- This study took advantage of the considerable effort already expended on developing and evaluating likely future nutrient loading scenarios within the context of 1988 and 1989 hydrologic and hydrodynamic conditions.

Future nutrient loading scenario conditions were passed from SWEM to EcoWin. The data were based on 1988 and 1989 hydrologic and hydrodynamic conditions but with post-TMDL loading conditions that include:

- The 2000 TMDL nitrogen and associated carbon load reductions (shown as TMDL + C on Figure 20 and Figure 21).
- Low level reductions to urban and agricultural runoff and STP nitrogen loadings in three states (VT, NH, MA) imposed upon the 2000 TMDL nitrogen and carbon load reductions (shown as Run 5 on Figure 20 and Figure 21).
- High level reductions to urban and agricultural runoff and STP nitrogen loadings in five states (NY, CT, VT, NH, MA) imposed upon the 2000 TMDL nitrogen and carbon load reductions (shown as Run 7 on Figure 20 and Figure 21)

- High level reductions to atmospheric nitrogen deposition imposed upon the 2000 TMDL nitrogen and carbon load reductions (shown as Run 2 on Figure 20 and Figure 21)

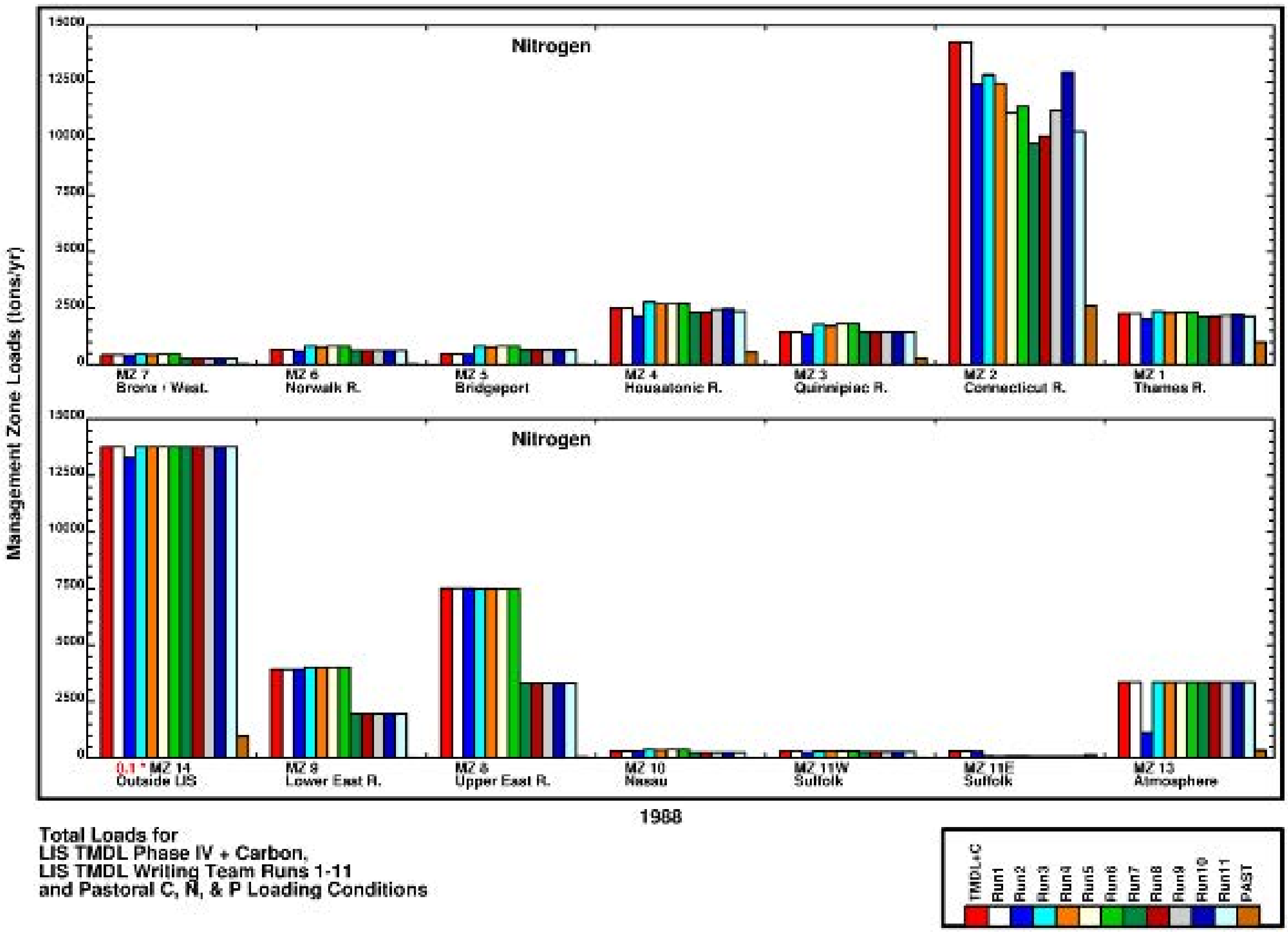


Figure 20: Annual total nitrogen loadings modeled with SWEM for Long Island Sound presented by TMDL geopolitical loading management zones (MZs) for 1988 hydrodynamic conditions. Loadings for TMDL+C and Runs 5, 7, and 2 were shared with EcoWin.

The EcoWin work was focused on evaluating aquaculture and bioextractive technologies within the context of the expected nutrient loading conditions in LIS that are reflective of the fully implemented 2000 TMDL nitrogen and carbon load reductions. It is expected that these mandated TMDL load reductions will be fully implemented by the end of 2014. Detailed descriptions of the loading conditions are found in HydroQual (2007 and 2009).

The key nutrient loading conditions passed from SWEM to EcoWin are for fully implemented 2000 TMDL nitrogen and carbon load reductions. The other loading conditions shared between SWEM and EcoWin represent possible, but undecided upon, options for later TMDL phases. The development of the 2000 TMDL loading conditions for modeling purposes is described in HydroQual (2007). Descriptions of the development of the other SWEM loading conditions shared with EcoWin are found in HydroQual (2009).

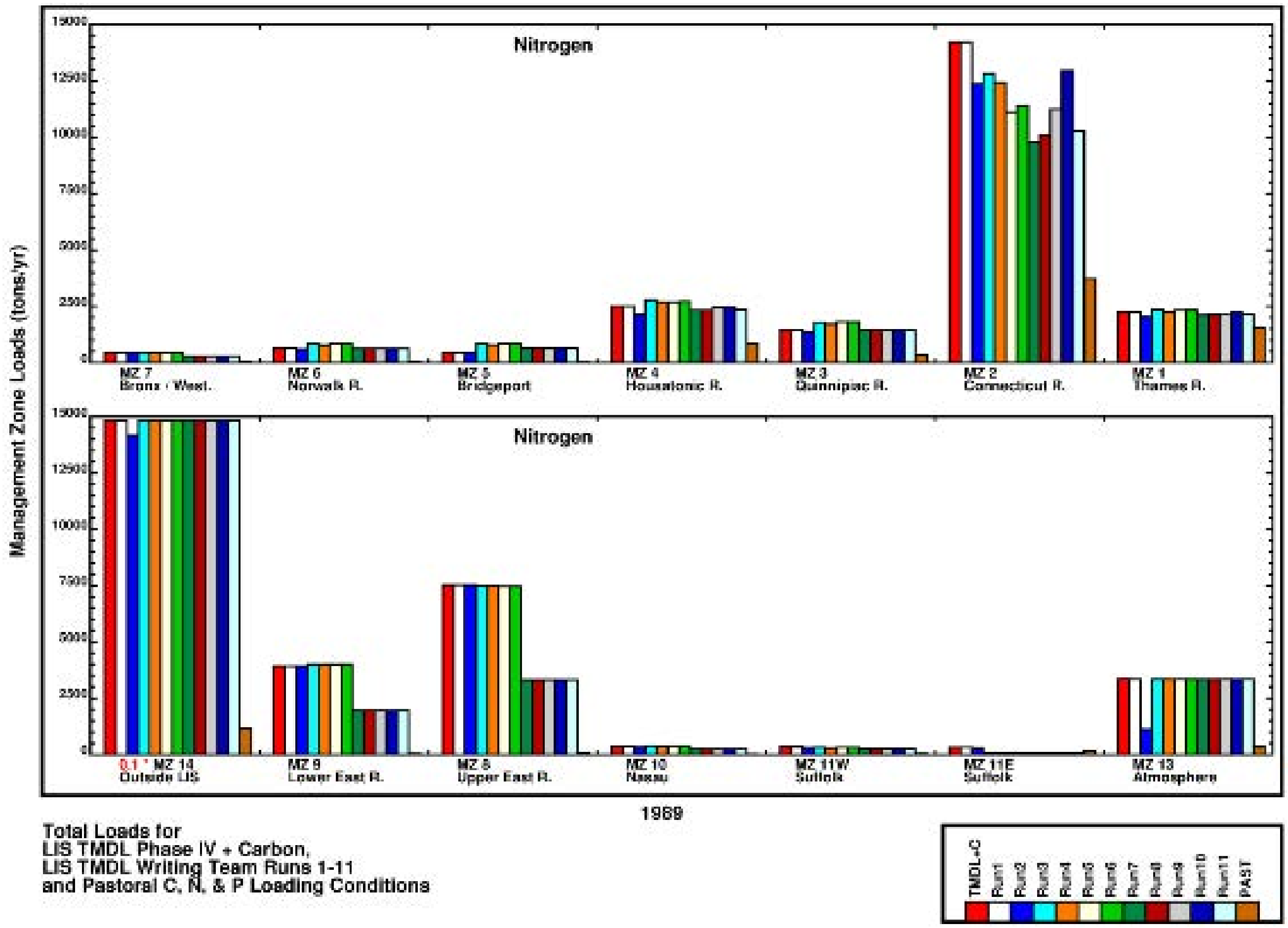


Figure 21: Annual total nitrogen loadings modeled with SWEM for Long Island Sound presented by TMDL geopolitical loading management zones (MZs) for 1989 hydrodynamic conditions. Loadings for TMDL+C and Runs 5, 7, and 2 were shared with EcoWin.

Figure 20 and Figure 21 adopted from HydroQual (2009) depict all four shared nitrogen loading conditions on an annualized summary basis for several TMDL geopolitical loading management zones designated across the LIS study area.

Model Calibration Method Using SWEM Outputs

This study relied upon SWEM specifically for the specification of hydrologic and hydrodynamic conditions in Long Island Sound. Selected demonstrations of the existing SWEM hydrodynamic calibration are provided here (see section below ‘Model Calibration Method Justification‘). These demonstrations include model and measurement comparisons for water elevation, current velocity, temperature, and salinity specifically within LIS, rather than other portions of the SWEM domain, for conditions that actually occurred.

The combined hydrologic/hydrodynamic and nutrient loading conditions provided to EcoWin by SWEM are projected representations of conditions in Long Island Sound with the 2000 TMDL fully implemented and without the implementation of bio-extractive technologies. The likely near-future conditions, including the implementation of the TMDL and pre-selected hydrodynamic/hydrologic conditions, although realistic, have not yet fully occurred or been measured. The calibration strategy for EcoWin therefore necessarily involved achieving favorable comparisons between EcoWin outputs and the provided SWEM projection outputs for both hydrodynamics and water quality. The inference was made that EcoWin results are valid to a similar degree as the SWEM projections in the absence of measurements reflecting the full implementation of the TMDL paired with the 1988 and 1989 hydrodynamic conditions that could be directly compared to EcoWin results. SWEM calibration demonstrations for water quality conditions that actually occurred are also provided here as further evidence of SWEM calibration and reliability of SWEM projections (see section below 'Model Calibration Method Justification'). These include model and measurement comparisons for DO, total nitrogen, and particulate organic carbon, inclusive of algal biomass.

It would not be reasonable to expect EcoWin to match these historical SWEM measurements and model comparisons as they represent different conditions than those modeled by EcoWin. The conditions modeled commonly by both EcoWin and SWEM should generally represent post-2014, allowing for variations that may occur in specific future years related to hydrodynamics rather than nutrient loadings. The various SWEM results shared with EcoWin and the intended purposes are identified in Table 10.

Table 10: SWEM Information Sharing Strategy

SWEM INFORMATION SHARED	STUDY PURPOSE
Calendar years 1988 and 1989 water volume fluxes provided in spreadsheets	Physical transport for EcoWin calculations
Various post-TMDL/2014 external nutrient loading forcing conditions developed using calendar years 1988 and 1989 hydrology and hydrodynamics provided in spreadsheets	Nutrient loadings for EcoWin calculations
Calculated nutrient, algal biomass, and oxygen concentrations for the post-TMDL/2014 external nutrient loading forcing conditions provided in spreadsheets	Calibration targets for EcoWin calculations, use with FARM
Various water years of model and measurement comparisons for water elevations, velocity, temperature, and salinity	Demonstrate SWEM hydrodynamic calibration and justify use of SWEM physical transport in EcoWin
Various water years of model and measurement comparisons for dissolved oxygen, nutrients, and algal biomass	Demonstrate SWEM water quality calibration, provide justification for relying on SWEM projection results and, by inference, EcoWin results

Model Calibration Method Justification

Demonstrations of SWEM calibration specifically within Long Island Sound for water elevation are summarized in Blumberg et al. (1999) and include model and measurement statistical comparisons for Acoustic Doppler Current Profilers (ADCPs) to measure current speeds, deployed during 1994-95 at Willet's Point, NY (WPNY); Bridgeport, CT (BCT); Montauk Point, NY (MPNY); and New London, CT

(NLCT) along with fifteen locations outside of LIS. It is noted that the Montauk deployment was in an embayment area slightly outside of the area covered by the model computational grid and there was a one hour phase shift between the model-calculated water elevations in the nearest grid cell and the Montauk measurements. Up to 8,640 instantaneous measurements of water elevation were made at each of these locations and model results were compared to the measurements instantaneously and sub-tidally (34-hr and 5-day low-passed filtered). These results as presented in Blumberg et al. (1999) are included here as Table 11.

Table 11: SWEM Calculated and Measured Water Elevation Comparisons

Station	Number of Measurements	Water Elevation Measurement Range (cm)	Water Elevation Measurement and Model Comparisons					
			Instantaneous		Sub-tidal 34-hr Low-Passed		Sub-tidal Five-Day Low-Passed	
			RMS Error %	Correlation Coefficient	RMS Error Cm	Correlation Coefficient	RMS Error Cm	Correlation Coefficient
WPNY	8640	254.5	10.5	0.95	12.9	0.79	8.7	0.84
MPNY	5682	80.5	29.8	0.60	7.0	0.81	5.0	0.85
BCT	8640	236.5	8.2	0.97	9.2	0.86	7.1	0.86
NLCT	8640	96.3	11.1	0.94	7.7	0.83	6.1	0.83

Demonstrations of SWEM calibrations specifically within LIS for current velocity are also summarized in Blumberg et al. (1999) and include two moorings along with thirteen other measurement locations. At the eastern LIS mooring in particular, the measured U and V velocity ranges were 108 and 57 cm/s at a depth of 3 m (near surface) and 85 and 55 cm/s at a depth of 25 m (near bottom). The RMS errors of the model and measurements comparisons were from 8%-13% with correlation coefficients between 0.53 and 0.92.

Regarding model and measurement comparisons for temperature and salinity in LIS, Blumberg et al. (1999) present correlations for numerous stations near surface and near bottom. For salinity model and measurement comparisons in LIS, the correlation coefficients for near surface and near bottom are 0.74 and 0.83, respectively. For temperature model and measurement comparisons in LIS, the correlation coefficients for near surface and near bottom are 0.99 and 1.00, respectively.

Demonstrations of SWEM calibrations for DO, total nitrogen, and particulate organic carbon presented in HydroQual (1999, 2002) include comparisons between measurements and model calculations along a spatial transect running from the Battery through the thalweg of the East River and LIS to the ocean and as time series for individual locations for nine synoptic sampling events conducted between October 1994 and September 1995. These comparisons have been simplified for summary purposes to simple x-y plots of calculated vs. observed as presented in Figures 22, 23, 24, and 25.

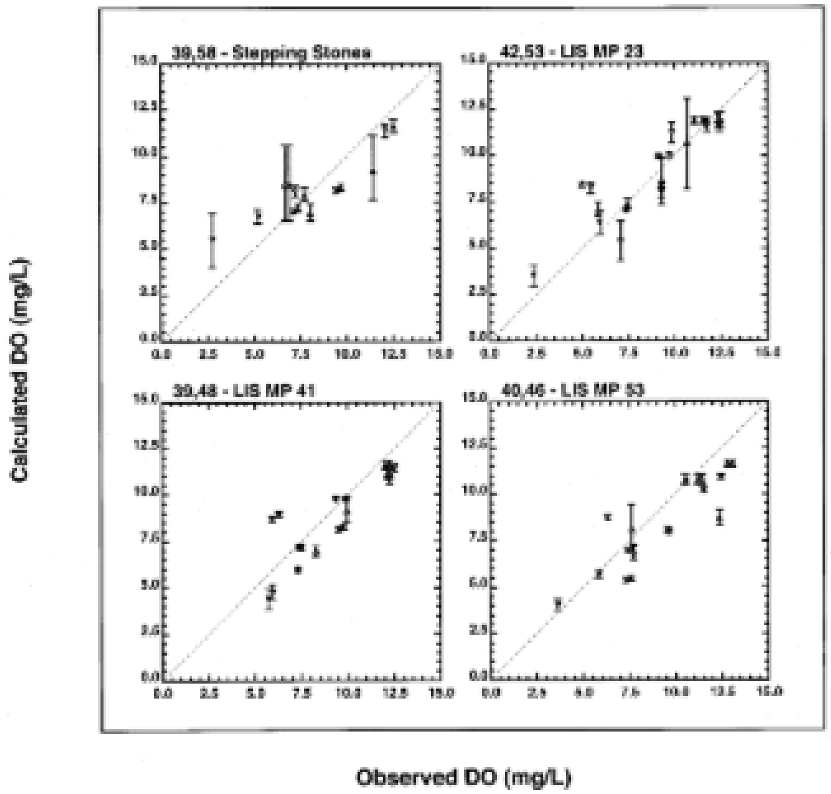


Figure 22: Summary demonstration of SWEM calibration for dissolved oxygen at selected locations (MP = mile point from East River) in Long Island Sound using 1994-95 model results and measurements.

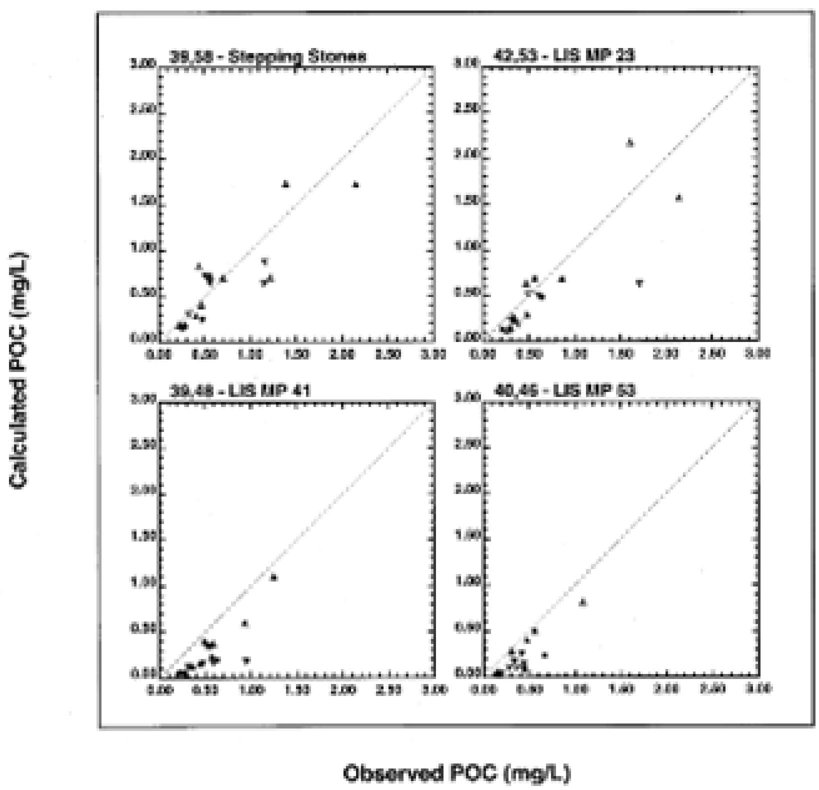


Figure 23: Summary demonstration of SWEM calibration for particulate organic carbon at selected locations (MP = mile point from East River) in Long Island Sound using 1994-95 model results and measurements.

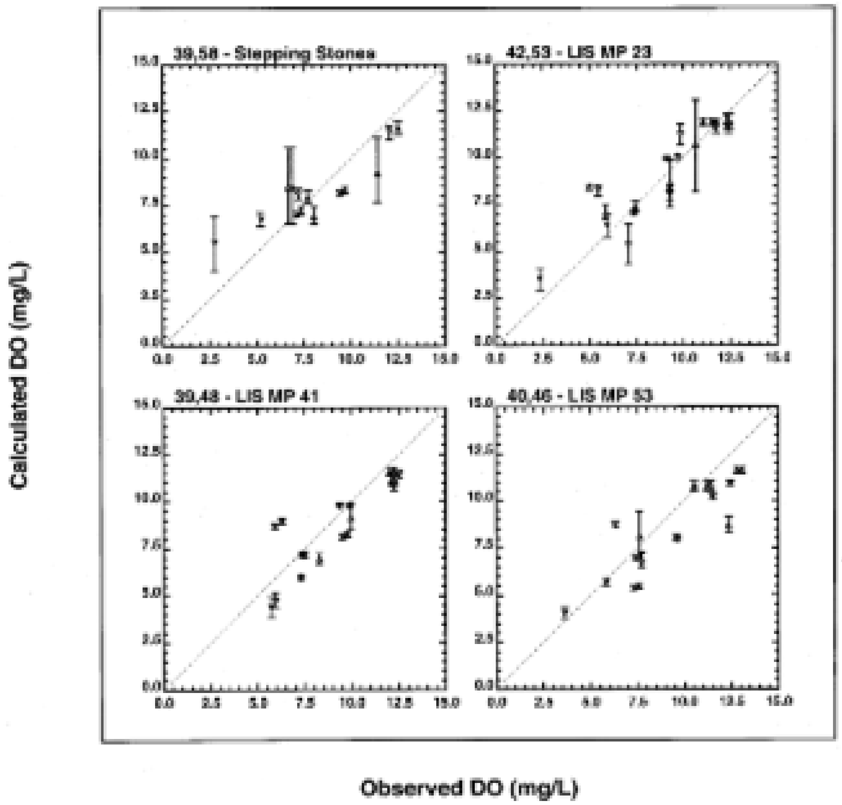


Figure 24: Summary demonstration of SWEM calibration for total nitrogen at selected locations (MP = mile point from East River) in Long Island Sound using 1994-95 model results and measurements.

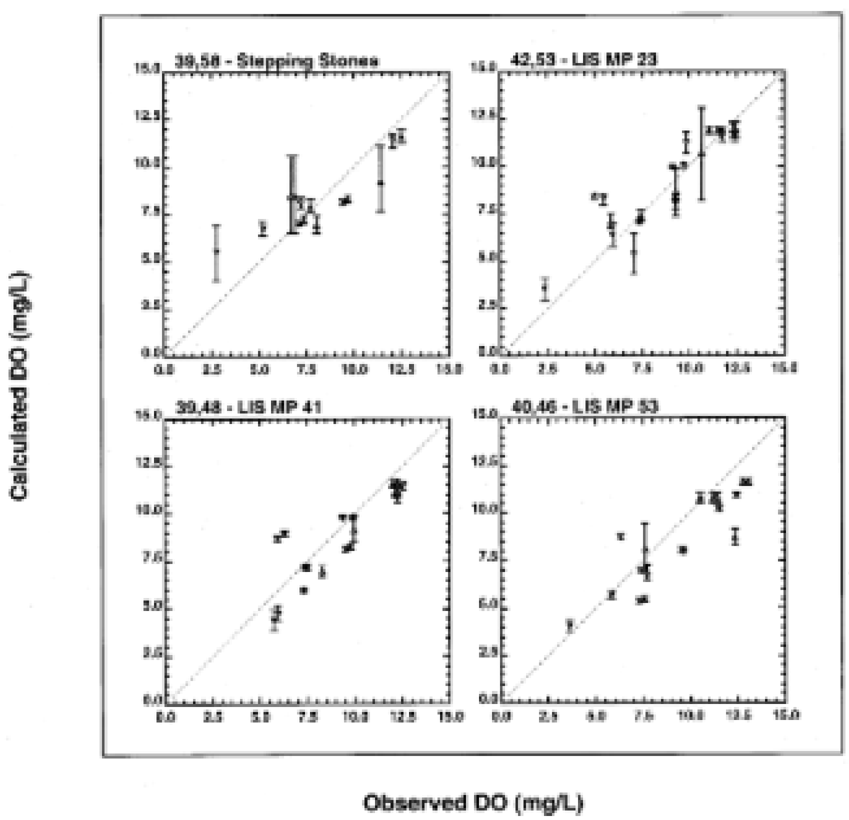


Figure 25: Summary demonstration of SWEM calibration for dissolved oxygen, total nitrogen, and particulate organic carbon throughout Long Island Sound during summer conditions using 1994-95 model results and measurements.

Figures 22 to 25 collectively demonstrate that while SWEM does not have a perfect representation of the measurements, it does not have a constant bias and it is balanced in somewhat equally and consistently representing DO, nitrogen, and particulate organic carbon simultaneously. Dissolved oxygen is the Sound's integrated response to external nitrogen and carbon loadings and algal dynamics. Nitrogen is the nutrient known to be a limiting factor in algal growth in LIS. Particulate organic carbon represents both allochthonous organic carbon and algal biomass. As demonstrated by frequent near zero measurements of dissolved inorganic nitrogen in the Sound, the phytoplankton populations consume all of the available dissolved inorganic nitrogen and create an oxygen demand at die-off. While Figures 22-25 each show comparisons between calculations and measurements at a few locations for nine surveys, Figure 25 shows the comparisons for all locations during a summer period when nitrogen depletion and hypoxia are likely to occur.

EcoWin.NET Model

Overview

The EcoWin.NET ecological modeling package is an object-oriented framework designed to simulate key components of biogeochemical cycles, including processes in the water column and sediments. It targets specific issues of relevance to integrated coastal zone management, including eutrophication and aquaculture. The EcoWin framework has been completely redesigned in 2012-2013 and runs on 64 and 32 bit Windows systems (Figure 26). A typical run for a ten year period to simulate shellfish aquaculture in LIS takes approximately 3 minutes.

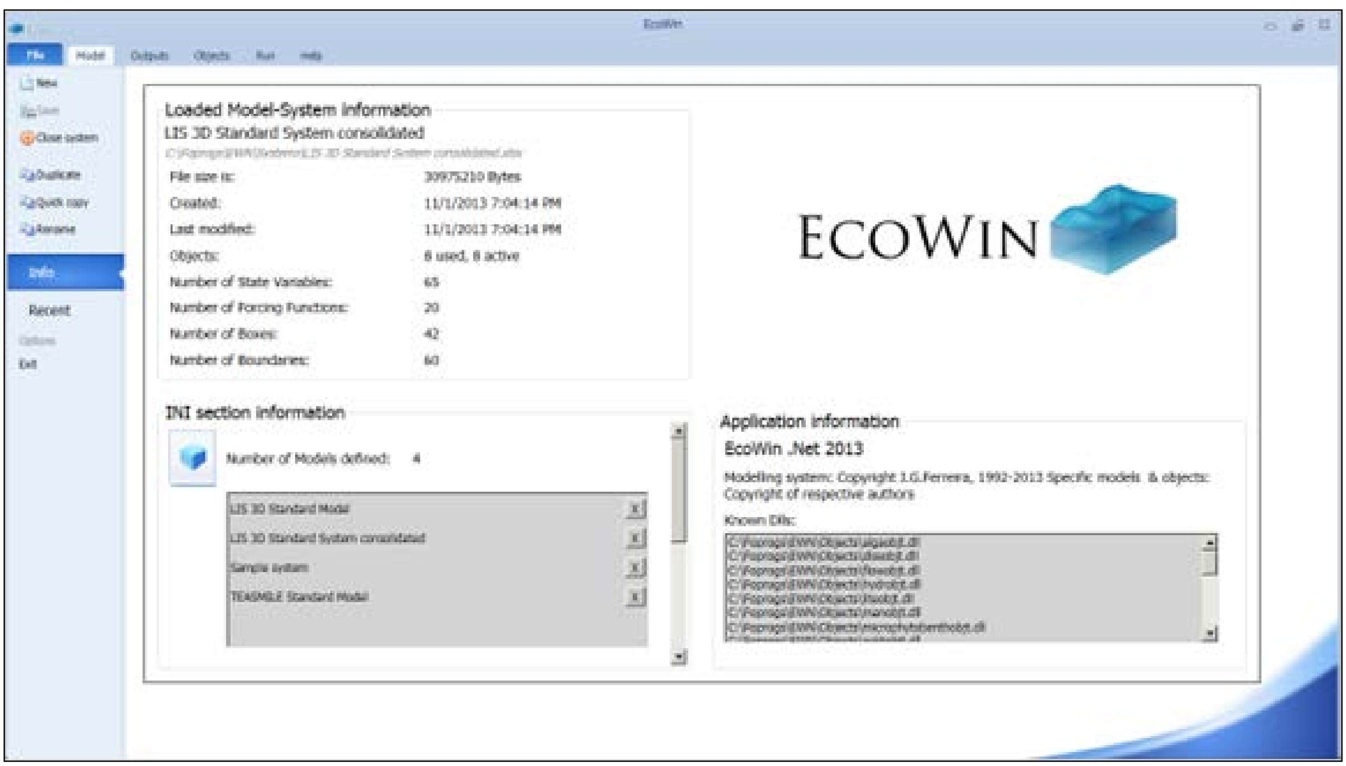


Figure 26: EcoWin.NET modeling software – file menu and general definitions.

Coupling with SWEM

In order to simulate shellfish culture over multiple culture cycles, to gain insights into the consequences of aquaculture activities in LIS, and the interaction of these with the other components of the ecosystem, the detailed grid used in SWEM was simplified into a coarser grid, and the vertical resolution limited to an upper and lower layer, maintaining the s-coordinate approach used in SWEM. Box definition or division is an important step in the development of an ecosystem model using the EcoWin.NET (EcoWin) platform. For the EcoWin box division of LIS, the criteria and methods are described below.

General criteria and methods

Administrative region: LIS is an area shared by CT and NY, therefore the division of boxes should consider the boundaries of different states.

Physical data: Boxes should be defined according to homogenous physical conditions. For this purpose, morphology, currents and vertical stratification are evaluated. Morphology is analyzed through bathymetry data. Currents are considered based on the SWEM hydrodynamic model, on which the EcoWin model boxes will be developed, and each box outline coincides with SWEM grid limits. Vertical stratification assessment is carried out by comparing surface and bottom salinity.

Aquaculture sites: The definition of the model boxes should aggregate the aquaculture areas in the system. These areas were included in the GIS and information on the status and type of shellfish beds should be analyzed for the purpose of potential spatial aggregation.

Coupling of SWEM and EcoWin

Administrative region: Long Island Sound is shared by the states of CT and NY; the research area defined in this study is shown in Figure 27. The state border crosses LIS from the west to its northeast corner, and the whole southwest corner is in NY. Since CT and NY have different coastal planning, and thus different aquaculture management approaches, the present proposal has taken this into consideration. The shellfish farm distribution shown in Figure 27 indicates that most of the farms in CT and NY are located within the 50 feet depth contour, and since it is known that the shellfish farms in NY are largely closed or restricted, this suggests LIS shellfish aquaculture management policies differ between CT and NY.

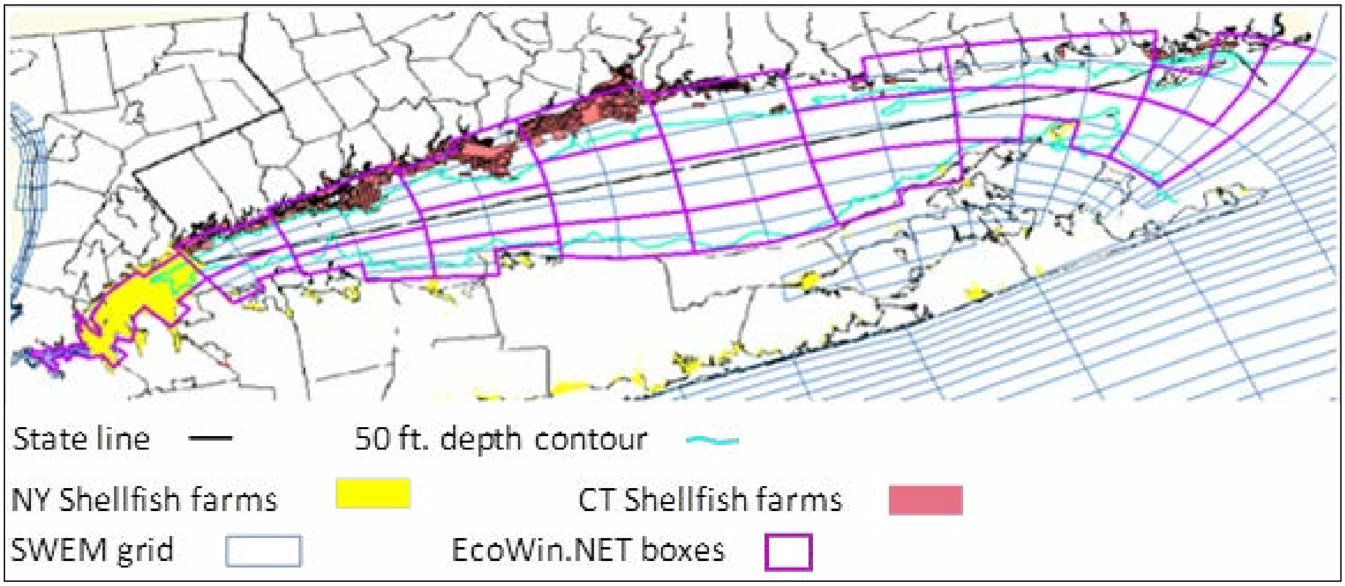


Figure 27: Long Island Sound region and state border between CT and NY.

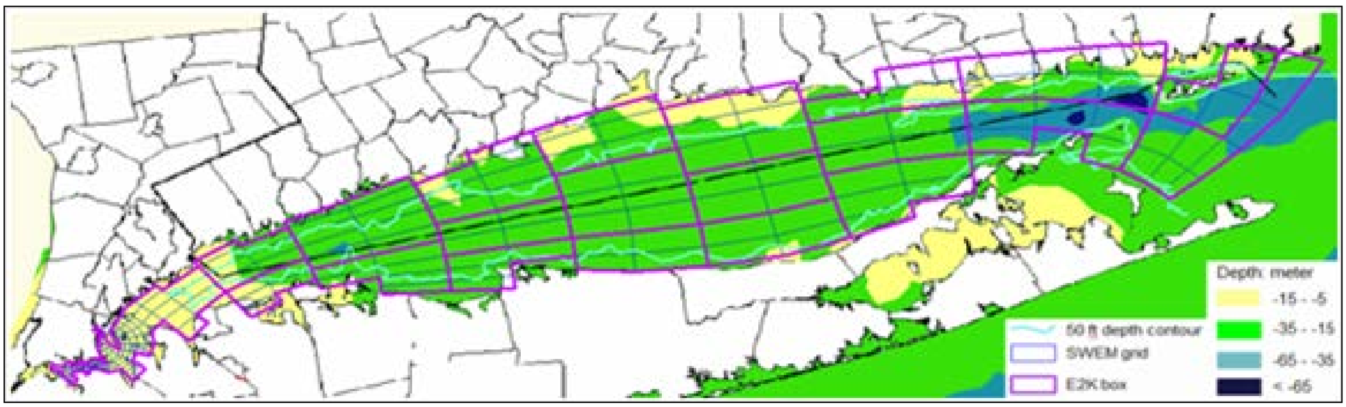


Figure 28: Bathymetry and EcoWin boxes

Physical data: In order to seamlessly couple EcoWin boxes with the SWEM grid, all EcoWin box limits must coincide with SWEM grid limits.

Long Island Sound morphology was analyzed through a bathymetry reclassification (Figure 28). Based on bathymetry data and on the circulation patterns, EcoWin model boxes aggregated with the SWEM grid are also shown in Figure 28.

Summertime vertical stratification was analyzed through the difference between bottom and surface salinity. Results are shown in Figure 29a. The salinity difference is fully considered during the EcoWin box division and in each box area results show that the water column was almost homogenous (Figure 29b).

Aquaculture sites: There are three datasets (ArcGIS shape file) on the project website (http://www.re-serv.org) for the shellfish aquaculture sites. The mapping of shellfish beds layout in CT is shown in Figure 30 upper (source file: Shellfish_Bed_Managed_DEEP.zip), and the layout of EcoWin boxes is shown in Figure 30 lower.

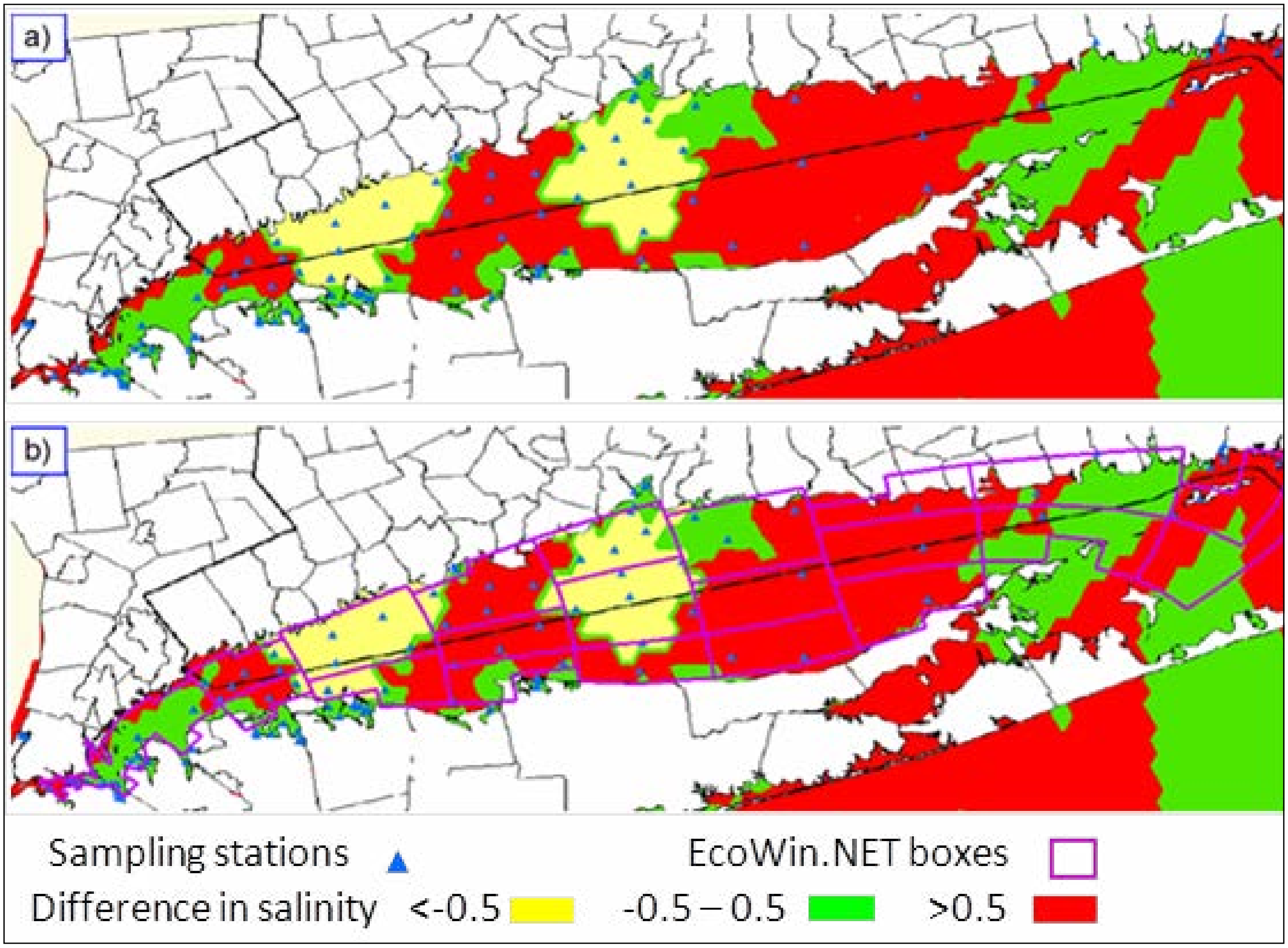


Figure 29: Vertical stratification and EcoWin boxes. a) salinity difference between bottom and surface water layer (bottom – surface); b) EcoWin box division considered with vertical stratification (ocean box only partially shown).

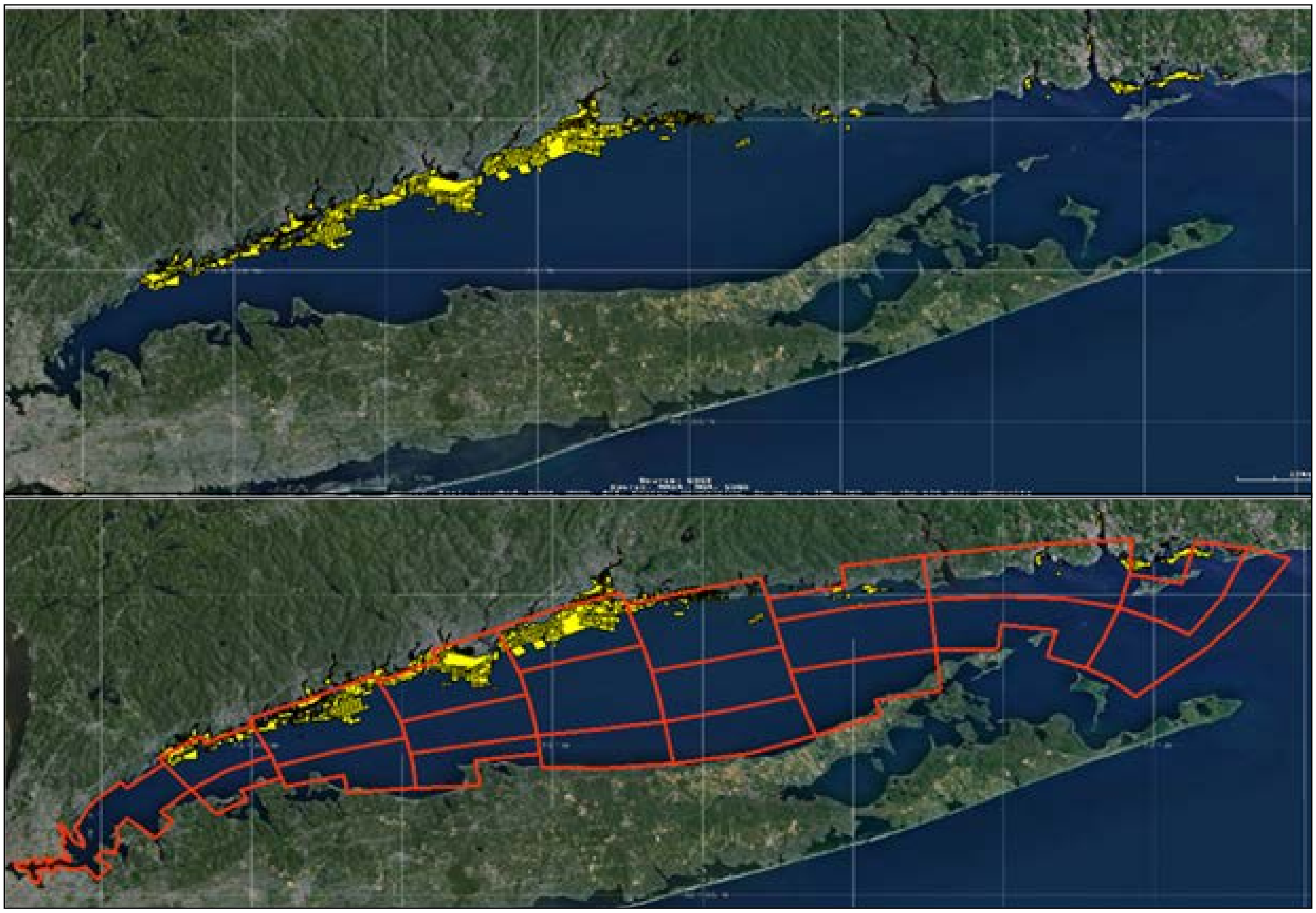
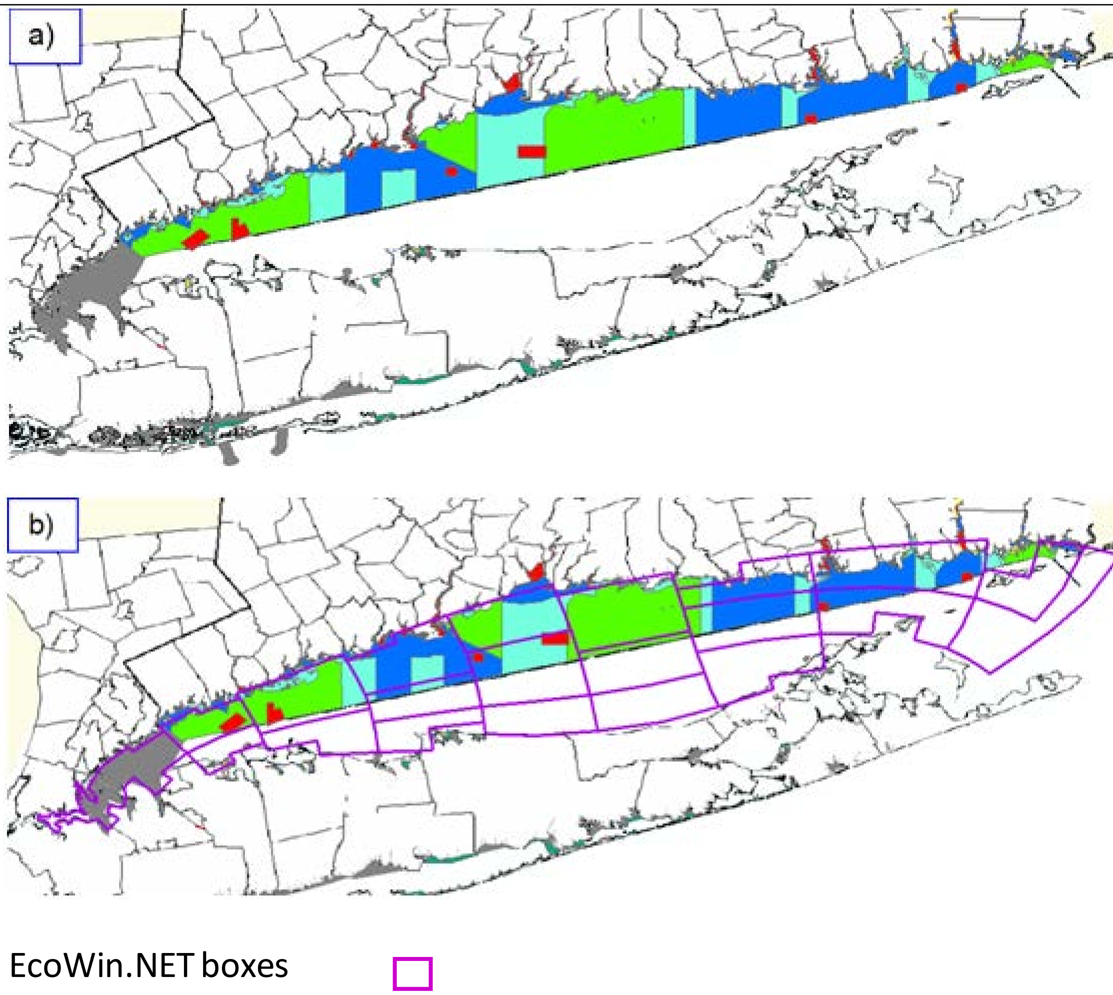


Figure 30: Mapping of CT shellfish beds along Long Island Sound and EcoWin grid layout; (upper panel). CT Shellfish beds (in yellow) along Long Island Sound coast, (lower panel) and EcoWin box layout (in red).

The shellfish beds status layout is shown in Figure 31a. The relationship between EcoWin boxes and the shellfish beds with different status is shown in Figure 31b. It can be clearly seen that shellfish bed status is a key factor associated with EcoWin box division, and the current box division proposal has mostly avoided aquaculture management confusion regarding farm status/types. Note that both CT and NY shellfish beds are shown for completeness, though the modeling only includes the CT shellfish areas.



Class	Acres	Class	Acres
CT_Restricted-Relay/Depuration	6.6	NY_Seasonal	11349.7
CT_Restricted-Relay	136219.1	NY_Holiday	833.8
CT_Prohibited	23336.9	NY_Closed	83256.1
CT_Conditionally Restricted-Relay	266.6		
CT_Conditionally Restricted-Rel/Dep	2292.2		
CT_Conditionally Approved	93763.5		
CT_Approved	134038.5		
Total	389923.5		95439.6

Figure 31: EcoWin box division based on shellfish beds status: a) Layout of shellfish beds status in CT and NY; b) relationship between EcoWin boxes and shellfish beds status. (source: HYDROQUAL).

Summary of EcoWin box division for Long Island Sound

In summary, considering administrative region, physical data and aquaculture sites, the whole LIS area has been horizontally divided into 21 sub-areas/boxes, and they all match the SWEM grid boundaries. Since most of the shellfish aquaculture activities along the LIS shore occur within the 50 feet depth, two vertical layers would be appropriate for the EcoWin.NET model. As a result, the EcoWin box division proposal for LIS includes 21 horizontal boxes, two vertical layers, 42 boxes in total.

Feasibility of the current box division scheme

In order to seamlessly couple with the SWEM hydrodynamic model, each EcoWin box outline coincides with SWEM grid edges, therefore the quality of SWEM coverage inevitably affects the EcoWin box division.

As shown in Figure 32, the northeast region of LIS has quite significant shellfish bed coverage, together with intensive sampling stations, which is not covered by SWEM. Figure 32b presents the EcoWin boxes and the layout of different status of shellfish beds in CT, which shows that the northeast area (marked with red circle) not included in SWEM inside Long Island Sound is also an approved shellfish farming area, i.e. it should be covered by the model. As a compromise, the EcoWin model will consider that missing grid as coastline, and the shellfish beds there will be put into adjacent EcoWin box for subsequent simulation.

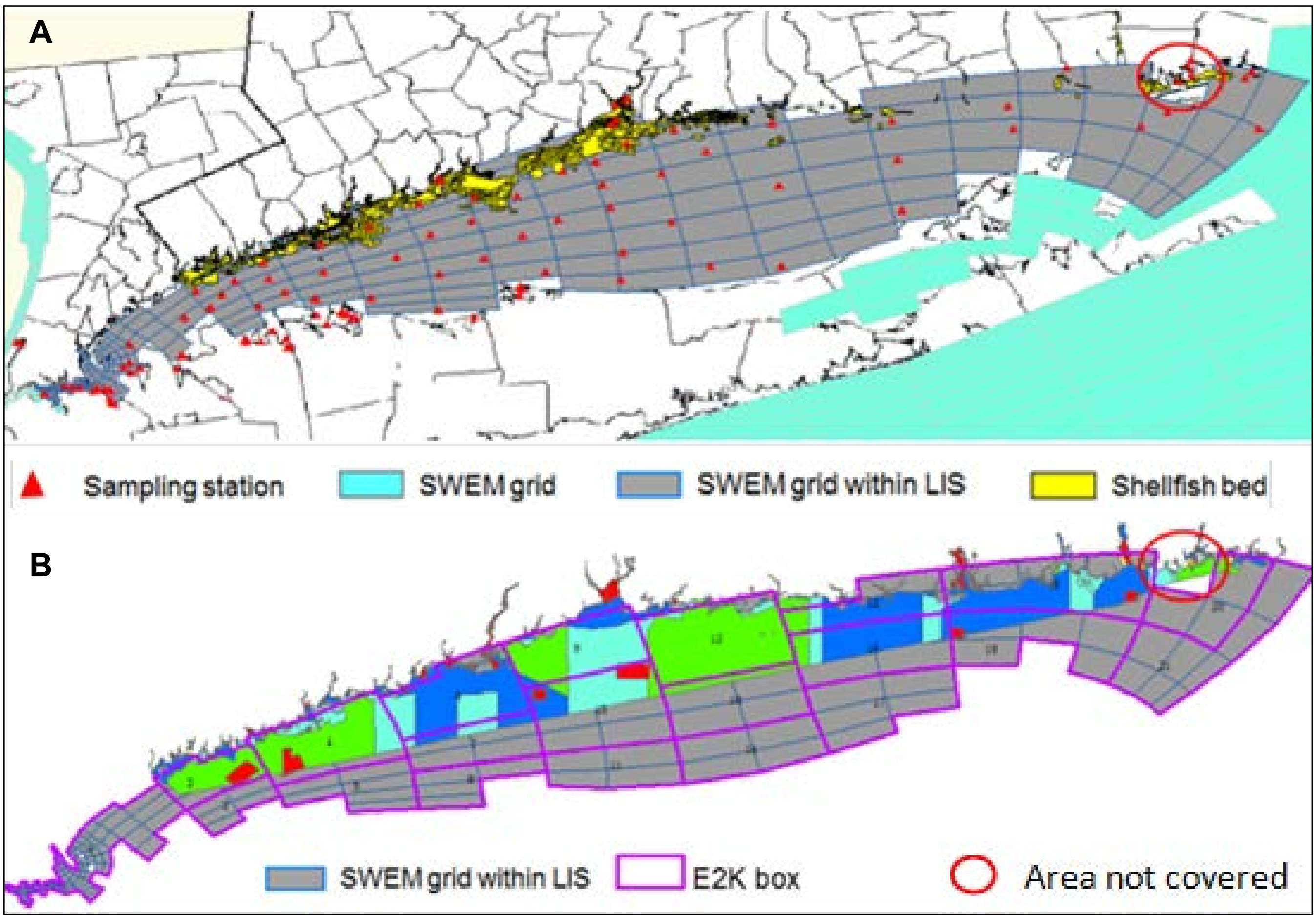


Figure 32: Limitation of SWEM grid coverage in NE Long Island Sound; a) shellfish beds and sampling stations shown with SWEM grid and b) SWEM grid boxes and EcoWin boxes are shown with missing grid area marked in red circle.

Calibration and Validation

Hydrodynamics: The hydrodynamic component of EcoWin uses aggregated water fluxes for each box generated by the SWEM model. For the EcoWin domain of LIS, the two models share the same geographic coverage, loading information of river input, treatment plant, combined sewer overflow (CSO), storm water overflow (SWO), and atmospheric loading. The only difference is the EcoWin model has 42 boxes distributed in two vertical σ-layers, whereas SWEM uses a grid with a horizontal resolution of 230 cells and 10 vertical σ-layers. For example at the boundaries, there are 50 treatment plants and 66 CSO and/or SWO diffusers in SWEM that discharge to the EcoWin domain, which means at least 111 SWEM grid cells receive loads from them, but on an EcoWin box basis there are only 12 boxes that receive treatment plant loads and 15 that receive CSO/SWO loads.

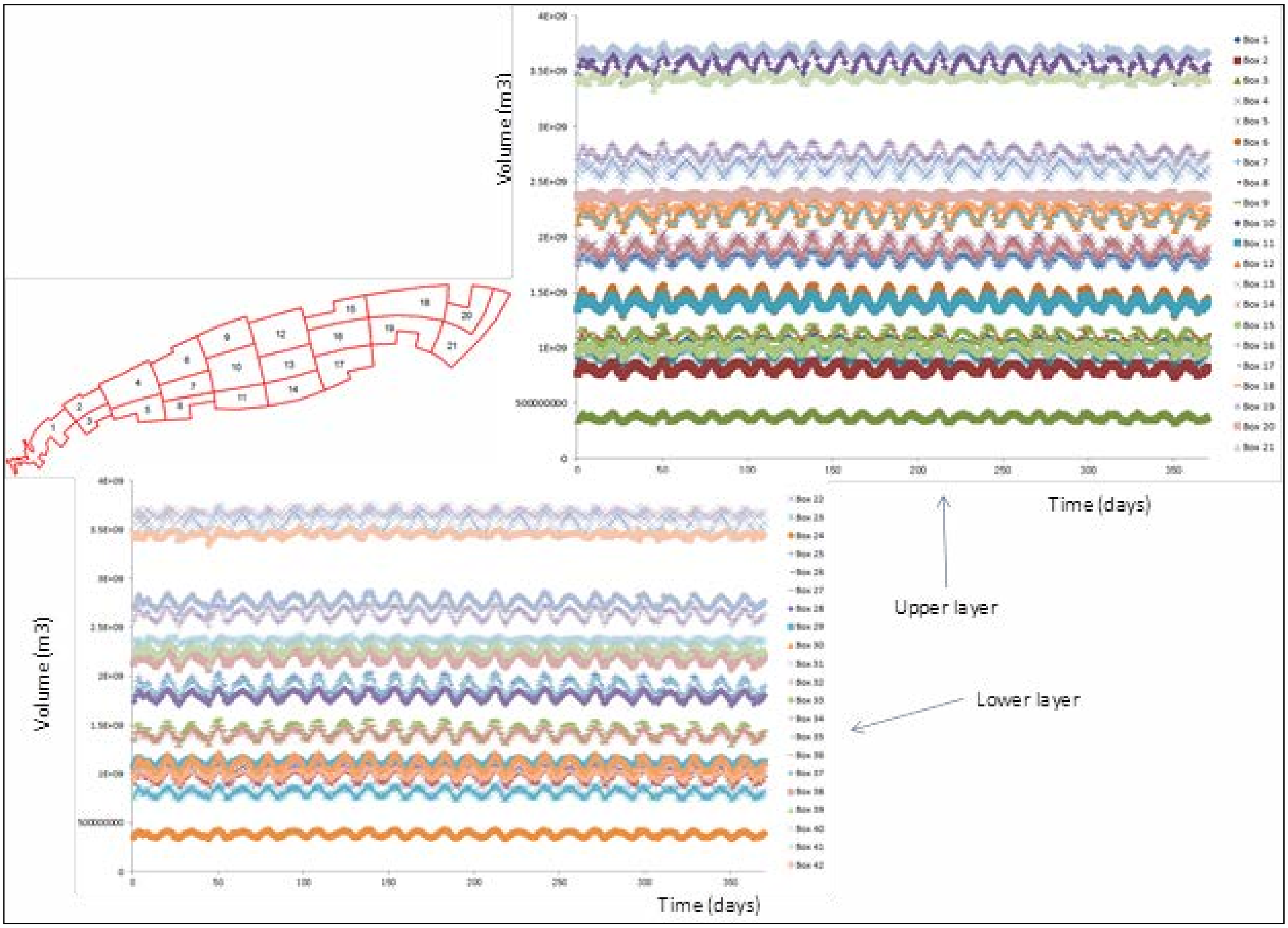


Figure 33: EcoWin.NET output of tidal cycles over a period of 1 year for all 42 model boxes.

Figure 33 shows the EcoWin output of water volume of all 42 LIS boxes, which is in agreement with the SWEM output, and EcoWin has reproduced identical tidal cycles for each box at both water layers.

Salinity: In EcoWin.NET, salinity is one of the state variables of the hydrodynamic object. For the LIS model, salinity for each EcoWin box uses the same initial value as SWEM, and its annual cycle could be precisely reproduced (Figure 34). In the LIS EcoWin model, the salinity of each box stabilizes from year 2 onwards, and the output has been well validated using data measured in 2007.

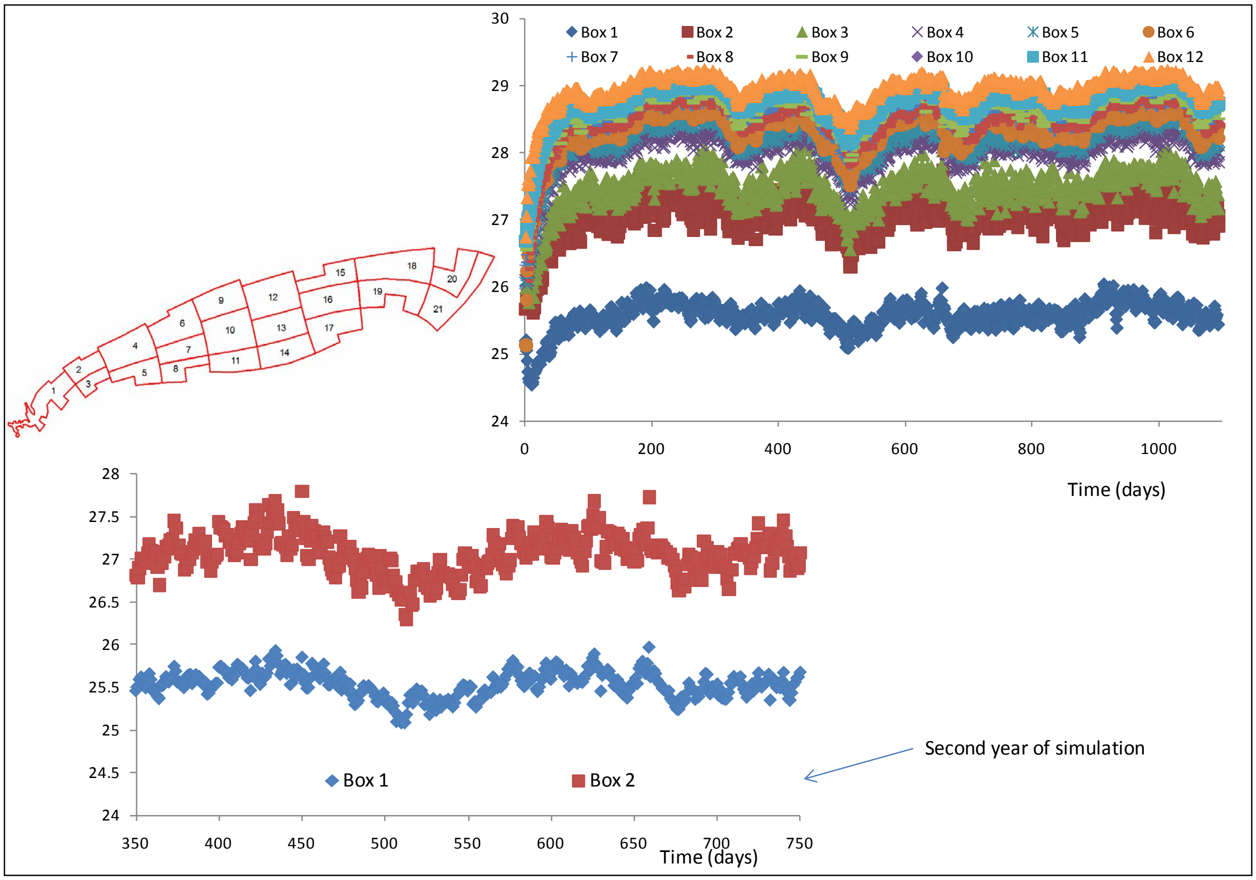


Figure 34: EcoWin.NET simulation of salinity for a subset of boxes (top right), and a detailed annual cycle for boxes 1 and 2, in western LIS (bottom left).

Temperature: Although EcoWin.NET, like SWEM, and many other models, can explicitly simulate water temperature by means of the heat budget equations, the focus of this work was on the aquaculture component of LIS, and we opted to apply a simple polynomial fit to data available for the Sound, since this was sufficient to force the relevant state variables, including individual shellfish growth, and optimized the model runtime.

In order to get optimal polynomial equations for water temperature simulation of LIS, polynomial regression of average water temperature against annual Julian days was made for EcoWin boxes based on historical data of 51 observation stations, and the results showed a very clear cubic curve pattern for all boxes over years, which indicated that the polynomial simulation method was feasible for LIS. In fact, the simulation results of other state variables e.g. phytoplankton biomass, shellfish growth etc. had fully justified our measurement on water temperature

Shellfish culture practice: Calibration of non-conservative water quality state variables, i.e. dissolved nutrients, phytoplankton, suspended inorganic and organic particulates, and DO, could only be addressed with the full shellfish model in place, because top-down control by filter-feeders affects the concentrations of a number of these state variables.

The definition of culture practice focused on the Eastern oyster and quahog, both of which are farmed in LIS—quahogs are sometimes considered more of a managed fishery. Culture practice includes data such as shellfish stocking density, mortality, and areas farmed. All of these items are typically forcing functions in carrying capacity models, and therefore will have a substantial effect on simulation results. Furthermore, accurate information on the data above may be used to indirectly estimate production, and therefore harvest. These two estimates can be compared with landings data, and with model outputs—these aspects help to increase confidence in simulation results.

The full description of culture practice is described in the previous section 'Shellfish Aquaculture Practices and Cultivation Areas', but a synthesis of the final options incorporated in EcoWin is presented below.

Due to these challenges, different approaches were considered in order to obtain a reasonable estimate of (1) the areas actually farmed; (2) their spatial distribution; (3) the production of oysters and quahogs. Culture practice can be obtained by four different methods, or combinations thereof:

1. Stakeholder interviews and consultation;
2. Direct sampling, in some cases complemented e.g. through remote sensing (Nobre et al, 2010);
3. Modeling;
4. Heuristics.

Method 1 was initially chosen, but there were difficulties in accessing the cultivation data required for this part of the model to operate successfully:

1. The shellfish lease areas for both CT and NY were made available in GIS (Figure 30), but there was no information as to what proportion of those areas is effectively cultivated;

2. There was no reliable information on landings in recent years, or their spatial distribution, for either Eastern oysters or quahogs;

Method 2 was not considered for budgetary reasons, and would have been a challenge because it requires close collaboration with industry. These same issues made it impossible to obtain area-specific individual shellfish growth curves from farms, and reliable local estimates of mortality.

There appears to be a serious governance problem with respect to the relationship between the shellfish industry and policy-makers and regulators, which in practice made dialog with the industry members almost impossible. Given the aims of this study, a strong link with industry stakeholders would have been extremely beneficial.

Consequently, a modeling approach (Method 3) was considered, and the results compared to Expert Knowledge (Method 4; see 'Shellfish Aquaculture Practices and Cultivation Areas').

The spatial distribution of leases that are effectively cultivated is required because the EcoWin.NET model assigns those areas to the appropriate model boxes, and subsequently simulates shellfish individual and population growth within them—as referred previously, the areas are a forcing function in EcoWin, and they therefore need to be estimated as accurately as possible.

The following steps were followed to estimate spatial distribution of shellfish culture through modeling:

I.) The lease areas supplied (Figure 30) were superimposed on the EcoWin model boxes in order to calculate the lease area (active or inactive) in each box;

II.) The potential annual harvest for the LIS oyster industry was calculated based on total lease area, stocking density, mortality, and culture period;

III.) The proportion between (II) and the estimated actual harvest was determined. A caveat for the methodology is the uncertainty in actual harvest, since landings are not formally reported to the regulator;

IV.) The spatially distributed lease area was proportionally reduced to provide final estimates of farmed areas to be assigned to EcoWin boxes.

The results obtained with this approach were discussed, and compared with expert knowledge (see 'Shellfish Aquaculture Practices and Cultivation Areas'); the team finally opted to use the heuristic method, leading to the areal distribution presented in Table 12.

Table 12: Spatial distribution of active Eastern oyster and quahog leases in Long Island Sound used for the standard model. Low and high density quahog areas estimated from harvest data (based on total revenue, which allows calculation of number of individuals harvested). For quahogs, 40% of harvest is considered to come from low density (10 individuals m^{-2}), and 60% from high density (100 individuals m^{-2}) cultivation areas.

		Totals	Box 23	Box 25	Box 27	Box 30	Box 33	Box 41
Eastern oyster	**% Area**	100	2	77	1	15	1	4
Eastern oyster	**Area (m^2)**	21246015 (5250 acres)	424920	16359432	212460	3186902	212460.2	849841
Quahogs	**% Area**	100	10	25	28	32	4	1
High stocking density quahogs (60%)	**Area (m^2)**	8333788 (2059 acres)	833379	2083447	2333461	2666812	333352	83338
Low stocking density quahogs (40%)	**Area (m^2)**	55404257 (13691 acres)	13851064	15513192	17729362	2216170	554043	55404257
Total quahogs	**Area (m^2)**	63738045 (15750 acres)	6373805	15934511	17846653	20396174	2549522	637381

The Eastern oyster (*Crassostrea Virginica*) was explicitly simulated (see individual model section above), providing model outputs of production. Quahogs (*Mercenaria mercenaria*) were not explicitly simulated, but their impact on the ecosystem was included in the EcoWin.NET ecosystem model.

The effect of quahog growth was simulated through a mechanistic model of the drawdown of both phytoplankton and organic detritus, using the following approach:

1. Four size classes were considered: Littleneck, Topneck, Cherry, Chowder. The mean live weight for each class was estimated by R.Rheault (see 'Shellfish Aquaculture Practices and Cultivation Areas'). Total clam harvest value and volume were estimated based on the historical trend (Figure 3). Landings were extrapolated to 2012 based on interviews with growers and managers that projected a modest increase in landings over this period. Based on these interviews we were also able to estimate the typical size distribution which we used to define the relative numbers landed for the various size classes in Table 13.

Table 13 shows the live weight and respective conversion to length for the hard clam. This was determined following McKinney et al. (2004), by first converting the live weight to tissue wet weight, and then tissue wet weight to length (Equations 1 and 2).

$$W_f = 0.372{W_t}^{0.97} W_f = 0.372{W_t}^{0.97}$$ **Eq. 1**

$$L = 10^{(\log(\frac{W_f}{0.00005})/3.09)} L = 10^{(\log(\frac{W_f}{0.00005})/3.09)}$$ **Eq. 2**

Where:

W_f: fresh tissue weight (g)

W_t: total fresh weight with shell (live weight, g)

L: length (mm)

Table 13: Morphometric data for hard clam.

