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The role of shellfish aquaculture in reduction of eutrophication in an urban estuary

Suzanne B. Bricker^{1*}, Joao Gomes Ferreira², Changbo Zhu^{2,11}, Julie M. Rose³, Eve Galimany³, Gary Wikfors³, Camille Saurel², Robin Landeck Miller⁴, James Wands⁴, Philip Trowbridge^{5,12}, Raymond Grizzle⁶, Katharine Wellman⁷, Robert Rheault⁸, Jacob Steinberg^{1,13}, Annie Jacob¹, Erik D. Davenport¹, Suzanne Ayvazian⁹, Marnita Chintala⁹, and Mark A. Tedesco¹⁰

^{1*} National Centers for Coastal Ocean Science, Silver Spring, MD 20910, corresponding author Suzanne.bricker@noaa.gov (email), +240-533-0347 (ph), +301-713-4388 (fax)

² Environmental Engineering, Faculty of Sciences and Technology, Universidade Nova de Lisboa, Portugal

³ NOAA Fisheries NEFSC Milford Laboratory, 212 Rogers Avenue, Milford, CT 06460

⁴ HDR, Inc., 1 International Boulevard, 10th Floor, Suite 1000, Mahwah, NJ 07495-0027

⁵ New Hampshire Dept. of Environmental Services, Durham, NH 03824

⁶ Department of Biological Sciences, Jackson Estuarine Laboratory, 85 Adams Point Road, Durham, NH 03824

⁷ Northern Economics, Inc., 1455 NW Leary Way, Suite 400, Seattle WA 98107

⁸ East Coast Shellfish Growers Association, 1121 Mooresfield Rd., Wakefield, RI 02879

⁹ EPA, ORD, National Health and Environmental Effects Research Laboratory, Atlantic Ecology Division, 27 Tarzwell Dr., Narragansett, RI 02882

¹⁰ EPA Long Island Sound Office, Government Center, Suite 9-11, 888 Washington Blvd., Stamford, CT, 06904-2152

¹¹ *present address* South China Sea Fisheries Research Institute, CAFS, 231 Xin Gang Xi Road, Guangzhou 510300, China

¹² *present address* San Francisco Environmental Institute, 4911 Central Ave, Richmond, CA 94804

¹³ *present address* School of Oceanography, University of Washington, 7616 Latona Ave NE, Seattle, WA 98115

Supporting information

Additional information on acronymns, abbreviations, databases, eutrophication assessment and oyster model methodological details, validation of models and detailed model production results referenced in text.

41 **Abstract**

42 Land-based management has reduced nutrient discharges, however, many coastal waterbodies
43 remain impaired. Oyster ‘bioextraction’ of nutrients and how oyster aquaculture might
44 complement existing management measures in urban estuaries was examined in Long Island
45 Sound, Connecticut. Eutrophication status, nutrient removal, and ecosystem service values were
46 estimated using eutrophication, circulation, local- and ecosystem-scale models, and an avoided
47 costs valuation. System-scale modeling estimated that 1.31% and 2.68% of incoming nutrients
48 could be removed by current and expanded production, respectively. Up-scaled local-scale
49 results were similar to system-scale results, suggesting this upscaling method could be useful in
50 waterbodies without circulation models. The value of removed nitrogen was estimated using
51 alternative management costs (e.g. wastewater treatment) as representative, showing ecosystem
52 service values of \$8.5 and \$470 million per year for current and maximum expanded production,
53 respectively. These estimates are conservative; removal by clams in Connecticut, oysters and
54 clams in New York, and denitrification are not included. Optimistically, calculation of oyster-
55 associated removal from all leases in both states (5% of bottom area) plus denitrification losses
56 showed increases to 10% – 30% of annual inputs, which would be higher if clams were
57 included. Results are specific to Long Island Sound but the approach is transferable to other
58 urban estuaries.

59 **Introduction**

60 Eutrophication is among the most serious threats to the function and services supported by
61 coastal ecosystems.^{1,2} Waterbodies worldwide have experienced nutrient-related degradation^{3,4}
62 including excessive algal blooms, hypoxia⁵, and loss of seagrass habitat⁶ that can have cascading
63 negative effects on fisheries.^{7,8,9} In the United States (U.S.; Table S1 for acronyms), 65% of
64 estuaries and coastal bays are moderately to severely degraded by nutrients from agricultural and
65 urban runoff, atmospheric deposition and wastewater treatment plant (WWTP) discharge.¹⁰ U.S.
66 and European legislation aimed at mitigating eutrophication is focused mainly on reductions of
67 land-based discharges.^{11,12} Practical limits on existing point and nonpoint source controls suggest
68 that additional innovative nutrient management measures are needed.¹³

69

70 The use of shellfish cultivation for nutrient remediation, called ‘bioextraction,’ has been
71 proposed in the U.S. and Europe.^{14, 15, 16, 17} Research investigating shellfish related nutrient
72 removal is consistent with U.S. policies promoting shellfish aquaculture and ecosystem service
73 valuation.^{18, 19} Removal of phytoplankton and particulate organic matter (POM) from the water
74 column by shellfish filtration short-circuits organic degradation by bacteria and consequent
75 depletion of dissolved oxygen (DO) which can lead to death of fish and benthic organisms and
76 losses of fish habitat. Nutrients are sequestered into tissue and shell, and shellfish may enhance
77 denitrification and burial.^{21, 22, 23, 24} Local, state, and federal agencies have been exploring the use
78 of shellfish aquaculture as a nutrient management measure in the Northeastern U.S.^{15, 25, 26}
79 Recent research has shown that the costs and removal efficiencies of nitrogen (N) through
80 shellfish cultivation compare favorably with approved Best Management Practices (BMPs).^{13, 15}
81
82 Nutrient credit trading has been proposed, and in some states implemented, as a tool to achieve
83 water quality goals.^{27, 28} These programs establish a market-based approach to provide economic
84 incentives for achieving nutrient load reductions to meet pollution reduction targets. They could
85 create new revenue opportunities for farmers, entrepreneurs, and others who are able to reduce
86 discharges below allocated levels at low cost and sell credits received to dischargers facing
87 higher-cost reduction options. A credit is the difference between the allowed nutrient discharge
88 and the measured nutrient discharge from a nutrient source (e.g. wastewater treatment plant).
89 Credits must be certified by a regulatory agency such as the Connecticut Department of Energy
90 and Environmental Protection before inclusion in a credit trading program such as the
91 Connecticut Nitrogen Credit Exchange. The Connecticut Nitrogen Credit Exchange (CT NCE)
92 was created in 2002 to improve nutrient-related hypoxia conditions in Long Island Sound (LIS),
93 providing an alternative compliance mechanism for 79 wastewater treatment plants (WWTPs)
94 throughout the state. During 2002-2009, 15.5×10^6 N credits were exchanged at a value of \$46
95 million, with estimated cost savings of \$300-400 million.²⁹ The CT NCE trading between point
96 sources is active and successful but the program does not yet include non-point sources.
97
98 The inclusion of shellfish bioextraction in non-point nutrient credit trading programs has been
99 proposed.^{13, 15, 16} This study examined the role of shellfish bioextraction in the control of

100 eutrophication symptoms and the ecosystem service value of nutrient removal using an
101 integrated modeling framework at system- and farm-scales in CT waters of LIS, an urban
102 estuary. This study is an example of the potential use of shellfish aquaculture to supplement
103 nutrient management in urban estuaries, which often require additional nutrient reductions and
104 also support shellfish populations. LIS is a good representative of urban estuaries because the
105 high-level eutrophication impacts are well known and is among the 65% of U.S. estuaries with
106 Moderate to High level eutrophication.¹ LIS has higher nitrogen (N) loads and chlorophyll a
107 (Chl), and lower dissolved oxygen (DO) concentrations than the median of U.S. estuaries (Table
108 S3).¹ It is also representative of urban estuaries in the European Union, which have these same
109 characteristics.^{30, 31} While results are specific to LIS, the approach is transferable and thus
110 relevant to other estuaries where nutrient reductions are required. The focus was nitrogen (N)
111 because it is typically the limiting nutrient in estuarine waterbodies.³² While there are thriving
112 industries of both oysters (*Crassostrea virginica*) and clams (*Mercenaria mercenaria*) in LIS, the
113 focus of the study was oysters because they are the main shellfish being farmed in LIS. An
114 individual growth model was developed for oysters and was integrated into the local- and
115 ecosystem-scale models. While clams are also a productive cultivated species in LIS, it was not
116 possible to develop an individual growth model due to time and resource limitations and thus
117 only gross N removal by clams could be estimated. This was an additional reason that clam
118 results were not included in the analysis. Denitrification was also not included in the model
119 because the model focus is the oyster and denitrification is a downstream process, and
120 additionally because of the high variability among published rates.^{24, 33} Project goals were to: (1)
121 determine the mass of N removed through oyster cultivation at current and expanded production;
122 (2) assess how significant oyster related N removal is in relation to total N loading under current
123 and expanded production scenarios; (3) estimate the economic value of this ecosystem service;
124 and (4) evaluate whether oyster related N removal may be significant enough to support a role
125 for shellfish growers in a nutrient credit trading program, taking into account the present
126 situation and potential expansion of aquaculture.

