2016

Fifteen Years of Rhode Island Oyster Restoration: A Performance Evaluation and Cost-Benefit Analysis

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FIFTEEN YEARS OF RHODE ISLAND OYSTER RESTORATION: A PERFORMANCE EVALUATION AND COST-BENEFIT ANALYSIS

BY

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A THESIS SUBMITTED IN PARTIAL FULFILLMENT OF THE

REQUIREMENTS FOR THE DEGREE OF

MASTER OF SCIENCE

IN

ECOLOGY AND ECOSYSTEM SCIENCE

UNIVERSITY OF RHODE ISLAND

2016

MASTER OF SCIENCE THESIS

OF

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2016

ABSTRACT

Federal, state and local non-profit organizations have long recognized the ecological and socioeconomic importance the oyster, *Crassostrea virginica*, represents to coastal communities. Shellfish restoration programs in Rhode Island date to the early 1900s and have been making considerable progress and gaining popularity in the past decade. To better understand both short and long term performance of oyster restoration in Rhode Island a compilation of all oyster restoration activities from 2000 to 2015 was undertaken. Restoration performance was assessed by comparisons of growth, survival, disease and recruitment over eleven years in two distinct programs; Roger Williams University's Oyster Gardening for Restoration (2006 - 2014) and the *North Cape* Shellfish Restoration Program (2003 - 2008). Mean costs of restoration were weighed against cumulative value of ecosystem services provide by oyster reefs. Over 26 million oysters, encompassing 6.6 acres have been seeded in thirteen distinct restoration sites in Rhode Island waters including salt ponds, tidal creeks and open coves in Narragansett Bay. Mean growth of oysters in restoration sites was between 30-50 mm annually with mean survival of 22% and 55% for year one and two+ oysters, respectively. Mortality varies among sites and appears to be driven largely by disease. Mortality outpaces recruitment at all monitored sites leading to a decline of the population once seeding has ceased, driving the need for continued restoration to maintain desired ecosystem services. A cost-benefit model indicates Rhode Island oyster restoration is not equitable in terms of

ecosystem services provided, as the cost of restoration is higher than the cumulative value of ecosystem services provided by the oyster reefs, thus, questioning the economic feasibility of restoration and emphasizing the importance of proper site selection coupled with alternate management strategies.

ACKNOWLEDGMENTS

This body of work would not have been possible without the support from the National Oceanic and Atmospheric Administration, Roger Williams University, and the Rhode Island Department of Environmental Management. Dale Leavitt provided invaluable guidance during field efforts. Dr. Roxanna Smolowitz and Kathryn Markey provided insight on shellfish diseases and pathology testing. Thanks to Roger Williams University students and Rhode Island Department of Environmental Management student researchers for putting in countless hours monitoring oyster reefs. A special thanks to Dale Leavitt, Bryan DeAngelis, Boze Hancock and Michael Rice for their dedication to shellfish aquaculture and restoration science and facilitating my continued learning in the field. This work was supported by funding from the National Oceanic and Atmospheric Administration (Grant # NA10NMF4630362) awarded to RWU (T. Scott and D. Leavitt, Principal Investigators). Thanks to Senator Jack Reed for his support with this funding.

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CHAPTER 1

INTRODUCTION

The eastern oyster (*Crassostrea virginica*) is an epifaunal bivalve, distributed along the Atlantic and Gulf coasts from the Yucatan Peninsula, Mexico to the Gulf of St. Lawrence, Canada (Buroker 1983). Eastern oysters play a critical ecological role within our coastal environment. Often dubbed *'ecosystem engineers'*, this role has been recognized as early as Moebius's (1883) pioneering monograph on oysters and oyster culture. Oysters are capable of benthic-pelagic coupling by filtering phytoplankton and seston and transporting this organic matter to the benthos, thus supplementing benthic food webs and accelerating nutrient cycling within the system (Dame 1993, Smaal and Prins 1993, Pietros and Rice 2003). Through filter feeding activities, *C. virginica* increases water clarity, reduces turbidity (Cloern 1982, Newell 1988) as well as reduces carbon, nitrogen, (Hargis and Haven 1999) and pollutants from the water column (Tolley *et al.* 2005). Oyster beds create complex biogenic structures, which increase species density, biomass and richness over nearby mud habitats (Tolley and Volety 2005, Manley *et al.* 2010, Abeels *et al.* 2012, Quan *et al.* 2012) and serve as essential fish habitat (Coen *et al.* 1999, Peterson *et al.* 2003); ultimately increasing productivity within our coastal waters (Grabowski *et al.* 2004, Grabowski *et al.* 2008).

Oyster beds, when healthy, can provide a direct economic benefit to coastal communities through both commercial and recreational fisheries and the infrastructure which support them. In the late 1800s and early 1900s, Narragansett Bay housed over 21,000 acres of private oyster beds resulting in annual landings of 14 million pounds (DeAlteris *et al.* 2000). Oyster populations are of the most degraded ecosystems in the world, with a global reef loss of 85% and reefs in New England have been considered functionally extinct (Beck *et al.* 2011). Since the mid-1900s, Rhode Island's oysters stocks have dramatically decreased due to overharvest, habitat and water quality degradation coupled with the spread of disease (DeAlteris 2000). Federal, State and local non-profit organizations have long recognized both the ecological and socioeconomic importance the oyster represents to Rhode Island. Various shellfish restoration programs in Rhode Island date to the early 1900s (Rice *et al.* 2000) and have been making considerable progress and gaining popularity in the past fifteen years.

Since 2000, four distinct oyster restoration programs have been initiated in Rhode Island, including: 1) the *North Cape* restoration program (Rhode Island Department of Environmental Management/National Oceanic and Atmospheric Administration), aimed to address the natural resource injuries resulting from the release of 828,000 gallons of heating oil into Block Island Sound during the 1996 *North Cape* oil spill; 2) the Oyster Gardening for Restoration and Enhancement (OGRE) program (Roger Williams University), aimed at increasing spawning stock biomass of oysters through community involvement; 3) the Environmental Quality and Incentives program (United States Department of Agriculture - Natural Resource Conservation Service), seeding oysters through cooperative help of aquaculturists; and 4) The Nature Conservancy in collaboration with the Rhode Island Department of Environmental Management, focusing on restoring oyster populations through increasing suitable settlement substrate. Each program has employed different approaches to restoration, including: direct seeding efforts of both single set oysters and spat on shell with varied density regimens and broodstock lines, targeting different habitat to increase oyster performance (i.e. substrate type, hydrodynamics, reef height, tidal height) as well as the construction of artificial reefs to increase suitable settlement substrate. Coupled with the aforementioned restoration programs, the Rhode Island Department of Environmental Management has addressed restoring oyster populations through the use of permanent closures and the creation of spawner sanctuaries.

Monitoring restored populations and the associated habitat is a fundamental part of the restoration process and allows practitioners and managers to learn from previous efforts and progress toward more successful restoration (Brumbaugh *et al.* 2006). Despite the increase in shellfish restoration in Rhode Island, careful monitoring of the restored populations and associated habitat has, in some cases, taken a back seat to efforts of introducing shellfish into estuaries. Prior to 2011, monitoring of oyster restoration in Rhode Island was completed without standardized metrics, resulting in mixed techniques and performed on varied temporal and spatial levels. Results from restoration monitoring have been, in some cases, reported in grant progress reports and, in other cases, have not been formalized or made available to the public. This lack of organization of data has resulted in the lack of ability to understand project performance in the context of oyster restoration on the state level.

It has long been recognized the effectiveness of restoration projects must be evaluated against a reference (Fagan *et al.* 2008). To progress toward more effective restoration, managers must understand both, the reference of target or 'un-degraded' ecosystems as well as the reference of previous restoration methods and performance. Based on number of oysters seeded on an annual basis, Rhode Island restoration has increased by a factor of twenty in the past fifteen years. The number of restoration sites, methodology used and entities involved has also increased dramatically. Despite the increase in restoration, communication between practitioners (i.e. government, NGOs and academia) of basic achievements, performance results and research remains minimal. Two informal oyster restoration summits (2003 and 2007) were organized by restoration scientists at Roger Williams University (RWU) to attempt to coordinate activities and share information, which led to the formation of the Rhode Island Shellfish Technical Working Group (RISTWG), a volunteer advisory council to the Rhode Island Coastal Resource Management Council (RI-CRMC). The RISTWG was created to provide a framework for coordination and communication between the agencies and groups involved in various shellfish restoration activities. The RISTWG is represented by federal, state, academic, NGO, wild-harvesters, and aquaculturists, acting as a centralized body commenting on Rhode Island shellfish restoration activities and collaboratively working together to further state-wide shellfish restoration planning, prioritization, and goal setting. The RISTWG recognizes the lack of a centralized document detailing all oyster restoration practices, performance and shortfalls within the state, thus, hindering our ability to analyze oyster restoration across projects, sites, and methods; ultimately creating a bottleneck of knowledge and encumbering our ability to progress towards more successful restoration.

The ability of oyster reefs to provide ecosystem services, including but not limited to increased water quality, habitat and fish production has gained recognition in both the scientific and political communities. Increasing water quality, habitat and fish production have been identified as federal priorities, set forth by the National Oceanic and Atmospheric Administration (NOAA). Many approaches to achieve these goals exist including (e.g. managing combined sewage treatment outflows, restoring upland and shoreline grass habitats, implementing artificial reefs and restoring shellfish beds) and are practiced locally. Oyster restoration has been touted as a cost-effective approach (Piehler and Smyth 2011, Grabowski *et al.* 2012) and is often funded on this basis. Values of ecosystem services are likely to be highly context specific, dependent upon practice, scale of restoration, population dynamics, biophysical and chemical parameters of the given habitat and management of the restored area. Valuation of ecosystem services, in economic terms, by oyster reefs has been published in primary literature (Henderson and O'Neil 2003, Piehler and Smyth 2011, Grabowski *et al.* 2012). The published values have not been fit to Rhode Island oyster restoration data and an understanding of the cost-benefit analysis of Rhode Island restoration efforts does not exist. Quantifying the value associated with ecosystem services of Rhode Island oyster reefs and understanding the changes in cost-benefit ratios dependent upon practice and site location will enhance our ability to maximize our investments and to appropriately allocate limited funding.

This work aims to: 1) document past oyster restoration in Rhode Island from 2003 to present, including methods and completed effort; 2) measure and analyze restoration performance and 3) incorporate these data into a cost-benefit analysis based on ecosystem services provided by oysters. This information will allow us to comment on the efficacy of oyster restoration in the state and provide suggestions for future efforts. Ultimately this document will provide an additional tool for authorities to adaptively manage oyster restoration to optimize both ecological services and economic investment.

CHAPTER 2

METHODOLOGY

Data Compilation

To allow for analysis of restoration performance across sites, years and projects, data was compiled from multiple sources including direct field work, annual reports, progress reports and personal communication. Monitoring and restoration data from the *North Cape* Shellfish Restoration Program (NCSRP) was personally collected from 2004 to 2009. Monitoring and restoration data from Roger Williams University's Oyster Gardening for Restoration and Enhancement (OGRE) was accessed through annual reports from 2006 to 2011 and personally collected from 2011 to 2014. Restoration data from The Nature Conservancy (TNC) and the United States Department of Agriculture – Natural Resources Conservation Service, Environmental Quality Incentives program (EQIP) were generated from annual reports, progress reports and personal communication. Raw monitoring data from TNC and EQIP restoration efforts were not available, therefore, these programs were excluded from calculations of restoration performance and cost-benefits.

Population Structure and Site Characteristics

North Cape restoration sites were surveyed between July and October of 2004 to 2008 and 2011 to 2013. Site boundaries were re-established using a hand held Garmin Global Positioning System and direct observation to determine the limits of oysters seeded in previous years. Seeded boundaries were then marked with surface floats and boundary edges measured with a 100 m tape, ensuring the area surveyed was accurately calculated. An average of 1.9% of total site area was sampled using 1 m^2 quadrats. Boats traveled an approximate grid within the site boundary, evenly distributing quadrats in a haphazard un-biased distribution. Divers excavated all live and recently dead 'boxes' (hinges still intact) oysters within each quadrat, in which a subsample of 50 live specimens and 50 boxes were measured from umbo to lip to the nearest mm. To assess recruitment to the site, oyster recruits or 'over-set' were tallied independently from the seeded cohorts. Total oyster abundance (± SE) within each site was estimated from mean densities sampled, using total site area as the basis for extrapolation.

Oyster Gardening for Restoration and Enhancement (OGRE) sites were monitored between July and October each year from 2011 to 2014. Due to the varied seeding practices between *North Cape* and OGRE restoration programs, monitoring methods differed slightly. OGRE sites were seeded with multiple, highly dense oyster beds with negligible presence of oysters between beds. *North* Cape sites were seeded with a lower density of oysters over a large area. In efforts to keep density variances to a minimum OGRE oyster bed were sampled independently from one another. Oyster beds were sampled using evenly distributed, haphazardly deployed quadrats. Large beds, generally greater than 100 m^2, with highly variable oyster distributions were sampled with 1 m^2 quadrats, while smaller beds with evenly distributed oysters were sampled with 0.25 m^2 quadrats. The smaller seeded footprint in comparison to the *North Cape* sites, allowed for greater overall sampling coverage. On average 5.2% of total bed area was sampled. Divers excavated all live oysters and boxes within each quadrat and in which a subsample of 50 live specimens and 50 boxes were measured from umbo to lip to the nearest mm. As an indicator of recruitment to the site, oyster recruits or ‘over-set’ was tallied independently from the seeded cohorts. Density was calculated independently for each bed and number of oysters (± SE) per bed was estimated from mean densities sampled, using bed area as a basis for extrapolation.

The majority of oyster beds in both programs (OGRE and *North* Cape) were over seeded on an annual or biannual basis, creating a reef of mixed cohorts. Tracking growth and survival of individual cohorts was accomplished through assessments of length distributions. Due to the reefs composition of mixed cohorts discerning precise 1st year survival and growth was not reliable. To track annual growth and survival on a finite scale, experimental reefs were seeded with single cohorts in Quonochontaug Pond, Smelt Brook Cove and Bissel Cove from 2011 to 2014. Within the experimental reefs first year survival was calculated by dividing the mean density of live oysters by the sum of dead oysters including scars (presence of $CaCO_3$ shell deposits on the setting media but absence of both valves). Annual growth was calculated by comparing length distributions between survey years.

Relative index of recruitment to restoration sites and surrounding areas was monitored with the use of artificial spat collectors. These consisted of individual polyethylene mesh bags filled with approximately 4 L of surf clam (*Spisula solidissima)* shell, moored to the seabed and hung in mid-water column with a surface float. Spat collectors were located in close proximity to each restoration site and spread throughout the water body, up to three kilometers away from the site of restoration, in efforts to observe spatial distribution of recruitment. Locations were chosen based on local hydrodynamics and wind patterns. Collectors were deployed prior to the first seasonal oyster spawn and retrieved in the fall of each survey year, as to represent one season of recruitment activity. Upon retrieval, collectors were transported to the RIDEM Coastal Fisheries Laboratory (*North Cape* 2004 – 2008) or Roger Williams University (OGRE 2011 – 2013) for analysis, where number of oyster spat per collector was enumerated. Depending on site size, five to ten spat collectors were placed within each body of water annually.