Size class	Live weight (g)	Tissue wet weight (g)	Length (mm)
Liitlenecks	51.6	17.1	61.7
Tops	93.2	30.2	74.3
Cherries	147.5	47.2	85.9
Chowders	246.2	77.7	100.8

2. For each class, a clearance rate was calculated based on length and on water temperature in each of the relevant model boxes for every model timestep. Hibbert (1977), and Doering & Oviatt (1986) provide equations relating clearance rate to these parameters—we chose the latter formulation, which gives slightly higher clearance rates, and therefore has a slightly greater effect on food limitation for oysters (Equation 3).

$$CR = L^{0.96}T^{0.95} CR = L^{0.96}T^{0.95}$$ **Eq. 3**

Where:

CR: clearance rate (ml ind^{-1} min^{-1})

L: length (cm)

T: temperature (^{o}C)

3. The culture practice component of this work provides the abundance (individuals per m^{-2}) of quahogs in LIS, but not the frequency distribution of the different size classes. We used data from Fegley (2001) to derive the distribution shown in Table 14, and refined the dataset to obtain an approximate match to the four size classes in Table 13.

Table 14: Frequency distribution of shell length in Mercenaria mercenaria from eastern Long Island Sound (data from Malinowski, in Fegel, 2001).

Length (mm)	%	wet tissue weight (g)	Total wet weight (g)
55	5	11.9	35.7
65	10	20.0	60.8
75	20	31.1	95.9
85	35	45.8	142.9
95	25	64.6	203.6
105	5	88.0	280.1

On that basis we estimated the proportions as: 15% littlenecks, 20% topnecks, 35% cherries, and 30% chowders. This confirms the results of the determination of ratios of size classes based on historical trends (R. Rheault, East Coast Shellfish Growers Association, pers. comm.).

4. The clearance rate determined for each size class at each model timestep was scaled to an overall clearance based on abundance, proportion of that size class, and habitat area within 'quahog boxes'. The overall volume cleared was integrated for the four size classes, and used to determine the CHL and organic detritus (detrital Particulate Organic Matter [POM]) removed at every model timestep.

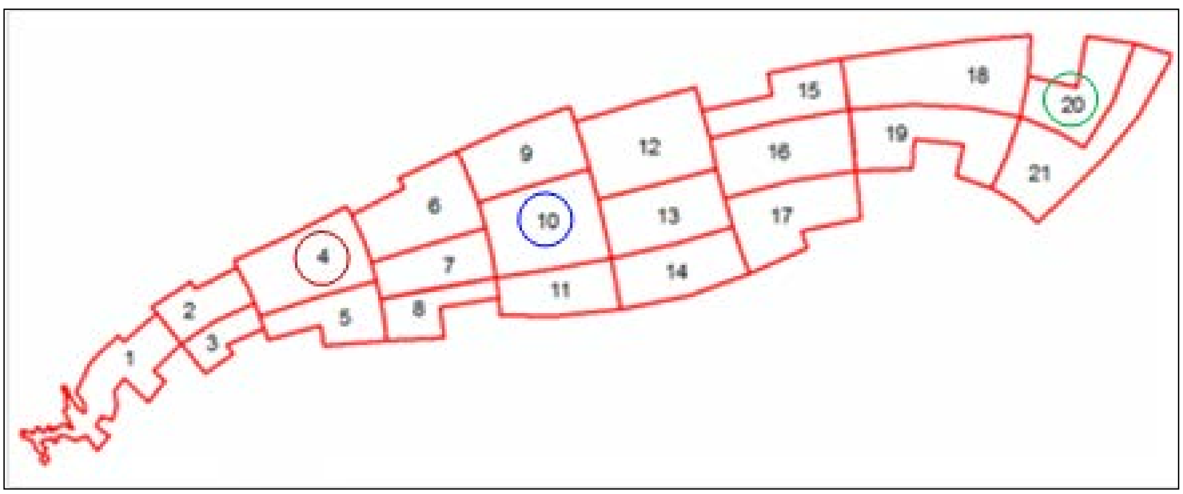


Figure 35: Boxes chosen for presentation of validation curves for nutrients and phytoplankton (Box 25 is below Box 4).

The model assumes a fixed proportion of the four size classes throughout the year, which is incorrect, but nevertheless considerably better than using both a constant weight and clearance rate for that weight. The aggregate clearance rate varies between 0.5 and 3.2 L ind^{-1} h^{-1}, where one 'individual' is a composite of the proportion of each size class and its clearance rate at a particular water temperature.

Dissolved nutrients and phytoplankton: Non-conservative water quality state variables simulated by EcoWin.NET were validated against SWEM. The validation curves are shown for boxes 10, 20, and 25 (Figure 35, Figure 36).

The three boxes were selected to broadly represent the eastern, central, and western parts of the Sound. Box 25 was chosen because it contains the largest area of cultivated oysters, and a substantial area of quahogs, and is therefore a bottom box, below Box 4.

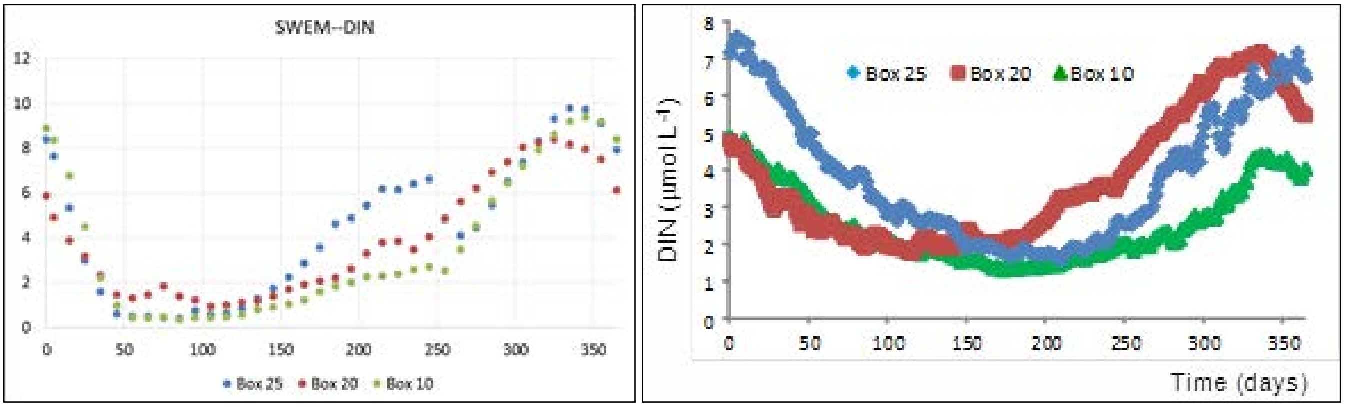


Figure 36. Comparison of annual cycle for dissolved inorganic nitrogen (DIN) simulated in SWEM and EcoWin.

EcoWin and SWEM show similar concentration ranges and annual pattern for dissolved inorganic nitrogen (DIN); this is encouraging, especially since no model coefficients (such as half-saturation constants or potential rate for primary production) were shared between the two models. This was an intentional option, since the two models are quite different in scale, formulations, and number of state variables. In the version of SWEM used for comparisons, bivalves are included by considering a constant biomass of 2.8 g DW m^{-2} in all of LIS.

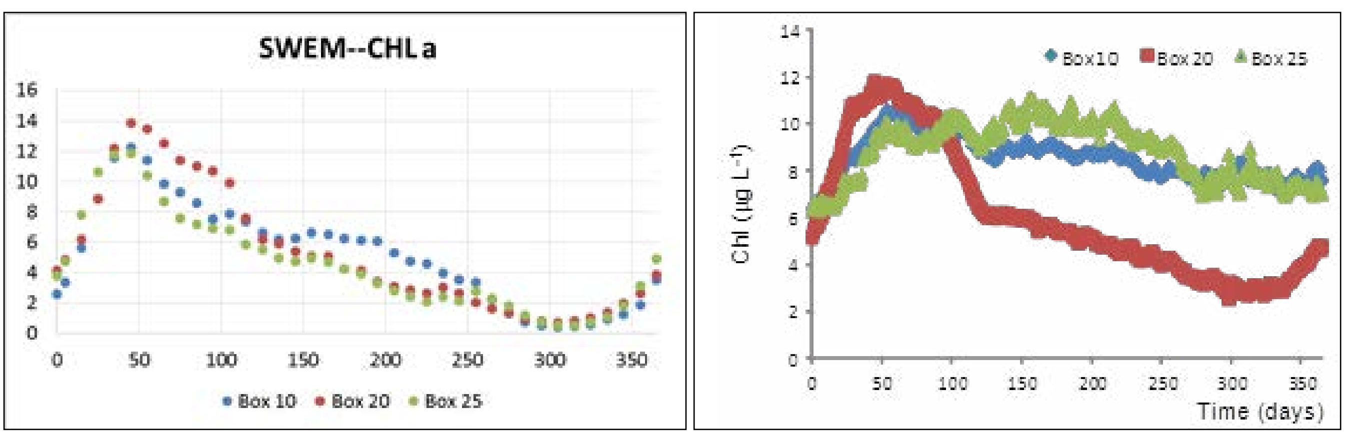


Figure 37: Comparison of annual cycle for chlorophyll simulated in SWEM and EcoWin.

The same exercise was done for CHL (Figure 37), with somewhat different results. As before, there is an approximate match for concentration ranges, and the two model curves are a good match in the eastern part of Long Island Sound. However, in the central Sound (Box 10) EcoWin.NET does not reproduce the drop in concentration observed in SWEM for the latter part of the year, and values in the western Sound remain elevated in EcoWin for most of the year.

We have opted to accept this deviation for two reasons:

(1) because given the main nutrient loading to the Sound is from the western part, including the city of New York, it seems inconsistent to arrive at lower simulated concentrations of CHL in the western section. Data from CT DEEP confirm that CHL concentrations are higher in the western Sound (i.e. see Figure 7; Rice and Stewart, 2013).

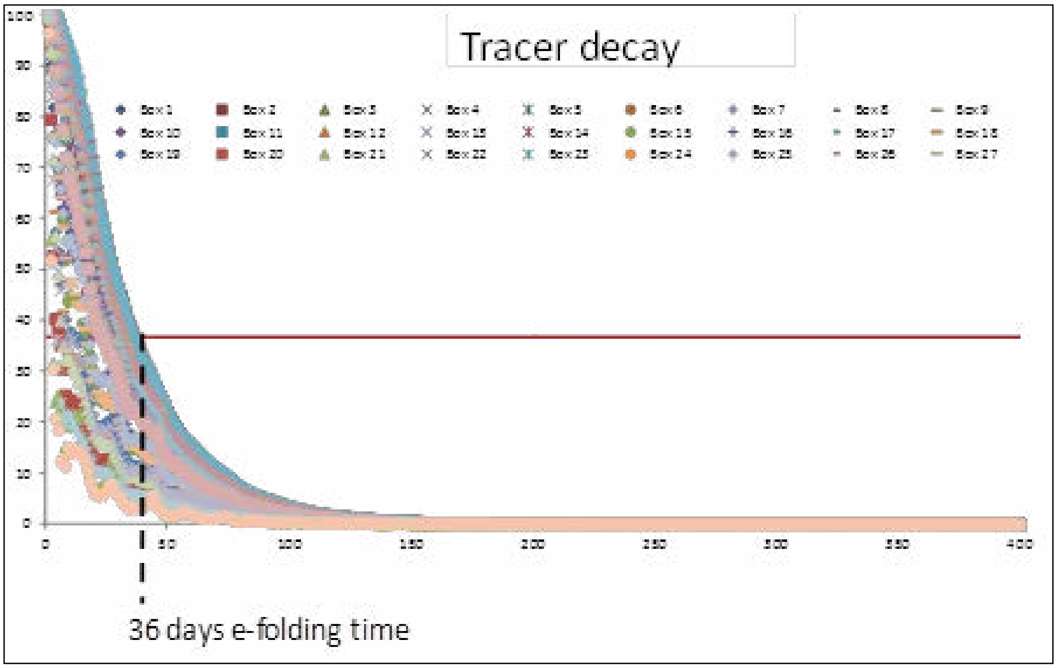


Figure 38: Simulation of e-folding time in EcoWin, based on hydrodynamics from SWEM.

Higher CHL concentrations in the eastern Sound stations might occur if LIS was a fast-flushing system, such that nutrients discharge to the western sound resulted in phytoplankton blooms further to the east, but this is not the case (Figure 38); estimated water residence times vary from 56 days (NEEA, 1999) to 60-90 days (Bricker et al., 1997, 2007). Neither is suspended matter sufficiently elevated to impose severe light limitation on primary production, with a mean concentration in 1996 of 6.5 ± 3.2 mg L^{-1} (Rose, pers. com.).

(2) the second reason is that peak CHL concentrations in LIS vary widely in value (1.8-65 mg L^{-1}) and timing (March-September; Rice & Stewart, 2013), which makes it unreasonable to attempt to identify a 'typical' year.

There is a further possibility that top-down control of phytoplankton biomass by commercially grown filter-feeding bivalves (oysters and quahogs), which are stocked primarily in the western Sound, might explain a reduction in CHL concentrations in that region. However simple budget calculations based on areal coverage, stocking density, and clearance, do not support this.

In any case, the bivalve component is not included in the SWEM model, which shows the decrease in spring and summer, and is explicitly simulated over multi-year cycles in EcoWin.NET, where this effect is only marginally reflected in algal concentration changes (see 'Results and Conclusions' section).

Oyster cultivation model

Modeling of oyster populations requires an adequate understanding of culture practice, as detailed previously. In EcoWin.NET, this is included in the input file (Table 15), and used to drive the interaction between the *Man* object and the *Zoobenthos* object. The individual model shown earlier was coded into the EcoWin framework, and (using object-oriented programming, (see background in Ferreira, 1995), a population class for *C. virginica* was built which considers 20 weight classes, spanning 0-100 g total fresh weight. The lower weight classes have a smaller amplitude of 3 g, to reduce numerical dispersion artifacts. Class 1 (0-3 g) is used for seeding, and animals in classes 18 and above are considered harvestable.

Seeding takes place from Julian Day 121 (May 1) onwards, for a period of 90 days, during which seed is applied uniformly to reach an overall stocking density of 62 individuals m^{-2}. The calculated stocking area is weighted using a factor of 1/3, as is the case in other model applications (e.g. Ferreira et al., 2008; Nunes et al, 2011), to account for the approximate duration of the culture cycle—in the model. The annually seeded area is thus one third of the total area nominally under cultivation, which will allow the farmers to harvest their crop annually, since every year there will be a section of the farmed area containing market-sized oysters. On the basis of interviews with shellfish farmers (Table 5), harvest is simulated for the entire year, rather than a specific period.

The population model applies a well-established equation (Eq. 4) relating population growth to both scope for growth, obtained through the individual model, and to mortality.

$$\frac{\partial n(s,t)}{\partial t} = -\frac{\partial\left[n(s,t)g(s,t)\right]}{\partial s} - \mu\left[(s)n(s,t)\right] \qquad \textbf{Eq. 4}$$

where t, time; s, weight class; n, number of animals; g, scope for growth (growth rate); u, mortality

Harvest is regulated by the availability of market-sized animals, rather than by the activity of the watermen. Better data with respect to effort (area, period and duration), would allow a complementary approach to be tested, with harvesting based on the boats and crews that work the cultivation areas over the year.

Scenario analysis

Three different stocking area scenarios were considered, apart from the standard model setup shown in Table 7 for oysters and Table 8 for clams. The rationale for the choices of low, high, and potential areas under cultivation is given in section ‘Shellfish Aquaculture Practices and Cultivation Areas.’ In each case, input files for the EcoWin model were prepared, with alterations only to the stocking areas for oysters and quahogs (see Table 7 and Table 8). No other changes were made to the input files—the calibration described earlier was used, i.e. the setup for the standard model.

Table 15: Excerpt from the parameters input file for EcoWin.NET, listing the names and values for some of the parameters used in the Long Island Sound model.

Object	Parameters	Parameter name	Units	Value
Zoobenthos	**18**	Enable Eastern oysters		Yes
		Enable high density Quahogs		Yes
		Enable low density Quahogs		Yes
		Enable wild species		No
		Number of Eastern oyster classes		20
		Eastern oyster class amplitude	g TFW ind^{-1}	3
		Eastern oyster mortality	d-1	0.0005
		Eastern oyster natural seeding		No
		Eastern oyster first seeding day	d	121
		Eastern oyster individual growth period	d	977
		TotalFW to meatDW for wild species		27
		Wild species individual weight	gTFW	4.6
		Wild species filtration rate	L $animal^{-1}$ h^{-1}	0.4
		TotalFW to meatDW for quahog		24.7192673
		Littleneck quahog individual weight	g TFW	51.6
		Topneck quahog individual weight	g TFW	93.166
		Cherry quahog individual weight	g TFW	147.49
		Chowder quahog individual weight	g TFW	246.24
Man	**13**	Active aquaculture area	No units	1
		Crop seeding area	No units	0.33
		Culture cycle	d	1098
		Nominal APP	No units	0
		Enable Eastern oyster harvest		Yes
		Number of Eastern oyster classes		20
		Eastern oyster limit seed class	Class number	1
		Eastern oyster limit harvest class	Class number	18
		Eastern oyster first seeding day	d	121
		Eastern oyster seeding period	d	90
		Eastern oyster first harvest day	d	1
		Eastern oyster harvest period	d	364
		Eastern oyster first harvest year	y	3

An identical set of model runs was executed for each scenario, in order to obtain oyster harvest and a general mass balance for the variables of interest from an environmental point of view, with a particular focus on nitrogen removal by oysters.

Farm Aquaculture Resource Management model (FARM)

A local farmscale modeling approach was used to estimate the potential use and benefit of shellfish aquaculture in nutrient remediation without the cost and time required for implementation and for comparison to the EcoWin ecosystem level model results. Specifically, we used the well-tested Farm Aquaculture Resource Management (FARM) model to evaluate the potential for cultivation of Eastern oyster (*Crassostrea virginica)* to reduce eutrophic symptoms. Note that while there are similarities, the local-scale FARM model has significant differences from the EcoWin ecosystem-scale model. For example, in FARM there is no 'man' object to harvest the shellfish, there is also no interaction (i.e. potential food depletion) with adjacent farms as there is in EcoWin. FARM is therefore suitable to analyze local aspects of nutrient drawdown, and potential valuation of individual farms for nutrient credit trading purposes, but it is not appropriate for scaling exercises where seston depletion due to increased stocking density at a particular farm may affect the carrying capacity at nearby farms.

While there are robust clam and oyster industries in LIS, here we apply the FARM model to simulate oyster production only, because an individual model for quahogs was not developed in this study (see 'AquaShell Individual Model'). This is a potential area for further research, but an improved description of culture practice for both species remains a higher priority. Additionally, the simulation will be restricted to CT waters since little is known about NY aquaculture practices. Aquaculture practices based on interviews with CT growers are described in Table 5 and 'Aquaculture Practices and Cultivation Area'. To summarize, CT oyster farm sizes vary from 5 to greater than 10,000 acres (Kristen DeRosia Banick, CT Department of Agriculture, Bureau of Aquaculture, pers. comm.; Table 5), and culture practice typically includes harvest of one and two inch oysters from restricted areas and growout to the 90 g at least three inch, legal harvest size on lease areas for one to two years. A small number (~10%) of growers use hatchery spat. Most aquaculture operations are subtidal with no gear. There is interest, however, in expanding to rack and bag and /or floating bags which are currently used by about 10% of growers in the Sound.

The FARM model has been tested in the United States, European Union, China, and elsewhere (Ferreira et al., 2007, 2009; 2011; 2012; Nunes et al., 2011; Bricker et al., 2014; Saurel et al, 2014; Rose et al., 2014a). The model combines physical and biogeochemical models, shellfish growth models, and screening models for determination of shellfish production and assessment of water quality changes on account of shellfish cultivation. The model is useful for decision support for aquaculture siting (e.g., Silva et al., 2011; Ferreira et al., 2012), because it evaluates both production and ecological carrying capacity (e.g. changes in dissolved and particulate outputs due to aquaculture activity). It can be used for marginal analysis of farm production potential and profit maximization, while assessing potential credits for carbon and nitrogen trading (Ferreira et al., 2007, 2009, 2011; Rose et al., 2014; www.farmscale.org). The model converts estimated nitrogen removed by the oysters to human population equivalents, and calculates the potential value of the ecosystem service represented, providing a substitution or 'avoided' cost of land-based nutrient removal that would serve as additional revenue to the farmer in a nutrient-trading program (see section 'Economic framework'). Also evaluated are changes in the eutrophication indicators, CHL and DO, resulting from the filtration by oysters during the culture period, using components of the ASSETS model (Bricker et al., 2003). The general layout for the FARM model is shown in Figure 39 and is applicable to bottom culture but also to suspended culture from rafts or longlines. Inputs for shellfish modeling include data on culture practice (e.g. farm layout, species, stocking densities) and environmental parameters, including shellfish food particles in the water column (i.e., phytoplankton and detritus; Figure 39). The model output of interest here is the net mass of nitrogen removed through uptake of phytoplankton and detritus, which results from the balance of shellfish filtration and return of nitrogen to the environment through pseudofeces and feces, natural mortality, and excretion.

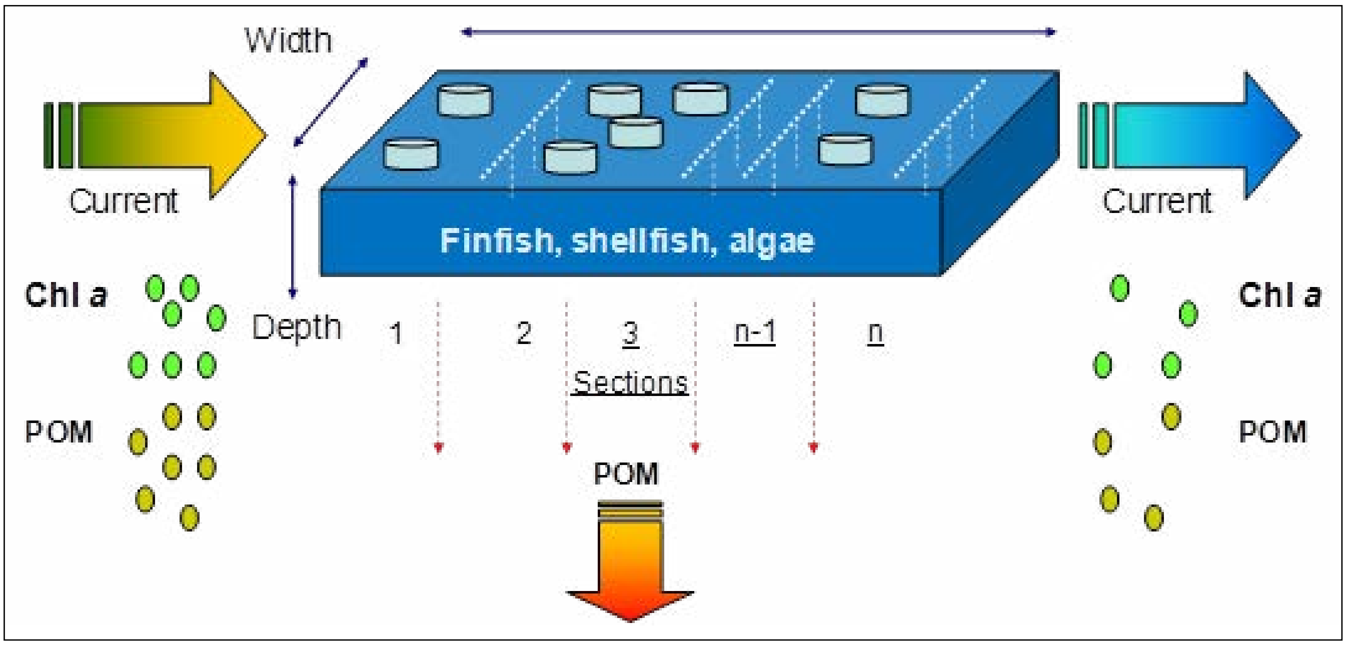


Figure 39: Farm layout (rope and bottom culture) used in FARM model (Ferreira et al., 2007, 2009).

Typical 'extensive' bottom culture practices for Eastern oyster that are employed by Northeast region growers as described above were used for the simulation. There is variability in the density of oysters used for seeding (30 – 200 oysters m^{-2}; Rose et al., 2014a). Mortality varies inter-annually within a broad range of 30-80% over the culture period, relating for instance to *Vibrio* outbreaks.

In addition to removal of nitrogen by oyster aquaculture, the avoided cost of wastewater treatment, the ecosystem service that is provided by the shellfish filtration, was estimated based on a substitution value of \$12.40-14.40 kg^{-1} (\$5.63 - \$6.55 lb^{-1}) for the estimated removed nitrogen (Lindahl et al., 2005, 2011). Local estimates suggest this value may be as high as \$130 kg^{-1} (\$60 lb^{-1}) based on WWTP costs to remove N from wastewater (see description in section 'Economic Framework'). The FARM model uses the lower value for the calculation, but the valuation may be easily scaled up if required. People-equivalents were calculated based on an annual per person N load of 3.3 kg (Ferreira et al., 2007). The results for the simulated farm were scaled up to evaluate potential removal using; 1) the existing acres of oyster habitat, and 2) estimates of potential future oyster grounds based on legal, policy, and environmental criteria in the manner of Silva et al. (2011) to provide insight about the potential improvement of water quality with expansion scenarios (Table 7).

The results for the simulated farms were scaled up to evaluate total present removal using estimates of current leased acres of oyster habitat. Estimates for potential removal were made using estimates of the potential acres available for expansion scenarios.

FARM application in Long Island Sound

The FARM model was used to simulate growth, production and nutrient removal at the farm scale using the four long term CT DEEP monitoring stations, Stations A4, 09, H2 and M3. The stations were used to represent the four Long Island Sound zones, the Narrows, Western, Central and Eastern, respectively, as the locations of simulated farms (Figure 1). As much as possible, input variables used for FARM are the same as those used for the EcoWin simulations (see Table 15). We have used a seeding density of 62 oysters m^{-2}, a density commonly used by growers in this area. Mortality is highly variable from year to year; we used 55% mortality per cycle, the same as for the system scale model, for a 1098 d cycle on a 4-acre farm where 3 acres are in cultivation. Current speeds and environmental data were representative of the 4 stations. Estimated current and potential future removal of nitrogen was evaluated using cultivated acres from Table 7 and is described in 'Upscaling FARM model results in Long Island Sound.'

As noted, the avoided cost of wastewater treatment, was estimated based on a substitution value of \$12.40-14.40 kg^{-1} estimated by Lindahl et al. (2005, 2011). Local estimates suggest this value may be as high as \$130 kg^{-1} based on WWTP costs to remove N from wastewater and in any case, are an underestimate based on more recent work by Rose et al. (2012). The FARM model uses the lower value for the calculation and thus may underestimate the ecosystem service value by as much as an order of magnitude.

FARM application in Great Bay Piscataqua

The FARM model was applied to two simulated GBP farms, Choice Oyster and Granite State, using the same environmental data but different current speeds. Seeding density used was 100 oysters m^{-2}, typical of growers in the Northeast region, with a cultivation period of 1095 days, the typical period of growth in GBP. We used a morality of 50%, an average since it is highly variable from year to year. Current and potential future nitrogen removal was estimated based on current and potential expanded acres of cultivation described in 'Upcaling FARM model results in Great Bay/Piscataqua Region Estuaries.' The same substitution value used in the LIS simulation for the ecosystem service provided by the removal of nitrogen by oysters was used for the GBP simulations.

Assessment of Estuarine Trophic Status (ASSETS)

The assessment of eutrophication status for these waterbodies was evaluated with the Assessment of Estuarine Trophic Status (ASSETS) assessment method. It is a Pressure-State-Response model that is straightforward in both required parameters and calculations and is designed to provide broad management-level guidance, including for poorly sampled coastal systems. It was originally developed to evaluate conditions in 140 U.S. coastal system assessment (Bricker et al., 2003, 2007) and has been tested extensively in European systems (e.g., Ferreira et al., 2003, 2007a; Devlin et al., 2011; Garmendia et al., 2012), and in China (e.g., Xiao et al., 2007). Both LIS and GBP have been evaluated in the two previous national assessments (Bricker et al., 1999, 2007) and in a study of Northeast estuaries (Bricker et al., 2006), and these results can be compared to the results of the assessments in this study.

Briefly, the ASSETS model evaluates three components of eutrophication: (1) *Influencing Factors* combines natural susceptibility and human-related nutrient inputs; (2) *Eutrophic Condition* estimates level of nutrient related impact based on five indicators (CHL, macroalgae, DO, nuisance/toxic blooms, changes in seagrass distribution) that, for each one, combine the concentration, spatial coverage and frequency of occurrence of problem levels and then combine into a single value; and (3) *Future Outlook* evaluates potential changes that may occur based on natural susceptibility and expected changes in nutrient load. The final step combines the categorical (i.e., High, Moderate, Low) results for the three components into a single overall rating (Figure 40; Bricker et al., 2003, 2008). The assessment uses quantitative and qualitative data to determine trophic status. For example, to evaluate the 'typical' extreme concentrations over the annual cycle, algal bloom concentrations are represented as the 90^{th} percentile of annual CHL data, and bottom water DO by the 10^{th} percentile of annual measures. Note also that the CHL and DO indices in the ASSETS methodology include concentration, spatial coverage and frequency of occurrence of extreme concentrations which result in a rating indicating a worse impact than it would be using concentration alone. For example, concentrations of CHL are considered moderate but are observed over a large spatial area and occur on a seasonal basis, thus the rating is High rather than Moderate. Additionally, the rating scale for Influencing Factors groups nutrient ratings of Moderate High and High into the High category in a precautionary approach. This may result in a seemingly higher impact rating than would be the result if the Influencing Factors were on a five grade scale.

For the ASSETS analysis the LIS was divided into 4 zones of equal area (each 815 km^2; see Figure 1), all are considered seawater zone where salinities are >25 psu. The GBP was divided into two zones for the analysis, a mixing and a seawater zone (Figure 4).

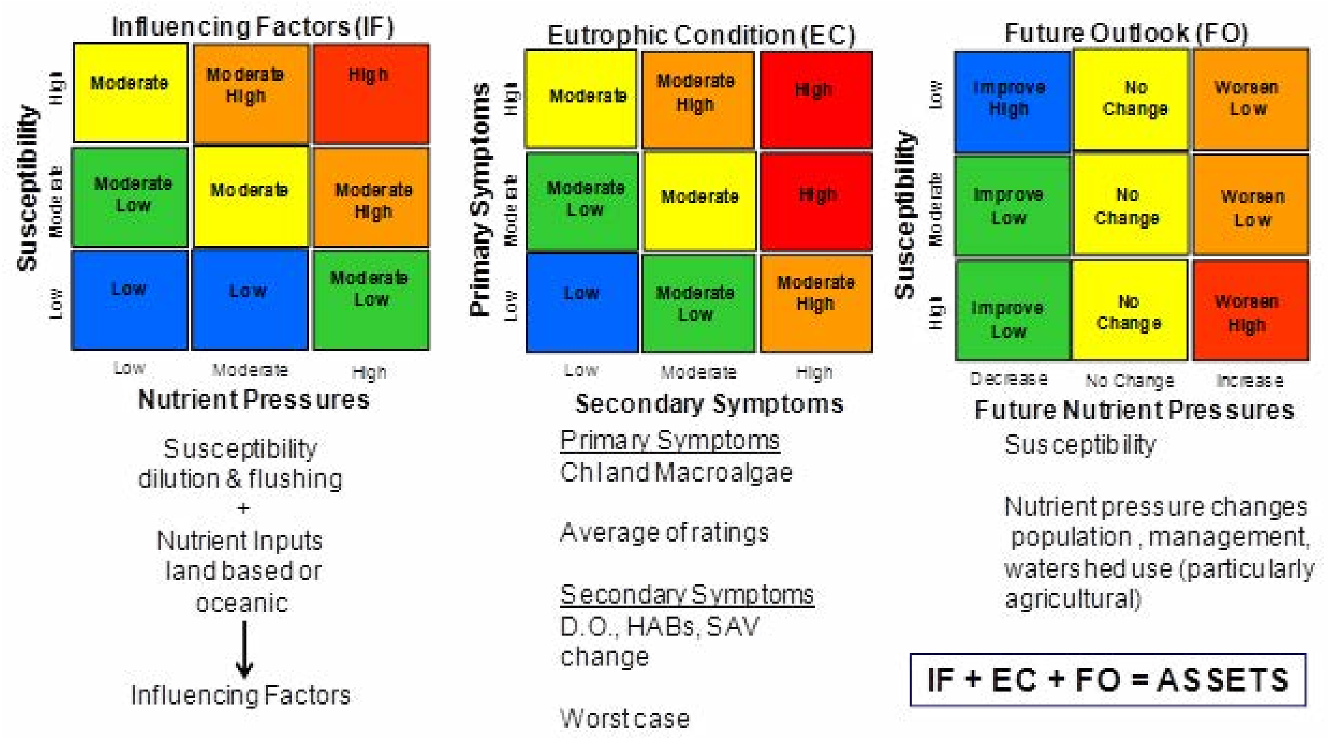


Figure 40: Summary of Assessment of Estuarine Trophic Status (ASSETS) method, a Pressure-State-Response framework that uses matrices in each of the three components, Influencing Factors, Eutrophic Condition and Future Outlook. From: Bricker et al., 2003. Bioextraction of Nitrogen.

Economic Framework

This section describes the ecosystem service benefits of bioextraction based on a literature review prepared by Northern Economics (2012; Appendix 1) and an analysis of the economic value of nitrogen removal by bioextraction using the replacement cost or cost-avoided method. The case study areas include the CT part of LIS, and GBP with a focus on oyster aquaculture. A related policy question evaluated here is whether or not shellfish producers should be included in the current Connecticut Department of Energy and Environmental Protection Nitrogen Credit Exchange.

Bioextraction of Nitrogen

Within the scientific literature there is growing recognition of the central role shellfish can play in the maintenance and stability of coastal ecosystems. Oysters, in particular, have been labeled ‘keystone’ or ‘cornerstone’ species in selected marine environments (Isaacs et al., 2004) because they support a complex community of species by performing functions essential to the diverse array of species that surround them. There is also increasing recognition that some shellfish species may impact or control ecological processes, so much so that they are included on the list of ’ecosystem engineers’—organisms that physically, biologically or chemically modify the environment around them in ways that influence the health of other organisms (Jones et al., 1994). The ecological functions and processes performed or affected by shellfish (see section ‘Introduction and Background’) contribute to human well-being by providing a stream of valuable services over time and are commonly referred to as ’ecosystem services’ (The Millennium Ecosystem Assessment, 2005).[5] Table 16 outlines the four broad categories of ecosystem services relevant to shellfish production and restoration.

Table 16: Ecosystem services provided by shellfish

Provisioning	Commercial, recreational and subsistence fisheries
	Aquaculture
	Fertilizer and building materials (lime)
	Jewelry and other decoration (shells)
Regulating	Water quality maintenance
	Protection of coastlines from storm surges and waves
	Reduction of marsh shoreline erosion
	Stabilization of submerged land by trapping sediments
Supporting	Cycling of nutrients
	Nursery habitats
Cultural	Tourism and recreation
	Symbolic of coastal heritage

Shellfish Bioextraction and Water Quality Trading Markets[6]

The concept of a nutrient credit trading program is the development of a 'market-based' approach to help control nutrient discharges by developing economic incentives for achieving water quality standards within a defined area (Lindahl et al., 2005, Lal, 2010; Jones et al., 2010; Profeta and Daniels, 2005). A nutrient trading program allows a source discharging nutrients to the coastal zone to meet its regulatory obligations by using nutrient reductions created by another source that has lower pollution control costs (EPA, 2003, 2008). Typically there is an upper limit for nutrient discharges, determined by a Total Maximum Daily Load (TMDL) analysis such as the one conducted for LIS (NYSDEC and CTDEP, 2000), that can be discharged to a waterbody above which there will be negative consequences to water quality. Each discharger within the watershed is allocated a specific amount that they can emit, the total equal to the TMDL limit. Dischargers who reduce their nutrient discharges below their allocated target levels can sell their surplus reductions or 'credits' (also called 'offsets') to other dischargers in the same watershed. The identification and quantification of the value of nitrogen removal through shellfish harvest can used be used by policy makers in this scenario to develop a nutrient trading market since shellfish are a sink for nutrients thus generating 'credits' that could be sold to dischargers (Ferreira et al., 2007; Lindahl et al., 2005).

The economic argument for including shellfish producers is that it may be possible to achieve the desired environmental outcome more efficiently (i.e., at a lower unit cost of abatement) than traditional 'command-and-control' approaches because it gives sources the flexibility to search for the lowest cost pollution reduction option. One potential option available to sources is the purchase of nutrient (e.g.,

[5]A number of authors (e.g., Boyd and Banzhaf 2007; Fisher and Turner, 2008) have pointed out shortcomings in the way ecosystem services were defined and classified by The Millennium Ecosystem Assessment. While this report acknowledges these critiques, it accepts The Millennium Ecosystem Assessment's definition and classification scheme for simplicity's sake.
[6]For a literature review on the use of Water Quality Trading please see Appendix 1.

nitrogen or phosphorus) credits from shellfish enhancement projects which are nutrient sinks (Ferreira et al., 2007). For example, the costs for controlling nitrogen loads at a point source such as a WWTP compared to restoring shellfish production can vary significantly (Rose et al., 2014a), creating the impetus for water quality trading. Through water quality trading, municipal and industrial WWTP that face higher nitrogen removal costs—particularly as nitrogen concentration limits are tightened—could meet their regulatory obligations by purchasing 'pollutant reduction credits' from, say, a shellfish aquaculture operation if the cost is lower and the nitrogen reductions achieve targets. The establishment of a market in which nutrient credits or offsets are freely traded can generate a market price for nutrient reduction. This market price represents the interests of WWTP affected by nitrogen concentration limits to reduce nitrogen emissions in the most cost-effective manner. Although the programs that are presently in place focus on point source trading, this concept includes trading among non-point sources though the establishment and certification of 'credits' is more difficult than for point sources (Lal, 2010; STAC, 2013a, 2013b; Walker and Selman, 2014).

Nitrogen credit trading at the watershed scale is now a reality in parts of the United States (Ferreira et al., 2011). In 2003, the EPA issued a national water quality trading policy to support the development and implementation of trading in water quality management (EPA, 2003). The agency advocates water quality trading as a cost-effective means to preserve and improve water quality, and a number of programs are in various stages of development (e.g. Chesapeake Bay, Long Island Sound and others in STAC, 2013a). While the policy doesn't specifically refer to shellfish enhancement as part of a nutrient trading scheme, it does support the creation of water quality trading credits in ways that achieve ancillary environmental benefits beyond the required reductions in specific pollutant loads, such as the creation and restoration of wetlands, floodplains and wildlife and/or waterfowl habitat. Using shellfish enhancement as part of a nutrient trading scheme would fulfill this objective, as shellfish enhancement would provide multiple economic and environmental benefits (Golen, 2009).

The Connecticut Nitrogen Credit Exchange was created in 2002 to improve nutrient related hypoxia conditions in Long Island Sound bottom waters. The Exchange provides an alternative compliance mechanism for 79 WWTP throughout the state. During 2002-2009 the total value of credits bought and sold was $45.9 million, representing 15.5 million nitrogen credits exchanged. The cost savings due to trading through the Exchange is estimated to be $300 - $400 million over the alternative of implementing nitrogen removal projects at all 79 facilities. The cost savings due to trading through the Exchange is estimated to be $300 - $400 million over the alternative of implementing nitrogen removal projects at all 79 facilities (CT DEP, 2010). At present the Connecticut Exchange includes only point sources. The CTDEP has conducted an evaluation of the potential of stormwater non-point sources to be included in the trading program and found that costs to generate a nitrogen credit far exceed those applicable to WWTPs. Further, there would be accountability constraints because of the difficulty of tracking and monitoring diffuse sources within Connecticut's 169 municipalities. However, investigation continues on how to effectively include non-point sources in this program.

Similarly, the Chesapeake Bay states are developing water quality trading programs to provide regulated sources options to remain in compliance with their individual nutrient waste load allocations, permit requirements, or aggregate mass load caps (Maroon, 2011; Stephenson and Shabman, 2011, Branosky et al., 2011). Furthermore, a number of US states and countries outside of the US are considering the use of shellfish in nutrient trading schemes in addition to other nitrogen reduction measures. As such, researchers are applying theoretical models to assess the nitrogen reduction potential of shellfish farms and for the valuation of nitrogen credits (Getchis and Rose 2011; Rose et al., 2014a). The nitrogen reduction costs directly associated with a shellfish enhancement project are the project's costs of production, including the costs of labor and capital. If the shellfish produced by a project can be sold on the open market, the income from these sales can offset project costs (Stadmark and Conley, 2011). The costs for nitrogen reduction measures also depend on the amount of nitrogen removed during the harvesting of shellfish (Stadmark and Conley, 2011). The nitrogen reduction potential of shellfish enhancement projects is the subject of intense (sometimes controversial) research (e.g. Rose et al., 2012).

Brumbaugh and Toropova (2008) and Lindahl et al. (2005) postulate that if a well-designed and regulated market existed for trading the nitrogen removed by shellfish, it should have the effect of spurring further investments in shellfish enhancement. By providing shellfish enhancement projects an opportunity to earn additional revenue when they sell nutrient reduction credits, a trading program allows projects involved in culturing shellfish and restoring beds to capitalize on the nitrogen removal services provided by their activities. However, Stephenson and Shabman (2011) note that participation of shellfish enhancement projects in a trading program is contingent upon buyers (e.g., WWTP) and sellers (e.g., shellfish growers) reaching an agreement on the amount of credits to be provided to the buyer, how it will be documented that the credits were provided, and the payment terms for the provision. Stephenson and Shabman (2011) suggest that these agreements will be described in a contract between the buyer and seller. The contract also will assure the buyer that the conditions set by the regulatory authorities are being met by the seller so that the credits can be used as offsets in the trading program. The authors note that nitrogen and other nutrient offsets lend themselves to measurement and verification, making credits especially amenable to a contracting process where payment is contingent on demonstrated results (Stephenson and Shabman, 2011). In a similar vein, DePiper and Lipton (2011) discuss how an enforced contract can result in a switch in how an oyster reef is managed, e.g., in the absence of a contract, oysters may be harvested more aggressively, and consequently the desired nutrient removal levels may not be achieved.

Stephenson and Shabman (2011) also suggest that a time-referenced benchmark would appear necessary in order to prevent an existing shellfish enhancement project from claiming credits for past investments. Once such time-referenced baselines are established, expansions of nutrient credit services (e.g., expanded oyster aquaculture production beyond the referenced date) could be counted as new (additional) services and credited.

Some researchers are concerned with participation of shellfish enhancement projects in nutrient trading schemes. Golen (2009) raises the issue of how to satisfactorily define and measure nitrogen removal when using shellfish. Shellfish enhancement can remove nitrogen from the water column in two ways: 1) removal of nitrogen incorporated in the shell and tissues during harvesting and 2) the action of denitrifying bacteria on shellfish biodeposits (pseudofeces and feces). Golen (2009) notes that while it is relatively easy to account for nitrogen removed in shell and flesh based upon annual harvest levels, the factors that govern the magnitude of nitrogen removal via denitrification are too complex and variable to allow the use of fixed removal rates that can be applied across all shellfish aquaculture facilities. Nitrogen trading using shellfish could proceed based solely upon the quantification of nitrogen removal via biomass at harvesting, but this could substantially underestimate the nutrient reduction value of shellfish enhancement: nitrogen removal by oysters via nutrient burial and denitrification of biodeposits is estimated to be seven times the weight of nitrogen contained in the shellfish meat (Stephenson and Brown (2006); cited in Golen (2009).

Stadmark and Conley (2011) suggest that the primary focus for nutrient mitigation should be on nutrient reductions from land-based sources before nutrients enter into the coastal zone. However, Rose et al. (2012) argue that this view may be somewhat simplistic. Rose et al. (2012) agree that nutrient mitigation should not be used in lieu of land-based nutrient reductions, but they note that as conventional treatment methods approach limits of technology (wastewater treatment plant upgrades) or become very expensive (urban stormwater retrofits), many estuaries will still need to find additional nutrient removal methods to meet water quality objectives. Lindahl and Kollberg (2009) support shellfish enhancement as a nutrient compensation measure but argue that it should be reserved for diffuse (non-point source) emissions into the coastal zone coming from, for example, agriculture, because for these there are few other effective options available. Rose et al. (2012) also note that nutrient removal by shellfish becomes more important as the main nutrient loading from land increasingly shifts to diffuse sources. According to the authors, as the majority of point source contributions are reduced, the law of diminishing returns makes it far more expensive to reduce the remainder of the nitrogen load to coastal ecosystems, largely attributable to agricultural and diffuse urban sources.

While shellfish enhancement projects have yet to become active participants in a US nutrient trading program, a number of studies have examined the credit generation potential of these projects. Stephenson et al. (2010) describe a bioeconomic model used to estimate the cost of providing a nitrogen offset from commercial oyster aquaculture. The model estimated cost by calculating the supplemental compensation necessary for a commercial aquaculture operation to produce additional oysters. The study calculated total nitrogen removed from ambient waters based on the amount of nitrogen sequestered in oyster tissue and shell at harvest and nitrogen removed through denitrification of oyster biodeposits. The estimated annual cost to generate one pound of nitrogen offset ranged from $0 to $150, depending on assumptions about the market price of harvested oysters, project input costs, and oyster growth and mortality rates. According to the authors, conceptually, costs may be zero because in some economic circumstances oyster growers would be willing to expand production without any compensation for nutrient removal services. Estimated total nitrogen removed for every one million market-sized oysters harvested ranged between 290 and 815 lb yr^{-1} (132 and 370 kg yr^{-1}) depending on assumptions of denitrification rates.

Piehler and Smyth (2011) compared the ecosystem service value of nitrogen removal in five of the major habitat types found in temperate estuaries (salt marsh, submerged aquatic vegetation, oyster reef, intertidal flat and subtidal flat). The authors estimated a dollar value of nitrogen removal using the rates from the North Carolina nutrient offset program. Under this program, the nutrient offset payment price is not set by the market but rather is a regionally derived number that had significant stakeholder input in its determination. The trading price of the North Carolina nutrient offset program at the time of the study was $13 per kg ($5.90 per lb) of nitrogen removed. To best estimate the annual value of habitat-specific nitrogen removal, the authors multiplied this price by the mean annual rates of denitrification and the standard error of these means. The authors estimated that submerged aquatic vegetation and oyster reefs provide about $3,000 per acre per year in nitrogen removal services, while wetlands provide about $2,500 per acre per year of nitrogen removal service.

Outside of the US one can find examples of shellfish enhancement projects being incorporated into nutrient trading schemes. For instance, Lindahl and Kollberg (2009) described what they called the first case in Sweden of trading a nitrogen emission. The Swedish community of Lysekil was permitted, as a trial between 2005 and 2011, to continue to emit nitrogen from its wastewater treatment plant into an estuary provided that the same amount of nitrogen was ‘harvested’ and brought ashore by 3,900 metric tons of farmed blue mussels. According to the authors, the payment of €150,000 ($168,000) that Lysekil made to the mussel farming enterprise contracted for the nitrogen removal was far below the cost for nitrogen removal in the local wastewater treatment plant.

Here we explore the potential for shellfish aquaculture to be included in the generation of non-point source credits or offsets in the CT Nitrogen Credit Exchange. A recent analysis of nutrient removal capabilities in shellfish farms by Rose et al. (2014a) showed that removal efficiencies were comparable to agricultural and urban best management practices (BMPs). They showed additionally that the production of non-point nitrogen source credits by shellfish compares favorably to agricultural non-point nutrient management strategies and is more cost effective than urban stormwater strategies and wastewater treatment upgrades. Inclusion of shellfish aquaculture in a nutrient trading program seems desirable based on removal efficiencies and cost effectiveness in comparison to these other non-point source strategies, though the difficulties in measurement and certification of amounts removed by tissue and / or denitrification are noted.

Economic Valuation of Ecosystem Services

To determine the total economic value of an activity such as shellfish bioextraction of nitrogen in LIS and GBP, economists consider consumer and producer surplus. Consumer surplus is the net value consumers receive from a good or service over and above what they actually pay for the good or service. Producer surplus is the difference between what producers actually receive when selling a product and the amount they would be willing to accept for the product. While not an exact measure of social welfare, the sum of the consumer and producer surplus that results from the ecosystem services provided by a shellfish bioextraction provides a useful approximation of net benefits.

Converting the benefits of ecosystem services to a common comparable unit (dollars) so as to sum them represents a major challenge to economists (Peterson and Lipcius, 2003). While economists possess an array of methods to assess consumer and producer surplus in monetary terms, no single method can capture the total value of the many disparate ecosystem services provided by a complex natural asset such as an oyster bed or reef (Johnston et al., 2002). Moreover, although the value of some services can be readily monetized, the value of others can only be monetized with great difficulty and uncertainty (Johnston et al., 2002). For example, estimation of consumer surplus is relatively straightforward if services are traded in traditional markets with market prices and values (e.g., commercial shellfish, shell for road building). However, many of the benefits of the ecosystem services provided by shellfish enhancement (e.g. water filtering and enhanced water quality, habitat for other organisms) accrue directly to people without passing through the market economy.

Methods / Approaches to determining ecosystem service value

There are three general types of approaches for estimating economic value of ecosystem services, each with strengths and weaknesses. Each service has an appropriate set of valuation methods and some services may require that several methods be used jointly (Farber et al., 2002; Carson and Bergstrom, 2003). The first approach, to conduct primary research, can be subdivided into indirect (revealed preference) and direct (stated preference) methods and requires the collection of new data, which may be costly and time-consuming (Dumas et al., 2005). Consequently, some researchers have adopted the second approach, commonly called benefit transfer (Smith et al, 1999), whereby researchers estimate the value of an ecosystem service using existing valuation information for a similar ecosystem service. The third includes cost-based methods (serving as a proxy for value), which is the method used in this study.

Cost-based methods such as the avoided cost, replacement cost, and substitute cost are related methods that estimate values of ecosystem services based on either the costs of avoiding damages due to lost services, the cost of replacing ecosystem services, or the cost of providing substitute services (King and Mazzotta, 2000). These methods do not provide strict measures of the economic value of ecosystem services, but rather provide approximate indicators of the value by assuming that, if people incur costs to avoid damages caused by lost ecosystem services, or to replace the services of ecosystems, then those services must be worth at least what people paid to protect or replace them. For example, the replacement cost method might identify a project that provides the same services as the lost ecosystem services and calculate the cost of construction for that project (Carson and Bergstrom, 2003).

King and Mazzotta (2000) note that the key assumption of cost-based methods may not necessarily be valid. The elimination of an ecosystem service is no guarantee that the public would be willing to pay for the identified least-cost alternative merely because it would supply the same benefit level as that service. King and Mazzotta (2000) conclude that cost-based methods are most appropriately applied in cases where damage avoidance or replacement expenditures have actually been, or will actually be, made.

The first step of a cost-based approach in the valuation of shellfish removal of nutrients is to quantify the removal capacity of shellfish bioextraction. This assessment determines the current level of water quality and the expected level of water quality after shellfish harvest. The EcoWin model was applied in this study to assess the nitrogen reduction potential of bioextraction in our case study sites (see section below: 'EcoWin.NET').

The next step is to estimate the costs of providing a substitute or replacement for the nutrient removal through biotechnology. The dollar value of providing replacement nutrient reduction services by, for example, a wastewater treatment plant, can be compared to shellfish aquaculture or restoration costs to determine whether biotechnology is worthwhile in terms of water quality improvements. This comparison may be presented in terms of per-unit nutrient reduction costs (Lindahl et al., 2005; Rose et al., 2014). The replacement value is calculated as the savings in costs from using bioextraction instead of more costly nutrient abatement options.

Applications of Cost-based Methods

Several studies have used cost-based methods to estimate the water quality improvement benefits of shellfish production and restoration, although not all of the studies follow the above methodological steps. In an early application of the cost-based method, Newell et al. (2005) used EPA estimates of the cost of reducing nutrient inputs necessary for meeting the water quality goals in the Chesapeake Bay. After converting this cost figure to an average per kilogram (kg) cost of removing nitrogen, these authors used modeled nutrient reduction estimates to derive an annual replacement value of nitrogen removal by oyster reefs in the Choptank River, Maryland, of $314,836, or $181 per hectare ($73.30 per acre). The authors postulated that over the ten-year life span of the oysters, the value of the nitrogen removal by the oyster reefs increased to $3.1 million, or over twice the dockside value of those same oysters.

Ferreira et al. (2011) estimated that in a 11.4 hectare Manila clam farm on the southern coast of Portugal, shellfish filtration provides a net nitrogen removal of about 28,867 kg (63,641 lb) per year. The authors estimated the replacement value of the clam farm to be about €0.26 million ($0.33 million) per year, about 10 percent of the direct income of the farm.

Using production and nutrient removal data to calculate the role of all European Union (EU) shellfish farms in removing nutrients, Ferreira et al. (2009) estimated that the farms removed a total of over 55,000 metric tons of nitrogen per year, that is, a PEQ of 17 million people, or the population of the Netherlands. The estimated replacement value is estimated to be €0.4 billion ($0.5 billion) per year. Based on the worldwide reported shellfish aquaculture landings and modeling results of nitrogen removal for a typical range of cultivated species, the authors estimated a global replacement value of €7.5 billion ($9.4 billion).

Gren et al. (2009) estimated the replacement value of nutrient cleaning by mussel cultures in the Baltic Sea using a nonlinear programming model that compares costs and impacts of mussel farms with other nutrient abatement measures such as wastewater treatment plants, changes in land use and fertilizer practices, increased cleaning by households and industries not connected to municipal wastewater treatment, and creation of wetlands. The results indicate that calculated marginal cleaning costs of nutrients by mussel farming can be considerably lower than other abatement measures, with cost savings between 2 and 11 percent. The authors note that the cost-effectiveness of mussel culture in nutrient removal from the water column depended on mussel growth, sales options, assumptions about mussel farming capacity and the nutrient load targets.

Stadmark and Conley (2011) reported that compared to other potential nutrient abatement measures, mussel farming is a relatively expensive measure. Their calculations show that nitrogen reduction in mussel farms (industry mussels) is about two times more expensive per kg of nitrogen than nitrogen reduction from wastewater treatment plants. However, Rose et al. (2012) point out that in estimating the cost of nitrogen removal, Stadmark and Conley (2011) did not count the revenue from the sale of shellfish biomass as an offset to costs. Rose et al. (2012) note that other studies (e.g., Gren et al., 2009; Miller, 2009) have shown that the cost of nitrogen removal from shellfish production is highly contingent upon the returns that could be achieved through shellfish biomass (either for animal or human consumption). In addition, Rose et al. (2012) note that the costs quoted by Stadmark and Conley (2011) fail to reflect the wide range of costs (and cost uncertainty) associated with nitrogen removal, and that they substantially understate the upper bound limits of nitrogen removal costs from wastewater treatment plants. According to Rose et al., (2012) extensive evidence elsewhere demonstrates that these costs can increase rapidly as treatment approaches limits of technology and as smaller plants add tertiary nitrogen removal processes.

Data: Costs of Abatement Technologies and Best Management Practices

Cost information based on increasing the effectiveness of WWTPs and furthering the adoption of agricultural best management practices (BMPs) and non-point urban controls derived from Evans (2008) and Kessler (2010) was used for LIS and GBP, respectively. These two studies were originally conducted in order to assess the cost-effectiveness of WWTP upgrades as compared to alternative nitrogen removal options (including agricultural and urban BMPs).

Long Island Sound Study

Evans (2008) evaluated potential load reductions that might be achieved via the implementation of both point and non-point nitrogen source controls throughout the CT River Basin[7]. A primary objective of this study was to estimate costs associated with achieving incremental reductions in nitrogen loading that could be subsequently used to develop point and non-point 'cost curves' that could depict varying levels of nitrogen reduction versus money spent in implementing reduction strategies.

Point Source Controls: Evans (2008) used estimates of potential nitrogen reductions that could be achieved by bringing all WWTPs to three specific 'levels' or target concentrations of nitrogen in the effluent (i.e., 8 mg/l, 5 mg/l, and 3 mg/l). The study reported the costs and nitrogen reductions incrementally from their current effluent concentration levels to 8 mg/l, then from 8 to 5mg/l, and finally from 5 to 3mg/l. Costs associated with upgrading treatment plants for each of the treatment levels were calculated using an approach developed as part of a study to estimate the cost of plant upgrades in the Chesapeake Bay Basin (Chesapeake Bay Program, 2002). We adjusted the cost estimates to 2013 dollars using the Engineering News-Record Construction Cost index (ENRCC) index to account for inflation.

[7]*Note that WWTPs are a point source of nitrogen as the nitrogen enters the water system at a single point, the effluent pipe, and fertilizers are a non-point source of nitrogen as the nitrogen enters the water system via runoff.*

The average per unit costs in 2013 dollars are \$14.63/lb (8 mg/l level)[8], \$16.82/lb (5mg/l level), and \$44.81/lb (3mg/l level). shows the total capital cost, annual O&M costs, the combined annualized capital cost (20 year depreciation), and average per unit costs per pound of nitrogen removed at the various nitrogen effluent limits.

Table 17: Incremental Costs and Reductions from Point Source Controls (2013 $)

	Capital Cost ($ million)	O&M ($ million)	Annualized Cost ($ million)	Nitrogen Removed (lb yr^{-1})	Avg. Cost* ($ lb^{-1} yr^{-1})
8 mg/l	433.59	8.67	30.35	2,074,788	14.63
5 mg/l	143.34	4.22	11.38	676,682	16.82
3 mg/l	316.39	12.61	28.43	634,471	44.81

**note that the CT Nitrogen Credit Program is paying \$5.01 lb^{-1}*
Source: Evans, 2008 with Northern Economics, Inc. analysis

Controls in Agricultural Areas: Evans (2008) also estimated potential nitrogen reductions and annual costs for agricultural controls that include the use of nutrient management (riparian buffers) and cover crops. The estimated average annual cost per acre reported by Evans (2008) for agricultural controls, adjusted for inflation using the ENRCC index, is \$38.92. The report estimated that the use of such controls would result in a maximum nitrogen load reduction of 1,313,636 lb/yr for the entire CT River Basin at an estimated adjusted annual cost of \$7.68 million. Given a current estimated agricultural load of 3,886,961 lb/yr this would equate to a maximum potential reduction of about 34.1 percent at a unit cost of about \$5.90 lb/yr (Table 18).

Table 18: Costs of Agricultural BMP Implementation (2013$)

Annualized Cost ($ yr^{-1})	Nitrogen Removed (lb yr^{-1})	Avg. Cost ($ lb^{-1} yr^{-1})
7,680,000	1,313,636	5.90

Source: Evans, 2008 with Northern Economics, Inc. analysis

Controls in Urban Areas: In order to assess reductions in urban areas Evans (2008) evaluated a representative urban BMP to estimate potential load reductions throughout the basin. The two most cost-effective BMPS are wet ponds and submerged gravel wetlands. The average per acre-drained construction costs for these BMPS are \$7,000 and \$11,000 respectively and the associated nitrogen removal is 55 and 85 percent respectively. In addition to actual construction costs, the cost of land acquisition was also considered. The total costs for full urban BMP implementation within all of the sub-basins were estimated to be \$3,261,709,912 in 2013 dollars. To allow for better comparison to other mitigation measures Evans divided this total cost by 20 years (similar to point source costs) to derive an estimated annual cost of \$163,085,496. Evans estimated the maximum nitrogen reduction that might be obtained via full imple-

[8]Evans (2008) estimated potential point source reductions for treatment plants in the upper part of the Connecticut River Basin so as to provide states in this area with sufficient information to adequately frame their options in terms of point source versus non-point sources controls.

mentation of urban BMPS in the entire Connecticut River basin to be about 1,035,150 lb/yr. The per-unit cost for urban controls is approximately $159 per pound of nitrogen removed as shown in Table 19.

Table 19: Costs of Full Urban BMP Implementation (2013$)

Annualized Cost ($ yr^{-1})	Nitrogen Removed (lb yr^{-1})	Avg. Cost ($ lb^{-1} yr^{-1})
163,085,496	1,035,150	159

Source: Evans, 2008 with Northern Economics, Inc. analysis

Great Bay Estuary Study

Point Source Controls: Kessler (2010) assessed the Capital plus Operation and Maintenance (O&M) costs for removing varying amounts of nitrogen at 18 municipal WWTP currently discharging into GBP. O&M costs were primarily based on 2009 operating budgets from WWTP operators. Additionally, existing operating budgets included estimated costs associated with more advanced treatment levels for nitrogen removal based on an EPA publication. The nitrogen effluent limits that were considered were 8 milligrams per liter (mg/l), 5mg/l, and 3mg/l[9]. Different approaches were used for estimating cost depending on the degree of planning that municipalities had already invested toward achieving more advanced treatment levels. The cost of nitrogen removed due to WWTP upgrades was calculated as the annualized cost divided by the pounds (lb) of nitrogen removed from (or not entering)[10] the Great Bay, Little Bay, and Upper Piscataqua. Amortized capital costs plus annual O&M costs were combined to estimate the total annual costs for each treatment level for nitrogen removal. A range of interest rates from 2 to 5 percent were selected to bracket the potential rates for a 20 year bond.

Kessler (2010) estimates the cost per pound of nitrogen removed from the estuary by improving each of the WWTPs in the watershed to 8mg/l, 5mg/l, and 3mg/l nitrogen effluent limits[11]. The study team adjusted Kessler's cost adjustments for inflation (Table 20), using the Engineering News-Record Construction Cost index (ENRCC). Overall, Kessler reports if all the WWTPs were given nitrogen effluent limits of 8 mg/l at design flow, a total of 116.8 tons of nitrogen per year would not enter GBP at an adjusted annualized cost of $16.13–$20.23 million with an average per unit cost of nitrogen removed of $77/lb. For the 5 milligrams of nitrogen per liter (mg N/L) nitrogen limit, Kessler estimates that 215.2 tons of nitrogen would be removed for an adjusted annualized cost of $26.13–$31.69 million with an average per unit cost of nitrogen removed of $68/lb. Finally, for the 3 mg N/L nitrogen limit, Kessler reports 280.8 tons of nitrogen could be removed for an adjusted annualized cost of $35.39–$42.76 million, and an average per unit cost of nitrogen removed of $70/lb. Kessler suggests that the costs per pound of nitrogen removed are

[9]The first two thresholds were calculated such that total nitrogen concentration in the sub estuary would be equal to the numeric criteria for the prevention of low dissolved oxygen (0.45 mg N/L) and for protecting eelgrass habitat (0.30 mg N/L). The third threshold was only calculated for the tidal river sub estuaries. The purpose of this third threshold was to establish watershed nitrogen loading limits that would protect eelgrass habitat in the downstream sub estuaries of Great Bay, Little Bay and Upper Piscataqua River. This third threshold can be described as a "downstream protective value". In general, the thresholds for preventing occurrences of low dissolved oxygen and protecting eelgrass in downstream areas were higher than the thresholds for protecting eelgrass habitat in tidal river sub estuaries.
[10]Wastewater treatment facilities remove nitrogen from wastewaters thus abating nitrogen delivery into the water body.
[11]These costs are incremental and assume that all WWTP move to 5mg/l once all have achieved 8 mg/l.

likely overestimates because the nitrogen load reductions at most WWTPs are partially cancelled out by several WWTPs for which the nitrogen load would increase under the permitting scenarios.

Table 20: Annual Costs of WWTF Upgrades at Various Levels of Nitrogen Removal (2013 $)

Level	Capital Cost ($ millions)	O&M ($ millions)	Annualized Cost ($ millions)	Nitrogen Removed (lb/year)	Avg. Cost ($ lb^{-1} yr^{-1})
8 mg/l	215.12	23.55	16.13-20.23	233,600	77
5 mg/l	291.29	28.90	26.13-31.69	430,400	68
3 mg/l	385.66	32.40	35.39-42.76	561,600	70

Source: Kessler, 2010 with Northern Economics, Inc. analysis

Controls in Agricultural Areas: For non-point source offsets, Kessler used estimates from Stevenson et al. (2010) outlined in Table 21. The estimates from Stephenson et al. (2010) are also adjusted for inflation using the ENRCC index. Estimates provided by Stephenson et al. (2010) are from an analysis done in Virginia on the nitrogen load entering Chesapeake Bay which compare favorably to costs estimated by other studies in the US and EU (Table 21), but there is no indication of how they would apply in GBP. Unfortunately, contributions of these sources to the existing GBP nitrogen load are not currently known.

Table 21: Annualized Costs per Pound of Non-Point Source Offsets (2013 $)

Nitrogen Removal Method Compilation for several EU and US locations+	Cost ($ lb^{-1})
Shellfish	5.7 – 150
Agricultural	0.1-470
Urban Stormwater Treatment	30-3629
Wetlands	0.6-214
For Chesapeake Bay only* Agriculture	9–512
Stormwater Treatment	59–2,414
Connecting Homes on Septic to WWTP	33–610

Source: Rose et al., 2014a+ compilation of studies in VA, NH, Denmark and 9 Baltic countries. Rose et al., 2014a includes
**Stevenson et al., 2010 with Northern Economics, Inc. analysis*

RESULTS AND DISCUSSION

Long Island Sound

System Wide Eutrophication Model (SWEM)

Interfacing Achieved between SWEM and EcoWin

A key project result was the successful interface of the SWEM and EcoWin models for LIS. The interfacing was achieved by developing a computational grid for EcoWin that represented an aggregation of smaller SWEM grid computational elements and also captured designated shellfishing waters. SWEM and EcoWin computational grids and shellfishing designations are shown on Figure 15.

Another important outcome of this study was the development of spreadsheets for passing water volume fluxes calculated by SWEM's hydrodynamic model to EcoWin. Nutrient loadings developed previously for SWEM and nutrient concentrations calculated by SWEM's eutrophication model for four likely future nutrient loading conditions were also passed to EcoWin through spreadsheets. The SWEM related information contained in the spreadsheets might be useful to LIS managers and researchers for purposes other than the EcoWin application in this study. Having this SWEM input and output information for LIS extracted in spreadsheets, albeit in a spatially aggregated format mapped to EcoWin, from large, user-unfriendly binary and ascii files used and generated by SWEM, is in and of itself a meaningful result for potentially facilitating further LIS investigations.

Conditions Provided by SWEM to EcoWin and Regulatory Implications

Per SWEM calculations presented in HydroQual, (2007), the 2000 TMDLs, which targeted cost-effective nitrogen removal actions and dissolved oxygen improvements (NYSDEC and CTDEP, 2000), are expected to benefit LIS ecological resources in terms of reducing the 'volume days' occurring in the Sound of DO levels that would produce organism mortality and biomass reduction. Based on SWEM results for the 2000 TMDL, however, it is unlikely that enforceable water quality standards for DO in LIS will be fully achieved by the mandated nitrogen TMDLs allocations, even if credit is taken for associated, but not mandated, reductions to carbon loadings.

Accordingly, from a Clean Water Act regulatory perspective, managed shellfish aquaculture in LIS could operate as a nutrient removal mechanism, either reducing or eliminating the need for later TMDL phases. EcoWin results address whether or not bio-extractive shellfish aquaculture technologies can be successful in further nutrient and dissolved oxygen management in LIS.

SWEM results for the TMDL loading conditions shared with EcoWin are presented in HydroQual (2007, 2009) and Landeck Miller and Wands (2009). Figure 41 to Figure 44, adopted from Landeck Miller and Wands (2009), include map displays of the minimum DO and maximum total nitrogen results calculated by SWEM for TMDL loading conditions.

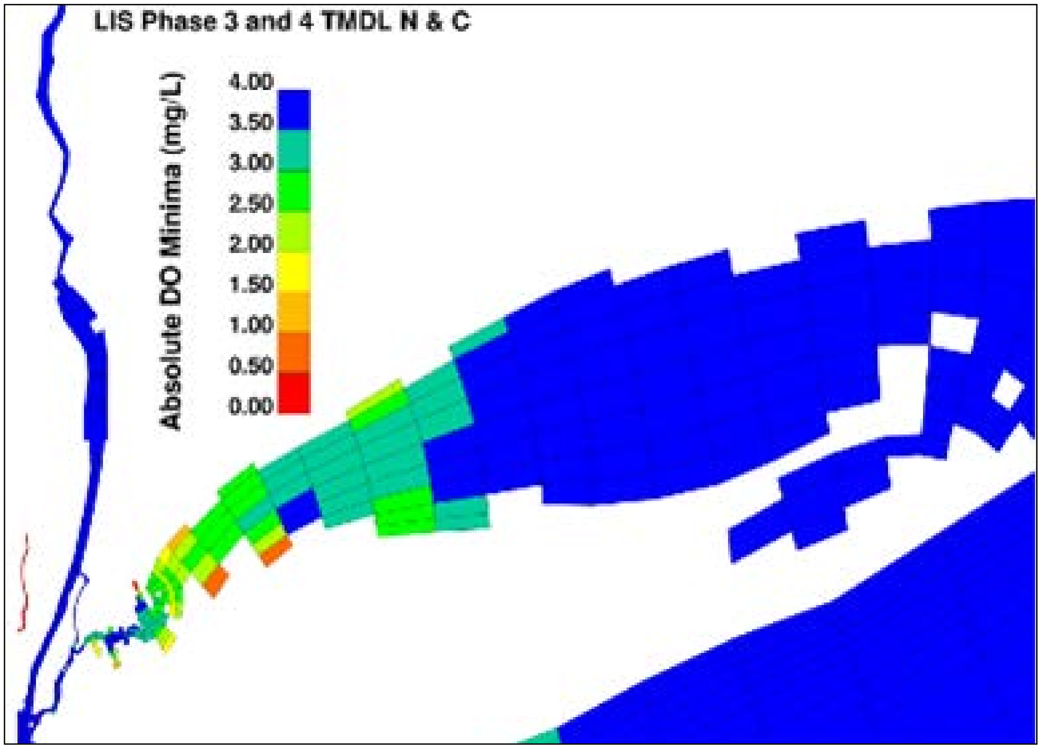


Figure 41: SWEM calculated minimum dissolved oxygen for 1988 hydrodynamics with full implementation of the 2000 TMDLs nitrogen loading reductions and associated carbon reductions.