127 **Materials and Methods**

128 **Study site and cultivation practices**

129 Long Island Sound (LIS; Figure 1) is a large estuary (3,259 km²) with an average depth of 20 m,
130 shared by the states of CT and New York (NY). The waterbody has historically received large
131 nutrient loads from its highly developed, intensely populated (8.93 x 10⁶ people in 2010)
132 watershed. The N load to LIS is estimated to be 50 x 10³ metric tons y⁻¹ with point sources
133 accounting for 75% and the remaining 25% attributed to non-point sources.³⁴ Summer thermal
134 stratification and a residence time of 2-3 months^{10, 35} combined with N loads have resulted in
135 notable water quality degradation including areas of regular summertime hypoxia³⁶ and loss of
136 seagrass habitat.³⁷ The Assessment of Estuarine Trophic Status (ASSETS) model was applied to
137 monitoring data (Table S2) to update the eutrophication status of LIS (Figures S1, S2).^{10, 38, 39}
138 Eutrophication condition improved from High to Moderate High since the early 1990s.¹
139 Improvements resulted from increased bottom water DO concentrations reflecting load
140 reductions from 60.7 x 10³ to 50.0 x 10³ metric tons y⁻¹.⁴⁰ However, Chl concentrations did not
141 change, receiving a rating of High in both timeframes. As nitrogen loads continue to decrease,
142 further improvements are expected, but may be counterbalanced by increasing population.

143
144 Hypoxia was used in the 2000 Total Maximum Daily Load analysis (TMDL) to guide
145 development of a plan for 58.5% N load reduction (by 2017) to fulfill water quality objectives
146 (NYSDEC and CTDEP, 2000). Implementation of the TMDL resulted in >40% reductions in N
147 loads by 2012, 83% of final reduction goals, primarily through WWTP upgrades to biological
148 nutrient removal.³⁶ Atmospheric and agricultural loads also decreased.³⁶ While water quality
149 improvements have been documented, they have been slow and masked by weather-driven
150 variability and continued population growth.⁴¹ The TMDL analysis concluded that full
151 attainment of desired water quality standards would require additional reductions or increased
152 assimilative capacity. The updated eutrophication assessment results confirm this conclusion.
153 The TMDL identified alternative management methods, such as bioextraction, as potential
154 measures to help achieve DO standards. The well-established Eastern oyster (*Crassostrea*
155 *virginica*; hereafter 'oyster') industry makes LIS a compelling site to test the potential for N
156 removal through cultivation and harvest and a useful example for other urban estuaries that

157 support oyster growth. Recent CT shellfish harvests have provided over 300 jobs and \$30 million
158 in farmgate revenue (where farmgate price is the sale price of oysters that is received by the
159 grower) annually, with oyster harvest exceeding 40×10^6 oysters.⁴²

160

161 Oyster cultivation practices in LIS typically involve collection of one- and two-inch oyster seed
162 from restricted areas and relay, or replanting, for on-bottom growout for one to two years in
163 approved areas. We have used seed of less than one inch in our model simulations to include
164 nutrient removal by early lifestages. Seed planting densities of 62 oysters m^{-2} are reduced by an
165 estimated 55% mortality. Stocking area does not typically include the entire farm area, rather
166 planting occurs on a rotational basis on 1/3 of the farm annually within the three-year culture
167 cycle.⁴³ Oyster seed planting takes place over 90 days beginning on May 1. Harvest occurs
168 throughout the year and a farmgate price of \$0.40 per oyster is used to calculate harvest value.
169 Growers have not reported harvest since 2008, so previous harvests were used to estimate
170 landings based on interviews with growers and managers. Growers did not specify what
171 proportion of the current > 61,200 acres of lease area is being used for cultivation. Thus, a
172 bracketing approach was used to capture the range of possible areas being cultivated within LIS
173 where the mid-range estimate (5,250 acres [21 km^2]) was used to represent current production
174 area and was used as the standard model scenario. The total potential area that could be
175 cultivated (11,116 acres [45 km^2]) was determined as one half of all suitable area (e.g. all areas
176 that support oyster growth and are not classified as prohibited for legal or contaminant reasons)
177 within the 12 m (40 ft) bottom contour. The spatial distribution of production was estimated by
178 superimposing known harvests from different locations onto model grid boxes. Culture practices
179 and monthly monitoring data (temperature, salinity, total particulate matter [TPM], POM, Chl,
180 DO) from 17 stations in the LIS Water Quality Monitoring Program were used to support model
181 applications (Table S2, Figure S1).⁴⁴ Five years of data (2008-2012) were used to provide a
182 robust dataset and to reduce bias due to anomalous weather years. Jonckheere-Terpstra (JT)
183 tests⁴⁵ were applied using a standard α - level of 0.05 indicating no trends in any variable at any
184 station. Other data (e.g. macroalgal abundance, occurrences of nuisance and toxic blooms) were
185 acquired from the LIS Study (LISS).⁴⁶ Additional methodological and analytical details are
186 available.³⁹

187

188 **Modeling Framework**189 ***System-scale aquaculture model***

190 The first step toward modeling aquaculture at the system-scale involved coupling of two models:

191 1) a high resolution 3-D coupled hydrodynamic-eutrophication-sediment nutrient flux model

192 (System Wide Eutrophication Model [SWEM]) that operates on the timescale of one year, and 2)

193 the broader scale EcoWin.NET (EWN)⁴⁷ ecological model that operates on a decadal timescale.

194 The resulting model framework was used to simulate aquaculture practices and to support

195 economic analyses, both which require timescales greater than one year. SWEM was used to

196 describe the main features of annual water circulation and nutrient loading to LIS by means of

197 2300 grid cells divided into 10 vertical (σ) layers. The hydrodynamic model solves a system of

198 differential, predictive equations describing conservation of mass, momentum, heat and salt, but

199 does not include shellfish. SWEM was calibrated to data collected during 1994-95 and validated

200 for data collected in 1988-89.^{48, 49, 50, 51, 52} The SWEM grid layout was superimposed on a two-

201 vertical-layer set of 21 larger boxes that were used for the EWN system-scale ecological

202 modeling. The EWN model includes oysters, simulated by a population model based on the

203 individual growth model developed for *Crassostrea virginica* which was integrated into the

204 ecosystem (and local scale) model (Figures S3 and S4). The larger EWN boxes, and the

205 simplified two-layer vertical formulation, were defined through consultation with the project

206 team and local stakeholders. The resulting EWN framework took into consideration state

207 boundaries, physical data and locations of aquaculture leases, and followed a well-established

208 methodology for merging the two types of models⁵³ Water flows across the EWN model grid

209 box boundaries were calculated from SWEM to obtain as accurate a representation of the

210 circulation pattern as possible, using a four-stage process: (i) a Geographic Information System

211 (GIS) representation of EWN boxes was used to map these to the SWEM grid; (ii) SWEM flow

212 outputs for a one-year model run were integrated to provide hourly fluxes across the EWN box

213 boundaries (horizontal and vertical); (iii) external inputs at the land boundaries (rivers, WWTPs)

214 were added to the list of flows; (iv) the final data file was checked for volume conservation.

215 The use of an annual run from a detailed hydrodynamic model, which is the general approach for

216 upscaling hydrodynamics in EWN, captures all the relevant physical signals, i.e. the freshwater

217 component determined by the annual hydrological cycle, and the tidal current component,
218 including the high tide/low tide semi-diurnal cycle in LIS, and the spring-neap tide signal.
219 An annual cycle will never provide complete volume conservation, because the tidal state at the
220 end of the run will not be an exact match to that in the beginning, EWN makes a small volume
221 adjustment (in this case, the average per hourly timestep is 0.00016%), based on the deviation
222 from the closure condition, to allow mass balance conservation in multi-annual runs.

223
224 The 42 box EWN model grid was used to simulate system-scale oyster production, and
225 associated drawdown of Chl, POM, and N using relevant transport, biogeochemistry, and
226 shellfish model components. The EWN oyster aquaculture model combined hydrodynamic
227 outputs from SWEM, as described above, with external nutrient loads that represented the level
228 of loads expected once the 2000 TMDL N and carbon load reductions had been fully
229 implemented.^{54, 55} Note that these are model predicted future values for 2017, not measured
230 values. Oyster populations in EWN are modeled using standard population dynamics equations
231 driven by individual growth and mortality (Figure S4)⁵³, using 20 heterogeneous weight classes
232 spanning 0-100 g live weight. EWN explicitly simulates seeding and harvest, defined from
233 expert knowledge of local growers. Seeding takes place annually from Year 1, with first harvest
234 in Year 3. Harvest is regulated by the availability of market-sized animals and market demand.
235 EWN was calibrated and validated for a standard area of oyster farms within each of the six
236 boxes that contain aquaculture (Figure 1) using a standard stocking density. There were three
237 other scenarios simulated for sensitivity testing using the areas described above, but only the
238 standard (5,250 acre [21 km²]) and potential expanded aquaculture (11,116 acre [45 km²]) area
239 scenarios are discussed here due to space considerations.