Disease Monitoring

Samples of 25 oysters were collected for disease testing from all *North Cape* and OGRE restoration sites in each survey year. Oysters sampled were from cohorts seeded in previous years, with the exception of Spectacle and Potter Coves (2011 – 2013), where the size and condition of shell hindered the proper protocol for analysis. At these sites, wild oysters in close proximity to the restoration areas were collected for disease sampling. All oysters were collected between September 1st and October 5th within each year and ranged from 60 to 90 mm valve height. Samples were transported on ice to the Aquatic Diagnostic Laboratory at Roger Williams University, where presence and severity of *Perkinsus marinus*, the pathogen causing dermo disease, was assessed using Ray's fluid thioglycollate medium and visual inspection of tissue. Results were reported in percent prevalence of the disease in each sample as well as intensity; a measure of concentration of *P. marinus* spores in infected individuals. Samples were also assessed through traditional histopathology for presence and severity of *Haplosporidium nelsoni* (MSX), *Haplosporidium costale* (SSO) and trematodes. Mean disease prevalence per site was compared using normal quantile transformations followed by Analysis of Variance and Tukey's post-hoc test ($\alpha = 0.05$). Regression analysis was used to test the relationship between density of oysters and percent prevalence of dermo ($\alpha = 0.05$).

Performance Evaluation

Performance metric evaluations were calculated for all *North Cape* and Oyster Gardening for Restoration and Enhancement sites. Due to the lack of available data, performance evaluations were not calculated for The Nature Conservancy and Environmental Quality and Incentives Program restoration efforts. Performance evaluations include first year survival, year two plus survival, recruitment to the restoration footprint and prevalence of dermo

(Perkinsus marinus). First year survival and annual growth were calculated using only data from the experimental reefs, described above (*see Site Monitoring)*, and reefs in which first year cohorts were discernable (Bissel Closed 2, 2007; Potter Cove, 2004 – 2006; Saugatucket River, 2004 – 2006; Smelt Brook Cove, 2004 – 2006 & experimental reefs; Spectacle Cove 2004 - 2006). Evaluations were computed by calculating the mean value of the given metric (e.g. recruitment) for each restoration site within all years and projects where data were available. Mean performance metrics were compared between sites using normal quantile transformations followed by Analysis of Variance and Tukey's post-hoc test ($\alpha = 0.05$).

Despite potential differences in mean growth, survival, recruitment and disease between restoration sites and practices, the ultimate success of restoration hinges on sustainability of oyster reefs post implementation. Sustainability was assessed based on the level of natural mortality (year 2+) weighed against recruitment. A sustainability index (SI) was calculated for each site each year post restoration implementation. The index is based on the following equation.

$$SI = R_i - M_i$$

Where: R_i = percentage recruits of the total population
M_i = percentage of mortality

Mean SI was compared between sites with normal quantile transformations followed by Analysis of Variance and Tukey's post-hoc test ($\alpha = 0.05$).

Cost-Benefit Analysis

To assess the effectiveness of Rhode Island's oyster restoration, based on financial investments and ecological returns, a cost-benefit model was fit to Rhode Island oyster restoration performance over the past 15 years. The model weighed projected annual dollar value returns associated with the sum of water quality improvement, fish production, and submerged aquatic vegetation enhancement per acre of oyster reef against the cost of implementing one acre of oyster reef and extrapolated over a fifty year time frame. To account for loss of ecosystem services provided by oyster reefs due to oyster mortality, the cumulative value of ecosystem services was discounted at a rate of 3% per year between seeding intervals. It was assumed inflation will impact the value of ecosystem services and costs of restoration equally.

Costs associated with Rhode Island oyster restoration were derived from the *North Cape* Shellfish Restoration Program and Oyster Gardening for Restoration and Enhancement annual budgets. Annual operation costs associated with the OGRE program were calculated by multiplying the total cost of hatchery operation budgets and OGRE field staff by the percentage of overall effort to complete oyster restoration. Annual operation costs of the *North Cape* oyster restoration program was derived from the sum of individual line items associated with oyster restoration efforts. Staff salary was divided by the percentage of overall effort to complete oyster restoration activities. Educational, research, and other extraneous services outside of oyster production, nursery, and seeding were excluded from cost calculations within both programs. Overhead and indirect expenses were not included in staff salary costs. Cost of restoration was converted to dollar value per acre of reef by dividing the annual restoration cost by the annual acreage seeded within each program. Total mean cost per acre of restored reef was calculated using data from both programs.

The economic value of ecosystem services provided by oyster reefs was adapted from Grabowski *et al.* (2012); the procedures from his work are outlined below (see Grabowski *et al.* (2012) for precise methodology). It should be noted, due to lack of data within the Northwest Atlantic, the ecosystem services provided by oyster reefs (nitrogen removal, fish production, and submerged aquatic vegetation enhancement) were calculated using data from estuaries in the southeastern United States.

Proxy measures were used to determine the value of water quality services (i.e. the cost of providing the same ecosystem service through alternative means). To determine the amount of incremental nitrogen removed from the system by oyster reefs, the nitrogen flux in soft-sediment bottom was subtracted from the nitrogen flux in oyster reefs. The net hourly rate of nitrogen removal was determined to be 246 and 12 micromoles of nitrogen per square meter per hour during the day in oyster reefs and in mud habitat, respectively (Piehler and Smyth 2011). Nitrogen removal by 1 m^2 of oyster reef was converted to annual kilograms of nitrogen removed per acre of oyster reef and multiplied by the rate of nitrogen removal by the trading price per kilogram of nitrogen removed for estuarine sites in the North Carolina Nutrient Offset Credit Program.

Nitrogen removal through the consumption of phytoplankton was based on the estimated removal of 40 micrograms per liter (µg/L) of chlorophyll-a. A carbon:chlorophyll-a ratio of 30 (Wienke and Cloern 1987) was used to convert chlorphyll-a removal to carbon removal, followed by converting carbon removal to nitrogen removal using the Redfield ratio (Redfield 1958). The estimated value of nitrogen removal was calculated using the trading price per kilogram of Nitrogen in the North Carolina Nutrient Offset Credit Program. Nitrogen stored in oyster shell and tissue was not accounted for, as harvest is prohibited from all restored oyster reefs in Rhode Island.

Newell and Koch (2004) suggest that the oyster's ability to reduce turbidity and by depositing nutrients in biodeposits enables oyster reefs to promote the growth of submerged aquatic vegetation in shallow estuarine waters at an estimated rate of 0.005 hectare of SAV per one hectare of oyster reef. The importance of submerged aquatic vegetation as nursery ground for many coastal species is well understood (Thayer *et al.* 1978). Grabowski *et al.* (2012) used surveys of local resident's willingness-to-pay to determine the value of eelgrass habitat in the Peconic River Estuary, coupled with the value of ecosystem services provided by seagrass habitat, to estimate the value of seagrass beds per hectare. This value was multiplied by the estimated rate of growth of seagrass beds created by one hectare of oyster reef.

Peterson *et al.* (2003) estimated 10 m^2 of restored oyster reef habitat creates an additional 2.6 kilograms of fish and mobile crustacean production annually. This figure was derived by comparing densities of all species of fish and commercially important crustaceans on oyster reefs versus mud bottom throughout the Gulf of Mexico and southeast Atlantic states. Enhanced value of commercial fish landings per m^2 of oyster reef was estimated using the data above as augmented fish production estimates from oyster reefs (Grabowski and Peterson 2007).

Oyster harvest value was not taken into account for the Rhode Island model as all reefs are protected from harvest. Oyster reefs can function as natural living erosion protection, however, all Rhode Island oyster reefs are low lying with very little relief so this benefit is not realized; therefore, it was not included in the model.

Mapped Footprint of Restoration Reefs

Discrete footprints of restoration efforts were mapped in ArcMap 10.4.1 for each restoration site and project (i.e. *North Cape*, OGRE, EQIP and TNC). *North Cape* and OGRE restoration reefs were measured in-situ via direct observation to determine the extent of reef. Survey stakes were placed on the reef boundaries and perimeters measured with a handheld tape measure. Coordinates of survey stakes were recorded with a handheld GPS. Reef coordinates and boundaries were transposed in ArcMap followed by calculations of reef area. Reef area calculations for EQIP sites were accomplished in-situ for Smelt Brook Cove and measured by telemetry via aerial satellite imagery where possible (i.e. Ninigret Pond, southern sanctuary and Potter Pond). Mean area was calculated with observed measurements (n=15) and extrapolated across total number of reefs seeded. Locations of EQIP and TNC reefs were derived from a combination of direct observation to determine reef extent, satellite imagery, progress reports, and personal communication.

CHAPTER 3

RESULTS

PROGRAM INTRODUCTION AND SEEDING HISTORY

North Cape Restoration Program

The *North Cape* Restoration program aimed to address the natural resource injuries resulting from the release of 828,000 gallons of heating oil into Block Island Sound during the 1996 *North Cape* oil spill (DeAngelis *et al.* 2009). Following a legal settlement in 2000, the trustees established a Shellfish Restoration Program, implementing projects focusing on the enhancement of the northern quahog (*Mercenaria mercenaria*) and restoring bay scallop (*Argopecten irradians*) and eastern oyster (*Crassostrea virginica*) populations to Rhode Island waters. The goal of the shellfish restoration program was to restore lost wet-tissue biomass and lost ecological services due to the oil spill. Field efforts commenced in 2002 and were carried out through 2008. Oyster restoration components of the program focused on increasing the spawning stock biomass of *C. virginica* to areas of suitable habitat, ultimately aiming to increase recruitment to the population. Site locations were initially selected based on local benthic substrate, hydrodynamics, fishing history and presence and abundance of predators and diseases. Oyster larvae for the program were set on surf clam, *Spisula solidissima*, shell using remote setting techniques (Jones and Jones 1998, Kennedy 1996), raised in a nursery grow-out for one season (June – November) and seeded on unprepared or un-cultched sites (Hancock *et al.* 2004, 2006, 2007; DeAngelis *et al.* 2008). An exception to this was the season of 2008, in which oysters were set as singles and raised in an upweller for one season prior to seeding. A total of seven sites have been seeded since 2003 including: Saugatucket River, Narragansett; Smelt Brook Cove, South Kingstown; Bissel Deep, Bissel Channel and Bissel Cove Closed, North Kingstown; Spectacle Cove, Portsmouth and Potter Cove, Prudence Island (Figure 1, Appendix A-K). All sites are subtidal with a depth range between 0.2 – 2.0 meters at mean low tide. Within each site, oysters were seeded in a large contiguous area with mean site size of 2,733.8 m^2 ± 293.9 m^2 and range of 2,016 – 3,324 m^2 (Table 1). Average density of seeded oysters was 107 oysters m^{-2}. Over 5.4 million oysters were seeded over the course of the program, encompassing 13,699 m^2 or 4.0 acres (Table 2). All oysters seeded during the *North Cape* program were sourced from Muscongus Bay Hatchery, Bremen, ME and set on shell at the RI-DEM Coastal Fisheries Laboratory in Jerusalem, RI. Monitoring of restoration activities took place at most sites from 2004 – 2008 and 2011 – 2013.

Oyster Gardening for Restoration and Enhancement

Roger Williams University's Oyster Gardening for Restoration and Enhancement program (OGRE), aims to increase the spawning stock biomass of *C. virginica* in suitable habitats within Rhode Island waters, as well as promote the education and stewardship of our estuarine resources. The program is a cooperative effort between University scientists and citizens of the state, in which waterfront property owners maintain an oyster nursery off their docks or moorings, rearing viable oysters for restoration. All oyster broodstock for the program comes from a native Rhode Island line; originating from Blue Bill Cove, Portsmouth, RI and Greenhill Pond, Narragansett RI. Oysters are conditioned, spawned and remotely set on *S. solidissima* shell within the RWU hatchery. Oysters are then transported to OGRE volunteers who maintain the bivalves in Taylor floats during the nursery phase, for one summer prior to seeding in the fall of each year. The OGRE program began as a pilot project in 2006, enlisting the help of 18 volunteers, producing 54,000 oysters for restoration and has since grown to over 100 volunteers' state-wide, producing between 200,000 – 500,000 oysters annually. Since 2006 ten sites have been seeded: Jenny's Creek, Prudence Island; Bristol Harbor, Bristol; Town Pond, Portsmouth; Sandy Point, Greenwich; Bissel Cove, North Kingstown; Smelt Brook Cove, South Kingstown; Ninigret Pond, Charlestown; Quonochontaug Pond, Charlestown, Winnapaug Pond, Westerly and Great Salt Pond, Block Island (Figure 1, Appendix A-K). Oysters were seeded directly on un-cultched benthic substrate in each site with the exception of Town Pond. Within the Town Pond restoration site four rectangles (9 m x 20 m) were clutched with 10 cm of surf clam shell prior to seeding oysters. All sites with the exception of Town Pond are subtidal with a mean ranging from 0.0 – 2.0 meters at mean low tide. Sites were selected based on local benthic substrate, hydrodynamics, fishing history and presence and abundance of predators and diseases. In contrast to the *North Cape* program, oysters were seeded in multiple small beds within each site. Mean bed size is 514 m^2 with a range of 215 – 1,074 m^2 (Table 2). Mean density of seeded oysters at the outset of restoration was 786 oysters m^{-2}. Nearly 3.4 million oysters have been seeded over the course of the program encompassing 1.2 acres (Table 3). Monitoring of restoration sites seeded in the OGRE program took place between 2011 – 2013 and selected sites were also monitored in 2014 (i.e. Town Pond, Bissel Cove)

Environmental Quality and Incentives

The Environmental Quality Incentives Program, run by the Natural Resource Conservation Service (NCRS) began oyster restoration efforts in Rhode Island waters in 2008. The program ran from 2008 to 2010 and started again in 2015. This program aims to increase spawning stock biomass of oysters through the direct seeding of oyster spat on shell. Commercial aquaculturists were hired to nursery rear spat on shell for one season prior to seeding in designated restoration sites chosen by the Rhode Island Department of Environmental Management. Between 2008 and 2010 oysters were seeded in Jenny’s Creek, Prudence Island; Bissel Cove, North Kingstown; Smelt Brook Cove, South Kingstown; Ninigret Pond, Charlestown; Quonochontaug Pond, Charlestown, Winnapaug Pond, Westerly, Potter Pond, South Kingstown and Great Salt Pond, Block Island (Figure 1, Appendix A-K).

Prior to seeding, benthic substrate in all sites were cultched with surf clam shell. All sites are subtidal with depth range of 0.2 – 1.5 meters at mean low tide. Multiple high density oyster beds were created within each restoration site. Mean bed size seeded was 24 m^2 with a range from 13 – 52 m^2. Over seventeen million oysters have been seeded through EQIP efforts between 2008 and 2010 encompassing an estimated seeded area of 0.71 acres (Table 5).

The Nature Conservancy

The Rhode Island Chapter of The Nature Conservancy (TNC) commenced oyster restoration efforts in 2012 and employed a different approach from that of the *North Cape,* OGRE and EQIP restoration efforts. The Nature Conservancy aims to enhance remnant populations of oysters through increasing suitable settlement substrate. This assumes the bottleneck of the population exists not in viable broodstock and larval availability, rather a lack of appropriate substrate for successful settlement. Four 3 m x 24 m reefs were created in Foster's Cove and two 3 m x 24 m reefs were created at Grassy Point, Ninigret Pond (Figure 1, Appendix H), in 2012 with 17.2 yd^3 of steamed surf clam and oyster shell, encompassing a total reef area of 0.11 acres. All reefs were located in subtidal waters with a depth of less than 1 foot at mean low tide. In 2013 the reefs were repurposed with the addition of Oyster Castles®, a specialized manufactured concrete unit using a blend of proprietary material, placed on top of the 2012 shell. These reefs were monitored in the summer of 2015.

