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Contact Name and Telephone Number: Robert Hazen, Ph.D. (Retired); Nicholas A. Procopio, Ph.D., 609-940-4077; nick.procopio@dep.nj.gov

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Project Contributors:

Robert Hazen, Ph.D., Retired. Research Scientist, Division of Science and Research. New Jersey Department of Environmental Protection

Michael P. Weinstein, Ph.D., Retired Senior Research Associate. Montclair State University

Meiyin Wu, Ph.D. Professor. Department of Biology. Montclair State University

Danlin Yu, Ph.D. Associate Professor. Department of Earth and Environmental Studies. Montclair State University.

Nicholas A. Procopio, Ph.D. Bureau Chief, Division of Science and Research. New Jersey Department of Environmental Protection

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DEVELOPING A WATERSHED-SCALE BASELINE FOR TIDAL WETLANDS

Abstract:

Tidal wetlands are critical habitat for the maintenance of secondary production of nekton. This study addresses two key factors: the specific structural and functional traits of coastal wetlands that make them conducive to supporting the secondary production of nekton and how individual components of the marsh are inter-connected to form a functional whole. Structural characteristics of relatively undisturbed tidal wetlands have been determined by meta-analysis from more than 500 papers and in-situ measurements and remote sensing of New Jersey wetland sites. Meta-analysis (Hedges' d values and 95% confidence intervals) has shown that restored sites compare favorably with undisturbed sites in catch per unit effort (CPUE) for forage and predatory species but the average size of nekton captured at Phragmites-dominated sites was generally smaller than those of comparable species at reference sites. In-situ ecological data collected within the Wading River Complex show that quadrats along Transects A and B, situated in the upper part of the estuary and reflecting lower saline conditions, were dominated by *Phragmites australis, Spartina cynosuroides*, and *Typha augustifolia*. The quadrats along the lower Transect C in more saline conditions were dominated by *Spartina alterniflora* and *Spartina patens*. Water level readings from two transects on a single day showed that the marsh may flood up to 4.75 inches and flood stage over the marsh can last upwards of 7 hours. High resolution satellite images for a 20 km^2 area of the Wading River Complex were used to determine the landscape composition of the area and model future trends. Remote sensing analysis by a Grey System Series coupled system dynamic simulative model has shown in a 2019 to 2033 simulation that an increase in *Phragmites australis* will pose a severe threat to local fauna and flora. We present an underlying framework for restoration success criteria that optimize secondary production and connectivity to adjacent habitats including the open waters of the estuary.

Report Organization:

This report is divided into three major sections including a meta-analysis of tidal wetland literature with a focus in the Delaware Bay, the Wading River as a regional priority area where in-situ ecological and hydrologic data were collected to establish baseline conditions, and an evaluation of remotely sensed data in the Wading River estuary that models habitat change through time. Each section is relatively stand alone and a project-wide summary and conclusion is provided at the end of this report.

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Overview of Need:

Tidal wetlands are critical habitats that contribute to the economy of coastal communities through production of "charismatic" taxa that support a $60 billion national fishery. Both President Obama's Executive Order 13547 and Congress' Magnuson-Stevens Act recognized that anthropogenic activities have negatively impacted coastal communities that rely on natural resources for their livelihood and economic prosperity. A coupled framework for metric development by *in-situ* measurements of the marsh drainage network, and dominant vegetation combined with photointerpretation of satellite imagery is needed to better understand the distribution and functional attributes of tidally influenced marshes and establish baseline conditions in reference environments. The reference baseline will help quantify and forecast potential alterations in tidal wetland structure and function due to sea level rise, climate change, and future land-use and land-cover changes.

A central tenet of the role of tidal wetlands is their support of secondary production of nekton, especially those taxa that contribute to coastal fisheries worldwide. It has been variously estimated that up to 80% of commercial and/or recreational fishes, shellfish and their forage base in coastal regions are believed to have "estuarine dependent" early life stages (Gunter, 1967; McHugh, 1976; Lellis-Dibble et al., 2008). Because they are rich in nutrients, provide abundant food, and likely shelter from predators, coastal wetlands are thought to be "essential" - *sensu* the United States Magnuson Stevens Fishery and Conservation Act - for the early life stages of commercial and recreational taxa that contribute to the stocks of major fisheries (MSFCA 2007) and consideration of "essential habitat" is being built into fishery management plans.

Not only do many ecosystems differ in structure and function from those in the past, but the ecosystem concept itself is increasingly framed in the context of climate change, land use, invasive species, reduced biodiversity, and other outcomes of human endeavors. These new ecosystem states, including those in coastal wetlands, are often less desirable, and are described as "novel, no-analog, or emerging" states (Hobbs et al., 2009, Higgs, 2012). As a consequence, the challenges of wetland ecosystem preservation, restoration, and rehabilitation have reached new levels of complexity. It is critical that we establish a meaningful range of baselines for the least disturbed systems as a frame of reference for efforts to quantify altered states and attempt to reverse centuries of impacts on tidal wetland quality and function.

Recognizing the many potential restoration end-points are influenced by prior human activities in wetland landscapes, as well as natural ones, the endeavor of restoration must be undertaken in the context of *multiple* stable endpoints, or what can be referred to as a "bound of expectation" (Weinstein et al., 1997; French, 2005). Aronson (1995) warns that by failing to inject 'a dynamic perspective into real world restoration efforts', i.e., a range of reference ecosystems, the outcome would be 'failed restorations and frustrated restorationists'. Nowhere is this more important than the realization that tidal wetlands do not function in isolation, but rather are integrated parts of watersheds and coastal ecosystems (Weinstein et al., 2014). The problem is to identify metrics that constitute a primary driver of the link between primary and secondary production and the exchange of materials and organisms between the wetland and surrounding ecosystem. We need to know how wetlands support the resource base and health of coastal fisheries; a $60 billion industry nationwide, and a nearly $3 billion industry in New Jersey. This area of evaluation remains virtually untouched in the vast number of wetland assessments that have been developed to date.