240
241 EWN model results for non-conservative water quality state variables (dissolved nutrients and
242 phytoplankton) were validated against SWEM results. In the version of SWEM used for
243 comparison, bivalves were included by considering a constant biomass of 2.8 g DW m⁻² over all
244 of LIS.⁵⁰ Note that the SWEM results are projected representations of full implementation of the
245 2000 TMDL with 1988 – 1989 hydrodynamic conditions without bioextractive technologies;
246 they are not measured data. The two models showed similar concentration ranges and annual

247 patterns for dissolved inorganic nitrogen (DIN) which was encouraging given that no model
248 coefficients (e.g. half-saturation constants or primary production rates) were shared between the
249 two models (Figure S5). The use of unique modeling coefficients was intentional, because the
250 two models are different in scale, formulations, and number of state variables. Comparisons
251 between both models for Chl concentrations also showed a match for ranges and spatial
252 distribution represented by model curves in the eastern part of LIS, but in central LIS the EWN
253 results did not reproduce the drop in concentration observed in SWEM for the latter part of the
254 year, and values in western LIS remained elevated in the EWN simulation for most of the year
255 (Figure S6). We accepted this deviation because the main nutrient loading is at the western end
256 of LIS, thus it seemed inconsistent to arrive at lower simulated concentrations of Chl in the
257 western Sound. Measured data confirm that Chl concentrations are higher in the western
258 Sound.⁵⁶ Higher Chl concentrations might occur in eastern LIS if it was a fast-flushing system
259 that would transport phytoplankton blooms from the west to the east, but residence times of 2-3
260 months estimated by EWN and other studies^{10, 35} suggest that this is not the case. More likely, the
261 overestimate of Chl is the result of the absence of zooplankton grazers in the model which are
262 estimated to reduce primary production by up to 50% throughout the year.⁴⁰

263

264 ***Local-scale aquaculture model***

265 Local-scale oyster production and N removal was estimated by application of the Farm
266 Aquaculture Resource Management (FARM) model which includes the oyster growth model
267 developed for LIS and used in EWN.^{57, 58, 59} Results were up-scaled to provide system-scale
268 estimates to compare to EWN model results. FARM takes into account food conditions inside a
269 farm, shellfish ecophysiological characteristics, and farming practices. Potential nutrient removal
270 by the farms was estimated and compared to results from EWN simulations. The system scale
271 EWN model differs from the local scale FARM model in that FARM does not have: a)
272 harvesting, but harvestable biomass is estimated, b) overlapping shellfish year-class populations,
273 c) multiple species of shellfish, or d) system-scale feedbacks.⁵³

274

275 A three-year culture cycle was simulated using data from one long-term monitoring station
276 located within each of the four LIS zones (Figure S1), and the same inputs (e.g. seeding density,

277 mortality, etc.) as were used for the EWN simulations. Nutrient removal was determined for each
278 simulated farm. Results were up-scaled in an approach developed previously for Potomac
279 River⁶⁰ to evaluate total area-weighted current and potential removal using the same standard
280 and expanded cultivation areas used by EWN. Additional assumptions were used for upscaling:
281 i) there were no additional reasons that identified bottom area could not be cultivated, ii) all lease
282 areas within a zone had the same oyster growth and N removal rates despite potential differences
283 in water quality among farm locations, iii) and there was no interaction among adjacent farms,
284 i.e., food depletion.

285

286 ***Ecosystem service valuation***

287 An intriguing aspect of the bioextraction discussion is the potential economic value of the water
288 filtering ecosystem service provided by oysters and whether growers should be paid for the
289 oyster related N removal capacity within a nutrient credit trading program. We used the cost
290 avoided, or replacement cost method, to estimate the value of N removal by oysters.⁶¹ This
291 method assumes that the costs of restoring a part of the ecosystem — in this case, clean water —
292 through N removal by wastewater, agricultural and urban BMPs, provides a useful estimate of
293 the value of the ecosystem service of N removal by oyster bioextraction. The use of the
294 replacement cost method assumes that if oysters are no longer harvested, the N removal services
295 they have been providing would need to be replaced. At present, WWTP upgrades, and
296 agricultural and urban BMPs are the most likely candidates to replace the service that the oysters
297 provide.

298

299 The value of shellfish aquaculture as a N removal device is estimated by taking the difference in
300 minimum total costs for nitrogen reduction targets in the watershed with and without the
301 inclusion of shellfish farms.⁶² In this case, the value of shellfish aquaculture production is
302 determined not only by its marginal cost in relation to other abatement measures (e.g., WWTPs),
303 but also by its cleaning capacity. Marginal costs increase rapidly with higher N reduction levels
304 due to the higher implementation costs of abatement measures required to meet reduction targets.
305 In the case of LIS, where aquaculture operations already exist and the costs of production are a
306 given (and are offset by oyster sales by the farmers), the value of the removed N is equal to the

307 minimum total cost without shellfish production (or the costs of WWTPs, agricultural and/or
308 urban BMPs that include wet ponds and submerged gravel wetlands).

309

310 Costs used in this analysis were estimated for incremental upgrades of N reduction from current
311 wastewater effluent concentration levels to 8 mg L⁻¹, from 8 to 5 mg L⁻¹, and from 5 to 3 mg L⁻¹
312 using an approach developed in Chesapeake Bay.^{63, 64} Total capital costs, annual operating and
313 maintenance costs, and the combined annualized capital cost (20 year depreciation) associated
314 with plants of different sizes were used to determine average cost per kilogram (2.2 pound) of N
315 removed. These were adjusted to 2013 dollars with the Engineering News-Record Construction
316 Cost index (ENRCC) to account for inflation.⁶⁴ Average annual costs of the N removal by the
317 three treatment levels were \$32.19 kg⁻¹ (\$14.63 lb⁻¹; 8 mg L⁻¹), \$37.00 kg⁻¹ (\$16.82 lb⁻¹; 5 mg L⁻¹),
318 and \$98.58 kg⁻¹ (\$44.81 lb⁻¹; 3 mg L⁻¹, Table 1).

319

320 The estimated average annual cost for agricultural controls including riparian buffers and cover
321 crops, adjusted for inflation using the ENRCC Index, was \$38.92 acre⁻¹. Use of such controls
322 was estimated to result in a maximum N load reduction of 0.59 x 10⁶ kg yr⁻¹ (1.31 x 10⁶ lb yr⁻¹)
323 for the entire CT River Basin and an estimated adjusted annual cost of \$7.68 million.⁶⁴ Given a
324 current estimated agricultural N load of 1.76 x 10⁶ kg yr⁻¹ (3.89 x 10⁶ lb yr⁻¹), the maximum
325 potential reduction would be 34.1% at a unit cost of about \$12.98 kg⁻¹ yr⁻¹ (\$5.90 lb⁻¹ yr⁻¹).

326

327 The two most cost-effective urban BMPs are wet ponds and submerged gravel wetlands with
328 average construction costs of \$7,000 and \$11,000 acre⁻¹ drained, respectively, resulting in N
329 removal of 55% and 85%, respectively.⁶⁴ Total costs including construction costs and the cost of
330 land acquisition for full implementation within all of the sub-basins were estimated to be \$3,262
331 million in 2013 dollars. The total cost was divided by 20-year amortization period to derive an
332 estimated annual cost of \$163 million. The maximum N reduction that might be obtained in the
333 CT River Basin was estimated as 0.47 x 10⁶ kg yr⁻¹ (1.04 x 10⁶ lb yr⁻¹) with an annual per unit
334 cost of \$349 kg⁻¹ yr⁻¹ (\$159 lb⁻¹ yr⁻¹, Table 1).

335

336 **Results and Discussion**

337 **System-scale oyster aquaculture bioextraction**

338 Output for the 10-year standard (5,250 acres) EWN model simulation shows a spin-up period in
339 the first 4 years, followed by a stable cycle with alternating years of higher and lower harvest
340 (the fluctuations result from slight variations in water volumes in consecutive years; Figure S7).
341 The reason for the variability is that EWN uses water flux outputs from SWEM superimposed on
342 a 365-day cycle (see *System-scale aquaculture model* in the Methods section). These are due to
343 natural year-to-year fluctuations which this modeling scheme does not consider and which
344 cannot be forecast by any model due to limitations in predicting weather patterns.

345
346 Year 9 of the EWN standard model run, after stabilization of the model, was chosen for a mass-
347 balance analysis of oyster cultivation and estimation of nutrient removal (Table 2; Figure S8).
348 The calculation of N removal and other eutrophication-related ecosystem services (e.g. Chl and
349 POM drawdown,) integrates physiological growth processes of the: (i) Year 3 cohort, much of
350 which will be physically removed (harvested) from the Sound; (ii) Year 2 cohort, which will be
351 harvestable only the following year; and (iii) Year 1 cohort, which will take an additional two
352 years to reach harvestable size. Nutrient removal is based on the filtration rate of the oysters
353 based on the outputs of the AquaShell individual growth model, calibrated and validated using
354 experimental data from this and other studies (Figure S3). EcoWin (and FARM) calculate the
355 total, or gross, phytoplankton and detrital carbon filtered by the oysters and then convert those
356 values to N. The net removal of N from the water is represented as the total N removed minus N
357 returned to the water as pseudofaeces, faeces, excretion, mortality, and spawning (Figure S8).
358 The model works internally in carbon units, and from those outputs other terms are calculated
359 (Table 2). The focus is N because carbon is not a limiting nutrient and thus poses no direct
360 concern for eutrophication.

361
362 Overall, the standard model suggests that the combined volume of the boxes under cultivation is
363 filtered by oysters roughly once per year (0.95 y^{-1}), though there is greater filtration in some
364 boxes (e.g. Box 25; Table 2). The total filtered volume of boxes that include aquaculture
365 corresponds to an annual clearance of $9 \times 10^9 \text{ m}^3$ of water (more than 300×10^9 cubic feet). Note

366 that the clearance rate and mass balance outputs represent the role of all cultivated shellfish, as
367 opposed to only the harvested biomass. The N removal role of oysters is typically estimated by
368 applying a conversion factor (usually about 1%)⁶⁵ to the harvested biomass.