In 2015, eight reefs were built in the Ninigret Pond spawner sanctuary (Figure 1, Appendix G), using 131 tons of a mixture of steamed surf clam shell and recycled oyster shell. All reefs were subtidal with a mean depth of 0.75 m at low water. Mean reef size was 40.8 ± 5.0 m^2, with a range of 25 m^2 to 89 m^2, encompassing a total reef area of 0.08 acres. An estimated 38,700 spat on shell oysters were seeded on half the reefs with mean shell height of 28.1 ± 6.1 mm. Oyster seed for the project was sourced from Aquacultural Research Corporation (Dennis, MA).

Two 0.25 acre reefs, comprised of surf clam shell, were installed in the Quonochontaug Pond eastern spawner sanctuary in 2014 (Appendix I). No oysters were seeded on these reefs.

The combined efforts of the *North Cape*, OGRE, EQIP and TNC restoration programs have resulted in over 26 million seeded oysters on 6.6 acres within Rhode Island coastal waters (Figure 2, Table 8). The Natural Resource Conservation Service is currently compiling seeding data from the 2015 EQIP restoration efforts. Monitoring of the 2015 EQIP reefs is ongoing. Total number of oysters seeded and acres restored, reported herein, does not account for the 2015 EQIP restoration efforts (expected data availability, May 2017).

MONITORING RESULTS

Raw monitoring data of oyster performance within the Environmental

Quality Incentives Program between 2008 – 2014 does not exist. Raw monitoring data from The Nature Conservancy projects was not available. Due to unavailability of contiguous data from these projects (TNC and EQIP), they were excluded from calculations of restoration performance and cost-benefits. Performance of oysters within TNC, EQIP, OGRE and *North Cape* sites is assumed to be similar, as comparable practices were implemented (see discussion). Specifics of restoration efforts (i.e. area seeded and oysters planted) for TNC and EQIP programs was provided in this work, as it is important within the context of overall restoration efforts in the state. The following data and analysis presented is from the *North Cape* (2003 – 2008) and OGRE (2006 – 2014) efforts.

Estimated Population and Length Distribution

Total number of oysters in all *North Cape* sites in the fall of 2013 was 8,439 ± 1,922 oysters with a site range of 912 – 4,638 oysters. As of 2013, Bissel Cove has the highest estimated population of oysters (4,639 ± 630), followed by Smelt Brook Cove (1,732 ± 285), Spectacle Cove (1156.9 ± 504) and Potter Cove (912 ± 603) (Table 1). Total number of oysters in all OGRE sites in the fall of 2013 was 211,722 ± 29,453. Bissel Cove had the highest estimated population of oysters (68,445 ± 4,755), followed by Smelt Brook Cove (63,622 ± 2,986), Town Pond (55,363 ± 18,199), Quonochontaug Pond (22,357 ± 1,943) and Jenny's Creek (1,935 ± 571) (Table 2).

Mean valve height (umbo to lip) of live oysters was 106.5 ± 3.4 mm across all *North Cape* sites and 96.3 ± 1.0 mm across all OGRE sites during 2013 monitoring events. Tracking growth of single cohorts within Bissel Cove, Smelt Brook Cove and Quonochontaug Pond from 2011 to 2013 revealed a mean annual growth of 32 ± 0.6 mm (shell height). Experimental sites were seeded in 2011 within the same cohort and tracked until 2014. First year growth post planting was significantly different between sites ($p<0.0001$) (Figure 3). First year shell height was largest in Bissel Cove (59.6 ± 1.6 mm) followed by Smelt Brook Cove (44.9 ± 1.5 mm) and Quonochontaug Pond (40.1 ± 1.3 mm). Second year shell height was significantly larger in Bissel Cove (86.3 ± 2.1 mm) compared to Smelt Brook Cove (78.2 ± 2.4 mm) ($p<0.0001$) and Quonochontaug Pond (71.8 ± 1.8 mm) ($p=0.354$). Third year shell height was significantly larger in Bissel Cove (118 ± 3.0 mm) compared to Smelt Brook Cove (108.4 ± 2.7 mm) ($p=0.0152$). Year three growth data is not available from Quonochontaug Pond.

Survival

Based on data from the experimental reefs where single cohorts were tracked, mean first year survival was 32 ± 23%. Including available data from all sites and years, mean first year survival drops to 21.9 ± 2.2%. Using data from all years, first year survival was highest in the Saugatucket River (32.8 ± 12.6%) followed by Bissel Cove (26.8 ± 5.7%), Smelt Brook Cove (24.5 ± 9.0%), Quonochontaug Pond (19.4 ± 3.5 %), Potter Cove (13.8 ± 3.4%) and Spectacle Cove (9.7 ± 1.4%) (Figure 4). Differences in mean first year survival were not significant between sites (p=0.2167)

Year two plus survival varied greatly between sites and years with a mean yr. 2+ survival across all sites of 55 ± 5%. Year 2+ survival was highest within Town Pond (74.2 ± 20.8%) followed by Quonochontaug Pond (69.5 ± 22.5%), Bissel Cove (68.2 ±12.3%), Saugatucket River (67.8 ± 22.6%), Spectacle Cove (63.7 ± 17.7%), Smelt Brook Cove (58 ± 4.4%), Jenny's Creek (43 ± 11%) and Potter Cove (39.2 ± 15.4%) (Figure 5). Differences in mean survival in year 2+ were not significant (p=0.6319). Length distribution of oyster boxes reveals 40% of overall mortality occurs between 5 – 50 mm valve heights, during the first year post planting. Twenty seven percent of overall mortality occurs between 80 – 120 mm height (Figure 6a), which will be discussed later. Excluding first year mortality in distribution plots of oyster boxes, 75% of mortality occurs between 80 – 120 mm valve height, as would be expected in areas of high disease pressure (Figure 6b).

Recruitment

Monitoring relative recruitment via artificial spat collectors yielded positive results between 2011 and 2012 with five of the seven water bodies studied showing recruitment events. Between early June and October of each year, between 2011 – 2013, five spat collectors were deployed in the vicinity of Bissel Cove; seven spat collectors were deployed in Point Judith Pond; four spat collectors were deployed in Potter Cove; seven spat collectors were deployed in Quonochontaug Pond; four spat collectors were deployed in

Spectacle Cove; five spat collectors were deployed in Town Pond and four spat collectors were deployed in Jenny's Creek. Recruitment to each site was measured as mean number of spat per bag. Mean recruitment across all sites and years was 0.89 ± 0.62 oysters per bag. Mean recruitment for each site across all years was highest in Bissel Cove (2.10 ± 1.84) followed by, Spectacle Cove (1.50 ± 0.87), Town Pond (1.40 ± 0.58), Potter Cove (0.81 ± 0.56), Point Judith Pond (0.13 ± 0.25), Quonochontaug Pond (0.07 ± 0.07) and Jenny's Creek (0.0) (Figure 7).

Artificial spat collectors can demonstrate the relative abundance and settlement distribution pattern, but do not represent actual recruitment rates on the bottom (Brumbaugh *et al.* 2006). A more appropriate measure of recruitment rates on the bottom can be calculated from the number of new recruits to the actual restoration sites, derived from density monitoring. This method revealed a mean density of recruitment across all sites and years of 0.83 ± 0.23 oysters m^{-2}, representing 3.2 % of the total population of oysters in all monitored sites. The highest mean density of recruits across all years was Bissel Cove (2.63 ± 0.22) followed by Town Pond (1.75 ± 0.4), Potter Cove (0.56 ± 0.03), Quonochontaug Pond (0.55 ± 0.22), Spectacle Cove (0.25 ± 0.05), Smelt Brook Cove (0.11 ± 0.03) and Jenny's Creek (0.1 ± 0.1) (Figure 8). Recruitment was significantly higher in Town Pond compared to Jenny's Creek (p=0.042). Recruitment in all other sites was not significantly different.

Disease Prevalence

Disease testing was completed within the *North Cape* restoration sites from 2004 to 2008 and again from 2011 to 2013 and within all OGRE sites between 2011 and 2013 (Table 7a-b). Oysters sampled had a mean valve height of 86.9 ± 2.6 mm and a mean mass of 78.9 ± 6.9 g. Test results revealed high prevalence of *Perkinsus marinus* in five of the ten sites sampled on an annual basis. Mean prevalence of dermo and intensity across all years was highest in Jenny's Creek (98.7 ± 1.3%, intensity 2.5 ± 0.2), followed by Smelt Brook Cove (95.3 ± 1.7%, intensity 3.1 ± 0.2), Saugatucket River (90 ± 8.9%, intensity 3.2 ± 0.4), Bissel Cove (81.3 ± 9.2%, intensity 2.4 ± 0.4), Ninigret Pond (80 ± 16.2%, intensity 1.5 ± .3), Spectacle Cove (64.8 ± 15.4%, intensity 3.2 ± 0.5), Potter Cove (32.5 ± 18.3%, intensity 1.0 ± 0.4), Town Pond (30.7 ± 19%, intensity 0.7 ± 0.4), Quonochontaug Pond (17.3 ± 17.3%, intensity 0.3 ± 0.3) and Great Salt Pond (12 ± 10.1%, intensity 0.9 ± 0.6) (Figure 9). Mean percent prevalence of *P. marinus* was significantly different across sites (p=0.0007). Dermo prevalence in Smelt Brook Cove was significantly higher than Quonochontaug Pond (p=0.0202), Great Salt Pond (p=0.0472) and Potter Cove (p=0.0326). Dermo was significantly higher in the Saugatucket River compared to Quonochontaug Pond (0.0406). Regression analysis showed no correlation between presence of dermo and density of oysters within Rhode Island restoration sites (r^2=0.034).

Other diseases appear to have very little impact on the oysters within the restoration sites monitored. *Haplosporidium nelsoni* (MSX) was only found at two sites (Spectacle Cove and Town Pond), both of which had a prevalence of 4%. *Haplosporidium costale* (SSO) was not found in any sites. Trematode infections within oyster tissue were found at the following sites, Great Salt Pond with a prevalence of 4%; Jenny's Creek with a prevalence of 8% and Spectacle Cove with a prevalence of 8%. *Roseovarious crassostreae* (ROD) was not monitored within the restoration sites.

Sustainability Index

Ultimately the success of restoration hinges on the ability of the reef to become self-sustaining post planting. Sustainability was assed based on the level of natural mortality (year 2+) weighed against recruitment. A sustainability index (SI) was calculated for each site each year post restoration implementation. A negative SI represents a reef in population decline, where mortality outpaces recruitment. An SI of zero represents a stable population, while a positive SI represents population growth. SI was negative or zero within all years and sites where data was available. Mean SI, across all years, was highest in Town Pond (-23.02 ± 26.36), followed by Quonochontaug Pond (-30.01 ± 27.89), Bissel Cove (-31.10 ± 12.59), Saugatucket River (-32.00 ± 32.00), Spectacle Cove (-36.30 ± 17.74), Smelt Brook Cove (-41.94 ± 4.56), Jenny's Creek (-57.00 ± 11.00) and Potter Cove (-60.80 ± 15.37) (Figure 10). The mean sustainability index across sites was not significantly different (p=0.6022).

Cost-Benefit

Mean costs of restoration per acre of restored reef (OGRE and *North Cape* Programs, 2004 – 2014) was \$71,366 ± \$8,592. The lack of recruitment to reefs hinders sustainability, prompting the need for maintenance seeding to preserve adequate oyster density to provide desired ecosystem services. Mortality estimates, indicate restoration reefs need to be reseeded every six years to maintain reef integrity, causing a stepwise linear slope of cumulative restoration costs (Figure 11). Maintaining one acre of oyster reef over 50 years within the current confines of our restoration sites is estimated at a value of \$642,295. Estimated annual value of nitrogen removal per acre of oyster reef was \$1,639. Estimated annual value of submerged aquatic vegetation enhancement based on the creation of 0.005 hectares of SAV per hectare of oyster reef was \$1,046. Estimated annual value of fisheries production was \$1,669 per acre of oyster reef. Total estimated annual value of ecosystem services provided by one acre of oyster reef was calculated at \$4,353 (Table 9). Cumulative value of ecosystem services provided by one acre of oyster reef over 50 years was estimated at \$209,917. Modeling cumulative costs of restoration with cumulative value of associated ecosystem services indicates a net negative value of restoration from time of oyster seeding to 50 years post planting. The slope of cumulative value of ecosystem services is lower than cumulative value of restoration causing an increase in monetary deficit each year restoration is continued (Figure 11). The smallest monetary deficit (cumulative cost of restoration subtracted by cumulative value of ecosystem services) occurs during year six after the initial seeding event (\$45,495). After

25 years of restoration the deficit increases to $251,802 and after 50 years of restoration the deficit increases to $432,378.

CHAPTER 4

DISCUSSION

Rhode Island has long recognized the socioeconomic and ecological importance oysters represent to local communities and have invested substantial effort in restoring lost stocks to Narragansett Bay and coastal salt ponds. Since 2003, four distinct programs have seeded over 26 million oysters, encompassing 6.6 acres into Rhode Island coastal waters. Analysis of project performance and cost-benefit analysis of restoration efforts was undertaken using eleven years of data compiled from the *North Cape* and OGRE programs. Monitoring data for the Environmental Quality and Incentives Program and The Nature Conservancy's restoration efforts was unavailable for this analysis. In most cases, TNC and EQIP efforts were sited in close proximity to *North Cape* and OGRE restoration with a common line of oysters and similar restoration practices, including: size of oysters at planting, height of restored reef, timing of planting and density of oysters seeded. Due to the lack of differences in restoration practices and locations, it is assumed TNC and EQIP restoration programs have performed similar to *North Cape* and OGRE programs. This assumption has been backed by qualitative observation of EQIP reefs and personal communication with TNC staff (Pers. Comm. Sara Coleman, Bryan DeAngelis). The compilation and analysis of monitoring results indicates acceptable survival and excellent growth of oysters post planting, however, limited recruitment hinders overall project success.

Assessment of size structure of all boxes within restoration sites reveals high mortality in the first year post seeding, with a mean survival across all years and sites of 22% (oyster valve length between 10 – 50 mm). High rates of mortality in the first year is expected and typically driven by predation and sedimentation. Oysters set on all sides of the setting media; subsequently, high mortality occurs in the act of planting, as oysters on the bottom of the media can become smothered by sediment. Pre-seeding oyster height was targeted at 20 mm to mitigate predation pressure, however, a large variance in the size of oysters during seeding events has been observed (Hancock *et al.* 2004, 2006, 2007; DeAngelis *et al.* 2008), leading to increased predator pressure on oysters which have not reached a size of predator refuge. Initial density of remote set oysters, within Rhode Island, on media (i.e. surf clam or oyster shell) is typically between 10 – 200 oysters per shell leading to high inter-specific competition. After two to three years of growth post-seeding oyster density ranges between 0 and 20 oysters per shell media (Griffin, unpublished data). The precipitous drop in oyster density per shell media is largely a factor of physical space limitation. Observations of mortality in the first year post seeding also include sedimentation, as oysters can be smothered in areas of high sediment deposition and shell subsidence. Observed first year mortality on Rhode Island oyster reefs does not appear out of the ordinary, as year one morality of 20 – 30 % has been observed in other regional oyster restoration efforts (Griffin 2015).

Excluding first year mortality, highest mortality is observed in individuals with a shell length between 80 – 120 mm, which is indicative of mortality caused by *Perkinsus marinus.* Levels of *P. marinus* infection build with age, as does associated percent mortality (Encomio *et al.* 2005), explaining mortality of the older cohorts. Survival of year 2+ oysters varied greatly between sites and within sites between years. Mortality rates of oysters can vary across space and time due to differences in habitat quality, disease and predator pressure. Part of the observed variance of mortality between years is undoubtedly due to sampling error. Mortality was based on the change of oyster density observed during annual sampling events. Oyster density on restored reefs varies greatly due to the nature of seeding, which often involves dumping totes of spat on shell off the side of boat in a predetermined area; a less than precise operation. Limited recruitment to restored reefs does not allow oyster density to become homogeneous across the site as time passes. Haphazard quadrat sampling of reefs was employed during surveys, keeping the sample size high and consistent between years to reduce variance; however, the large standard errors associated with observed oyster densities greatly effects mortality estimates and confounds analysis comparing mortality across sites and years leading to non-significant results. Monitoring of oyster restoration efforts in the Chesapeake Bay has demonstrated year 2+ survival rates between 30 - 70% (Mann and Powell 2007). We observed a mean annual survival of 55% with a range from 25% to 100%, which appears to be on par with highly intensive efforts in the Chesapeake Bay.