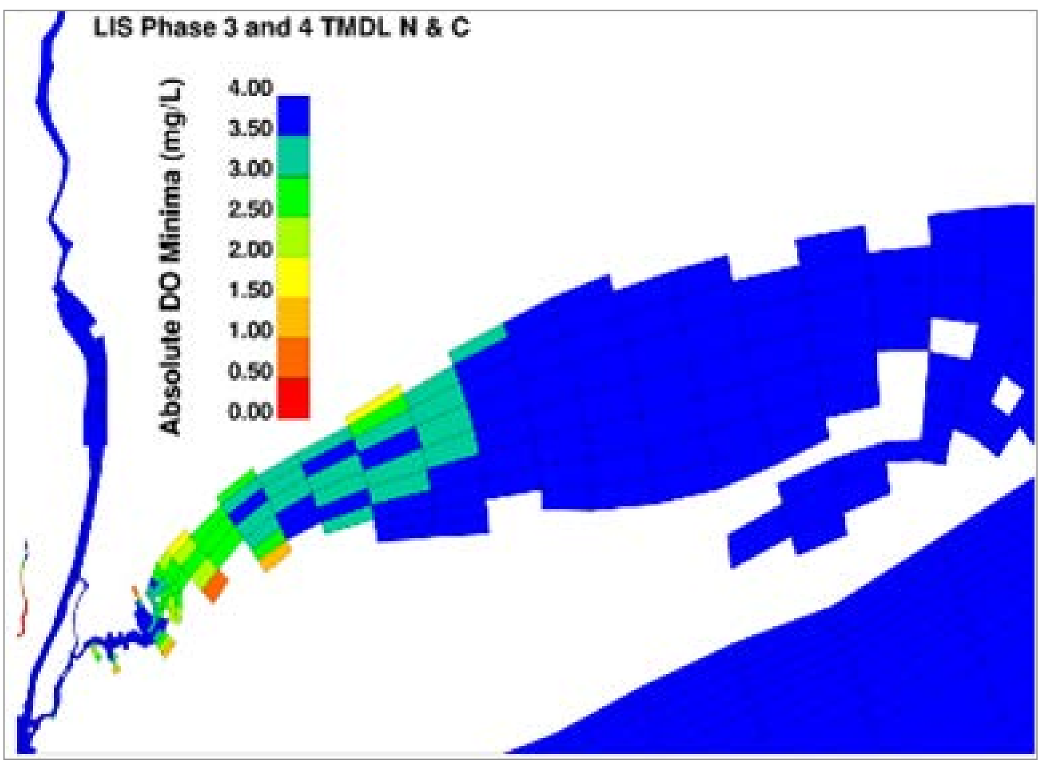


Figure 42: SWEM calculated minimum dissolved oxygen for 1989 hydrodynamics with full implementation of the 2000 TMDLs nitrogen loading reductions and associated carbon reductions.

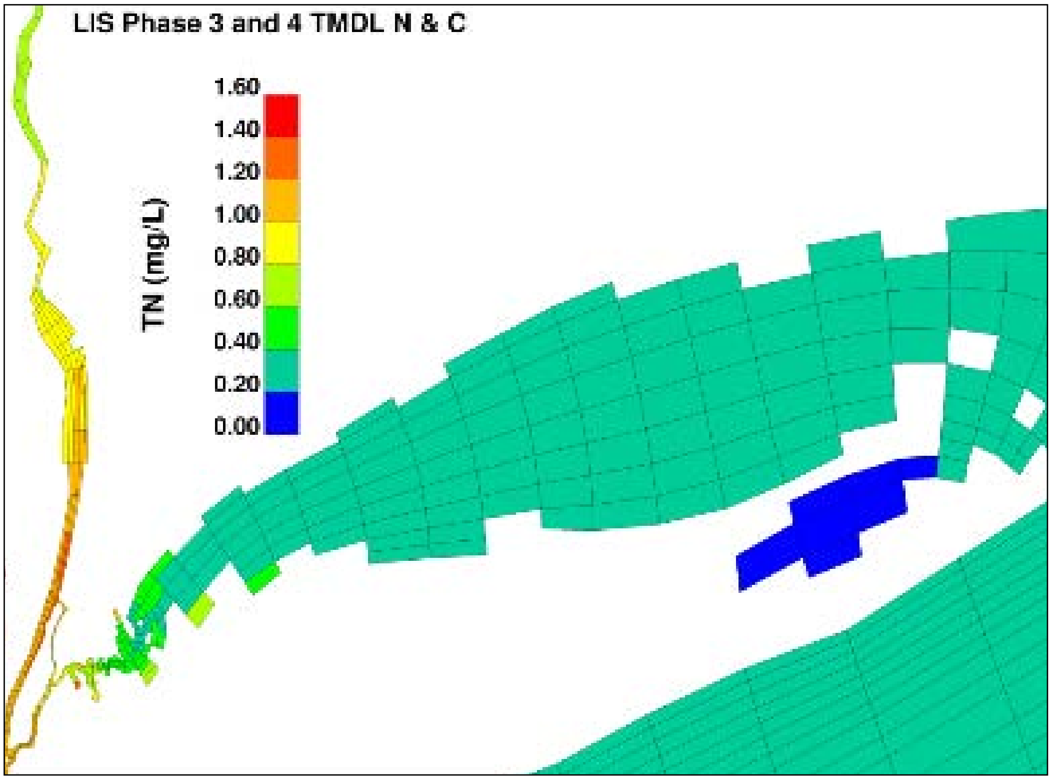


Figure 43: SWEM calculated total nitrogen associated with minimum dissolved oxygen for 1988 hydrodynamics with full implementation of the 2000 TMDL nitrogen loading reductions and associated carbon reductions.

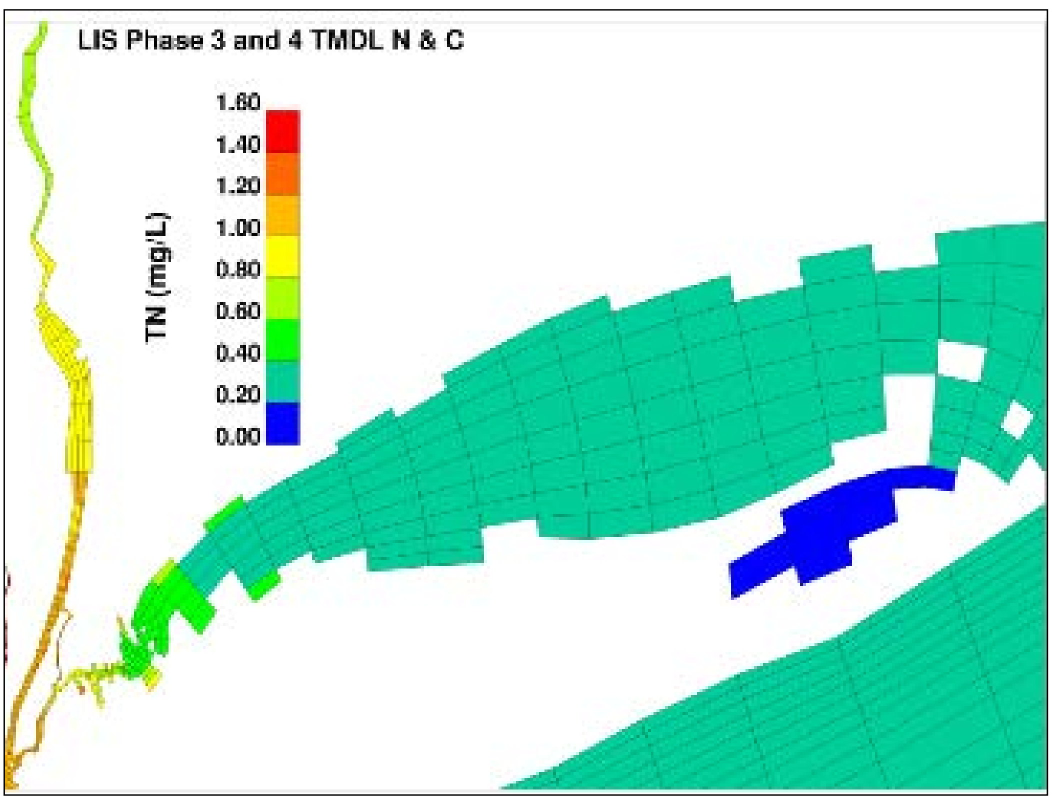


Figure 44: SWEM calculated total nitrogen associated with minimum dissolved oxygen for 1989 hydrodynamics with full implementation of the TMDL nitrogen loading reductions and associated carbon reductions.

EcoWin.NET

The general pattern of results for the conservative variables in the EcoWin model (water volume, water temperature, salinity), together with some of the key water quality variables, were shown in the previous section (see section 'Methods and Supporting Experiments and Data'), because the outputs are part of the process of model validation. The results shown here deal with the observed patterns for shellfish production and harvest, where the focus is primarily on the Eastern oyster, because in this case the individual growth, population dynamics, and interactions with the human elements of the activity (planting and harvesting), were explicitly simulated.

Oyster harvest

The harvest yield obtained in the model, over a ten year run, is shown in Table 22. There are interannual fluctuations due to slight variations in water volumes in consecutive years. EcoWin uses an output series of water fluxes from SWEM superimposed on a 365 day cycle. Although a flux correction was introduced to compensate for the fact that the calendar year does not have an integer number of spring-neap cycles, there are always minor differences observed in long model runs. This is not considered a problem, particularly in the light of natural year-to-year fluctuations, which this modeling scheme does not consider—these are in any case, by definition, unpredictable by any model.

Table 22: Annual oyster harvest in Long Island Sound, simulated with EcoWin.NET, for the last 4 years of a 10 year model run. The oyster growth period simulated is 977 days per cycle.

Year[12]	Box 23 (tonnes)	Box 25 (tonnes)	Box 27 (tonnes)	Box 30 (tonnes)	Box 33 (tonnes)	Box 41 (tonnes)	Total (tonnes)
7	636	24623	311	4574	309	1235	31688
8	607	20945	258	3745	239	845	26638
9	631	24319	306	4486	303	1224	31268
10	607	20912	257	3714	236	838	26565
Mean	620	22700	283	4130	272	1035	29040
Standard deviation	15.4	2049.1	29.6	463.6	39.7	224.0	
Coef. of variation (%)	2.5	9.0	10.5	11.2	14.6	21.6	

The modeled harvest varies between about 26.5 kt and 31.5 kt per year, with an average of 29 kt. Box 25 has by far the largest harvest, but this still occurs (Table 23) at a very attractive average physical product (APP) of 45 (1 lb input converts to 45 lb output).

[12]*Ordinal year, e.g. Year 7 means the 7th year, the period between years 6 and 7*

Table 23: Mean annual oyster harvest in Long Island Sound, seeding (considering only 1/3 of the stocking area is seeded per year), and average physical product (output/input), an indicator of return on investment.

	Box 23	Box 25	Box 27	Box 30	Box 33	Box 41	Total
Harvest (t y^{-1})	620	22700	283	4130	272	1035	29040
Seed (t y^{-1})	13	507	7	99	7	26	659
APP	47	45	43	42	41	39	44

The simulated pattern of oyster harvest over a ten year period is shown in Figure 45, and illustrates the spin-up period in the first 3 to 4 years, followed by a stable limit cycle with alternating years of higher and lower harvest, as described above. Seeding takes place annually from the first year, but the first harvest year is Year 3, due to the time taken to grow the first crop. The harvest is represented cumulatively throughout the year, and then restarts the following year at zero. The marked difference between Box 25 and the other boxes led to the choice of a different scale in order to represent the whole Sound in one graph.

Year nine of the model run was chosen for a mass balance analysis of cultivation. In all mass balance outputs, calculated values represent the role of all the cultivated animals, as opposed to the harvested biomass. This key difference merits more detailed discussion.

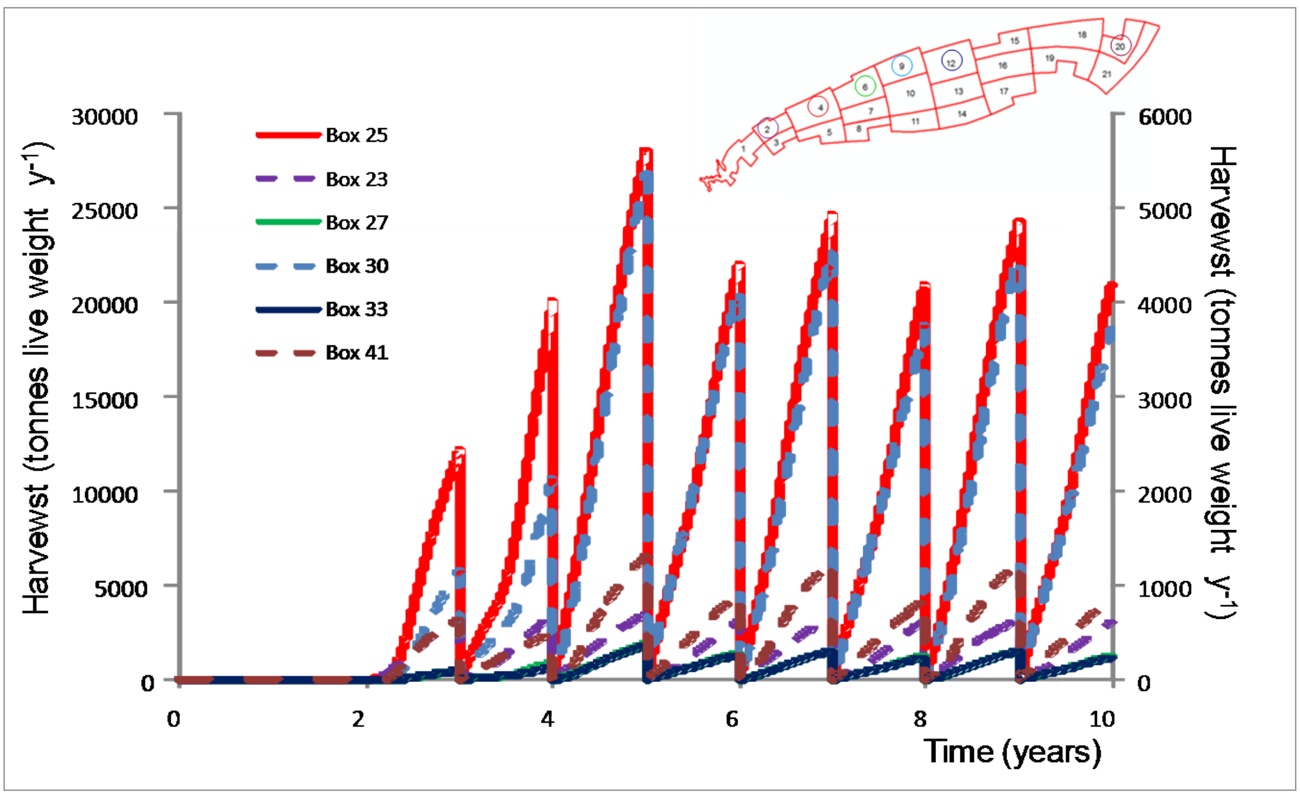


Figure 45: Simulated harvest in EcoWIn.NET over a ten year period. Box 25 is on the left scale, all others on the right.

Table 24. Harvested oyster biomass (live weight), and the role of oysters in water clearance in Long Island Sound.

Year 9	Box 23	Box 25	Box 27	Box 30	Box 33	Box 41	Totals
Tons harvested	**630.72**	**24318.94**	**305.91**	**4486.10**	**303.22**	**1223.56**	**31268.46**
Clearance (m3 y^{-1})	159696072	6649705181	93793687	1492544845	105844035	463171322	8964755143
Box volume (m^3)	766535810	1795089300	1346708900	1032006100	2134304300	2362068500	9436712910
Clearance/ Volume (% y^{-1})	20.8	370.4	7.0	144.6	5.0	19.6	95.0

It is common to determine the role (for instance in nitrogen removal) of shellfish, whether cultivated, or in restored reefs, by considering the harvested or collected biomass, and applying a conversion factor (usually about 1%; e.g. Higgins et al, 2011) for the nitrogen component. In other words, to consider only the fraction of shellfish physically taken out of the water. The difficulty in this approach is obvious when we consider an oyster or mussel reef, particularly in a pristine natural system with little or no human intervention, where the gathering of shellfish is either nonexistent or insignificant at the scale of the reef. Under such conditions, shellfish remove both phytoplankton and organic detritus from the water column through filtration, and return a part of that as particulate (uningested pseudofeces, and undigested feces) and dissolved (excreted) materials (see Figure 19). The rest is used for somatic and gonad growth, and for shell formation—given shellfish are organic extractors, the net removal is obviously positive.

One of the key ecosystem services supplied by shellfish reefs and aquaculture alike is the removal of primary symptoms of eutrophication (*sensu* Bricker et al, 2003), converting a significant part of both phytoplankton, and detrital organics, into tissue and (to a small extent) shell. Although the shellfish that are not cultivated, or that form a reef, remain in the water, they still contribute to the reduction of eutrophication symptoms. This reduction promotes increased water clarity, and the combined effect of shellfish, *while in the water*, is to short-circuit the eutrophication process, greatly reducing secondary symptoms of eutrophication such as hypoxia (through lack of detrital organic supply) and loss of submerged aquatic vegetation (through improved underwater light climate). In other words, nutrient bioextraction by means of shellfish aquaculture or reef restoration is not the physical removal of nutrients from an aquatic ecosystem, but the nutrient bioextraction from a particular component of that ecosystem, in order to modify the processes or rates within that ecosystem.

Whereas for the shellfish harvest approach referred above, there is no need to apply a complex ecosystem model such as EcoWin, the evaluation of the true eutrophication-related ecosystem service provided by shellfish aquaculture or reefs does require an approach that takes into account the full complement of sources and sinks of carbon, nitrogen, and phosphorus related to bivalve physiology. The AquaShell individual model described in the Methods-Individual Model section (see Figure 19) and elsewhere (e.g. Silva et al., 2011) incorporates all the required processes, and EcoWin builds those into a population model, which also includes the losses (i.e. nutrient sinks) from both spawning and mortality.

Table 25: Role of oysters in the nutrient mass balance of Long Island Sound, determined from EcoWin.NET mass balance.

Year 9	Box 23	Box 25	Box 27	Box 30	Box 33	Box 41	Totals
Metric tons harvested (t y^{-1})	630.72	24318.94	305.91	4486.10	303.22	1223.56	31268.46
Total phyto uptake (kg C y^{-1})	96269.49	3151479.79	38520.63	559622.33	36612.82	143135.11	4025640.17
Total POM uptake (kg C y^{-1})[13]	157673.40	5441272.50	72771.11	1177455.29	93129.22	533546.88	7475848.39
Total POM release (kg C y^{-1})	49955.85	1775192.31	24829.36	420840.95	35580.76	223832.91	2530232.14
Total POM mortality (kg C y^{-1})	872.93	35927.03	488.23	7498.47	507.24	2093.29	47387.19
Total DIN excretion (kg N y^{-1})	2269.02	80799.62	1040.41	15705.66	1075.84	4679.36	105569.92
Total phyto uptake (kg N y^{-1})	14975.25	490230.19	5992.10	87052.36	5695.33	22265.46	626210.69
Total POM uptake (kg N y^{-1})	24526.97	846420.17	11319.95	183159.71	14486.77	82996.18	1162909.75
Total POM release (kg N y^{-1})	7770.91	276141.03	3862.35	65464.15	5534.79	34818.45	393591.67
Total POM mortality (kg N y^{-1})	135.79	5588.65	75.95	1166.43	78.90	325.62	7371.34
Total N intake (kg N y^{-1})	24526.97	846420.17	11319.95	183159.71	14486.77	82996.18	1162909.75
Total N loss (kg N y^{-1})	10175.72	362529.30	4978.71	82336.24	6689.53	39823.44	506532.93
Net N removal (kg N y^{-1})	14351.25	483890.87	6341.24	100823.48	7797.24	43172.74	656376.82
Population-Equivalents (PEQ y^{-1})	4349	146634	1922	30553	2363	13083	198902
Nitrogen removal (%)	2.28	1.99	2.07	2.25	2.57	3.53	2.10

The results shown in Table 24 and Table 25 thus consider one year (Year 9) for the mass balance calculations, but these integrate the eutrophication-related ecosystem service provided by: (1) the Year 3 cohort, much of which will be physically removed (harvested) from the Sound; (2) the Year 2 cohort, which will be harvestable only the following year; and (3) the Year 1 cohort, which will take a further two years to reach harvestable size.

EcoWin calculates the clearance rate, defined as the volume of water cleared by oysters over a set period of time, taking into account the population that exists in each box. The results in Table 24 are thus integrated annual values from hourly model outputs for the 20 weight classes used in the population model, at all the boxes where cultivation takes place. In Box 25, where most of the cultivation occurs, the oysters filter the equivalent of the whole box volume about once a quarter; in Box 30, which has the second highest harvest, the total volume is filtered roughly 1.5 times per year. In other boxes, where the stocking density and harvest are lower, the role of oysters in ‘cleaning’ the water is less pronounced. Overall, the standard model suggests that the combined volume of the boxes under cultivation is filtered roughly once a year (0.95 y^{-1}), corresponding to an annual clearance of 9 X 10^9 m^3 of water, or over three hundred billion cubic feet.

[13] *Includes both phytoplankton and detrital organic material*

Smaal & Prins (1993) proposed two simple conditions for rapid analysis of whole water body carrying capacity: the first considers the relationship between the bivalve clearance time (CT) and the water retention time (RT) for a given volume, and the second compares CT to the primary production time (PPT). At the scale of LIS, the shellfish cultivation simulated in the standard model would take over 8 years to clear the total water volume (77.43 X 10^9 m^3), which is significantly greater than the water retention time (RT), estimated in EcoWin (Figure 38) to be of the order of one month. Even at the highest potential cultivation scenario, the system is well below carrying capacity (see scenario discussion below). The PPT aspect is far better considered using an ecological model such as EcoWin than with generic calculations, partly because shellfish also use detrital particulate organic matter (POM). We return to this component in the analysis below.

Nutrient mass balance

A similar analysis was performed to establish the role of oyster aquaculture in mitigating the impact of eutrophication in LIS, through a mass balance for both carbon and nitrogen (Table 25). Although the model works internally in carbon units, and it is therefore from those outputs that the terms in Table 25 are mainly calculated (excluding for instance shellfish excretion which is directly output in nitrogen), the main focus for discussion in this study is the nitrogen balance, because carbon is not a limiting nutrient, and therefore not a direct concern with respect to primary production and eutrophication.

Overall, for an annual harvest of about 31 x 10^3 metric tons of oysters, the shellfish population removes over 650 tonnes of nitrogen (Figure 46), an ecosystem service corresponding to about 200,000 population-equivalents, If we consider a unit value of $40 per PEQ (Lindahl et al., 2005), this represents a potential annual income of approximately $9,000,000 from nutrient credit trading, adding 6.4% to income from product sales (calculated using $0.40 per piece); this number may well underestimate the potential value of shellfish in eutrophication abatement—Stephenson et al. (2010) present a range of unit costs for non-point source nitrogen removal at least one order of magnitude above those reported by Lindahl et al (2005) using estimates from wastewater treatment.

Figure 46 represents a synthesis of the role of the Eastern oyster in top-down control of eutrophication symptoms. The total POM uptake data in Table 25 have been divided, for both carbon and nitrogen balance, into the algal and detrital components. This is possible because the AquaShell individual growth model discriminates between the two food sources (phytoplankton and detritus). Filtration of detrital organic matter corresponds to 46% of the total organic removal, which is a good reason to avoid a bulk assessment based on the primary production time (PPT) approach referred earlier.

Although eutrophication assessment models such as ASSETS do not consider the detrital organic material in the water column as part of the primary symptoms, detritus then appears implicitly through its dual influence on underwater light climate, through the indicator Submerged Aquatic Vegetation (SAV), and on DO through the indicator bottom water hypoxia.The average percentage of nitrogen removed, calculated against the harvested biomass is in the range 2-3.5%, with an aggregate value of 2.1%. This is double the usual reported value of 1% by weight, which reflects the fact that the whole population is included, rather than the biomass physically removed (harvested) from the water (e.g. Bricker et al., 2014, 2014a). Table 26 shows the calculated nitrogen removal per unit area.

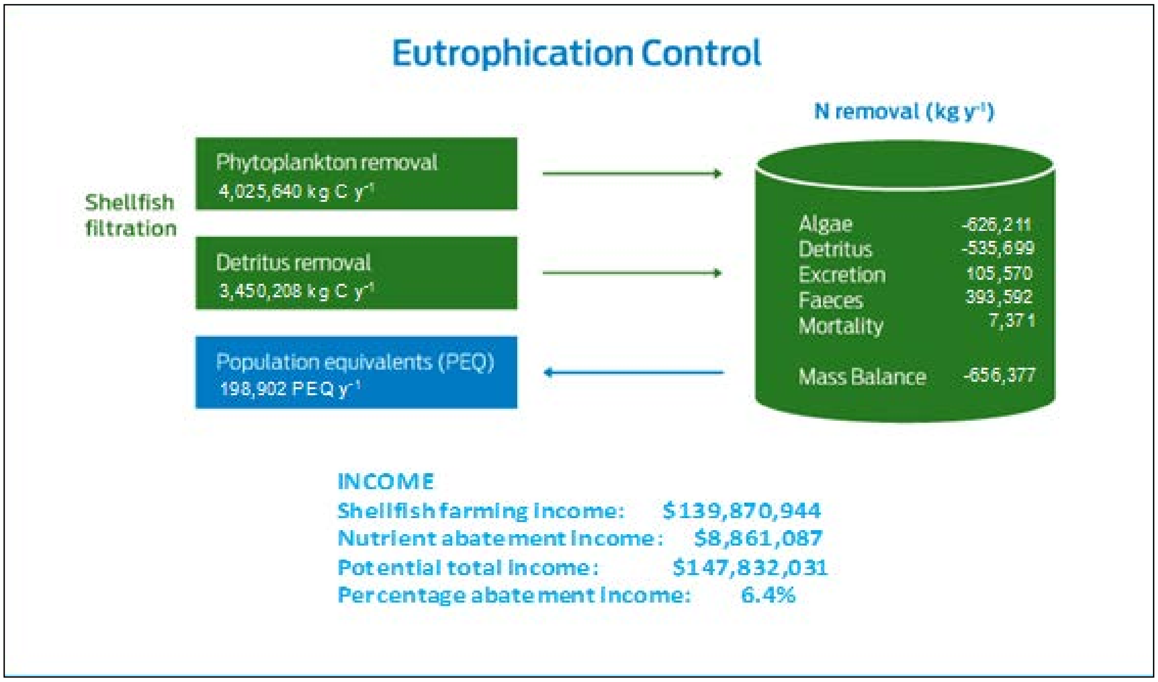


Figure 46: Mass balance for the total cultivated area of oysters in Long Island Sound, based on results from EcoWIn.NET.

These results reflect the spatial variability in (1) food composition, evident in Table 25 with respect to the uptake ratios of phytoplankton and total POM; and (2) shellfish growth rate, which varies between an endpoint of 50 g live weight for Box 41 (eastern, less eutrophic area), and over 150 g in Box 23, at the New York end of LIS. These model outputs suggest some areas of LIS perform better than others in terms of nitrogen removal per unit area, under identical conditions of stocking density per acre. Box 25, where the modeled harvest is 1-2 orders of magnitude above the other boxes, shows the lowest removal, at about 264 lb N acre^{-1} y^{-1}.

The average removal is 318 ± 79 lb N acre^{-1} y^{-1}, which can be compared with a simple calculation based on a final stocking density at harvest of 30 individuals m^{-2}, with an individual live weight of 91 g, and a nitrogen content of 1% of total live weight. This provides a value of about 243 lb N acre^{-1} over the whole culture cycle, or about 91 lb N acre^{-1} y^{-1}. The higher value (275 lb N acre^{-1} y^{-1}) obtained in this work is consistent with the alternative approach used to estimate nitrogen removal (i.e. removing the primary symptoms of eutrophication through top-down control by shellfish, rather than physically removing shellfish through harvest).

Table 26: Annual net nitrogen removal per unit area in Long Island Sound, determined from EcoWin.NET mass balance.

	Box 23	Box 25	Box 27	Box 30	Box 33	Box 41	Totals
Net N removal (kg N y^{-1})	14351	483891	6341	100823	7797	43173	656377
Net N removal (lb N y^{-1})	31611	1065839	13968	222078	17175	95094	1445764
Acres	105	4043	53	788	53	210	5252
Net N removal (lb N acre^{-1} y^{-1})[14]	301	264	266	282	327	453	1893

Chlorophyll drawdown

The role of oyster aquaculture in mitigating the impact of eutrophication can be addressed through models by using a mass balance approach (as above), but the CHL drawdown as determined from CHL concentration can also indicate top-down control. Such changes, given the large masses and volumes involved are typically more difficult to detect.

We used the ASSETS percentile 90 for CHL as an indicator (Bricker et al., 2003), calculated for Model Year 9, taking advantage of the object-oriented architecture of EcoWin, which allows objects to be switched off or on (Figure 47) for analysis.

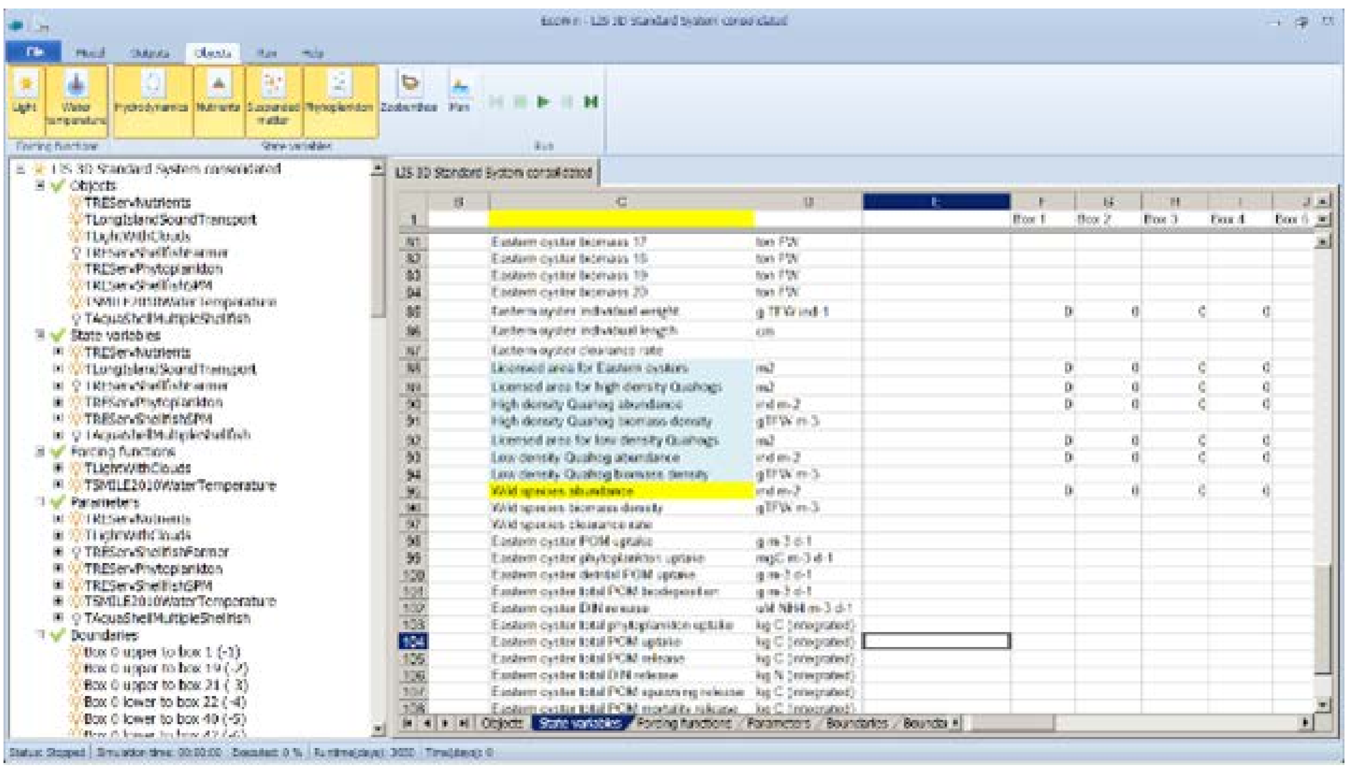


Figure 47: Approach for testing the effect of Eastern oyster aquaculture on chlorophyll concentrations in EcoWIn.NET by switching off the Zoobenthos and Man objects (top right ribbon, objects with yellow background are active).

[14]*The whole acreage is used, rather than the annual seeded acreage, because bioextraction is evaluated as a contribution of all year classes.*

The results are shown in Table 27. The highest percentage change is in the western part of the Sound, which reflects the difference in stocking density compared to the eastern area.

Table 27: Comparison of Chl P90 in EcoWin.NET model boxes containing oysters and quahogs, with shellfish active and inactive for the Standard model.

Scenario	Box 23	Box 25	Box 27	Box 30	Box 33	Box 41
Full model with oysters and quahogs	13.7	10.5	9.0	8.3	8.5	10.5
Model without shellfish	14.7	11.5	9.8	9.0	8.7	10.5
Percentage change of percentile 90 (%)	7.3	9.8	9.4	8.1	3.0	0.4

At these stocking densities, well below production and ecological (but perhaps not social) carrying capacity, the model suggests there is a significant change to CHL concentration in the western Sound, with a drawdown of almost 10%.

It is useful to analyze phytoplankton drawdown over an annual cycle (Figure 48). The general pattern shows a much greater CHL reduction (as a percentage) in the warmer months of the year. This is due to the higher shellfish clearance rates driven by the temperature increase, and is ecologically significant—higher community metabolism during this period, associated with lower oxygen solubility, means that from a perspective of eutrophication control, the ecosystem service provided by shellfish in the summer months is particularly valuable.

The west-east variation in percentage drawdown is consistent with model outputs for production and for other environmental factors.

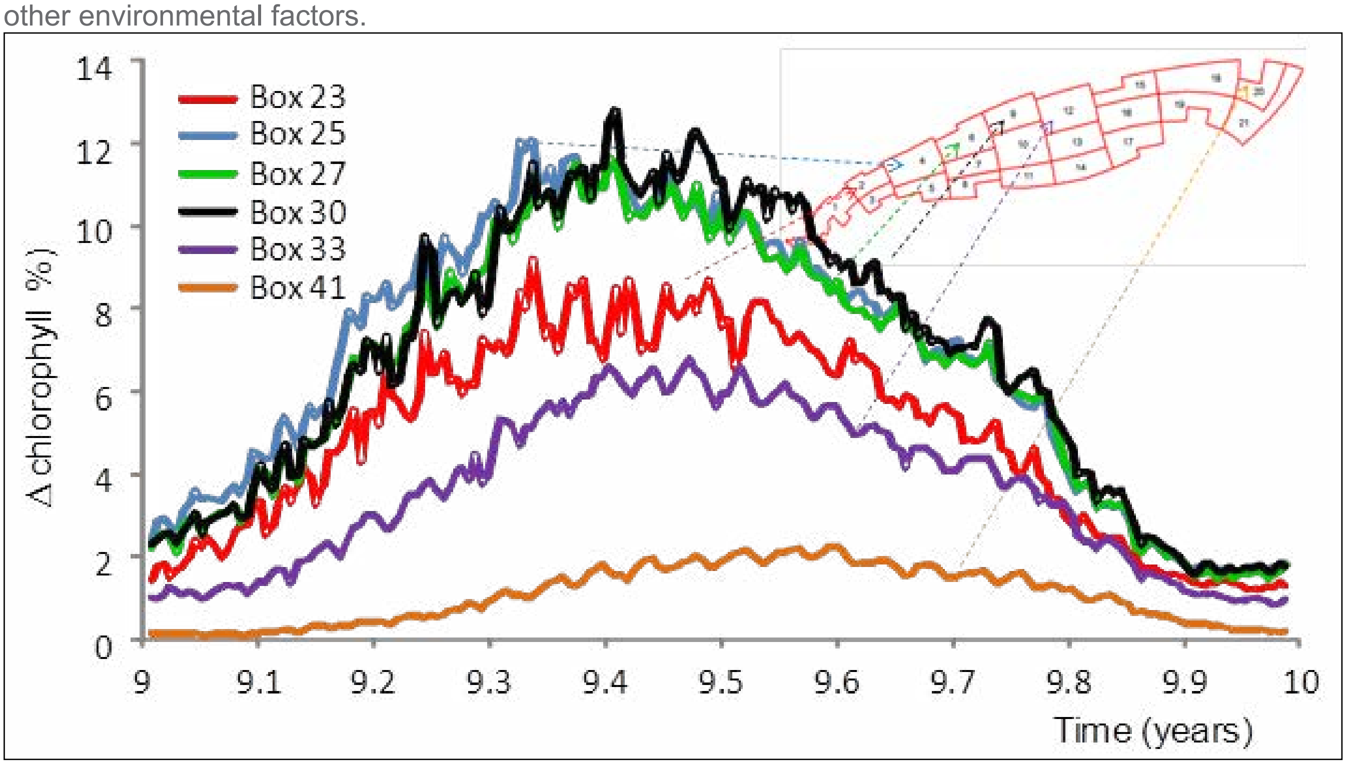


Figure 48. Temporal variation in phytoplankton drawdown simulated in EcoWin.NET by switching off the Zoobenthos and Man objects; the graph shows the percentage increase of Chl concentration in the cultivated boxes when the shellfish are inactive.

Scenario Analysis

The results below show EcoWin model outputs for the various scenarios, indicated as *Low*, *Standard*, *High*, and *Potential* (see Methods and Supporting Data section).

Table 28 represents the key production factors, considering the 4 different modeled stocking areas, which include the Standard and 3 other scenarios.

The predicted harvest for the maximum stocking areas (*Potential* scenario) is about double (204%) the results obtained with the standard model. Since the area increases by 112%, there is a very small effect of food depletion at the higher density. This is reflected in the Average Physical Product (APP), which decreases by about 0.8 between the *Low* and *Potential* scenario for the aggregate set of cultivated boxes.

Nevertheless, the APP does not fall below 15 (i.e. 1 kg of seed yields 15 kg of product), which makes the cultivation financially attractive even for the largest stocking area scenario under consideration. This suggests that even in the *Potential* situation, ecological balance is maintained or improved—CHL is lower, but oyster production appears to remain in the Stage I section of the carrying capacity curve (Ferreira et al., 2007).

It is worth noting that this analysis is executed at an ecosystem scale, and therefore shellfish stock (per box) is uniformly distributed in relatively large model cells; consequently (see for instance Nunes et al., 2011), the results obtained here are less constrained than those obtained with a farm-scale analysis for a particular box. On the other hand, EcoWin takes multiple culture cycles into account, and a model such as FARM does not, which to some extent reduces the disparity between the two approaches.

The correct sequence, however, when looking at the potential for bioextraction in an area such as LIS, is to examine carrying capacity from a system perspective, and only after that apply a local-scale model at selected sites. The reverse approach does not take into account the interactions among farms in any expansion scenario, which are particularly important for organically extractive aquaculture—the EcoWin results presented herein explicitly account for those interactions.

Table 28: Areas, seeding, harvest, and Average Physical Product (output/input, APP) for the four sets of model outputs for oysters.

	Box 23	Box 25	Box 27	Box 30	Box 33	Box 41	Totals
Scenario	Areas (m^2)						**Total area (m^2)**
Low	35612	1371076	17806	267093	17806	71225	1780618
Standard	424920	16359432	212460	3186902	212460	849841	21246015
High	619008	23831797	309504	4642558	309504	1238015	30950385
Potential	899698	34638370	449849	6747734	449849	1799396	44984896
Scenario	Areas (acres)						**Total area (acres)**
Low	8.8	339	4.4	66	4.4	18	440
Standard	105	4043	53	788	53	210	5250
High	153	5889	76	1147	76	306	7648
Potential	222	8559	111	1667	111	445	11116
Scenario	Seed (tonnes)						**Total seed (tonnes)**
Low	3.3	128	1.7	25	1.7	6.6	166
Standard	40	1521	20	296.3819	20	79	1976
High	58	2216	29	431.7579	29	115	2878
Potential	84	3221	42	628	42	167	4184
Scenario	Harvest (tonnes Year 9)						**Total harvest (tonnes)**
Low	52	2061	26	392	26	104	2662
Standard	631	24319	306	4486	303	1224	31268
High	922	35070	437	6366	435	1772	45002
Potential	1345	49935	614	8865	619	2555	63932
Scenario	APP (Year 9)						**Aggregate APP**
Low	15.85	16.16	15.93	15.78	15.77	15.67	16.07
Standard	15.96	15.98	15.48	15.14	15.35	15.48	15.83
High	16.01	15.82	15.19	14.74	15.12	15.39	15.63
Potential	16.07	15.50	14.68	14.13	14.79	15.27	15.28

Table 29 summarizes the environmental effects associated with the four scenarios, for the six model boxes where shellfish are cultivated. The ratio of annual water clearance to aggregate volume, an indicator of the role of oysters in bioextraction, increases from 7.8 % in the *Low* case, to 207.6 % at the highest potential expansion. At the *Potential* scenario, oysters therefore filter the total volume of the cultivated boxes twice per year.

The net nitrogen removal through cultivation of *C. virginica* corresponds to only 17,000 people at the lowest stocking conditions, and increases to over 400,000 population-equivalents for the *Potential* cultivated area.

Nitrogen removal per acre always (except in Box 41) decreases as the stocked area increases, although there is only a small reduction in APP (Table 28). The decrease in N removal through the whole range of stocking densities is only 7.5 % in aggregate, and it is probably due to the shift in population distribution, with more oysters in the lower weight classes, as the cultivation area increases.

Table 29: Water clearance and nitrogen removal by oysters for the four sets of model outputs.

Item	Box 23	Box 25	Box 27	Box 30	Box 33	Box 41	Totals
Low (440 acres)							
Clearance (106 m3 y-1)	13.3	547.3	7.6	120.6	8.7	38.7	736.1
Clearance/Volume (% y-1)	1.7	30.5	0.6	11.7	0.4	1.6	7.8
Net N removal (kg N y-1)	1249.97	42530.00	546.92	8646.06	658.17	3618.36	57249.48
Net N removal (lb N y-1)	2753.24	93678.41	1204.66	19044.18	1449.72	7969.95	126100.17
PEQ y-1	379	12888	166	2620	199	1096	17348
Acres	8.80	338.80	4.40	66.00	4.40	17.60	440.00
N removal (lb N acre-1 y-1)	312.87	276.50	273.79	288.55	329.48	452.84	286.59
Standard (5250 acres)							
Clearance (106 m3 y-1)	159.7	6649.7	93.8	1492.6	105.8	463.2	8964.8
Clearance/Volume (% y-1)	20.8	370.4	7.0	144.6	5.0	19.6	95.0
Net N removal (kg N y-1)	14351.25	483890.87	6341.24	100823.48	7797.24	43172.74	656376.82
Net N removal (lb N y-1)	31610.68	1065838.92	13967.50	222078.14	17174.54	95094.15	1445763.93
PEQ y-1	4349	146634	1922	30553	2363	13083	198902
Acres	105.00	4042.50	52.50	787.50	52.50	210.00	5250.00
N removal (lb N acre-1 y-1)	301.05	263.66	266.05	282.00	327.13	452.83	275.38
High (7648 acres)							
Clearance (106 m3 y-1)	233.4	9798.9	139.0	2214.1	155.5	676.1	13217.1
Clearance/Volume (% y-1)	30.5	545.9	10.3	214.5	7.3	28.6	140.1
Net N removal (kg N y-1)	20511.45	690867.40	9128.32	145385.73	11320.21	62896.67	940109.78
Net N removal (lb N y-1)	45179.40	1521734.36	20106.42	320232.88	24934.39	138538.93	2070726.39
PEQ y-1	6216	209354	2766	44056	3430	19060	284882
Acres	152.96	5888.96	76.48	1147.20	76.48	305.92	7648.00
N removal (lb N acre-1 y-1)	295.37	258.40	262.90	279.14	326.02	452.86	270.75
Potential (11116 acres)							
Clearance (106 m3 y-1)	340.9	14524.5	207.0	3302.1	228.8	985.6	19588.9
Clearance/Volume (% y-1)	44.5	809.1	15.4	320.0	10.7	41.7	207.6
Net N removal (kg N y-1)	29018.21	979372.36	13061.14	208399.53	16375.65	91432.99	1337659.88
Net N removal (lb N y-1)	63916.76	2157207.85	28769.03	459029.80	36069.71	201394.25	2946387.40
PEQ y-1	8793	296780	3958	63151	4962	27707	405351
Acres	222.32	8559.32	111.16	1667.40	111.16	444.64	11116.00
N removal (lb N acre-1 y-1)	287.50	252.03	258.81	275.30	324.48	452.94	265.06

Table 30 shows the changes in the 90th percentile of CHL at the highest stocking density. There is a shift in the pattern of reduction when compared to the standard model (Table 27), with a more pronounced change in Box 23, when compared to Box 30.

In addition, the changes in CHL percentile 90 in the standard model were all below 10 %, whereas at the higher stocked areas of the *Potential* scenario, the four boxes further to the west all show reductions above 15 %.

Table 30: Comparison of Chl P90 in EcoWin.NET model boxes containing oysters and quahogs, with shellfish active and inactive for the Potential scenario.

Scenario	Box 23	Box 25	Box 27	Box 30	Box 33	Box 41
Full model with oysters and quahogs	12.7	9.5	8.3	7.8	8.2	10.4
Model without shellfish	14.7	11.5	9.8	9.0	8.7	10.5
Percentage change of percentile 90 (%)	15.9	20.9	18.7	15.3	6.4	0.7

Nutrient bioextraction by quahogs

Quahogs were modeled using a hybrid approach (see 'Methods and Supporting Experiments and Data') which did not include individual growth or population dynamics, but did partition the abundance of different size classes following Fegley (2001), and calculate clearance rate as a function of individual length and temperature (Doering & Oviatt, 1986). The role of quahogs in nitrogen removal has been calculated for the standard model and the three scenarios (Table 31) for both phytoplankton and detrital organic matter.

Although the table shows similar parameters to those calculated for oysters in Table 30, the results presented here should be considered as indicative only. In particular, the nitrogen removal estimates are based solely on the filtration of phytoplankton and detritus, and are therefore gross rather than net. A proportion of the nitrogen removal estimated for quahogs will be returned to the water column in particulate form as pseudofeces and feces, and in dissolved substances excreted by the animals; there will be additional losses due to mortality and spawning, since an individual model for growth of quahogs was not developed in this study, these terms could only be estimated from field data, if available.

Since the PEQ estimates are based on gross nitrogen rather than net removal, these values will be exaggerated. A conservative estimate for bioextraction would be 50% of the PEQ (the modeled value for oysters is 56%, data in Figure 46), i.e. accounting for organic and inorganic losses. For the standard model, the combined culture of oysters and quahogs would provide an ecosystem service corresponding to over 600,000 PEQ. The *Potential* scenario roughly doubles this number, to 1,250,000 PEQ.

Despite the caveats inherent in this simplified approach, it is worth noting that (1) bioextraction appears to be higher for the quahogs than the oysters, despite the lower clearance rates, because the area occupied is substantially greater and this simplified approach does not consider natural mortality; (2) the removal rate per acre is lower than for oysters, although this may be partly due to the fact that both constant length and abundance are used for each of the size classes.

Marginal analysis of harvestable oyster biomass

EcoWin can be used to perform a marginal analysis in a similar way to local-scale models such as FARM. For the system-scale approach, rather than increase the stocking density at the farm site, the stocking density (or lease area) in each box is increased. Ferreira et al (2014) have shown that a detailed optimization analysis for different stocking densities (S), or its equivalent, increased lease areas, for the various boxes (B), will require S^B model runs.

This rapidly reaches a limit in terms of computational time. For example an analysis of the 13 stocking densities (or lease areas) used to make Figure 49 (for Box 25 only) in the six EcoWin boxes presently cultivated in Long Island Sound would require 13^6, i.e. over four million model runs, roughly eighteen years of continuous model simulations.

Table 31: Nitrogen removal by quahogs for the four sets of model outputs.

Item	Box 23	Box 25	Box 27	Box 30	Box 33	Box 41	Totals
Low (440 acres)							
Gross N removal (kg N y^{-1})	28198	59220	62482	72235	9672	2988	234797
Gross N removal (lb N y^{-1})	12802	26886	28367	32795	4391	1357	106598
PEQ	8545	17946	18934	21890	2931	906	71151
Acres	124	310	347	396	50	12	1239
Gross N removal (lb N $acre^{-1}$ y^{-1})	103	87	82	83	89	110	86
Standard (5250 acres)							
Gross N removal (kg N y^{-1})	341949	708794	751440	869605	120172	37853	2829814
Gross N removal (lb N y^{-1})	155245	321793	341154	394801	54558	17185	1284736
PEQ	103621	214786	227709	263517	36416	11471	857519
Acres	1575	3938	4410	5040	630	158	15750
Gross N removal (lb N $acre^{-1}$ y^{-1})	99	82	78	78	87	109	82
High (7648 acres)							
Gross N removal (kg N y^{-1})	487150	1003308	1066288	1234553	173215	55055	4019570
Gross N removal (lb N y^{-1})	221166	455502	484094	560487	78640	24995	1824885
PEQ	147621	304033	323118	374107	52489	16683	1218052
Acres	2295	5736	6425	7342	918	229	22945
Gross N removal (lb N $acre^{-1}$ y^{-1})	96	79	75	76	86	109	80
Potential (11116 acres)							
Gross N removal (kg N y^{-1})	686227	1400340	1493683	1730577	248075	79834	5638737
Gross N removal (lb N y^{-1})	311547	635754	678132	785682	112626	36245	2559987
PEQ	207948	424345	452631	524417	75174	24192	1708708
Acres	3335	8337	9337	10671	1334	333	33347
Gross N removal (lb N $acre^{-1}$ y^{-1})	93	76	73	74	84	109	77

The best way to optimize this type of analysis is therefore by producing a family of curves, and using Monte Carlo methods for optimization. It is important to note that while management decisions can be well informed by models of this type, policies that affect the livelihood of shellfish farmers should be fully participative, since there is a strong element of social choice that must be enacted.

Figure 49 illustrates how changes in stocking density or lease area in Box 25 will affect harvestable oyster biomass in that box. This is a typical representation of the law of diminishing returns, such as presented for FARM e.g. in Ferreira et al. (2007), and both FARM and EcoWin in Nunes et al. (2011). The seeding currently used in the standard model is low when compared to many other areas used for oyster cultivation throughout the world, and the availability of particulate organics in Box 25 means that from the perspective of growth potential, the oyster stock in the box could be substantially increased.

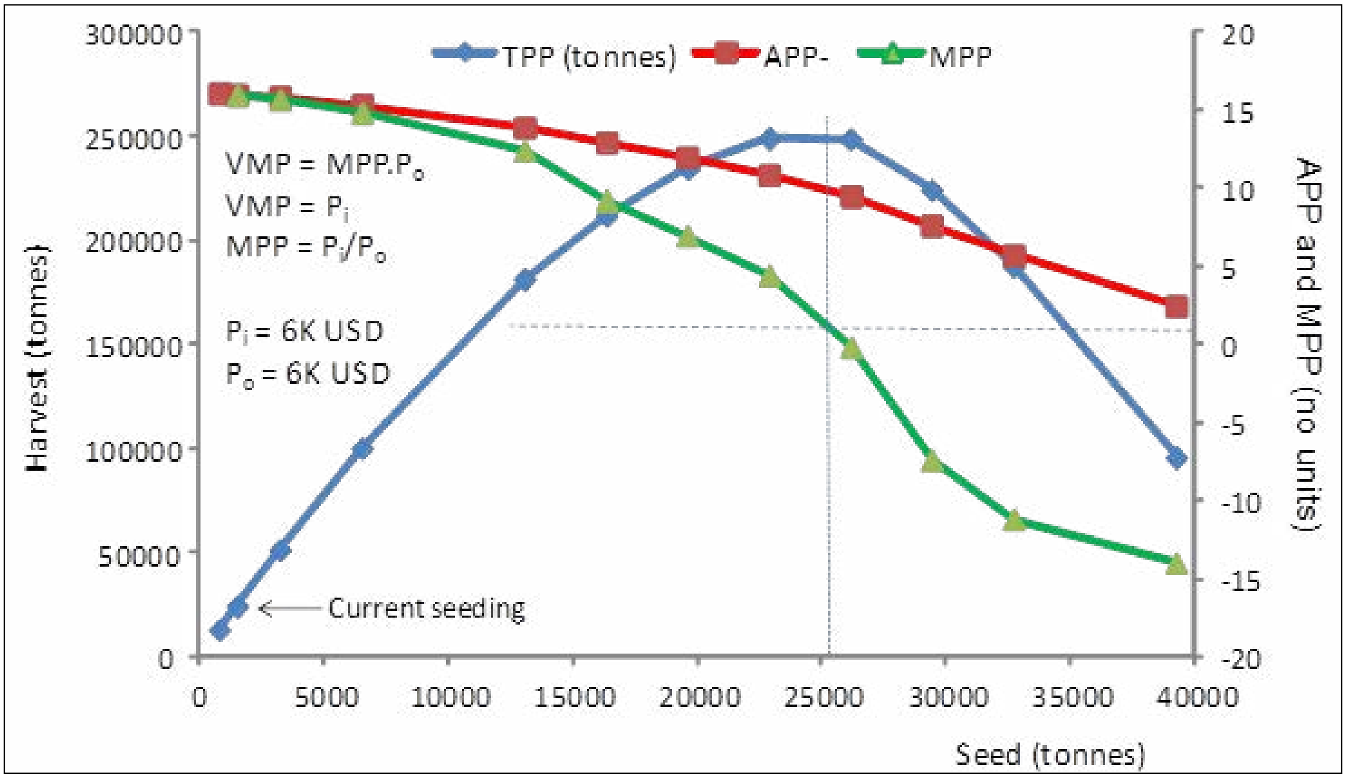


Figure 49: Marginal analysis of the increase in lease area and/or stocking density of Eastern oysters in Box 25 of the EcoWin model, keeping all other boxes unchanged – production data are from Year 9 of the model run. TPP – Total Physical Product (harvest), APP – Average Physical Product (Harvest/Seed), MPP – Marginal Physical Product, VMP – Value of Marginal Product, Pi – price of input (seed), Po – Price of output (harvest).

The marginal curve only starts to flatten with an annual stocking of 20,000 tonnes, and the optimum profit point, considering $P_i = P_o$, is roughly at the maximum production level. There is not enough data on industry costs or revenue for this system to be able to extend this analysis. It is however clear that the more P_i is in excess of P_o, the greater the Marginal Physical Product (MPP) for the optimum profit point will be, and this will shift the stocking density for profit maximization to the left of the production of curve. An increase in stocking density will also have a potential effect on the nearby boxes where oysters are cultivated.

This is shown in Table 32, which includes the total harvest for the Sound, excluding Box 25. Three points are of note: (1) the harvest in Box 23 increases in the early stages of increased stocking in Box 25. This is the only box where this happens, and presumably is linked to the additional subsidy of particulate organics from increased cultivation in Box 25; (2) the percentage change in harvestable biomass in each box, based on the maximum and minimum seeding shown for Box 25, leads to a maximum reduction in Box 27, and Box 23 shows a relatively small reduction even compared to Box 33, which is substantially further east; (3) even Box 41, which is on the eastern side of the Sound, is affected by the changes in stocking in Box 25. This is observed not only at the highest stocking of Box 25, but throughout every scenario of increased stocking of the test box.

Table 32: Effect of stocking increase in Box 25 only on harvest in other oyster aquaculture areas of Long Island Sound (all values in tonnes). Data are for the same period as the marginal analysis, i.e. Year 9 of the 10 year model run.

Seed in Box 25	Box 23	Box 25	Box 27	Box 30	Box 33	Box 41	Total excluding Box 25
1521	631	24319	306	4486	303	1224	6950
3272	632	51671	302	4442	301	1220	6898
6544	634	100195	295	4359	297	1215	6799
13088	635	181240	278	4184	289	1204	6589
16359	633	211272	268	4093	285	1199	6478
19631	629	234001	258	4002	281	1194	6365
22903	623	248621	249	3915	277	1189	6254
26175	616	248066	240	3834	274	1185	6149
29447	607	223779	232	3758	270	1181	6049
32719	598	187162	223	3683	267	1177	5948
39263	576	95798	208	3553	261	1170	5768
Percent change	-8.6		-32.1	-20.8	-13.8	-4.4	-17.0

Overall, the model suggests that stakeholders who farm in other boxes would be affected by a decreased yield of 17%. This reinforces the fact that decisions on expansion and redistribution of organically extractive aquaculture in different zones need to reflect a social consensus, as well as the appropriate environmental and production aspects.

Data used for ASSETS and FARM in Long Island Sound

In general, we did not find any significant trends at any station for any of the parameters examined for the years 2008-2012. At specific locations for specific water quality variables the test showed very small trends that on closer analysis were seen to be driven by anomalous observations. All data for 2008-2012 were used for the application of the ASSETS eutrophication model to assess nutrient related water quality within LIS and for the FARM model simulations. Additional to these data, seagrass areal trends, occurrence of macroalgal and nuisance/toxic algal blooms, and other relevant information for the assessment were acquired from ongoing and previous studies for completion of the eutrophication assessment (e.g. Bricker et al., 2006; LIS Health Report, 2012, Gobler et al., 2011; Gobler and Hattenrath-Lehmann, 2013; Hattenrath-Lehmann and Gobler, 2011, 2010; Lopez et al., 2014; Tiner et al. 2003, 2007, 2010, 2013, M. Tedesco, Long Island Sound Study, pers. comm.).

Plots of the annual cycle for temperature, CHL, DO, show typical patterns seen in the Northeast US with temperatures lowest in the winter months (January-March) and highest in summer months of June - August, consistent with temperatures on land (Figure 6 shows 4 LIS stations as an example). In many estuaries in the Northeast region of the US an annual cycle is observed where CHL concentrations peak in February or March during the winter-spring bloom period and may also peak in the fall with the fall turnover of the system (Lopez et al., 2014). This pattern is seen in CHL concentrations in some of the stations but the pattern is highly variable from year to year. Note also that concentrations in the western Sound (i.e. Narrows zone) are greater than in the other 3 zones; concentrations decrease from west to east with distance from the New York metropolitan area (Figure 7).

Dissolved oxygen concentrations measured in grab samples were more consistent than for CHL, showing highest concentrations in the cooler months, lower during the summer period (July – September; Figure 8). Concentrations in the Narrows and Western zones consistently dropped below 5 mg L^{-1}, the threshold considered 'biologically stressful', and sometimes to less than 2 mg L^{-1}, the hypoxic threshold (Bricker et al., 2003). In the Central zone, biologically stressful concentrations were observed in July – September, but concentrations did not drop to hypoxic levels. In the Eastern zone, observed DO concentrations were all above 6 mg L^{-1}.

These observations are consistent with descriptions of dissolved oxygen conditions in the 2012 LIS Health Report (LISS, 2012) where hypoxia was reported to occur in summer months, usually in the Western Sound and Narrows though the total area with variable duration and areal extent from year to year. After seven years of below average severity, hypoxia increased slightly in 2011 and more significantly in 2012. The area affected by hypoxia in the summer of 2012 was the fifth largest since 1987— 289 square miles, an area about 13 times the size of Manhattan. The duration of hypoxia in 2012 was 80 days, 23 days more than the average between the years 1987 and 2012 (LISS, 2012).

Farm Aquaculture Resource Management (FARM) model in Long Island Sound

The FARM model was applied to the data from Stations A4, 09, H2 and M3 from the CT DEEP monitoring network (Figure 1) using culture practices and environmental data shown in Table 33 and Table 34, assuming that 3 acres of a 4-acre simulated farm were in cultivation at each location, consistent with culture practices used by local growers. The model runs were executed with input drivers as shown in Figure 6 - Figure 8 and culture practices listed in Table 33. Note that while typical practices include harvest of one and two inch seed from restricted areas and growout for one to two years in approved areas (Figure 12), we have used a smaller seed for the simulation in order to include the removal capacity that includes the early lifestages of the oyster. A farmgate price of $0.40 per oyster was used to calculate harvest value. Neap and spring current speeds for 2012 were calculated from NOAA tides and currents stations corresponding to the location of the sampling stations: Greenwich Point, Piney Creek, Charles Island, Little Gull Island for Stations A4, 09, H2 and M3, respectively (http://tidesandcurrents.noaa.gov/noaacurrents/Stations?g=444).

Table 33: Culture practice data for Long Island Sound used in the FARM model simulations.

Long Island Sound shellfish cultivation practice	
Species	*Crassostrea virginica*
Cultivation practice	Bottom culture no gear
Seed density (number oysters/ sq meter)	62
Seed size (from hatchery+) mm (g)	14 (0.55)
Cultivation period (d)	1095 (day 1 = May 1 = Julian day 122)
Harvest size mm (g)	76 mm (90 g)
Seed price	150 – 250 in a bushel, a bushel is \$10-\$15 = \$20 / kg
Farmgate price	\$0.40 / oys = \$4.40 /kg for 90g oys
Mortality (% per cultivation cycle)	55
Environment	
Peak current at spring tide (m s^{-1})	0.54 (Station A4) 0.46 (Station 09) 0.31 (Station H2) 2.86 (station M3)
Peak current at neap tide (m s^{-1})	0.48 (Station A4) 0.26 (Station 09) 0.15 (Station H2) 2.77 (Station M3)
Nitrogen loading (metric ton y^{-1})	50,000

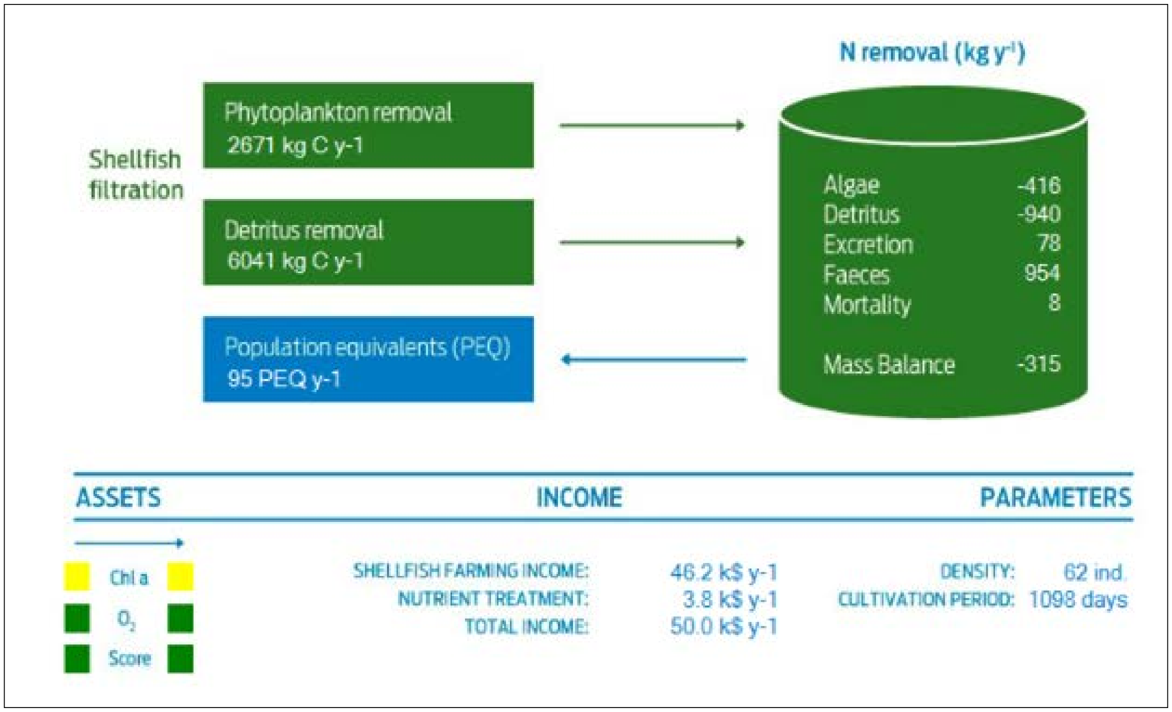
+While hatchery seed is typically not used since seed oysters of one and two inches are harvested from restricted areas and grown out in approved areas, for the model to account for nutrient removal through the life of the oyster, we have used spat as the starting oyster size and weight and use measured hatchery size and weight.

Table 34: Annual average temperature, salinity, chlorophyll, POM and TPM concentrations LIS (average 2008-2012 from Connecticut Department of Energy and Environmental Protection) and GBP (average 2005-2010 from NH Department Environmental Services).

Station	Temperature (°C)	Salinity (psu)	Chlorophyll (ug L^{-1})	POM (mg L^{-1})	TPM (mg L^{-1})
Long Island Sound					
A4	12.6	25.5	14.82	2.45	6.62
9	12.3	26.2	6.09	1.56	6.20
H2	13.0	26.8	5.21	1.50	6.34
M3	11.8	29.7	2.50	0.85	5.10
Great Bay Piscataqua					
GRBAP	11.2	21.8	3.59	2.16	21.7

FARM model results in Long Island Sound

The results for the four stations using site-specific current speeds and environmental variables varied, with harvest being greater at the westernmost stations where there is more food available. Harvest accounts for a culture period of 1098 days at a simulated 4 acre farm (3 acres in cultivation) at each location. Results are shown for all stations in Table 35. Harvest at Station 09 was 31.6 metric tons (3.5 tons acre^{-1} year^{-1}; Figure 50) with removal of about 0.105 metric ton N acre^{-1} year^{-1} (Figure 50). This removal represents an equivalent ecosystem service of nutrient treatment for 32 people acre^{-1} y^{-1} with a value of $1,300 acre^{-1} y^{-1} which would be additional income to the farmer if there were a nutrient trading program in place (Table 35). Simulation results for Station A4, further to the west and with greater annual average CHL, POM and TPM concentrations show greater harvest and removal than at the other stations. Note that the value of the shellfish product at Stations A4, 09 and H2 are greater than the value of the ecosystem service represented by the removal of nitrogen. Harvest is much smaller at the easternmost station and value of the ecosystem service is greater than the value of the product.



		Ecosystem service equivalent	
Oyster harvest (metric ton acre^{-1} yr^{-1})	N removed* (metric ton acre^{-1} yr^{-1})	PEQ (acre^{-1} yr^{-1})	Value of service ($10^3 acre^{-1} yr^{-1})
3.5	0.105	32	1.3

Figure 50: FARM simulation results for simulated farm at LIS Station 09 for cultivation of 3 acres of a 4 acre farm; oyster harvest, N removed, People Equivalents (PEQ) and ecosystem value of nutrient removal by oyster filtration at the simulated site.

The FARM model results show that CHL concentrations decreased at all stations except Station M3 but only slightly. DO concentrations changed only slightly at Stations 09 and H2 and thus the ASSETS rating did not change for either CHL or DO for any station (Table 35). These small changes suggest that there is no negative impact on water quality from the aquaculture operation and that there is margin for additional density of seed at all. These results are consistent with the EcoWin results and marginal analysis showing that carrying capacity can accommodate additional seeding density which would also improve the nutrient removal capacity.

Table 35: FARM simulation results for a) Stations A4, 09, H2 and M3 for harvest, N removal, PEQs represented by N removal, changes in Chlorophyll and Dissolved Oxygen concentration and value of harvested product and ecosystem service (i.e. nutrient treatment) represented by N removal. In parentheses is the EcoWin model grid box that corresponds to the location of the station, for comparison in upscaling.

Station Results	Station A4 (EcoWin box 23)	Station 09 (EcoWin box 25)	Station H2 (EcoWin box 27)	Station M3 (EcoWin box 41)
Harvest (tons/acre*yr)	3.7	3.5	3.2	0.0
N removed (tons/ acre*yr)	0.317	0.105	0.097	0.021
PEQ /acre*yr	96	32	29	6
change in Chl concentration (ug/l)	-0.04	-0.02	-0.01	0
change in DO concentration (ug/l)	0	-0.01	-0.01	0
$ for product /acre* yr	$16,067	$15,400	$14,233	$200
$ for ecosystem service (nutrient removal) / acre* yr	$3,833	$1,267	$1,167	$267
APP	44	42	39	0.5

These are local scale simulations showing N removal of 0.32 – 0.021 metric ton acre^{-1} yr^{-1}, decreasing from west to east. These bioextraction values are within the range of removal rates estimated in other places for various other species (Rose et al., 2014). The results of the FARM model are for farmscale only. It would be more useful to calculate the total removal systemwide based on present aquaculture acreage in cultivation and also to estimate potential significance if the LIS aquaculture industry were to expand.

Upscaling FARM model results in Long Island Sound

The per acre local scale FARM model results were used in combination with corresponding acreages under cultivation in EcoWin boxes as described in section 'Aquaculture Practice and Cultivation Area' to estimate the potential significance of removal by current and potential oyster aquaculture on the CT side of LIS. In the upscaling area-weighted calculations, results from Station A4 were used to represent conditions in the Narrows (EcoWin box 23), while results from Station 09, H2 and M3 were used to represent Western (EcoWin box 25), Central (EcoWin boxes 27, 30, 33) and Eastern (EcoWin box 41) LIS, respectively (Table 35). However, upscaling from farm to system scale requires additional assumptions and caveats. The assumption is made that there are no additional reasons that the identified bottom area could not be leased; that all lease areas have the same oyster growth and N removal rates, despite potential differences in water quality among farm locations; and that there is no interaction, i.e., food depletion (as discussed in 'Marginal analysis of harvestable oyster biomass', where adjacent boxes may have a 17% reduction in harvest), among adjacent farms. Results of upscaling of FARM and comparison to EcoWin results for N removal, PEQs and values of product and ecosystem services are shown in Table 35. There is good correspondence between the two methods of estimation.

Upscaled FARM results show that present oyster cultivation (440 – 7648 acre scenarios) removes an estimated 46 - 799 metric t N y^{-1} or 0.09 – 1.6% of the total N input to Long Island Sound ($50x10^3$ t y^{-1}). This represents an ecosystem service of nutrient treatment for 14,000– 243,000 PEQs with possible supplement to grower's income through nutrient trading of $0.555 x 10^6 – $9.65 x$10^6$. Upscaling to the maximum acres of potential cultivated area (11,116 acre scenario), assuming that all areas would have the same production and bioextraction capabilities, would lead to an annual N removal through shellfish harvest of 1162 metric t N y^{-1}, 2.36% of the total input. This is equivalent to an ecosystem service of nutrient treatment for 353,000 PEQs with an ecosystem service value of $14 x$10^6$ (Table 36). It would take more than 478 x 10^3 acres in active oyster cultivation, the equivalent of ~1,936 km^2 bottom area to remove the total input to LIS at the estimated rates of removal. This represents about 60% of the area of the Sound.

Table 36: Upscaled FARM results for low, mid, high and potential cultivated oyster leases using areas from EcoWin grid boxes and individual station results. FARM upscaled results are compared to EcoWin results. (Note that total N input to Long Island Sound is $50x10^3$ t y^{-1}).