369
370 Current cultivation results in an estimated annual harvest of 31×10^3 metric tons of oysters and
371 removal of more than 650 metric tons of N (Table 2; Figure S8), the equivalent of 1.3% of total
372 annual inputs. This removal estimate represents an ecosystem service corresponding to about
373 200,000 Population Equivalents (PEQ) considering a per-person annual load of 3.3 kg N y^{-1} . The
374 N removed, compared to total harvested biomass of oysters in the six model boxes that include
375 shellfish is 2-3.5%, with an aggregate value of 2.1% (Table 2). This is double the usual reported
376 value of 1% by weight that includes only harvested biomass, reflecting inclusion of the whole
377 population (see [17] for details). These results suggest some areas perform better than others in
378 terms of N removal per unit area, under identical conditions of seeding density. The area-
379 weighted average removal estimated by the model is $125 \text{ kg N acre}^{-1} \text{ y}^{-1}$ ($275 \text{ lb N acre}^{-1} \text{ y}^{-1}$). By
380 comparison, a calculation based on final oyster stocking density at harvest of $30 \text{ individuals m}^{-2}$,
381 an individual harvestable fresh weight oyster of 91 g, and an N content of 1% of total fresh
382 weight gives $110 \text{ kg N acre}^{-1}$ ($243 \text{ lb N acre}^{-1}$), or $41.3 \text{ kg N acre}^{-1} \text{ y}^{-1}$ ($91 \text{ lb N acre}^{-1} \text{ y}^{-1}$). The
383 higher value obtained by the EWN model is consistent with the alternative approach that
384 considers removal of N from the water column by all shellfish, not just those that are harvested.

385
386 The EWN model outputs for standard (discussed above) and potential scenarios estimate that the
387 ratio of annual water clearance to aggregate volume increases from 0.95 to 2.08, meaning that
388 oysters filter the total volume of the cultivated boxes up to twice per year in the expanded
389 production scenario compared to less than one time per year in the standard scenario. The
390 percent reduction of Chl through filtration increases from an average of 1.3% to 2.1% from
391 current to expanded production, with Box 25 showing the greatest removals at 2.8% and 5.3% in
392 current and maximum production, respectively. The EWN model outputs for the potential
393 scenario estimate that harvest, and net N removal and PEQs would also double to $64 \times 10^3 \text{ y}^{-1}$,
394 1,340 metric tons N y^{-1} , and over 400,000 PEQ, respectively (Table 2). The N removal represents
395 about 2.68% of total annual inputs. Nitrogen removal per acre, except in Box 41, decreases as the

396 stocked area increases, probably due to the shift in population distribution with more oysters in
397 the lower weight classes. There is only a small effect on food depletion at the higher density.
398 This smaller effect was reflected in the Average Physical Product (APP), the harvested biomass
399 divided by total seed weight, which decreases by 1.63 between the standard and potential
400 scenarios for the aggregate set of cultivated boxes (Table 2). The APP does not fall below 45 (i.e.
401 1 kg of seed yields 45 kg of product), which makes cultivation financially attractive even for the
402 largest stocking area scenario. These results suggest that even at potential expanded production
403 (11,116 acres in cultivation), ecological balance is maintained or improved—Chl is lower, but
404 oyster production appears to remain in the Stage I section of the carrying capacity curve, where
405 Marginal Physical Product (MPP) is greater than APP, suggesting capacity for additional seeding
406 density.⁵⁷

407 **Shellfish carrying capacity of Long Island Sound**

408 There is great interest in expanding aquaculture for greater N removal and to increase domestic
409 production of seafood.^{18, 19, 20} An important consideration is whether there is capacity to increase
410 production without causing detrimental impacts to the environment. We have used the EWN
411 model to assess whether LIS is at carrying capacity following the overall definition⁶⁶ and
412 focusing on production and ecological categories.⁶⁷ When evaluating the potential for increased
413 bioextraction, the carrying capacity at a system perspective should be considered first and after
414 that a local-scale model should be applied at selected sites. The reverse approach does not take
415 into account the interactions among aquaculture farms in an expansion scenario, which are
416 particularly important for organically extractive aquaculture.⁶⁹ The EWN results explicitly
417 account for those interactions. Executed at an ecosystem scale, the shellfish stock (per EWN
418 box) is uniformly distributed in relatively large model cells, consequently results obtained here
419 are less constrained (from a food depletion perspective) than those obtained with a farm-scale
420 analysis for a particular box.⁵³ On the other hand, EWN takes multiple culture cycles into
421 account while the local-scale FARM model does not (see *Farmscale oyster aquaculture*
422 *bioextraction*), which to some extent reduces the disparity between the approaches.

423
424 Overall, the standard EWN model suggests that the combined volume of boxes under cultivation
425 is filtered by oysters about once a year (Table 2). At the scale of LIS, the shellfish simulated in

426 the standard model would take over 8 years to clear the total water volume ($77.4 \times 10^9 \text{ m}^3$). This
427 estimate for water clearance is greater than the *overall* residence time for LIS estimated to be on
428 the order of months^{10, 35} and similar to the e-folding time of 2-3 months estimated in EWN by
429 means of Lagrangian tracers.³⁹ Even at the highest potential cultivation scenario, the system is
430 below carrying capacity. Thus, from a food depletion perspective, there appears to be potential
431 for expanded cultivation and increased oyster bioextraction, the challenge being more related to
432 social license aspects and reduction of conflicts with competing water uses such as recreation.
433 The modeling framework developed in this project is appropriate for testing different
434 management strategies, however, results would be more complete if EWN included bioextractive
435 nutrient removal by clams, and other autochthonous benthic filter-feeders that compete for the
436 same food resource.⁶⁸ As noted previously, it was not possible to develop a growth model for
437 clams (or other filter feeders), thus we were unable to include estimates of their N removal
438 capacity in the model.

439
440 We have extended the analysis and used the EWN model to perform a marginal analysis to
441 indicate potential for increasing production by means of an optimization analysis. The analysis
442 considers different stocking densities (S; here we use increased lease areas with the same
443 stocking density) for various boxes (B), and would require S^B model runs.⁷⁰ The number of
444 required model runs rapidly reaches a limit in terms of computational time, thus the best way to
445 optimize this analysis would be to produce a family of curves and use Monte Carlo methods for
446 optimization. We have not conducted a Monte Carlo analysis due to lack of appropriate input
447 from the management and grower communities, but have done a marginal analysis for one box
448 (Box 25; Figure 1) to highlight what changes in seeding that box might do to harvest within the
449 other boxes. Note that management decisions can be well informed by models of this type but
450 policies that affect the livelihoods of shellfish farmers should be fully participative as there is a
451 strong element of social choice that must be enacted.

452
453 The marginal analysis showed that changes in seeding of Box 25, which contains the majority of
454 leases (Table 2), results in changes to harvest in all boxes (Figure S9, Table S5). The changes in
455 harvest are a typical representation of the law of diminishing returns, such as presented for

456 FARM and EWN.^{53, 57} The seeding density currently used in the standard model is low compared
457 to other oyster cultivation operations throughout the world⁷¹, and the carrying capacity
458 calculation above showed that stock could be increased. The marginal curve shows increased
459 harvest (Total Physical Product [TPP] which is harvest) with increased seed but starts to flatten
460 with an annual stocking in Box 25 of 20.0×10^3 metric ton seed. The optimum profit point,
461 considering $P_i = P_o$, is roughly at the maximum production level (where P_i is price of seed, P_o is
462 harvest value). There are not enough data on industry costs or revenue to extend this analysis,
463 however, note that that the more P_i is in excess of P_o , the greater will be the MPP for the
464 optimum profit point. This will shift the stocking density for profit maximization to the left of
465 the production curve. The simulated increase in stocking density in Box 25 caused decreased
466 harvest in all other boxes. In all boxes except Box 23, directly west of Box 25, harvests
467 decreased at all seeding levels. Harvest in Box 23 increased in early stages of increased stocking
468 before decreasing, which is likely linked to additional subsidy of particulate organics from
469 increased cultivation in Box 25 (Table S5). Maximum harvest reduction (32%) is seen in Box 27
470 directly east of Box 25. But even Box 41, on the eastern end of LIS, showed decreases in harvest
471 with increased stocking in Box 25. Overall, the model suggests that stakeholders with
472 aquaculture farms in other boxes would be affected by an overall decreased yield of 17%. This
473 decrease in yield reinforces that decisions on expansion and redistribution of aquaculture among
474 zones should reflect a social consensus, as well as appropriate environmental and production
475 aspects.⁶⁹ This analysis also indicates that production could be increased, from a perspective of
476 ecological sustainability. With respect to the use of this kind of tool, models should support
477 decision-makers, rather than replace them.

478 **Farmscale oyster aquaculture bioextraction**

479 The FARM model estimated N removal at Station 09 in western LIS (Figure S1) of 0.105 metric
480 ton N acre⁻¹ y⁻¹, representing a population equivalent of 32 PEQ acre⁻¹ y⁻¹. Nitrogen removal and
481 harvestable biomass in the farms simulated in the Narrows, Western and Central areas were 2 -3
482 times greater than in Eastern LIS. Results showed that Chl and DO concentrations changed only
483 slightly (0.3% decrease in both) over the three-year culture cycle. The slight change suggests no
484 negative effect on water quality from the aquaculture operation and that there may be a margin
485 for increased stocking density. The local-scale simulations showed a range in N removal of 0.32

486 – 0.021 metric ton acre⁻¹ yr⁻¹, decreasing from west to east, consistent with EWN results and
487 within the range of removal rates estimated in other ecosystems.⁷¹

488
489 FARM model results provided an opportunity to compare local results to those from EWN.
490 Results were up-scaled to represent potential system-scale impacts using acreages for current
491 (5,250 acres) and potential (11,116 acres) production. Results from each station were used to
492 represent conditions of the zone in which they reside for a system-wide area-weighted total N
493 removal estimate of 549 metric tons N y⁻¹, or 1.10% of the total annual input at current
494 cultivation and 1,160 metric tons N y⁻¹, 2.32% of inputs at expanded production. The removal
495 estimate corresponds to land-based nutrient removal for 167,000 and 353,000 PEQ for current
496 and potential production, respectively. These results are within 16% of EWN results for oyster
497 related N removal and PEQs. In locations with no system-scale circulation model upscaling farm
498 level results may provide reasonable estimates for bioextraction capabilities, provided overall
499 system stocking remains low enough that farms do not significantly interact with respect to food
500 depletion.