Perkinsus marinus has been observed in eastern oysters for over 50 years along the eastern and southeastern seaboards of the United States (Smolowitz 2013). Andamari *et al*. (1996) found no presence of *P. marinus* in oysters within four distinct locations (Pawcatuck River, Narrow River and Charlestown Pond) between 1991 and 1992. Mareiro *et al.* (2001) showed high prevalence and intensity of *P. marinus* in wild and cultured oyster populations throughout Narragansett Bay and Rhode Island salt ponds in 1998, suggesting dermo made its presence in Rhode Island waters between 1992 and 1998. *Perkinsus marinus* is now fairly ubiquitous within wild, restored, and cultured oysters in Rhode Island. Mean dermo prevalence between 2003 and 2014 was over 60% within six of ten monitored restoration sites. Markey and Gómez-Chiarri (2007) found similar results within five wild sites sampled in Narragansett Bay and the Coastal Ponds (Bissel Cove, Spectacle Cove, Saugatucket River, Narrow River, Great Salt Pond) between 1998 and 2007, where disease testing showed dermo prevalence ranging between 62 – 100% with the exception of Great Salt Pond (7% in 2001).

Prevalence of dermo is highly variable and directly correlates with temperature and salinity. Prevalence and intensity are generally highest in salinities greater than 12 ppt. Temperature also regulates the disease, as the prevalence and intensity oscillates with seasonal fluctuations in water temperature. Maximum prevalence and intensity generally lags 1-2 months behind maximum summer water temperature and minimum prevalence and intensity lags 1-2 months behind minimum winter water temperatures (Burreson and Ragone Calvo 1996). Prevalence and intensity of dermo was similar across most sites with the exception of Smelt Brook Cove and Saugatucket River which exhibited significantly higher dermo rates compared to Potter Cove, Great Salt Pond, and Quonochontaug Pond. There is not enough variability in salinity or water temperature within the current restoration sites to influence the presence of dermo. All restoration sites with the exception of Saugatucket River experience salinities between 22 - 35 ppt depending on tidal cycle and amount of precipitation. Salinity at Saugatucket River varies between 4 - 24 ppt depending on tidal cycle and rainfall. The short pulses of low salinity in Saugatucket River are apparently not sufficient to extricate *P. marinus*, as the site has consistently high infection rates. Dermo is transmitted directly between oysters, as new infections are acquired as oysters feed and the parasite infects its host though gut epithelial tissue (Villalba *et al.* 2004, Bushek *et al.* 2002). This mechanism of transmission can cause densely populated oyster beds to be particularly susceptible to high levels of dermo. Regression analysis showed no correlation between presence of dermo and density of oysters within Rhode Island restoration sites.

Two broodstock lines were used for restoration sites which were assessed for disease; Muscongus Bay's selected hatchery line (*North Cape*) and wild stock from Blue Bill Cove, Portsmouth and Green Hill Pond, Narragansett (OGRE). The wild stock was chosen for potential disease resistance, as it resides naturally in areas of high dermo prevalence.

Restoration from both programs (*North Cape* and OGRE), encompassing both oyster lines, were undertaken in close proximity to one another in Bissel Cove and Smelt Brook Cove. Dermo prevalence was initially lower in the wild line compared to the Muscongus line in Bissel Cove (100% versus 32% after two years post planting) but then climbed to 100% in the wild line in subsequent years. Both lines had similar dermo infection rates in Smelt Brook Cove and similar mortality rates were observed in both lines across sites. These data suggest using a native oyster line did not have appreciable effects on disease prevalence and survival, however, this should be considered preliminary. Disease testing on year two cohorts did not take place during the same years for both lines (wild and Muscongus), therefore, a direct comparison of performance is difficult as disease can be highly ephemeral (Pers. Comm. R. Smolowitz). Gomez-Chiarri *et al.* (2010) compared survival of three different lines of oysters; disease resistant NEH, a local stock from Green Hill Pond (GHP) and a hybrid cross between the two (HYB) on ten commercial farms in Rhode Island between 2008 and 2010. Their data showed significantly higher survival of the NEH line (11 to 76%) compared to the local GPH and HBY stocks (2 – 62%) depending on the farm (Gomez-Chiarri *et al.* 2010). Impacts of dermo have clear and wide ranging effects on restoration efforts stemming from the associated mortality of older cohorts, thus, reducing spawning stock biomass, filtration capacity and associated ecological benefits. Further investigation on the efficacy of using native or modified lines to reduce disease pressure and increase survival within restoration settings is warranted.

No restoration program can become self-sustaining without adequate recruitment. While recruitment to spat collectors was low it is encouraging considering the same sites had been monitored with similar methods from 2004 to 2008 without documenting a single recruitment event. Due to sparse recruitment to collectors, spatial and temporal settlement events were not detectable. Recruitment to restored reefs was modest with less than one recruit m^{-2} on an annual basis. Bissel Cove and Town Pond had consistently higher recruitment rates compared to all other sites but lacked significance due to high variance.

Self-sustaining oyster reefs need a positive net balance of shell aggregate and accretion to allow for suitable settlement substrate. Many factors affect the rate of shell life including water chemistry, sulfide rich substrates, the presence of various sponges, polychaetes, mollusks and some algae (Pafford 1988). The half-life of oyster shell varies between 3 – 10 years depending on the given environment (Powell *et al.* 2006). Although the observed recruitment to Rhode Island reefs is a positive sign, a consistent set over multiple years has not been observed. History in Rhode Island indicates large recruitment events such as that observed in the 1990s and the modest event in 2010 occur on a decadal or multi-decadal pattern. The sporadic nature of recruitment events in Rhode Island leads to the loss of shell habitat and hinders reef building efforts. To overcome the hurdle of limited settlement substrate the Nature Conservancy has built fifteen reefs in Ninigret Pond with a combination of oyster/surfclam shell and Oyster Castles® and two reefs in

Quonochontaug Pond with surfclam shell. Results of monitoring indicated no or low recruitment in all but one location (Foster's Cove) (Pers. Comm. Sara Coleman), which is known for consistent oyster sets.

The cause of low recruitment to Rhode Island oyster restoration sites is not fully understood. DeAngelis *et al.* (2008) monitored temporal development of oyster gonads and estimates of larvae within the water column at two restoration sites in Point Judith Pond (Smelt Brook Cove and Saugatucket River) during the spawning season of 2008. Both sites were monitored weekly for oyster condition index and twice weekly for presence of veliger stage oyster larvae between June and September 2008. Gonadal development and larval abundance indicated regular and distinct periods of veliger stage larvae in the water column (Saugatucket River max = 1350 $\pm$ 340 m^{-3}; Smelt Brook Cove max = 8,575 $\pm$ 4,400 m^{-3}) with a peak in mid-July. Presence of veliger stage larvae and lack of recruitment suggest that the recruitment bottleneck exists between the free-swimming stage and recruitment of spat. This bottleneck may be driven by a myriad of factors including predation, disease, siltation or inadequate settlement substrate, or larval displacement greater than the study area (Dickie 1955, Hancock 1973, Wolf 1988).

When put in the context of historical oyster landings in Rhode Island salt ponds, appropriate salinity regimes for successful oyster set, is a particularly compelling argument. Native oysters were abundant in all Rhode Island salt ponds prior to the construction of permanent breachways. Post-breaching, oyster populations began to dwindle and reliable sets were only observed in the back coves and along the edges of the pond where freshwater inundation lowered salinities (Lee 1980). Lower survival of oyster broodstock and settled spat in higher salinity waters is likely a function of increased predation, as higher salinity water houses a myriad of predators not found in less saline environments (e.g. starfish, whelk, mud crabs, Asian shore crabs, ctenophores). Furthermore, dermo MSX, SSO and ROD are more common in higher salinity environments. Few locations in Rhode Island's coastal waters support a consistent wild oyster population. All of these locations (i.e. Green Hill Pond, Narragansett; Narrow River, Saunderstown; Seekonk River, Providence and Quicksand Pond, Little Compton) are located in low salinity environments. During small pulse events of oyster recruitment, as noted in 2010, recruitment was highest along the fringes of salt ponds and back coves of Narragansett Bay where ground water inundation was observed (personal observation, Griffin). Recruitment events and sustainable oyster populations have been linked to low salinity environments outside of Rhode Island. Morality and river flow estimates in the James River, VA have been recorded since 1994. Data shows, in years of low flow, oyster mortality rates exceed 70% and recruitment was hindered (Mann and Powell 2007). Extant subtidal oyster communities in the Chesapeake Bay are limited to upper sub-estuaries where lower salinity regimes exist (Mann and Powell 2007). Tolerated salinity ranges for oyster larval rearing is widely reported between 3 and 33 ppt (Calabrese and Davis 1970, Amemiya 1926, Carriker 1951, Davis 1958). Optimal salinity ranges for larval rearing has been reported between 17 and 29 ppt (Calabrese and Davis 1970, MacInnes and Calabrese 1979, Amemiya 1926). These reported values have a wide range and encompass studies from broad geographic regions. These values do not take post settlement survival into account so miss the mortality link which is pertinent in the context of restoration. Further knowledge is needed to quantify how salinity relates to post-settlement survival on Rhode Island oyster reefs. With the current body of oyster restoration science, we are unable to pinpoint what is causing the recruitment bottleneck to our local reefs.

Despite the mechanisms of recruitment failure, oyster settlement is consistently outpaced by natural mortality in all monitored restoration sites in Rhode Island, leading to a decline of the population once seeding has ceased. As a result of disease and subsequent mortality, our data suggest we lose a functional oyster reef, in-terms of ecosystem services, within six years post seeding. This has clear and wide-ranging implications. The loss of ecosystem services stems from the reduction of biomass, thus reducing total filtration capacity (loss of nitrogen removal and submerged aquatic vegetation enhancement) as well as the negative impact on fisheries production through the loss of biogenic structure of the reef. Remnant populations may persist for ten years post seeding as we observed in both Potter Cove and Spectacle Cove, but both of these sites had densities of less than 1 oyster m^{-2}. While this wouldn't provide much, in terms of ecosystem services, they might yet contribute to the total spawning stock biomass of oysters within our coastal waters. If we further assume that these remnant populations have been disease challenged and survived, they may contain some as yet unknown aspect of disease resistance to offer to future populations.

The cost-benefit model indicates Rhode Island oyster restoration is not equitable in terms of ecosystem services provided, as the cost of restoration is higher than the cumulative value of ecosystem services provided by the reef. Mortality outpaces recruitment within all restoration sites, prompting the need for maintenance seeding to preserve a functioning reef in terms of ecosystem services; thus, the cost of restoration is not fixed and the cumulative cost of restoration rises at a steeper slope than the cumulative value of ecosystems services. It should be noted the ecosystem services described herein (nitrogen removal, fish production, and submerged aquatic vegetation enhancement) were calculated using data from estuaries in the southeastern United States. Due to differences in temperature, sediment chemistry, fish assemblages and oyster productivity, we cannot assume reported values are directly comparable to oyster reefs in Rhode Island. It is, however, safe to assume reefs located in the southeastern United States perform at a higher level, in terms of production and nitrogen removal, than those found in New England waters; thus, fitting our data to this model would overestimate the ecological value of reefs. This assumption is based on warmer water temperatures in southern estuaries compared to Rhode Island, leading to a longer filtration season and higher levels of de-nitrification coupled with higher fish productivity.

Grabowski *et al*. (2012) predict the initial investment of restoring one acre of oyster reef will be recouped through ecosystem services within 10 years of seeding. His model is based on self-sustaining reefs which are common to South Carolina but not Rhode Island. The lack of recruitment driving the need for maintenance seeding in Rhode Island tips the balance to net negative. If our reefs were self-sustaining, initial restoration investments would be recouped through ecosystems services in 17 years with an annual capital gain of $4,200 thereafter.

Town Pond and Bissel Cove consistently performed higher than other sites in terms of growth, survival and recruitment but sustainability indices did not score significantly higher compared to other sites; likely due to the stochastic nature of the data. Specifically why Town Pond and Bissel Cove have excelled in terms of recruitment compared to other locations is unknown; however, we suspect that a combination of suitable settlement substrate, larval retention and fresh water inflow are among the responsible factors. Rhode Island's highest performing restoration sites do not come close to maintaining a self-sustaining population, suggesting a re-prioritization of restoration goals (i.e. conducting restoration without the end goal of population recovery) or adaptively managing our restoration practices is needed.

CHAPTER 5

CONCLUSIONS

This study revealed the non-self-sustaining nature of oyster restoration sites within Rhode Island and the lack of a positive cost-benefit in terms of ecosystem services. Proper site selection is critical to successful restoration. Recruitment limitations and disease prevalence are currently thought to be the governing factors in the success of Rhode Island oyster restoration. It is suggested practitioners closely assess recruitment patterns and levels of disease within the body of water of interest prior to undergoing restoration efforts. In addition to recruitment monitoring, practitioners should assure adequate settlement substrate is available within the site or in the near vicinity (e.g. cobble substrate, boulders, rip-rap etc.). There is currently no silver bullet to address the problem of poor recruitment. Historically Narragansett Bay does not receive continuous and heavy sets of oysters. With the exception of a very limited number of coves and rivers, large recruitment events appear to occur on a decadal or multi-decadal pattern, thus, restoration sites will require maintenance seeding. At present, the Rhode Island Department of Health prohibits restoration from occurring within water that is prohibited to shellfishing. While the “attractive nuisance” aspect of this approach is admirable, this has the effect of blocking the implementation of restoration in sites with more appropriate salinity regimes for optimal recruitment (i.e. 12 –

20 ppt) or with known reliable recruitment (e.g. Narrow River, Green Hill Pond, Quicksand Pond, and Seekonk River).

The science community needs to continue to study recruitment and mortality patterns as well as optimal genetic oyster lines for restoration within Rhode Island to better understand how we can maximize ecosystem services from our investment. Self-sustaining populations of oysters may never be realized within the current framework of oyster restoration within Rhode Island. While this sounds dire, it is not necessarily so – depending on how one views the problem. As our knowledge base increases, the restoration community in Rhode Island has the where-with-all to produce outstanding restoration sites leading to abundant ecosystem and community services within the life time of the reef. This may require maintaining the current level of restoration through maintenance seeding leading to a net revenue loss; which begets a social question of the willingness to pay for such restoration by the citizens of our State.

TABLES

Table 1. Results of monitoring *North Cape* restoration sites in 2013 – the last year of full monitoring. Saugatucket River, Bissel Cove Deep and Bissel Cove Channel were not monitored in 2013 due to negligible population of oysters.

Site	Mean No. Alive (m^{-2})	SE	Seeded Area (m^{-2})	Estimated Total Live	SE	Mean No. Recruits (m^{-2})	SE	% Recruits of Total Estimated Population
Saugatucket River	-	-	2,048	-	-	-	-	-
Smelt Brook Cove	0.9	0.2	2,016	1,732	285	0.1	0.1	8.5
Spectacle Cove	0.3	0.2	3,317	1,157	504	0.2	0.0	0.6
Potter Cove	0.3	0.1	3,324	912	283	0.2	0.0	0.7
Bissel Cove Deep	-	-	2,520	-	-	-	-	-
Bissel Cove Channel	-	-	1,780	-	-	-	-	-
Bissel Cove R2/R3	0.3	0.2	2,964	4,639	630	0.3	0.1	16.6

Table 2. Results of monitoring Oyster Gardening for Restoration and Enhancement sites in 2013 – the last year of full monitoring. Ninigret and Great Salt were not monitored in 2013.