	Ecowin	FARM (upscaled)
N removal t y^{-1} (% total load)		
440 acres	57.3 (0.11)	46.0 (0.09)
5250 acres	656 (1.31)	549 (1.10)
7648 acres	940 (1.88)	799 (1.6)
11116 acres	1338 (2.68)	1162 (2.32)
PEQ (1000s)		
440 acres	17	14
5250 acres	199	167
7648 acres	284	243
11116 acres	405	353
Treatment value ($1000)		
440 acres	---	555
5250 acres	---	6,622
7648 acres	---	9,647
11116 acres	---	14,021

It is possible that harvest and N removal would be greater with lower mortality or higher seeding densities meaning that there would be a greater removal per acre. However, higher seeding densities might also cause depletion of dissolved oxygen if over loaded. Another thing to note is that this includes only the area within the CT side of LIS and may underestimate the potential removal since there are 412,018 acres of certified shellfish area in the NY side of LIS. If this area was included in the calculation, assuming the same removal rate, and that only 1/3 of the acres are actively cultivated, potential removal increases by 14.4 x 10^3 tons per year or by 28% of the total input.

This suggests that bioextraction, although it does not remove a significant amount of the total N load at present, is a promising management measure in LIS. It could play a significant role in nitrogen reduction strategies if expanded and combined with traditional point and non-point management measures. Ways to increase the removal through bioextraction are to increase the cultivated areas, including the NY acres as noted above and to increase seeding densities. Since water quality is not negatively impacted by present cultivation practices, it is an indication that seeding could increase without negative impacts, as shown in Figure 49. The seeding densities are low compared to other places (typically 100 oyster m^{-2}; Rose et al., 2014a) and it is possible that nitrogen removal would be more significant if seeding densities were increased to those typical of other places. The FARM model simulation using the 100 seed oyster densities, with all other inputs the same, suggests that nitrogen removal would increase by 38%, while water quality would not decline at any station. The maximum removal in the scenario with potential 11,116 acres in oyster cultivation in CT waters would remove 3.2% of the total N load (50,000 tons y^{-1}).

The level of bioextractive N removal is small compared to the 33% reduction (removal) of point source loads (35% of total loads) anticipated in the CT River Basin if all WWTPs above CT were brought to a discharge concentration of 3 mg/l of total N (NEIWPCC, 2008). Compared to the 7% anticipated reduction of non-point source N loads if all agricultural BMPs were 100% implemented and the 5.5% anticipated reduction of non-point source N loads if all urban BMPs were 100% implemented (NEIWPCC, 2008) shellfish bioextraction of N cannot be considered insignificant.

Prospects for Expansion of Shellfish Aquaculture in Long Island Sound

In LIS there is a general consensus that the traditional production of oysters is limited by the availability of seed and the availability of suitable growing grounds (areas in protected waters with firm bottom that keeps oysters from being smothered in silt or buried in hurricanes). If the growers or the state were to expand efforts to plant shell, and nature cooperated by providing ample sets of larvae, it is believed that production could as much as double. However, this assumes that there is adequate firm bottom lease area and that the plagues of disease, hurricanes and starfish will not ruin the crop. Additionally, shell for cultch is in tight supply up and down the east coast causing the price of shell to double in the last two decades. While purchase of cultch from elsewhere could help with increasing production, it would require assurances that the shell is disease free. Shell recycling programs with local restaurants has been successful for re-claiming shell in Rhode Island and might be useful as a model.

It is unlikely that the existing 70,000+ acres of leased bottom will ever all be devoted to oyster or clam planting. However advances in production techniques, predator control, prices or permitting could lead to substantial investments in planting and harvests that would increase production. Connecticut managers speculate that future growth of oyster production in LIS will depend primarily on the expansion of intensive culture efforts, utilizing hatchery-reared seed, cages and (potentially) long-lines of floating cages. The latter would involve permitting issues as floating farms can meet aggressive opposition from boaters and other fishermen as well as from waterfront property owners who are protective of their view. Any plans to expand aquaculture operations in the future will thus take some time to address concerns (called 'social carrying capacity,' Angel and Freeman, 2009) of waterfront landowners and boaters who do not want to see commercial harvest activities, floating gear or risk having boats become entangled in culture gear.

These techniques noted for expansion are well suited to protected waters, however, most of the protected waters on the CT shoreline have poor water quality and are ill-suited for harvest for direct consumption. Using protected waters for nursery culture, followed by relay and growout in clean waters, will require a change in the regulatory framework because of anticipated user conflicts and interference with protected eelgrass beds. Nonetheless, intensive oyster culture in CT continues to grow and existing farms continue to increase production. As permitting challenges are resolved it is conceivable that this segment of the industry will grow at double-digit rates much like it has in other states (i.e. Rhode Island). Potential inclusion of NY in an expansion would increase the bioextraction capabilities. However, understanding of even the present aquaculture situation in NY waters is very limited due to a lack of communication thus it is not possible to speculate about what the NY contribution might be in an aquaculture expansion scenario.

Assessment of Estuarine Trophic Status (ASSETS)

The ASSETS evaluates eutrophication causes (Influencing Factors), conditions (Eutrophic condition) and forecasts future changes in nutrient related conditions (Future Outlook). The ASSETS model was applied using data and information described in 'Data used for ASSETS and FARM in Long Island Sound.'

Influencing Factors

Susceptibility: The susceptibility in LIS is determined as the combination of the ability to dilute and to flush nutrients. This is a relatively well-mixed system, though there is some stratification at the western Sound at times of high flow, thus the calculation uses the volume of the estuary as the volume available for dilution of incoming freshwater and associated substances. The ratio of freshwater inflow to the estuary volume gives a rating for dilution potential of High. The flushing potential is considered Low due to the small tidal height and small ratio of freshwater to estuarine volume (Bricker et al., 1999). The combined flushing and dilution potentials give a susceptibility rating of Moderate.

Nutrient Input: Estuaries can receive nutrient inputs from land and ocean influxes, though typically human activities on land result in land-based inputs being much greater than oceanic inputs. The load to LIS from the NY and CT parts of the watershed is estimated to be 50 x 10^3 metric tons N per year (NY SDEC and CT DEP, 2000), with 75% from point sources (WWTP and CSOs) and 25% from non-point sources (agricultural runoff, atmospheric deposition, etc; NYSDEC and CTDEP, 2000). The calculation of the ratio of human versus oceanic influence on N inputs in this system gives a rating of Moderate, meaning that land based sources are the dominant source of nutrients to LIS (Bricker et al., 2003; Ferreira et al., 2007). Note that despite a decline in loading from 60.7 x 10^3 metric tons N per year in the 1990s (P. Stacey, CT Department of Environmental Protection, retired, pers. comm.; Latimer et al., 2014), the rating has not changed since it is still mostly attributable to human related sources.

The combined Moderate susceptibility and Moderate nutrient input gives an overall Influencing Factor rating for this system of Moderate (Figure 51).

Eutrophic Condition

The calculation of Eutrophic Condition was based on an analysis of the condition of primary (CHL, macroalgal abundance) and secondary (DO, nuisance/toxic bloom occurrence, changes in seagrass coverage) indicators.

Chlorophyll a: Phytoplankton growth is directly stimulated by increases in nutrients and thus CHL, as a measure of phytoplankton biomass, is used as one of the first indications of nutrient enrichment. The 90th percentile concentration of CHL a for 2008 - 2012 ranged from 5.53- 22.82 µg/l among the four zones and for LIS as a whole was 14.95 µg/l (Table 37). All stations with the exception of one in the Eastern zone and two in the Narrows have concentrations within the Moderate (5 – 20 µg/l) category. Concentrations at one station in the Eastern zone are within the Low, and at two stations in the Narrows are within the High category. Spatial coverage determined from the number of stations with concentrations at Moderate or above, was High for all zones. The frequency of occurrence is considered periodic since there is typically a winter spring bloom in this estuary. These three characteristics (concentration, spatial coverage, frequency of occurrence) combined give a rating for CHL of High for all four zones with the overall system-wide rating determined as High.

Table 37: Chlorophyll a 90th percentile concentrations, Bottom dissolved oxygen 10th percentile concentrations in Long Island Sound zones and overall. Chlorophyll concentrations 5 – 20 µg/l are considered Moderate, > 20 µg/l are considered High. Bottom dissolved oxygen concentrations >0 – 2 mg/l are considered Hypoxic, concentrations >2 – 5 mg/l are considered Biologically Stressful (Bricker et al., 2003).

Zone	Surface Chlorophyll a (µg/l)	Bottom Dissolved Oxygen (mg/l)
	(90% percentile)	(10th Percentile)
Narrows	22.82	3.19
Western	13.6	4.2
Central	9.55	5.18
Eastern	5.53	6.84
Long Island Sound	14.95	4.28

Macroalgae: Macroalgae such as *Ulva* and *Gracilaria* species are used as a primary symptom indicators in addition to CHL since in some systems nutrient increases will result in macroalgal growth rather than increases in phytoplankton (e.g. Nobre et al., 2005). Time series measurements of macroalgae in LIS are lacking, except along the eastern CT shoreline (Keser et al., 2010). Since there are no published reports of frequent or extensive problems with macroalgal blooms (Lopez et al., 2014), the rating for macroalgae is No Problem. The rating for Primary symptoms, the average rating of CHL (High) and macroalgae (No Problem), is High.

Dissolved Oxygen: The 10th percentile of DO concentrations using monthly samples for 2008– 2012 ranged from 3.18 - 6.84 mg/L among the four zones, and for the Sound as a whole was 4.28 mg/L (Table 37). The Narrows and Western zones and the Sound as a whole have concentrations considered Biologically Stressful while concentrations in the Central and Eastern zones are considered as No Problem (>5 mg/l; Bricker et al, 2003). The three characteristics of bottom DO (concentration, spatial coverage, frequency of occurrence) combined give a rating of Moderate for the Narrows and Western zones, Low for the Central and Eastern zones, with the overall system-wide rating determined as Moderate.

Indices	Methods	Parameters	Rating	Level of expression	Index
Influencing Factors (IF) ASSETS: 3	Susceptibility	Dilution potential	High	Moderate	MODERATE
		Flushing potential	Low		
	Nutrient inputs		Moderate		
Eutrophic Condition (EC) ASSETS: 1	Primary	Chlorophyll *a*	High	High	MODERATE HIGH
		Macroalgae	No Problem		
	Secondary	Dissolved Oxygen	Moderate	High	
		Submerged Aquatic Vegetation	No Problem		
		Nuisance and Toxic Blooms	High		
Future Outlook (FO) ASSETS: 4	Future nutrient pressures	Future nutrient pressures decrease			IMPROVE LOW

ASSETS SCORE: POOR

Estuary Characteristics

Population (X 10^3) 2010	4,900
Nutrient loading (tN y^{-1})	50 X 10^3
Mean depth (m)	19.5
Mean tidal range (m)	1.9
Water residence time (d)	60-90

Main impacts

Dissolved oxygen
Nuisance/toxic blooms

Figure 51: Results of ASSETS eutrophication assessment Long Island Sound. (See text for details.)

Nuisance/toxic blooms: Nuisance and toxic blooms have negatively impacted LIS shellfish fisheries episodically since the 1950s. Blooms still occur, but only on an episodic basis lasting from days to weeks.

The brown tide organism, *Aureococcus anophagefferens*, a very small organism that occurs at very high densities has been detected in LIS since 1986 but has not bloomed densely in LIS since 1995. Dense blooms of this organism decrease water transparency, kill off seagrasses, and clog shellfish siphons causing shellfish to starve to death (Gastrich and Wazniak, 2002). Of note is that *Aureococcus* has been shown to be closely related to changes in dissolved organic nitrogen rather than inorganic nitrogen, which is typically related to growth and blooms of phytoplankton and dinoflagellates (Gobler et al., 2011; Glibert et al., 2007). *Aureococcus* is not considered a problem in the Sound at present, though there are blooms in LIS embayments.

The overall rating for nuisance/toxic blooms is determined to be Low because the various nuisance/toxic blooms that occur do so only on an episodic basis and for a duration of days to weeks.

and widgeon grass (*Ruppia maritima*) with eelgrass being dominant. Prior to the 1930s when disease caused an Atlantic coast-wide dieoff of seagrasses, with >90% losses in LIS, eelgrass was prevalent throughout the shallow areas of the Sound (Lopez et al., 2014). By the 1950s, eelgrass had re-established beds in the eastern Sound, however, in the western Sound regrowth was not successful and eelgrass beds continued to suffer additional losses during the 1980s and 1990s. These losses have been attributed to effects of nitrogen loading and seagrasses have been effectively eliminated in the western Sound (Lopez et al., 2014). Recent studies were conducted in LIS to evaluate changes in eelgrass (*Zostera marina*) beds since 2002 (Tiner et al., 2003, 2007, 2010, 2013). The studies used true color aerial photographs and field surveys to delineate and compare areal limits and density of *Zostera* beds. The 2012 survey located 240 eelgrass beds in eastern LIS totaling 2,061 acres with >75% being of moderate to high density (Tiner et al., 2013). An additional 80 beds (~584 acres) of undetermined submerged aquatic vegetation were also identified. Compared with the 2009 survey, eelgrass in this portion of LIS had increased by 83 acres. The largest gains occurred in the Fishers Island and Niantic Bay sub-basins (+57 and +32 acres, respectively). Quiambog Cove and Little Narragansett Bay sub-basins experienced the most losses (-19 and -15 acres, respectively). Though there were losses as well as gains in seagrass acreage from 2002 to 2012, the balance was an increase in seagrass spatial coverage of 343 acres, a gain of about 30% (Tiner et al., 2013).

The rating for seagrasses is Moderate due to the combination of significant historical losses and complete elimination in the western sound, while recognizing recent gains.

Secondary symptoms receive an overall system-wide rating of Moderate based on the precautionary approach which selects the worst case (Dissolved Oxygen) of the three secondary symptoms (Figure 51).

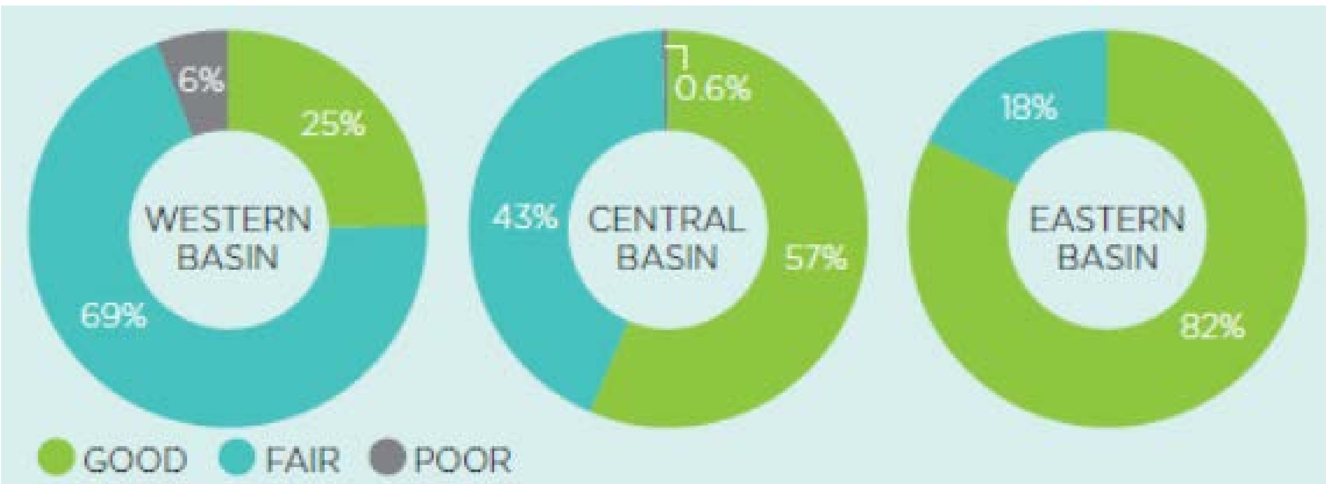
The Eutrophic Condition rating for this system is Moderate High based on High primary and Moderate secondary symptom expression ratings. These results are fairly consistent with the results from the Sound Health 2012 report (LISS, 2012) for the Water Quality Index (Figure 52), though the method varies slightly from the ASSETS model. The Water Quality Index uses nutrient concentrations and does not use macroalgal, nuisance/toxic blooms and changes in seagrass. The ASSETS assessment uses biological indicators, while nutrient concentrations are not used. However, both methods showed a gradient in water quality with highest quality in the Eastern Basin and lowest in the Western Basin. The Report also noted concerns about the effects of nutrient inputs on water quality and habitats in the estuary.

Future Outlook

Susceptibility: The susceptibility of LIS is Moderate (see discussion in Influencing Factors section above) which means the system is somewhat sensitive to increases in nutrient loads.

Future nutrient inputs: In response to observed nutrient related water quality degradation, in 2000 the states of New York and Connecticut developed a Total Maximum Daily Load (TMDL) plan to reduce nitrogen loads by 58.5 percent by 2017 (NYSDEC and CTDEP, 2000; CT DEEP, 2001; NYC DEP, 2000). The TMDL specifies the allowable pollutant loading from all contributing sources (e.g., point sources, non-point sources, and natural background) that can be discharged to LIS while still allowing attainment of water quality standards. In the 1990s, the Study determined that the largest single source of nitrogen came from WWTPs. The TMDL targeted the WWTPs requiring upgrades to biological nutrient removal (BNR) which greatly increases the removal of nitrogen. In 2012, these facilities discharged 35 million fewer pounds of nitrogen to the Sound in 2012 compared to 1990, 83 percent of the final reduction goal (Figure 53). Nitrogen loads from these sources will continue to decrease. Additionally, atmospheric deposition and agricultural nutrient loads have been decreasing across the watershed since the early 2000s and the trend is expected to continue (LISS, 2013).

Figure 52: Water Quality Index - The index includes five water quality component indicators—dissolved inorganic nitrogen, dissolved inorganic phosphorus, chlorophyll a, water clarity, and



dissolved oxygen. Monthly data (from May to October, when pollution has the greatest effect on water quality) are summarized Soundwide from 1991 to 2011 (adapted from: LISS, 2103).

While these drivers are expected to continue to decrease, other drivers may increase pressures on LIS water quality in the future. For example, future population changes such as the expected ~10% increase in population in CT and NY coastal counties between 2005 and 2025 (Woods and Poole Economics, Inc., 2007) will lead to continued development in the watershed. This may result in increases in septic system nutrient loads and loads from impermeable surfaces that are already known to be an issue. A land cover change analysis in the lower watershed confirms that development is increasing (CLEAR http://clear.uconn.edu/projects/landscapeLIS/index.htm). These changes could counter-balance the expected improvements in WWTP and non-point source reductions but the overall expectation is that loads will decrease in the future.

The combination of decreases in nitrogen load and Moderate susceptibility results in a Future Outlook rating of Improve Low.

Overall ASSETS Rating: The combination of Moderate influencing factors, High eutrophic conditions and Improve Low future outlook gives an overall rating of Bad according to the standardized ASSETS methodology.

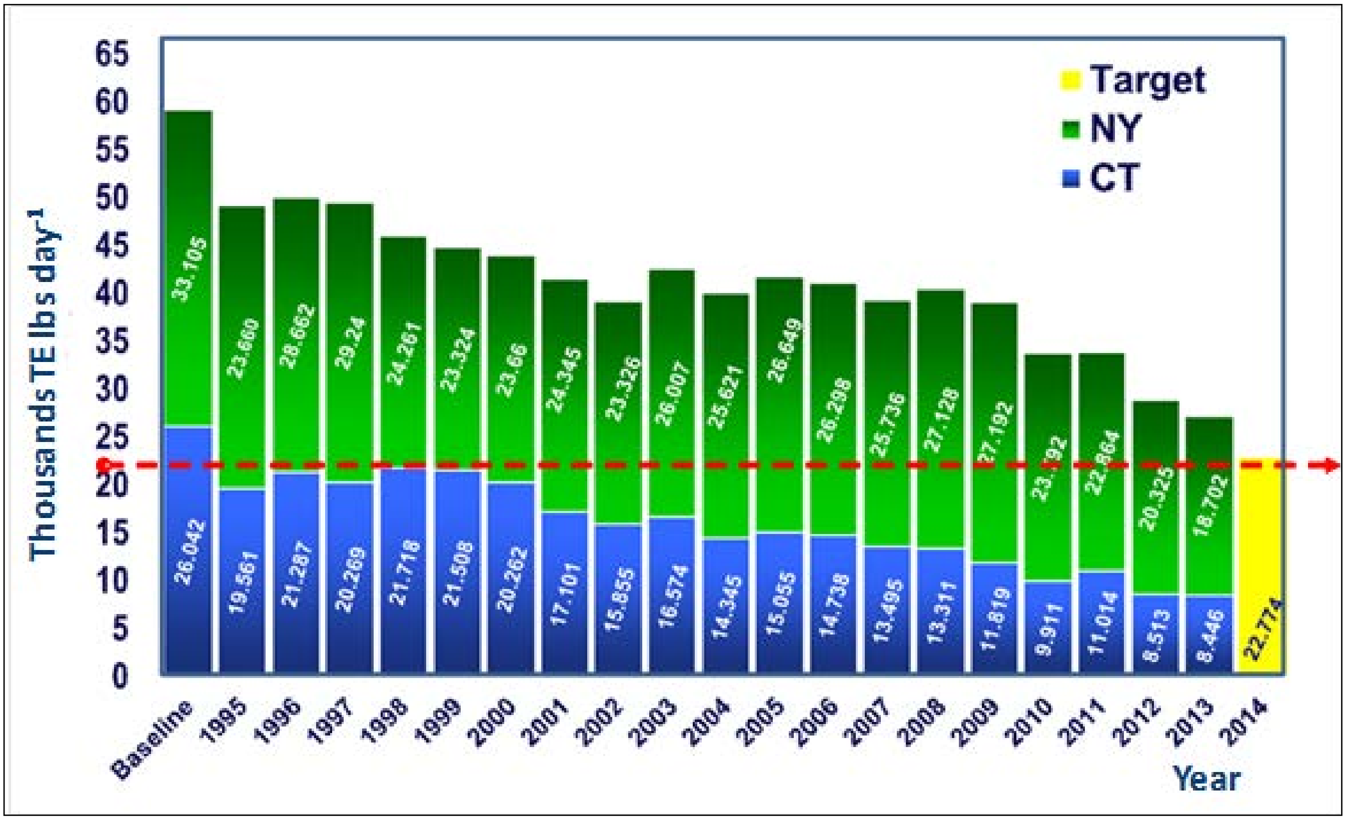


Figure 53: Comparison of study results to previous eutrophication assessment results (from Bricker et al., 1999, 2006, 2007, Ferreira et al., 2007).

Comparison to previous studies in Long Island Sound

Results here show that eutrophication conditions in LIS have improved since previous eutrophication assessment studies, likely the result of declines in nutrient loads (Figure 54). While CHL conditions have remained the same, DO conditions have improved from High to Moderate impact since the early 2000s. Overall eutrophic condition has improved from the High to Moderate High category. The increase in acreage of seagrasses and the continuing declines in nutrient loads from WWTPs and nonpoint atmospheric and agricultural loads are encouraging signs that management efforts are working and suggests that conditions will improve in the future.

Year	Nut Load	IF	CHL	Macro	Primary	DO	SAV	Nuis/Tox	Secondary	EC	FO
1991	M	M	H	No Data	H	M	H	No Data	H	H	IL
1999	M	M*	H	H	H	M	No Prob	L	M	H	NC
2002	M	M	H	No Data	H	No Prob	No Data	No Data	L	M	IL
2007	M+	M+	H	No Prob	H	H	L	L	H	H	IL
This Study	M	M	H	No Prob	H	M	No Prob	L	M	MH	IL

*, + indicate changes to ratings from previous studies:
*Bricker et al., 1999 results were modified from High IF to Moderate since susceptibility was reported as High when it is Moderate thus IF should be Moderate as shown here
+ Bricker et al., 2007 results were modified from High to Moderate Nut Load and from High to Moderate IF since High loads should have been noted previously as Moderate, as shown here
Nut Load = nutrient load, IF = Influencing Factors, CHL = Chlorophyll a, Macro = Macroalgae, Primary = Primary symptoms, DO = dissolved oxygen, SAV = submerged aquactic vegetation (seagrass), Nuis/Tox = Nuisance and Toxic blooms, Secondary = Secondary symptoms, EC = Eutrophic condition, FO = Future Outlook., L = Low, M = Moderate, H = Nigh, No Prob = No Problem, WH = Worsen High, WL = Worsen Low

Figure 54: Comparison of study results to previous eutrophication assessment results (from Bricker et al., 1999, 2006, 2007; Ferreira et al., 2007).

Economic Analysis: Value of Nitrogen Removal

In this section we use the replacement-cost method to assess the value of nitrogen removal through bioextraction. This method assumes that the costs of restoring a part of the ecosystem—in this case, clean water through nitrogen removal technologies of wastewater and agricultural BMPs, provides a useful estimate of the value of the ecosystem services of nitrogen removal by bioextraction. The use of the replacement-cost method assumes that if shellfish are no longer harvested, the nitrogen removal services they have been providing would need to be replaced. In this case, water quality regulations would require that the ecosystems services would be replaced. At present, WWTP upgrades and agricultural BMPs are the most likely candidates.[15]

As indicated by Gren et al. (2009) the value of shellfish aquaculture as a nitrogen removal device is estimated by taking the difference in minimum total costs for predetermined nitrogen targets in the watershed with and without the inclusion of shellfish farms. In this case, the magnitude of the value of shellfish aquaculture production is determined not only by its marginal cost in relation to other abatement measures (e.g., WWTPs), but also by its cleaning capacity. It is clear that marginal costs increase rapidly at higher reduction levels due to the need to implement high cost abatement measures in order to meet the predetermined nitrogen targets. In the case of LIS and GBP, where aquaculture facilities already exist and the costs of production are a given (and offset by sales of product by the firms), the value of bioextraction is equivalent to the minimum total cost without shellfish production (or the costs of WWTP, agriculture and/or urban BMPs).

Nitrogen removal estimates from bioextraction in Long Island Sound using EcoWin model simulations are discussed in greater length in the section 'EcoWin.Net: Scenario Analysis'. The EcoWin model separated LIS into six different areas or boxes, of a total 42, in which shellfish are cultivated. The standard

[15]Data indicate that controls on nitrogen loads through the use of urban non-point sources are significantly more costly.

scenario (which includes an estimate of 5,250 acres in shellfish production) nitrogen removal results of the EcoWin modeling are presented in Table 37.

Table 38: EcoWin Standard Scenario Modeling Results, Long Island Sound.

Standard (5,250 acres)	Box 23	Box 25	Box 27	Box 30	Box 33	Box 41	Totals
Net N removal (kg N y^{-1})	14,351	483,891	6,341	100,823	7,797	43,173	656,377
Net N removal (lb N y^{-1})	31,611	1,065,839	13,968	222,078	17,175	95,094	1,445,764
Acres	105	4,043	53	788	53	210	5,250
1Net N removal (lb N acre-1 y^{-1})[16]	301	264	266	282	327	453	275

In addition to the standard scenario, EcoWin modeling also estimated low, high, and potential scenarios. These results are reported in Table 30. Long Island Sound cost estimates for abatement technologies and best management practices are summarized in Table 39.

Table 39: Annualized Cost per Pound per Year for All Nitrogen Removal Alternatives (2013$).

WWTF 8 mg/l	WWTF 5 mg/l	WWTF 3 mg/l	Agricultural BMPs	Urban BMPs
14.63	16.82	44.81	5.90	159

Source: Evans, 2008 with Northern Economics, Inc. analysis

Applying cost estimates in Table 39 to the EcoWin modeling results, we can calculate the costs avoided in terms of alternative nutrient abatement options due to biotechnology (Table 40). As aforementioned, these values can alternatively be viewed as the savings in costs from using bioextraction, compared to more costly alternative nutrient abatement options.

We estimate that the annual cost to replace the removal of nitrogen through biotechnology to range between $8.5 million and $230.3 million per year (depending on the abatement technology considered) under the standard acreage scenario. Under all acreage scenarios, the costs range from $0.7 million under the low scenario, to $469.3 million per year under the potential acreage scenario. In addition to the total value of nitrogen removal in LIS through bioextraction, the weighted average value per acre per year is calculated for each scenario and nitrogen removal methods, ranging from $1,565 to $45,648. As shown in Table 39, as the number of acres increase, the average value for each effluent nitrogen level decreases. For example, under the low scenario (440 acres) at the 8 mg/l level, an average value per acre per year is $4,192. If we move to the standard scenario (5,250 acres) under the same 8 mg/l level, the value per acre decreases to $4,028.

[16] The whole acreage is used, rather than the annual seeded acreage, because bioextraction is evaluated as a contribution of all year classes.

Table 40: Average Biotechnology Nitrogen Removal Value: Long Island Sound (2013$).

Scenario	Level	Box 23	Box 25	Box 27	Box 30	Box 33	Box 41	Total Value ($/yr)	Weighted Average ($/acre/yr)
Low (440 acres)	8mg/l	40,274	1,370,320	17,622	278,579	21,206	116,585	1,844,579	4,192
	5mg/l	46,313	1,575,763	20,264	320,345	24,386	134,063	2,121,124	4,821
	3mg/l	123,371	4,197,648	53,981	853,361	64,960	357,129	5,650,427	12,842
	Ag. BMP	16,256	553,087	7,113	112,440	8,559	47,056	744,506	1,692
	Urb. BMP	438,537	14,921,000	191,880	3,033,363	230,909	1,269,454	20,085,058	45,648
Standard (5,250 acres)	8mg/l	462,393	15,591,135	204,318	3,248,502	251,225	1,391,035	21,148,285	4,028
	5mg/l	531,717	17,928,612	234,950	3,735,529	288,890	1,599,583	24,318,909	4,632
	3mg/l	1,416,431	47,759,724	625,879	9,951,011	769,568	4,261,103	64,782,727	12,340
	Ag. BMP	186,630	6,292,873	82,466	1,311,156	101,399	561,447	8,535,842	1,626
	Urb. BMP	5,034,859	169,767,149	2,224,754	35,371,954	2,735,515	15,146,555	230,277,273	43,863
High (7,648 acres)	8mg/l	660,888	22,259,458	294,118	4,684,300	364,733	2,026,541	30,290,037	3,961
	5mg/l	759,970	25,596,672	338,213	5,386,586	419,416	2,330,366	34,831,224	4,554
	3mg/l	2,024,472	68,186,538	900,961	14,349,235	1,117,274	6,207,824	92,786,304	12,132
	Ag. BMP	266,747	8,984,332	118,712	1,890,671	147,213	817,949	12,225,623	1,599
	Urb. BMP	7,196,207	242,376,490	3,202,564	51,005,921	3,971,472	22,066,386	329,819,040	43,125
Potential (11,116 acres)	8mg/l	934,975	31,555,493	420,836	6,714,744	527,618	2,946,000	43,099,893	3,877
	5mg/l	1,075,149	36,286,401	483,929	7,721,442	606,721	3,387,674	49,561,577	4,459
	3mg/l	2,864,072	96,662,724	1,289,131	20,569,017	1,616,233	9,024,367	132,026,242	11,877
	Ag. BMP	377,373	12,736,384	169,857	2,710,196	212,957	1,189,060	17,395,919	1,565
	Urb. BMP	10,180,656	343,598,201	4,582,358	73,114,815	5,745,077	32,078,096	469,301,681	42,218

Source: Northern Economics Analysis 2014

Great Bay Piscataqua Estuaries

Data used for ASSETS and FARM in Bay/Piscataqua Region Estuaries

In general, we did not find any significant trends at any station for any of the parameters examined for the years 2005-2010. At specific locations for specific water quality variables (e.g. river stations NH-0057, GBRLR, GBRCL for salinity) the statistical test showed very small trends that on closer analysis were seen to be driven by anomalous observations. All data for those years were used for the application of the ASSETS eutrophication model to assess nutrient related water quality within GBP. The results of the trend analysis for CHL were in agreement with an analysis performed by the Piscataqua Region Estuaries Program that showed no trends from 1975 – 2010, nor any short term changes during the past three years (PREP, 2013). Additional to these data, seagrass areal trends, occurrence of macroalgal and of nuisance/toxic algal blooms, and other relevant information for the assessment were acquired from on-going and previous studies (e.g. NH DES 2009, 2010; NHEP 2007; PREP, 2013).

Plots of the annual cycle for temperature, CHL, DO, show typical patterns seen in the Northeast US with temperatures lowest in the winter months (January-March) and highest in summer months of June - August, consistent with temperatures on land (Figures 9 to 11 shows station GRBAP as an example, other stations only sample April-December). In many estuaries in the Northeast region of the US an annual cycle is observed where CHL concentrations peak in February or March during the winter-spring bloom period and may also peak in the fall with the fall turnover of the system. This pattern is seen in CHL concentrations in some of the stations but the pattern is highly variable from year to year (Figure 10). It is possible that CHL concentrations in all other stations are low since they are not sampled during the typical timing of the winter spring bloom and thus may miss the highest concentrations of CHL.

Dissolved oxygen concentrations measured in surface grab samples were more consistent, showing highest concentrations in the cooler months, lower during the summer period (Station GRBAP as an example; Figure 11). While DO concentrations decreased during the summer, the percent saturation typically did not decrease as much suggesting that observed declines are partially caused by the increase in temperature, in addition to oxygen consumption from other processes, which lessens the amount of oxygen that can be dissolved. Station GRBAP shown in Figure 4 did not show any concentrations or saturations below state water quality thresholds. However, the DO concentration fell below the state water quality standard (5 mg L^{-1}) in a few surface grab samples (16 of 360 observations [4%] among the 5 stations in the 6 year dataset) collected during the months of June – September at stations GRBCL, GRBLR, GRBOR, GRBSQ, and NH-0057A, causing concern that biological stress from low DO may occur at some times and in some areas of the bay. Note that the low DO concentrations and low % DO saturation were observed only in June – September and only in riverine stations.

These results are consistent with findings in the State of the Estuary report which shows that state standards for DO are nearly always met in the large bays and harbors but are not met in some tidal rivers sometimes for periods of several weeks during the summer (PREP, 2013). Note that the state DO concentration threshold (5 mg L^{-1}) and the ASSETS threshold for biologically stressful category of DO impairment (>2 to 5 mg L^{-1}) is the same.

AquaShell and FARM model parameterization

Figure 55 shows the driver data used for the AquaShell and FARM models. Temperature at station GRBAP varied from 1.2 ^{o}C in the winter to 21.9 ^{o}C in the summertime. Salinity varied from 15.9 – 30.2 psu with lowest values in March – May, corresponding to months of highest rainfall. Annual averages are shown in Table 33 and show that salinity is lower than in LIS but Particlate Organic Matter (POM) is much higher in GBP than in LIS. The AquaShell individual oyster growth model was run to determine the weight of an 86mm (3.4 inch) ‘regular’ harvestable size oyster in GBP. AquaShell results give a weight of 67g for an 86mm oyster which compares favorably with measured weights of 70g reported by Grizzle et al. (2014). While there is no legal harvest size in New Hampshire, three – four inch oysters are typical market size. For our modeling GBP we use the harvest size of 86mm (3.4 inch) of 70g as reported for a typical 'regular' size oyster. We use a starting seed height of 10-15mm (0.5 – 0.65g) of hatchery derived spat (R.Grizzle, University of New Hampshire, pers. comm.). These data (Figure 55) were used as drivers for the FARM model with local aquaculture practices (Table 41).

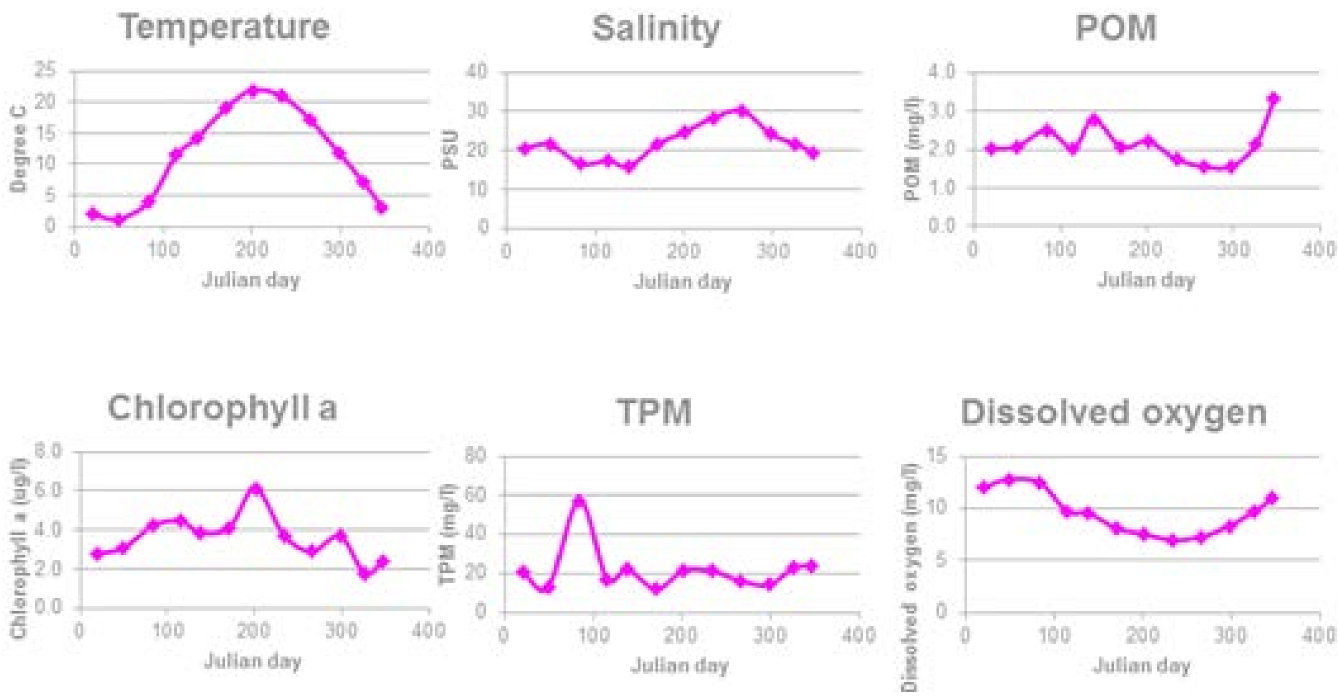


Figure 55: Environmental data used to drive the AquaShell individual model FARM model for simulations of aquaculture production and environmental effects (i.e. removal of nutrients). The data are averages for 2005-2010 at station GRBAP (see Figure 1 for location; Table 33 for annual average and comparison to LIS stations).

FARM model culture practice

The FARM model was applied to the data from station GRBAP (Figure 4) using culture practices shown in Table 41. Two model runs were executed with all input drivers the same except for current speeds; current speeds from Granite State were used in one simulation and from Choice Oyster in the other. The model was run using seeding density of 100 oysters m^2, which is typically used by growers in this estuary, spat of 10-15mm height and average of 0.58g starting weight, 50% mortality per cycle, a minimum harvest weight of 70g, for a 1095d cycle. Oyster farmgate price of $0.60 each was used to calculate harvest value (Table 41).

Table 41: Culture practice data for Great Bay/Piscataqua Region Estuaries used in the FARM model simulations.

Great Bay/Piscataqua Region Estuaries Shellfish cultivation practice	
Species	*Crassostrea virginica*
Cultivation practice	**Bottom culture with gear**
Seed density (number oysters/ sq meter)	**100***
Seed size (from hatchery) mm (g)	**10-15mm; 0.5 – 0.65g +**
Cultivation period (d)	**1095 (day 1 = May 1)**
Harvest size mm (g)	**76 – 102 mm for 'regular' (average 86mm ~70g) +**
Seed price	**$40/1000 = $ 69 kg^{-1} for 0.58g seed**
Farmgate price	**$0.50 - $0.70 ea ($0.60 ea = $8.57 kg^{-1} based on 70g oyster at harvest)**
Mortality (% per cultivation cycle)	**50**
Environment	
Peak current at spring tide (m s^{-1})	**0.60 (Granite State) #** **0.29 (Choice Oyster) #**
Peak current at neap tide (m s^{-1})	**0.47 (Granite State) #** **0.19 (Choice Oyster) #**
Nitrogen loading (metric ton y^{-1})	**1113 (TN), 542 (DIN) @**

**Calculated from seeding density 400 x 10^3 over 1 acre (R. Grizzle, personal communication)*

+ from Grizzle, Ward, Peter, Cantwell, Katz, Sullivan. 2014. Growth, morphometrics and nitrogen and carbon content of farmed Eastern oysters (Crassostrea virginica) in New Hampshire, USA. Final report to National Oceanic and Atmospheric Administration, National Ocean Service, NCCOS, Silver Spring, MD 20910

 # personal communication from R. Grizzle, measured at location of existing oyster farms in GBP

@ From PREP, 2013

Farm Aquaculture Resource Management model (FARM)

The two model simulations, using different current speeds, showed no significant difference in results; harvest for culture period of 1095 days (3 years) on a one acre farm was about 20 metric tons of oysters with nitrogen removal through filtration of about 0.35 ton N y^{-1} (Figure 56) at both sites. This removal represents an equivalent ecosystem service of nutrient treatment for 105 people acre-1 y^{-1} with an ecosystem service equivalent value of $4,200 acre-1 y^{-1} which would be additional income to the farmer if there were a nutrient trading program in place.

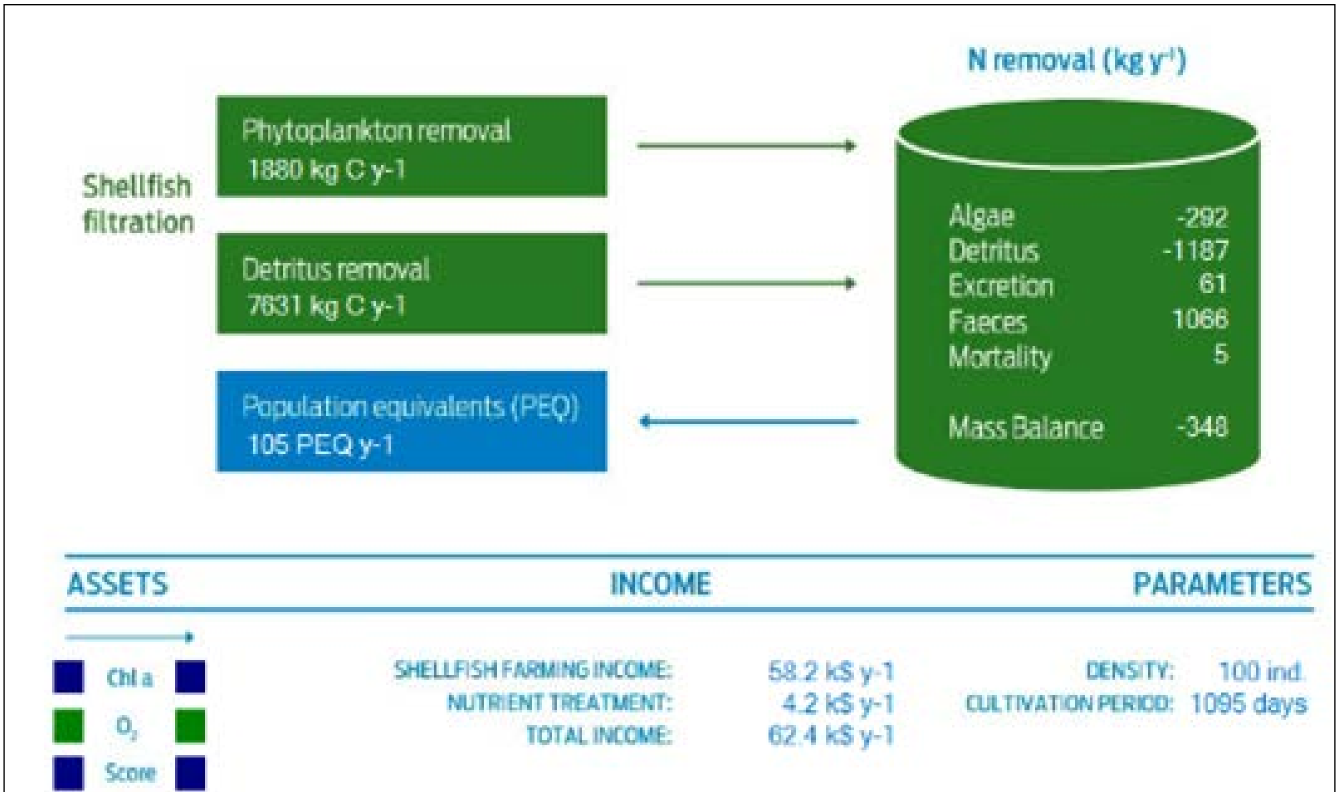
The FARM model shows that CHL concentrations decreased slightly (90th percentile changed from 4.86 to 4.83 ug L^{-1}). DO concentrations did not change at all (10 percentile 7.21 mg L^{-1} both inflow to and outflow from farm). The ASSETS rating did not change for either CHL or DO (Figure 56). The small changes suggest that there is no impact on water quality from the aquaculture operation and there is margin for additional density of seed.

This is a local scale simulation showing N removal of 0.35 ton acre-1 yr^{-1} which compares favorably with removal in other places and various other species (Rose et al., 2014a). However, note that the per acre harvest, nitrogen removal and thus the ecosystem service value of that removal are all greater in the simulated GBP farm than for the LIS simulated farms (Table 34). There are several things that might explain this difference. First, average annual temperature and average annual salinity are favorable to oyster growth in both places, and current speeds are sufficient to assure there is no local scale food depletion (Table 32, Table 40). However, Particulate Organic Matter (POM) is a little higher and Total Particulate Matter (TPM) is almost 3 times higher in GBP (Table 33), which means there is more food available, despite the lower CHL concentrations. Additionally, the higher seeding density in GBP (100 oysters m^{-2}) than in LIS (62 oysters m^{-2}) also likely to contributes to the higher production and nutrient removal capacity observed in these simulated local scale results (Figure 50 and Figure 56). While a marginal analysis has not been done for GBP, these results confirm the conclusions for LIS that there is margin to increase the seeding density which would increase both production and nutrient removal with no detrimental impacts to water quality.

The results of the FARM model are for farmscale only, in this case for 1 acre. It would be more useful to calculate the total removal for the whole GBP based on present aquaculture acreage and also to estimate potential significance if the GBP aquaculture industry were to expand.

Upscaling FARM model results in Great Bay/Piscataqua Region Estuaries

We have used the per acre local scale results to estimate the significance of removal by oyster filtration on the 25.5 acres that are presently in cultivation within GBP. For potential expansion, an analysis was done to evaluate the suitable bottom areas in GBP using a GIS based approach that excludes unsuitable area in the manner of Silva et al. (2011). Nash and Elliott (2012) show that the presence of eelgrass and legal constraints imposed by the Great Bay National Estuarine Research Reserve Program boundaries eliminates most of GBP as potential cultivatable area. Water depth suggests that there are 800 acres of suitable bottom area but shellfish closures, mooring fields and existing farms further restrict suitable areas with a total potential aquaculture area identified as ~413 acres. Given the 50m buffers between farms used by Nash and Elliot (2012), there are 36 hypothetical 4 acre farms (144 acres) or 62 hypothetical 3 acre farms that could be located within the suitable area and would be useful for upscaling the localscale results (Figure 57). However, upscaling requires some caveats. Since most farms are 4 acres, we use the scenario with 4 acre farms and the assumption is made that unidentified constraints eliminate one half to three quarters of the 36 hypothetical farms, thus we evaluated scenarios including expansion to an additional 9 – 18 farms. We further assume for the exercise that oyster growth and nitrogen removal rates would be the same for all suitable bottom areas. Finally, we assume that only 50% of each farm is harvestable at any time given typical aquaculture practices where there are several year classes of oysters growing at the same time (R. Grizzle, University of New Hampshire, pers. comm.).



			Ecosystem service equivalent (for nutrient removal)	
Farm Location	Oyster harvest (metric ton acre^{-1} yr^{-1})	N removed* (metric ton acre^{-1} yr^{-1})	PEQ (acre^{-1} yr^{-1})	Value of service (\$10^{3} acre^{-1} yr^{-1})
Granite State	6.8	0.348	105	4.2
Choice Oyster	6.8	0.355	108	4.3

Figure 56: FARM simulation results for simulated farm at GBP Granite State farm site for cultivation of a 1 acre farm; oyster harvest, N removed, People Equivalents (PEQ) and ecosystem value of nutrient removal by oyster filtration at the simulated site. Estimate of physical carrying capacity

Upscaled results show that present oyster cultivation (25.5 acres; 12.8 acres in cultivation) removes an estimated 8.9 metric t N y^{-1} or 0.73% of the total input to GBP (Table 42). Upscaling to the maximum acres of potential and existing farms (97 acres; 48.4 acres in cultivation), assuming that all areas would have the same production and bioextraction capabilities, would lead to an annual N removal through shellfish harvest of ~34 metric t N y^{-1}. This is equivalent to an ecosystem service of nutrient treatment for 10,238 PEQs with an ecosystem service value of \$410 x10^{3}. This represents a maximum removal of 2.79% of the total N load (Figure 58). It would take about 3,500 acres in active oyster cultivation, or 7,000 farm acres (since only 50% of a farm is cultivated) which is equivalent to ~28 km^{2} bottom area to remove the total input to GBP at the estimated rates of removal. This is an area equal to nearly 65% of the bottom area of GBP. It is possible that harvest and N removal would be greater with lower mortality or higher seeding densities, though higher seeding densities might also cause depletion of DO if over loaded.

- Hypothetical four-acre farm
- 50m buffer: 14.2 acres
- 36 (buffered) farms
- 144 acres harvestable area
- Likely an overestimate
- depth, substrate, others
- Assume unidentified constraints eliminate half of the 36 farms
- 9-18 4 acre farms

Source: Nash and Elliott, 2012

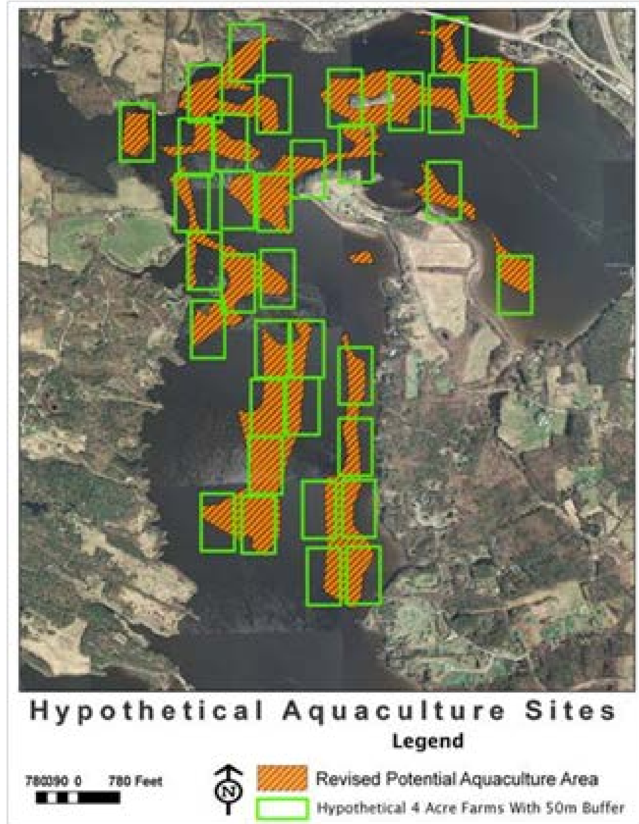


Figure 57: Potential expansion of aquaculture sites in Great Bay/Piscataqua Region Estuaries (Nash and Elliott, 2012)

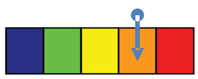
Table 42: Results of upscaling farmscale FARM model results to present and potential expanded aquaculture in GBP

Upscaling N removal in Great Bay/Piscataqua Region Estuaries				
	Acres	**N removed (metric tons)**	**% of total input**	**People Equivalents**
Simulated one acre farm	1	0.35	0.03	105
Present acres	25.5 (12.8 acres in cultivation)	8.9	0.73	2,678
Future acres (minimum = present acres + additional 36 acres)	61.5 (30.8 acres in cultivation)	21.5	1.76	6,458
Future acres (maximum = present acres + additional 72 acres)	97.5 (48.8 acres in cultivation)	34.1	2.79	10,238

While a seemingly promising alternative management measure, removal of nitrogen through cultivation of oysters at typical densities in GBP, even in the maximum potential expansion scenario, does not appear to be feasible as the only form of nutrient treatment in this waterbody. However, nitrogen removal by native oyster beds and oyster aquaculture can play a role in nitrogen reduction strategies given removal rates are similar to other BMPs as is implementation cost (Rose et al., 2014, 2014a).

Indices	Methods	Parameters	Rating	Level of expression	Index
Influencing Factors (IF) ASSETS: 1	Susceptibility	Dilution potential	Moderate	Moderate	HIGH
		Flushing potential	Moderate		
	Nutrient inputs		Moderate High		
Eutrophic Condition (EC) ASSETS: 2	Primary	Chlorophyll *a*	High	High	MODERATE HIGH
		Macroalgae	Moderate		
	Secondary	Dissolved Oxygen	Low	Moderate	
		Submerged Aquatic Vegetation	Moderate		
		Nuisance and Toxic Blooms	Low		
Future Outlook (FO) ASSETS: 4	Future nutrient pressures	Future nutrient pressures decrease			IMPROVE LOW

<u>ASSETS SCORE: POOR</u>



<u>Estuary Characteristics</u>

Population (X 10^3) 2010	287.7
Nutrient loading (tN y^{-1})	1225
Mean depth (m)	3.20
Mean tidal range (m)	2.38
Water residence time (d)	28.5

<u>Main impacts</u>

Macroalgae have replaced seagrass in some areas
Seagrass declines (35% area lost)

Figure 58: Results of ASSETS eutrophication assessment for Great Bay/Piscataqua Region Estuaries. See text for details.

Comparison of FARM results to removal estimated from tissue concentration and harvest

To get an independent estimate of the size of harvestable size oysters and associated removal of nitrogen in GBP, an estimate was made based on harvest estimates and nitrogen content of oysters. Typical oyster harvest is estimated to be 1,000 bags of oysters per acre per year where each bag weighs about 15 pounds. This is a total harvest of about 7 metric tons of oysters per acre per year, close to the results of the FARM application of 6.8 metric tons per acre per year. The size of harvestable oysters based on the 1,000 bags of 15 pounds each is 54 g, close to the estimated weight of a 76 mm oyster from the AquaShell simulation (53 g) but lighter than the 70g measured oysters (note that the 70g is for a 86mm height oyster [Grizzle et al, 2014]). The AquaShell simulation gives a weight of 70g for an 84mm oyster, close to Grizzle et al.'s (2014) measured values.

Based on results of a study of oysters in GBP Estuary, the percent nitrogen (N) in oysters varied from 6.9 to 8.4 %N in soft tissue and 0.07 to 0.18 %N in shell (Grizzle et al, 2014). The grand means for soft tissue N for regular size (3 inch, 7.62 cm) oysters were 7.5%; for shell N the grand means were 0.13% (7.68% total). Applying those percentages to the total harvest suggests removal of N based on tissue and shell concentrations of 1,152 lbs per acre per year (0.552 metric tons $acre^{-1}$ yr^{-1}). This estimate is greater than the estimate of the FARM model (0.35 metric tons $acre^{-1}$ yr^{-1}). There is good concurrence of the measured and modeled shellfish size and weights, suggesting that the FARM and WinShell models are simulating oyster growth in GBP reasonably well, however, the disparity between the N content requires close consideration.

Assessment of Estuarine Trophic Status (ASSETS) for Great Bay Piscataqua

The ASSETS evaluates eutrophication causes (Influencing Factors), conditions (Eutrophic condition) and forecasts future changes in nutrient related conditions (Future Outlook). The ASSETS model was applied using data and information described in 'Data used for ASSETS and FARM in Great Bay/Piscataqua Region Estuaries.'

Influencing Factors

Susceptibility: The susceptibility in GBP is determined as the combination of the ability to dilute and to flush nutrients. This is a well-mixed system, though there is some stratification at times of high flow, thus the calculation uses the volume of the estuary as the volume available for dilution of incoming freshwater and associated substances. The ratio of freshwater inflow to the estuary volume gives a rating for dilution potential of Low. The flushing potential is considered High due to the large tidal height and moderate to large ratio of freshwater to estuarine volume (Bricker et al., 1999). The combined flushing and dilution potentials give a susceptibility rating of Moderate.

Nutrient Input: Estuaries can receive nutrient inputs from land and ocean influxes, though typically human activities on land result in land-based inputs being much greater than oceanic inputs. The annual input of Total Nitrogen (TN) to GBP is 1,113 metric tons. While the preference is to use Dissolved Inorganic Nitrogen (DIN) for the calculation of nutrient load influences because it is believed to be more accessible for algal uptake than TN, here we have used TN for consistency with previous results when DIN data were not available. The calculation of the ratio of human versus oceanic influence on TN inputs in this system gives a rating of Moderate High (0.63) meaning that land based sources are the predominant source of nutrients to GBP (Bricker et al., 2003; Ferreira et al., 2007).

The combined Moderate susceptibility and Moderate High nutrient input in the ASSETS model, which groups the Moderate High inputs with High inputs, gives an overall Influencing Factor rating for this system of High (Figure 58).

Eutrophic Condition

The calculation of Eutrophic Condition was based on an analysis of the condition of primary (CHL, macroalgal abundance) and secondary (DO, nuisance/toxic bloom occurrence, changes in seagrass coverage) symptoms.

Chlorophyll a: Phytoplankton growth is directly stimulated by increases in nutrients and thus CHL, as a measure of phytoplankton biomass, is used as one of the first indications of nutrient enrichment. The 90th percentile concentrations of CHL for 2005 - 2010 were at Moderate levels in the mixing zone (10.0 $\mu g\ L^{-1}$) and were Low in the seawater (2.60 $\mu g\ L^{-1}$) zone. Each station in the mixing zone shows 90th percentile concentrations above 5 $\mu g\ L^{-1}$, the lower limit of the Moderate category, and thus the spatial coverage is considered High. When 90th percentile concentrations are Low, as in the seawater zone, the spatial coverage is assumed to be High. The frequency of occurrence is considered periodic since there is typically a winter spring bloom in this estuary. These three characteristics (concentration, spatial coverage, frequency of occurrence) combined give a rating of High in the mixing zone and Low in the seawater zone with the overall system-wide rating determined as High (Figure 59).

It is important to note that the concentrations in the mixing zone are moderate but the large spatial area over which these concentrations are observed and the seasonal periodicity gives this indicator a higher rating than might be expected if only concentrations were considered.

Year	Nut Load	IF	CHL	Macro	Primary	DO	SAV	Nuis/Tox	Secondary	EC	FO
1999	L	L	M	M	M	No Prob	No Prob	L	M	M	WH
2006	M+	M+	H	No Data	H	No Prob	No Prob	No Data	L	M	WH
2007	M+	M+	H*	H	H	No Prob	L	L	L	M	WH
This Study	MH	H	H	M	H	L	M	L	M	MH	IL

*, + these variables have been recalculated with new data and corrected, they were incorrect in the original documents, *CHL data for the 2007 NEEA report (Bricker et al., 2007) was reported incorrectly as Low. The 90th percentile value for the mixing zone is 10.4 ug L^{-1} putting it into the moderate category; with a spatial coverage of high and periodic frequency the rating is High for the mixing zone. The rating for the seawater zone is Low (4.0 ug L^{-1}) and the corrected rating for the Great Bay/Piscataqua Region Estuaries is High. (P. Trowbridge, N.H. Dept. of Environmental Services – now San Francisco Estuary Institute, pers. comm.)
+Nutrient loads reported in the 2006 Gulf of Maine report (Bricker et al., 2006) and the 2007 NEEA (Bricker et al., 2007) were incorrectly reported as Low, both based on incorrect nutrient load values for 2002-2004. Using the correct load, 1,113 metric tons TN y^{-1} (P. Trowbridge, N.H. Dept. of Environmental Services, pers. comm.), resulted in a change in rating from Low to Moderate. This also changed the Influencing Factor rating from Low to Moderate.
Nut Load = nutrient load, IF = Influencing Factors, CHL = Chlorophyll a, Macro = Macroalgae, Primary = Primary symptoms, DO = dissolved oxygen, SAV = submerged aquactic vegetation (seagrass), Nuis/Tox = Nuisance and Toxic blooms, Secondary = Secondary symptoms, EC = Eutrophic condition, FO = Future Outlook., L = Low, M = Moderate, H = Nigh, No Prob = No Problem, WH = Worsen High, WL = Worsen Low

Figure 59: Comparison of study results to previous eutrophication assessment results from Bricker et al., 1999, 2006, 2007.

Macroalgae: Macroalgae such as *Ulva* and *Gracilaria* species are used as a primary symptom indicator in addition to Chl since in some systems nutrient increases will result in macroalgal growth rather than increases in phytoplankton (e.g. Nobre et al., 2005). In GBP, spatial coverage of macroalgal blooms has increased in the past two decades, a cause for concern. For example, at one site in GBP, the mean percent cover of Ulva increased from 0.8% to 39% of area covered between 1979-1980 and 2008-2010 and covered an estimated 3% of the GBP surface in 2007 in areas previously occupied by seagrasses (PREP, 2013). Biomass of Ulva and Gracilaria collected at three sites in the mixing zone showed average biomasses exceeding 100 g dry weight per m^2 (Nettleton et al., 2011), a level that may cause dieoff of seagrasses (Bricker et al., 2003; Short, 2012). The rating for macroalgae in both GBP zones is Moderate, in agreement with an assessment by the PREP Technical Advisory Committee and with the State of the Estuary report (PREP, 2013).

The rating for Primary symptoms, the average rating of CHL (High) and macroalgae (Moderate), is High.

Dissolved Oxygen: The 10th percentile of DO concentrations, using monthly samples for 2005 – 2010, was determined for the mixing zone (6.0 mg L^{-1}; range of 5.16 – 7.0 mg L^{-1}) and for the seawater zone

(6.95 mg L^{-1}). All stations in both zones showed concentrations above the ASSETS biologically stressful DO threshold (5 mg L^{-1}; Bricker et al., 2003). This indicates a rating of No Problem. However, as noted, there are times of the year when DO is below state water quality thresholds shown both by the monthly sampling and the datasonde records (5 mg L^{-1}) suggesting that further, confirmatory analysis is needed.

Here we compare the results of the ASSETS determination using monthly records with analysis of datasonde records to further evaluate DO conditions within GBP. Although the ASSETS analysis of grab samples show no station with 10th percentile values below the state water quality threshold, the 2013 State of Our Estuaries assessment of datasonde measurements shows for stations in the Squamscott, Lamprey, and Oyster Rivers that there were DO concentrations below 5 mg L^{-1} at some point during the day on more than 10% of the days during the summer season for years with complete data records (PREP, 2013). There were also low DO events in the Salmon Falls River, but there were no years with complete data records at this station. DO concentrations less than 5 mg L^{-1} were rare in the middle of GBP and at the mouth of estuary in Portsmouth Harbor (Figure 4).

An additional analysis of the datasonde DO measurements where daily minimum DO concentration (mg L^{-1}) and daily average DO percent saturation were compiled for all days in 2005 – 2010. The 10th percentile concentrations were determined from these data for all stations. In summary, the 10th percentile of the daily minimum DO is less than 5 mg L^{-1} at one station in the mixing zone (GRBLR;). Considering the datasonde DO analysis within the ASSETS decision matrix, the observed biological stress (<5 mg/L), low spatial coverage (one of 5 datasondes), and periodic frequency, would indicate a DO rating of Low (Table 43).

Given this analysis, using a precautionary approach, the DO rating for GBP is Low (Figure 58).

Table 43: Comparison of DO assessment using monthly grab and datasonde observations with ASSETS criteria.

Station	ASSETS 10th percentile	Datasonde 10th percentile
GRBGB	6.8	6.9
GRBLR	5.9	4.0
GRBOR	5.5	5.4
GRBSF	Insufficient data	5.4
GRBSQ	5.2	5.3
All Mixing Zone	6.0	5.3
GRBCML Seawater zone	7.0	6.8

Nuisance/toxic blooms: The overall rating for nuisance/toxic blooms is determined to be Low because blooms, primarily Alexandrium spp. occur episodically only within the seawater zone and they do not originate in the estuary but are swept in from offshore (NH Department of Environmental Services Shellfish Program; Bricker et al., 1997). These blooms cause fisheries closures in offshore coastal waters and have been detected as far into the estuary as Fox Point, near the boundary of the seawater and mixing zones (Figure 1) during high bloom periods. The occurrence of these blooms in the seawater zone has limited the potential expansion of aquaculture to areas within the mixing zone where nuisance/toxic blooms do not occur (R.Grizzle, University of New Hampshire, pers. comm.)

Submerged Aquatic Vegetation: The 2013 State of Our Estuaries report shows that 35% of the eelgrass in GBP has been lost (PREP, 2013; Short, 2012). Further analysis shows that losses were 33% (loss of 850 acres) in the mixing zone and 45% (loss of 155 acres) in the seawater zone from 1996 to 2009-2011 (P. Trowbridge, N.H. Dept. of Environmental Services, pers. comm.; Short, 2012). The magnitude of these losses corresponds to a rating of Moderate.

Secondary symptoms receive an overall systems-wide rating of Moderate based on the precautionary approach which selects the worst case of the three secondary symptoms (Figure 58).

The Eutrophic Condition rating for this system is Moderate High based on High primary and Moderate secondary symptom expression ratings. This is consistent with the evaluation of the 2013 State of Our Estuaries report (PREP 2013) that noted concern about nutrient inputs potential detrimental impacts on water quality and habitats within GBP.

Future Outlook

Susceptibility: The susceptibility of Great Bay is Moderate (see discussion in Influencing Factors section above) which means the system is somewhat sensitive to increases in nutrient loads.

Future nutrient inputs: At present the total load of nitrogen to GBP is 1113 metric tons y^{-1} (542 metric tons as DIN) with 32% from WWTP and 68% from non-point sources (PREP, 2013). There are presently plans to upgrade the major WWTPs that discharge to GBP and two WWTPs that discharge to the GBP already have National Pollution Discharge Elimination System (NPDES; http://www.epa.gov/region1/npdes/permits/2012/finalnh0100871permit.pdf, http://www.epa.gov/region1/npdes/permits/2012/finalnh0100196permit.pdf) permits that will reduce nitrogen loading to the bay in the next 5-10 years. Nitrogen permit limits for other communities are being negotiated by EPA and the communities. Planned upgrades to WWTPs will significantly reduce the DIN load to the estuary. Additionally, many communities in the region are working to reduce non-point sources of nitrogen from the landscape. Education efforts to increase awareness and preventative actions of the public to help reduce nutrients from sources they can help control, such as reducing us of lawn fertilizers, have been shown successful in nearby Maine (e.g. Lawns to Lobsters, http://www.lawns2lobsters.org/) and are expected to make a difference in the GBP watershed as well (PREP, 2013).

Other conservation and restoration efforts such as restoration of salt marsh and eel grass beds and oyster reefs, will also contribute to reductions in nutrient loads (PREP, 2013).

Other future changes such as the expected >20% increase in population in NH and Maine by 2030 (Woods and Poole Economics, Inc., 2007) will lead to continued development in the watershed including increases in impermeable surfaces that are already known to be an issue (PREP, 2013). These changes could counter balance improvements in WWTP and some of the non-point source reductions, but the overall expectation is that loads will decrease.

The combination of decreases in load and Moderate susceptibility results in a rating Improve Low.

Overall ASSETS Rating: The combination of High influencing factors, Moderate High eutrophic conditions and Improve Low future outlook gives an overall rating of Poor according to the standardized ASSETS methodology.

Comparison to previous studies

The results here show Great Bay to have a greater nutrient related degradation than any of the previous eutrophication assessment studies (Figure 59; Bricker et al. 1999, 2006, 2007). Some of the observed differences are due to differences in condition but also to new and better data. For example, the new data shows a considerable increase in macroalgae abudance and losses of seagrass. As nutrient loads have increased, some eutrophication indicators have changed showing a greater level of nutrient related impacts. It should be noted, however, that though loads have increased over time there is a strong relation to rainfall and thus to non-point source runoff. High rainfall years show a higher level of impact making the trends difficult to discern due to the high variability. We have tried to reduce this variability and potential bias of high or low flow years by using combined 6 years of data. On an optimistic note, although eutrophic impacts are greater than in previous studies, expected improvements in the future is more hopeful than in the previous studies.

Economic Analysis: Value of Nitrogen Removal in Great Bay Piscataqua

There are currently 25 acres under cultivation in GBP and in 2013 it was estimated that the production was about 100,000 oysters per acre per year or approximately seven tons of oysters harvested per year (Grizzle, 2014) which is confirmed by results of the FARM model (Figure 56 and associated text). Based on Nash et al. (2012), GBP has the carrying capacity and zoning requirements to allow for approximately 72 additional acres to be added over time, potentially reaching 97 acres under lease (Figure 57, Table 42). Using the nitrogen removal estimate of 0.3 metric tons per year per acre (see section ‘ Great Bay/Piscataqua Region Estuaries: FARM model results’; Figure 56) we estimate the total pounds of nitrogen removed per year per acre to be 661 lbs. Using the cost estimates provided in Table 20, we estimate the nitrogen removal value at the present cultivation level to be between $1.12 million and $1.28 million per year (Table 44) depending on TMDL levels and assuming that the entire lease area is cultivatable. If GBP reaches its potential of 97 acres of lease and we assume that all of the lease acres are in production, we estimate nitrogen removal values to range between $4.3 million to $5.0 million per year. These values represent costs avoided in terms of alternative nutrient abatement options because of biotechnology. Please note that agriculture and urban BMPs are not included in the valuation summary for Great Bay. This is due to lack of data for these alternative abatement alternatives in the region. Note also that this assumes that all lease acres are under cultivation while we have previously assumed that only half of a lease is cultivated, thus these values may be maximum estimates.