501 **Ecosystem service valuation**

502 Annualized cost estimates for removal of one kilogram (2.2 pound) of N via WWTPs and
503 agricultural and urban BMPs were applied to the estimates of current and potential N removal
504 estimated by EWN. The annual cost to replace the removal of N through bioextraction is
505 estimated to range from \$8.5 million y⁻¹ to \$230.3 million y⁻¹ (depending on the abatement
506 technology considered) under the standard acreage scenario (Table 3). Under the potential
507 production scenario, avoided costs range from \$17.4 million to \$469 million y⁻¹. Note that these
508 costs are a proxy for the value of N removal through bioextraction. These values could be
509 considered as potential payment in a nutrient credit trading program for ecosystem services
510 provided by the oyster aquaculture production. A weighted average value per acre per year is
511 calculated for each scenario and N removal method where the lowest is for agricultural BMPs at
512 current (\$1,630 acre⁻¹ yr⁻¹) and potential production (\$1,570 acre⁻¹ yr⁻¹) and the greatest is for
513 urban BMPs at current (\$43,900 acre⁻¹ yr⁻¹) and potential expanded production (\$42,200 acre⁻¹
514 yr⁻¹). As the number of acres increases, the average value for each effluent N level decreases; in
515 the standard scenario at the 8 mg L⁻¹ level, an average value per acre per year is \$4,030 while at

516 the potential scenario under the same 8 mg^{-1} level, the value per acre decreases to \$3,880 (Table
517 3).

518

519 **Could oyster aquaculture bioextraction help nutrient management in urban** 520 **estuaries?**

521 Oyster aquaculture is a promising complement to land-based nutrient management measures in
522 LIS, an urban estuary. Model results of N removal at current and potential oyster production
523 seem small compared to total inputs (1.31% – 2.68% of total input) but per-acre removal is
524 relatively large and represents an ecosystem service that would need to be replaced by source
525 load reductions such as WWTP upgrades and enhancement of agricultural and urban BMPs.
526 Note that this model approach includes bioextractive N removal by all oysters, not just those that
527 are harvested resulting in estimates that are about double what is typically estimated, which
528 could be a useful approach for estimation of N removal by restored reefs. Per-acre bioextractive
529 removal ($0.13 \text{ metric tons acre}^{-1} \text{ y}^{-1}$) is comparable to approved BMPs and may be more cost
530 effective than some abatement alternatives.^{13, 71, 72} Based on these results it would take
531 cultivation of > 60% of the bottom area to remove the total N input to LIS, though it is unlikely
532 that such a large area would be approved for cultivation due to suitability and use conflicts.
533 However, these results show that LIS is not at carrying capacity and bioextraction could play a
534 more prominent role in N reduction strategies if cultivation area or seeding densities were
535 expanded. Consistency between the local- and system-scale model results suggests that the local-
536 scale approach could provide a reasonable estimate of bioextractive services in waterbodies that
537 lack a circulation model.

538

539 The ecosystem value of oyster mediated N removal in LIS is estimated to range from \$8.5 to
540 \$230 million y^{-1} under current production and up to \$469 million y^{-1} if production is increased.
541 The values are significant compared to CT NCE activity between 2002 and 2009 where $15.5 \times$
542 10^6 credits were traded representing \$45.9 million in economic activity. Currently, only WWTPs
543 participate in the CT NCE, which allows WWTPs around the State to share in the costs and
544 benefits of removing N from wastewater.

545

546 The concept of using oysters and other filter feeding shellfish for nutrient removal directly from
547 the water is gaining momentum. The Chesapeake Bay Program is evaluating the science
548 supporting the assignment of nutrient credits to cultivated oysters and restored oyster reefs and
549 recently approved the use of harvested oyster tissue as a nutrient reduction BMP.²⁶ The town of
550 Mashpee, MA has already begun to use oysters for nutrient reduction to address TMDL N
551 reduction requirements, targeting cultivation and harvest of 500,000 oysters to remove 50% of
552 the 5000 kg N per year required by the TMDL.⁷³ The Mashpee, MA management plan includes
553 additional clam harvest areas for the same use. Bioextraction appears to be a promising
554 management strategy in impacted waterbodies of all sizes – LIS is 3,259 km², the Mashpee River
555 complex is <5 km², the Chesapeake Bay region is >11,000 m².

556
557 Note that our calculations for LIS underestimate the total N removal capability and thus the
558 economic value of shellfish bioextraction because the model was unable to include N removal by
559 clams in CT and by clam and oyster aquaculture in >400,000 acres of shellfish lease area in NY.
560 Denitrification, which could be a significant N loss based on the range of previous estimates (648
561 lb acre⁻¹ yr⁻¹, [295 kg acre⁻¹ yr⁻¹]²³, 2.16 lb acre⁻¹ yr⁻¹, [0.98 kg acre⁻¹ yr⁻¹]³³) was also not
562 included in the analysis. Using the same ratio of lease (400,000 acres) to current cultivated acres
563 in NY as for CT, we estimate that an additional 34,300 acres of cultivated oysters could be
564 removing N from LIS. Assuming the same per acre N removal rate by NY oyster aquaculture as
565 was determined for CT oyster farms, we estimate an additional 4,460 metric tons of N could be
566 removed by oysters in NY for a total removal of 5,110 metric tons per year, 10% of total annual
567 inputs to LIS. Based on the range of published areal denitrification rates and the total oyster
568 aquaculture acreage in CT and NY, denitrification losses of N could be between 38.7 and 11,700
569 metric tons N yr⁻¹. Thus, oyster sequestration into tissue and shell plus denitrification losses
570 could potentially remove as much as 16,800 metric tons N yr⁻¹, or about one third of the total N
571 input to LIS by cultivation of 5% of the bottom area of LIS. The total could be greater if N
572 removal by clams was also included.

573
574 While these optimistic results are specific to LIS, physical and biogeochemical process equations
575 and shellfish growth models used in this study are transferable, although typically the growth

576 models require recalibration to local oyster growth conditions. The physics of a system-scale
577 model must also be calculated on a case-by-case basis, since circulation is different in each
578 system, but an up-scaled local-scale model can be used in waterbodies that lack a circulation
579 model. Shellfish culture practices (including species, use of triploids, etc) also vary across
580 different systems, so transferability is not direct but the EWN and FARM models accommodate
581 most of these differences. Despite expected differences in results in different systems, even in
582 adjacent boxes in LIS there are differences, the overall result shows that bioextraction provides
583 net removal of N and is thus relevant as a potential management strategy in impacted estuaries.
584

585 The potential use of bioextraction as a nutrient management measure can complement existing
586 measures - a positive externality of commercial shellfish production shown in this study and in
587 previous work in the U.S.^{23, 24, 25, 60, 65, 73, 74}, in Europe^{14, 53, 58}, and in China.^{43, 75} While it is not
588 possible to compare the percent of incoming N removed by cultivated shellfish in these studies,
589 farm-scale modeled N removal at 14 locations in 9 countries across 4 continents and from
590 several different species of bivalves ranged from 105 - 1356 lbs acre⁻¹ yr⁻¹ (12 – 152 g m⁻² yr⁻¹)
591 with mean N removal of 520 lbs acre⁻¹ yr⁻¹ (58 g m⁻² yr⁻¹).⁷¹ By comparison, the average areal
592 removal of N by oyster aquaculture in LIS is 275 lb N acre⁻¹ yr⁻¹ (31 g m⁻² yr⁻¹; Table 2); within
593 in the range but on the lower side of reported removal rates. The ecosystem service value
594 associated with oyster related nutrient removal is also highlighted^{73, 74}. The use of bioextraction
595 as a water quality management tool is gaining support in the U.S. and elsewhere, though
596 inclusion of growers in economic nutrient credit trading programs requires further study.
597 Regardless of whether shellfish farmers become eligible for payment, they are already
598 contributing to required nutrient reductions in several U.S. jurisdictions.^{26, 73, 74} and thus could be
599 used elsewhere. The valuation of ecosystem services associated with shellfish cultivation has the
600 benefit of enhancing public awareness of water quality issues and could help shift attitudes to
601 allow increased opportunities for shellfish aquaculture, jobs creation and reduction of U.S.
602 dependency on imported shellfish aquaculture products in addition to improving water quality.
603

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614 **References**

(1) Bricker, S.B.; Longstaff, B.; Dennison, W.; Jones, A.; Boicourt, K.; Wicks, C.; Woerner, J. *Effects of Nutrient Enrichment in the Nation's Estuaries: A Decade of Change, National Estuarine Eutrophication Assessment Update*; NOAA Coastal Ocean Program Decision Analysis Series No. 26; NOAA, National Ocean Service, National Centers for Coastal Ocean Science, Silver Spring, MD; 2007.

(2) Bricker, S.; Devlin, M. Eutrophication – International comparisons of water quality challenges. *Biogeochemistry* **2011**, 105(2), 135-302.

(3) Zaldivar, J.M.; Cardoso, A.C.; Viaroli, P.; Newton, A.; de Wit, R.; Ibanez, C.; Reizopoulou, S.; Somma, F.; Razinkovas, A.; Basset, A.; Holmer, M.; Murray, N. Eutrophication in transitional waters: an overview: *Transitional Waters Monographs* **2008**, 1, 1-78.

(4) Xiao, Y.; Ferreira, J.G.; Bricker, S. B.; Nunes, J. P.; Zhu, M.; Zhang, X. Trophic assessment in Chinese coastal systems - Review of methods and application to the Changjiang (Yangtze) Estuary and Jiaozhou Bay. *Estuaries and Coasts*, **2007**, 30 (6), 901-918.