Site	Mean No. Alive (m^{-2})	SE	Seeded Area (m^{-2})	Estimated Total Live	SE	Mean No. Recruits (m^{-2})	SE	% Recruits of total Estimated
Jenny's Creek	6	2	299	1,935	571	0.00	0.00	0.00
Quonochontaug Pond	226	6	500	22,357	1,943	0.39	0.05	0.88
Bissel Cove	134	25	503	68,445	4,755	5.79	0.38	4.25
Town Pond	57	14	1,074	55,363	18,199	0.84	0.52	1.62
Smelt Brook Cove	102	5	742	63,622	3,984	0.27	0.03	0.31
Ninigret Pond	-	-	266	-	-	-	-	-
Great Salt Pond	-	-	205	-	-	-	-	-

Table 3. Estimated number of oysters seeded by site and year during the *North Cape* Restoration Program.

Site	2003	2004	2005	2006	2007*	2008	TOTAL
Saugatucket River	48,700	137,400	272,800	575,642		276,115	1,310,657
Smelt Brook Cove	114,400	86,900	372,900	425,600		276,115	1,275,915
Bissel Cove Deep		137,400					137,400
Bissel Cove Channel	112,400	137,400					249,800
Bissel Cove Channel R1				439,362			439,362
Bissel Cove Channel R2						552,231	552,231
Spectacle Cove	96,600		361,200	222,389			680,189
Potter Cove	140,800		370,900	288,389			800,089
TOTAL	*512,900*	*499,100*	*1,377,800*	*1,951,382*		*1,104,461*	*5,445,643*

* Oysters were not seeded in 2007 due to prioritizing resources on scallop restoration.

Table 4. Estimated number of oysters seeded by site and year during the Oyster Gardening for Restoration program.

Site	2006	2007	2008	2009	2010	2011	2012	2013	2014	TOTAL
Jenny's Creek	27,000	128,000	286,000							441,000
Bristol Harbor	27,000	102,000	95,000							224,000
Town Pond			143,000	144,000	308,000	135,000	94,000	116,000	70,000	1,010,000
Sandy Point		56,000								56,000
Bissel Cove		40,000	57,000	40,000		84,000	74,000	93,000	27,000	415,000
Smelt Brook Cove		52,000	48,000	48,000	49,000	50,000	21,000	38,000	35,000	341,000
Quonochontaug Pond		52,000	57,000	48,000	63,000	50,000	41,000	116,000	42,000	469,000
Ninigret Pond			38,000	40,000	35,000	41,000	34,000	58,000		246,000
Winnapaug Pond									8,000	8,000
Great Salt Pond			9,500	8,000	21,000	45,000	21,000	29,000		133,500
TOTAL	*54,000*	*430,000*	*733,500*	*328,000*	*476,000*	*405,000*	*285,000*	*450,000*	*182,000*	*3,343,500*

Table 5. Estimated number of oysters seeded by site and year during the Environmental Quality and Incentives Program.

Site	2008	2009	2010	TOTAL
Jenny's Creek	79,560	145,869	279,372	504,801
Bissel Cove	499,140	2,466,672	703,581	3,669,393
Potters Pond		666,666	1,013,856	1,680,522
Smelt Brook Cove	378,000			378,000
Ninigret Pond	1,835,280	5,478,097	1,630,219	8,943,596
Quonochontogue Pond		439,178	593,861	1,033,039
Winapaug Pond		626,971	291,983	918,954
Great Salt Pond		336,142	57,253	393,395
TOTAL	*2,791,980*	*10,159,595*	*4,570,125*	*17,521,700*

Table 6. Estimated number of oysters seeded by site and year during The Nature Conservancy restoration efforts.

Site	2015
Ninigret Pond	38,700
TOTAL	*38,700*

Figure 7a. Prevalence and intensity of *Perkinsus marinus* within *North Cape* restoration sites from 2004 – 2013.

Site	2004		2005		2006		2007		2008		2011		2012		2013	
	% Prevalence	Makin Index	% Prevalence	Makin Index	% Prevalence	Makin Index	% Prevalence	Makin Index	% Prevalence	Makin Index	% Prevalence	Makin Index	% Prevalence	Makin Index	% Prevalence	Makin Index
Saugatucket River	28	2	100	4	100	5	100	2	100	4	96	2.8	96	3.3	100	2.5
Smelt Brook Cove	86	3	100	3	100	4	92	2	100	3	100	3.8	100	3.4	NA	NA
Bissel Channel	0	0	11	1	10	1	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Bissel Closed	NA	NA	NA	NA	NA	NA	79	1	100	3	100	4	100	3.1	87	3.1
Spectacle Cove	13	1	60	1	40	1	100	2	NA	NA	0	0	0	0	76	0.7
Potter Cove	14	1	24	1	0	0	92	1	NA	NA	25	0.4	60	2.4	NA	NA

Figure 7b. Prevalence and intensity of *Perkinsus marinus* within Oyster Gardening for Restoration and Enhancement Sites from 2011 – 2013.

Site	2011		2012		2103	
	% Prevalence	Makin Index	% Prevalence	Makin Index	% Prevalence	Makin Index
Town Pond	20	0.1	4	0.5	68	1.5
Bissel Cove	32	0.9	52	2.8	100	1.1
Quonnie Pond	0	0.0	0	0	52	0.88
Smelt Brook Cove	100	3.1	88	2.8	NA	NA
Jenny's Creek	96	2.1	100	2.7	100	2.8
Great Salt Pond	0	0.0	4	2	32	0.7
Ninigret Pond	48	0.9	92	1.9	100	1.6

Table 8. Total estimated number of oysters seeded by site between 2003 and 2015. All programs combined.

Site	Estimated No. Oyster seeded
Sandy Point	56,000
Bristol Harbor	224,000
Great Salt Pond	526,895
Spectacle Cove	680,189
Potter cove	800,089
Winnapaug Pond	926,954
Jenny's Creek	945,801
Town Pond	1,010,000
Saugatucket River	1,310,657
Quonochontaug Pond	1,502,039
Potter pond	1,680,522
Smelt Brook Cove	1,994,915
Bissel Cove	5,463,186
Ninigret Pond	9,228,296
TOTAL	*26,310,843*

Table 9. Total estimated value of ecosystem services proved by oyster reefs per acre and mean costs of restoration. Ecosystem service values adapted from Grabowski *et al.* (2012). Cost of restoration per acre represent the mean operating cost of *North Cape* and OGRE programs from 2003 to 2014 to maintain one acre of oyster reef.

Ecosystem Service	Annual per acre value	
Fisheries production	$	1,669
Nitrogen removal	$	1,639
SAV enhancement	$	1,046
TOTAL	*$*	*4,353*
Restoration Implementation	Per acre cost	
Mean	$	71,366.12
se	$	8,591.58

Figure 1. Location of oyster restoration sites by program between 2003 and 2015.

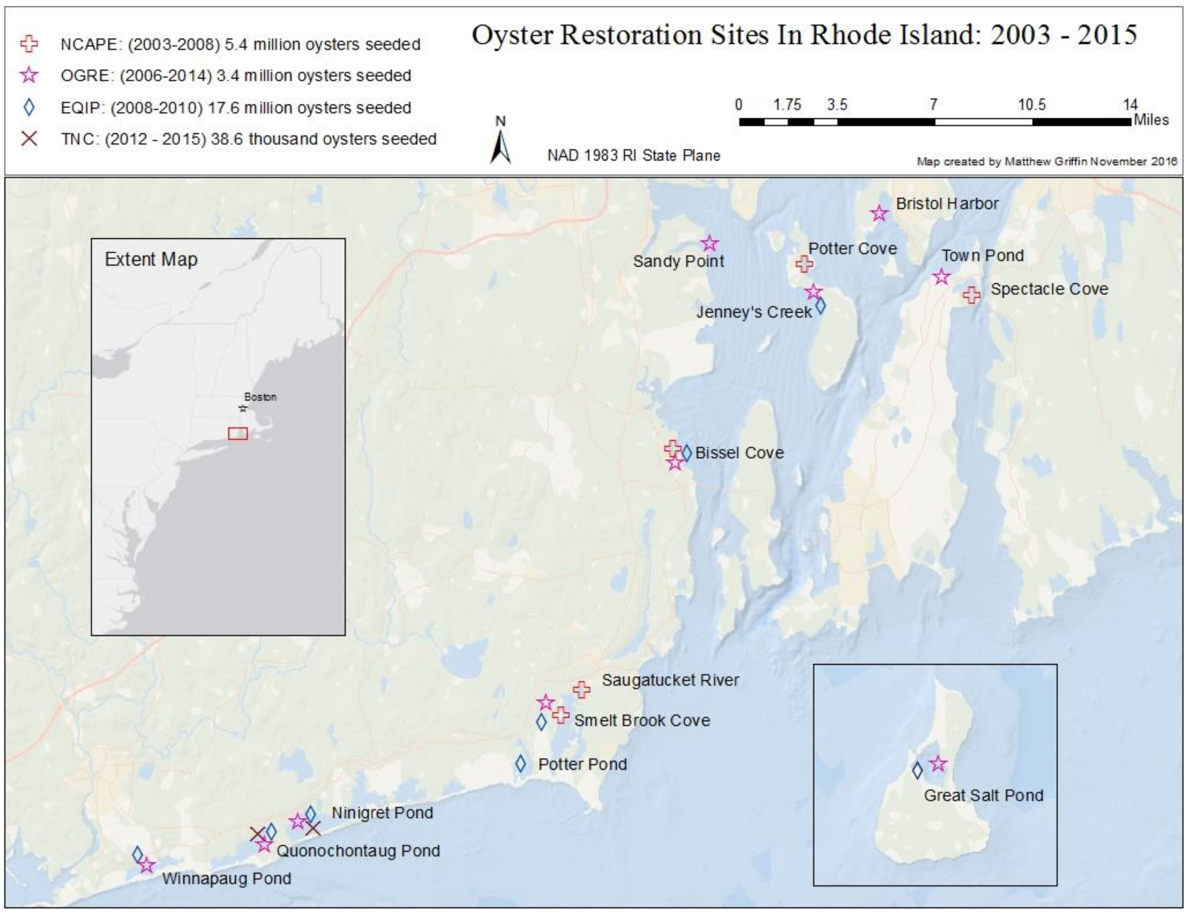


Figure 2. Total number of oysters seeded by project and year from 2003 to 2015.

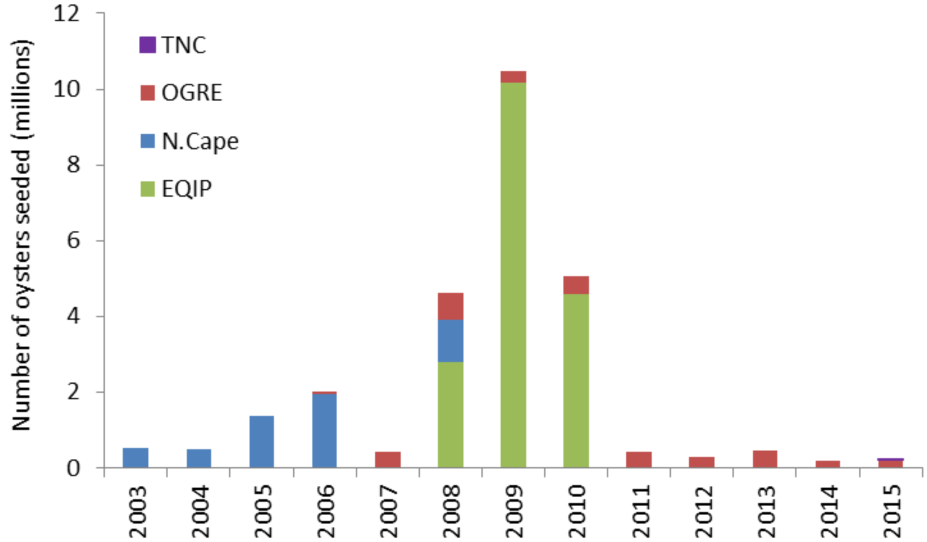


Figure 3. Mean valve height of oysters, measured in millimeters, in three restoration sites; Bissel Cove, Quonochontaug Pond and Smelt Brook Cove. Sites not connected by same letter are significantly different for the given year [p = <0.0001, α = 0.05 (2012)], [p = <0.0001, α = 0.05 (2013)], [p = 0.152, α = 0.05 (2014)].

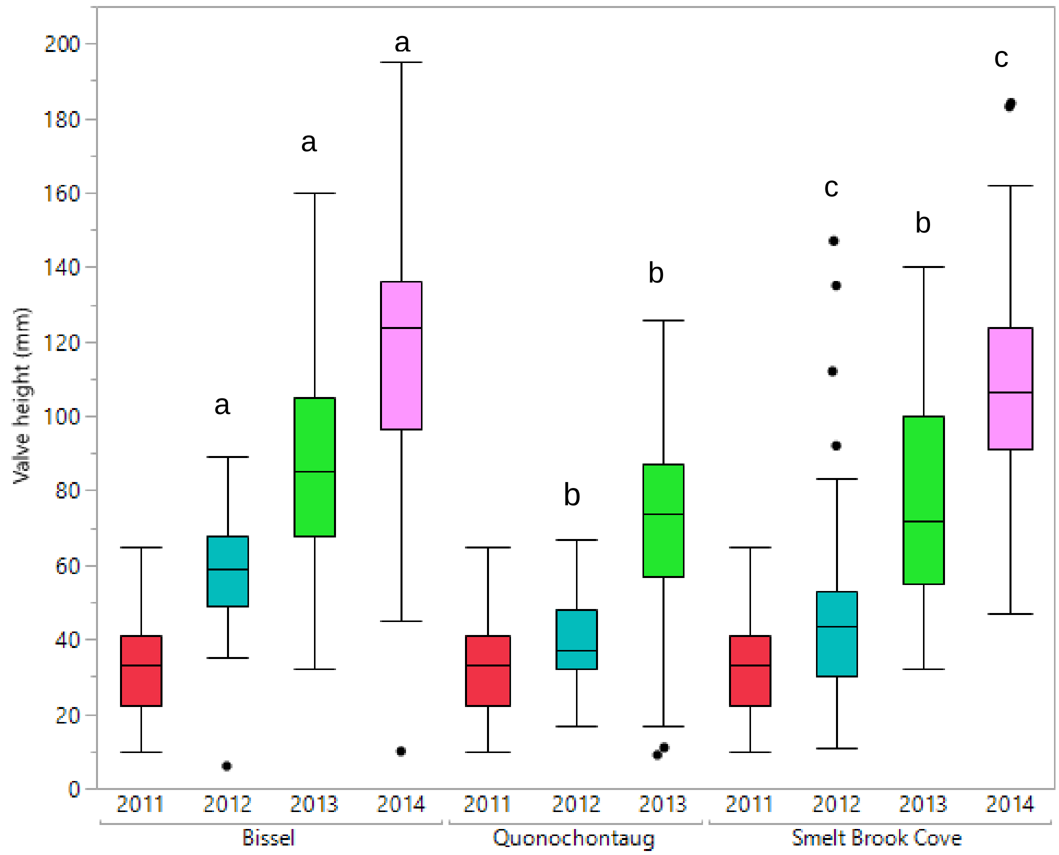


Figure 4. Mean percent survival of oysters during the first year post seeding ($p = 0.2167$, $\alpha = 0.05$).