Table 44: Average Biotechnology Nitrogen Removal Value: Great Bay (2013$)

Scenario	Level	Total Value ($ y^{-1})	Value per Acre ($ y^{-1})
Current (25 acres)	8 mg/l	1,278,545	51,142
	5mg/l	1,116,476	44,659
	3 mg/l	1,152,491	46,100
Potential (97 acres)	8 mg/l	4,960,755	51,142
	5mg/l	4,331,927	44,659
	3 mg/l	4,471,665	46,100

Source: Northern Economics Analysis 2014

The average value per acre per year for each of the nitrogen limits in GBP is the same for each scenario, with the per acre value ranging from $44.7 to $51.1 thousand. The value per acre in GBP is simply the total value divided by the number of acres.

CONCLUSIONS

Methods and Modifications

Individual Eastern oyster model: The AquaShell individual model for Eastern oyster was developed for incorporation into the EcoWin and FARM models using data from this studygrowth experiments in LIS collected as part of this project. Growth and nutrient content experiments were also conducted in GBP for comparison to LIS results and for validation of FARM model results in GBP. This project added an Eastern oyster model as a component of the EcoWin and FARM, which has modeled several other species in previous applications. The individual model was developed, calibrated and validated in this project. The individual Eastern oyster model worked well in both the EcoWin and FARM models, providing good reproduction of growth and estimated time to harvest size. This allowed modeling of nutrient removal at system and local scales with a high level of confidence.

SWEM and EcoWin and FARM models: The removal of nitrogen by oyster and clam aquaculture was estimated through the use of the two system scale models, SWEM and EcoWin, and by the local scale model, FARM. Note that the SWEM and EcoWin were run only for LIS and only for CT; GBP did not have a systemwide hydrodynamic model needed for EcoWin simulations and no data were available for aquaculture culture practices in NY.

The high resolution SWEM model is typically run for two years and does not include the aquaculture simulation. The EcoWin model includes aquaculture and is run for a decade which is a timeframe that allows for full development of the aquaculture simulation and is also of relevance to the economic analysis. The high resolution SWEM modeling grid was simplified into the coarser grid, and the vertical resolution limited to an upper and lower layer for EcoWin, reducing the number from 2,300 to 42 boxes (21 upper, 21 lower). The merging of boxes considered CT and NY state boundaries, morphology, currents and vertical stratification, and the location of aquaculture sites. SWEM was run in the reduced grid configuration and water volumes, fluxes and associated substances (e.g. salinity, nitrogen, carbon) were passed to EcoWin. The EcoWin model performance was calibrated and validated using salinity, temperature and CHL data. Successful reproduction of salinity, temperature and CHL conditions, compared to measured data, provided confidence in the use of EcoWin for the systemwide simulations.

EcoWin simulation including shellfish aquaculture in LIS: The EcoWin model requires shellfish aquaculture practices including the area of cultivation, seeding densities, bottom or floating, culture period, size of seed and harvest size oysters in order to reproduce the growth and nutrient removal through aquaculture activities. This proved difficult to characterize, with particular difficulty in determining the areas of cultivation, given the reticence of growers to provide information on their industry practices. A bracketing method was used whereby a lower and upper estimate for the cultivation area was determined by 1) calculation using known harvest and densities and 2) total known lease areas and areas cultivated based on interviews with growers. There is cultivation of both oysters and clams. Oysters were simulated explicitly. Clams were considered a wild species, though upper and lower areas of harvest were also determined since the filtration of clams adds to the removal capacity in the system. The mid-value of oyster cultivation acres was used as the standard model (5,250 acres) and scenarios were simulated using the upper and lower estimates of cultivation acres as well as potential expanded acres, for both oysters and clams (Table 6).

Nutrient related water quality

The eutrophic conditions in LIS and GBP were assessed using the ASSETS eutrophication model. Nutrient-related water quality degradation continues to be a serious issue in LIS and GBP (Figures 51, 58). Application of the ASSETS eutrophication assessment to LIS identifies water quality as impaired by nutrients, despite nutrient management efforts having decreased nitrogen inputs. In GBP, the ASSETS eutrophication assessment also identifies significant impairment. There is no discernable trend in nitrogen loads in GBP, and the variability appears to be weather dependent (i.e. in years of high runoff there is high loading). In both waterbodies the assessment indicates moderate to high level eutrophic impacts. Dissolved oxygen is a moderate level problem but algal blooms, indicated by CHL and by occurrence of nuisance and toxic algal blooms, are rated as a high level impact. In GBP, CHL is a high level impact, while macroalgal blooms are a moderate problem that have been increasingly killing off seagrasses.

Improvements in water quality are expected due to the LIS TMDL and the nutrient criteria requirements in GBP. Given they are both areas of active shellfish cultivation, there is the potential that nitrogen removal through this means could complement land-based management measures. In both LIS and GBP, our analyses show that there is capacity to increase shellfish aquaculture which would complement reductions in nutrient loading from land based sources.

The EcoWin results show that at the standard model conditions with stocking densities of shellfish below production and carrying capacity that there is already a significant change to CHL concentrations with a drawdown of almost 10% in western LIS. While this evaluation was specific to oysters and quahogs, the results suggest that water quality conditions would be improved by expansion of shellfish aquaculture.

Bioextractive nitrogen removal in LIS and GBP

N removal in LIS: Removal of nitrogen by oysters estimated by simulation in EcoWin ranged from 57 – 940 metric tons N per year for the low to high cultivation scenarios, and increases to 1,338 metric tons per year in the expansion scenario. This equates to 0.11 – 2% of total N inputs, and up to 3% at the expanded scenario (Table 45). Removal of N by quahogs ranged from 234 – 4,020 metric tons N per year, and increases to 5,639 metric tons per year in the expansion scenario. Compared to the total N inputs, the current range of removal is 0.5 – 8% of total inputs and 11% at the expanded scenario (Table 45). Together the oyster and quahog N removal could account for up to about 10% of total inputs at present and up to almost 14% in an expansion scenario. The combined removal seems very promising. While bioextraction appears higher for the quahoqs, the area occupied is substantially greater than the area occupied by oysters. Since this model approach using quahogs as wild species does not include natural mortality, removal may be overestimated, and there is no validated quahog model to provide confidence in the quahog growth and removal rates. Due to the lack of confidence in the quahog removal rates, we use only the oyster removal for the economic analysis.

The FARM model is local scale but given assumptions and caveats, the local scale LIS results were up-scaled to estimate removal of N on system wide basis. Table 45 shows that removal estimates are similar to results for the simulation by EcoWin for both the N removed and for the people equivalents (PEQs) represented by the removal. This provides some confidence that in areas such as GBP where there is no circulation and water quality model, that scaling up local values can be useful for estimating the impact to the system given appropriate assumptions and caveats.

N removal in GBP: Removal of nitrogen by oysters estimated by simulation in the FARM model was about 9 metric tons N per year at both farm locations, and increases to 34 metric tons per year in the expansion scenario. This equates to 0.73 % of total N inputs, up to almost 3% at the expanded scenario (Table 45).

Table 45: Impact of shellfish aquaculture on Long Island Sound and Great Bay/Piscataqua Region Estuaries nutrient inputs and ecosystem service value.

	EcoWin	EcoWin Mass Balance and FARM (upscaled) estimates	Cost based estimate	Production based estimate
N removal as % total load				
LIS 440 acres	0.11	0.09		
LIS 5250 acres	1.31	1.10		
LIS 7648 acres	1.88	1.6		
LIS 11116 acres	2.68	2.32		
GBP current	---	0.73		
GBP potential	---	2.79		
PEQ (1000s)				
LIS 440 acres	17	14		
LIS 5250 acres	199	167		
LIS 7648 acres	284	243		
LIS 11116 acres	405	353		
GBP current	---	1.0		
GBP potential	---	10		
Treatment value ($1000)				
LIS 440 acres	---	555	704 – 20,085	
LIS 5250 acres	---	6,622	8,535 – 230,277	267,400
LIS 7648 acres	---	9,647	12, 225 – 329,819	
LIS 11116 acres	---	14,021	17,395 – 469,302	545,000
GBP current	---	107	1,116 - 1,279	
GBP potential	---	410	4,332 - 4,961	

How important is N removal in LIS and GBP: While seemingly small (0.11 – 3% for oysters only), these estimated N removal rates in LIS (0.12 – 0.13 metric tons acre^{-1} y^{-1}) and in GBP (0.35 metric tons acre^{-1}

y^{-1}) are similar to those of other BMPs and may be more cost effective than some abatement alternatives (Rose et al., 2012, 2014a). As noted above, the removal by shellfish aquaculture is not insignificant compared to the expected reductions due to fully implemented agricultural and urban BMPS in the CT River Basin (NEIWPCC, 2008).

Note that the value of nutrient removal estimated here should be considered a low estimate of the total value of the ecosystem service since 1) we have not included the contribution from clam bioextraction, 2) it does not include the value attributed to certainty of sustainable domestic production of seafood, and 3) we have not estimated the value of services that accrue directly to people such as enhanced water quality and habitat for other organisms that indirectly or directly improve human well being.

Implications to nutrient management

In this study we estimated the monetary value of bioextraction technologies as measured by costs avoided. These values represent the costs of providing a substitute or replacement for the nutrients removed through biotechnology, that is, the value of bioextraction is the costs associated with alternative nutrient removal methods that are avoided or replaced by the shellfish aquaculture.

In the case of GBP, it would cost between $1.1 and $1.3 million (see Table 46) under the current scenario of 25 production acres to replace the nitrogen removal services through source controls. The cost of alternative nitrogen removal technologies depends on the method of nitrogen removal (abatement) chosen. As seen in Table 45, under the current scenario, upgrading WWTPs to 5mg/l would be the least costly of the upgrade alternatives. Therefore, it can be assumed that to maintain current water quality levels, users would be willing to pay this amount. It follows, that if shellfish farming reaches the potential scenario of 97 production acres, the cost to offset bioextraction would increase as the amount of nitrogen removed increases; the corresponding range of values is estimated to be $4.3 - $5.0 million.

A similar summary is shown for LIS in Table 4, note that as described, the value includes only the oyster cultivation and thus may be larger if clams were included. In the case of LIS, costs avoided could range from $8.5 to $230.3 million under the current acreage scenario. Under the potential acreage scenario costs could range between $17.4 and $469.3 million, again depending on the alternative abatement approach considered.

Table 46: Nitrogen Removal Summary (Note that because of the different methods of determination, the costs cannot be compared between Great Bay/Piscataqua Region Estuaries and Long Island Sound; see text section 'Costs of Abatement Technologies and Best Management Practices').

Scenario	Abatement Source	Great Bay/Piscataqua Region Estuaries			Long Island Sound		
		Total Value ($/yr)	Nitrogen Removed (lb/yr)	Value ($/lb/yr)	Total Value ($/yr)	Nitrogen Removed (lb/yr)	Value per AcreCost ($/lb/yr)
Current	8mg/l	1,278,545	16,525	77.37	21,148,285	1,445,764	14.63
	5mg/l	1,116,476	16,525	67.56	24,318,909	1,445,764	16.82
	3mg/l	1,152,491	16,525	69.74	64,782,727	1,445,764	44.81
	Ag. BMP	-	-	N/A	8,535,842	1,445,764	5.90
	Urb. BMP	-	-	N/A	230,277,273	1,445,764	159.28
Potential	8mg/l	4,960,755	52,219	77.37	43,099,893	2,946,387	14.63
	5mg/l	4,331,927	52,219	67.56	49,561,577	2,946,387	16.82
	3mg/l	4,471,665	52,219	69.74	132,026,242	2,946,387	44.81
	Ag. BMP	-	-	N/A	17,395,919	2,946,387	5.90
	Urb. BMP	-	-	N/A	469,301,681	2,946,387	159.28

Source: Northern Economics Analysis 2014

We also estimated the cost of nitrogen removal through calculation of costs of oyster production. However, this estimate is not comparable to the costs of abatement practices, as oyster production is not conducted for the sole purpose of nutrient removal (as it is in some countries). Instead the costs of production reflect the provision of ecosystems services as well as a food commodity. Because we were not able to collect data on revenues and costs from LIS shellfish growers, we used data from a survey of Rhode Island growers engaged in intensive oyster aquaculture (the practice of some of the LIS growers, though most practice extensive culture with no gear; see Appendix 2). This information in reported more formally in Rhode Island Intensive Oyster Aquaculture Economic Impact Analysis (Northern Economics, 2014). Nitrogen removal from the harvest of oysters in LIS was calculated using production and removal estimates from EcoWin modeling results. Using the Rhode Island survey data as a proxy for LIS we derived a per pound cost of oyster production of $3.70. Total oyster production costs for LIS were then derived using production estimates for LIS from the Connecticut Department of Agriculture (4,402,000 pounds in 2010). The costs were then divided by an estimate of 88,040 pounds of nitrogen removed through shellfish harvest. We estimate that the total annual costs of nitrogen removal in LIS, through shellfish production or biotechnology, to be $267.4 million for the current acreage scenario and $545 million for the potential acreage scenario at an annual cost per pound of nitrogen removed of $184.97.

The EcoWin provides a mass balance and economic analysis and the FARM model also calculates the value of N removal. Table 45 shows that for each case in both waterbodies the values are significantly under estimated compared to the cost and production based estimates. The FARM model uses a conservative literature value for the estimation of the costs avoided value (see section 'FARM Aquaculture Resource Management model (FARM)' and these results suggest that the value used should be increased to reflect higher values as demonstrated by this economic analysis.

Currently only WWTPs participate in the Connecticut Nitrogen Credit Exchange (NCE) which allows WWTP around the State to share in the costs and benefits of removing nitrogen from wastewater. Between 2002 and 2009 15.5 million credits were traded on the (NCE) representing $45.9 million in economic activity. The price of nitrogen credits has ranged from $1.65 to $5.01[17]. This is similar to the trading price of $13 per kilogram ($5.90 per lb) of N removed in the North Carolina nutrient offsets program (Piehler and Smyth, 2011).

Whether shellfish producers should be included in the CT River Water Quality Trading Program however remains a policy decision. We know that on a per-acre basis, shellfish aquaculture removes a relatively large quantity of nutrients from ambient waters - an ecosystem service that can only be replaced currently via WWTP upgrades and enhancement of agricultural BMPs. We also know that there is an ecosystem service value associated with biotechnology - a positive externality of commercial shellfish production. The use of shellfish biotechnology as a water quality management tool will require further verifications of actual production and revenues of shellfish harvesters and modifications of existing public cost-share programs or inclusion in economic nutrient trading programs. Regardless of whether shellfish farmers become eligible for nutrient credit trading, the valuation of the ecosystem services associated with shellfish culture should have the benefit of enhancing public awareness of water quality issues, and could help shift attitudes to allow increased opportunities for shellfish culture, and stimulate local economies.

[17]*In March of 2013, the Nitrogen Credit Advisory Board approved a value of $5.01 per nitrogen credit.*

RECOMMENDATIONS

- Improve governance and relationships among regulators, resource managers and the industry as a prerequisite for further development of bioextraction.

- Improve estimates of bioextraction using more accurate data on growers' culture practices, costs and revenue. Actual harvest data and areas under cultivation are needed to estimate the N removal through aquaculture. This should include quahog in addition to oyster.

- Develop an individual model for quahog to add to the EcoWin and FARM models to improve bioextraction estimates in LIS, GBP and other estuaries where this approach will be applied.

- Acquire better data on grower costs and revenues for shellfish production in order to more accurately determine the value of the ecosystem service provided by shellfish aquaculture and to fully understand the incorporation of growers into a nutrient trading program.

- Increase sampling measurements to better understand local effects of global climate change and changes in all estuarine habitats including salt marshes; this will reduce uncertainties in understanding of impacts of climate change on all habitats.

- Undertake high resolution bathymetry and topography measurement to support predictions of coastal hazards and flooding from storms/sea level rise coupled with higher density sea level measurements.

- Determine the fate of primary production and linkages to hypoxia. There are uncertainties with respect to the understanding of sinking and horizontal export of primary production, and imbalances between sources and sinks of carbon in western LIS.

- Research how climate and weather affect the amount and timing of nitrogen delivery to the estuary. Climatic trends, including extreme rain and snow events, can affect the delivery of nitrogen loads to estuaries. More nitrogen is "flushed" from the landscape during wet periods. Additionally, rainfall patterns in addition to temperature variations are known to impact phytoplankton growth, e.g. extending the timing and duration of toxic blooms, and also WWTP processes; e.g. improving removal during warmer time periods. New England is experiencing more frequent higher intensity rain storms, and this trend is anticipated to continue.

- Conduct research to better understand trophic linkages between primary production and apex predators; food web dynamics are relatively poorly known.

- Conduct research to better understand the consequences of changes in climate on timing and fate of primary production.

- Support research to identify and optimize nitrogen load reduction actions to improve conditions in the estuary.

- Monitor macroalgae spatial distribution patterns consistently from year-to-year to better characterize long-term trends on an estuary wide basis.
- Fully explore policy options for including shellfish culture into N trading programs and other bioextraction opportunities such as seaweed culture, ribbed mussel culture or harvest of other N-containing noncommercial species, such as invasive green crabs, Crepidula (limpets) or starfish.

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APPENDICES

Appendix 1: Northern Economics, Inc. Valuation of Ecosystem Services from Shellfish Enhancement: A Review of the Literature. Prepared for NOAA National Ocean Services: EPA Regional Ecosystem Services Resarch Program. May 2012.

Appendix 2: Northern Economics, Inc. Rhode Island Intensive Oyster Aquaculture Economic Impact Analysis. NOAA, National Centers for Coastal Ocean Science. February 2014.

Appendix 3a: Grizzle R., K Ward, C Peter, M Cantwell, D Katz, J Sullivan. 2014. Nitrogen and Carbon Content of Farmed Eastern Oysters (Crassostrea virginica) in the Great Bay Estuary, New Hampshire. A Final Report to: National Oceanic and Atmospheric Administration, National Ocean Service, NCCOS, Silver Spring, MD.

Appendix 3b: Grizzle, R.E., K.M. Ward, C.R. Peter, M. Cantwell, D. Katz and J. Sullivan. 2016. Growth, morphometrics and nutrient content of farmed eastern oysters, *Crassostrea virginica* (Gmelin), in New Hampshire, USA. Aquaculture Research 1-13.

Appendix 1:

Valuation of Ecosystem Services from Shellfish Enhancement: A Review of the Literature

Prepared for

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1 Introduction

This report is an expanded version of a literature review prepared by Northern Economics, Inc. for the Pacific Shellfish Institute in 2009 under a NOAA National Marine Aquaculture Initiative Grant. As with the earlier version, the purpose of this updated report is to provide planners and decision makers an overview of the existing literature on the valuation of ecosystem services provided by shellfish enhancement.[1] In order to orient non-economists to economic valuation, the report also describes the economic concepts and methods that have been applied in this literature. The primary differences between the current literature review and previous one are a greater focus on aquaculture production, an expanded review of the valuation of water quality improvement benefits and an added discussion of best management practices and standards.

The types of shellfish this report is concerned with are the bivalve mollusks, including clams, mussels, and oysters.[2] These shellfish are found throughout the coastal United States; some of the species are native to the areas where they occur, while others have been deliberately or inadvertently introduced. The emphasis of the literature review is on socioeconomic and biological studies of U.S. shellfish resources; however, studies of shellfish enhancement projects outside the United States are also discussed.

Within the scientific literature there is growing recognition of the central role shellfish can play in the maintenance and stability of coastal ecosystems. For example, oysters have been labeled “keystone species” (or “cornerstone species”) in selected marine environments (Isaacs et al. 2004). As a keystone supports an arch, keystone species support a complex community of species by performing a number of functions essential to the diverse array of species that surround them. There is also increasing recognition that some shellfish species may impact or control many ecological processes; so much so that they are included on the list of “ecosystem engineers”—organisms that physically, biologically or chemically modify the environment around them in ways that influence the health of other organisms (Jones et al. 1994). Many of the ecological functions and processes performed or affected by shellfish contribute to human well-being by providing a stream of valuable services over time. Such services are commonly referred to as “ecosystem services” (The Millennium Ecosystem Assessment 2005).[3]

Although there is increasing recognition that shellfish provide multiple ecosystem services, management of shellfish and their habitats for objectives beyond recreational and commercial harvest has not yet become widespread (Brumbaugh and Toropova 2008). The remaining sections of this chapter examine why many of the ecological services provided by shellfish are in short supply, and how the economic valuation of these services can help rectify this problem.

[1] Following the definition provided by Caddy and Defeo (2003), the term “enhancement” is used here to mean any intervention that improves the productivity of a shellfish resource and renders the productive activity more sustainable. It includes shellfish restoration and aquaculture as well as management (i.e., the sustainable use of natural shellfish beds).

[2] The bivalve mollusks currently or historically cultured in the United States as food include oysters of several species (*Crassostrea virginica, C. gigas, C. ariakensis, C. sikamea, Ostrea lurida* and *O. edulis*), mussels (*Mytilus edulis, M. trossulus and M. galloprovincialis*), several venerid (family Veneridae) clams (*Mercenaria mercenaria, Protothaca staminea* and *Venerupis phillipinarum*), scallops (*Argopecten irradians*), geoducks (*Panopea generosa*), soft-shell clams (*Mya arenaria*), cockles (*Clinocardium nuttallii*), rock scallops (*Hinnites giganteus*), arks (*Anadara transversa* and *A. ovalis*) and razor clams (*Siliqua alta, S. costata* and *S. patula*) (National Research Council 2010).

[3] A number of authors (e.g., Boyd and Banzhaf 2007; Fisher and Turner 2008) have pointed out shortcomings in the way ecosystem services were defined and classified by The Millennium Ecosystem Assessment. While this report acknowledges these critiques, it accepts The Millennium Ecosystem Assessment’s definition and classification scheme for simplicity’s sake.

1.1 Why the Production of Ecosystem Services May Be Suboptimal

In his book *Economic Values and the Natural World*, the economist David W. Pearce noted: "If the Earth's resources were available in infinite quantities, and if they could be deployed at zero cost, there would be no economic problem. Everyone could have everything they wanted without compromising each other's or later generations' wants and needs. It would not be necessary to choose" (Pearce 1993). But resources are finite, and we do have to choose among the goods and services we derive from those resources. The trade-offs people make are based on their preferences about the goods in question. If the good is bought and sold in a market, the marginal value of the good to society is reflected in its market price. The price acts as a "signal" that leads to the optimal allocation of resources among competing uses and production of goods.[4]

However, most of the services provided by a healthy, functioning ecosystem are what economists call "pure public goods" (Costanza et al. 1997). Specifically, these services exhibit the public good characteristics of non-rivalry and non-excludability. In the context of ecosystem services, non-rivalry means that more than one person can enjoy the benefits of an intact ecosystem at the same time. Non-excludability means that it is difficult (costly) to prevent one individual from enjoying the benefits created by another individual's actions to protect and preserve an ecosystem.

A consequence of these two characteristics of public goods is that the goods are rarely exchanged through markets, and thus the market system fails to allocate and price the goods correctly (Perman et al. 2011). The lack of market prices is often interpreted as if the public goods have no value, a fact that in the case of ecosystems leads to overuse or excessive exploitation (or underinvestment in protection and restoration) even though they provide services that people find beneficial.

1.2 How Economic Valuation of Ecosystem Services Offers a Possible Solution

For an economist, the question of the value of ecosystem protection or restoration and the larger question of how much protection or restoration should occur are central to the understanding of whether society should spend money on ecosystem protection or restoration or some other important "good" (Hicks 2004). Economic valuation of ecosystem services can be defined as the process of expressing a value for these services in monetary terms. Estimating the economic value of resources is frequently an important element in the formation and institution of efforts to prevent the twin problems of overexploitation and under-provision of public goods (Isaacs et al. 2004).

Today, the identification and quantification of ecosystem values is not only possible, it is increasingly seen as essential for the efficient and rational allocation of environmental resources among competing social and political demands (National Research Council 2004). Once described and accounted for, ecosystem service values can be used to make comparisons between competing economic tradeoffs. Furthermore, by estimating and accounting for the economic value of ecosystem services, costs or benefits that otherwise would remain hidden are revealed and vital information that has often remained outside of the decision-making calculus at local, regional and national scales can be brought to the foreground of decision making. A good example is the population and development pressures that estuarine areas are now experiencing. These pressures raise significant challenges for planners and decision makers (Wilson and Farber 2008). Communities must often choose between the myriad services provided by intact functioning ecosystems and competing uses of the coastal environment. To choose from these competing options, it is important to know what ecosystem services will be

[4] These economic principles truly apply only in a "perfect" market, i.e., one that is highly competitive, with many buyers and sellers all of whom have complete knowledge about the market. In reality, markets are not perfect, and therefore prices may reflect values only imperfectly (Gittinger 1982).

affected by coastal development or management and how these services create value for different members of society (Wilson and Farber 2008). Moreover, economic valuation of these ecosystem services gives them a common currency with marketed goods (e.g., beachfront condominiums constructed on landfill) so that ecosystems are not underrepresented in decision making (Carson and Bergstrom 2003).

While most of the ecosystem services provided by shellfish enhancement are not sold in markets and, therefore, not priced, there are exceptions, the principal one being the commercial harvest of shellfish for food. The relative importance of this ecosystem service may be relatively minor. An assessment of shellfish meetings over five years conducted by Luckenbach et al. (2005) revealed more than 300 presentations related to oyster restoration, with fewer than 25 percent focused solely on the protection of oyster fisheries. However, in the absence of information on the value of unpriced ecosystem services, the default measure of project success tends to be fisheries-based metrics such as harvest of market-sized oysters. The results are often disappointing due, in part, to a mismatch between the scale of restoration and measured outcomes. Citing a U.S. Army Corps of Engineers report, Brumbaugh et al. (2007) note that even relatively small shellfish restoration projects can be costly (e.g., > $100,000 per acre for restored oyster reef) relative to the value of oyster landings measured on the same area. However, valuation of non-fishery-related ecosystem services may reveal that the cost of restoration actions is justified even though it could not be justified by considering fishery benefits alone.

Valuation of ecosystem services is also important because not all services may be compatible with each other. A choice is often required among them, and this choice will dictate the total value of the ecosystem. For wild stocks of oysters to be of value as a food source, they must be harvested, thus affecting their ability to provide some other ecosystem services, such as providing habitat for themselves or other valued species, reducing shoreline erosion and contributing to water quality through their filtering capability (Sequeira et al. 2008). The valuation of the full array of services would enable planners and decision makers to better understand tradeoffs inherent in areas managed for harvest versus other uses (Brumbaugh and Toropova 2008). Lipton et al. (2006) note that the failure to acknowledge the total costs of oyster harvest in Chesapeake Bay management strategies contributed to the long term decline of the resource and, consequently, its asset value. Today, oysters in Chesapeake Bay have been reduced to one percent of their former abundance (King and McGraw 2004). Of course, a shellfish aquaculture operation can replant once harvest is complete and thereby reestablish the ecosystem services generated by the operation.

In short, economic valuation is a critical factor in ensuring that ecosystems will be maintained in a sustainable manner. Putting a dollar value on ecosystem services does not devalue them—on the contrary, it gives these services a currency so that they can be compared with the economic values of activities that may compromise them (Carson and Bergstrom 2003; National Research Council 2004). Furthermore, the results of economic valuation can be utilized by policy makers to achieve social equity in putting costs of ecosystem service losses on those responsible and using fees paid for lost services to restore those ecosystem services and thereby preserve them for the general public trust (National Research Council 2010).

2 Description of Ecosystem Services

Economic valuation of the ecosystem services provided by shellfish enhancement must start by identifying and describing those services. Based on four broad categories of ecosystem services developed by The Millennium Ecosystem Assessment (2005), Brumbaugh and Toropova (2008) organized the ecosystem services relevant to shellfish restoration or maintaining historical natural shellfish beds (Table 1). The literature describing the biological, physical and social processes underlying the provision of these ecosystem services is summarized below. It is important to note that the study of the environmental effects of shellfish enhancement is a large and rapidly expanding field, and the articles cited here represent only a small sample of the available literature. For more in-depth literature reviews the reader is referred to the comprehensive articles by Fisher and Mueller (2009), the National Research Council (2010) and Coen et al. (2011) and the regional reviews by Dumbauld et al. (2009) and Rice (2008). Furthermore, the current list of ecosystem services provided by shellfish may be incomplete—other benefits of shellfish doubtless exist without due recognition (Peterson and Lipcius 2003).

Table 1 Ecosystem Services Provided by Shellfish Enhancement

Provisioning	Commercial, recreational and subsistence fisheries Aquaculture Fertilizer and building materials (lime) Jewelry and other decoration (shells)
Regulating	Water quality maintenance Protection of coastlines from storm surges and waves Reduction of marsh shoreline erosion Stabilization of submerged land by trapping sediments
Supporting	Cycling of nutrients Nursery habitats
Cultural	Tourism and recreation Symbolic of coastal heritage

Source: Adapted from Brumbaugh and Toropova (2008)

2.1 Provisioning Services

Provisioning services are the products or goods people obtain from a restored or maintained shellfish population. They include foodstuffs and raw materials for building and manufacturing. The provisioning service of commercial shellfish fisheries and aquaculture production is often the predominant objective of enhancement programs. Shellfish are bought and consumed for their nutritional benefits as well as their taste—they are healthy sources of protein, rich in vitamins and minerals, low in fat and a good source of omega-3 fatty acids (Canadian Aquaculture Industry Alliance 2008). Despite substantial declines in U.S. shellfish landings,[5] the value of the shellfish industry to regional economies remains significant. Annual dockside value of oysters reached highs of more than $90 million per year in the 1990s (NOAA Fisheries Office of Habitat Conservation undated-a). Moreover, the full economic value of shellfish harvests goes well beyond dockside value; in addition to primary sales of the raw, unshucked product, there are economic benefits from secondary products

[5] U.S. landings of all oyster species in 1880 totaled around 154 million lbs. By the end of the 20th century, harvests had declined to just under 31 million lbs (King and McGraw 2004).

and services (e.g., shucking and packing houses, transport, manufacture of prepared oyster products and retail sales) (NOAA Fisheries Office of Habitat Conservation undated-a). In some areas shellfish fisheries and cultivating operations also contribute to public revenues through licensing and lease fees.

Some consumers choose to gather the mollusks themselves. Shellfish support major recreational fisheries in which people derive pleasure from an outdoor outing that offers them the opportunity to prepare and eat their own "catch." For example, shellfishing is a popular activity in Washington State, primarily along the Pacific Coast and the shoreline of Puget Sound. Thousands of local residents and visitors enjoy digging shellfish at local beaches and shoreline environments (U.S. Environmental Protection Agency 2006). It is estimated that the recreational razor clam fishery alone generates on average about 250,000 digger trips per year in southwest Washington counties. This recreational fishing effort represents about a $12 million influx of tourist and resident spending (on motels, food, gasoline, souvenirs, etc.) in coastal communities during winter and spring when business is otherwise slow (ORHAB Partnership 2002).

For thousands of years, the coastal indigenous peoples of North America have relied on the sea for most of their needs, and today shellfish from tidal flats remain an essential subsistence food source for Tribes and First Nations (NOAA Fisheries Northwest Fisheries Science Center and Washington Sea Grant Program 2002). In the Pacific Northwest the relative ease with which large amounts could be harvested, cured, and stored for later consumption made shellfish an important source of nutrition—second only in importance to salmon (U.S. Environmental Protection Agency 2006). For example, Donatuto (2003) states that shellfish in the tidelands adjacent to the Reservation of the Swinomish Indian Tribal Community represent a vital subsistence and commercial resource for the Tribe, as well as an important point of cultural association for the Tribe's identity. Shellfish are employed in cultural ceremonies, incorporated in the common diet, and sold to support families on the Reservation. Similarly, in the Quinault language, the words ta'aWshi xa'iits'os mean "clam hungry", indicating the traditional dependence of this Tribe on shellfish as a subsistence food (ORHAB Partnership 2002).[6]

In addition to being harvested in commercial, recreational and subsistence fisheries for their edible meat, clams, oysters and other mollusks are removed for other purposes such as chicken grit (ground-up shells fed to chickens to help their gizzards digest food and to provide calcium for egg shells), natural water filters and buffers, calcium carbonate food supplements, and even as mulch for growing lavender plants (NOAA Restoration Center undated). Shucked oyster shell is also an ideal material with which to protect shorelines because the shell becomes tightly packed and is lighter than traditional shoreline protection materials (i.e., limestone rock). Furthermore, in some areas oyster larvae quickly recruit to the created reefs, indicating that reef maintenance is not likely to be a problem (Piazza et al. 2005).[7]

Pearls are by far the best known decorative product obtained from shellfish. Though not as hard or durable as mineral gems, pearls are nonetheless valued for their luster and scarcity; moreover, the naturally formed softness of pearls is part of their appeal, and has earned them the name "queenly gem" (Hisada and Fukuhara 1999). Natural and cultured pearls are manufactured into earrings/studs, necklaces, pendants, bracelets, rings and other jewelry. In 1997, the estimated world production value of fresh and salt water pearls was $472 million (this figure does not include China's production value) (Hisada and Fukuhara 1999). Also used for all kinds of decorative purposes is the inner shell

[6] Other Native American words also suggest the traditional importance of shellfish. For example, Paolisso and Dery (2008) note that the name "Chesapeake" is derived from Algonquin word "Chesepioc" (or Tschiswapeki), which translates into English as "great shellfish bay."

[7] While most oyster-producing states lack laws requiring oyster shells to be returned to the water, Washington is an exception. Washington regulations require that all oysters taken by sport harvesters on public tidelands be shucked (opened) on the beach, and the shells left on the same tideland and tide height where they were taken.

layer of oysters, which is called nacre or mother of pearl. Most jewelry made from shellfish is bought and sold in global markets. However, North American Tribes and First Nations still use the shells of mussels, clams, abalone and oysters to decorate woodcarvings and ceremonial apparel, as they did thousands of years ago (NOAA Fisheries Northwest Fisheries Science Center and Washington Sea Grant Program 2002).

2.2 Regulating Services

Regulating services provided by shellfish enhancement are the benefits people obtain from the regulation of ecosystem processes. They are derived from the capability of shellfish to improve water quality through filtration, reduce shoreline erosion, and stabilize estuarine sediments.

2.2.1 Water Quality Maintenance

The water quality maintenance services of shellfish is a direct result of their suspension-feeding activity, which serves to reduce concentrations of microscopic algae (phytoplankton) and suspended inorganic particles in surrounding waters (Brumbaugh et al. 2006). In some coastal systems shellfish, through their feeding activity and resultant deposition of organic material onto the bottom sediments, are abundant enough to influence or control the overall abundance of phytoplankton growing in the overlying waters.[8] This control is accomplished both by direct removal of suspended material and by controlling the rate that nutrients are exchanged between the sediments and overlying waters (Ulanowicz and Tuttle 1992; Newell 2004). Encrusting, or fouling, organisms (e.g., tunicates, bryozoans, sponges, barnacles, and some polychaete worms) on oyster reefs and in the surrounding benthos are also suspension feeders and contribute to the overall filtering capacity of a reef (NOAA Fisheries Office of Habitat Conservation undated-a).

Due to this filtering of estuarine waters through feeding activity, shellfish can enhance water clarity and allow sufficient light penetration to support expansion of seagrass habitat (Newell and Koch 2004), an important estuarine "nursery area," where juvenile invertebrates and fish are protected from predators. Ecosystem modeling suggests that restoring shellfish populations to even a modest fraction of their historic abundance could improve water quality and aid in the recovery of seagrasses, which can further reduce sediment resuspension and improve light conditions (Newell and Koch 2004). In addition, researchers suggest that robust populations of shellfish can suppress harmful blooms of phytoplankton, such as the "brown tides" that have occurred along the mid-Atlantic coast (Cerrato et al. 2004), and help modulate blooms of other types of harmful plankton, including "red tides" (Peabody and Griffin 2008).

As suspension feeders, shellfish also enhance water quality by concentrated deposition of feces and pseudofeces (particles collected on its gills that the oyster does not use as food) (Newell 2004; Newell et al. 2005). Increased biodeposition of organic matter in sediments leads to increased bacterial denitrification that can help to remove nitrogen from estuarine systems. The associated bacteria in sediments of an oyster bed can remove 20 percent or more of the nitrogen in oyster wastes, using the same process that is used in modern wastewater treatment plants (Shumway et al. 2003). Further,

[8] Field observations demonstrate that large populations of shellfish are capable of consuming a considerable fraction of the phytoplankton from overlying waters. For example, Haamer, and Rodhe (2000) showed that water passing a mussel bed in the sound between Sweden and Denmark was depleted of phytoplankton in the entire height of the fully mixed water column over a several kilometer long more or less continuous mussel bed. Similarly, field measurements by Grizzle et al. (2006) reveal a considerable fraction of suspended particulate matter is removed—up to 62 percent for the bivalve species *Mercenaria*—from waters overlying dense shellfish assemblages.

filter-feeding shellfish not only remove nitrogen from the water column; they also incorporate a high proportion of it into their tissues. Shellfish are approximately 1.4 percent nitrogen and 0.14 percent phosphate by weight. When the shellfish are harvested, the nitrogen is removed from the system, thereby recycling nutrients from sea to land (Shumway et al. 2003). Much of this nitrogen is in the relatively large shell and so when species with lighter shells, such as blue mussels, are harvested, less nitrogen will be removed (Newell 2004).

By mediating water column phytoplankton dynamics and denitrification, shellfish are likely to reduce excess nutrients that stimulate excessive plant growth in coastal waters, which often leads to low dissolved oxygen levels (hypoxia) as the phytoplankton die, a serious environmental problem in many aquatic ecosystems worldwide (Atlantic States Marine Fisheries Commission 2007). For example, 65 percent of U.S. coastal rivers, bays are moderately to severely degraded by nutrient pollution from agricultural practices, urban runoff, septic systems, sewage discharges and eroding streambanks (Bricker et al. 2008).

Shellfish filtration rates are a function of several environmental factors, including temperature, salinity, and suspended particulate concentration (Cerco and Noel 2005).[9] In addition, different shellfish species exhibit significant variation in filtration rates, with the rates generally increasing with species size (Powell et al. 1992). For large species, such as the eastern or American oyster, *Crassostrea virginica,* filtration rates have been estimated at 163 liters per gram (of oyster tissue) per day (NOAA Fisheries Office of Habitat Conservation undated-a). Newell (1988) calculated that the abundance of this species in Chesapeake Bay before 1870 was high enough that oysters could filter the entire volume of the bay in about three days (but after nearly a century of exploitation and habitat destruction, the reduced populations require 325 days to perform the same activity).

2.2.2 Protection of Shorelines and Sediment Stabilization

The protective influence of intertidal shellfish habitats on adjacent shorelines has been widely noted in the literature (Meyer et al. 1997; Piazza et al. 2005; Atlantic States Marine Fisheries Commission 2007; Borsjea et al. 2011). In some locations, oyster reefs and, likely, mussel beds that fringe the estuarine shoreline serve as natural breakwaters that protect the shoreline against the erosive force of wind- and boat-generated waves, thereby reducing bank erosion, protecting fringing salt marsh and decreasing loss of aquatic vegetation beds, such as eelgrass, behind the reefs.

Grabowski and Peterson (2007) note that oyster reefs are a living breakwater; consequently, they can rise at rates far in excess of any predicted sea-level rise rate. Moreover, a living, ecologically functional oyster reef can provide a more aesthetically pleasing and ecologically sound solution to coastal erosion problems than groins, breakwaters, sea walls or jetties (Marsh et al. 2002).

Oyster reefs that fringe the shoreline also tend to stabilize sediment (Peabody and Griffin 2008). The reduction in suspended sediment in adjacent waters improves water clarity, thereby possibly providing better opportunities for establishment of seagrasses and other species. Using a predictive model of Chesapeake Bay, Cerco and Noel (2007) assessed the impact of a tenfold increase in oyster biomass on three spatial scales, and suggested that the enhanced abundance of submerged aquatic vegetation is the most significant improvement to be attained from the water clarity benefits of oyster restoration.

It is also important to note that an oyster or mussel reef protruding only several centimeters above the bottom can help delaminate water flow across the bottom and assist in water column mixing processes (Atlantic States Marine Fisheries Commission 2007). This mixing and the formation of

[9] Filtration activity is defined in terms of clearance rate, i.e., the volume of water cleared of particles per time unit.

eddies around the reef structure functions to further reduce development of hypoxic conditions (NY/NJ Baykeeper undated). In addition, by disrupting flow on open bottoms or within tidal channels, reefs create depositional zones, usually downstream of the reef structure, that accumulate sediment and organic material (Lenihan 1999). These changes in hydrodynamics and material transport directly influence recruitment, growth, and other biotic processes of the shellfish and other organisms (e.g., finfish) that live on the reef (Atlantic States Marine Fisheries Commission 2007).

2.2.3 Carbon Sequestration

One potential regulating service that was not listed by Brumbaugh and Toropova (2008) is the role shellfish may play in sequestering (storing for long periods of time) carbon in the calcium carbonate of their shells, thereby reducing concentration of carbon dioxide, a "greenhouse gas" that contributes to global climate change (Rodhouse and Roden 1987; Peterson and Lipcius 2003; Hickey 2008; Tang et al. 2011). Carbon is absorbed naturally from the ocean as the calcium carbonate shell of the shellfish grows. Shells incorporated deep into the sedimentary strata beneath the seafloor and shells buried in soils on land will remain intact indefinitely, allowing the shellfish to provide a long-lasting service of preventing the carbon from re-entering the atmosphere (National Research Council 2010). In other words, it has been suggested that shellfish enhancement through the production of shells, can act as a "carbon sink." However, Allison et al. (2011) point out that this suggestion is based on the shell carbon content without considering factors such as respiration or the calcification process. Calcification induces shifts in the seawater carbonate equilibrium to generate dissolved carbon dioxide, and therefore actually releases carbon dioxide into the water (Chauvaud et al. 2003). While this process provides local chemical buffering against growing ocean acidification caused by increasing concentrations of atmospheric carbon dioxide (National Research Council 2010), the result is that shellfish enhancement may on balance prove to be a "carbon source."

Coen et al. (2011) conclude that the issue of whether the carbonate-rich deposits of live and dead shellfish are net carbon sinks or sources needs to be addressed in greater detail before it can be resolved to everyone's satisfaction as we discuss carbon sequestration and reduction of ocean acidification as a potentially important ecosystem service of shellfish enhancement. Noting the complexity of carbon cycles in marine systems, Allison et al. (2011) conclude that one should not "expect piles of oyster shells to be eligible for carbon payments anytime soon." According to the authors, the most useful contributions of shellfish enhancement to climate change mitigation is its compatibility with the maintenance of coastal environments that have known large carbon sequestration capacity. These coastal environments include tidal salt marshes, mangroves and seagrass meadows (Laffoley and Grimsditch 2009). In many cases the carbon sink capacity of these environments to mitigate climate change is being depleted by human activities (Macreadie et al. 2011).

2.3 Supporting Services

While not providing direct services themselves, supporting services are necessary for the production of all other ecosystem services. By creating structurally complex shell habitat and performing a wide array of ecological functions, shellfish populations can substantially modify benthic and pelagic communities at different trophic levels and alter energy flow and nutrient cycling over the scale of entire coastal ecosystems (Cranford et al. 2007).

2.3.1 Cycling of Nutrients

Because they are filter feeders, shellfish can greatly influence nutrient cycling in estuarine systems and maintain the stability of the ecosystem (NOAA Fisheries Office of Habitat Conservation undated-a). As noted earlier, oysters filter large amounts of phytoplankton and detritus (small organic particles) from the water column. As grazers of phytoplankton and other particles, these filter feeders couple, or join, the oyster reef to the water column. Some of the organic components resulting from shellfish metabolism serve as a nutrient source for benthic infauna, some enters the microbial loop, and some re-enters the water column. This flux or cycling of carbon, nitrogen and other essential materials is vital for the continuity and stability of any living system and acts to keep the system in balance (Peabody and Griffin 2008; NOAA Fisheries Office of Habitat Conservation undated-a).

In addition, Norkko and Shumway (2011) state that the burrowing, feeding and other activities of shellfish in the sediment can result in high levels of bioturbation and bioirrigation (i.e., mixing and flushing of sediments). These processes increase the transport of particulates, nutrients and oxygen across the sediment-water interface, and increase oxygen penetration into the sediment, thereby producing complex and favorable microhabitats that facilitate benthic communities with higher overall diversity.

2.3.2 Nursery Habitats

In addition to playing an important nutrient cycling role, some species of bivalve shellfish such as oysters and mussels form complex structures that provide refuge or hard substrate for other species of marine plants and animals to colonize, thereby enhancing biodiversity (Brumbaugh et al. 2006; NOAA Fisheries Office of Habitat Conservation undated-a). Lenihan and Grabowski (1998) note that these structures represent a temperate analog to coral reefs that occur in more tropical environments. Both kinds of structures are "biogenic", being formed by the accumulation of colonial animals, and both provide complex physical structure and surface area used by scores of other species as a temporary or permanent habitat (Brumbaugh et al. 2006).

Older, maturing oyster reefs become larger and more complex, and provide greater habitat diversity. The extensive irregular surfaces of a reef provide 50 times the surface area of a similar sized flat bottom. These crevices provide good nursery habitat for a wide diversity of vertebrate and invertebrate organisms—worms, snails, sea squirts, sponges, crabs, and fish (Henderson and O'Neil 2003). By overcoming a survival bottleneck in the early life history of many fish and invertebrate species, oyster reefs enhance recruitment in those species, while other research indicates that oyster reef habitat contributes to fish and invertebrate production by providing refuge from predation and access to reef-associated prey resources (Peterson et al. 2003). The overall ecological result is greatly enhanced biodiversity in shellfish habitat compared to surrounding areas of the seabed (Atlantic States Marine Fisheries Commission 2007).

Perhaps most important from an ecosystem service perspective, certain types of shellfish offer the unique service of creating important habitats for other commercially or recreationally important species, particularly when they occur at high densities (Atlantic States Marine Fisheries Commission 2007). In fact, an oyster reef is the only habitat type that is itself a commercial edible species and a refuge and food for other marketable species (NOAA Fisheries Office of Habitat Conservation undated-a).[10] Among the commercial species of fish and invertebrates that use Atlantic and Gulf Coast oyster reefs at some time in their life cycles or prey upon oysters and associated fauna are flounder,

[10] The habitats created by shellfish can be classified into three major types: (1) reefs (veneer of living and dead animals), (2) aggregations (living and dead) and (3) shell (dead) accumulations (often called 'shell hash') (Atlantic States Marine Fisheries Commission 2007).

menhaden, herring, anchovies, spadefish, striped bass, cobia, croaker, silver perch, spot, speckled trout, Spanish mackerel, pinfish, butter fish, harvest fish, blue crab, stone crab, penaeid shrimp, black drum, and several species of mullet (Atlantic States Marine Fisheries Commission 2007; NOAA Fisheries Office of Habitat Conservation undated-a). On the Pacific coast, shells of the Pacific oyster placed at high density in the intertidal zone may provide excellent habitat for newly recruited Dungeness crab (Ruesink et al. 2005).

The potential size of the structure created by shellfish depends on the species. Available evidence suggests that reefs created by the Pacific oyster, *C. gigas* (mostly in the intertidal) and suminoe or Asian oyster *C. ariakensis* (mostly subtidal) are much smaller in size, occupy less area in estuaries, and are a more heterogenous mix of shell and sediment compared with eastern oyster reefs (Ruesink et al. 2005). The Olympia or West Coast oyster, *Ostreola conchaphila*, is not a reef builder, but instead is usually found attached to rocks or dead shells (Peabody and Griffin 2008). Nonetheless, Olympia oyster aggregations (beds) have high biodiversity because they provide a physical habitat structure ideal for juvenile fish and crustaceans, worms, and foraging nekton and birds (Peabody and Griffin 2008). Shellfish habitat—whether it is a living assemblage or an accumulation of dead shells—provides hard substrate for the attachment of many species that would not be present in areas consisting only, or mainly, of soft sediments. The overall ecological result is greatly enhanced biodiversity in shellfish habitat compared to surrounding areas of the seabed (Atlantic States Marine Fisheries Commission 2007).

The structures used in shellfish aquaculture (racks, cages, nets, ropes, trays and lines) also provide habitat by providing surfaces for attachment of other organisms (Shumway et al. 2003). For example, macroalgae and epifauna growing upwards from protective plastic mesh used in bottom clam culture can substitute for natural seagrass habitat as a nursery area for mobile invertebrates and juvenile fish (although this nursery habitat is removed and cleaned at harvest) (Coen et al. 2007). In comparison to unplanted adjacent sandflat, the epibiotic habitat growing on aquaculture bottom netting had a 42 (fenced lease) to 46 (open lease)-fold enhancement of mobile invertebrates and a 3 (fenced lease) to 7 (open lease)-fold enhancement of juvenile fishes (Coen et al. 2007). The subtidal rack and bag systems used to rear oysters in Southern New England and parts of the Northeast can act as refugia for a variety of marine organisms, including the juvenile stages of various species of commercially valuable finfish (Rice 2008).

The influences of shellfish habitat on associated populations, assemblages and ecological processes can extend beyond the shellfish reefs and beds into adjacent habitats. For example, shellfish reefs and beds diversify the seascape to enhance the synergistic benefits of multiple habitat types, such as creating corridors between shelter and foraging grounds (Peterson and Lipcius 2003). Furthermore, shellfish reefs and beds can support the creation of other habitat types. As discussed previously, oyster reefs can dampen waves and thus reduce the erosion of salt marsh faces and help stabilize submerged aquatic vegetation beds; they can filter estuarine waters, thereby enhancing their clarity and allowing sufficient light penetration to support expansion of seagrass habitat; and they can increase the nutrients available to seagrasses through the deposition of organic matter and waste by-products (Peterson and Heck Jr. 1999). Oysters probably generate greater per capita organic deposition than other bivalve types because of their high filtration rate and capacity to discharge pseudofeces and thereby continue filtration under conditions of high turbidity (National Research Council 2010). Thus, oyster reefs can support at least two other important habitat types within an estuary (NY/NJ Baykeeper undated). Seagrasses and salt marshes constitute additional key habitats in estuaries and provide food, habitat and nursery areas for many species (NOAA Fisheries Office of Habitat Conservation undated-a). These improvements to the ecological integrity of other habitats may ultimately lead to additional increases in the production of finfish and invertebrates targeted in commercial and recreational fisheries (National Research Council 2010).

2.4 Cultural Services

Cultural services provided by shellfish are the nonmaterial benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences.

As discussed above, some shellfish support recreational fisheries which offer participants the opportunity to gather edible shellfish themselves. People who engage in this pastime typically do so because they derive aesthetic and social benefits from the experience as well as sustenance. In addition, shellfish can play an important role in recreation and tourism by creating habitat for fish, allowing recreational fisher use, and can improve adjacent beach water quality, resulting in more desirable areas for tourists and local residents to visit (Henderson and O'Neil 2003). As an example of the sociocultural importance of recreational shellfishing, Bauer (2006) noted that razor clam harvesting, cleaning, cooking, eating and canning have been an important focus of family relationships and local culture in Washington coastal communities for many generations. To underscore this importance Bauer provides a quote from Dan L. Ayres, Coastal Shellfish Lead Biologist, Washington Department of Fish and Wildlife: "The cultural significance of joining with friends or family to successfully brave the natural elements to take home a sport limit of fresh razor clams cannot be understated."

Community support and involvement in shellfish enhancement projects can heighten public awareness of the need to rehabilitate and conserve marine and estuarine ecosystems. For example, Reynolds and Goldsborough (2008) argue that hands-on oyster reef restoration projects are not only complimentary to, but a critical component of, advancing a broader environmental policy agenda aimed at reducing nutrient loading in coastal waters. Public involvement and ownership of shellfish restoration projects is expected to ensure continuation of the projects into the future and the ability to address additional environmental issues as these arise.

In recent years, several U.S. coastal communities have become involved in restoring shellfish habitats in their areas, often in partnership with state and federal agencies and private, non-profit organizations. *Promoting Oyster Restoration Through Schools* is a community-based restoration and educational program focusing on the importance of oyster populations in the Delaware Bay ecosystem. The program utilizes the oyster as a vehicle to acquaint school children, grades K-12, with the Delaware Estuary and basic scientific concepts (Haskin Shellfish Research Laboratory 2008).

The *South Carolina Oyster Restoration and Enhancement Program* has developed numerous education tools to convey the importance of oysters to its volunteers, including online tutorials, presentations, laboratory exercises, simple experiments, observational field studies and volunteer-friendly sampling and monitoring techniques to educate and involve individuals and groups of all ages (Hodges et al. 2008). The program's underlying premise is to empower citizens to take responsibility for water quality and to encourage them to acquire a "vested interest" in water resources through investment of personal time and energy (Hadley et al. 2008).

A project to restore about one acre of oyster reef in the West Branch of the Elizabeth River, Virginia, provided an opportunity for local middle-school students to become actively engaged in a resource restoration project in their home waters (NOAA Chesapeake Bay Office 2008). The project stimulated public awareness of the ecological value of oyster reefs while instilling a sense of community stewardship in local restoration work. According to the NOAA Chesapeake Bay Office (2008), the project succeeded because of the development of a partnership approach to oyster restoration between a local community, a conservation organization, a state agency and a federal agency.

Mass et al. (2008) note that one of the major successes of the eastern oyster restoration project in the Bronx River has been the involvement of multiple Bronx River community groups. Various groups in the Bronx have helped construct the pilot reefs, monitored for spat settlement, maintained oyster "gardens" in the area, and educated the public about the benefit of oyster reefs and their importance to the ecosystem. A community-based project to transplant blue mussels into a degraded tidal salt pond in Portsmouth, New Hampshire, garnered city involvement and increased local awareness of the pond as an ecosystem rather than a sewage lagoon (McDermott et al. 2008).

According to Burke and Menzies (2010), an indication of the value of shellfish resources to the residents of Drayton Harbor, Washington is the volunteer contributions of time and money these residents made to improve water quality in the area and restore a community oyster farm. Between 1998 and 2008, over 38,000 volunteer hours were spent on this shellfish restoration project. The authors report that this public investment in volunteerism has contributed to the social capital necessary for effective local government in the area.

In addition to these community-based shellfish restoration efforts, thousands of people participate each year in shellfish celebrations that raise money for non-profits, promote community engagement in local environmental issues and segue nicely with the growing interest in local food and connecting people with local foods and traditions (U.S. Environmental Protection Agency 2006). A good example of such a celebration is the Annual West Coast Oyster Shucking Championship and Washington State Seafood Festival held in Shelton, Washington. "OysterFest" is the primary fund raising activity of the Shelton SKOOKUM Rotary Club Foundation, which uses the earnings to support a broad array of community organizations and events (Shelton SKOOKUM Rotary Club Foundation 2008).

Shellfish fisheries and aquaculture can also indirectly bring local environmental problems to the attention of nearby communities. U.S. public health standards under which shellfish fisheries and aquaculture operate demand clean waters and commercial shellfish harvest can only take place in waters that have been certified under the National Shellfish Sanitation Program (Shumway et al. 2003). The standards of this program fostered the first estuarine/marine monitoring programs, and are the most stringent of all U.S. water quality classifications, far exceeding those required for swimming. As a result, the presence of shellfish fisheries and aquaculture often results in increased monitoring of environmental conditions of estuaries and coastal waters. Moreover, the economic hardships suffered by communities following closure of shellfish fisheries and culture operations due to water contamination have often provided the political impetus for improvement in sewage treatment plants or programs to fix local septic systems (Shumway et al. 2003).

Public participation in shellfish restoration projects can also foster an appreciation of cultural heritage. In coastal communities throughout the U.S., shellfish are cultural icons, reflecting traditions and a way of life dating back generations (Brumbaugh et al. 2006). For example, the "watermen" in Chesapeake Bay area communities who make a living by harvesting oysters derive more than material gain from their work—for them it is entire lifestyle. What's more, there is a growing recognition among the broader public that the livelihood of these individuals is an integral part of what is worth preserving in America's coastal areas (Wasserman and Womersley undated). Similarly, shellfish aquaculture is part of the cultural history in parts of Europe. Shellfish culture in Europe has become a family tradition, where the rights and also the skills stayed within families (Wijsman 2008).

As discussed in Section 2.1, the harvesting of shellfish for food and cultural purposes is also a longstanding practice deeply rooted in some aboriginal communities in North America and elsewhere (Kingzett and Salmon 2002). For instance, the Wampanoag Tribe, Massachusetts' only federally recognized Native American Tribe, is investing time and money into culturing bay scallops, as bay scallops historically played an important cultural role for the Tribe (Trauner 2004). Such tribal projects can help preserve traditional ecological knowledge as well as provide an important source of cash income (Harper 2008).

Western Washington Tribes continue to use shellfish for subsistence, economic and ceremonial purposes (U.S. Environmental Protection Agency 2006), and the Tribes are closely involved in efforts to rebuild stocks of Puget Sound's native oysters (Peter-Contesse and Peabody 2005; Kay 2008). Moreover, shellfish grounds have become important to some Washington Tribes by affirming treaty rights that entitle them to fish and hunt for subsistence and commerce on traditional lands and water (Barry 2008). Federal courts have upheld the Tribes' treaty rights to harvest shellfish in Puget Sound, including on private tidelands under certain circumstances.[11] The exercise of these treaty rights helps Tribes preserve their self-sufficiency and cultural autonomy (ORHAB Partnership 2002).

2.5 Ecological Uncertainty

Freeman (1999) states that the first steps in economic valuation of ecosystem services are to determine the size of the environmental change affecting ecosystem structure and function and determine how these changes affect the quantities and qualities of ecosystem service flows to people. However, it is important to recognize that many of the reported changes in ecosystem structure and function associated with shellfish enhancement are a source of scientific debate. Major gaps in knowledge include how native and introduced shellfish species influence nutrient cycling, hydrodynamics, and sediment budgets; whether other native species use them as habitat and food; and the spatial and temporal extent of direct and indirect ecological effects within communities and ecosystems (Ruesink et al. 2005).

For example, one of the reported ecosystem services provided by restoration of oyster reefs and other bivalve-dominated habitats, the grazing of phytoplankton populations, was the focus of recent articles (Pomeroy et al. 2006; Pomeroy et al. 2007), which concluded that filtration by the eastern oyster in Chesapeake Bay, either at historical densities or at current restoration target densities, is insufficient to reduce the severity of phytoplankton blooms and resulting hypoxia in the Bay because of temporal and spatial mismatches between eastern oyster filtration and phytoplankton (see reply by Newell et al. (2007)). In a literature survey following this debate, Coen et al. (2007) found that several researchers expressly noted that the system-level effects of oyster filtration have been poorly quantified, especially as they might relate to any specific native oyster reef restoration project. The authors noted that the potential benefits of filtration by oysters as stated in the popular press ignore the realities of the scale of restoration required to achieve such benefits. More recently, a debate arose in the literature over the potential of mussel farming as a nutrient reduction measure in the Baltic Sea (Stadmark and Conley 2011; Rose et al. 2012; Stadmark and Conley 2012).[12] The issue of shellfish and carbon sequestration brings up yet another area of controversy (see Section 2.2.3).

Peterson and Lipcius (2003) suggest that planners and decision makers would be able to make more informed decisions about future restoration actions if they were armed with explicit estimates of the probabilities of a suite of alternative outcomes associated with each restoration alternative. However, due to the interconnectedness of the various elements of an ecosystem and the variety and

[11] Although the Tribes have no claim to the oysters, native or otherwise, that are planted on privately owned or leased tidelands, they do have a right to 50 percent of the shellfish that existed prior to any stock enhancement efforts. An agreement was signed between the 13 treaty Tribes of Puget Sound and the commercial shellfish growers in Washington State. Under this agreement the Tribes have agreed not to harvest from qualifying shellfish beds from qualified commercial growers (Barth 2008).

[12] Rose et al. (2012) argue that the considerable uncertainties surrounding the application of shellfish enhancement, "'nutrient bioextraction'' strategies to nutrient management are attributable in part to the limited number of direct field trials and also to the inherent variability across coastal ecosystems worldwide. The authors conclude that a more thorough evaluation of nutrient assimilation and regeneration resulting from shellfish biodeposition within an ecosystem context is needed to reach informed conclusions for each location under consideration.

complexity of ecological outputs (National Research Council 2004), it is extremely difficult to identify the probability that an impact on the biological and human environment will occur and to weigh this probability against the magnitude of the possible impacts (Evans 1979). The ability of economists to place economic valuations on ecosystem services is contingent on a concerted effort to measure and document these services in the field. Consequently, ecological uncertainty propagates through to uncertainty about economic outcomes (Dorrough et al. 2008).

In order to start to address uncertainties about the ecological effects of shellfish enhancement projects emphasis has been placed on project monitoring, i.e., systematic data collection that indicates progress toward identified criteria, performance standards and ecological goals (NOAA Fisheries Office of Habitat Conservation undated-b). Peterson and Lipcius (2003) argue that monitoring the performance of the restoration and then adaptively modifying the scale or even type of restoration in response to documented performance of the restoration could legitimately be included among the costs of restoration. In 2004, participants at a Sea Grant sponsored workshop proposed a set of sampling criteria and methodologies to provide standardized population and ecosystem measures for assessing the success of oyster restoration projects (Coen et al. 2004). In addition, The Nature Conservancy and partners have developed a nationwide network of shellfish restoration sites where quantitative approaches are used to monitor ecosystem services and outcomes associated with restoration projects (Brumbaugh and Toropova 2008).

In recent years, modeling has also been increasingly used as a means to acquire a better understanding of the impacts of shellfish enhancement on water quality, nutrient cycling and benthic processes. The application of mathematical models provide a means of quantitatively evaluating ecosystem goods and services which result from shellfish farming in coastal environments. For example, nutrient enrichment interactions with shellfish aquaculture only recently began to be assessed through modeling efforts, mostly within the past decade (Burkholder and Shumway 2011). One such modeling effort was the use by Landeck-Miller and Wands (2009) of a mechanistic numerical model of Long Island Sound eutrophication processes, the System Wide Eutrophication Model, to assess the potential of shellfish aquaculture as an additional means of removing nitrogen from Long Island Sound. For reviews of modeling studies pertaining to the environmental impacts of shellfish aquaculture see Burkholder and Shumway (2011) and Grant and Filgueira (2011).

3 Economic Valuation of Ecosystem Services

The economic concept of value has been broadly defined as any net change in human well-being or welfare. In economic analysis, any action which increases welfare is a benefit and any action which decreases welfare is a cost. In assessing the value of a project to protect or restore a shellfish population, the economist is interested in estimating how much the welfare of one person or society at large would change as a result of that project.

Economic welfare includes what economists call consumer surplus and producer surplus. Consumer surplus is the net value consumers receive from a good or service over and above what they actually pay for the good or service. Producer surplus (also called economic rent) is the difference between what producers actually receive when selling a product and the amount they would be willing to accept for the product. While not an exact measure of social welfare, the sum of the consumer and producer surplus that results from the ecosystem services provided by a shellfish enhancement project provides a useful approximation of the project's net benefits.

Converting the benefits of ecosystem services to a common comparable unit (dollars) so as to sum them often represents a major challenge to economics (Peterson and Lipcius 2003). While economists possess an array of methods to assess consumer and producer surplus in monetary terms, no single method can capture the total value of the many, disparate ecosystem services provided by a complex natural asset such as an oyster reef (Johnston et al. 2002). Moreover, although the value of some services can be readily monetized, the value of others can be done so only with great difficulty and uncertainty (Johnston et al. 2002). For example, estimation of consumer surplus is relatively straightforward if services are traded in traditional markets with market prices and values (e.g., commercial shellfish, shell for road building). However, as discussed in Section 1.1, many of the benefits of the ecosystem services provided by shellfish enhancement (e.g. water filtering and enhanced water quality, habitat for other organisms) accrue directly to people without passing through the market economy.

The first section of this chapter provides an overview of methods to estimate dollar measures of the value of ecosystem services. Each economic valuation method has strengths and weaknesses, and each service has an appropriate set of valuation methods. Furthermore, some services may require that several methods be used jointly (Farber et al. 2002; Carson and Bergstrom 2003). The second section demonstrates how economic valuation methods can be used to place a value on the services provided by shellfish enhancement. Where possible, examples drawn from the literature are used to illustrate applications of these methods.

3.1 Methods

There are two general types of approaches for estimating economic welfare gains (or losses). The first approach, which is to conduct primary research, can be subdivided into indirect (revealed preference) and direct (stated preference) methods. This approach requires the collection of new data, which may be costly and time-consuming (Dumas et al. 2005). Consequently, some researchers have adopted the second approach, commonly called benefit transfer, whereby existing valuation information for an ecosystem service is used to estimate the value of a similar ecosystem service.

3.1.1 Primary Research Approach

3.1.1.1 Revealed Preference Methods

Revealed preference techniques estimate the value of an ecosystem service using market data and consumer characteristics, activities and purchases (Isaacs et al. 2004). The major strength of indirect approaches is that they are based on data reflecting actual market choices, where individuals bear the actual costs and benefits of their actions (Dumas et al. 2005). The most common revealed preference methods are the market price method, hedonic pricing method, travel cost method and cost-based methods.

The market price method uses the prices of goods and services that are bought and sold in commercial markets to determine the value of an ecosystem service (King and Mazzotta 2000a). By measuring the change in producer and consumer surplus after the application of a change in production or price, the value can be determined (Carson and Bergstrom 2003). The primary shortcoming of this method is that it only takes into account the market components of the value of ecosystem services.

The travel cost method estimates the number of recreational trips an average person takes to a specific site, as a function of the cost of travelling to that site, the comparative costs of travelling to substitute sites, and the quality of the recreational experience at the sites.[13] The basis of the method is the assumption that the recreational experience is enhanced by high quality sites (e.g., clean water, abundant recreational fisheries), hence the net willingness to pay for—and value of—recreational trips depends on site quality. Travel cost models require data on participation rates, cost of travel to sites and site quality (Johnston et al. 2002).

Cost-based methods, which include the damage cost avoided, replacement cost and substitute cost methods, are related methods that estimate values of ecosystem services based on either the costs of avoiding damages due to lost services, the cost of replacing ecosystem services, or the cost of providing substitute services (King and Mazzotta 2000c). These methods do not provide strict measures of the economic value of ecosystem services, but rather provide rough indicators of the value by assuming that, if people incur costs to avoid damages caused by lost ecosystem services, or to replace the services of ecosystems, then those services must be worth at least what people paid to replace them. For example, the replacement cost method might identify a project for providing the same services and calculate the cost of construction for that project (Carson and Bergstrom 2003).

King and Mazzotta (2000c) note that that the key assumption of cost-based methods may not necessarily be valid. Just because an ecosystem service is eliminated is no guarantee that the public would be willing to pay for the identified least cost alternative merely because it would supply the same benefit level as that service. Without evidence that the public would demand the alternative, cost-based methods are not valid estimators of ecosystem service value. King and Mazzotta conclude that cost-based methods are most appropriately applied in cases where damage avoidance or replacement expenditures have actually been, or will actually be, made.

[13] The random utility model is a variation of the travel cost method. Unlike the traditional travel cost model, which focuses on one recreation site, the random utility model uses information from multiple recreation sites. Individuals choose a recreation site based on differences in trip costs and site characteristics (e.g., water quality) between the alternative sites. Statistical analysis of the relationship between site characteristics and recreationists' site choices enables estimation of any consumer surplus changes arising from any changes in site characteristics (Dumas et al. 2005)

3.1.1.2 Stated Preference Methods

Stated preference methods involve questioning survey respondents to determine changes in consumer surplus. The contingent valuation and conjoint/choice analysis methods are examples of these methods. The contingent valuation directly elicits net willingness to pay as a measure of the value of a specified level of environmental quality. As an example, a survey might ask respondents' willingness to pay for water that is acceptable for swimming and other activities (Henderson and O'Neil 2003). The conjoint/choice survey format asks respondents to choose between bundles of environmental assets, which differ across their physical, biological, aesthetic and/or money dimensions (Johnston et al. 2002).

The major weakness of stated preference methods is their hypothetical nature. The results of the methods are often highly sensitive to what people believe they are being asked to value, as well as the context that is described in the survey.[14] In most cases, the survey is designed to include a description of a realistic or plausible scenario that is comprehensible to the respondent. For example, for previously highly polluted systems such as the Chesapeake and the Upper Narragansett, where beach closings due to pollution are common, the hypothetical nature of stated preference methods is not a limit—beach closures and swimming bans actually happened (Henderson and O'Neil 2003). In some surveys, however, respondents may be placed in unfamiliar situations in which complete information may not be available. At best, respondents give truthful answers that are limited only by their unfamiliarity. At worst, respondents give unconsidered answers due to the hypothetical nature of the scenario (Dumas et al. 2005).

Notwithstanding these problems, stated preference methods may be the only kind of valuation technique suitable in many circumstances. In particular, they are the only way of measuring non-use values, also referred to as passive-use values, which include such values as existence value and option value. Existence value is the welfare obtained from the knowledge that an environmental asset exists in a certain condition, without directly using it. Option value is the welfare obtained by retaining the option to use an environmental asset at some future date.

3.1.2 Benefit Transfer Approach

As noted above, the benefit transfer approach is used when limited time or funding preclude costly data collection and the development of new consumer surplus estimates. With benefit transfer, value estimates from one or more previously conducted valuation studies that used revealed or stated preference methods are spatially and/or temporally transferred to a new study (Dumas et al. 2005). The literature on benefit transfer generally describes four types of approaches: value (benefit estimate) transfer; value function transfer, meta regression analysis and preference calibration (Rosenberger and Loomis 2001; Dumas et al. 2005; Ready and Navrud 2005).

Value transfer uses summary measures of the environmental benefit estimates directly. (Dumas et al. 2005). The approach encompass the transfer of a single (point) benefit estimate from an existing study, or a measure of central tendency for several benefit estimates from a previous study or studies (such as an average value). The primary steps to performing a single point estimate transfer include identifying and quantifying the changes in, say, recreational use at a study site, and locating and

[14] Economists acknowledge that questions of validity, bias, and reliability persist in the use of the contingent valuation method and other survey-based methods to value the non-market components of ecosystem services. In 1992, the National Oceanic and Atmospheric Administration commissioned a blue ribbon panel to advise the agency on the use of the contingent valuation method (Arrow et al. 1993). The panel concluded that the contingent valuation method can produce estimates reliable enough to be the starting point for a judicial or administrative determination of natural resource damages as long as certain sampling and survey design guidelines are followed.

transferring a "unit" consumer surplus measure (Rosenberger and Loomis 2001). Consequently, this approach is best suited for situations where the projected impacts of a project or policy can be measured in fairly homogeneous, divisible units (Ready and Navrud 2005).