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

(6) Orth, R. J.; Moore, K. A. Distribution and abundance of submerged aquatic vegetation in Chesapeake Bay: An historical perspective. *Estuaries* **1984**, 7(4B), 531-540.

(7) Breitburg, D. L.; Hondorp, D. W.; Davias, L. A.; Diaz, R. J. Hypoxia, nitrogen, and fisheries: Integrating effects across local and global landscapes. *Annual Review of Marine Science* **2009**, 1(1), 329-349.

(8) Lipton, D. W.; Hicks R. The cost of stress: Low dissolved oxygen and recreational striped bass (*Morone saxatilis*) fishing in the Patuxent River. *Estuaries* **2003**, 26, 310-315.

(9) Mistiaen, J. A.; Strand, I. E.; Lipton, D. Effects of environmental stress on Blue Crab (*Callinectes sapidus*) harvests in Chesapeake Bay tributaries. *Estuaries* **2003**, 26 (2A), 316-322.

(10) Bricker, S. B.; Longstaff, B.; Dennison, W.; Jones, A.; Boicourt, K.; Wicks, C.; Woerner, J. Effects of Nutrient Enrichment in the Nation's Estuaries: A Decade of Change. *Harmful Algae* **2008**, 8:21-32.

- (11) United States Clean Water Act. 1972. PUBLIC LAW 92-500-OCT. 18, 1972
<https://www.gpo.gov/fdsys/pkg/STATUTE-86/pdf/STATUTE-86-Pg816.pdf>.
- (12) European Union Water Framework Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000.
- (13) Stephenson, K.; Aultman, S.; Metcalfe, T.; Miller, A. An evaluation of nutrient nonpoint offset trading in Virginia: A role for agricultural nonpoint sources?: *Water Resour. Res.* **2010**, 46: WO4519.
- (14) Lindahl, O.; Hart, R.; Hernroth, B.; Kollberg, S.; Loo, L. O.; Olrog, L.; Rehnstam-Holm, A. S.; Svensson, S.; Syversen, U. Improving marine water quality by mussel farming: a profitable solution for Swedish society. *Ambio* **2005**, 34(2), 131–138.
- (15) Rose, J. M.; Bricker, S. B.; Tedesco, M. A.; Wikfors, G. H. A Role for Shellfish Aquaculture in Coastal Nitrogen Management. *Environ. Sci. Technol.* **2014**, 48, 2519–2525.
- (16) Ferreira, J. G.; Hawkins, A.; Bricker, S. B. The role of shellfish farms in provision of ecosystem goods and services. In: *Shellfish Aquaculture and the Environment*; S. E. Shumway, Ed.; Wiley-Blackwell: Hoboken, NJ **2011**, p. 3 – 31.
- (17) Ferreira, J. G.; Bricker, S. B. Goods and services of extensive aquaculture: shellfish culture and nutrient trading. *Aquaculture International* **2016**, 24 (3), 803–825.
- (18) National Oceanic and Atmospheric Administration (NOAA) Marine Aquaculture Policy. 2011.
http://www.nmfs.noaa.gov/aquaculture/docs/policy/noaa_aquaculture_policy_2011.pdf.
- (19) National Oceanic and Atmospheric Administration (NOAA) National Shellfish Initiative. 2011.
http://www.nmfs.noaa.gov/key_issues/17_natl_shellfish_initiative_homepage.html.
- (20) Executive Office of the President of the United States. Memorandum for Executive Departments and Agencies. M-16-01. Incorporating Ecosystem Services into Federal Decision Making. 2015.
<https://www.whitehouse.gov/sites/default/files/omb/memoranda/2016/m-16-01.pdf>.
- (21) Kellogg, M. L.; Cornwell, J. C.; Owens, M. S.; Paynter, K. T. Denitrification and nutrient assimilation on a restored oyster reef. *Marine Ecology Progress Series* **2013**, 480, 1-19.
- (22) Carmichael, R. H.; Walton, W.; Clark, H. Bivalve-enhanced nitrogen removal from coastal estuaries. *Canadian Journal of Fisheries and Aquatic Sciences* **2012**, 69, (7), 1131-1149.
- (23) Humphries, A. T.; Ayvazian, S. G.; Carey, J. C.; Hancock, B. T.; Grabbert, S.; Cobb, D.; Strobel, C. J.; Fulweiler, R. W. Directly Measured Denitrification Reveals Oyster Aquaculture and Restored Oyster Reefs Remove Nitrogen at Comparable High Rates. *Frontiers in Marine Science* **2016**, 3:74.
- (24) Pollack, J. B.; Yoskowitz, D.; Kim, H-C.; Montagna, P. A. Role and value of nitrogen regulation provided by oysters (*Crassostrea virginica*) in the Mission-Aransas Estuary, Texas, USA. *PLoS ONE* **2013**, 8(6), e65314.
- (25) Kellogg, M. L.; Smyth, A. R.; Luckenbach, M.W.; Carmichael, R. H.; Brown, B. L.; Cornwell, J. C.; Piehler, M. F.; Owens, M. S.; Dalrymple, D. J.; Higgins, C. B. Use of oysters to mitigate eutrophication in coastal waters. *Estuarine, Coastal and Shelf Science* **2014**, 151, 156-168.

(26) Oyster BMP Expert Panel. *Panel Recommendations on the Oyster BMP Nutrient and Suspended Sediment Reduction Effectiveness Determination Decision Framework and Nitrogen and Phosphorus Assimilation in Oyster Tissue Reduction Effectiveness for Oyster Aquaculture Practices*. Report submitted to the Chesapeake Bay Partnership Water Quality Goal Implementation Team, September 22, 2016.

www.chesapeakebay.net/calendar/event/24330/.

(27) Lal, H. Nutrient credit trading- a market-based approach for improving water quality. In *Advances in Nitrogen Management for Water Quality*; Delgado, J. A., Follett, R. F., Eds.; Soil and Water Conservation Society: Ankeny, IA 2010; pp. 344-361.

(28) Branosky, E.; Jones, C.; Selman, M. *Comparison Tables of State Nutrient Trading Programs in the Chesapeake Bay Watershed*. Washington, DC: World Resources Institute. 2011. www.wri.org/publication/comparison-tables-of-state-chesapeake-baynutrient-trading-programs.

(29) Connecticut Department of Environmental Protection (CT DEP). *An Incentive-based Water Quality Trading Program*. The Connecticut Department of Environmental Protection, Bureau of Water Protection and Land Reuse, Hartford, CT., 2010.

(30) Andersen, J. H.; Axe, P.; Backer, H.; Carstensen, J.; Claussen, U.; Fleming-Lehtinen, V.; Järvinen, M.; Kaartokallio, H.; Knuuttila, S.; Korpinen, S.; Laamanen, M.; Lysiak-Pastuszak, E.; Martin, G.; Møhlenberg, F.; Murray, C.; Nausch, G.; Norkko, A.; Villnäs, A. Getting the measure of eutrophication in the Baltic Sea: towards improved assessment principles and methods. *Biogeochemistry* **2011**, 106, 137–156.

(31) Garmendia, M.; Bricker, S.; Revilla, M.; Borja, A.; Franco, J.; Bald, J.; Valencia, V. Eutrophication Assessment in Basque Estuaries: Comparing a North American and a European Method. *Estuaries and Coasts* **2012**, 35 (4), 991-1006.

(32) Malone, T. C.; Conley, D. J.; Fisher, T. R.; Glibert, P. M.; Harding, L. W.; Sellner, K. G. Scales of nutrient-limited phytoplankton productivity in Chesapeake Bay. *Estuaries* **1996**, 19, 371–385.

(33) Lunstrum, A. Oyster aquaculture impacts on sediment nitrogen cycling and efficacy as a nutrient bioextraction tool in a tributary of Chesapeake Bay. Master's Thesis, University of Virginia, 2015.

(34) New York State Department of Environmental Conservation (NYSDEC) & Connecticut Department of Environmental Protection (CTDEP). *A Total Maximum Daily Load Analysis to Achieve Water Quality Standards for Dissolved Oxygen in Long Island Sound*. Prepared in Conformance with Section 303(d) of the Clean Water Act and Long Island Sound Study, 2000.

(35) Bricker, S. B.; Clement, C.; Frew, S.; Harmon, M.; Harris, M.; Pirhalla, D. *NOAA's National Eutrophication Survey. Volume 2: Mid- Atlantic Region*. NOAA, National Ocean Service, Office of Ocean Resources Conservation and Assessment, Silver Spring, 1997.

(36) Long Island Sound Study (LISS). *Protection and Progress: LISS Biennial Report 2011–2012*. NE IWGCC/ LISS, 2013.