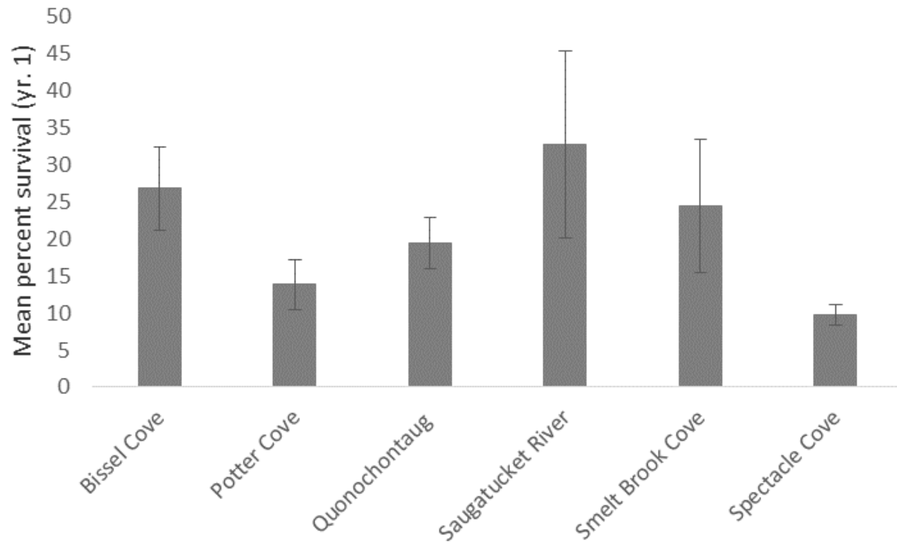


Figure 5. Mean percent survival of oysters after 2+ years post seeding ($p = 0.6319$, $\alpha = 0.05$).

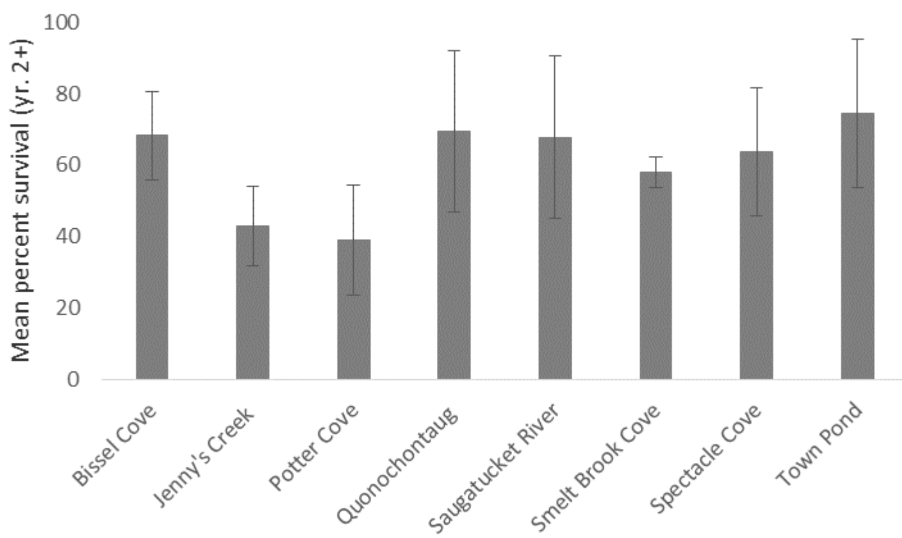


Figure 6a. Length frequency of live and dead oysters across all Oyster Gardening for Restoration and Enhancement sites. All cohorts included. Solid line represents lambda smoothing.

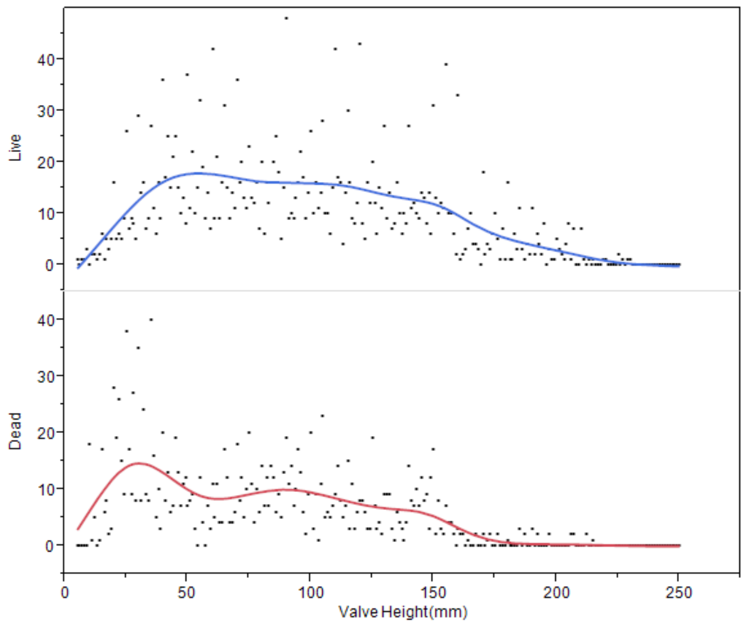


Figure 6b. Length frequency of live and dead oysters, excluding first year cohorts, across all Oyster Gardening for Restoration and Enhancement sites. Solid line represents lambda smoothing.

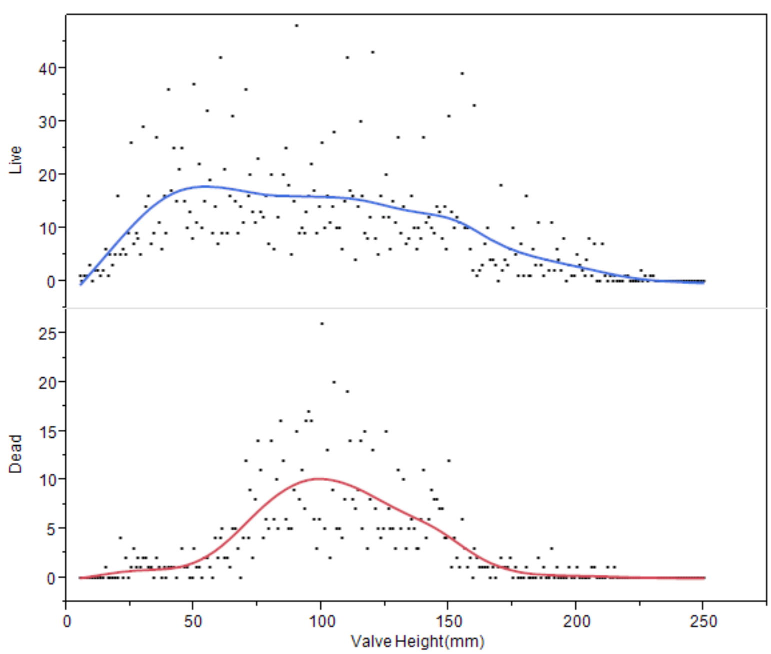


Figure 7. Mean number of oysters per spat collector by site. Data represents monitoring between 2011 and 2013. Recruitment was not observed on spat collectors between 2004 and 2008.

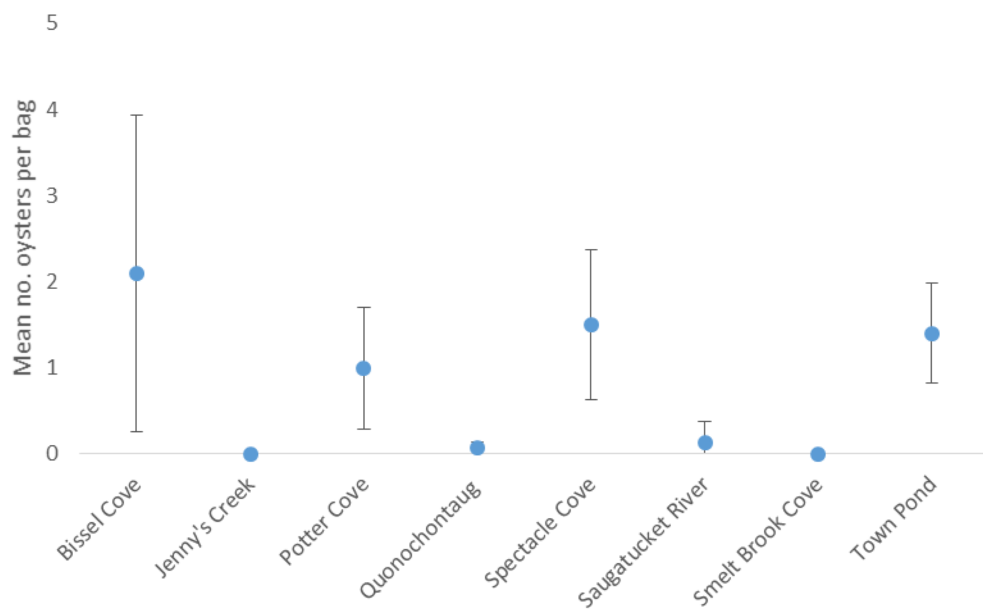


Figure 8. Mean recruitment to restored oyster reefs by site between 2011 and 2013. Recruitment was not observed on reefs between 2004 and 2008. Sites not connected by same letters are significantly different (p = 0.0514, α = 0.05).

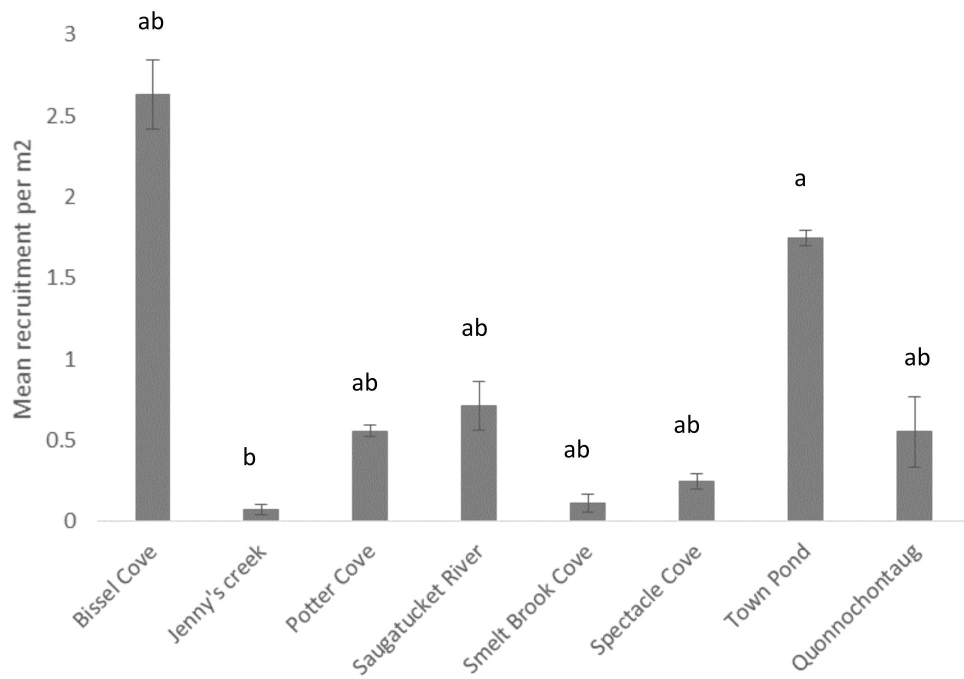


Figure 9. Mean percent *Perkinsus marinus* prevalence by restoration site. Data compiled from 2004 to 2013. Sites not connected by same letters are significantly different (p = 0.0007, α = 0.05).

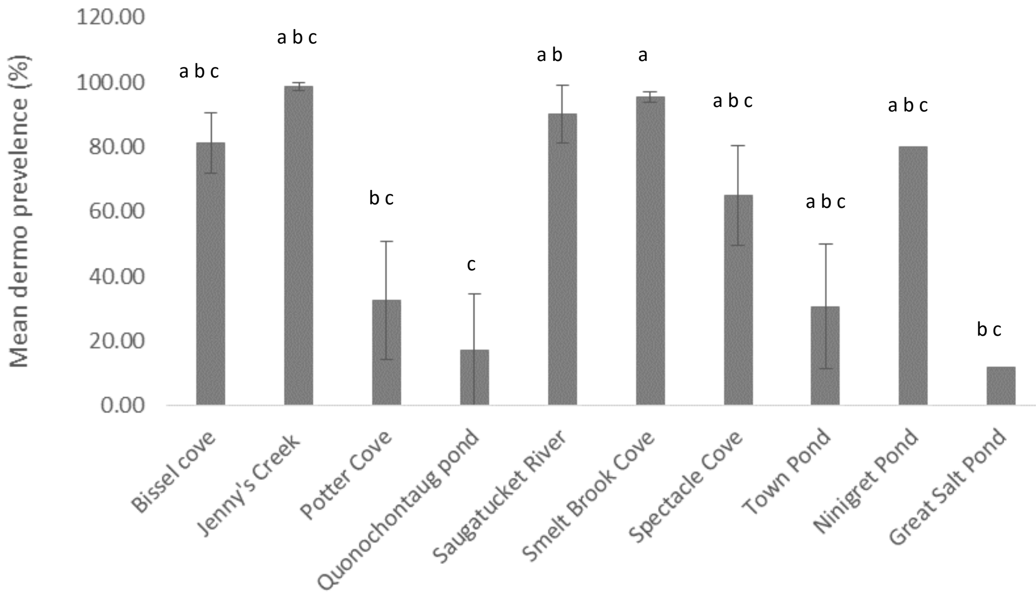


Figure 10. Mean sustainability index by site from 2004 to 2013. Negative values represent population decline ($p = 0.6022$, $\alpha = 0.05$).