Function transfers are a more rigorous approach whereby a benefit or demand function is transferred from another study. The benefit function statistically relates peoples' net willingness to pay to characteristics of the ecosystem and the people whose values were elicited (King and Mazzotta 2000b). When a benefit function is transferred, it can be adapted to fit the specific characteristics of the site of interest, thus allowing for more precision in transferring benefit estimates between contexts (King and Mazzotta 2000b; Rosenberger and Loomis 2001). Ready and Navrud (2005) note that value function transfer will work well only if a) there is sufficient variation at the existing study site in the attributes of the ecosystem, b) there is sufficient variation at the existing study site in the characteristics of the user population, c) the attributes of the ecosystem and the population at the site of interest fall within the range of the original data at the study site, and d) consumer preferences for the ecosystem services are similar at the study site and site of interest.

Meta regression analysis offers the advantage of combining results from several original valuation studies. With this method, benefit estimates gathered from multiple studies serve as the dependent variable in regression analysis, and characteristics of the individual studies serve as the independent variables. According to Dumas et al. (2005), meta regression analysis may be used to control for differences in functional form and other methodological differences across studies as well as differences between the study sites and site of interest. Problems with this method include reporting errors and omissions in the original studies; inconsistent definitions of environmental commodities and values; and large random errors (Dumas et al. 2005).

Smith et al. (1999) proposed the method of preference calibration as a solution to the problems associated with benefit function transfer and meta regression analysis. Rather than computing a unit value or constructing a statistical function describing how unit values change with economic or demographic variables, the same existing studies can calibrate a specific preference function. Dumas et al. (2005) note that a major benefit of preference calibration is its recognition that net willingness to pay is constrained by income in situations involving large changes in policy variables. However, the authors also list several problems with preference calibration: it does not tailor the benefit estimates to the demographics and other characteristics of the site of interest as does benefit function transfer and meta regression analysis; it is more time consuming than benefits function transfer due to the increased analytical burden; and it has yet to be vetted by tests of transfer accuracy.

Benefit transfer studies have been carried out at various geographical scales and for a wide range of ecosystem services. However, there is still no consensus over the accuracy and precision of the technique for a number of reasons, including disagreement over the principles that should underlie benefit transfer, over the required standards of accuracy for transfers, and over the best valuation methods for developing benefit transfer (Hanley et al. 2006). Furthermore, the number of original studies available is insufficient for many purposes. It is true that the international peer-reviewed literature in the field of economic valuation of ecosystem services has grown substantially in recent decades (Wilson and Farber 2008). In addition, access to this literature is facilitated by the availability of searchable online databases.[15] However, the lack of benefit studies across multiple contexts remains one of the significant challenges to the growth and sustainability of these online databases

[15] These online databases include the one maintained by the National Ocean Economics Program (2012), which is the first portal dedicated to compiling and organizing bibliographic information on valuation studies specific to coasts and oceans, and the Environmental Valuation Reference Inventory (2011), an international database of studies on the economic value of environmental benefits and human health effects that has been developed specifically as a tool for the benefit transfer approach.

(Wilson and Farber 2008).[16] For example, with particular reference to the nutrient cycling benefits of oyster reefs and other estuarine habitats, Piehler and Smyth (2011) note that transferability needs to exist within a system (e.g., are rates of nitrogen removal the same across habitat types and landscapes) and between systems (e.g., are habitat-specific nitrogen removal rates similar from estuary to estuary) before these processes can be modeled effectively at the ecosystem level and extended to economic evaluations.

3.2 Applications

This section reviews studies that have used economic valuation methods to place a value on the services provided by shellfish enhancement. A sample of these studies and their findings is provided in Table 2.

Table 2 Examples of Valuation of Ecosystem Services Provided by Shellfish Enhancement

Commercial fisheries and aquaculture	Lipton (2008c) projected that the net returns to harvesting oysters in Maryland and Virginia over a 10-year time horizon were $12.8 million. Lipton (2008a) estimated the annual gain in consumer surplus from an increase in Chesapeake Bay harvest of 2.57 million bushels of oysters to be $11.6 million.
Recreational fisheries	English (2008) reported an average per-trip value of $21.40 for recreational shellfishing in southeastern Massachusetts.
Water quality maintenance	Hicks (2004) determined that the water quality improvements resulting from restoring 1,890 acres at 73 reef sites in Chesapeake Bay would annually generate $640 thousand in benefits for recreational anglers. Newell et al. (2005) derived an annual replacement value of nitrogen removal by oyster reefs in the Choptank River, Maryland of $314,836, or $181 per hectare.
Cycling of nutrients and creation of habitat	Grabowski and Peterson (2007) estimated the value of enhanced commercial fish production by oyster reefs in the southeast United States to be $3,700 per hectare per year. Isaacs et al. (2004) found that the average annual net willingness to pay among resident saltwater recreational fishermen to maintain access to fishing over Louisiana's oyster reefs was $13.21.

3.2.1 Provisioning Services

As discussed in Section 2.1, shellfish can support important commercial, recreational and subsistence fisheries where their growth is abundant and the waters are suitable (Henderson and O'Neil 2003).

[16] Smith and Pattanayak (2002) describe one factor that has limited the number of economic valuation studies: "To be published, non-market valuation research generally must introduce a new method. Field journals in environmental economics are usually not interested in new estimates of the benefits from improving a given environmental resource for their own sake. Updating results for a specific application, such as the demand for sport-fishing recreation or new estimates of the marginal willingness to pay for improvements in air quality may have policy value but usually will not be considered important enough to occupy scarce journal space."

This section reviews studies that have placed a value on these provisioning services of shellfish enhancement.

3.2.1.1 Commercial Fisheries and Aquaculture Operation

Using the market price method, the value of commercial shellfish fisheries and aquaculture operations is comparatively easy to evaluate, as they generate products that are bought and sold in markets and, therefore, have observable prices. Welfare received by society from these activities is represented by the producer surplus that accrues to the harvesters, processors and retailers of the products and by the consumer surplus accruing to those who purchase and consume the harvested products.

Producer Surplus Derived from Commercial Shellfish Fishing or Culture

The appropriate measure of producer surplus for a shellfish fishing or aquaculture enterprise attempts to determine income or gross revenues net of all the firm's costs of production, including the costs of labor and capital. Gross revenues can be estimated based on the dockside price paid to the fisher/farmer for the shellfish and the quantity sold. Price and landings data can generally be obtained from government agencies or trade journals. Calculating expenses is typically more difficult due to a lack of publically-available cost data. Even with cost data an issue often arises related to the cost of a person's labor. Labor costs must consider the opportunity cost of labor, that is, what an individual could have earned if they had not spent that time shellfish fishing or farming (Lipton 2008b). It is conceivable that some individuals have greater earning potential when they are not on the water, but forgo higher income because they prefer working the water to other forms of employment (Lipton 2008b).

The production value of shellfish goes well beyond dockside value of the raw, unshucked product (King and McGraw 2004); the economic benefits of shellfish fishing or aquaculture to other market levels such as processors, wholesalers and retailers must also be considered.[17] Estimates of welfare measures for the wholesaler/processor segment of the industry must take into account that wholesale product prices double count the dockside value, that is, the price of a wholesale shellfish includes the price paid to the fisher/farmer for the shellfish plus the expense of adding value by processing, packaging and transporting the product, plus the profit to the processor (Lipton 2008b). Only that increase in profit to the processor is a potential welfare gain from shellfish enhancement, and even of that, only the profit that they earn from shellfish over and above they might earn from investing in processing some other product would count. For example, if the processors earn greater profits because they no longer have to transport shellfish from other regions, it would be that increased profit that would be the measure of the welfare gain, not the total value of their processing output. Even then, over time, market factors might shift to eliminate these benefits, and the welfare gains to local processors might lead to welfare losses to other processors outside the region (Lipton 2008b).

A number of studies have estimated, at least partially, the producer surplus generated by shellfish restoration projects, or collected industry information that would support such an analysis. Lipton (2008a) estimated the net returns to harvesting oysters in Maryland and Virginia over a 10-year time horizon using a harvesting cost estimate by Wieland (2006). The net present value of the stream of net revenues using a 2.6 percent rate of discount was $12.8 million. In an economic assessment of an oyster restoration project in Apalachicola Bay, Florida, Berrigan (1990) calculated the revenues from

[17] Some shellfish fishing or cultivating operations harvesters deliver directly to restaurants or other retail outlets, but it is more common for harvesters to sell shellfish either to wholesalers or to processors. Wholesalers repack shellstock into sacks, boxes, or bushels and sell them to processors or directly to restaurants or retailers. Processors produce raw shucked shellfish; prepared raw halfshell shellfish; and smoked, cooked, or canned shellfish (Muth et al. 2002).

shellfish harvest and revenues generated by added value through wholesale and retail sales. The British Columbia Ministry of Agriculture and Lands (2005) estimated the production economics for seeded sub-tidal geoduck grow-out production in British Columbia using a farm model. Coffen and Charles (1991) investigated the determinants of shellfish aquaculture production in Atlantic Canada through the estimation of Cobb-Douglas production functions, relating production output to several independent input variables. The statistical analysis was carried out for both mussel and oyster culture, based on data collected in a survey of aquaculturists. Muth et al. (2002) estimated the per unit processing costs of post-harvest treatment of Gulf-harvested oysters. The production costs of shellfish culture in Willapa Bay, Washington, were examined by Bonacker and Cheney (1988). Frank Harmon Architect and Olympus Aquaculture Consulting (2008) estimated the profitability of a hatchery in Virginia for the production of oyster larvae and spat. Adams et al. (1993) projected positive net returns for a hard shell clam culture operation in Cedar Key, Florida.

Lipton et al. (1992) note that the producer surplus generated from commercial shellfish fisheries will depend on how the shellfish resource is managed. Net benefits to producers will likely be less under an open access fishery management regime than if a bottom leasing program is instituted.[18] This expectation follows from the well-known result of "rent dissipation" in open access management regimes (Lipton et al. 1992).

In recent years, researchers have made greater use of modeling tools to improve the economic performance of shellfish aquaculture operations. For example, the Farm Aquaculture Resource Management (FARM) modeling framework applies a combination of physical and biogeochemical models, bivalve growth models and screening models for determining shellfish production (Longline Environment Ltd. 2012). The FARM model will eventually enable a detailed analysis of the production of a full range of cultivated shellfish species, including various species of mussels, oysters and clams. In a study of five different shellfish aquaculture systems, ranging from bays to lochs, where different species and aquaculture techniques are being used, Ferreira et al. (2007) added a Cobb–Douglas function to the FARM model to determine the optimal stocking density in terms of maximum economic profit. Silva et al. (2011) used the model in conjunction with other modeling tools to predict potential production, economic outputs and environmental effects at proposed shellfish aquaculture sites.

Consumer Surplus Derived from Commercial Shellfish Fishing or Culture

Consumer surplus will often be increased by shellfish enhancement because the seafood eating public will have available a quality and greater quantity of shellfish at a lower price. Lipton (2008a) and Lipton (2008c) approximated the consumer surplus benefit from a restored Chesapeake Bay oyster fishery using an estimated inverse demand curve. The author notes that this is a simple approach and it has limitations caused by failure to consider the entire system of demand and supply equations; the system of equations for each of the different types of oysters; a demand specification inconsistent with traditional economic theory; and extremely limited data (Lipton 2008a). The annual gain in consumer surplus from an increase in Chesapeake Bay harvest of 2.57 million bushels of oysters was estimated to be \$11.6 million.

3.2.1.2 Recreational and Subsistence Fisheries

Comparatively few studies have assessed the recreational value of shellfishing. Hayes et al. (1992) used the contingent valuation method to estimate benefits to recreational shellfishing from

[18] Sedentary fishery resources such as oysters and mussels have long been subject to property rights. Sergius Orata reportedly cultivated oysters in Lake Lucrine during the early Roman empire (Bolitho 1961).

improvements to the water quality of Upper Narragansett, after expenditures for infrastructure to reduce pollution by Rhode Island communities. Survey respondents were asked their net willingness to pay for acceptable swimming and shellfishing (versions of the survey asked for swimming and shellfishing separately or combined). The contingent valuation method was also used by Damery and Allen (2004) to estimate the value of recreational shellfishing on Cape Cod, Massachusetts. Survey questions were posed to elicit the shellfisher's willingness to pay to obtain the right to shellfish, or, alternatively, their willingness to accept compensation to give up their right to shellfish (in a case where they already own a shellfish permit). English (2008) applied the travel cost method to assess the value of recreational shellfishing in southeastern Massachusetts. In addition to differences in travel costs, English's model accounted for differences in annual recreational license fees that communities impose for access to local shellfish beds. The average per-trip value for recreational shellfishing was estimated to be $21.40.

As discussed in Section 2.1, subsistence shellfishing continues to be important to many coastal Tribes and First Nations in North America. Attaching a dollar value to wild food harvests is difficult, as subsistence products do not circulate in markets. However, if families did not have subsistence foods, substitutes would have to be purchased. A cost-based method that estimate a replacement expense per pound based on the price of store-bought substitutes of a similar nutritional content can be used to derive a replacement value of wild shellfish harvests. The World Bank (2000) used this methodology to estimate the value of subsistence fisheries in Pacific Island economies.

3.2.2 Regulating Services

3.2.2.1 Water Quality Improvements

As natural biofilters that improve water quality by removing suspended solids and nutrients and lowering turbidity, shellfish are analogous to wastewater treatment facilities (Grabowski and Peterson 2007). This section discusses two methods that have been used to estimate the water quality improvement benefits of shellfish enhancement.

An estimate of the benefits generated by a regulating ecosystem service can be derived from the values of the human activities that are supported and protected by the ecosystem service (National Research Council 2004). For example, the improved water quality that results from shellfish enhancement can, in turn, enhance the quality of recreational beach use, boating and fishing. However, the values of many of these human activities may not be produced and traded in the private market economy. Consequently, methods appropriate for the valuation of non-market components of ecosystem services must be used.

An alternative method of estimating the water quality improvement benefits of shellfish enhancement is the cost-based approach, which as discussed in Section 3.1.1.1, does not provide strict measures of the economic value of ecosystem services, but rather provides rough indicators of the value by assuming that, if people incur costs to avoid damages caused by lost ecosystem services, or to replace the services of ecosystems, then those services must be worth at least what people paid to replace them. For example, the nutrient reduction costs of an individual unit of oyster reef can be quantified and compared to the cost of processing a similar amount of nutrients with a wastewater treatment facility (Grabowski and Peterson 2007).

This section describing the valuation of water quality benefits concludes with a discussion of the proposed participation of shellfish enhancement projects in water quality trading programs. Nontraditional nutrient trading entities, including shellfish aquaculture and restoration, are increasingly proposed as deserving of nutrient credits (Maroon 2011). The establishment of a market in which entities are free to buy and sell nutrient credits or offsets among themselves would generate a market price for nutrient reduction, and thereby place a monetary value on the ability of an ecosystem, such as an oyster reef, to improve water quality.

Rose et al. (2012) caution that the search for innovative nitrogen removal pathways and technologies commences with very limited knowledge of the costs and feasibility of alternatives. Given this lack of understanding, the authors state that the challenge is to create incentives and a competitive process whereby people can test the costs and effectiveness claims of alternative nutrient-removal technologies against the hard reality of experience. Even if it demonstrated that shellfish enhancement can make a significant contribution toward improving water quality, Cerco and Noel (2007) emphasize that it will not be a panacea for the host of problems associated with nutrient overload in coastal waters. For example, Peter-Contesse and Peabody (2005) point out that the input of nutrients in many parts of Puget Sound far exceeds the processing capacity of filter feeders. On the other hand, filtering by shellfish may increase the water quality benefits resulting from other management actions such as wastewater treatment, land use changes and nonpoint pollution controls (Henderson and O'Neil 2003). Furthermore, shellfish enhancement and other bioremediation measures may have cost advantages by their direct impact on eutrophic coastal zones and the multifunctional abatement of several pollutants (Gren et al. 2009). As Burkholder and Shumway (2011) state, shellfish aquaculture not only can reduce nutrient loads and phytoplankton blooms, it can also remove toxic contaminants and reducing concentrations of microbial pathogens.

Benefit Valuation Methods

As noted above, the value of the water quality improvements obtained from shellfish enhancement can be estimated by measuring the recreational and other benefits derived from those improvements. For example, shellfish enhancement projects may increase the desirability for swimming and other water contact activities by improving water quality conditions through filtering out phytoplankton and fine sediment from the water column (Henderson and O'Neil 2003).

Numerous studies have estimated the value of the recreational benefits of water quality improvements using methods appropriate for the valuation of non-market components of ecosystem services. For example, Freeman (1995) reviewed studies of valuation of water quality improvement and various types of marine recreation, including sport fishing, swimming and related beach activities, and boating. This literature employed several revealed and stated preference methods, and the review showed that there is substantial variation in value measures across studies. The author attributed these differences to five sets of factors: intrinsic differences in the resource, that is, differences in sites (or species in the case of fishing); differences in the socioeconomic characteristics of the populations whose values have been estimated; differences in the way in which sites and markets have been defined; differences in the economic models of behavior and choice on which estimates are based (including functional forms of demand functions); and differences in the econometric estimation techniques used. As Freeman notes, the last three sets of factors involve choices made by the investigator in the course of designing and carrying out the study.

While numerous studies have been conducted on determining the value of the recreational benefits of water quality improvements, only a few of these studies focus specifically on the improvements likely to result from an enhanced population of shellfish. Hicks (2004) used a travel cost model of recreation demand to analyze the economic benefits to Chesapeake Bay's recreational fishermen from proposed oyster reef restoration programs. The model explicitly links historical oyster bottom conditions to recreational fishing catch to capture ecosystem and habitat benefits. Hicks determined that recreational anglers would realize benefits of approximately $640 thousand per year for restoring 1,890 acres at 73 reef sites in the Bay. The total cost of the restoration was determined to equal $27.0

million. When calculating the net present value of the 30-year stream of benefits, assuming a discount rate of three percent, it was determined that the recreational benefits would equal approximately half of the cost of the restoration.

Cost-based Methods

Following the methodology outlined by King and Mazzotta (2000c), the first step of a cost-based approach is to quantify the nutrient removal capacity of shellfish enhancement projects. This assessment would determine the current level of water quality, and the expected level of water quality after shellfish enhancement. As discussed above, researchers are applying theoretical models to assess the nitrogen reduction potential of shellfish enhancement.

The next step is to estimate the costs of providing a substitute or replacement for the nutrient removal services of shellfish enhancement. The dollar values of providing replacement nutrient reduction services by, for example, a wastewater treatment plant can be compared to shellfish aquaculture or restoration costs to determine whether shellfish enhancement is worthwhile in terms of water quality improvements. This comparison may be presented in terms of per unit nutrient reduction costs (Lindahl et al., 2005). The replacement value is calculated as the savings in costs from using shellfish enhancement instead of more costly nutrient abatement options.

Several studies have used cost-based methods to estimate the water quality improvement benefits of shellfish enhancement, although not all of the studies follow the above methodological steps. In an early application of the cost-based method, Newell et al. (2005) used U.S. Environmental Protection Agency estimates of the cost of reducing nutrient inputs necessary for meeting the water quality goals in the Chesapeake Bay. After converting this cost figure to an average per kg cost of removing nitrogen, Newell et al. used modeled nutrient reduction estimates to derive an annual replacement value of nitrogen removal by oyster reefs in the Choptank River, Maryland of $314,836, or $181 per hectare. The authors postulated that over the ten-year life span of the oysters the value of the nitrogen removal by the oyster reefs increased to $3.1 million, or over twice the dockside value of those same oysters.

Ferreira et al. (2007) used modeling results for a hypothetical 6,000 m^2 oyster farm to show an annual net removal of around 10,683 kg of nitrogen. If a standard population-equivalent (PEQ) of 3.3 kg of nitrogen per person per year is considered, the net nitrogen removal from the farm corresponds to an untreated wastewater discharge from over 3,000 PEQ. While noting that wastewater treatment costs per inhabitant are highly variable, the authors assumed an average unit treatment cost of about €650. The estimated replacement value of the oyster farm as regards nutrient removal is about €2 million per year, almost half of the estimated combined total annual income of the farm operation. In another modeling exercise, Ferreira et al. (2011) estimated that in a 11.4 hectare Manila clam farm on the southern coast of Portugal, shellfish filtration provides a net nitrogen removal of about 28,867 kg per year, which equates to the emissions of 8,748 PEQ. The authors estimated the replacement value of the clam farm to be about €0.26 million per year, about 10 percent of the direct income of the farm.

Using production and nutrient removal data to calculate the role of all European Union shellfish farms in removing nutrients, Ferreira et al. (2009), estimated that the farms removed a total of over 55,000 tons of nitrogen per year, that is, a PEQ of 17 million people, or the population of the Netherlands. The estimated replacement value is estimated to be €0.4 billion per year. Based on the worldwide reported shellfish aquaculture landings and modeling results of nitrogen removal for a typical range of cultivated species, the authors estimated a global replacement value of €7.5 billion.

Using a close approximation of the cost-based methodology described by King and Mazzotta, Gren et al. (2009) estimated the replacement value of nutrient cleaning by mussel cultures in the Baltic Sea using a nonlinear programming model that compares costs and impacts of mussel farms with other nutrient abatement measures such as wastewater treatment plants, changes in land use and fertilizer practices, increased cleaning by households and industries not connected to municipal wastewater treatment and creation of wetlands. The results indicate that calculated marginal cleaning costs of nutrients by mussel farming can be considerably lower than other abatement measures, with cost savings between 2 and 11 percent. The authors note that the cost-effectiveness of mussel culture in nutrient removal from the water column depended on mussel growth, sales options, assumptions about mussel farming capacity and the nutrient load targets.

Stadmark and Conley (2011) reported that compared to other potential nutrient abatement measures, mussel farming is a relatively expensive measure. Their calculations show that nitrogen reduction in mussel farms (industry mussels) is about two times more expensive per kg nitrogen than nitrogen reduction from wastewater treatment plants. However, Rose et al. (2012) point out that in estimating the cost of nitrogen removal, Stadmark and Conley did not count the revenue from the sale of shellfish biomass as an offset to costs. Rose et al. note that other studies (e.g., Gren et al. 2009; Miller 2009) have shown that the cost of nitrogen removal from shellfish production is highly contingent upon the returns that could be achieved through shellfish biomass (either for animal or human consumption). In addition, Rose et al. note that the costs quoted by Stadmark and Conley fail to reflect the wide range of costs (and cost uncertainty) associated with nitrogen removal, and that they substantially understate the upper bound limits of nitrogen removal costs from wastewater treatment plants. According to Rose et al. note, extensive evidence elsewhere demonstrates that these costs can increase rapidly as treatment approaches limits of technology and as smaller plants add tertiary nitrogen removal processes.

Water Quality Trading

Water quality trading is a "market-based" approach to achieving water quality standards within a defined area.[19] It allows a source discharging nutrients to the coastal zone to meet its regulatory obligations by using nutrient reductions created by another source that has lower pollution control costs (U.S. Environmental Protection Agency 2003). The economic argument for this approach is that it is able to achieve the resulting environmental outcome more efficiently (i.e., at a lower unit cost of abatement) than traditional "command-and-control" approaches because it gives sources the flexibility to search for the lowest cost pollution reduction option.[20] These nitrogen reductions are also called offsets because the reductions are made in order to compensate for or to offset an emission made elsewhere.

One potential option available to sources is the purchase of nutrient (e.g., nitrogen or phosphorus) credits from shellfish enhancement projects which are nutrient sinks (Ferreira et al. 2007). For example, the costs for controlling nitrogen loads at a point source such as a wastewater treatment plant compared to restoring shellfish production can vary significantly, creating the impetus for water quality trading. Through water quality trading, municipal and industrial wastewater treatment plants that face higher nitrogen removal costs—particularly as nitrogen concentration limits are tightened—could meet their regulatory obligations by purchasing "pollutant reduction credits" from,

[19] There are two basic forms of emission trading systems: cap-and-trade systems that based on an initial allocation of emission rights through a regulatory body, and baseline-and-credit systems (also known as offset systems) where emission credits are generated through the implementation of an activity which reduces emissions against a baseline (Profeta and Daniels 2005). The water quality trading approach discussed in this report is a baseline-and-credit system. The U.S. Environmental Protection Agency's Acid Rain Trading Program controlling sulfur dioxide emissions is an example of a cap-and-trade system.

[20] Under a command and control approach, the regulated entities are given little discretion in their pollution control efforts. For example, the government may mandate what pollution-reduction technologies they should use.

say, a shellfish aquaculture operation if the cost is lower and the nitrogen reductions achieve targets. The establishment of a market in which nutrient credits or offsets are freely traded can generate a market price for nutrient reduction. This market price represents the interests of wastewater treatment plants affected by nitrogen concentration limits to reduce nitrogen emissions in the most cost-effective manner.

Nitrogen credit trading at the watershed scale is now a reality in parts of the United States (Ferreira et al. 2011). In 2003, the U.S. Environmental Protection Agency issued a national water quality trading policy to support the development and implementation of trading in water quality management (U.S. Environmental Protection Agency 2003). The agency advocates water quality trading as a cost-effective means to preserve and improve water quality, and a number of programs are in various stages of development. While the policy doesn't specifically refer to shellfish enhancement as part of a nutrient trading scheme, it does support the creation of water quality trading credits in ways that achieve ancillary environmental benefits beyond the required reductions in specific pollutant loads, such as the creation and restoration of wetlands, floodplains and wildlife and/or waterfowl habitat. Using shellfish enhancement as part of a nutrient trading scheme would fulfill this objective, as shellfish enhancement would provide multiple economic and environmental benefits (Golen 2007).

In 2002, the Connecticut Department of Energy and Environmental Protection developed the Nitrogen Credit Exchange among 79 wastewater treatment plants to improve water quality in Long Island Sound, with over $30 million in economic activity in the first four years of trading (Ferreira et al. 2011). Similarly, the Chesapeake Bay states are developing water quality trading programs to provide regulated sources options to remain in compliance with their individual nutrient wasteload allocations, permit requirements or aggregate mass load caps (Maroon 2011; Stephenson and Shabman 2011).

Furthermore, a number of U.S states and countries outside of the United States are considering the use of shellfish in nutrient trading schemes in addition to other nitrogen reduction measures. As such, researchers are applying theoretical models to assess the nitrogen reduction potential of shellfish farms and for the valuation of nitrogen credits (Getchis and Rose 2011). The nitrogen reduction costs directly associated with a shellfish enhancement project are the project's costs of production, including the costs of labor and capital (see Section 4.1). If the shellfish produced by a project can be sold on the open market, the income from these sales can offset project costs (Stadmark and Conley 2011). The costs for nitrogen reduction measures also depend on the amount of nitrogen removed during the harvesting of shellfish (Stadmark and Conley 2011). As mentioned in Section 2.5, the nitrogen reduction potential of shellfish enhancement projects is the subject of intense (sometimes controversial) research.

Brumbaugh and Toropova (2008) and Lindahl et al. (2005) postulate that if a well-designed and regulated market existed for trading the nitrogen removed by shellfish, it should have the effect of spurring further investments in shellfish enhancement. By providing shellfish enhancement projects an opportunity to earn additional revenue when they sell nutrient reduction credits, a trading program allows projects involved in culturing shellfish and restoring beds to capitalize on the nitrogen removal services provided by their activities. However, Stephenson and Shabman (2011) note that participation of shellfish enhancement projects in a trading program is contingent upon buyers (e.g., wastewater treatment plant) and sellers (e.g., shellfish growers) reaching an agreement on the amount of credits to be provided to the buyer, how it will be documented that the credits were provided and the payment terms for the provision. Stephenson and Shabman suggest that these agreements will be described in a contract between the buyer and seller. The contract also will assure the buyer that the conditions set by the regulatory authorities are being met by the seller so that the credits can be used as offsets in the trading program. The authors note that nitrogen and other nutrient offsets lend themselves to measurement and verification, making credits especially amenable to a contracting process where payment is contingent on demonstrated results (Stephenson and Shabman 2011). In a similar vein, DePiper and Lipton (2011) discuss how an enforced contract can result in a switch in how an oyster reef is managed, e.g., in the absence of a contract, oysters may be harvested more aggressively, and consequently the desired nutrient removal levels may not be achieved.

In the aforecited article, Stephenson and Shabman also suggest that a time referenced benchmark would appear necessary in order to prevent an existing shellfish enhancement project to claim credits for past investments. Once such time referenced baselines are established, expansions of nutrient credit services (e.g., expanded oyster aquaculture production beyond the referenced date) could be counted new (additional) services and credited.

Some researchers are somewhat circumspect about participation of shellfish enhancement projects in nutrient trading schemes. Golen (2007) raises the issue of how to satisfactorily define and measure nitrogen removal when using shellfish. As discussed in Section 2.2.1, shellfish enhancement can remove nitrogen from the water column in two ways: 1) removal of nitrogen incorporated in the shell and tissues during harvesting and 2) the action of denitrifying bacteria on shellfish biodeposits (pseudofeces and feces). Golen notes that while it is relatively easy to account for nitrogen removed in shell and flesh based upon annual harvest levels, the factors that govern the magnitude of nitrogen removal via denitrification are too complex and variable to allow the use of fixed removal rates that can be applied across all shellfish aquaculture facilities. Nitrogen trading using shellfish could proceed based solely upon the quantification of nitrogen removal via biomass at harvesting, but this could substantially underestimate the nutrient reduction value of shellfish enhancement: nitrogen removal by oysters via nutrient burial and denitrification of biodeposits is estimated to be seven times the weight of nitrogen contained in the shellfish meat (Stephenson and Brown (2006); cited in Golen (2007)).

Stadmark and Conley (2011) suggest that the primary focus for nutrient mitigation should be on nutrient reductions from land-based sources before nutrients enter into the coastal zone. However, Rose et al. (2012) argue that this view may be somewhat simplistic. Rose et al. agree that nutrient mitigation should not be used in lieu of land-based nutrient reductions, but they note that as conventional treatment methods approach limits of technology (wastewater treatment plant upgrades) or become very expensive (urban stormwater retrofits), many estuaries will still need to find additional nutrient removal methods to meet water quality objectives.

Lindahl and Kollberg (2009) support shellfish enhancement as a nutrient compensation measure but argue that it should be reserved for diffuse (nonpoint source) emissions into the coastal zone coming from, for example, agriculture, because for these there are few other effective options available. Rose et al. (2012) also note that nutrient removal by shellfish becomes more important as the main nutrient loading from land increasingly shifts to diffuse sources. According to the authors, as the majority of point source contributions are reduced, the law of diminishing returns makes it far more expensive to reduce the remainder of the nitrogen load to coastal ecosystems, largely attributable to agricultural and diffuse urban sources.

While shellfish enhancement projects have yet to become active participants in a U.S. nutrient trading program, a number of studies have examined the credit generation potential of these projects. Stephenson et al. (2010) describes a bioeconomic model used to estimate the cost of providing a nitrogen offset from commercial oyster aquaculture. The model estimated cost by calculating the supplemental compensation necessary for a commercial aquaculture operation to produce additional oysters. The study calculated total nitrogen removed from ambient waters based on the amount of nitrogen sequestered in oyster tissue and shell at harvest and nitrogen removed through denitrification of oyster biodeposits. The estimated annual cost to generate one pound of nitrogen offset ranged from $0 to $150, depending on assumptions about the market price of harvested oysters, project input costs, and oyster growth and mortality rates. According to the authors, conceptually, costs may be zero because in some economic circumstances oyster growers would be willing to expand production without any compensation for nutrient removal services. Estimated total nitrogen removed for every one million market-sized oysters harvested ranged between 290 and 815 lbs per year, depending on assumptions of denitrification rates.

Piehler and Smyth (2011) compared the ecosystem service value of nitrogen removal in five of the major habitat types found in temperate estuaries (salt marsh, submerged aquatic vegetation, oyster reef, intertidal flat and subtidal flat). The authors estimated a dollar value of nitrogen removal using the rates from the North Carolina nutrient offset program. Under this program, the nutrient offset payment price is not set by the market but rather is a regionally derived number that had significant stakeholder input in its determination. The trading price of the North Carolina nutrient offset program at the time of the study was $13 per kg of nitrogen removed. To best estimate the annual value of habitat specific nitrogen removal, the authors multiplied this price by the mean annual rates of denitrification and the standard error of these means. The authors estimated that submerged aquatic vegetation and oyster reefs provide about $3,000 per acre per year in nitrogen removal services, while wetlands provide about $2,500 per acre per year of nitrogen removal service.

Outside of the United States one can find examples of shellfish enhancement projects being incorporated into nutrient trading schemes. For instance, Lindahl and Kollberg (2009) described what they called the first case in Sweden of trading a nitrogen emission. The Swedish community of Lysekil was permitted, as a trial between 2005 and 2011, to continue to emit nitrogen from its wastewater treatment plant into an estuary provided that the same amount of nitrogen was ''harvested'' and brought ashore by 3,900 mt of farmed blue mussels. According to the authors, the payment of €150,000 that Lysekil made to the mussel farming enterprise contracted for the nitrogen removal was far below the cost for nitrogen removal in the local wastewater treatment plant.

3.2.2.2 Shoreline Protection and Stabilization of Submerged Land

As described in Section 2.2.2, the physical structure of an oyster reef can protects shorelines and inland waters from the erosive force of waves. Henderson and O'Neil (2003) suggest that in the absence of other data, cost-based methods could be used to estimate the economic benefit of an oyster reef for shoreline protection or sediment stabilization. For example, cost-based methods might answer the questions, "if the oyster reefs weren't there, are there structures or other ways to provide the same services the reefs perform?" and "what is the cost of providing the substitute?"

Previous efforts to protect shorelines have largely involved constructing bulkheads and seawalls (National Research Council 2007a). Both of these methods have been shown to cause vertical erosion down the barrier, subsequent loss of intertidal zone and even increased erosion on adjacent properties (Scyphers et al. 2008). The benefits of oyster reefs as an alternative to the hard bulk-heading of shorelines have not been valued in the literature. However, Conservation International (2008) discusses the results of several studies that analyzed the economic contribution of shoreline protection services provided by coral reefs based on estimates of the extent of shoreline protected by coral reefs and the costs required to replace the reefs by artificial means.

As discussed in Section 3.1.1.1, cost-based methods do not provide strict measures of the economic value of ecosystem services, and may misrepresent the value in certain circumstances.

3.2.2.3 Property Values

Changes in the value of waterfront property may reflect the full range of positive (and negative) impacts of shellfish enhancement, including local changes in environmental amenities such as water quality and shoreline protection. Consequently, any expected effects on those other factors may be expressed in terms of effects on property values. Estimating the value accruing to private tideland owners is an especially critical part of the balance sheet for shellfish enhancement in states such as Washington, where 75 percent of tidelands is owned privately (Peter-Contesse and Peabody 2005). Moreover, the added value from waterfront properties not only has implications for added wealth for the buyers and sellers of these properties; it also has value to local communities and states through accompanying real estate taxes and property transfer taxes in many states (Kildow 2008). Yet, in comparison to activities directly benefiting from restoration projects such as recreational fishing and beach use, real estate values have received far less attention from researchers (Kildow 2008).

Kildow (2008) discussed a number studies that use various economic valuation methods to examine how coastal preservation and restoration influence housing prices for waterfront homes. Included in this literature survey are two hedonic property studies that demonstrate the impact of water quality on housing values along the Chesapeake Bay; one study was conducted in Anne Arundel County, Maryland (Leggett and Bockstael 2000), and the other was conducted in the St Mary's River Watershed (Poor et al. 2006). A third study by Braden (2004) that combined two methods, the hedonic method and conjoint analysis, used housing market data and also measured the preferences of homeowners in Lake County, Illinois.

The effect of shellfish enhancement on the value of waterfront property may not necessarily be positive. As aesthetic values associated with shorefront property have increasingly become more important, the so-called not-in-my-backyard (NIMBY) factor has come into play: waterfront property owners do not want their views affected by commercial ventures, including shellfish aquaculture operations (National Research Council 2010). This attitude is most likely to occur in communities or settings where shellfish aquaculture has not been part of the established or traditional waterfront or where there is a large influx of residents who do not share the community's cultural fishing traditions (National Research Council 2010). Landowner opposition may also arise when changes occur in the species cultivated that result in the use of more intensive culture methods (Dewey et al. 2011). Shellfish growers can design their operations to minimize visual and water access conflicts, and it is possible that prospects for permitting of shellfish enhancement projects can sometimes be improved by educating the local community about the ecological benefits (e.g., water filtration, nutrient removal, habitat enhancement for valuable finfish and crustaceans) of those projects (National Research Council 2010). However, anecdotal information presented in a number of studies (e.g., Kite-Powell 2009; Northern Economics 2010) suggests that shellfish aquaculture operations may lower the value of real estate in some coastal communities. As will be discussed in Section 4.2, any such decrease in property value should be considered a shellfish enhancement cost.

3.2.3 Supporting Services

As discussed in Section 2.3, the structure provided by shellfish reefs and beds serves as habitat for other species of fish and crab, and protects the ecological integrity of other habitats, such marshlands and seagrass beds, which also support a wide variety of marine life. These improvements may ultimately lead to measurable increases in the production of additional types of finfish and invertebrates targeted in commercial and recreational fisheries. Larger populations of important commercial and recreational species potentially mean significant contributions to economic welfare in the form of greater industry profits and consumer benefits.

Few studies have yet sought to quantify secondary production attributable to shellfish enhancement. Peterson et al. (2003) reviewed available empirical data on quantitative enhancement of nekton populations by restoring oyster reefs in the southeast United States and applied demographic and growth models to estimate the species specific augmentation of fish and crustacean production that is expected per unit area of oyster reef restoration. They estimated that 10 m^2 of restored oyster reef yielded an additional 2.6 kg per year (2,600 kg per hectare per year) of production of fish and large mobile crustaceans for the functional lifetime of the reef. Grabowski and Peterson (2007) converted the amount of augmented production per each of the species groups that were augmented by oyster reef habitat in Peterson et al. to a commercial fish landing value. According to the researchers, for fish of commercial significance, the enhanced production by the reefs equates to $3,700 per hectare per year and, over a 50 year time span, the fish productivity would exceed the anticipated value of directed oyster harvest from the same area by more than 34 percent.

A study in North Carolina (West Bay, Neuse River) by Lenihan and Grabowski (1998) compared the value of fish and crab from three oyster reefs to the value of harvest from adjacent unstructured sand bottom areas. A total of 15 commercially valuable species were found to utilize restored oyster reef habitat. The study results indicated that the long-term commercial value of these fish and crabs was greater than the value of the oyster production.

Recreational anglers who are aware of the species variety and abundance of fish available over oyster reefs value the reefs for the enhanced recreational fishing opportunities they provide. Isaacs et al. (2004) employed the contingent valuation method to estimate the value of Louisiana's oyster reefs as recreational fishing grounds, using a sample drawn from resident saltwater anglers who participated in the National Marine Fisheries Service's Marine Recreational Fishing Statistical Survey. A telephone survey was conducted featuring a dichotomous-choice net willingness to pay question. The average annual net willingness to pay among resident saltwater recreational fishermen to maintain access to recreational fishing over Louisiana's oyster reefs was $13.21.

3.2.4 Cultural Services

Wilson and Farber (2008) note that opportunities for recreation and natural amenities (e.g., productive sport fisheries, clean beaches) get an inordinate amount of attention in the economic literature, while other benefits such as aesthetic, spiritual and historic do not get much attention at all. Although a number of qualitative descriptions of the culture services provided by restored, enhanced or managed shellfish populations appear in the literature (see Section 2.4 for examples), no example of an attempt to measure the benefits of these services in monetary terms could be found.

3.2.5 Non-Use Values

As described in Section 3.1.1.2, non-use values include existence value and option value. For example, Hicks et al. (2004) suggested that people may benefit from oyster reefs in Chesapeake Bay even if they do not directly use the environmental asset by either deriving value from knowing that oyster reefs exist and provide ecosystem services (existence value) or from knowing that improved environmental conditions might make future use of the Bay more enjoyable should they choose to use the Bay directly (option value). To measure these non-use values Hicks et al. used the contingent value method. Based on the results of a mail survey of residents of New Jersey, Delaware, Maryland, Virginia, and North Carolina, the researchers determined that the non-use value of a ten-year oyster reef project, consisting of 10,000 acres of oyster sanctuary and 1,000 acres of artificial reef, to be at least $14.91 per household per year with a median estimate of $86.86 per household per year. Aggregating to the general population, the researchers estimated the total non-use value of the project to be at least $114.95 million.

3.3 Avoiding the Double Counting of Benefits

The above sections have examined appropriate methodologies for estimating the benefits arising from various categories of ecosystem services provided by shellfish enhancement. However, the benefits estimated by each methodology can not necessarily be added to arrive at a measure of the overall value of such projects—there may be some overlap between values estimated by different methodologies.

For example, the recreational value of improvements in the quality of coastal waters may be measured using travel cost methodologies, yet may also influence the value of local homes (e.g., residents may wish to live in close proximity to valued recreational resources), thereby influencing hedonic property value estimates. Moreover, a portion of the recreational value may be reflected in residents' net willingness to pay for water quality improvements (Johnston et al. 2002). In these and many other cases, summing of value estimates from different methodologies may double-count the same values. Analysts must consider this potential for overlap when developing a comprehensive, integrated approach to assessing the benefits of shellfish enhancement.

4 Costs of Shellfish Enhancement

Not all the effects of shellfish enhancement are necessarily beneficial. An economic analysis must consider the costs as well as the benefits, where cost is represented by the value of the next best alternative foregone as the result of making a decision, i.e., the opportunity cost. This section describes the potential costs that may be incurred in efforts to protect or restore a shellfish population. A summary of these potential costs is provided in the table below.

Table 3 Costs of Shellfish Enhancement

Project costs	Start-up and operating costs (capital, labor, fuel, seed, etc.)
Environmental costs	Aquaculture (changes in community composition, habitat transformation, genetic impacts on wild populations, coastal use conflicts, etc.) Introduced shellfish species (trophic interactions, habitat transformation, interactions with native species, etc.)
Human health costs	Shellfish poisoning (lost wages and work days, medical treatment, poisoning cause investigations, shellfish monitoring programs, etc.)

4.1 Project Costs

As discussed in Section 3.2.1.1, several studies have examined the costs (as well as the revenues) of commercial shellfish fisheries and aquaculture operations. However, Henderson and O'Neil ((2003) note that the costs incurred in designing and implementing shellfish restoration projects have been more difficult to fully account for because they are typically borne by public entities. The authors also observe that the opportunity costs (e.g., spending public monies for other purposes) and secondary and higher order costs are impossible to completely track. Restoration project costs include several major elements: planning and project development; the cost of eyed oyster larvae and remote setting equipment; bottom cultch and placement material; other equipment and maintenance; transportation and vehicle costs; a baseline survey; the cost of a monitoring program; data analysis and interpretation; enforcement; infrastructure; labor; and project implementation (these costs reflect the management personnel expense needed to manage project activities) (Industrial Economics 1999).

As noted in Section 1.2, Brumbaugh et al. (2007) indicate that even relatively small restoration projects can be costly (e.g., > US $100,000 per acre for restored oyster reef). According to The Nature Conservancy (2008), the surveying and up-front permitting requirements alone can be extensive, lengthy and costly. Project costs vary substantially with the purpose of the project as well as the size. For example, Henderson and O'Neil (2003) note that Maryland and Virginia both constructed oyster reefs in Chesapeake Bay using base material of oyster shell. The cost of shell base materials for both the Virginia and Maryland reefs was about $10,000 per acre. However, the Maryland reefs were also initially seeded at a cost of $10,000 per acre and maintained annually by addition of broodstock and more shell base material, creating a "put and take" fishery. The initial stocking and addition of maintenance broodstock resulted in higher productivity, but at an increased cost.

4.2 Environmental Costs

Though to a far lesser extent than finfish, shellfish aquaculture can also have undesirable effects on the environment (Sequeira et al. 2008; National Research Council 2010). It may result in changes in benthic community composition through a range of mechanisms, such as excessive partitioning of food resources, competition for space and increased sediment deposition (Sequeira et al. 2008). For example, Bendell-Young (2006) found that intertidal regions in British Columbia that had been used for intensive farming of Manila clams, *Venerupis philippinarum*, were characterized by a decrease in species richness, altered species abundance and distribution, change in community structure composed of surface species, sub-surface species and bivalves, to one composed primarily of bivalves, and greater accumulations of surface sediment silt and organic matter.

Submerged and floating shellfish cultivation gear may also have negative impacts on essential marine habitats. For example, the physical disturbance caused by oyster cultivation gear may cause deterioration of eelgrass beds, an essential habitat for juvenile fish and shellfish (Getchis 2005; National Research Council 2010), and physical alteration to prospective geoduck aquaculture sites through grading and rock removal may result in damage to ecological functions (Washington Department of Ecology 2009). Mechanical harvesters, commonly used for scallops and clams, can create significant environmental stress, damaging the ocean floor and harming benthic plant life and other wild species (Brumbaugh et al. 2006; Stokesbury et al. 2011; SeaChoice undated). However, harvesting of culture plots with mechanical harvesters is less destructive than harvesting wild shellfish because harvest is restricted to relatively small plots (SeaChoice undated).

In addition, there is the risk of contaminating wild shellfish populations with cultured genes. With culture of a native species, this risk centers on the potential loss of natural genetic variation, which serves to buffer the population against natural selective forces (Straus et al. 2008). In some aquaculture operations, “seed” (juveniles) can be taken from the wild; however, seed scarcity has forced other operations to turn to hatcheries (Cross et al. 2008). For example, geoduck growers obtain seed from hatcheries. Since the geoducks planted by aquaculture operations may reproduce before harvest and all bivalve molluscs are broadcast spawners, there is a potential for the cultured clams to interact with the genetics of the wild populations (Washington Department of Ecology 2009). Moreover, diseases and parasites carried by hatchery seed may be introduced into areas where they currently do not exist, with a consequent deleterious effect on wild shellfish populations (Cross et al. 2008; Straus et al. 2008; Washington Department of Ecology 2009).

The extensive use of nets, docks, cages and other gear by shellfish farms may also conflict with navigation, dredging, commercial and recreational fishing, swimming and other users of coastal waters and the adjoining shoreline. Moreover, shellfish farms may have to compete for space with residents who have shoreline vistas. Farms are usually hidden below the waterline, but at low tide much of the gear is exposed (Thacker 2006).[21] In a review of aquaculture siting issues in Washington State’s coastal zone during the 1970’s, Evans (1979) reports that among the complaints expressed by concerned citizens before the Kitsap County Board of Commissioners was that hydraulic harvesting of subtidal hardshell clams destroys subtidal aquatic environments and associated plants and animals; causes extraordinary amounts of broken clam shells to be washed up on nearby beaches; results in unacceptably high levels of turbidity (amount of suspended solid materials in the water); and is unacceptably noisy and interferes with the residential character of the area. At the time Evans’ report was written, harvesting of hardshell clams was only one example of a growing controversy over

[21] Sommers and Canzoneri (1996) note that Puget Sound growers have been increasingly using floating aquaculture structures. According to the authors, floating aquaculture is often associated with heightened use conflicts because it occupies surface water and allows more intensive growing operations.

aquacultural use of aquatic areas in Washington. Conflict also developed over the siting and development of mussel rafting operations in Island and San Juan Counties and oyster farming in Pierce County and Hood Canal (Evans 1979).

More recently, the conversion of some of Washington's intertidal beaches to geoduck aquaculture has resulted in conflicts with a number of shoreline residents who feel geoduck aquaculture alters the nature of their shorelines (Coalition to Preserve Puget Sound Habitat 2007). Some shoreline residents dislike having what they see as an industrial activity occurring near them, and many people are concerned that geoduck aquaculture will harm the ecological functions of the shorelines (Washington Department of Ecology 2009).

Because shellfish are ecosystem engineers, translocations of native species and introductions of nonnative species can have disproportionately high environmental impacts, many of which are potentially undesirable (Ruesink et al. 2005). Such translocations and introductions have been employed to augment or replace depleted natural stocks or to diversify the number of species used in aquaculture operations (National Research Council 2010). Non-native shellfish species that are brought into new environments may indeed provide such valuable services as water column filtration, habitat creation for non-shellfish species and stabilization of estuarine sediments, but they can also may compete with native species, negatively affect food webs and bring in new diseases and other undesirable species (Brumbaugh et al. 2006).

Competition between native and introduced shellfish is expected to be most intense if they share similar habitat. For example, the suminoe oyster was identified as a candidate for introduction to Chesapeake Bay because its salinity and temperature requirements closely match those of the eastern oyster; therefore, the two species would be likely to occupy the same habitat (U.S. Army Corps of Engineers 2008). Researchers have concluded that the risk is moderate to high that suminoe oysters would interact and compete with eastern oysters (U.S. Army Corps of Engineers 2008). Ruesink et al. (2005) indicate that on the western coast of North America, the native Olympia oysters tend to occur at lower depths with less temperature stress than does the introduced Pacific oysters. The authors note that it is reasonable to predict that Pacific oysters would occupy a higher tidal elevation than do Olympia oysters, and that, in places where Pacific oysters reached high density, they would transform habitat and increase epifaunal diversity. Thus, Pacific oysters would perform a novel ecosystem role in western North American estuaries (Ruesink et al. 2005). Moreover, Olympia oysters filter food particles that are smaller than those taken by Pacific oysters and, thus, serve slightly different ecological roles in controlling phytoplankton blooms (Peter-Contesse and Peabody 2005).

However, Ruesink et al. (2005) observe that competition between oyster species also occurs indirectly through habitat modification. Recent evidence suggests that Olympia oyster larvae disproportionately settle in areas with large accumulations of shell. Because intertidal Pacific oysters comprise most of the shell habitat in the bay, the native oysters only have the option of recruiting to zones where immersion times are too short for survival. Thus, the introduced oyster has developed into a recruitment sink for natives, particularly in the absence of remnant subtidal native-oyster reefs (Ruesink et al. 2005).

Species brought in with shellfish aquaculture can present problems for the continued production of the cultured shellfish species in addition to potentially interacting with native species and altering the structure and function of surrounding communities and ecosystems. For example, Ruesink et al. (2005) report that *Batillaria attramentaria*, an Asian snail introduced to the U.S. West Coast with the Pacific oyster, outcompetes the mud snail *Cerithidea californica*, which has caused local extinction of the native snail in a number of estuaries. In addition, the shell-boring sabellid polychaete, *Terebrasabella heterouncinata*, also introduced with the Pacific oyster in California, infested cultured red abalone, *Haliotis rufescens*, with great economic consequences to growers before it was successfully eradicated. Based on their review of the unintended consequences of shellfish introductions, Ruesink et al. (2005) conclude that the potential ecological costs of the deliberate introduction of shellfish suggest that native shellfish are a better option for ecosystem restoration.

4.3 Human Health Costs

Because they are capable of filtering large volumes of water relative to their size, shellfish may ingest and concentrate undesirable pathogens and other toxic pollutants in their tissues when filtering contaminated water. Although these pathogens have little effect upon the shellfish themselves, they can be harmful to the human consumer, causing diseases such as hepatitis (Busse 1998). Toxins produced by harmful algae can also be concentrated by shellfish through filter feeding to the point that the shellfish become dangerous or even deadly for humans to eat (NOAA Fisheries Northwest Fisheries Science Center and Washington Sea Grant Program 2002). Paralytic shellfish poisoning, neurotoxic shellfish poisoning, diarrhetic shellfish poisoning and amnesic shellfish poisoning are caused by eating scallops, mussels, clams, oysters and cockles contaminated with various toxins (Bauer 2006).

Shellfish enhancement projects must take into account the potential costs of shellfish consumption-related illnesses. Human sickness and death from eating tainted shellfish result in lost wages and work days. Costs of medical treatment and investigations for the cause of the sickness can also be significant. Further, individuals who are sick may experience pain and suffering (Hoagland et al. 2002). Hoagland et al. (2002) estimated the costs of various types of shellfish poisoning to be $1,400 per reported illness, $1,100 per unreported illness and $1 million per mortality. Based on these cost estimates and the estimated annual number of shellfish consumption-related illnesses and deaths in the United States, the researchers calculated that the public health costs of shellfish poisoning averaged $400 thousand per year.

In addition, the costs of shellfish monitoring programs must be considered. Most of the states in which shellfish poisoning is a significant problem operate such programs. Boesch et al. (1997) estimated that each program costs $100,00 to 200,000 per year, while Hoagland et al. (2002) estimated that the total annual cost of U.S. shellfish surveillance programs to be approximately $2 million.

Consumer concerns about the safety of shellfish consumption in regard to human health can lead to a downward shift in the demand for shellfish products and loss in consumer surplus. This loss reflects the fact that when consumers become concerned about seafood safety, the maximum amount they are willing to pay for any quantity of shellfish is less than when they are not concerned about safety (Lipton and Kasperski 2008). Keithly and Diop (2001) examined the extent to which the demand for Gulf-area oysters has been reduced as a result of mandatory warning labels and negative publicity. In general, their results suggested that since 1991 the “summer” dockside price had been reduced by about 50 percent as a result of warning labels and associated negative publicity, while the “winter” dockside price has been reduced by about 30 percent.

On the other hand, increases in demand for shellfish and gains in consumer surplus can be achieved if consumers perceive that certain shellfish products are considered safer and perhaps more environmentally-friendly (Shumway et al. 2003; Trauner 2004; Sequeira et al. 2008). A measure of the value of food safety is an individual’s net willingness to pay to reduce the health risks from eating shellfish. Lin and Milon (1995) and Lin et al. (1995) used the contingent valuation method to estimate consumers’ net willingness to pay for shellfish products that are less likely to cause illness. In both studies, a substantial number of consumers showed a positive net willingness to pay for safer oysters, and Lin et al. found that individuals who had been sick from eating unsafe oysters generally valued safer oysters more than others. The contingent valuation method was also used by Appéré and

Bonnieu (2003), who estimated how much recreational shellfish gathers would be willing to pay for a gathering site where the risk of contracting an illness is low. Instead of being asked directly to place a monetary value on the provision of a hypothetical site, respondents were asked how far they would drive to use such a site. This study also found a substantial number of consumers who held a positive net willingness to pay for safer shellfish.

Of course, if a price premium is charged in the retail market for shellfish that pose a lower health risk to consumers, the economic benefits accrue to the sellers rather than the buyers. For example, Sommers and Canzoneri (1996) state that shellfish from Washington command a premium price in the domestic market because they have gained a reputation as a superior product that is safer to eat than shellfish produced elsewhere in the United States. The authors note that this reputation rests largely on the strength of Washington's shellfish monitoring program and the emphasis that regulatory officials and shellfish growers in the state place on water quality.

4.4 Best Management Practices and Standards

The implementation of best management practices (BMPs) and regulatory standards represent one approach to reduce or minimize adverse environmental and social effects, food safety issues and other public concerns resulting from proposed shellfish enhancement projects. BMPs and standards are general overarching principles and specific procedures or methodologies used to guide the day-to-day operation of aquaculture businesses (Getchis and Rose 2011). They are often developed by the industry group (e.g., shellfish growers) to which they apply but also can be driven by regulatory agencies or other nongovernmental organizations. Adoption of and adherence to BMPs is usually voluntary, while regulatory standards usually are imposed by a public authority (i.e., federal, state, or local agencies responsible for permitting and oversight of shellfish aquaculture); compliance is often required by law as a condition of the permit (National Research Council 2010). Lists of BMPs and standards that can be applied across shellfish species and conditions are provided by Dewey et al. (2011), Hargreaves (2011) and the National Research Council (2010).

Expenditures of time or money in conforming with BMPs and standards increase the project costs of shellfish enhancement (National Research Council 2010). Some shellfish growers question whether BMPs and standards are attainable and affordable for the average aquaculture operation (Getchis and Rose 2011). However, some BMPs and standards have been proven to increase efficiency and hence profitability, while reducing environmental impacts (e.g., the use of triploid oysters in order to mitigate against successful reproduction and spread in areas where the oyster is nonnative) (National Research Council 2010). Moreover, some BMPs and certification standards for shellfish products (e.g., organic, sustainable, fair trade, domestically or even locally grown) can be used as a marketing tool to enhance the value of those products (National Research Council 2010). Hargreaves (2011) notes that product certification for shellfish aquaculture is a work in progress and provides a review of its current status.[22] Although consumer demand for "ecolabeled" seafood is growing, Hargreaves cautions that considerable uncertainty remains about the price premium, market access or other value that can be obtained by producers of ecolabeled shellfish.

The National Research Council (2010) noted that in settings where shellfish aquaculture is carried out by a number of small, independent operators, it makes sense to develop BMPs or standards that set parameters based on system-wide ecological carrying capacity (defined as the aquaculture biomass

[22] According to its website, the Aquaculture Stewardship Council (ACS), an independent, nonprofit organization, will be the world's leading certification and labeling program for responsibly farmed seafood (Aquaculture Stewardship Council 2012). The ACS certification process will allow for shellfish producers to use an ecolabel identifying the product as environmentally sustainable (Getchis and Rose 2011).

production above which unacceptable ecological impacts arisesee Inglis et al. (2000) and McKindsey et al. (2006)), thereby taking into account the cumulative effects of all farming operations. Ecological carrying capacity is typically calculated using ecosystem modeling techniques (see Section 2.5). Stakeholders (e.g., habitat and aquaculture farm managers, commercial and recreational fishermen, coastal land owners and nongovernmental organizations) can help identify ecosystem components that need to be evaluated in unbiased ecological carrying capacity assessments as well as contribute to decisions about what are acceptable or unacceptable environmental impacts (National Research Council 2010; Byron et al. 2011). According to the National Research Council (2010), BMPs or standards that target parameters related to ecological carrying capacity can be focused on shellfish aquaculture broadly and may not have to address each location, species and culture technique separately. However, given that the ability to quantify and measure ecological carrying capacity remains limited, adopting this approach will require careful consideration of the risks, the acceptable level of ecological change and the appropriate parameters to monitor.

5 Economic Valuation Issues and Considerations

5.1 Distribution of Benefits and Costs across Society

In assessing the economic value of a shellfish enhancement project, the economic efficiency criterion requires only that there be a net positive change in welfare for society as a whole in order for the project to be justified. While delivering net benefits to society as a whole is important and should be given due weight, planners and decision makers are well aware that it is only one consideration—the way in which the costs and benefits of the project are distributed within society (i.e., who receives the benefits, who pays the costs) can also be important. Typically, planners and decision makers have to consider the extent to which such distributional impacts are important and need addressing.

The benefits estimated by different economic valuation methods are, in many cases, realized by different segments of the population (Johnston et al. 2002). For example, the hedonic pricing method assesses benefits (and costs) realized by local coastal property owners, while the travel cost analysis estimates values received by both local residents and tourists.[23] The market price method, in contrast, measures economic values realized off-site by a range of resident and non-resident user groups who supply and consume shellfish products. Other ecosystem services may also occur off-site; for example, when juvenile finfish sheltered by an oyster reef are caught at a later life stage miles away, or improvements in water quality arising from an oyster reef extend far beyond the reef itself (Johnston et al. 2002; Henderson and O'Neil 2003). Still other benefits may accrue to people who receive enjoyment from simply knowing that the cultural values associated with the lifestyle of the watermen are being preserved (Wasserman and Womersley undated). In short, how certain individuals will be affected by a shellfish enhancement project that alters the flow of ecosystem services will likely depend on the type of service. Services that are actually traded in markets; services that are not marketed but consumed on-site; services that are not marketed and produce ecological effects off-site—each may have a different affected population (Plummer 2009).

5.2 Distribution of Benefits and Costs through Time

As described above, a protected or restored shellfish population may be viewed as an environmental asset that provides a stream of ecosystem services over time. An important factor for planners and decision makers to consider when weighing the benefits and costs of shellfish enhancement is the time frame over which the benefits and costs occur. When making decisions about changes to an ecosystem, a long time horizon is typically needed (Ludwig et al. 2005). For example, in discussing the role of the timing of benefits and costs related to the potential introduction of the Pacific oyster into Chesapeake Bay, Lipton (2008b) notes that it would take several years to restore oyster resources in Chesapeake Bay to a level that would support the level of a fishery. Since much of the restoration expense will occur early in the process, this timing will have an impact on calculation of net benefits.

Economic discounting is the process of weighting the sequence of costs or benefits over time (Ludwig et al. 2005). In general, individuals will discount values of things in the future in comparison to the same things in the present. To reflect this positive time preference, the standard economic approach is to use a discount rate to express a time series of effects as an equivalent present value. The level of the discount rate substantially affects the importance of services that occur far in the future—even

[23] As discussed in Section 3.2.2.3, it is possible that waterfront property owners in close proximity to large-scale shellfish cultivating operations may have negative values for this coastal zone use, related to such negative amenities as odors, noise, visual disturbance and other nuisances associated with such operations.

small differences in a discount rate for a long-term restoration project can result in order-of-magnitude differences to the present value of net benefits (National Research Council 2004).

Where the basis of comparison is strictly financial, economists feel confident using the opportunity cost of invested funds as the discount rate. However, for most ecosystem services, a financial opportunity cost is inappropriate and forces an emphasis on services that occur in the near future (National Research Council 2007b). Ludwig et al. (2005) argue that an appropriate consideration of economic and ecological uncertainties would lead planners and decision makers to discount benefits of ecosystem services at the lowest possible rate over long time horizons.

5.3 Limits of Economic Valuation

Apart from the technical problems with economic valuation methods discussed in Section 3.1, planners and decision makers should also be aware that there are criticisms of the basic principles underlying the application of these methods, especially with respect to valuing ecosystems. Some of these criticisms contend that economic valuation methods are inherently inadequate because they are based only on the preferences of the current generation and neglect the ethical issue of the intergenerational allocation of natural resource endowments. For example, Berrens et al. (1998) note that irreversible ecosystem losses involve intergenerational equity issues because they constrict the choice sets of future generations.

Other critics focus on the fact that economic valuation is rooted in anthropocentric or human-centered benefits, i.e., it rests on the basic assumption that value derives from what people find useful. Some would argue that human uses and the values to which they give rise are not deserving of any special consideration when it comes to a decision on whether or not to preserve an ecosystem or a certain species (National Research Council 2004). This non-anthropocentric or biocentric viewpoint assumes that all living things have value even if no human being thinks so. According to one interpretation of this notion of non-anthropocentric intrinsic value, non-human species have moral interests or rights unto themselves (National Research Council 2004).[24]

This reference to morals, rights and duties implies an ethic that rejects the assumption that humans even have a choice regarding whether or not to protect a species or ecosystem; rather, it is seen as an obligation (Mazzotta and Kline 1995; National Research Council 2004). These arguments are inconsistent with the economic principle of trade-offs between the provision of an ecosystem service and something else that can also be quantified by the dollar metric. Instead, they present individuals with the moral imperative that we ought to preserve all ecosystems and associated species (Mazzotta and Kline 1995). Brumbaugh and Toropova (2008) allude to these precepts when they note that some individuals around the U.S. who have initiated small-scale, community-based shellfish enhancement projects are motivated by the belief that the ecosystem services these projects provide have intrinsic value.[25]

As demonstrated by an increasing number of studies (e.g., Edwards 1986; Mazzotta and Kline 1995; Kotchen and Reiling 2000; Rekola et al. 2000; Spash 2000), the presence of such motivations can be a significant source of validity problems for economic valuation methods. As Costanza et al. (1997)

[24] The conceptual framework for The Millennium Ecosystem Assessment places human well-being as the central focus for assessment, but it recognizes that the actions people take that influence ecosystems result not just from concern about human well-being but also from considerations of the intrinsic value of species and ecosystems (The Millennium Ecosystem Assessment 2005).

[25] Flimlin (2008) succinctly summarizes the motivation question as follows: "… what, or who, has influenced these people to focus their careers, livelihoods, or hobbies toward small invertebrates locked within two opposing shells?"

and Pearce and Moran (1994) state, concerns about the preferences of future generations or ideas of intrinsic value translate the valuation of ecosystem services into a set of dimensions outside the realm of economics.

The issue of intrinsic value within economic valuation is currently unresolved (Hanley 2000). The National Research Council (2004) notes that this dilemma raises the question of what, if any, metric might be used to quantify, or at least rank, intrinsic values. According to the National Research Council, choices made by planners and decision makers are often made on the basis of information about many sources of value, including intrinsic and moral values, as well as economic values, and some decision rules seek to incorporate different types of values explicitly. As an example, the National Research Council describes decision rules that imply adherence to moral principles or a premise of intrinsic value unless the cost is too high; these decision rules incorporate concern about both intrinsic value and economic welfare, and implicitly allow some trade-offs between the two. Similar trade-offs are also implied by decision rules that apply a benefit-cost test to environmental policy choices but constrain the decisions to ensure that certain conditions reflecting intrinsic value are not violated (National Research Council 2004). Possible constraints include ensuring that levels of critical ecosystem services do not fall below standards necessary to ensure their continuation. In such cases, information about benefits and costs as determined by economic valuation will be a useful input into the policy decision but will not solely determine it (National Research Council 2004).

6 Economic Impact Analysis

Economic valuation and economic impact analysis are two widely used but distinctly different economic measures (TCW Economics 2008). As discussed in Section 3, economic valuation is a measure of the net economic welfare derived by society from policy or program changes. Economic impact analysis, on the other hand, provides planners and decision makers with information on how policy changes affect economic activity, as measured in terms of sales/output, value added, income, employment and tax revenues, in communities, counties, or even at the state or national level (TCW Economics 2008). Economic impact analysis is the focus of our Task 3 work.

Economic impacts are typically estimated using an input-output model—a methodology that models the linkages between input supplies, outputs and households in a regional economy that can be used to predict the impact of changes on economic activity within the region. Through the use of multipliers an input-output model provides measures of the total effects throughout the economy of a unit change in direct or initial spending, including indirect effects (businesses buying and selling to each other) and induced effects (household spending based on the income earned from the direct and indirect effects).

Several studies have measured the economic impacts of projects related to shellfish enhancement. Based on the results of an input-out model, it was estimated that oysters worth $1 million in dockside value in Chesapeake Bay generate an estimated $36.4 million in total sales, $21.8 million in income, and 932 person-years of employment (NOAA Fisheries Office of Habitat Conservation undated-a). Athearn (2008), Gardner Pinfold Consulting Economists Ltd. (2003) and O'Hara et al. (2003) estimated the economic impacts of shellfish aquaculture in Maine, while Frank Harmon Architect and Olympus Aquaculture Consulting (2008) projected the impacts of a hatchery in Virginia for the production of oyster larvae and spat. Philippakos et al. (2001) utilized an input-output methodology to estimate the direct, indirect and induced economic impacts of the cultured clam industry in Florida. Burrage et al. (1990) examined the regional economic impacts of a project intended to revitalize the northern Gulf Coast oyster industry by relaying oysters (moving oysters from leases under compromised water quality to leases in cleaner, approved waters before final harvest).

In Washington State, an early study by Bonacker and Cheney (1988) measured the direct economic impacts of shellfish culture in Willapa Bay. The study examined expenditure patterns of industry employees but did not calculate multiplier effects. According to a 1987 study of Washington's aquaculture industry conducted by the Washington State Department of Trade and Economic Development (Inveen 1987), the ratio of total jobs to direct jobs for the oyster industry was 1.17. That is to say, for every one job directly related to the industry, 0.17 additional indirect jobs were generated in other industries throughout the State. An economic impact analysis conducted in the early 1990s by Conway (1991) suggested that, on average, each job in Washington's oyster industry supported 1.13 additional jobs elsewhere in the state economy—this constitutes an employment multiplier for the oyster industry equal to 2.13. Wolf et al. (1987) of the Economic Development Council of Mason County estimated the economic impact of the County's oyster industry using the employment multiplier of 1.17 from the Washington State Department of Trade and Economic Development's 1987 study. The analysis was updated in 2002 using the same employment multiplier (Economic Development Council of Mason County 2002).

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Appendix 2:

Rhode Island Intensive Oyster Aquaculture Economic Impact Analysis

Prepared for

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February 2014

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Contents

Executive Summary

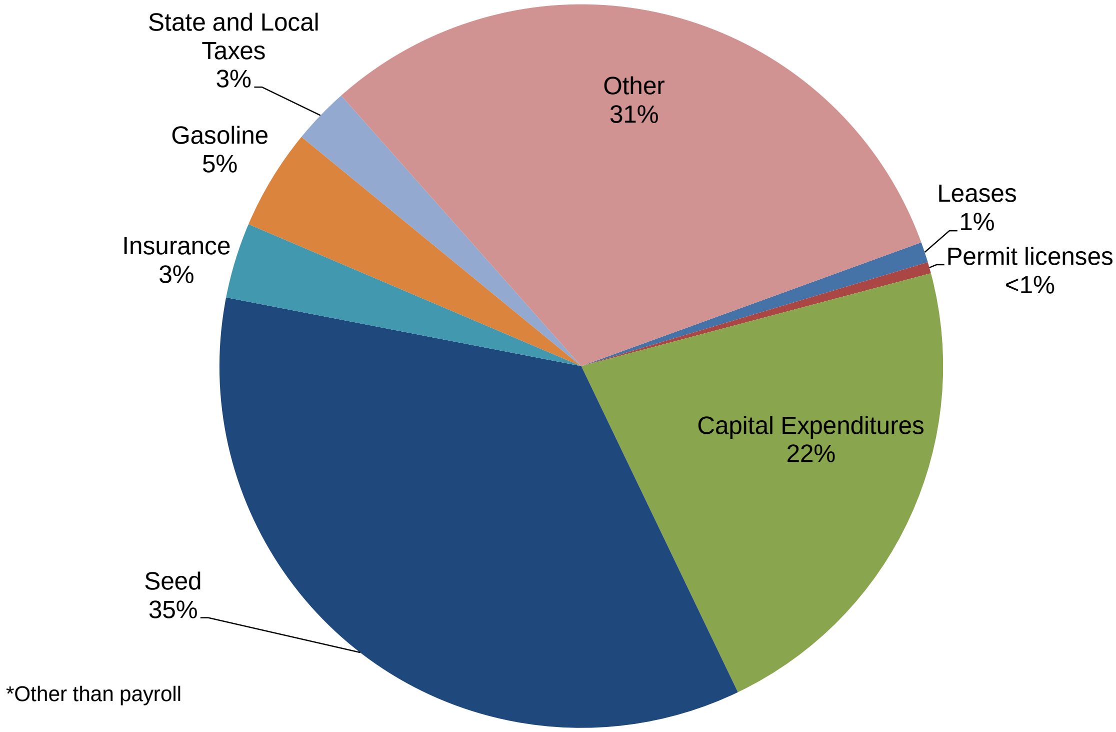
Northern Economics, Inc. conducted an Input-Output (I-O) analysis to gauge the economic impacts of the oyster aquaculture industry in Rhode Island. The purpose of the study is to better understand how oyster production impacts the statewide economy; impacts are estimated in the form of direct, indirect and induced employment, labor income and output. As a basis for the analysis the study team used results from a 2013 survey administered by Robert Rheault, Executive Director ECSGA, to gauge both the direct and downstream impacts of industry spending and employment[1].