(37) Lopez, G.; Carey, D.; Carlton, J. T.; Cerrato, R.; Dam, H.; DiGiovanni, R.; Elphick, C.; Frisk, M.; Gobler, C.; Hice, L.; Howell, P.; Jordaan, A.; Lin, S.; Liu, S.; Lonsdale, D.; McEnroe, M.; McKown, K.; McManus, G.; Orson, R.; Peterson, B.;

- Pickerrell, C.; Rozsa, R.; Shumway, S. E.; Siuda, A.; Streich, K.; Talmage, S.; Taylor, G.; Thomas, E.; VanPatten, M.; Vaudrey, J.; Yarish, C.; Wikfors, G.; Zajac, R. Chapter 6: Biology and Ecology of Long Island Sound. In *Long Island Sound: Prospects for the Urban Sea*; Latimer, J. S.; Tedesco, M. A.; Swanson, R. L.; Yarish, C.; Stacey, P. E.; Garza, C., Eds.; Springer: New York Heidelberg Dordrecht London; 2014; pp 285-480.
- (38) Bricker, S. B.; Ferreira, J. G.; Simas, T. An Integrated Methodology for Assessment of Estuarine Trophic Status. *Ecol. Modelling* **2003**, 169, 39-60.
- (39) Bricker, S. B., Ferreira, J. G.; Zhu, C.; Rose, J. M.; Galimany, E.; Wikfors, G. H.; Saurel, C.; Landeck Miller, R.; Wands, J.; Trowbridge, P.; Grizzle, R.E.; Wellman, K.; Rheault, R.; Steinberg, J.; Jacob, A. P.; Davenport, E. D.; Ayvazian, S.; Tedesco, M. A. *An Ecosystem Services Assessment using bioextraction technologies for removal of nitrogen and other parameters in Long Island Sound and Great Bay/Piscataqua Region*. NCCOS Coastal Ocean Program Decision Analysis Series No. 194. NOAA NCCOS, Silver Spring, MD and US EPA ORD, Atlantic Ecology Division, Narragansett, RI, 2015.
- (40) Latimer, J. S.; Tedesco, M. A.; Swanson, R. L.; Yarish, C.; Stacey, P. E.; Garza, C., Eds.; *Long Island Sound: Prospects for the Urban Sea*. Springer. New York Heidelberg Dordrecht London, 2014.
- (41) Connecticut Department of Energy and Environmental Protection (CT DEEP). 2013 Long Island Sound hypoxia season review. Connecticut Department of Energy and Environmental Protection: Hartford, CT 2013.
- (42) Connecticut Department of Agriculture. *Connecticut Shell Fishing industry profile, 2011*. <http://www.ct.gov/doag/cwp/view.asp?a=1369&q=316994>.
- (43) Ferreira, J. G.; Hawkins, A. J. S.; Monteiro, P.; Moore, H.; Service, M.; Pascoe, P. L.; Ramos, L.; Sequeira, A. Integrated Assessment of Ecosystem-Scale Carrying Capacity in Shellfish Growing Areas. *Aquaculture* **2008**, 275, 138-151.
- (44) Long Island Sound (LIS) Water Quality Monitoring Program. Undated. www.ct.gov/deep/cwp/view.asp?a=2719&q=325534.
- (45) Zar J. H. *Biostatistical Analysis*, Fourth edition. Prentice Hall: Upper Saddle River, New Jersey, 1999.
- (46) Long Island Sound Study (LISS). Undated. www.longislandsoundstudy.net/
- (47) Ferreira, J. G. EcoWin - An object-oriented ecological model for aquatic ecosystems. *Ecol. Modelling* **1995**, 79, 21-34.
- (48) HydroQual, Inc. *Newtown Creek WPCP Project East River Water Quality Plan, Task 10.0 - System-wide Eutrophication Model (SWEM), Sub-tasks 10.1 - 10.7*; Seven separate reports prepared under contract to Greeley and Hansen for the City of New York Department of Environmental Protection, 1999.
- (49) HydroQual, Inc. *Calibration Enhancement of the System-Wide Eutrophication Model (SWEM) in the New Jersey Tributaries*; Report to NJDEP, Final Technical Report April 23, 2001 through July 31, 2002. Prepared under subcontract to Passaic Valley Sewerage Commissioners, Newark, NJ, 2002.
- (50) Landeck Miller, R. E.; Wands, J. R. *Applying the System Wide Eutrophication Model (SWEM) for a Preliminary Quantitative Evaluation of Biomass Harvesting as a Nutrient Control Strategy for Long Island Sound*. HydroQual, Inc. Report prepared for

the Long Island Sound Study under contract agreement with the Hudson River Foundation, 2009.

(51) Blumberg, A. F.; Khan, L. A.; St. John, J. P. Three-Dimensional Hydrodynamic Model of New York Harbor Region. *J. Hydr. Engrg. ASCE* **1999**, 125 (8),799-816.

(52) Landeck Miller, R. E.; St. John, J. P. Modeling Primary Production in the Lower Hudson River Estuary. In *The Hudson River Estuary*; Levinton, J. S.; Waldman, J. R., Eds.; Cambridge: New York, NY, 2006; pp. 140-153.

(53) Nunes, J. P.; Ferreira, J. G.; Bricker, S. B.; O'Loan, B.; Dabrowski, T.; Dallaghan, B.; Hawkins, A. J. S.; O'Connor, B.; O'Carroll, T. Towards an ecosystem approach to aquaculture: assessment of sustainable shellfish cultivation at different scales of space, time and complexity. *Aquaculture*, **2011**, 315, 369-383.

(54) HydroQual, Inc. *Reassessment of the 2000 Total Maximum Daily Load Analysis to Achieve Water Quality Standards for Dissolved Oxygen in Long Island Sound. Additional SWEM Scenarios to Identify Dissolved Oxygen Responses to Load Reductions in between TMDL and Pastoral Loadings*. Summary Findings Report, Prepared for USEPA Long Island Sound Study, 2007.

(55) HydroQual, Inc. *System Wide Eutrophication Model (SWEM) Evaluation of Long Island Sound (LIS) TMDL Writing Team Recommended Loading Scenarios Results*. Prepared for USEPA Long Island Sound Study, 2009.

(56) Rice, E.; Stewart, G. Analysis of interdecadal trends in chlorophyll and temperature in the Central Basin of Long Island Sound. *Est. Coast Shelf Sci.* **2013**, 128, 64-75.

(57) Ferreira, J.G.; Hawkins, A. J. S.; Bricker, S. B. Farm-scale assessment of shellfish aquaculture in coastal systems – the Farm Aquaculture Resource Management (FARM) model: *Aquaculture* **2007**, 264, 160–174

(58) Ferreira, J. G.; Sequeira, A.; Hawkins, A. J. S.; Newton, A.; Nickell, T. D.; Pastres R.; Forte J.; Bodoy, A.; Bricker, S. B. Analysis of coastal and offshore aquaculture: Application of the FARM model to multiple systems and shellfish species: *Aquaculture* **2009**, 292, 129-138.

(59) Silva, C.; Ferreira; J. G.; Bricker, S. B.; DelValls, T. A.; Martín-Díaz, M. L.; Yáñez, E. Site selection for shellfish aquaculture by means of GIS and farm-scale models, with an emphasis on data-poor environments: *Aquaculture* **2011**, 318, 444–457.

(60) Bricker, S. B.; Rice, K. C.; Bricker III, O. P. From Headwaters to Coast: Influence of Human Activities on Water Quality of the Potomac River Estuary. *Aquat Geochem* **2014**, 20:291-324

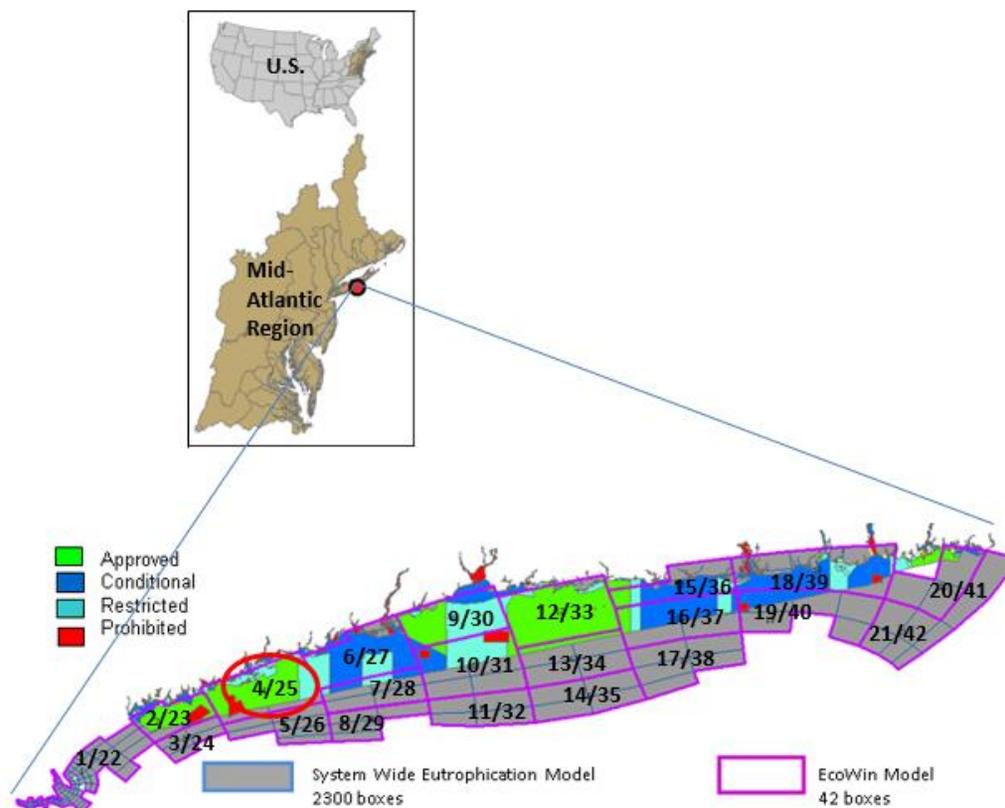
(61) King, D. M.; Mazzotta, M. J. *Ecosystem Valuation. Methods, Section 1. Market Price Method*. 2000. www.ecosystemvaluation.org/market_price.htm.

(62) Gren, I.-M.; Lindahl, O.; Lindqvist, M. Values of mussel farming for combating eutrophication: An application to the Baltic Sea. *Ecological Engineering* **2009**, 35 (5), 935-945.

(63) Chesapeake Bay Program. *Nutrient Reduction Technology Coast Estimates for Point Sources in the Chesapeake Bay Watershed*, 2002.