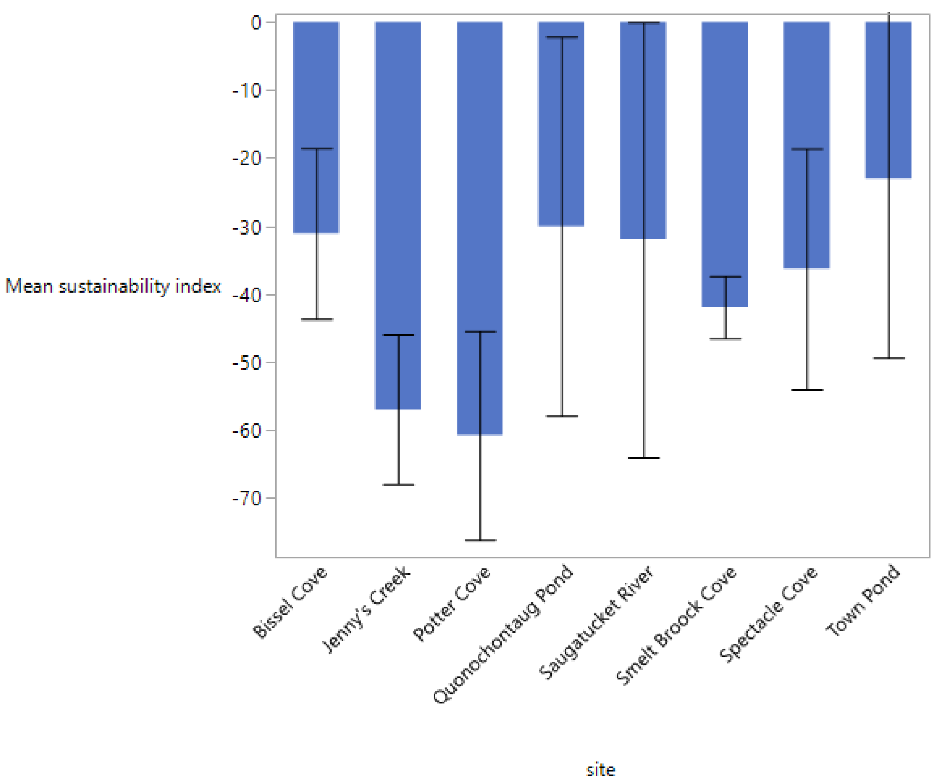
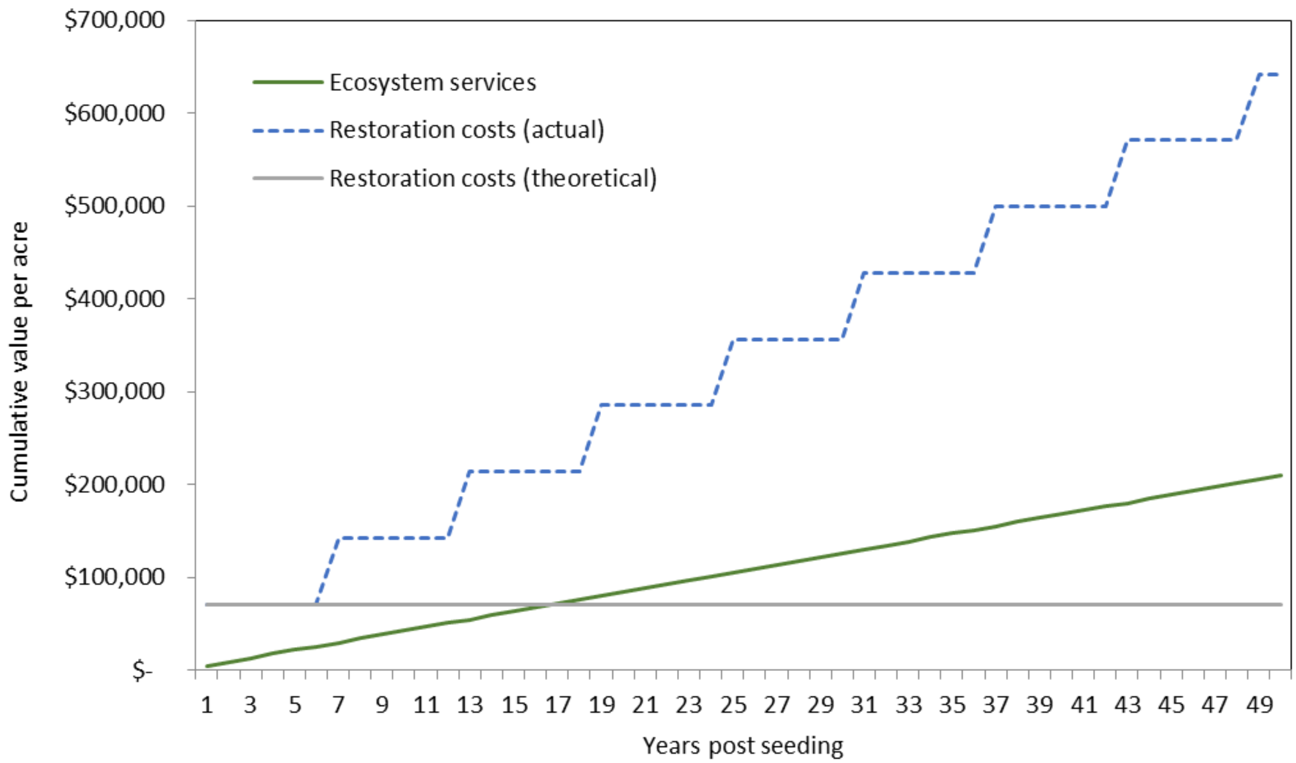


Figure 11. Cost-benefit model of cumulative ecosystem services provided from one acre of oyster reef versus costs of restoration. 'Actual' restoration costs represent annual operation costs from *North Cape* and Oyster Gardening for Restoration and Enhancement programs to maintain one acre of oyster reef. 'Theoretical' restoration costs represent a self-sustaining population after the initial seeding.



APENDICES

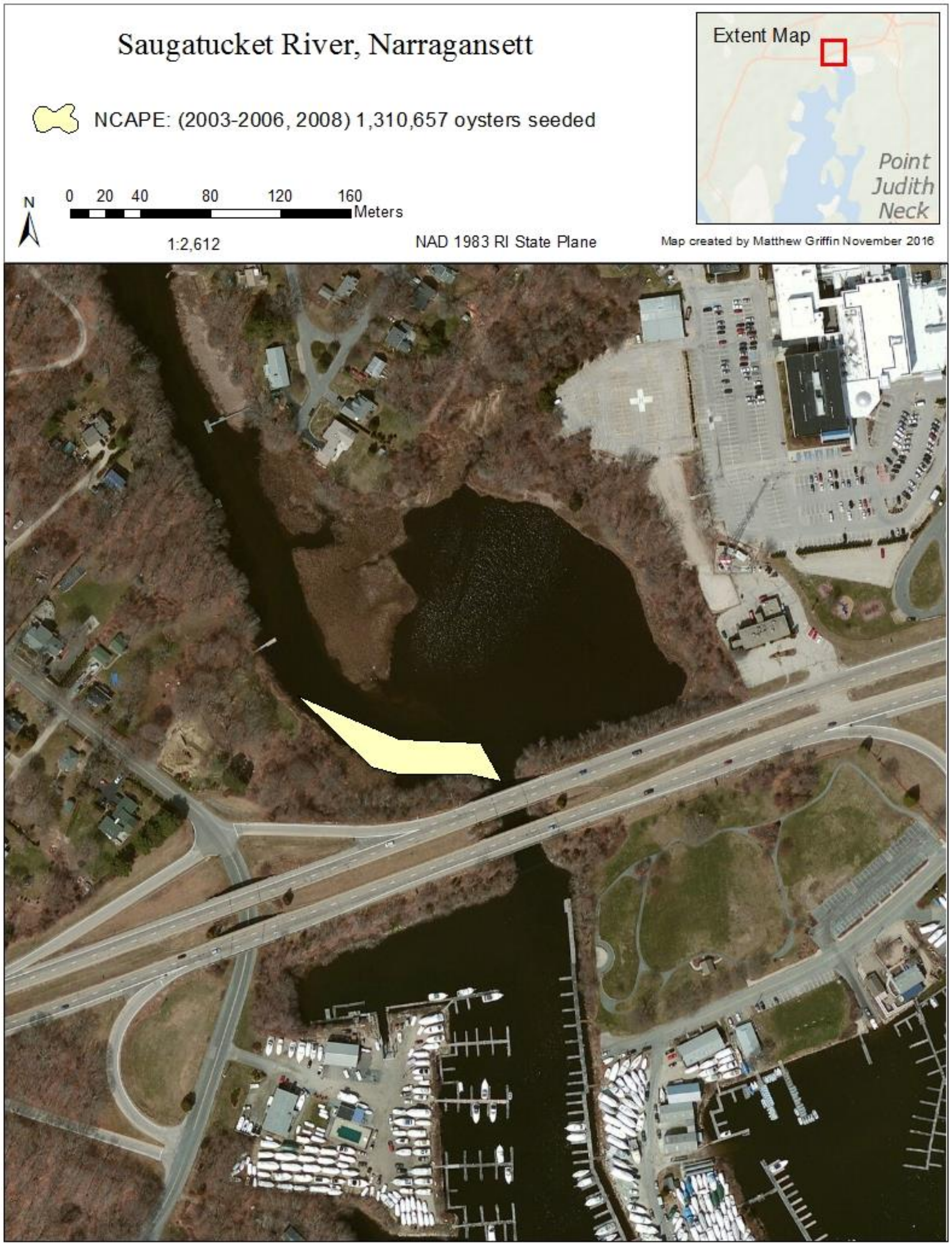
Appendix A. Oyster restoration in Town Pond, Portsmouth.

Appendix B. Oyster restoration in Spectacle Cove, Portsmouth.

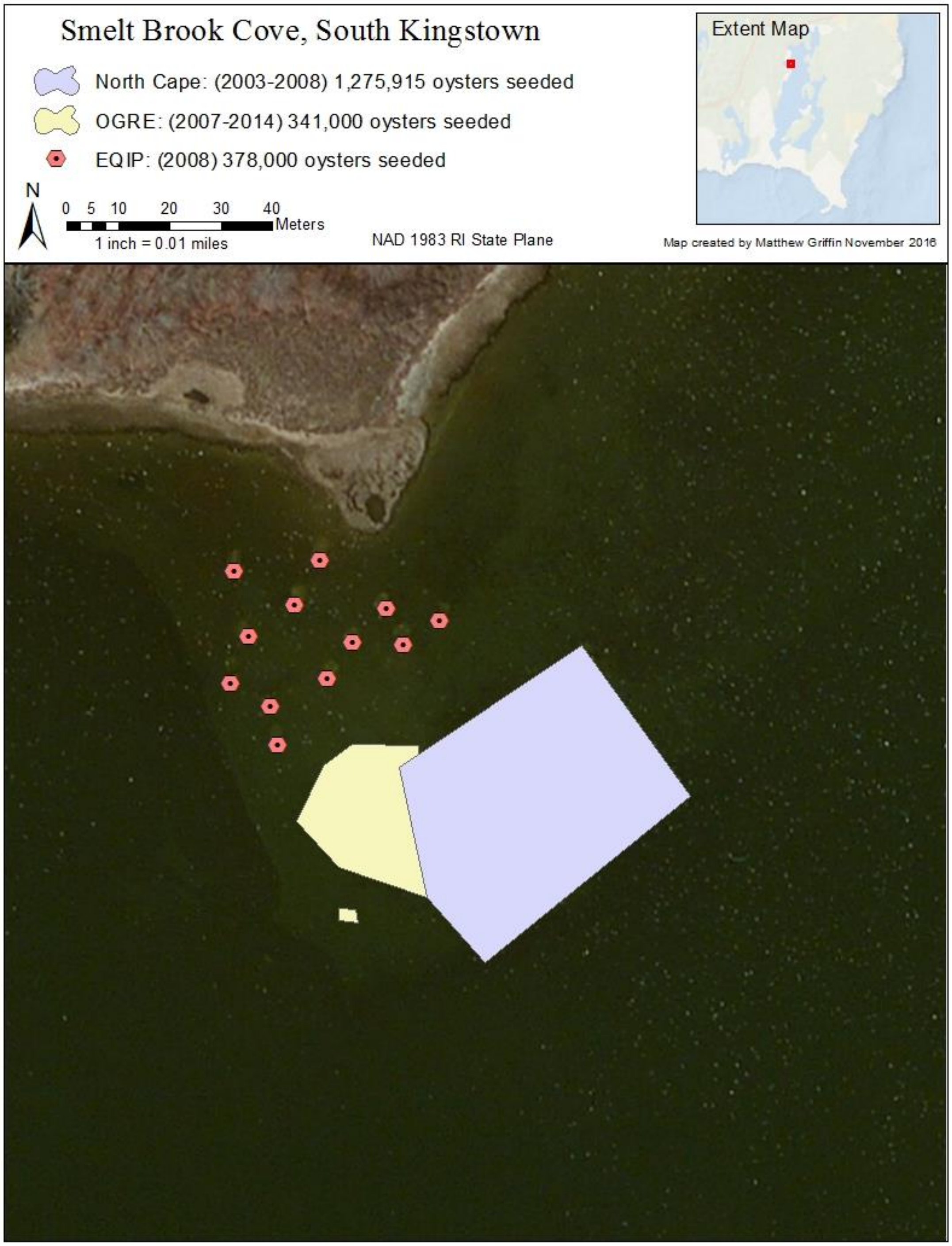
Appendix C. Oyster restoration in Jenny's Creek, Portsmouth.

Appendix D. Oyster restoration in Bissel Cove, North Kingstown.

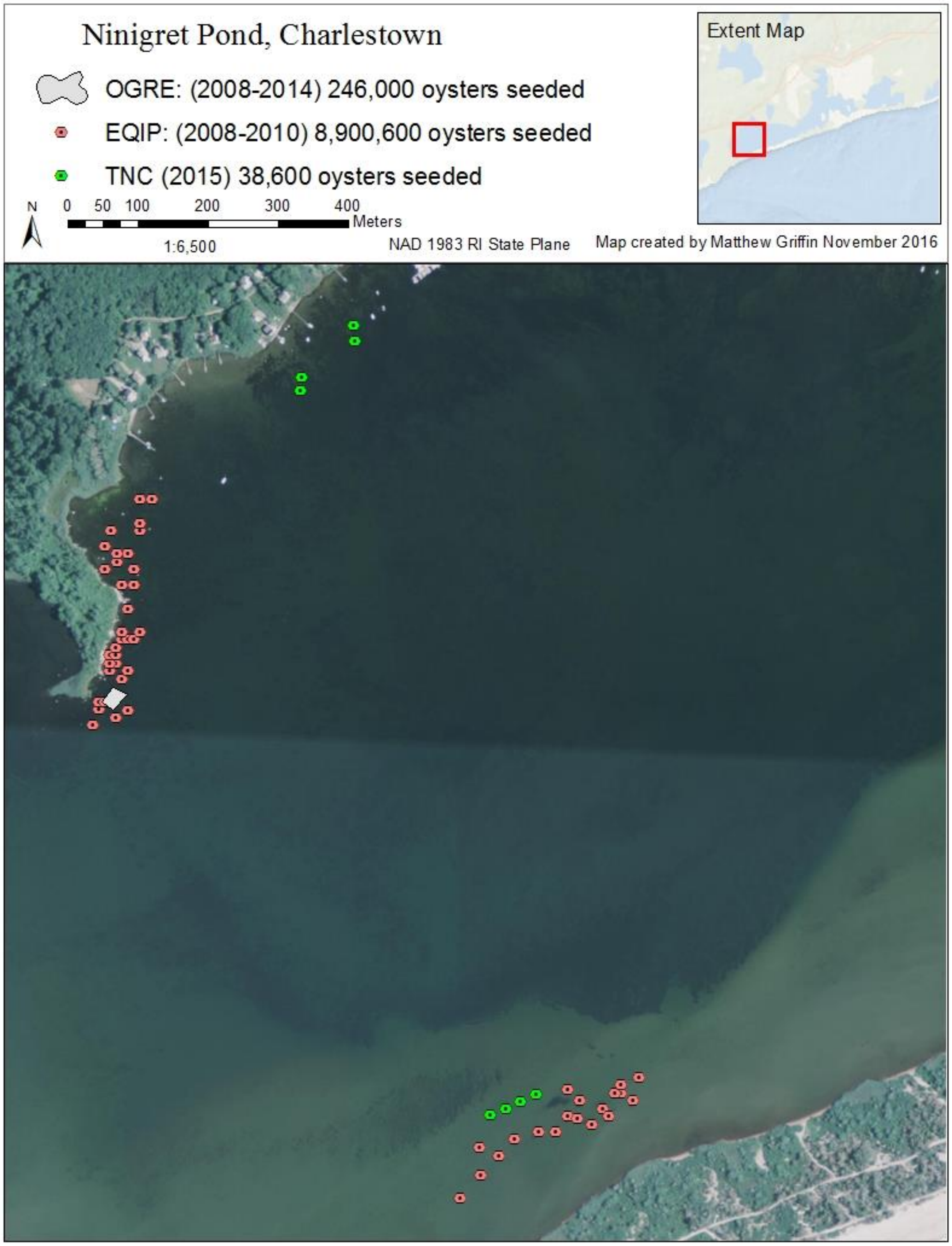
Appendix E. Oyster restoration in the Saugatucket River, Narragansett.