While the state of Rhode Island has approximately 50 oyster aquaculture farms (Beutel 2012), a small portion of total growers are responsible for the majority of the state's production (Rheault 2014). After interviewing industry contacts, our study team estimates that fourteen of the state's producers are responsible for 90 percent of total production. The remaining firms are either new to the industry, or are small or part-time producers who have comparatively low levels of spending and employment (Rheault 2014).

Of the fourteen major aquaculture producers in Rhode Island, half (or seven firms) responded to the survey. Survey respondents reported a total spending of $1.01 million on the goods and services shown below (Figure ES- 1).

Figure ES- 1. Survey Respondent Non-Labor Expenditures, by Category (2012)

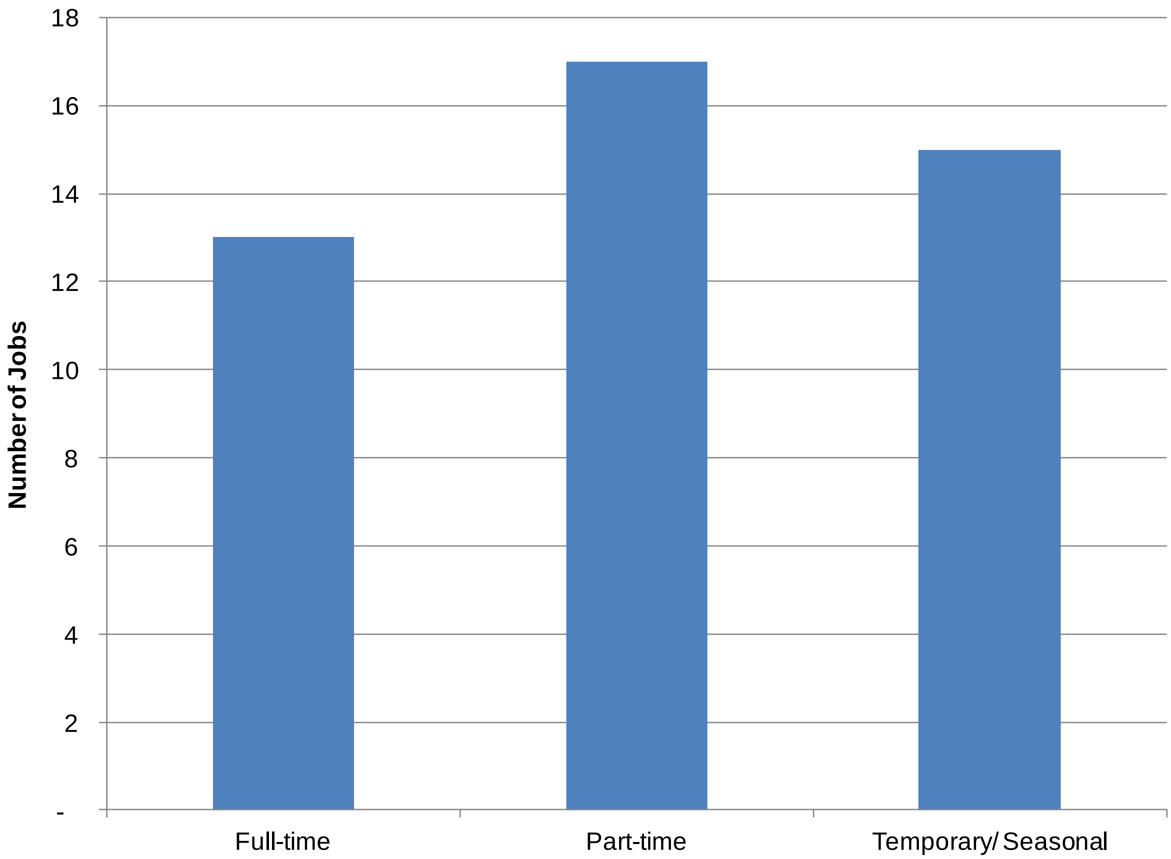
[1] Survey administered in 2013; data given was for 2012.



Note: Other includes miscellaneous purchases of clothing and equipment, repair services and transportation
Source: Northern Economics, Inc. using Rheault, 2013

In addition, respondents paid more than $472,000 in payroll expenses and supported 45 direct jobs within the state. It is worth noting that neither the payroll estimates nor the jobs shown in Figure ES- 2 include owner employment or salaries, which would increase the figures. According to the Coastal Resources Management Council (CRMC) 2012 Annual Report, in 2012 aquaculture farm workers numbered 105 (Beutel 2012).

Figure ES- 2. Survey Respondent Jobs by Type (2012)



Note: Does not include owner employees
Source: Rheault, 2013

In order to estimate the economic impact of the seven missing producers, the study team used the average expense figures reported by the survey respondents[2]. IMPLAN, an I-O software tool, uses industry expenditures to estimate backward linkages and an industry's total impact on jobs, labor income and economic output generated within a specified area (in this case, the State of Rhode Island). The two data sets (actual survey responses and estimates for missing respondents) were combined with IMPLAN multipliers to produce the economic impact estimates shown below in Table ES- 1.

[2] Respondents did not report expenditures for all categories; thus average expense per firm, when multiplied for the seven missing producers, does not exactly match figures for the seven respondents.

Table ES- 1. Economic Impact of Intensive Oyster Aquaculture Producers in Rhode Island

	Direct	Indirect	Induced	Total
Employment	90	41	13	144
Labor Income ($)	944,160	787,870	573,360	2,305,390
Output ($)	2,765,820	1,022,860	1,553,360	5,342,040

Notes: Includes payments to state and household impacts; Jobs are not full-time equivalents; labor income is included in output
Source: Northern Economics, Inc. using IMPLAN 2011

These figures imply that for every $1 million spent by oyster producers in Rhode Island, 52 jobs are created. In addition, $1.9 million in total economic output – or $1.90 for every dollar spent– is generated, almost half of which is wages or labor income.

Table ES- 2. Rhode Island Intensive Aquaculture per Million Dollar Impacts (Multipliers)

Jobs	Labor Income ($)	Output ($)
52	833,500	1,931,500

Notes: Jobs are not full-time equivalents; labor income is included in output
Source: Northern Economics, Inc. using Rheault (2013) and IMPLAN (2011)

As previously noted, this analysis estimates only the impact of the fourteen oyster producers currently responsible for the majority of production and spending within Rhode Island. In 2012 alone, the number of farms in Rhode Island increased from 43 to 50, a jump of 16 percent (Beutel 2012). As the new farms increase production and gain traction within industry, the economic impact of oyster aquaculture in Rhode Island will likely grow, supporting the need for continued tracking and analysis of the growing industry.

In the following sections we describe the data and analysis that produced these results.

1 Introduction

In 2012 and 2013 we attempted to collect survey data from aquaculture producers in Long Island Sound through the University of Connecticut Sea Grant with assistance from Robert Rheault, Executive Director of ECSGA using an NEI designed survey. The survey team attempted to conduct both mail and in-person surveys. Unfortunately the response rate was less than 10% of the existing industry. Four surveys were returned and these all lacked important expenditure data. As a result a decision was made by the REServe team to use data from Rhode Island intensive oyster aquaculture producers as a proxy for the LIS growers.[3] Robert Rheault distributed a modified version (less detailed) of the LIS survey developed by NEI to Rhode Island growers. Surveys were submitted directly to NEI for analysis with a final response rate of 58% (7 responses from 14 growers). These intensive oyster aquaculture producers represent approximately 80% of the total production of all aquaculture production (4,303,886 oysters in 2012 at a farm gate value of $2,822,734). With the data collected we conducted an input/output analysis using the IMPLAN model for the State of Rhode Island. The rest of this document describes the IMPLAN model and input/output analysis, describes the data used and provides results from our analysis.

[3] This decision was based on the assumption that future growth in the Connecticut (LIS) industry would come from intensive growers (similar to those in Rhode Island) as opposed to traditional extensive growers that rely on wild spat exclusively and bottom plant without cages or bags.

2 Data Collection

The analysis of intensive oyster aquaculture producers in Rhode Island was based on a survey administered by Robert Rheault over the course of 2013. Intensive aquaculture refers to the use of hatchery reared seed, floating upwellers, container culture for most or part of the growth cycle as opposed to traditional or extensive culture which involves natural spat collection and bottom culture without cages, bags or containers. Rheault is familiar will all growers in the state of Rhode Island. He selected growers to be surveyed based on whether they had been in business for more than 3 years (allowing for consistent sales to expenditure ratios), kept accurate records, were in the industry as a means of livelihood, and represented a large percentage of annual sales. Surveys were sent out twice over two months (May and June) and again in October of 2013. All data collected corresponded to farming activities that took place in 2012. While the study team focused on expenditure and employment data for the purpose of the Input-Output (I-O) modeling, the survey also yielded information on the demographics and business models of industry participants. These details are summarized in Section 0

2.1 Survey Results

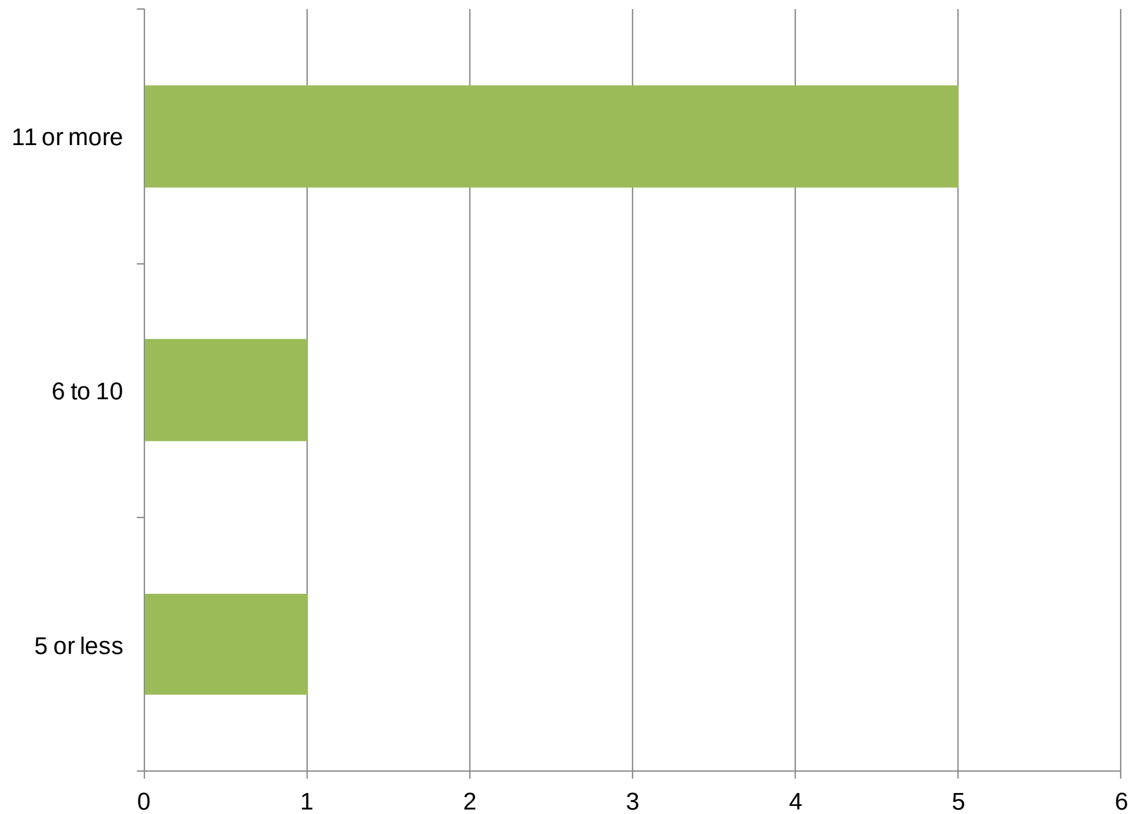
The survey results indicate that business operators are relatively young, but experienced. Though more than half report being in the 31-40 age bracket, only one reported less than three years of experience in the industry (Table 1 and Figure 1).

Table 1. Number of Responses by Age of Principal Operator of Business

Age of Principal Operator of Business	Number of Businesses
31 – 40	4
41 – 50	1
51 – 60	1
61+	1

As shown in Figure 1, respondents collectively have more than 60 years of experience in the industry.

Figure 1. Years of Industry Experience



Of the seven respondents, only one reported selling seeds to others, and none reported growing their own or naturally recruiting shellfish seed (Table 2 and Table 3). In speaking with industry experts, Northern Economics, Inc. learned that most Rhode Island producers purchase seed from outside of the state (Rheault 2014).[4]

Table 2. Shellfish Operations

Shellfish Operations	Number of Businesses
Hatchery (seed sales to others)	1
Hatchery (seed for personal use)	1
Upweller	4
Clams culture on bottom	2
Oysters on bottom	5
Oysters in floating or suspended cages	4
Oysters in cage on bottom	5
Oysters intertidal	0
Wild harvester	1
Shellfish dealer/ Shellstock shipper	5
Mussels suspension culture	1

Source: Rheault 2013

Table 3. Methods for Obtaining Shellfish Seed

Method	Number of Businesses
Grow own	0
Purchase from a hatchery	7
Deploy spat collectors or clutch	2
Rely on natural recruitment	0

Source: Rheault 2013

[4] There is only one non-commercial university hatchery in the state of Rhode Island. Therefore, all seed is purchased from out or state hatcheries. Some seed is purchased small, grown out and re-sold to local growers for a substantial profit (e.g. in 2013 seed sales from in-state nursery growers to other instate growers was $180,000).

Survey respondents collectively represent 76 acres of tidelands, 71.5 acres of which were reported to be under cultivation. According to the Coastal Resources Management Council (CRMC) 2012 Annual Report, this represents 41 percent of statewide acreage under cultivation[5]. While three firms produce products other than oysters, all firms report using at least 90% of their acreage for oyster operations (Table 4).

Table 4. Operation Size and Percent of Use by Operation (2012)

Firm	Operation Size (acres)	Percent use by Operation		
		Oysters	Clams	Mussels
A	3	95	0	5
B	20	100	0	0
C	22	100	0	0
D	18	50	0	0
E	3	70	0	0
F	4	90	10	0
G	7	95	0	5

Note: Percentages do not sum to 100 due to operations that are not listed and/or unused acres.
Source: Rheault 2013

In 2012, survey respondents reported total revenues of $1.6 million, or an average of $229 thousand per grower.

Table 5. Oyster Production and Revenue (2012)

Calculation	Oyster Production (count)	Oyster Revenue ($)
Total	3,691,339	1,604,745
Average	461,417	229,249

2.2 Employment and Expenditure Data

[5] Total 2012 acreage under cultivation reported at 172.55 acres

The survey asked firms to describe their spending in 2012. The survey listed the spending categories shown in Table 6, and survey respondents supplied responses by category. Total reported spending summed to $1.01 million on the goods and services shown below (Table 6 and Figure 2).

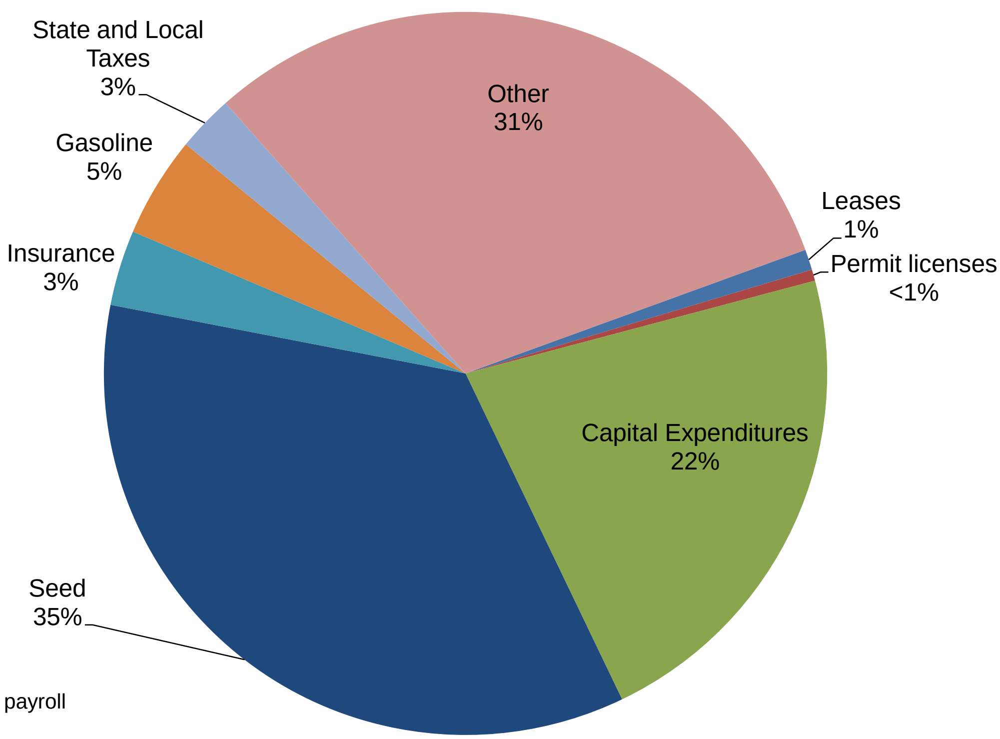
Table 6. Total and Average Expenses by Category, 2012 ($)

Expenditure	Leases	Permit licenses	Capital expenditures	Seed	Insurance	Gasoline	State and local taxes*	Other	Total
Total	9,250	5,083	222,000	354,730	34,044	45,491	25,466	312,636	1,008,700
Average	1,321	726	37,000	50,676	4,863	7,582	4,244	52,106	158,519

Note: The calculated totals above were adjusted to meet reported totals; where reported totals were greater than calculated totals (the sum of categories) and no 'other' expenditures were reported, study team increased other expenditure totals to maintain consistency.

Source: Northern Economics, Inc.

Figure 2. Survey Respondent Non-Labor Expenditures, by Category (2012)

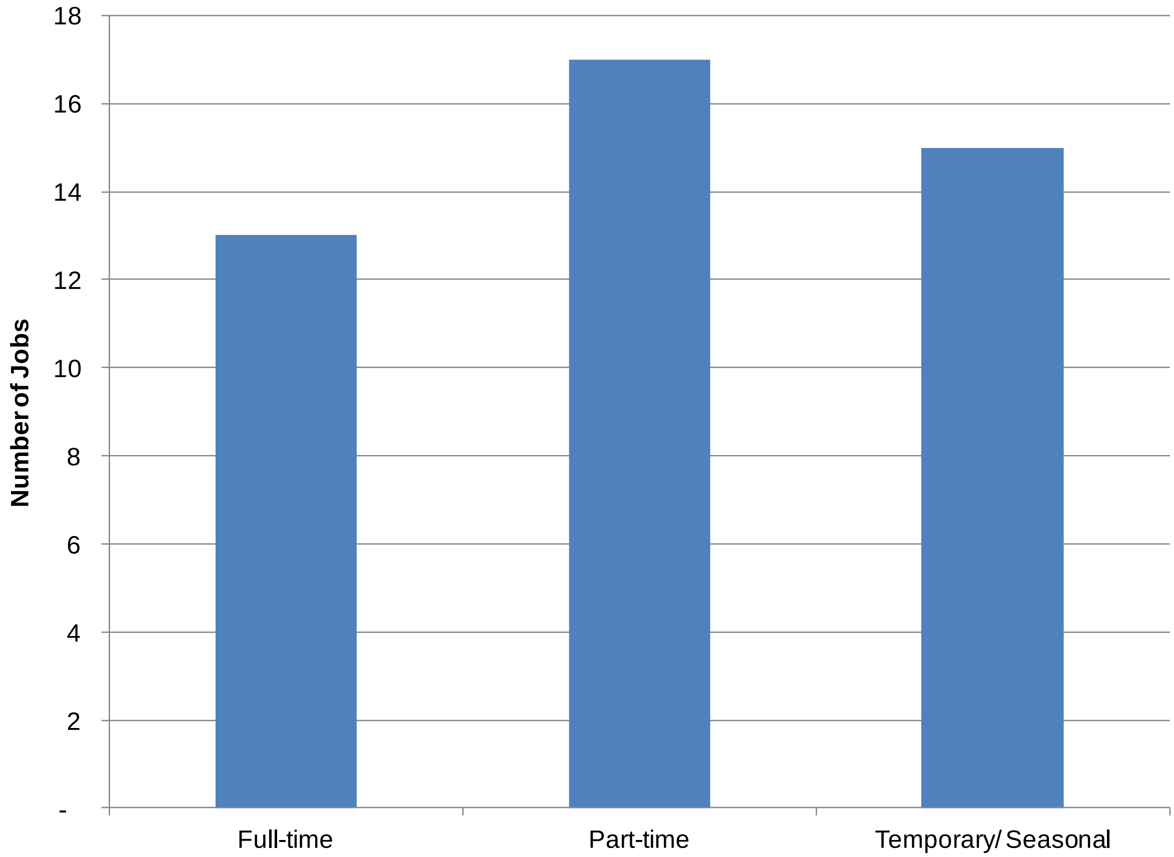


*Other than payroll

Note: Other includes miscellaneous purchases of clothing and equipment, repair services and transportation
Source: Northern Economics, Inc. using Rheault 2013

In addition to spending on goods and services, respondents paid more than $472 thousand in payroll expenses and supported 45 direct jobs within the state. It is worth noting that neither the payroll estimates nor the jobs shown in Figure 3 include owner employment or salaries, which would increase the figures. According to the CRMC 2012 Annual Report, in 2012 aquaculture farm workers numbered 105 (Beutel 2012).

Figure 3. Survey Respondent Jobs by Type (2012)



Note: Does not include owner employees

3 Input Output Analysis

The economic benefits of the intensive oyster aquaculture industry in Rhode Island was estimated by quantifying the direct, indirect, and induced effects of local spending on goods and services. The expenditure data described in Section 2 were used as the inputs to an I-O model used to gauge the multiplier effects of the industry.

3.1 How Input-Output Analysis Works

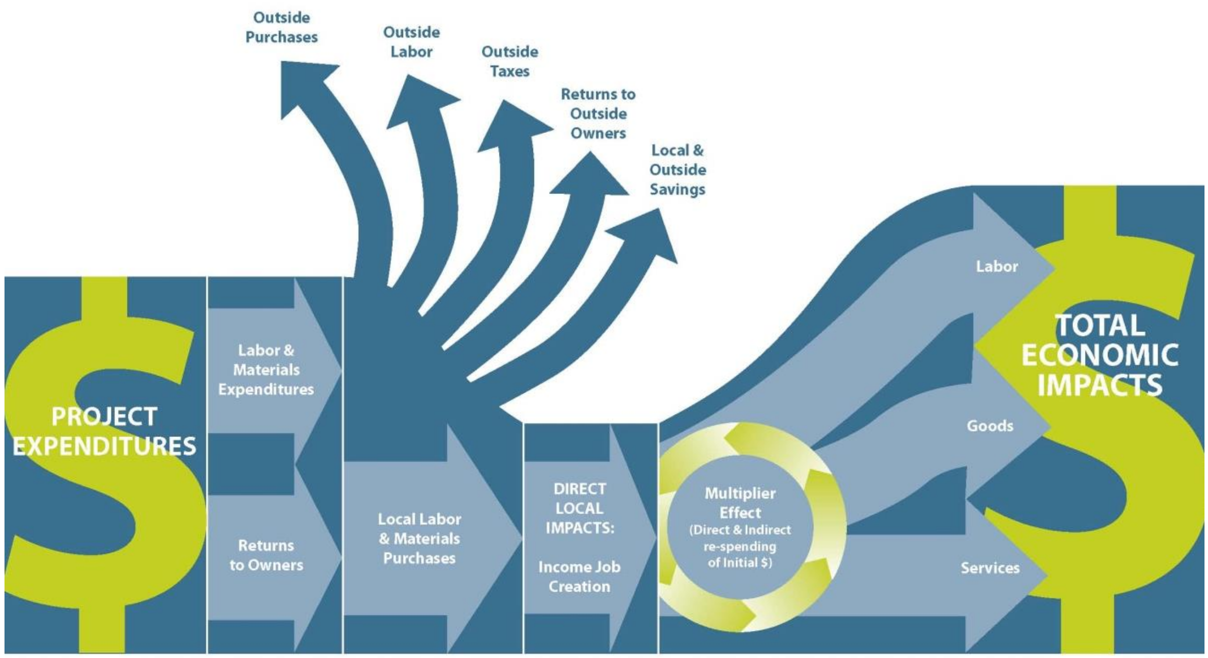
I-O models are economic tools used to measure the effects of an economic activity on a region; using expenditure patterns, tax payments, labor requirements and other economic data they aim to replicate the inter-industry transactions within a community, tracking the flow of money between the industries within a specified economic region of interest. An I-O model can measure how many times a dollar is re-spent in, or "ripples" through, the economic region before it leaks out.

The I-O model yields multipliers that are used to calculate the indirect and induced effects on jobs, income, and business sales/output generated per dollar of spending on various types of goods and services in the study area. To evaluate the economic effects to the state or a particular region, only the "local" (i.e., within the state or within the region) expenditures are used in the model; the rest are considered leakages. More leakages mean smaller multipliers; and the larger the local expenditures, the greater the multiplier effects. The multipliers for any given industry in any given location are unique, based on industry composition and geographic area. In this case, the IMPLAN™ software was used to model the Rhode Island state economy, which in turn allowed the study team to estimate the local impacts of intensive oyster aquaculture spending.

IMPLAN uses specific data on what inputs are needed to produce the goods or services for over 400 industries, and borough-specific data on what industries are available locally from which to purchase those inputs. The most recent (2011) IMPLAN data on multipliers for Rhode Island were applied for this analysis

Figure 4 illustrates conceptually how the total economic impacts or benefits are determined.

Figure 4. Illustration of Input-Output Analysis



3.2 Analysis of Intensive Oyster Aquaculture

The first step in conducting this analysis was to determine the quality and extent of the available data. As previously mentioned, in 2012 there were 50 recorded oyster producers in Rhode Island, of which fourteen were responsible for the vast majority of production. Of these fourteen 'intensive' aquaculture producers, half (or seven firms) responded to the survey.

The study team, in partnership with the survey administrator, determined that the seven collected surveys were representative of the seven missing intensive producers, but likely differed significantly from the remaining 36 (who are either small producers or new entrants to the industry). Consequently the team decided that the I-O analysis would be completed for the fourteen intensive oyster aquaculture firms only.

In order to estimate the economic impact of the seven missing respondents, the study team used average expenditure by category. In other words, the average expense per firm for gas, insurance, leases, etc. were used as data points for each of the missing respondents. Because some respondents did not report expenditures for all categories, the seven imputed responses do not exactly match the seven survey responses (Table 7).

Table 7. Expenditures by Category used for Analysis ($thousands)

Component	Leases	Permit Licenses	Capital expenditures	Seed	Insurance	Gas	State and local taxes	Other
Survey Respondents	9.3	5.1	222.0	354.7	34.0	45.5	25.5	312.6
Missing Respondents	9.3	5.1	259.0	354.7	34.0	53.1	29.7	364.7

Note: Expenditures do not total to direct output because some payments listed above are made to the state and contribute to induced impacts. In addition, adjustments to direct output were made for reported wages.
Source: Northern Economics, Inc. 2014

Oyster seed is a significant expense for aquaculture farmers, comprising about 35 percent of non-labor expenses in 2012. According to the survey administrator, all seed is purchased out of state 1mm seed at $12 per thousand). However, as indicated above, some seed is kept by growers, grown to 10-20 meters and then resold at $30,000–40,000. For this analysis we assume that 30 percent of seed costs may be attributed to in-state economic impacts; the rest (70%) was considered a leakage.

Spending on leases and deeds, permit licenses, and taxes become state, not industry revenues, generating induced economic impacts. In order to estimate the induced impact of these funds, the study team relied on IMPLAN's multipliers for state government spending impacts. According to the most recent IMPLAN data for the State of Rhode Island[6]:

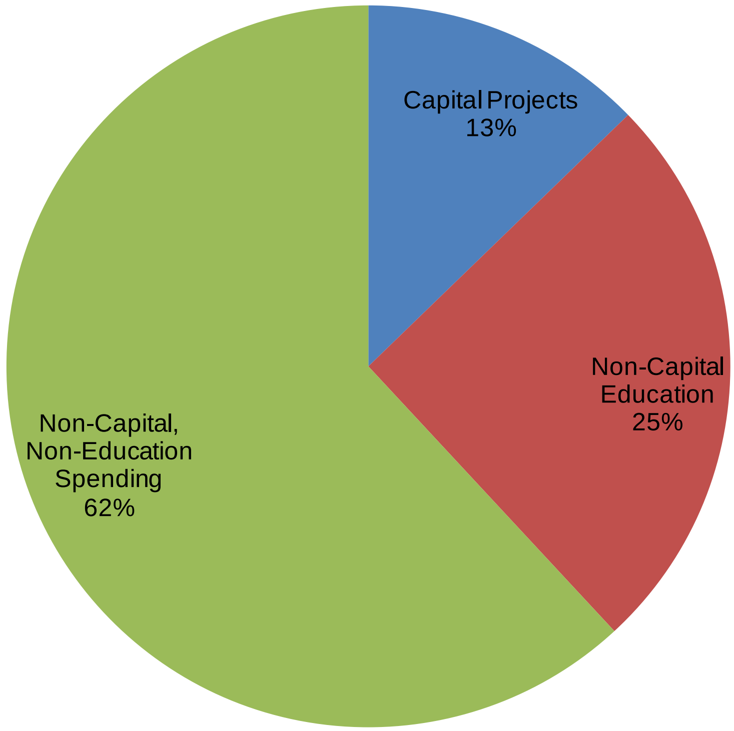
- $1 million spent on investments (capital goods) generate: 2.8 jobs, $172,977 in income; and $385,395 in output.
- $1 million spent on education programs generate: 15.3 jobs, $1,024,960 in income; and $1.63 million in output.
- $1 million spent on non-education programs generate: 13.6 jobs, $763,000 in income; and $1.4 million in output.

According to the Rhode Island Fiscal Year 2013 Budget as Enacted and the Fiscal Year 2013 Capital Budget, the state spends approximately 13 percent of its total revenues on capital projects, 25 percent of its total revenues on non-capital education, and the remaining 62 percent on non-capital, non-education items (Figure 5).[7]

[6] This data is more current than the industry multipliers available; state government spending impacts are estimates for 2014.

[7] FY 2013 documents used because funds gathered in 2012 are spent the following year.

Figure 5. Rhode Island State Expenditures by Category



Source: State of Rhode Island 2013a and 2013b

The study team estimated intensive oyster aquaculture spending on leases, permits and licenses, and taxes. By distributing those revenues according to the percentages shown in Figure 5, and combining them with the state government multipliers, the study team was able to estimate the induced economic impacts shown in Table 8.

Table 8. Economic Impact of Intensive Oyster Aquaculture Payments to State of Rhode Island

	Direct	Indirect	Induced	Total
Employment	-	-	2	2
Labor Income ($)	-	-	38,900	38,900
Output ($)	-	-	69,600	69,600

Notes: Jobs are not full-time equivalents; labor income is included in output
Source: Northern Economics, Inc. using IMPLAN 2014

3.3 Input-Output Results

Using the methods and adjustments outlined above, the two data sets (actual survey responses and estimates for missing respondents) were combined with IMPLAN multipliers to produce the economic impact estimates shown below in Table 9 and Table 10.

Table 9. Economic Impact of Survey Respondents

	Direct	Indirect	Induced	Total
Employment	45	20	6	72
Labor Income ($)	472,080	369,730	279,270	1,121,080
Output ($)	1,334,570	480,580	757,140	2,572,290

Notes: Jobs are not full-time equivalents; labor income is included in output
Source: Northern Economics, Inc. using IMPLAN 2011

Table 10. Estimated Economic Impact of Missing Respondents

	Direct	Indirect	Induced	Total
Employment	45	21	7	73
Labor Income ($)	472,080	418,140	294,090	1,184,310
Output ($)	1,431,250	542,280	796,220	2,769,750

Notes: Jobs are not full-time equivalents; labor income is included in output
Source: Northern Economics, Inc. using IMPLAN 2011

The above tables are summed in Table 11.

Table 11. Economic Impact of Intensive Oyster Aquaculture Producers in Rhode Island

	Direct	Indirect	Induced	Total
Employment	90	41	13	144
Labor Income ($)	944,160	787,870	573,360	2,305,390
Output ($)	2,765,820	1,022,860	1,553,360	5,342,040

Notes: Includes payments to state and household impacts; Jobs are not full-time equivalents; labor income is included in output
Source: Northern Economics, Inc. using IMPLAN 2011

These figures imply that for every $1 million spent by oyster producers in Rhode Island, 52 jobs are created. In addition, $1.9 million in total economic output – or $1.90 for every dollar spent– is generated, almost half of which is wages or labor income.

Table 12. Rhode Island Intensive Aquaculture per Million Dollar Impacts (Multipliers)

Jobs	Labor Income ($)	Output ($)
52	833,000	1,931,500

Notes: Jobs are not full-time equivalents; labor income is included in output
Source: Northern Economics, Inc. using Rheault (2013) and IMPLAN (2011)

4 References

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Appendix 3a:

Nitrogen and Carbon Content of Farmed Eastern Oysters (*Crassostrea virginica*) In the Great Bay Estuary, New Hampshire

A Final Report to:

National Oceanic and Atmospheric Administration
National Ocean Service, NCCOS
Silver Spring, MD 20910

Submitted by:

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March 17, 2014

Abstract: When harvested, farmed oysters represent a removal from the ecosystem of nutrients such as nitrogen (N) and carbon (C) they have assimilated. We measured over three separate years (2010, 2011, 2012), at six sites in the Great Bay Estuary, New Hampshire, USA the N and C content of oysters deployed in polyethylene bags typically used on oyster farms in the region. Sites were chosen to represent a range of ambient nutrient concentrations, water flow conditions, and location within the estuary. Approximately 1,000 “seed” size ~0.3 year old; (19-42 mm shell height) or 200, ~1.3-year old (28-79 mm shell height) oysters were placed into two separate bags which were suspended 10 cm off the bottom attached to plastic coated wire cages at each of the study sites. Samples of oysters from the bags were taken on seven occasions over the course of the 3-yr study, and analyzed for N and C content and size. A quadratic polynomial fit to the combined (all sites, both size classes) datasets for shell height indicated that on average a ‘cocktail’ size oyster (63 mm shell height) would be reached in the third growing season, and ‘regular’ size (76 mm) would require a full 3 years. However, there was wide variability in growth, with some individuals reaching regular size in the second growing season. There were significant differences in growth rates and %N and %C in shell and soft tissue among the study sites. Growth variations were positively correlated (but not significantly) with chlorophyll *a* concentrations (2.9 to 6.7 µg/L) and negatively correlated (but not significantly) with mean water flow speeds (14 to 35 cm/s). Percent N in soft tissue and shell were highest at the two sites with highest mean dissolved inorganic N ($NO_2/NO_3 + NH_4$) concentrations in the water column, with means varying from 6.9 to 8.4 %N in soft tissue and 0.07 to 0.18 %N in shell. Grand means of soft tissue N and C for regular size oysters were 7.5% and 37.1%, respectively; and for shell N and C were 0.13% and 12.0%, respectively. Although the overall means were similar to previous research in other areas, our data also suggest that oyster size, seasonality, and nutrient concentration in ambient water can strongly affect %N and %C content of oysters.

Introduction

Eutrophication of coastal waters and estuaries persists as a global problem (Boesch 2002; Bricker et al. 2008; Selman et al. 2008; Kroeze et al. 2013). Management of nutrients such as nitrogen and phosphorus that contribute to eutrophication typically has focused on control of land-based sources. Recent research, however, has begun to quantify the potential of bivalve shellfish aquaculture as a nutrient management tool. Suspension-feeding bivalves such as oysters and mussels remove suspended particulates and associated nutrients from the water as they feed, and when harvested represent a direct removal of the assimilated nutrients from the ecosystem (Hammer 1996; Newell 2004; Lindahl et al. 2005; Ferreira et al. 2007; Burkholder and Shumway 2011; Coen et al. 2011; Ferreira et al. 2011; Higgins et al. 2011; Carmichael et al. 2012). Farmed (and wild) bivalves also potentially cause nutrient removal by enhancing burial in sediments, and in the case of nitrogen, increased denitrification rates (Newell et al. 2005; Pietros and Rice 2009; Carmichael et al. 2012; Hollein and Zarnoch 2014).

In the mid-Atlantic and northeastern US, eastern oyster (*Crassostrea virginica*) aquaculture is a rapidly growing industry and recent research has begun to quantify the potential role oyster farming might play in nutrient management (see reviews by Newell and Mann 2012, and STAC 2013). In the Great Bay Estuary where the present study was conducted, the major nutrient of concern is nitrogen (PREP 2013) and research is underway to assess the role oyster farming might play in its management (Grizzle and Ward 2011; Rose et al. 2014).

The fundamental metric upon which estimates of nutrient removal by farmed oysters are based is the nutrient content of individual oysters. The meager but growing literature indicates substantial spatial variability in N content of oysters but the cause(s) or generality of such variability is not known (Higgins et al. 2011; Grizzle and Ward 2011). There are also seasonal variations in nutrient content due to physiological changes in the oyster but these effects have not been well quantified in farmed oysters (Newell and Mann 2012; STAC 2013). Finally, more data on size-related variations in nutrient content are needed in order to make nutrient removal predictions for different harvest scenarios (Higgins et al. 2011).

The present study had two overall objectives: (1) quantify the nutrient (N and C) content (”assimilated” N and C of some authors) and growth rates of farmed oysters held in gear typical of the region for 3 years in different areas of the estuary; and (2) assess the relations between oyster size, growth, nutrient content, and environmental variables.

Methods

Field Methods

Six sites were chosen to represent a range of ambient nutrient concentrations, water flow speeds, and location within the estuary: two oyster farm sites (Granite State Shellfish [GSS] at the mouth of the Oyster River and Little Bay Oyster [LBO] in Little Bay), Adams Point (AP) in Little Bay, the mouths of the Bellamy (BMY) and Squamscott Rivers (SQR), and Nannie Island (NI) in Great Bay (Fig. 1).

Two size classes (small: 19-42 mm initial shell height [0.3 year old]; large: 28-79 mm initial shell height [1.3 years old]) of hatchery-reared oysters were obtained from Little Bay Oyster in August 2010. Before deployment at the study sites, thirty (30) oysters were haphazardly selected from each of the two size classes, shell height and whole wet weight measured, then frozen and shipped to USEPA Atlantic Ecology Division for elemental analysis (see Laboratory Methods section below).

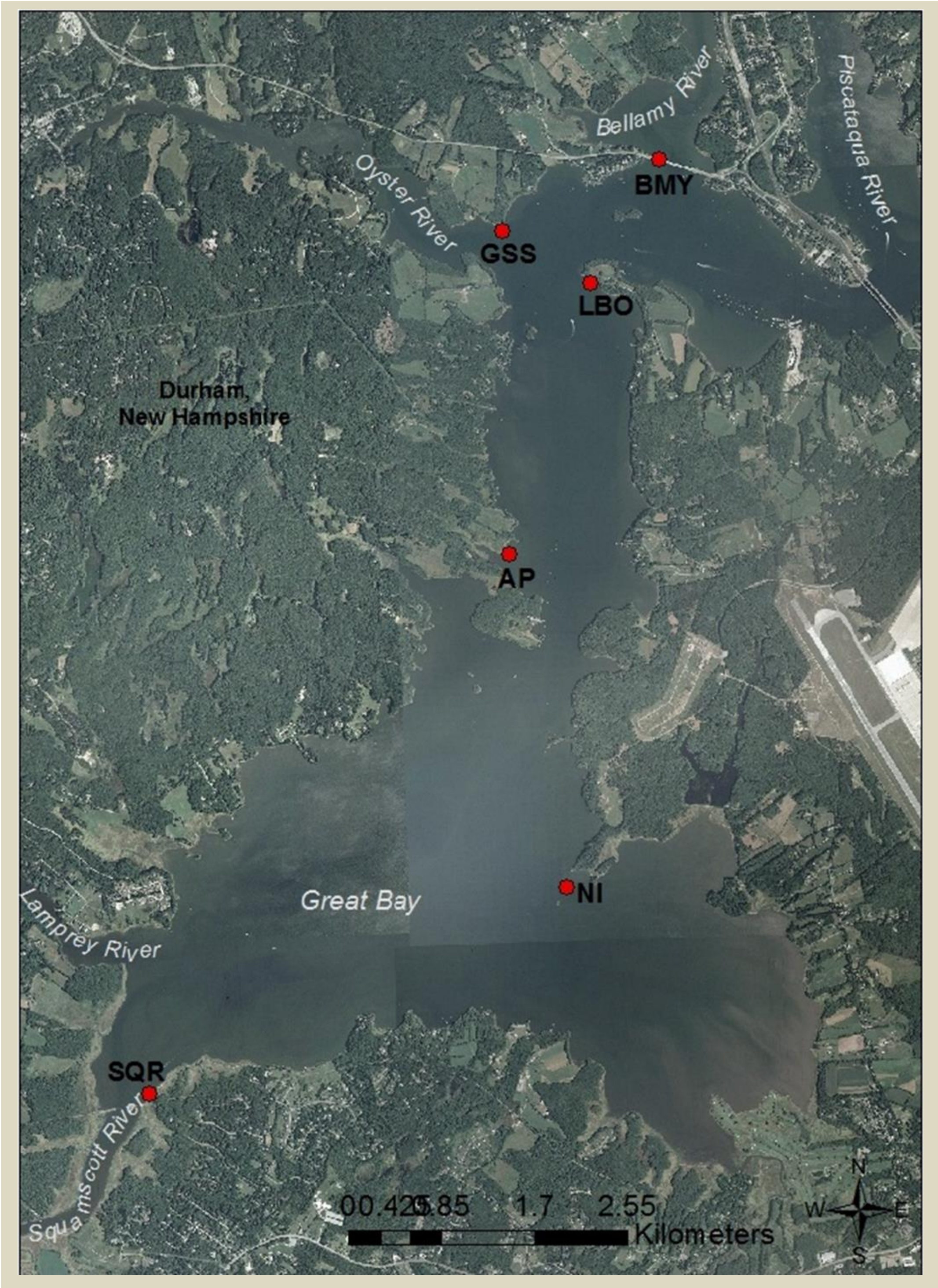


Fig. 1. Six sites (red circles) where oysters were deployed 2010-2012.

At each of the six study sites, ~1,000 small size class oysters were deployed in a 9 mm mesh and ~200 large size class oysters in a 24 mm mesh flat ("envelope" style) polyethylene bag. The two bags were suspended side-by-side 10 cm off the bottom attached to plastic coated wire cages. Table 1 below summarizes the timeline of sampling and analyses for the study period August 2010 – November 2012.

Table 1. Details on sampling and analysis of oysters by size class (small and large).

Date	Measurement	# of oysters analyzed	Oyster age, by size class (yr)
Aug 2010	Size	30 small 30 large	0.3 for small 1.3 for large
Nov 2010	Size	28 small 30 large	0.6 for small 1.6 for large
Apr 2011	Soft tissue N and C	47 small 30 large	1.0 for small 2.0 for large
Nov 2011	Size	17 small 11 large	1.7 for small 2.7 for large
Nov 2011	Soft tissue N and C	30 small 16 large	1.7 for small 2.7 for large
Apr 2012	Soft tissue N and C	8 small 10 large	2.1 for small 3.1 for large
Nov 2012	Size	22 small 25 large	2.7 for small 3.7 for large
Nov 2012	Soft tissue N and C	40 small 40 large	2.7 for small 3.7 for large
Nov 2012	Shell N and C	29 small 25 large	2.7 for small 3.7 for large

Water flow speed was measured in October 2012 to characterize tidal flow conditions at the six study sites. Measurements were made at ~hourly intervals for as much of one 12.5-hr tidal cycle as practical for one spring tide and one neap tide at each of the six sites using a Marsh-McBirney electromagnetic current meter. On each visit to a site, 5 – 10 replicate instantaneous measurements were made at each of two water depths (0.2 and 0.8 of the total water depth above the bottom) which were then averaged to yield a single depth-averaged flow speed.

Laboratory Methods

Size measurements consisted of shell height (calipers to nearest mm), shell and soft tissue dry weight (5+ days at 60^{o} C), and whole wet (fresh) weight (to nearest 0.1 g). For soft tissue N and C content (measured at USEPA laboratories, Narragansett, RI), the frozen oysters were re-measured (shell height), shucked (soft tissue) into aluminum pans, oven-dried at 60^{o}C and weighed (soft tissue dry weight), homogenized by mortar and pestle. The samples were analyzed following laboratory operating procedure LOP-AED/PEB/DK/2009-01-01 using a Thermo-Finnigan Flash EA112 CHN/O elemental analyzer which was calibrated prior to analysis with certified standards. Each tissue sample was analyzed in duplicate and the results reported as %N and %C. For shell N and C content (measured at UNH's Water Quality Analysis Laboratory in the Department of Natural Resources & the Environment), a 1-cm wide section of shell was cut using a bandsaw along the length dimension at a point about 2 cm from the umbo of both valves, dried, weighed, and ground with a mortar and pestle until homogeneous. Ground samples were analyzed for %N and %C on an elemental analyzer (Perkin Elmer CHN 2400 Series II, Waltham, MA, USA) using high temperature combustion and thermal conductivity detection.

Data Analysis

The effect of site on oyster growth was tested based on the final size data for each of the two size classes. All oysters used in the experiment came from the same population (i.e., initial sizes for both size classes were the same for all six study sites), and the experiment was run for the same time period (~3 yr.) for all six study sites. Thus, final size reflected overall growth variations; these data were examined using ANOVAs. Oyster growth also was assessed by plotting combined (all sites and seasons) size data over time and fitting appropriate models to the data; annual growth rates were determined for each site using linear regression.

Oyster soft tissue N and C content were analyzed across year, season, and site using ANOVAs and t-tests when appropriate. For all analyses, residuals were examined to ensure homogeneous variance and a normal distribution. Outliers were tested for and removed when appropriate. Oyster shell N and C content were only determined on the final sampling (Table 1), so those data were only analyzed across site. Shell %C was non-normally distributed despite transformations and analyzed with a non-parametric Kruskal-Wallis. Prior to analyses on tissue and shell chemistry, potential effects of oyster age were tested because oysters were from two distinct initial age groups (0.3 and 1.3 years). No discernable effects of oyster age were evident and thus the two size/age classes were analyzed together. A Tukey’s HSD post hoc test was run to determine differences among means for ANOVAs. Statistical significance for all tests was set at an alpha level of 0.05 to control Type I error.

The potential effects of environmental factors on oyster growth were assessed by scatterplots and correlation analysis comparing oyster growth metrics and appropriate environmental variables.

Results and Discussion

Oyster morphometrics and growth

For the present study, the focus was on metrics usually reported in the aquaculture literature, including widely used production models such as FARM (Ferreira et al. 2007) and AquaShell. Because shell height is the primary metric affecting marketability and selling price, other metrics are compared to it. Oyster shell height and whole oyster fresh (wet) weight data from the overall dataset (all years, sites, seasons combined) showed the expected nonlinear relationship, and a logarithmic model provided the best fit to the data (Fig. 2). In the New England region, most of the market is comprised of two size classes: < 3 inches (76 mm) (‘petite’ or ‘cocktail’), and 3 - 4 inches (76 to 102 cm) (‘regular’). An average petite oyster (assuming 63 mm height) would have a fresh weight of ~20 g; an average regular (89 mm) would be ~70 g (Fig. 2).

Soft tissue dry weight (DW) showed the same general relationship to shell height (Fig. 3) as fresh weight (Fig. 2). An average petite oyster (assuming 63 mm height) would have a soft tissue DW of ~0.4 g; an average regular (89 mm) would be ~1.3 g (Fig. 3). These values are similar but slightly lower than those reported for farmed oysters in the Chesapeake Bay by Higgins et al. (2011).

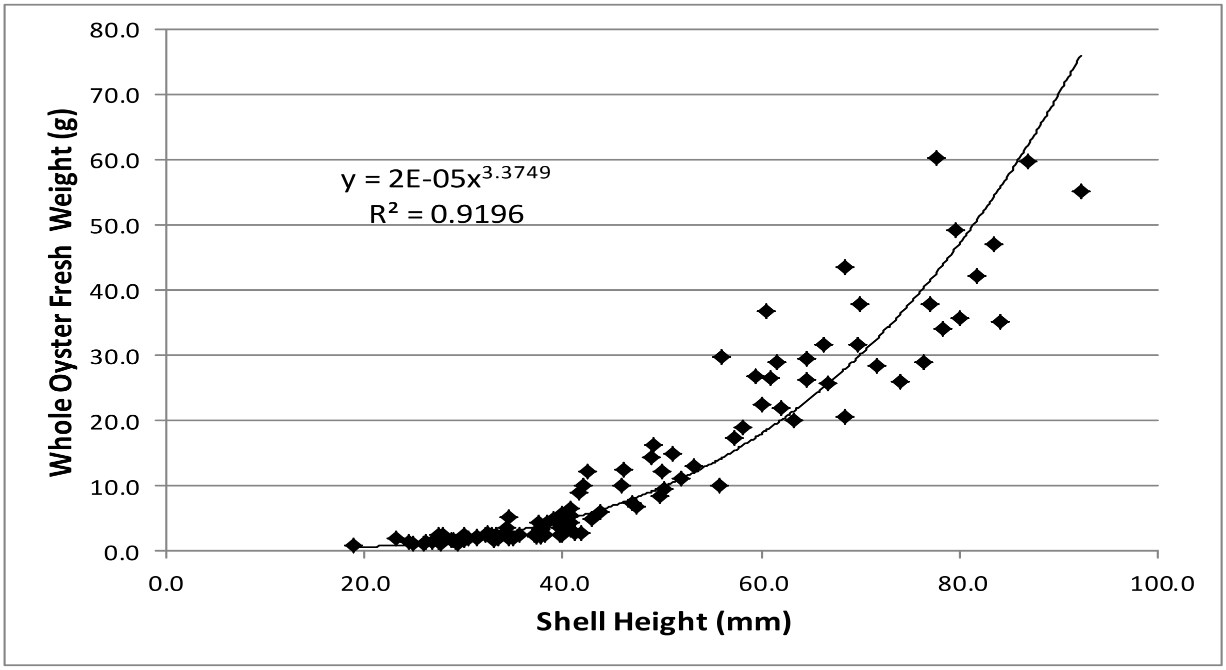


Fig. 2. Relationship between shell height and whole oyster wet (fresh) weight for overall dataset (all sites, seasons and size classes combined).

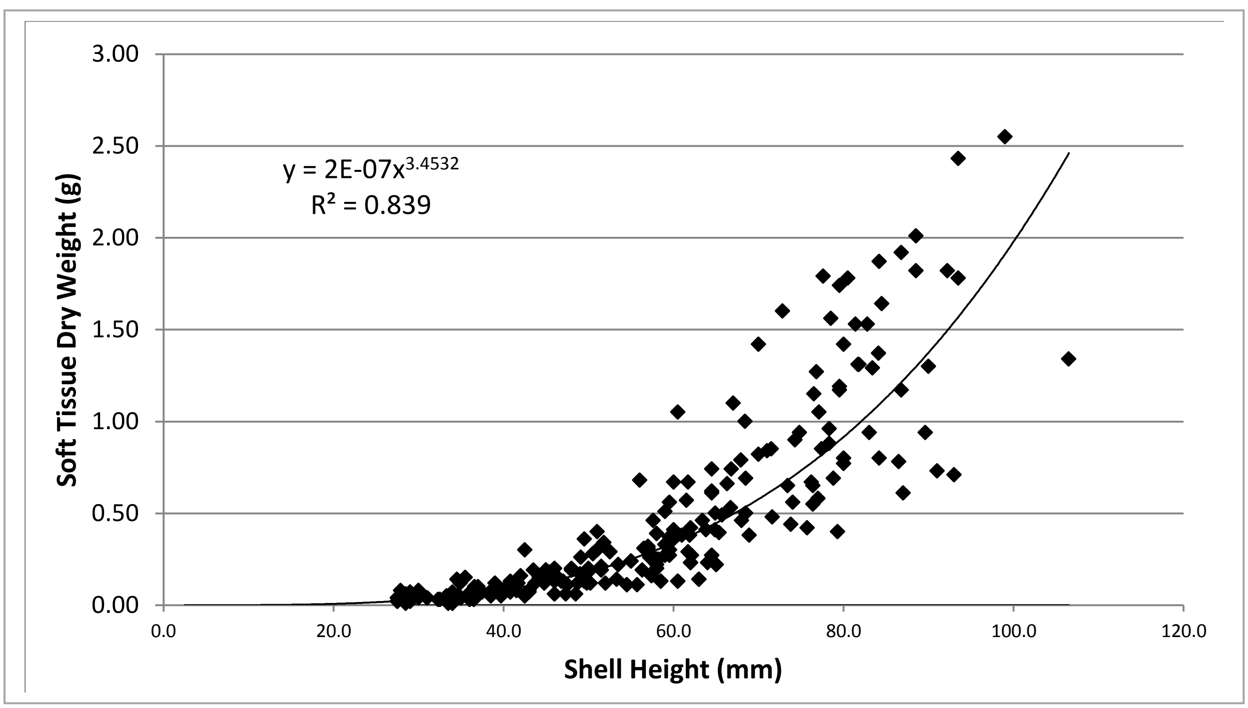


Fig. 3. Relationship between shell height and soft tissue DW for overall dataset (all sites, seasons and size classes combined).

Shell DW was only measured on oysters from the final sampling date (Table 1), so the shell height range was not as wide as the other datasets. Based on these final size measurements, an average petite oyster (63 mm height) would have a shell DW of ~17 g; an average regular (89 mm) would be ~37 g (Fig. 4). These values are very similar to those reported by Higgins et al. (2011) for farmed oysters, and much lower than Newell et al. (2005) for wild oysters which typically have thicker and heavier shells than farmed oysters (Paynter and Dimichele 1990; see discussion in Higgins et al. 2011, and Newell and Mann 2012). However, it should be noted that farmed oysters can be grown using methods ranging from confinement in bags from seed-to-market size (as in the present study), to various combinations of gear and bottom seeding. Also, in some areas oyster aquaculture involves harvesting of wild-set oysters on leased bottom areas, and no growout gear at all is used. Thus, the trend of "farmed" oysters having thinner shells than wild may be strongly dependent on culture methods used.

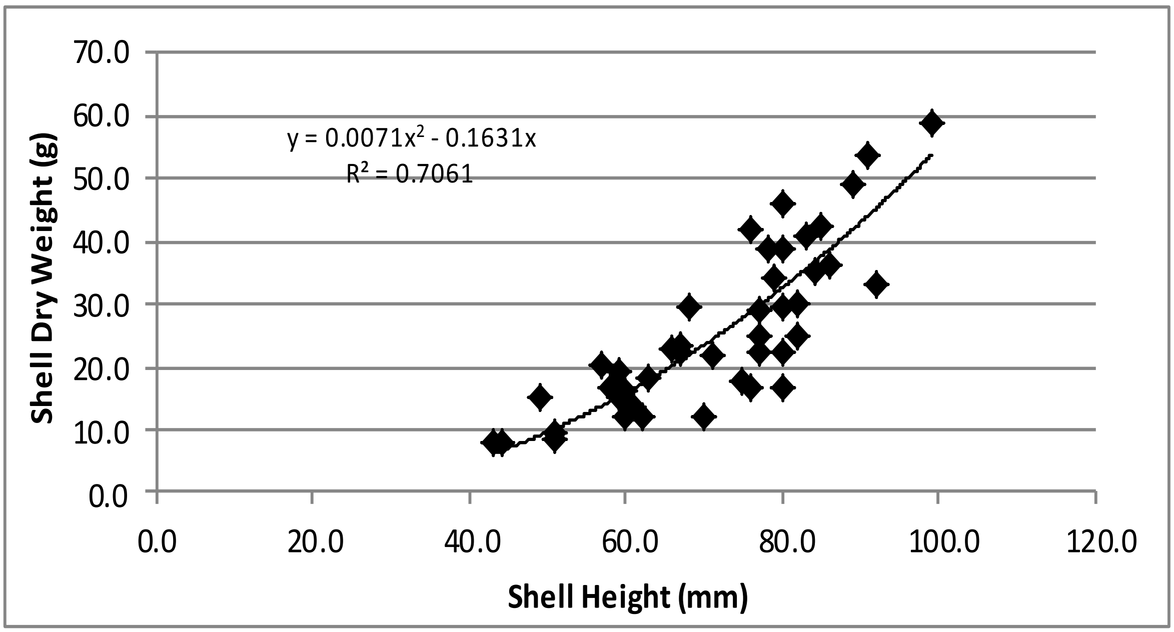


Fig. 4. Relationship between shell height and shell DW for overall dataset (all sites, seasons and size classes combined

Oyster growth was assessed in three ways: by combining size data from all six sites, doing this for both size classes and fitting models to the plotted data (Figs. 5 and 6); by fitting models to the size data from each of the study sites individually and computing average annual rates (Table 2); and by comparing final size metrics from Nov 2012 (Figs. 7, 8 and 9).

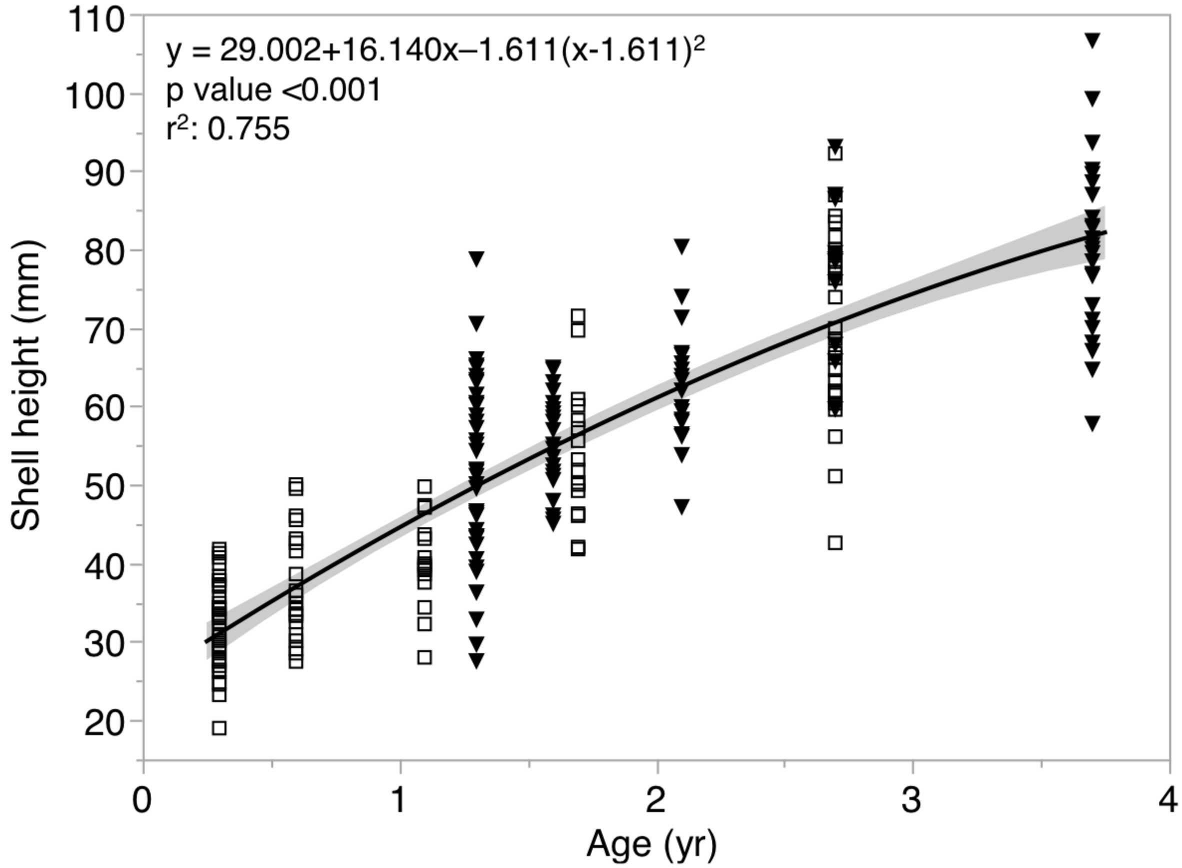


Fig. 5. Growth of oysters as shell height using combined datasets from all sites, seasons, size classes. The gray shaded line around the fit line is 95% confidence interval. Open squares = small size class; Black inverted triangles = large size class.

A quadratic polynomial fit to the combined (all sites, both size classes) size datasets for shell height indicated that a cocktail size oyster (63 mm) would be reached in the third growing season, and minimal regular size (76 mm) would require a full 3 years (Fig. 5). However, there was wide variability among individual oysters on each sampling date with some reaching regular size in the second growing season. Annual growth rates of the eastern oyster vary widely across its range with fastest growth in the warmer regions, probably largely a result of extended growing seasons (Kraeuter et al. 2007). In her review, Shumway (1996) noted that market size (90 mm) could be attained in 2 years in the Gulf of Mexico, compared to 4 or 5 years in Long Island Sound. Recent studies reported market size oysters were attained in 2 years in the Chesapeake (Harding et al. 2008, 2010) and less than 2 years in Rhode Island (Carmichael et al. 2012), both similar to our findings for Great Bay. Growth expressed as whole wet weight and soft tissue dry weight had similar non-linear growth patterns (Fig. 6).

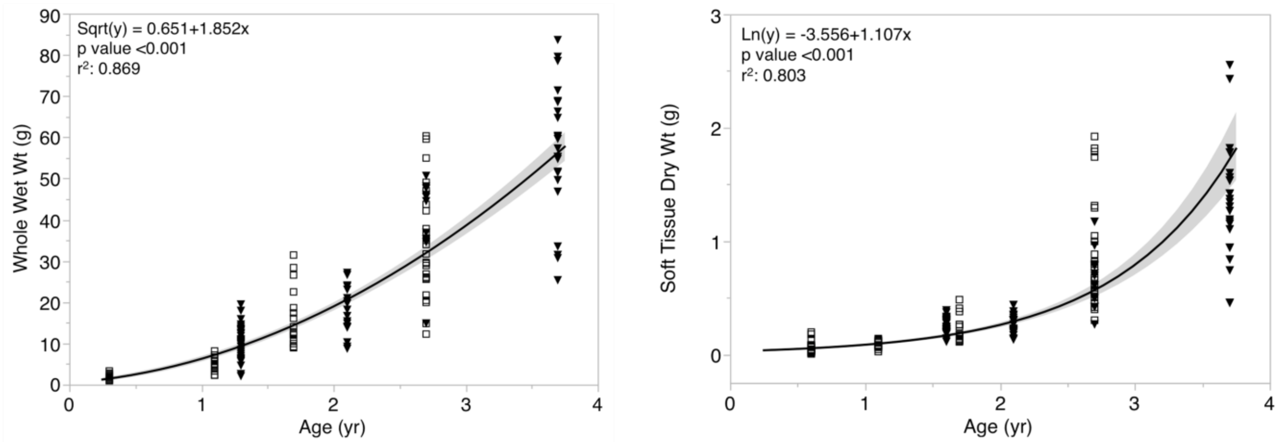


Fig. 6. Growth of oysters as whole wet (fresh) weight and soft tissue DW using combined datasets from all sites, seasons, size classes. The gray shaded line around the fit lines is 95% confidence interval. Open squares = small size class; Black inverted triangles = large size class.

The loss of some oyster bags resulted in incomplete (for the entire 3-year growth period) data from two of the study sites. Nonetheless, sufficient data were available for most sites to calculate growth rates for at least 2 years by fitting appropriate regression models to plots of all three size metrics and determining the mean annual growth rate for each (Table 2). As expected, there was wide variability in annual rates among the sites, and all three size metrics showed similar rankings relative to growth rates. The SQR and NI sites generally showed fastest growth and GSS and AP the slowest.

Table 2. Mean annual growth rates for all three oyster size metrics determined from individual Linear regressions for each site. For BMY, no oysters were measured for whole wet weight. * denotes number of observations were < 10.

Site	Age Measured (yr)	Shell Height (mm yr^{-1})	Whole Wet Weight (g yr^{-1})	Dry Tissue Weight (g yr^{-1})
AP	0.3 - 3.7	14.66	19.57	0.48
BMY	0.3 - 1.6	*25.10	*--	*0.22
GSS	0.3 - 3.7	11.45	14.14	0.32
LBO	0.3 - 3.7	16.67	20.74	0.38
NI	0.3 - 3.7	17.75	25.00	0.63
SQR	0.3 - 2.7	20.55	*44.90	0.31

…e class basis by comparing final (Nov 2012) …atasets were available. These data generally …at NI and slowest at GSS (Figs. 7, 8 and 9).

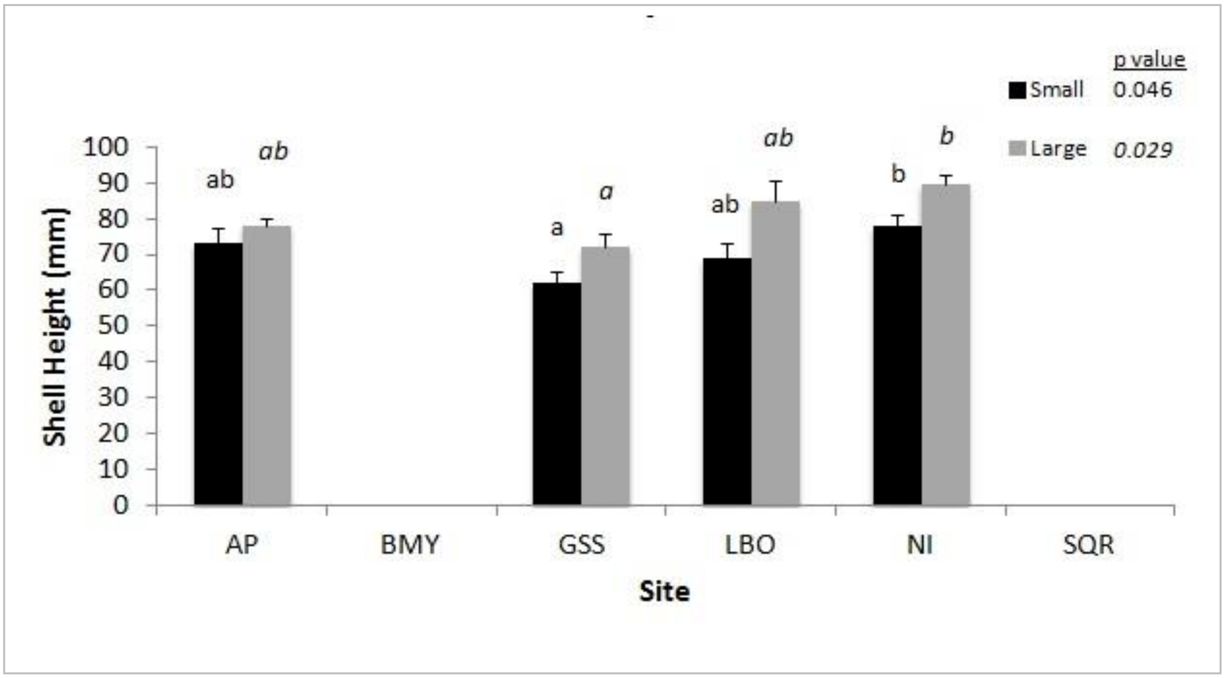


Fig. 7. Final shell heights (3-year growth) by size class for four sites (no final data were available for BMY and SQR) with complete datasets. P values shown for overall ANOVAs by size class and significance groups based on Tukey test (p<0.05) indicated with lower case letters (normal font for small size class, italicized for large class).