(64) Evans, B.M. *An Evaluation of Potential Nitrogen Load Reductions to Long Island Sound from the Connecticut River Basin*. Report Submitted to New England Interstate Water Pollution Control Commission, Lowell, MA, 2008.

- (65) Higgins, C. B.; Stephenson, K.; Brown, B. L. Nutrient bioassimilation capacity of aquacultured oysters: quantification of an ecosystem service. *J. Environ Qual.* **2011**, *40*, 271–277.
- (66) Inglis, G. J.; Hayden, B. J.; Ross, A. H. *An Overview of Factors Affecting the Carrying Capacity of Coastal Embayments for Mussel Culture*. National Institute of Water & Atmospheric Research (NIWA) Client Report CHC00/69, Christchurch, 2000.
- (67) McKindsey, C. W.; Thetmeyer, H.; Landry, T.; Silvert, W. Review of recent carrying capacity models for bivalve culture and recommendations for research and management. *Aquaculture* **2006**, *261*, 451–462.
- (68) Sequeira, A.; Ferreira, J. G.; Hawkins, A. J.; Nobre, A.; Lourenço, P.; Zhang, X. L.; Yan, X.; Nickell, T. Trade-offs between shellfish aquaculture and benthic biodiversity: A modelling approach for sustainable management. *Aquaculture* **2008**, *274*, 313–328.
- (69) Filgueira, R.; Guyondet, T.; Bacher, C., Comeau, L. A. Informing Marine Spatial Planning (MSP) with numerical modelling: A case study on shellfish aquaculture in alpeque Bay (Eastern Canada). *Marine Pollution Bulletin* **2015**, *100*, 200–216.
- (70) Ferreira, J. G.; Saurel, C.; Lencart e Silva, J. D.; Nunes, J. P.; Vazquez, F. Modelling of interactions between inshore and offshore aquaculture. *Aquaculture* **2014**, *426–427*, 154–164.
- (71) Rose, J. M.; Bricker, S. B.; Ferreira, J. G. Modeling shellfish farms to predict harvest-based nitrogen removal. *Marine Pollution Bulletin* **2015**, *453*, 135–146
- (72) Stephenson, K.; Shabman, L. *Nutrient assimilation services for water quality credit trading programs*. Discussion paper. Resources for the Future, Washington, DC, 2015, RFF DP 15-33.
- (73) Town of Mashpee Sewer Commission. *Final Recommended Plan / Final Environmental Impact Report. Comprehensive Wastewater Management Plan. Town of Mashpee*. Prepared by GHD Inc., Hyannis, Massachusetts, 2015.
- (74) Reitsma, J.; Murphy, D.C.; Archer, A. F.; York, R.H. Nitrogen extraction potential of wild and cultured bivalves harvested from nearshore waters of Cape Cod, USA. *Marine Pollution Bulletin*, **2017**, *116*, 175–181.
- (75) Nobre, A. M.; Ferreira, J. G.; Nunes, J. P.; Yan, X.; Bricker, S.; Corner, R.; Groom, S.; Gu, H.; Hawkins, A. J. S.; Hutson, R.; Lan, D.; Lencart e Silva, J. D.; Pascoe, P.; Telfer, T.; Zhang, X.; Zhu, M. Assessment of coastal management options by means of multilayered ecosystem models. *Estuarine Coastal and Shelf Science*, **2010**, *87(1)*: 43–62.



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617 Figure 1: Location map of Long Island Sound with inset U.S. and North Atlantic region maps. High resolution System Wide
 618 Eutrophication Model (SWEM) grid box boundaries shown in blue, and broader scale EcoWin (EWN) ecological model boxes shown
 619 in purple. Shellfish classification areas in EWN model boxes that included oysters also shown (see key at left). Surface and bottom
 620 boxes are enumerated where 1/22 indicates surface box 1 and bottom box 22. Oyster production uses bottom boxes only to simulate
 621 bottom culture with no gear, the typical cultivation practice in LIS where boxes 23, 25, 27, 30, 33, and 41 are the only boxes that
 622 include oyster aquaculture. Box 25 which includes the largest lease area and is the box used for marginal analysis is denoted with a red
 623 circle.

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Table 1: Incremental costs and reductions from point source controls at three levels of effluent nutrient concentration, and costs of implementation of agricultural and full urban best management practices (from Evans, 2008). Results are reported in 2013 U.S. dollars

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Alternative nutrient reduction measure	Capital costs (\$ million)	O&M (\$ million)	Annualized cost (\$ million)	Nitrogen removed 10 ³ lb yr ⁻¹ 10 ³ kg yr ⁻¹	Average cost \$ lb ⁻¹ yr ⁻¹ \$ kg ⁻¹ yr ⁻¹
WWTP 8 mg L ⁻¹	433	8.67	30.4	2,070 941	14.63 32.19
WWTP 5 mg L ⁻¹	143	4.22	11.4	677 308	16.82 37.00
WWTP 3 mg L ⁻¹	316	12.6	28.4	634 288	44.81 98.58
Agricultural BMP			7.68	1,310 595	5.90 12.98
Full Urban BMP			163	1,040 473	159 349

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637 Table 2: EcoWin model outputs for Standard model scenario, and specific results for Potential scenario (bold font), for oyster
 638 aquaculture impacts on water clearance and nutrient removal in Long Island Sound, Connecticut determined from model simulations
 639 for Year 9. Note: i) the whole acreage is used, rather than the annual seeded acreage, because bioextraction is evaluated as a
 640 contribution of all year classes, ii) Total POM uptake includes both phytoplankton and detrital organic material, phytoplankton uptake
 641 is also shown separately. Total N inputs are 50×10^3 metric tons y^{-1} .

Year 9	Box 23	Box 25	Box 27	Box 30	Box 33	Box 41	Total for Standard Scenario	Total for Potential Scenario
Acres (Standard scenario)	105	4,040	53	788	53	210	5,250	11,100
Oyster Harvest (ton y^{-1})	630	24,300	306	4,490	303	1,220	31,300	63,900
Clearance volume (%total volume y^{-1})	20.8	370	7.00	145	5.00	19.6	95.0	208
Total phytoplankton uptake (kg N y^{-1})	15,000	490,000	5,990	87,000	5,700	22,300	626,000	
Total POM uptake (includes phytoplankton) (kg N y^{-1})	24,500	846,000	11,320	183,200	14,500	83,000	1,160,000	
Total DIN excretion (kg N y^{-1})	2,270	80,800	1,040	15,700	1,080	4,680	106,000	
Total feces (kg N y^{-1})	7,770	276,000	3,860	65,500	5,540	34,800	394,000	
Total mortality (kg N y^{-1})	136	5,590	76.0	1.17	78.9	326	7,370	
Total N uptake (kg N y^{-1})	24,500	846,000	11,300	183,200	14,500	823,000	1,160,000	
Total N release (kg N y^{-1})	10,200	363,000	4,980	82,300	6,690	39,800	507,000	
Net N removal (kg N y^{-1})	14,400	484,000	6,340	101,000	7,800	43,200	656,000	1,340,000
N removal as % biomass	2.28	1.99	2.07	2.25	2.57	3.53	2.10	
Net N removal (lb N acre $^{-1}$ y^{-1})	301	264	266	282	327	453	275 ¹	265¹
(kg N acre $^{-1}$ y^{-1})	137	120	121	128	149	206	125 ¹	120¹
Average Physical Product (APP)* (harvest / seed)	47.9	48.0	46.5	45.4	46.0	46.4	47.5 ²	45.9²
Population-equivalents (PEQ y^{-1})	4,350	147,000	1,920	30,600	2,360	13,100	199,000	405,000
% reduction in percentile 90 Chl concentration (from phytoplankton loss)	1.40	2.80	1.50	1.40	0.800	0.100	0.100 - 2.80	0.500 - 5.30

642 ¹Net N removal for total is an average of all boxes; ²APP for total is aggregate value

643 *When APP is >1 this means that there is more than one kg of product that is harvested from one kg of seed where the profit margin will depend on the cost of the seed and the
 644 value of the harvested product. The breakeven point is dependent on the relative costs of seed and product, technically the threshold is the point where APP is equal to Pi/Po (price
 645 of input/price output).

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Table 3: Average bioextraction nitrogen removal value for Connecticut, Long Island Sound based on an avoided costs approach considering 3 wastewater treatment plant effluent levels, agricultural and urban best management practices (BMP) as alternate management measures. Boxes are bottom boxes where oyster aquaculture occurs, showing total for each box and value of lease acres (see figure 1). Results are reported in 2013 U.S. dollars.

Scenario	Level	1000 U.S. \$						Total Value (10 ³ U.S. \$ y ⁻¹)	Weighted Average (10 ³ US \$ acre ⁻¹ yr ⁻¹)
		Box 23	Box 25	Box 27	Box 30	Box 33	Box 41		
Standard (5,250 acres)	8mg/l	462	15,600	204	3,250	251	1,390	21,100	4.03
	5mg/l	532	17,900	235	3,740	289	1,600	24,300	4.63
	3mg/l	1,420	47,800	626	9,950	770	4,260	64,800	12.30
	Agricultural BMP	187	6,300	82	1,310	101	561	8,540	1.63
	Urban BMP	5,040	170,000	2,230	35,400	2,740	15,100	230,000	43.90
Potential (11,116 acres)	8mg/l	935	31,600	421	6,720	528	2,950	43,100	3.88
	5mg/l	1,080	36,300	484	7,720	607	3,390	49,600	4.46
	3mg/l	2,860	96,700	1,290	20,600	1,620	9,020	132,000	11.90
	Agricultural BMP	377	12,700	170	2,710	213	1,190	17,400	1.57
	Urban BMP	10,200	344,000	4,580	73,100	5,750	32,100	469,000	42.20

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