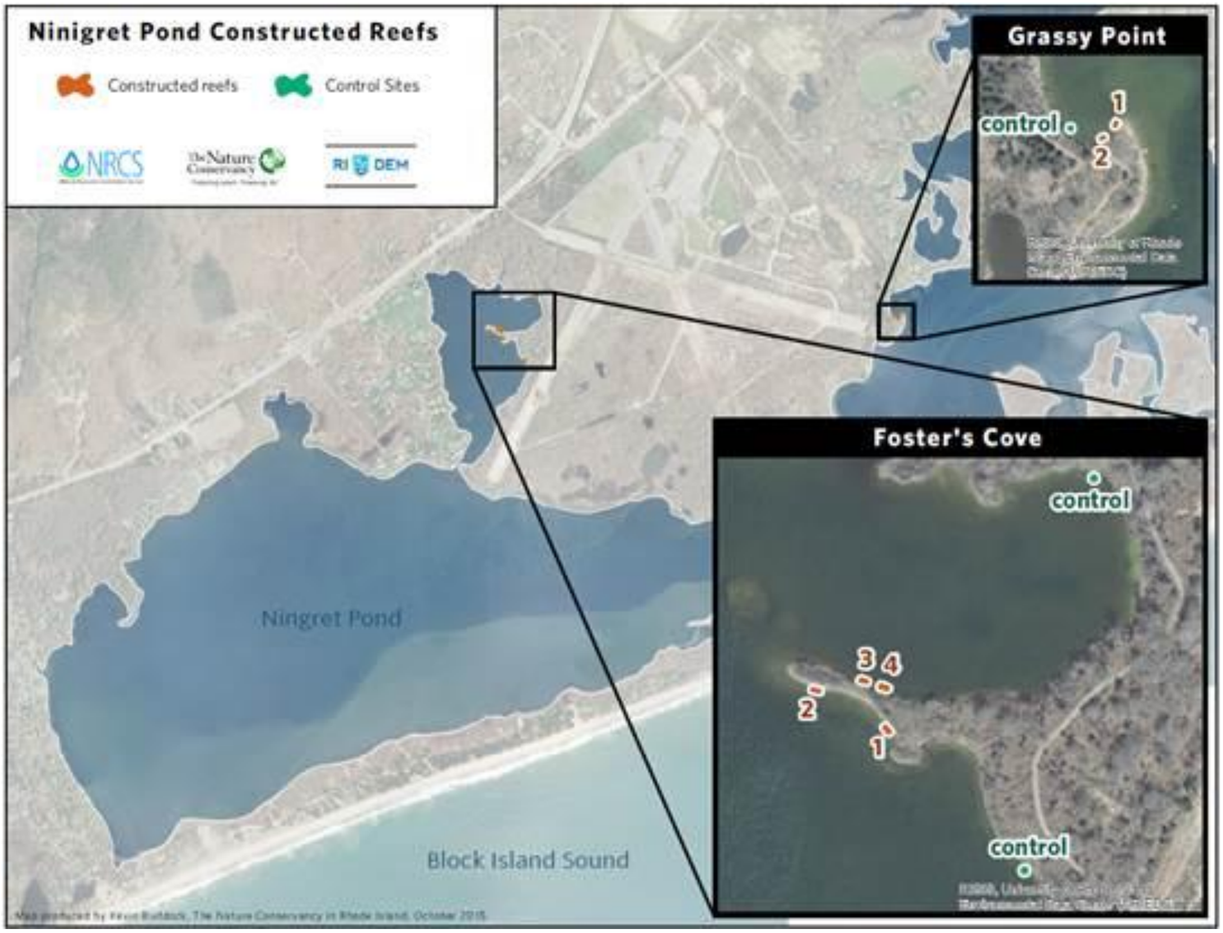
Appendix F. Oyster restoration in Smelt Brook Cove, South Kingstown.



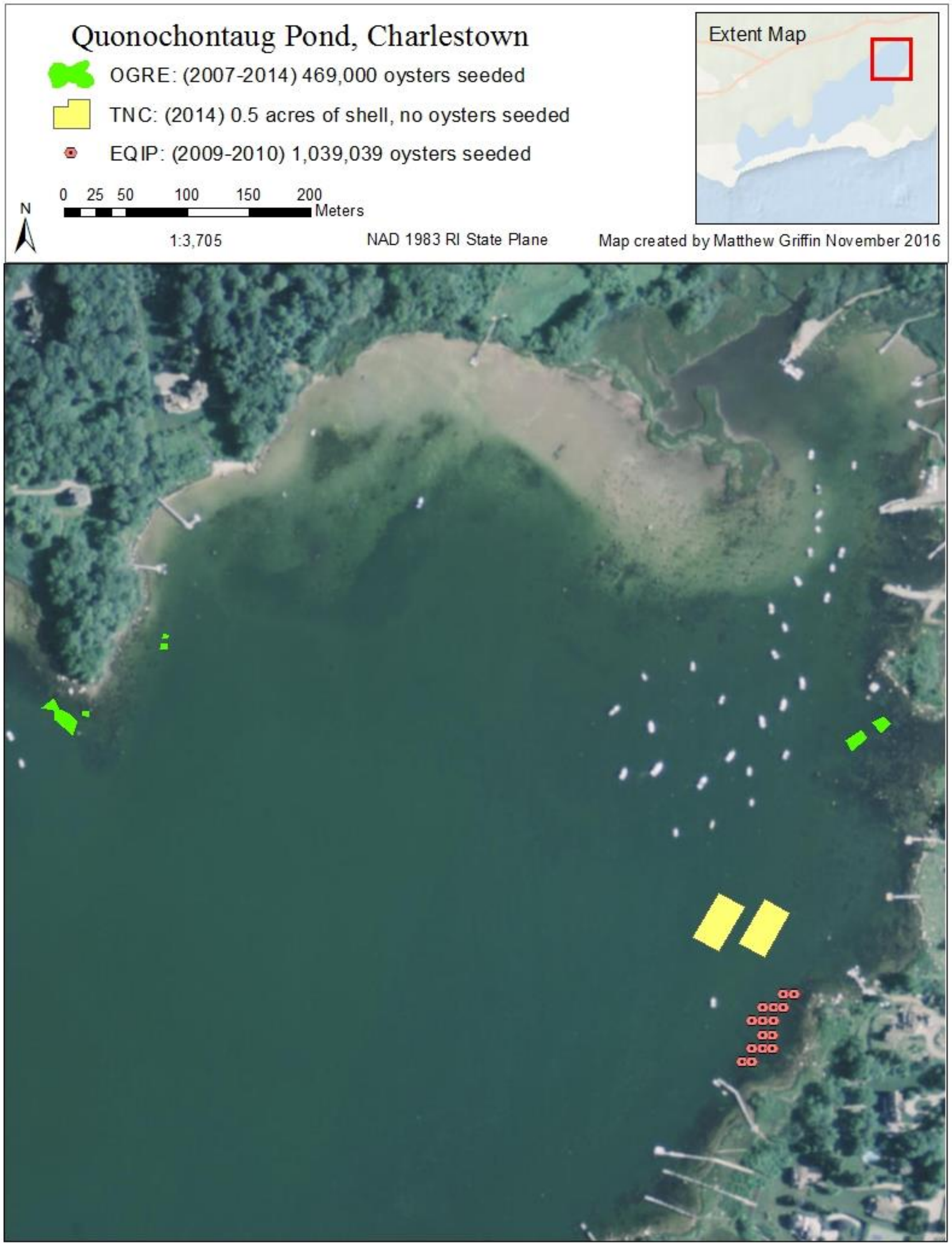
Appendix G. Oyster restoration in Ninigret Pond, Charlestown. Excluding TNC Oyster Castles (see Appendix H).



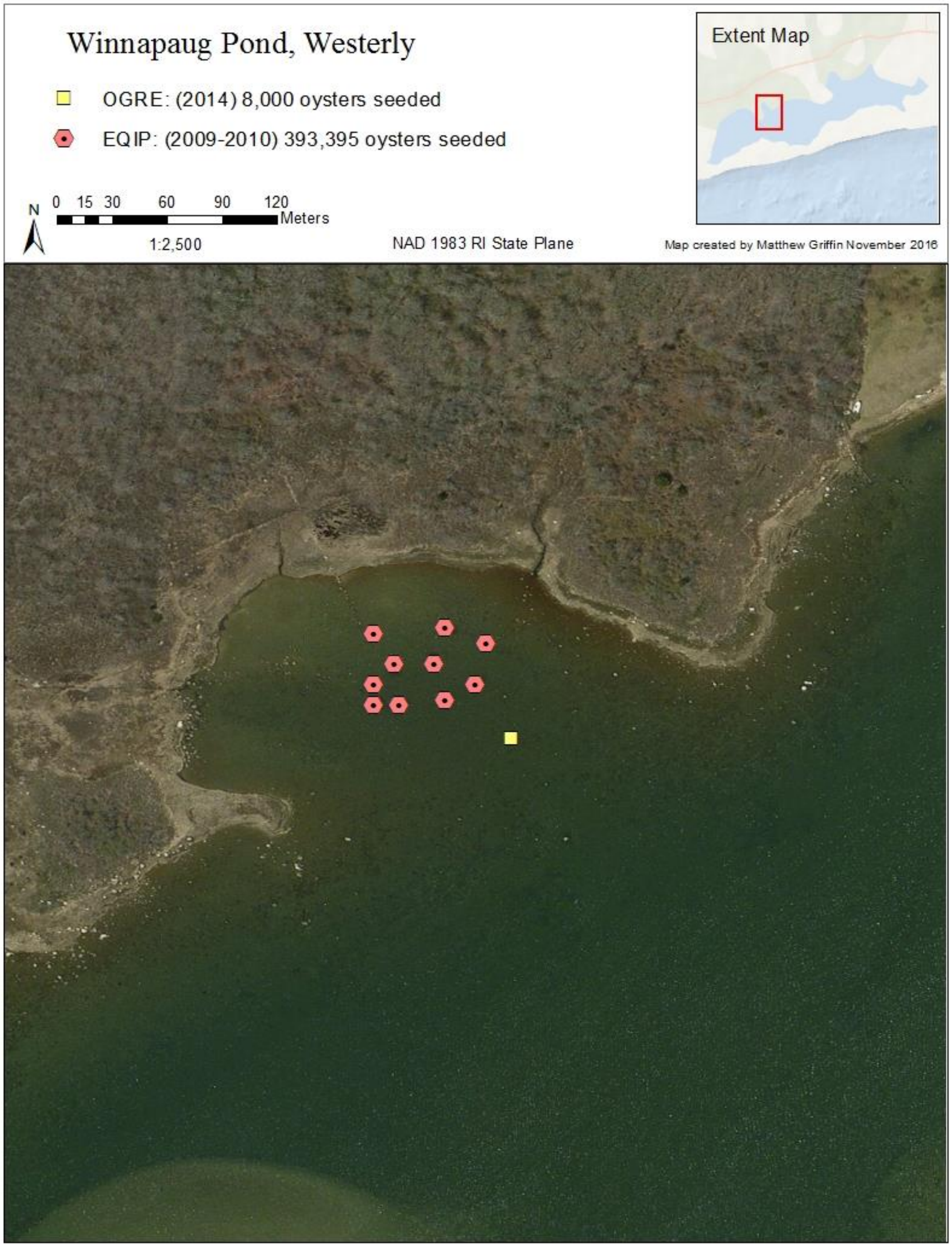
Appendix H. The Nature Conservancy Oyster Castle Reefs. Map provided by The Nature Conservancy. No oysters seeded on mapped reefs below.



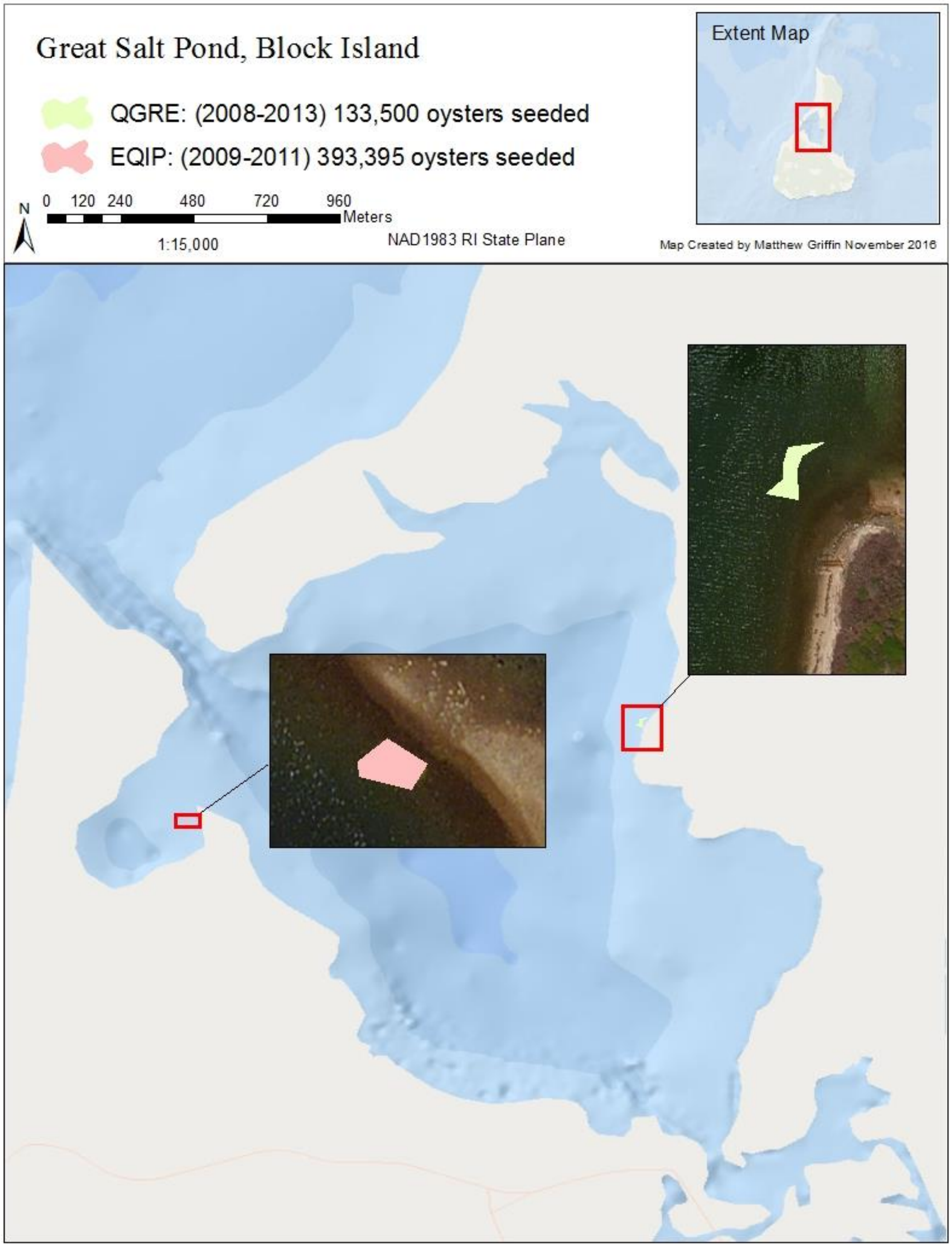
Appendix I. Oyster restoration in Quonochontaug Pond, Charlestown. TNC reef location is approximate.



Appendix F. Oyster restoration in Winnapaug Pond, Westerly. OGRE site location is approximate.



Appendix K. Oyster restoration in Great Salt Pond, Block Island.



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