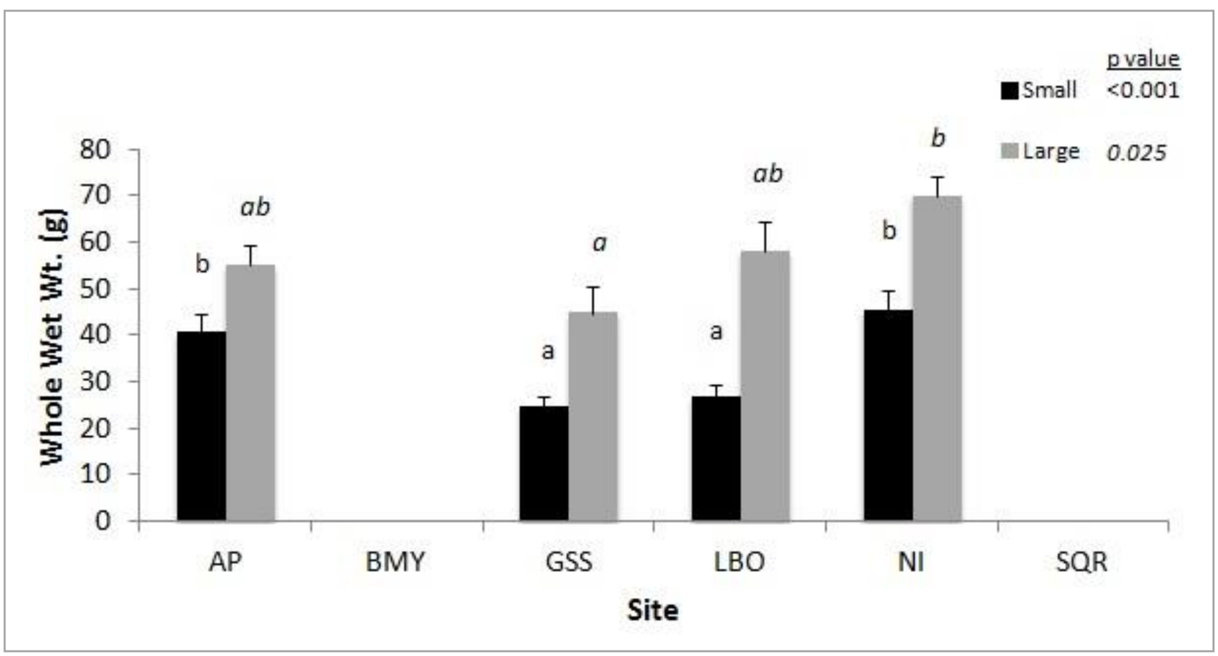
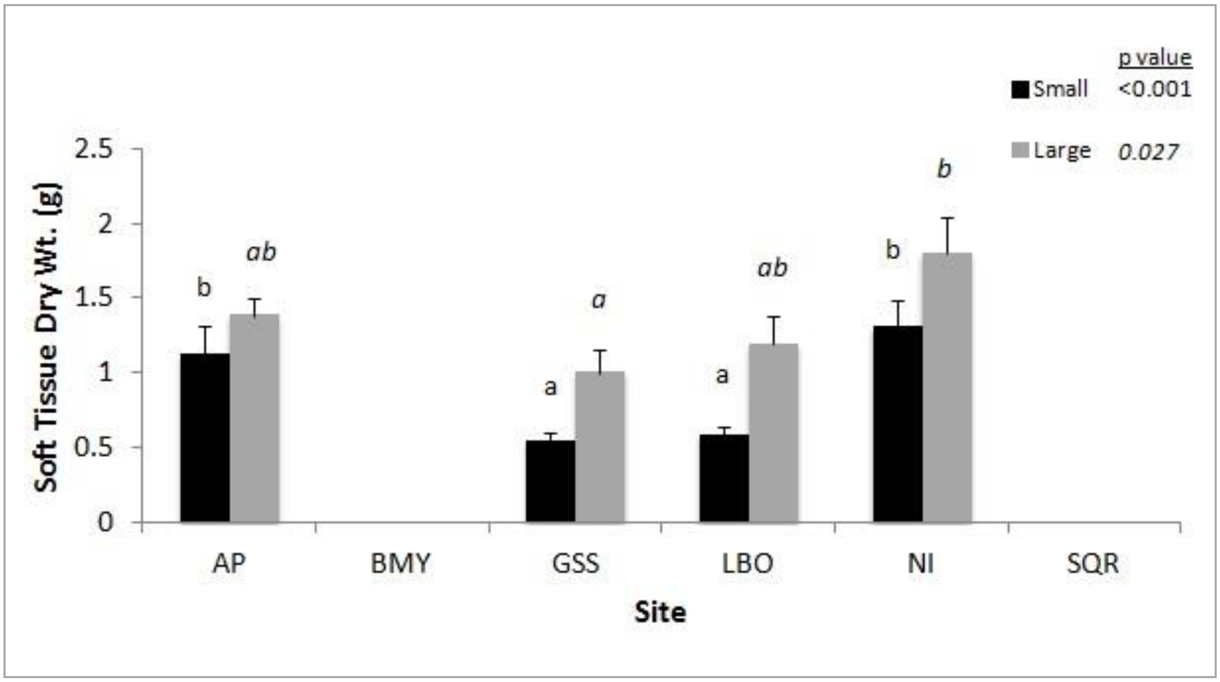


Fig. 8. Final whole wet (fresh) weight (3-year growth) by size class for four sites with complete datasets. P values shown for overall ANOVAs, etc. as in Fig. 7.

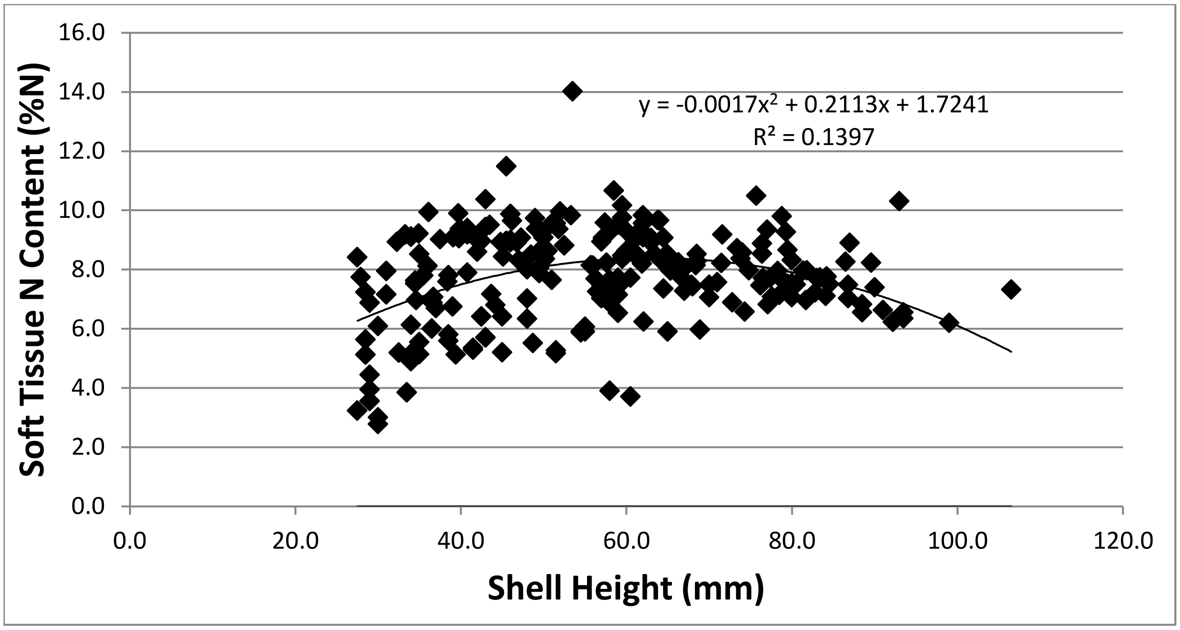


Growth of farmed and wild oysters typically varies widely among sites within an estuary, probably in most cases largely due to environmental variations (Shumway 1996). Thus, significant among-site differences in growth were expected (Brown et al. 1998; Higgins et al. 2011; Carmichael et al. 2012). See below for further discussion of the causes for the site effect for growth and nutrient content.

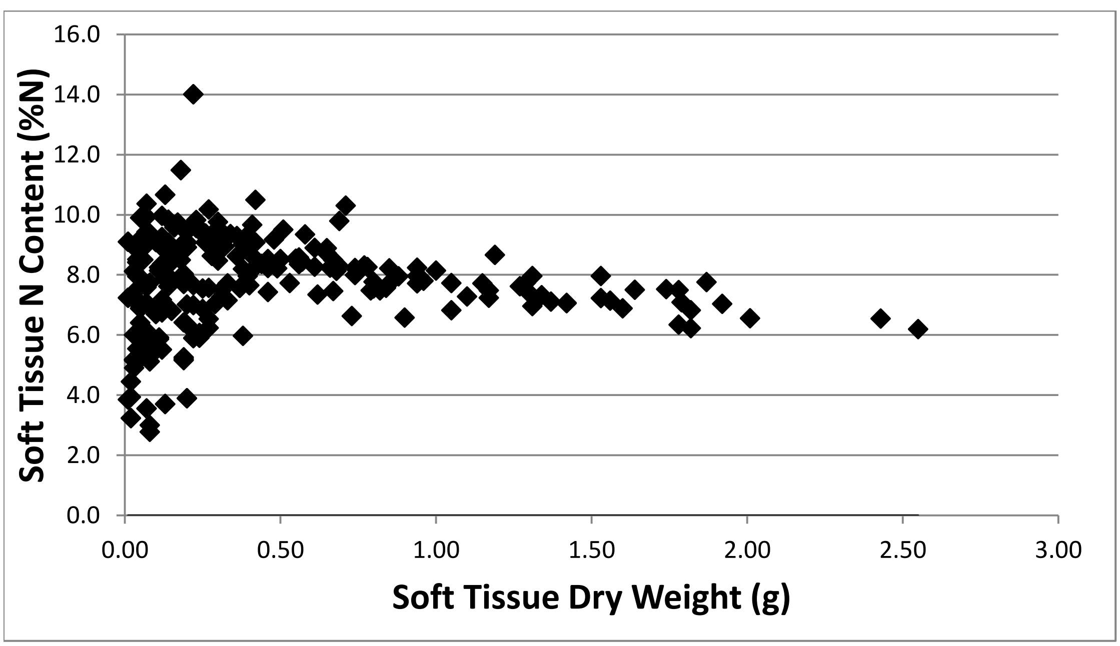
Oyster N and C content

Oyster N and C content were measured in soft tissue and shell on a per oyster basis. The data were assessed in three ways: in combined (overall dataset) form relative to various oyster metrics; by year, site, and season (spring and fall); and relative to various environmental measurements.

The combined dataset was initially assessed by scatterplots of various combinations of oyster size metrics and N and C content of soft tissue and shell. All plots showed no apparent pattern (and in some cases [e.g. %C of shell] little variability), except soft tissue %N relative to shell height and soft tissue DW. Shell height showed a weak relationship modeled by the quadratic equation shown in Figure 10, with an overall trend of decreasing N content on either side of the maximum: 8.3%N for an oyster of shell height 65 mm, which is the approximate size of a petite oyster. The model predicted 7.9%N for a 76 mm (regular) size oyster, and 5.6% for a jumbo (102 mm) oyster. Higgins et al. (2011) reported somewhat lower soft tissue %N in their 'regular' and 'jumbo' oysters (mean: 7.3%) compared to smaller sizes (mean: 8.1%), but not as wide a range as we found.



Soft tissue %N relative to soft tissue DW showed no discernable trend for those individuals less than ~0.5 g. However, in the range of 0.5 to 2.5 g DW there was a strong negative linear relationship with %N content decreasing from 9% to 6% (Fig. 11). Other than Higgins et al. (2011), we are aware of no previous research on the relationship between oyster size metrics and % nutrient content.



Percent N and C in soft tissue varied widely and significantly among study sites, with the same general pattern for both nutrients: lowest at site LBO and highest at SQR (Fig.12). Although no significant differences were found among the site means in %N and %C in the shell, the same overall trend was observed (Fig. 13). In a preliminary 3-month study, we reported significant differences for soft tissue nutrient content among the same six study sites (Grizzle and Ward 2011). Here we confirm this finding and quantify the site effect over a 3-year study period. To our knowledge, no previous studies have found a site effect for nutrient content. See below for further discussion of the site effect, including potential causes.

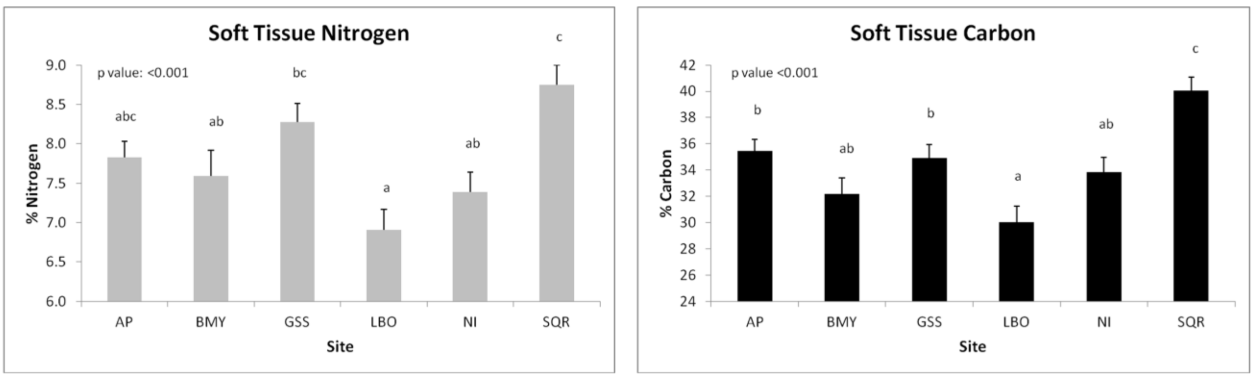


Fig. 12. Soft tissue %N and %C by site using combined data from all years and seasons. P values shown are for an ANOVA testing the effect of site. Letters indicate significance among means (Tukeys).

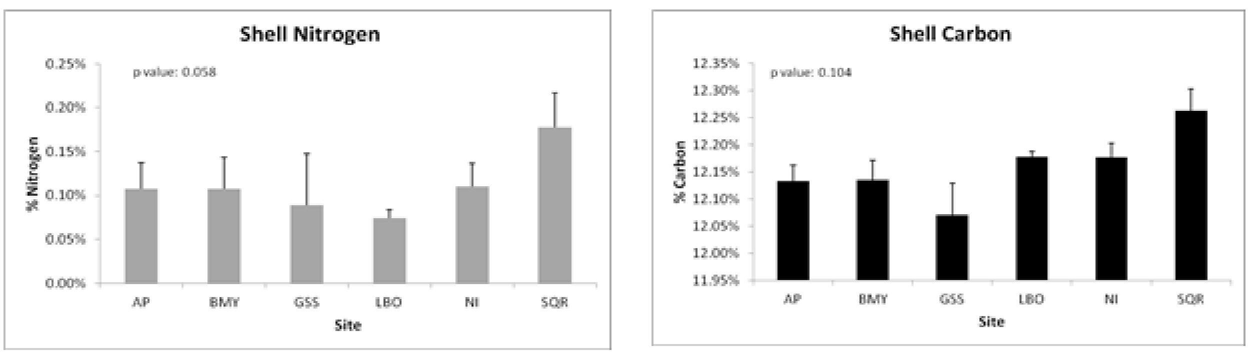
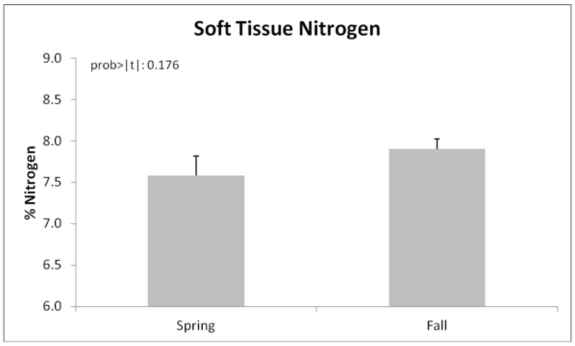


Fig. 13. Shell %N and %C by site; note: shell data were only available for Nov 2012 sampling (see Table 1). P values shown for overall ANOVAs, etc. as in Fig. 12.

Nutrient content was also assessed by season (only spring and fall data were available) by combining the data from all sites and both size classes. There was a significant difference between seasons for %C but not %N, though both showed the same trend: lowest in spring, highest in fall. Soft tissue nutrient content was expected to vary by season due to changes in overall condition related to reproductive state and perhaps over-winter stress. Thompson et al. (1996) found ~10% seasonal variation in protein content due to the oyster's reproductive cycle. Although we are not aware of other studies on seasonality of oyster nutrient content, our data (Fig. 14) suggest that it might be substantial.



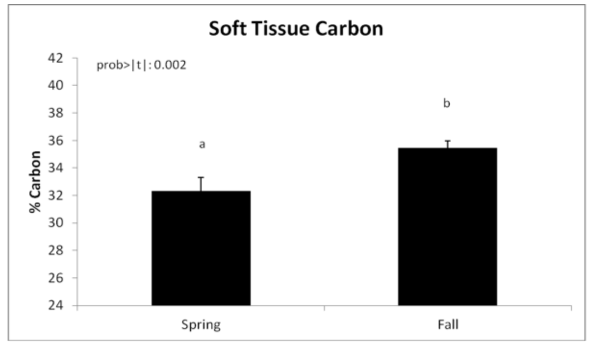
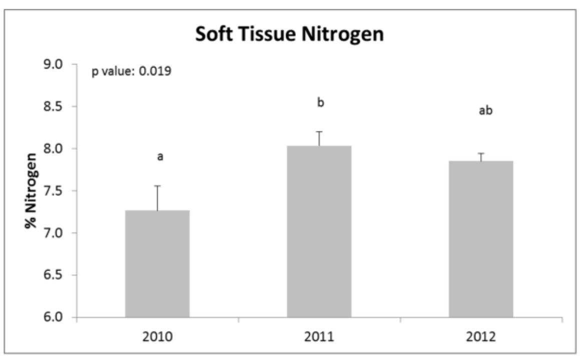


Fig. 14. Mean %N and %C in soft tissue by season (all sites, size classes combined). T-test results are shown.

The final oyster nutrient content assessment was by year. After combining data from all sites, sizes, and seasons, there were significant differences for soft tissue %N and %C, with the lowest levels for both in 2010 (Fig. 15). It should be noted, however, that these differences are confounded by oyster size; the oysters were smaller in 2010 than subsequent years and size was found to be related to nutrient content (see discussion above).



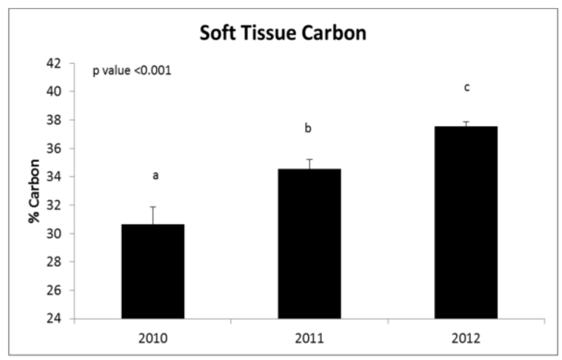


Fig. 15. Mean %N and %C in soft tissue by year (all sites, sizes, seasons combined). T-test results are shown.

Environmental variables and among-site differences in oyster metrics

The major environmental variables known to affect growth of bivalves were assessed as possible causes for the among-site differences (Shumway 1996): water flow, chlorophyll *a*, salinity, temperature, and dissolved oxygen. Nutrient concentrations (nitrate/nitrite and ammonia) in the water were also assessed as possible causes for among-site differences in N content. All data (except water flow) were provided by the UNH Tidal Water Quality Monitoring Program, which is funded by the Piscataqua Region Estuaries Partnership (PREP) and the Great Bay National Estuarine Research Reserve's System Wide Monitoring Program (SWMP; NOAA 2004). Referring to Table 3 below, the data presented are from monthly grab samples taken from 0.5 m below the water surface, and the months assessed represented the approximate time when oysters would likely be feeding (April through November). It should be noted that the six study sites for the present study did not coincide completely with the UNH monitoring locations. Referring to Table 3 below, there were no water monitoring sites near the Bellamy River, so only water flow data are provided for our BMY site. Their site GRBOR site is in the Oyster River about 3.5 km upstream from our GSS. Their GRBAP site is within 0.2 km of our AP site, and 3.0 km from our LBO site. Their GRBGB site is about

Table 3. Environmental data summary from water monitoring sites nearest each of the six study sites (see text for details).

Site	Water Flow Mean (cm/s)	Water Flow Max (cm/s)	Temp (°C)	Range	Salinity (ppt)	Range	Chl *a* (µg/L)	Range	Dissolved Oxygen (mg/L)	Range	Chl *a* Flux* (µg/cm²/s)	NO_2 + NO_3 (mg/L)	Range	NH_4 (mg/L)	Range
Bellamy River	19	34													
2010 (average) 04/08/2010-12/08/2010	-	-													
2011 (average)	-	-													
2012 (average)	-	-													
Granite State Shellfish (GRBOR)	35	60													
2010 (average) 04/09/2010-12/07/2010	-	-	16.8	2.4-30.3	22.8	.2-29	6.4	2.9-17.32	7.4	.6-12.9	223.5	0.111	.028-.202	0.059	.012-.175
2011 (average) 04/06/2011-11/29/2011	-	-	17.0	3.7-30.2	(no data)	(no data)	3.0	.66-6.48	7.7	2.4-14.4	105.5	0.102	.088-.215	0.102	.031-.082
2012 (average) 04/03/2012-12/10/2012	-	-	16.6	2.6-28.7	23.1	.2-29.5	3.4	1.33-9.49	8.0	2.4-13.2	117.9	0.069	.006-.141	0.048	.010-.098
Little Bay Oyster Co. (GRBAP)	16	34													
2010 (average)	-	-	11.1	1.6-22.9	23.9	11.7-30.7	4.1	1.22-16.93	9.4	6.46-11.25	64.8	0.094	<.005-.198	0.032	<.005-.092
2011 (average)	-	-	10.4	-0.7-23.1	21.1	6.9-29.4	2.9	.1-16.53	10.1	7.23-13.67	47.0	0.093	.025-.167	0.037	<.005-.079
2012 (average)	-	-	11.8	2.4-23.9	23.7	17.2-30.8	3.9	.88-12.97	9.4	7.0-11.38	62.5	0.089	<.005-.224	0.030	<.005-.083
Adams Point (GRBAP)	31	71													
2010 (average)	-	-	11.1	1.6-22.9	23.9	11.7-30.7	4.1	1.22-16.93	9.4	6.46-11.25	125.6	0.094	<.005-.198	0.032	<.005-.092
2011 (average)	-	-	10.4	-0.7-23.1	21.1	6.9-29.4	2.9	.1-16.53	10.1	7.23-13.67	91.1	0.093	.025-.167	0.037	<.005-.079
2012 (average)	-	-	11.8	2.4-23.9	23.7	17.2-30.8	3.9	.88-12.97	9.4	7.0-11.38	121.1	0.089	<.005-.224	0.030	<.005-.083
Nannie Island (GRBGB)	26	31													
2010 (average) 04/13/2012-11/23/2010	-	-	16.5	.1-27.2	24.0	.2-30.1	5.3	2.12-14.53	8.7	5.8-12.8	138.7	0.126	.06-.988	0.044	.02-.116
2011 (average) 04/19/2011-12/06/2011	-	-	16.1	3.9-27.2	20.4	10.3-29.9	3.4	.82-10.4	8.7	1.4-12.0	88.0	0.079	.007-.137	0.050	.014-.084
2012 (average) 04/13/2012-12/06/2012	-	-	16.4	2.2-26.6	24.6	10.6-31.4	3.5	1.32-10.61	8.7	1.1-43.5	90.4	0.065	.025-.122	0.023	.007-.058
Squamscott River (GRBSQ)	14	24													
2010 (average) 04/09/2010-12/03/2010	-	-	17.1	1.2-29.1	21.8	.1-177.5	6.7	2.6-13.17	8.2	.2-33.7	93.2	0.150	.107-13.17	0.101	.03-.215
2011 (average) 04/11/2011-12/07/2011	-	-	16.9	3.7-28.9	(no data)	(no data)	4.4	.36-10.11	8.4	3.4-14.9	61.6	0.105	.04-.131	0.122	.079-.179
2012 (average) 03/20/2012-12/06/2012	-	-	16.4	1.2-27.7	19.9	.3-30.9	6.2	2.06-28.65	8.6	2.5-23.8	86.2	0.113	.03-.131	0.150	.036-.311

*Chlorophyll flux = Mean water flow speed x Mean Chlorophyll a concentration

1.0 km from our NI site, and their GRBSQ is within 0.1 km of our SQR. In summary, our AP, NI and SQR sites were within 1 km of their corresponding UNH monitoring sites, and there were no known factors that might cause differences in water quality data between our sites and theirs. Although our LBO site was 3 km from its corresponding water quality site, both are in Little Bay in an area where no major tributaries enter or other potential confounding factors occur. Finally, although our GSS site was 3.5 km from its corresponding water monitoring site, it likely would have been strongly affected by discharges from the Oyster River on at least ebb tides.

Although some oyster metrics varied significantly by season and year, the focus here is on differences in oyster growth and nutrient content among the study sites relative to environmental factors in Table 3. It also should be noted that adequate growth data were only available for five of the six sites for this analysis (Table 2).

Two environmental factors were strongly (but not significantly) correlated with oyster growth metrics. Chlorophyll *a* (chl *a*) concentration was positively related to growth, and water flow speed negatively related. Chl *a* is often used as a proxy for overall food concentration, though other organic particulates are typically consumed and provide nutrition (Newell and Langdon 1996). Moreover, factors other than food concentration typically affect growth. Thus, although a positive relationship between growth and chl *a* was expected and has been reported in other field studies (e.g., Carmichael et al. 2012).

All three growth metrics were also negatively correlated with water flow speed, with shell height showing the strongest correlation. The relationship between water flow speed and bivalve growth in general typically shows an inverted parabolic shape if a wide enough range of speeds is tested (Kirby-Smith 1972; Wildish et al. 1992; Wildish and Kristmanson 1997; Peterson 2001). Although the full range of feeding and growth responses of the eastern oyster to water flow speed have not been well characterized, inhibition of feeding (and thus growth) are likely reached when speeds exceed 10 to 15 cm/s (Grizzle et al. 1992; Lenihan et al. 1996). Thus, the negative relationship we found might be expected. However, our oysters were held in bags which strongly diminish the actual water speed to which the oysters were likely exposed (Rheault and Rice 1996; Fredriksson et al. (2010)

Perhaps the most important finding relative to nutrient management was a positive correlation between soft tissue %N and shell %N and the mean dissolved inorganic N concentration in the ambient water. Neither correlation was significant and additional research is needed to further test and characterize the general relationship. Although we are aware of no previous studies linking bivalve nutrient content and ambient nutrient concentrations, the chemical composition of bivalve shells has been shown to be affected by a variety of environmental conditions (Rosenberg 1980). If the range of variation in soft tissue (6.9% - 8.6%) and shell (0.06% - 0.18%) N content observed among our study sites was mainly caused by variations in ambient N concentrations in the water, then better quantification of the relationship could be quite important in nutrient management involving bioextraction. In any case, our significant and substantial 'site effects' (Figs. 12 and 13) for oyster N and C content indicate that environmental factors are likely important in determining the bioextraction potential of farmed oysters.

Comparisons of summary data to previous studies

Table 4 below summarizes data from the present study for average N and C content on a per oyster basis, and compares our findings to previous research. Each of the studies differed somewhat in objectives and methods, so direct comparisons cannot be made for some metrics. When viewed from the perspective of assessing the bioextraction potential of farmed oysters, however, several observations can be made.

The mean values for %N and %C at the bottom of the table may be viewed as the nutrient content of the "average" farmed oyster based on studies from Alabama to New Hampshire. However, it should be noted that most of the research from which these means of means were calculated reported wide variability in nutrient content, and identified some of the causes for the variability. It seems reasonable to conclude at this time that oyster nutrient content (on a percent basis) can be expected to vary with environmental conditions (particularly dissolved nutrients in the water), from year to year, and seasonally. Much more research is needed to fully characterize the range of variability and the causal factors.

Higgins et al. (2011) and the present study report data on *percent* and *total* nutrient content of different size classes of oysters. As might be expected, there is a strong positive relationship between oyster size and *total* nutrient content. Higgins et al. (2011) provided models that relate size to total nutrient content based on a wide range of oyster sizes from their study area. The yellow shading in Table 4 highlights our directly comparable data, and indicate that for the two similar size classes (petites [65 and 68 mm] and regulars [86 and 80 mm]) the nutrient content values (percent and total) were very similar. Although we found a negative relationship between size and %N in both soft tissue and shell (Figs. 10 and 11), the decrease across the size range sampled was much less than the increase in total mass of the oyster. Thus, our larger oysters still had substantially more total N (and C) compared to smaller oysters, and all available data suggest this is likely the general trend.

Table 4. Summary data from relevant studies on N and C content of oysters, including wild and farmed.

Shell				**Soft Tissue**			**Whole Oyster**		
Height (mm)	**Dry Wt. (g)**	**%C**	**%N**	**Dry Wt. (g)**	**%C**	**%N**	**Total C (g)**	**Total N (g)**	**Source**
				Wild Oysters					
76	150		0.3	1		7		0.52	Newell et al. 2005
76				2.4 - 4.4		8.6		0.3 - 0.5	Carmichael et al. 2012
				Farmed Oysters					
44	4.8	11.8	0.2	0.20	43.3	8.2	0.65	0.03	Higgins et al. 2011
65	24.3	12.4	0.2	0.80	44.3	8.1	3.39	0.11	Higgins et al. 2011
86	37.6	12.4	0.2	1.58	45.1	7.3	5.38	0.18	Higgins et al. 2011
118	71.9	12.0	0.3	3.00	46.2	7.4	10.01	0.39	Higgins et al. 2011
114			0.21			9.3			Kellogg et al. 2013*
(>7 yr old)			0.15						Kellogg et al. (unpub data)*
42 and 98						9.1-13.5			Dalrymple & Carmichael (in prep)*
68	43.9	~12	~0.12	0.75	38.0	8.2	5.56	0.12	Present study
80	58.4	~12	~0.13	1.30	37.1	7.5	7.10	0.17	Present study
	MEAN:	**12.1**	**0.20**	**MEAN:**	**42.3**	**8.6**			

*Data taken from summary in STAC (2013)

Finally and as already discussed, farmed oysters are thought to typically have thinner shells compared to wild oysters and thus more total nutrient content. However, we are aware of only one study on wild oysters that provides comparative data (Newell et al. 2005; Table 4). The major recent research emphasis for wild oysters has been on their role in nutrient cycling (e.g., Piehler and Smyth 2011; Kellogg et al 2013; Hollein and Zarnoch 2014). However, when wild oysters are harvested for human consumption there is a removal of nutrients from the ecosystem. Thus, better quantification of the bioextraction potential of both farmed and wild harvested oysters as well as oysters grown using different aquaculture methods is needed.

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Appendix 3b:

Aquaculture Research

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Growth, morphometrics and nutrient content of farmed eastern oysters, *Crassostrea virginica* (Gmelin), in New Hampshire, USA

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Abstract

When harvested, oysters represent a removal from the ecosystem of nutrients such as nitrogen (N) and carbon (C). A number of factors potentially affect nutrient content, but a quantitative understanding across the geographical range of the eastern oysters is lacking. This study was designed to quantify the relationships among various metrics of farmed eastern oysters near its northern geographical range focusing on nutrient content. Hatchery-reared oysters were deployed in polyethylene bags at six sites, and were measured on multiple occasions from 2010 to 2012. A quadratic polynomial fit to the combined datasets for shell height indicated that on average a 'cocktail' size oyster (63 mm shell height) would be reached after 2 year, and 'regular' size (76 mm) would require 3 year. There significant differences in growth rates and oyster nutrient content among the sites; means for %N in soft tissue ranged from 6.9 to 8.6, and 0.07 to 0.18 in shell. Per cent N in soft tissue and shell were highest at two sites at the mouths of rivers with elevated dissolved inorganic N concentrations in the water. Grand means (all sites, seasons and years combined) of soft tissue N and C for regular size oysters were 7.3% and 38.5%, respectively; and for shell N and C were 0.13% and 12.0% respectively. Our study extends the range of data on nutrient content of the eastern oyster to northern New England, and indicates that oyster size, seasonality, and nutrient concentration in ambient water potentially affect %N and %C content of oysters.

Keywords: shellfish aquaculture, nutrient assimilation, bioextraction, eutrophication, water quality

Introduction

Eutrophication of coastal waters and estuaries persists as a global problem (Boesch 2002; Bricker, Wicks & Woerner 2008; Kroeze, Hofstra, Ivens, Löhr, Strokal & van Wijnen 2013). Management of nutrients such as nitrogen (N) and phosphorus (P) that contribute to eutrophication typically has focused on control of land-based sources. Recent research, however, has begun to quantify the potential of bivalve shellfish aquaculture as a nutrient management tool (Bricker, Rice & Bricker 2014; Rose, Bricker, Tedesco & Wikfors 2014; Saurel, Ferreira, Cheney, Suhrbier, Dewey, Davis & Cordell 2014). Suspension-feeding bivalves such as oysters and mussels remove suspended particulates and associated nutrients from the water as they feed, and when harvested represent a direct removal of the assimilated nutrients from the ecosystem (Hammer 1996; Newell 2004; Lindahl, Hart, Hernroth, Kollberg, Loo, Olrog & Rehnstam-Holm 2005; Ferreira, Hawkins & Bricker 2007, 2011; Burkholder & Shumway 2011; Coen, Dumbauld & Judge 2011; Higgins, Stephenson & Brown 2011; Carmichael, Walton & Clark 2012; Pollack, Yoskowitz, Hae-Cheol & Montagna 2013). Farmed (and wild) bivalves also potentially cause nutrient removal by enhancing burial in sediments, and in the case of nitrogen, increased denitrification rates (Newell, Fisher, Holyoke &

Cornwell 2005; Pietros & Rice 2009; Piehler & Smyth 2011; Carmichael, Walton *et al.* 2012; Hoellein & Zarnoch 2014; Hoellein, Zarnoch & Grizzle 2015).

In the mid-Atlantic and northeastern United States, eastern oyster (*Crassostrea virginica*) aquaculture is a rapidly growing industry and recent research has begun to quantify the potential role oyster farming might play in nutrient management (Newell & Mann 2012; STAC, 2013; Kellogg, Smyth, Luckenbach, Carmichael, Brown, Cornwell, Piehler, Owens, Dalrymple & Higgins 2014; Rose *et al.* 2014). In the Great Bay Estuary where this study was conducted, the major nutrient of concern is N (PREP 2013), and research is underway to assess the role oyster farming might play in its management.

Estimates of nutrient removal by farmed oysters are based on the nutrient content of individual oysters. The meagre but growing literature indicates substantial spatial variability in N content of oysters but the causes or generality of such variability is not known (Grizzle & Ward 2011; Higgins *et al.* 2011). There are also temporal variations in nutrient content due to physiological changes in the oyster but these effects have not been well quantified in farmed oysters (Newell & Mann 2012; STAC 2013). Finally, more data on size-related variations in nutrient content are needed to make nutrient removal predictions for different harvest scenarios (Higgins *et al.* 2011). This study was designed to quantify the relationships among various metrics of farmed eastern oysters near its northern geographical range focusing on nutrient content.

Materials and methods

Study area and field methods

Six sites were chosen to represent a range of ambient nutrient concentrations and other environmental conditions, and locations within the estuary: two oyster farm sites, Granite State Shellfish (GSS) at the mouth of the Oyster River and Little Bay Oyster Company (LBO) in Little Bay; Adams Point (AP) in Little Bay; the mouths of the Bellamy (BMY) and Squamscott Rivers (SQR); and Nannie Island (NI) in Great Bay (Fig. 1). Although we did not measure water quality parameters as part of this study, we did compare our oyster data to water data available during the same 2.3 year period from the Piscataqua Region Estuaries Partnership (PREP) and the Great Bay National Estuarine Research Reserve's System Wide Monitoring Program (see Grizzle, Ward, Peter, Cantwell, Katz & Sullivan 2014 for details). The aim here was to look for trends in oyster nutrient content and ambient water quality parameters in the study area.

Two size classes [small: 19–42 mm initial shell height (0.3 year old); large: 28–79 mm initial shell height (1.3 year old)] of hatchery-reared oysters were obtained from LBO in August 2010. Before deployment at the study sites, thirty (30) oysters were haphazardly selected from each of the two size classes, shell height and whole wet weight measured, then frozen and shipped to USEPA Atlantic Ecology Division laboratory for elemental analysis (see Laboratory Methods section).

At each of the six study sites, ~1000 small size class oysters were deployed in a 9 mm mesh and ~200 large size class oysters in a 24 mm mesh flat ('envelope' style) polyethylene bag. The two bags were suspended side-by-side 10 cm off the bottom attached to plastic coated wire cages. Table 1 below summarizes the timeline of sampling and analyses for the study period August 2010–November 2012; note that timing of sampling and sample size varied depending on oyster mortality, funding and other constraints.

Laboratory methods

Size measurements consisted of shell height (calipers to nearest mm), shell and soft tissue dry weight (DW; 5+ days at 60°C), and whole wet (fresh) weight (to nearest 0.1 g). For soft tissue N and C content (measured at USEPA Laboratories, Narragansett, RI, USA), the frozen oysters were re-measured (shell height), shucked (soft tissue) into aluminium pans, oven-dried at 60°C and weighed (soft tissue DW), and homogenized by mortar and pestle. The samples were analysed following a standardized laboratory operating procedure using a Thermo-Finnigan Flash EA1112 CHN/O elemental analyzer which was calibrated prior to analysis with certified standards. Each tissue sample was analysed in duplicate and the results reported as %N and %C. For shell N and C content (measured at UNH's Water Quality Analysis Laboratory in the Department of Natural Resources & the Environment), a 1 cm wide section of shell was cut using a bandsaw along the

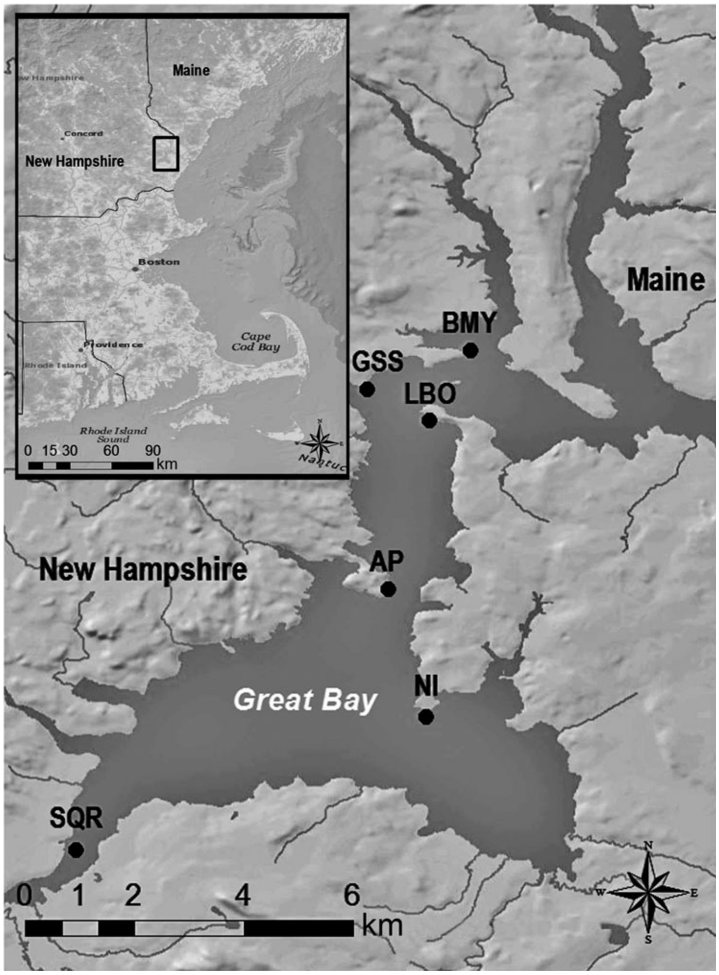


Figure 1 Six sites where oysters were deployed 2010–2012 (see text for details).

length dimension at a point about 2 cm from the umbo of both valves, dried, weighed and ground with a mortar and pestle until homogeneous. Ground samples were analysed for %N and %C on an elemental analyzer (Perkin Elmer CHN 2400 Series II, Waltham, MA, USA) using high temperature combustion and thermal conductivity detection.

Data analysis

The effect of site on oyster growth was tested based on the final size data for each of the two size classes. All oysters used in the experiment came from the same population (i.e. initial sizes for both size classes were the same for all six study sites), and the experiment was run for the same time period (~2.3 years) for all six study sites. Thus, final size reflected overall growth variations; these data were examined using ANOVAs. Oyster growth also was assessed by plotting combined (all sites and seasons) size data over time and fitting appropriate models to the data; annual growth rates were determined for each site using linear regression.

Oyster soft tissue N and C content were analysed across year, season and site using ANOVAs and *t*-tests when appropriate. For all analyses, residuals were examined to ensure homogeneous variance and a normal distribution. Outliers were tested for and removed when appropriate. Oyster shell N and C content were only determined on the final sampling (Table 1), so those data were only analysed by site. Shell %C was non-normally distributed despite transformations and analysed with a non-parametric Kruskal–Wallis. Prior to analyses on tissue and shell chemistry, potential effects of oyster age were tested because oysters were from two distinct initial age groups (0.3 and 1.3 year). No discernable effects of oyster age were evident, thus the two size/age classes were analysed

Table 1 Details on sampling and analysis of oysters by size class (small and large)

Date	Measurement	No. of oysters analysed	Oyster age, by size class (year)
August 2010	Size	30 small 30 large	0.3 for small 1.3 for large
November 2010	Size	28 small 30 large	0.6 for small 1.6 for large
April 2011	Soft tissue N and C	47 small 30 large	1.0 for small 2.0 for large
November 2011	Size	17 small 11 large	1.7 for small 2.7 for large
November 2011	Soft tissue N and C	30 small 16 large	1.7 for small 2.7 for large
April 2012	Soft tissue N and C	8 small 10 large	2.1 for small 3.1 for large
November 2012	Size	22 small 25 large	2.7 for small 3.7 for large
November 2012	Soft tissue N and C	40 small 40 large	2.7 for small 3.7 for large
November 2012	Shell N and C	29 small 25 large	2.7 for small 3.7 for large

together. A Tukey's HSD post hoc test was run to determine differences among means for ANOVAS. Statistical significance for all tests was set at an alpha level of 0.05 to control Type I error.

Results

Oyster growth and morphometrics

Oyster growth was assessed by combining data from all six sites for both size classes and fitting models to the plotted data, and by site and in the context of environmental differences. For shell growth, a quadratic polynomial fit to the combined (all sites, size classes, seasons) datasets for shell height indicated that, on average, a cocktail size oyster (63 mm) would be reached in the third growing season, and regular size (76 mm) would require a full 3 years (Fig. 2). However, there was wide variability among individual oysters on each sampling date with some almost reaching regular size in the second growing season. Growth as measured by whole wet weight and soft tissue DW showed similar trajectories and variability. However, because winter data (when both would have typically decreased) were not consistently available (see Table 1), they were not modelled.

Oyster growth also was assessed by site and size class by comparing final (November 2012) size metrics from the four sites for which complete datasets were available (oysters were lost during the final year at sites BMY and SQR). This assessment represented relative growth differences among the sites because each of the two size classes was drawn from the same population at the initiation of the study. Thus, the final sizes represented ~2.3 years of growth. Both size classes of oysters differed significantly among the sites, with slowest growth for both size classes at Site GSS and fastest at Site NI (Fig. 3). Although the data are not presented herein, the same overall trend was shown for whole wet weight and soft tissue DW.

The focus for this study was on metrics usually reported in the aquaculture literature. As shell height is the primary metric affecting marketability and selling price, other metrics were compared to it. Oyster shell height and whole oyster fresh (wet) weight data from the overall dataset (all years, sites, seasons combined) showed the expected non-linear relationship, and a power function provided the best fit to the data (Fig. 4). In the New England region, most of the market is comprised of two size classes: <76 mm (3 in) ('petite' or 'cocktail'), and 76–102 mm (3–4 in; 'regular'). Our data indicate an average petite oyster (assuming 65 mm height) would have a fresh weight of ~20 g; an average regular (86 mm) would be ~70 g (Fig. 4).

Soft tissue DW showed the same general relationship with shell height (Fig. 5) as fresh weight (Fig. 4). An average petite oyster (assuming 65 mm height) would have a soft tissue DW of 0.48 g; an average regular (86 mm) would be 1.20 g (Fig. 4). Shell DW was only measured on oysters from the final sampling date only (see Table 1), so overall shell height range was not as wide as the other datasets. Based on these final size measurements, an average petite oyster (65 mm height) would have a shell DW of 19.4 g; an average regular (86 mm) would be 38.5 g (Fig. 6).

Oyster N and C content relative to oyster metrics

Oyster N and C content were measured in soft tissue and shell on an individual oyster basis. The data mainly were assessed by combining data from the overall dataset (all sites, seasons and sizes) and assessing scatterplots produced from different combinations of oyster metrics. All plots showed no apparent pattern [and in some cases (e.g. %C of shell) little variability], except soft tissue %N

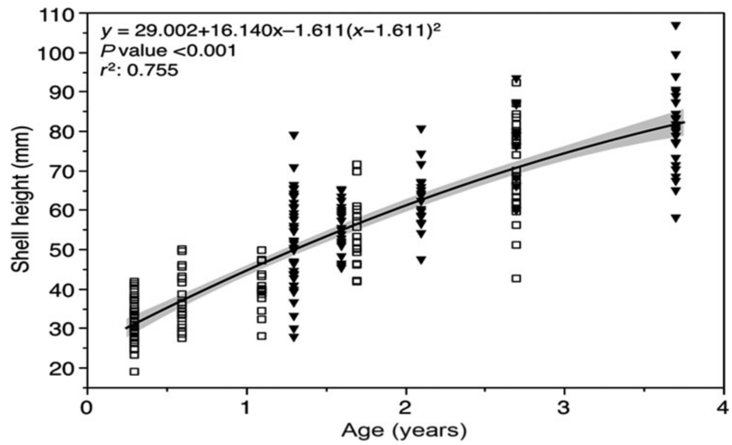


Figure 2 Oyster growth as shell height using combined datasets from all sites, seasons, size classes. The grey shaded area around the fit line is 95% confidence interval. Open squares = small size class; Black inverted triangles = large size class.

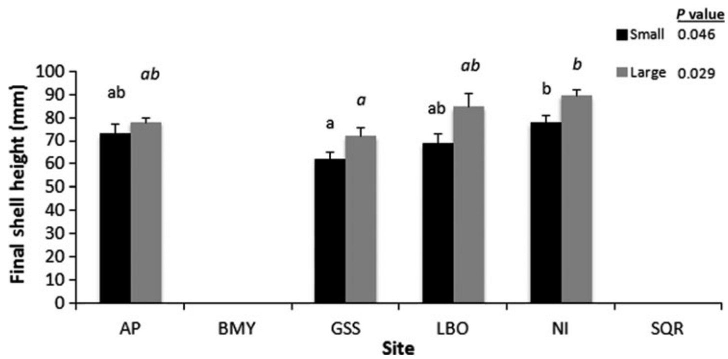


Figure 3 Final shell heights (2.3 years of growth) by size class for four sites (no final data were available for BMY and SQR) with complete datasets. P-values shown for overall ANOVAs by size class and significance groups based on Tukey test (P < 0.05) indicated with lower case letters (normal font for small size class, italicized for large class).

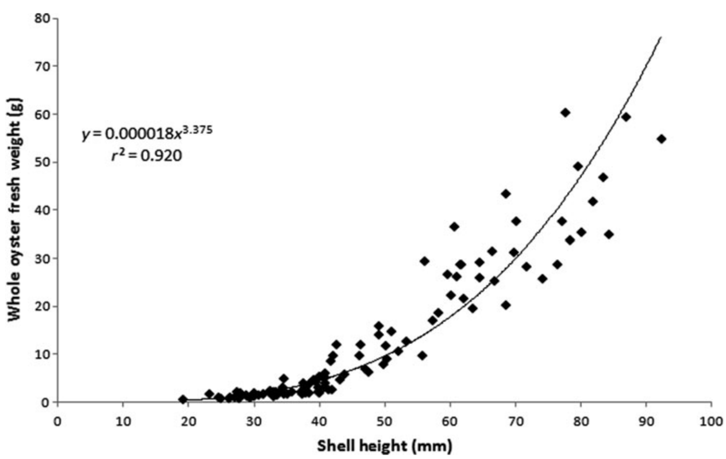


Figure 4 Relationship between shell height and whole oyster wet (fresh) weight for overall dataset (all sites, seasons and size classes combined). Best-fit power regression model shown.

relative to shell height and soft tissue DW. Shell height had a weak relationship modelled by the quadratic equation in Fig. 7, with an overall trend of decreasing N content on either side of the maximum: 8.3 %N for a petite oyster (shell height ~65 mm). The fitted model predicted 7.9 %N for a 76 mm (regular) size oyster, and 5.6% for a jumbo (102 mm) oyster.

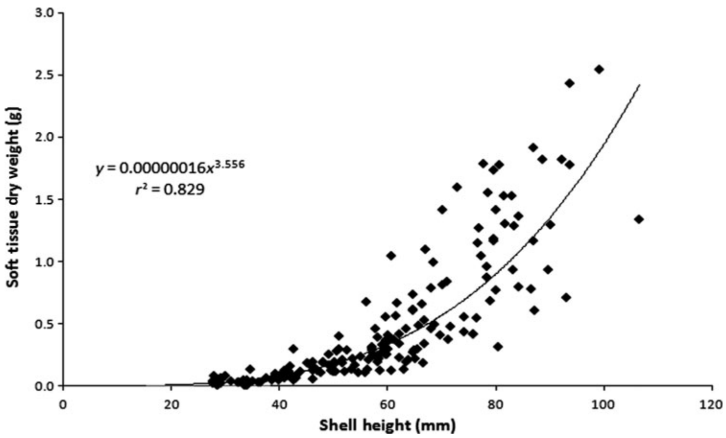
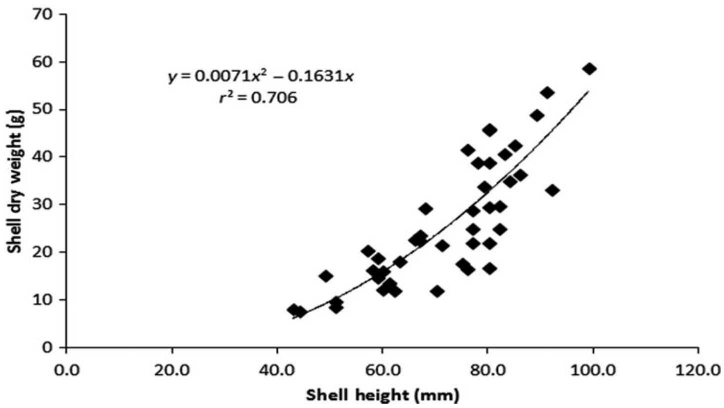
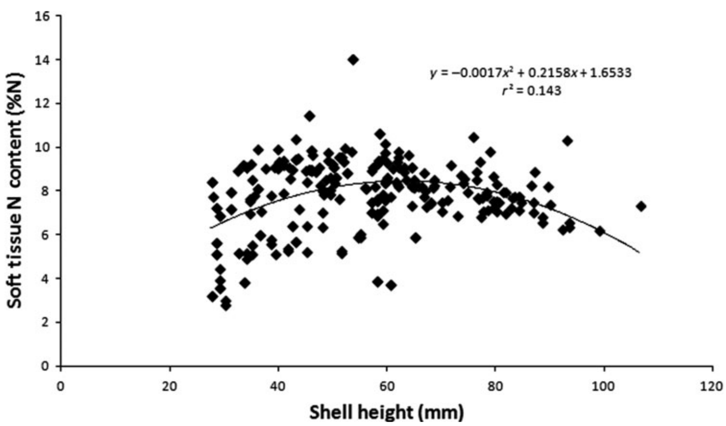


Figure 5 Relationship between shell height and soft tissue DW for overall dataset (all sites, seasons and size classes combined). Best-fit power regression model shown.





Soft tissue %N relative to soft tissue DW showed weak and widely variable positive trend for those individuals less than ~0.5 g. However, in the range of 0.5–2.5 g DW there was a strong negative linear relationship with %N content decreasing from 9% to 6% (Fig. 8). These data strongly suggest that there is a substantial decrease in soft tissue N content as an oyster ages.

Oyster N and C content by site

Soft tissue %N and %C varied widely and significantly among study sites, with the same general pattern for both nutrients: lowest at site LBO and highest at SQR (Fig. 9). Although no significant differences were found among the site means in %N and %C in the shell, the overall

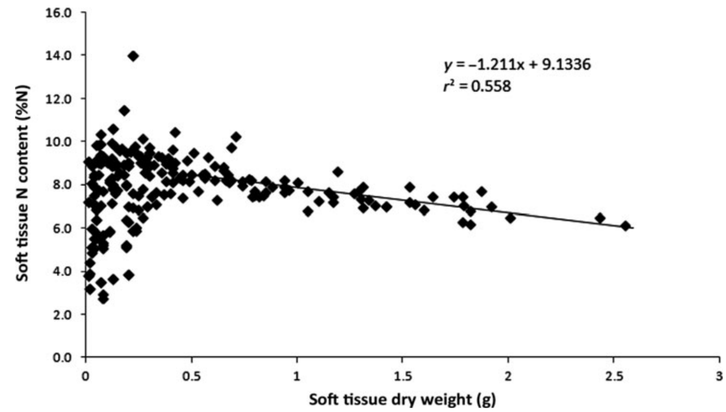


Figure 8 Relationship between soft tissue DW and soft tissue %N from overall dataset (all years, sites, seasons); linear model shown only for those individuals = or >0.5 g DW.

trends were similar to that for soft tissue (Fig. 9).

Discussion

Oyster growth and morphometrics

Oyster growth was assessed by combining data from all six sites for both size classes and fitting models to the plotted data, and by site. The two major findings with respect to growth were that on average harvest-size oysters would be reached in the third growing season (Fig. 2), and there was wide variability among individual oysters on each sampling date. Annual growth rates of the eastern oyster typically vary widely across its range with fastest growth in the warmer regions, likely due to extended growing seasons (Kraeuter, Ford & Cummings 2007). For farmed oysters, deployment methods may also strongly affect

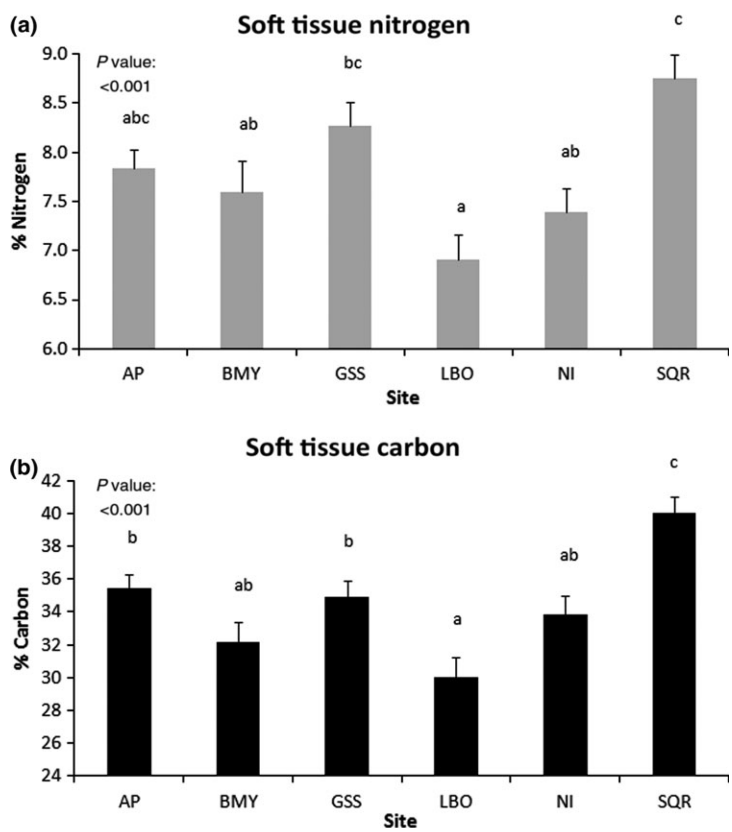


Figure 9 Soft tissue %N and %C by site using combined data from all years and seasons. *P*-values shown are for an ANOVA testing the effect of site. Letters indicate significance among means (Tukeys).

growth (Comeau 2013). In her review, Shumway (1996) noted that market size (90 mm) could be attained in 2 year in the Gulf of Mexico, compared to 4 or 5 year in Long Island Sound. Recent studies reported market size oysters were attained in 2 year in the Chesapeake (Harding, Mann & Southworth 2008; Harding, Mann, Southworth & Wesson 2010), similar to our findings for Great Bay.

Growth also varied widely among the six sites (Fig. 3). The smallest oysters at the end of the study (~2.3 year growth period) for both size classes occurred at Site GSS, with similar growth occurring at the other sites. Although the data are not presented herein, the same overall trend was shown for whole wet weight and soft tissue DW. Growth of farmed and wild oysters typically varies widely among sites within an estuary, probably in many cases due to environmental variations (Shumway 1996; Brown, Butt, Shelton & Paynter 1998; Higgins *et al.* 2011; Carmichael, Shriver & Valiela 2012). Thus, significant among-site differences in growth were expected. Although we do not have sufficient environmental data to fully explain the oyster growth variations observed among the study sites, some comments can be made. In Grizzle *et al.* (2014), we summarized environmental data from several routine water quality monitoring sites near our oyster study sites. The major differences relevant to oyster growth among the monitoring sites were in tidal current speeds and chlorophyll *a* concentrations, and both were correlated with oyster growth.

Morphometric relationships in our study focused on how shell height, which typically determines harvest time and is the most widely reported oyster metric in general, was related to other metrics (Figs 4–7). Combining data across all sites and seasons, an average petite oyster (assuming 65 mm height) would have a soft tissue DW of 0.48 g; an average regular (86 mm) would be 1.20 g. These values are slightly lower than those reported for soft tissue DW of farmed oysters in the Chesapeake Bay by Higgins *et al.* (2011). Reitsma, Murphy and Archer (2014) reported much heavier soft tissue DW (mean: 2.53 g) for regular size diploid oysters, but their triploids had similar values to ours and Higgins *et al.* (2011).

Shell DW was measured on oysters from the final sampling date only (Table 1), so overall shell height range was not as wide as the other datasets. Based on these final size measurements, an average petite oyster (65 mm height) would have a shell DW of 19.4 g; an average regular (86 mm) would be 38.5 g (Fig. 7). These values are very similar to those reported by Higgins *et al.* (2011) for farmed oysters, and much lower than Newell *et al.* (2005) for wild oysters which may typically have thicker and heavier shells than farmed oysters (Paynter & Dimichele 1990; see discussion in Higgins *et al.* 2011; Newell & Mann 2012). However, Reitsma *et al.* (2014) reported shell DW values for their wild oyster to be within the range reported for farmed (see Table 2). Moreover, it should be noted that farmed oysters can be grown using methods ranging from confinement in bags from seed-to-market size (as in this study), to various combinations of gear and bottom seeding. And in some areas, oyster aquaculture involves harvesting of wild-set oysters on leased bottom areas, with no grow-out gear at all. Thus, the trend of 'farmed' oysters having thinner shells than wild may be dependent on culture methods used and perhaps other factors.

Oyster N and C content

Oyster N and C content were measured in soft tissue and shell on a per individual oyster basis at multiple times over the ~2.3 year study period (Table 1). The data were assessed in three ways: combined (overall dataset) relative to various oyster metrics; by year, site, and season (spring and fall); and relative to environmental measurements.

The major overall finding relevant to oyster nutrient content was wide variability in most metrics, particularly soft tissue %N relative to shell height (Fig. 7) and soft tissue DW (Fig. 8). Shell height showed a weak relationship modelled by the quadratic equation shown in Fig. 7, with an overall trend of decreasing N content on either side of the maximum: 8.3 %N for a petite oyster (shell height ~65 mm). The model predicted 7.9 % N for a 76 mm (regular) size oyster, and 5.6% for a jumbo (102 mm) oyster. Higgins *et al.* (2011) reported somewhat lower soft tissue %N in their 'regular' and 'jumbo' oysters (mean: 7.3%) compared to smaller sizes (mean: 8.1%), but not as wide a range as we found. Our finding of a strong decline from 9% to 6 %N as soft tissue DW increased from 0.5 to 2.5 g (Fig. 8) has not to our knowledge been reported previously.

Soft tissue %N and %C varied widely and significantly among study sites, with the same general

Table 2 Summary data from relevant studies on N and C content of eastern oysters, including wild and farmed. Shaded data are from regular size farmed oysters

Shell				Soft tissue			Whole oyster			
Height (mm)	Dry Wt. (g)	%C	%N	Dry Wt. (g)	%C	%N	Total C (g)	Total N (g)	Study region	Source
Wild oysters										
76	150		0.30	1		7		0.52	Mid-Atlantic	Newell *et al.* 2005;
83	46		0.26	2.42		8.2		0.31	New England	Reitsma *et al.* 2014;
Farmed oysters										
44	4.8	11.8	0.18	0.20	43.3	8.2	0.65	0.03	Mid-Atlantic	Higgins *et al.* 2011;
65	24.3	12.4	0.19	0.80	44.3	8.1	3.39	0.11		Higgins *et al.* 2011;
86	37.6	12.4	0.17	1.58	45.1	7.3	5.38	0.18		Higgins *et al.* 2011;
118	71.9	12.0	0.26	3.00	46.2	7.4	10.01	0.39		Higgins *et al.* 2011;
76				2.4–4.4		8.6		0.20–0.38	New England	Carmichael, Walton *et al.* 2012;
114			0.21			9.3			Mid-Atlantic	Kellogg, Cornwell, Owens & Paynter 2013*
42 and 98						9.1–13.5			Gulf of Mexico	Dalrymple & Carmichael*
85†	47.4		0.26	2.70		7.9		0.32	Massachusetts	Reitsma *et al.* 2014;
83‡	35.7		0.21	2.36		8.0		0.26		Reitsma *et al.* 2014;
87§	22.3		0.32	1.36		8.5		0.19		Reitsma *et al.* 2014
65	19.4	12.0	0.12	0.48	38.0	8.3	2.51	0.06	New England	Present study
86	38.5	12.0	0.13	1.20	37.1	7.3	5.07	0.14		Present study
Shaded means	**36.3**	**12.1**	**0.22**	**1.84**	**41.1**	**7.8**	**5.23**	**0.22**		

*Data taken from summary in STAC (2013).
†Cultured on-bottom.
‡Cultured off-bottom.
§Off-bottom triploid.

pattern for both nutrients: lowest at site LBO and highest at SQR (Fig. 9). Although no significant differences were found among the site means in % N and %C in the shell, the overall trends were similar to that for soft tissue (Fig. 10). In a preliminary 3-month study, we reported significant differences in soft tissue N and C content among the same six study sites (Grizzle & Ward 2011). Here, we confirm this finding and quantify the site effect over a 2.3 year study period. To our knowledge, no previous studies have found a site effect for nutrient content. As already noted, although we did not measure water quality parameters during the study, the two sites with highest N concentrations in soft tissue and shell were in the general area of routine water quality monitoring sites that had the highest mean dissolved inorganic N concentrations (see Grizzle *et al.* 2014 for details). Carmichael, Shriver *et al.* (2012) found a positive relationship between N loadings to the estuary and tissue N content for hard clams and softshell clams, but not oysters. In their review, Kellogg *et al.* 2014 noted the need for more site-specific data due to the reported wide variations in N content of oysters.

N and C content were also assessed by season (only spring and fall data were available) by combining the data from all sites and both size classes. There was a significant difference between seasons for %C but not %N, though both showed the same trend: lowest in spring, highest in fall (see Grizzle *et al.* 2014 for data). Soft tissue nutrient content was expected to vary by season due to changes in overall condition related to reproductive state and perhaps over-winter stress. Thompson, Newell, Kennedy and Mann (1996) found ~10% seasonal variation in protein content due to the oyster's reproductive cycle. Although we are not aware of other studies on seasonality of oyster nutrient content, our data suggest it might be substantial.

Table 2 summarizes data from this study for average N and C content on a per oyster basis (all sites and years combined), and compares our findings to previous research. Each of the studies in Table 2 differed somewhat in objectives and methods, so direct comparisons cannot be made for

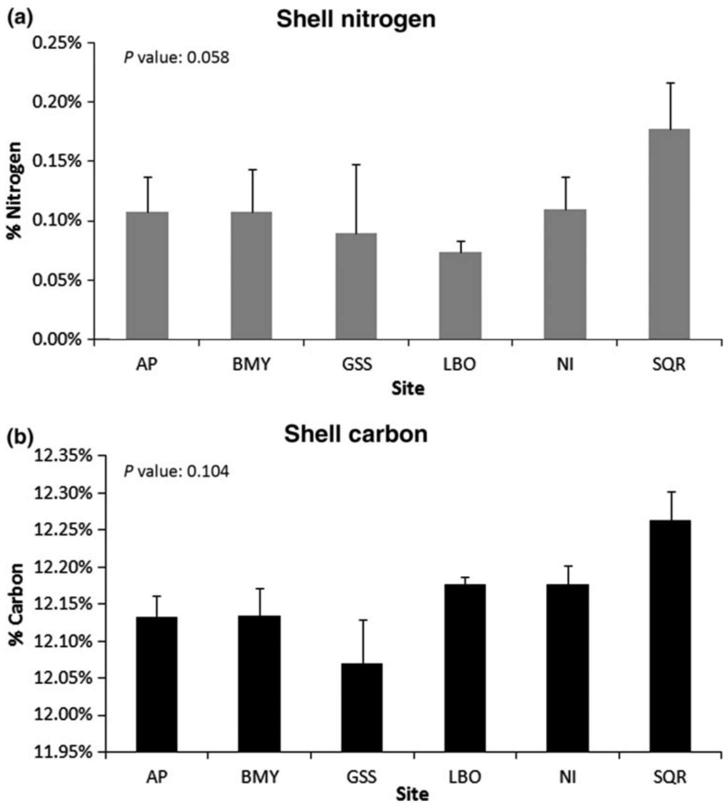


Figure 10 Shell %N and %C by site; note: shell data were only available for November 2012 sampling (see Table 1). *P*-values shown for overall ANOVAs, etc. as in Fig. 9.

some metrics. Nonetheless, trends for regular size (shaded data in Table 2) oysters are clearly evident. Although mean shell height only ranged from 83 to 87 mm, shell and soft tissue DW varied by >50%. Although %N in shell also varied by ~50%, soft tissue %N only ranged from 7.3 to 8.5. Thus, the wide range of mean total N (0.14–0.32 g) for a regular size oyster appears to be largely a factor of variations in weight and N content in the shell. Another trend were differences in total N in wild oysters (mean: ~0.4 g) compared to farmed (0.22 g). Although few data are available for wild oysters, these differences may largely be due to the heavier shell weight sometimes reported for wild oysters (see discussion above).

It should be noted that Higgins *et al.* (2011) and this study report data on *per cent* and *total* nutrient content of different size classes of oysters. As might be expected, there is a strong positive relationship between oyster size and *total* nutrient content. Higgins *et al.* (2011) provided models that relate size to total nutrient content based on a wide range of oyster sizes from their study area. They also report mean data for two size classes [petites (65) and regulars (86 mm)] similar to our findings in this study (Table 2). Although we found a negative relationship between size and %N in both soft tissue and shell (Figs 7 and 8), the decrease across the size range sampled was much less than the increase in total mass of the oyster. Thus, our larger oysters still had substantially more total N (and C) compared to smaller oysters, and it seems reasonable to expect this to be the general trend.

Finally, although we did not measure water quality parameters as part of this study, we did compare our oyster data to water data available during the same 2.3 year period (see Grizzle *et al.* 2014 for details). Data from the two water monitoring stations nearest our sites SQR and GSS had the highest mean dissolved inorganic N concentrations, which were the two stations with highest soft tissue N concentrations (Fig. 9), and SQR had the highest shell N concentration (Fig. 10).

Management implications

As noted above, N enrichment of Great Bay is a major concern of managers. Farmed oysters are directly involved in the movement of N through ecosystems via three physiological processes: filtration/assimilation, excretion and biodeposition. Oysters also affect subsequent microbially mediated processes such as denitrification (Newell 2004; Newell & Mann 2012; Kellogg *et al.* 2014). This study focused on the effects of filtration/assimilation by measuring N content in shell and soft tissue, which represents N removal from the ecosystem when oysters are harvested ('nutrient bioextraction'; Rose *et al.* 2014). Here, we briefly consider our data in the overall context of N management in the Great Bay estuarine system.

The data herein were used in a companion study to assess the N bioextraction potential of oyster aquaculture in Great Bay on various scales (Bricker, Ferreira, Zhu, Rose, Galimany, Wikfors, Saurel, Miller, Wands, Trowbridge, Grizzle, Wellman, Rheault, Steinberg, Jacob, Davenport, Ayvazian & Tedesco 2015). Based on measured environmental data from two of this study sites and other data typical of oyster farms in the area, the online FARM model (www.farmscale.org) predicted an average total N removal of 142 kg ha^{-1} year^{-1} of (=0.35 metric ton ac^{-1} year^{-1}) from the ecosystem via oyster harvest. A more direct approach using a mean of 0.14 g total N/regular size oyster (Table 2), and assuming a harvest of 1 000 000 oysters ha^{-1} year^{-1} (which is probably near maximal), indicates a total N removal upon harvest from the Great Bay system of 140 kg N. In comparison, Higgins *et al.* (2011) estimated a total N removal of 132 kg ha^{-1} year^{-1} based on data from farms in the Chesapeake Bay region. Thus, we think it reasonable to conclude that at the individual farm-scale realistic annual predictions of total N removal via harvest can be made. However, moving to larger spatial and temporal scales is more complicated.

Oyster aquaculture is a rapidly growing industry in New Hampshire with ~20 ha (50 ac) currently (2015) licensed for oyster aquaculture, but the potential areal extent is limited by a variety of constraints and has been estimated to probably not exceed 80 ha (Grizzle & Ward 2012). Assuming full production and harvest on only half the licensed bottom area (40 ha) per year, the probable maximum N removal via harvest would be ~5600 kg. In comparison, the annual total N loadings to the Great Bay system have been estimated at ~1 250 000 kg (PREP 2013). Thus, 40 ha of bottom area in maximum production would remove <1% of the annual total N loadings to the Great Bay system. Although this analysis indicates oyster farming may not have a major role to play in estuary-wide management of N, we suggest that oyster farms could be quite important in this respect in Little Bay, that portion of the estuary where they occur (see Fig. 1). Moreover, and as noted above, oysters are involved in nutrient dynamics in several ways, and oyster harvest only involves removal of assimilated nutrients. Thus, it makes sense for managers to consider the full complexity of how oysters affect N movements in the ecosystem. Although our understanding of how processes such as oyster excretion and biodeposition affect overall N movements remains incomplete in many respects, N bioextraction via oyster harvest only represents a portion of the total N removal from the ecosystem (see reviews by Kellogg *et al.* 2014; Rose *et al.* 2014).

Conclusions and future studies

Our study increases the geographical range of research on the nutrient content of farmed oysters to northern New England. Data are now available from Alabama in the Gulf of Mexico to New Hampshire. Focusing on N content of farmed oysters, the following conclusions seem reasonable based on research to date.

- Considering all studies to date, the N content of farmed oysters may be expected to vary widely around the average values, and is likely affected by environmental conditions, oyster age/size, and physiological changes in the oyster that occur seasonally.
- Based on studies in three geographical areas, the average %N content (relative to DW) of regular harvest-size (~85 mm shell height) oysters was 0.22% of the shell and 7.8% of soft tissue.
- Based on the same three studies, total N ranged from 0.14 to 0.32 g per oyster of regular harvest size (~85 mm shell height), due mainly to differences in shell and soft tissue DW.
- Based on two studies, %N content had a negative correlation with oyster size
- Based mainly on this study, %N content was highest in oysters grown in waterthat likely had the highest average dissolved inorganic N concentrations.

These findings suggest several areas for future research. More studies are needed on how oyster nutrient content is affected by ambient environmental conditions, different farming methods, different sizes of oysters, and oysters harvested at different times of the year. For some uses, mean values based on our current understanding may be sufficient. However, the reported range of 0.14–0.32 g of total N content for oysters of very similar shell size (but widely different weight) strongly indicates that better understanding is needed on the variety of factors controlling oyster nutrient content.

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