This study focuses on two key questions: (1) what are the *specific* structural and functional traits of coastal wetlands that make them conducive to supporting the secondary production of nekton; and (2) how are individual components of the marsh *inter-connected* to form a functional whole?

Project Goals:

New Jersey wishes to build its wetland programs to more effectively and efficiently address long-term wetland program planning and to carry out shorter-term projects that are consistent with the defined goals outlined through its Wetland Program Plan (WPP).

Thus, the primary goals of this research are (1) to establish a baseline for a relatively undisturbed tidal wetland that links its planform to secondary production, and the exchange of organisms and materials between the wetland and adjacent waters; and (2) to develop an underlying framework for restoration success criteria that optimize secondary production and connectivity to adjacent habitats including the open waters of the estuary.

Components of this research directly address Core Elements 1 and 3 of the current NJDEP WPP (NJDEP, 2019). Specifically, this work addresses:

- Core Element 1. Monitoring and Assessment
  - Objective 1 - Action 1: Identify program decisions and long-term environmental outcome(s) that will benefit from a wetland monitoring and assessment program;
  - Objective 1 - Action 4: Select a core set of indicators to represent wetland condition or a suite of functions;
  - Objective 2 - Action 5: Analyze monitoring data to evaluate wetlands extent and condition/function or to inform decision-making.
- Core Element 3. Voluntary Wetland Restoration, Creation, Enhancement and Protection and Improved Coastal Shoreline Resiliency
  - Objective 1 - Action 2: Consider watershed planning, wildlife habitat and other objectives when selecting restoration/creation/enhancement/protection sites; apply tools (GIS, color-infrared photography, mapping, modeling, field inspection of soil, vegetation, and hydrologic conditions) to identify projects.

Through this project we provide a summary of a meta-analysis of literature pertaining to links between wetlands and secondary production of nekton, identify quantitative metrics that relate geomorphological and ecological characteristics, utilize high resolution multi-spectral imaging to characterize vegetation (composition, biomass stem density, etc.) and hydrological characteristics, couple in-situ data with remotely sensed data, and identify in-situ data and vegetation monitoring methods that contribute to critical decision support for these and other sites.

Section 1. Meta-analysis of Wetland Literature: Review, Data, and Analysis:[1]

Meta-Analysis of Literature Pertaining to Links between Wetlands and Secondary Production
Project Lead: Michael P. Weinstein

Introduction:

Restoration success takes two general forms, (1) projects that restore ecological fidelity and longevity (self-organizing traits) to sites through the application of best scientific principles (Higgs, 1997), and (2) projects that rest on cultural foundations, restoring sites to some practical use as perceived by society (Cairns et al., 1975). For more than two decades, these tenets were adopted as part of an effort to restore 4,049 ha (10,000 ac) of tidal wetlands on Delaware Bay, USA. State and federal agencies were tasked with monitoring the progress of restoration in two types of degraded wetlands; formerly diked salt hay (*Spartina patens*) farms in the lower Bay, and brackish, meso-oligohaline, *Phragmites australis* dominated sites in the upper Bay. The underlying theme of the Estuary Enhancement Program (EEP), as the project is known, was to link primary production in the salt marshes to secondary production of nekton that were impacted by a power facility located on the New Jersey side of the upper Bay (Figure 1) (Weinstein et al., 1997; Weinstein et al., 2001). The presumption was that the area of salt marshes to be restored would offset losses at the power facility and contribute new generations of fish and shellfish (blue crabs, *Callinectes sapidus*) to support the regional economy (Frisk et al., 2011; Weinstein et al., 2012; Weinstein & Litvin, 2015).

Decades of data have been accumulated on tidal salt marsh restoration in Delaware Bay, New Jersey. Because of this data rich environment, it is an ideal location to summarize and evaluate various measures of restoration success. Restoration in such areas was necessary due to the historic construction of dikes in the early 20th century to provide for suitable agricultural conditions to produce salt hay (*Spartina patens*) for animal fodder. Restoration efforts have involved removing the dikes and channel dredging to restore hydrology and salinity to near-original conditions.

This meta-analysis utilized software to perform the scanning of hundreds of research papers for commonalities of location, time, and physical and biologic parameters which characterize the degree of restoration success. Because salt marshes do not function in isolation, a broad literature review of the linkages between wetlands to the estuary as a whole, including the coastal zone is provided. To this extent wetlands should be viewed as interactive components of the broader mosaic of habitats that exchange materials and organisms, and which together interactively support the secondary production of marine finfish and shellfish.

A meta-analysis of the relevant metrics of the salt marsh that help define what is "essential" about them in their structure and functions that make them conducive to secondary production of commercial/recreation species and their forage base was conducted**.** This analysis identifies metrics for the planform of coastal wetlands that support basic wetland functions, and that acknowledge the link between coastal wetlands and secondary production that ultimately support more than 80% of coastal fish stocks and a US marine fisheries economy that exceeds $60 billion nationwide. It reviews existing literature that relates the role of riverine fresh, tidal fresh, and tidal saline marshes of the WRC as essential habitat for the secondary production of fauna that use these habitats. The need to manage the introduced m-haplotype of *Phragmites australis*, an invasive species, *or biopollutant*, of national concern, is also recognized herein.

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[1] Portions of this section follow Weinstien, M. P., R. Hazen, and S. Y. Litvin. 2019. Response of Nekton to Tidal Salt Marsh Restoration, a Meta-Analysis of Restoration Trajectories. Wetlands. 39:575-585 which was published as part of this grant and included as supplemental material.

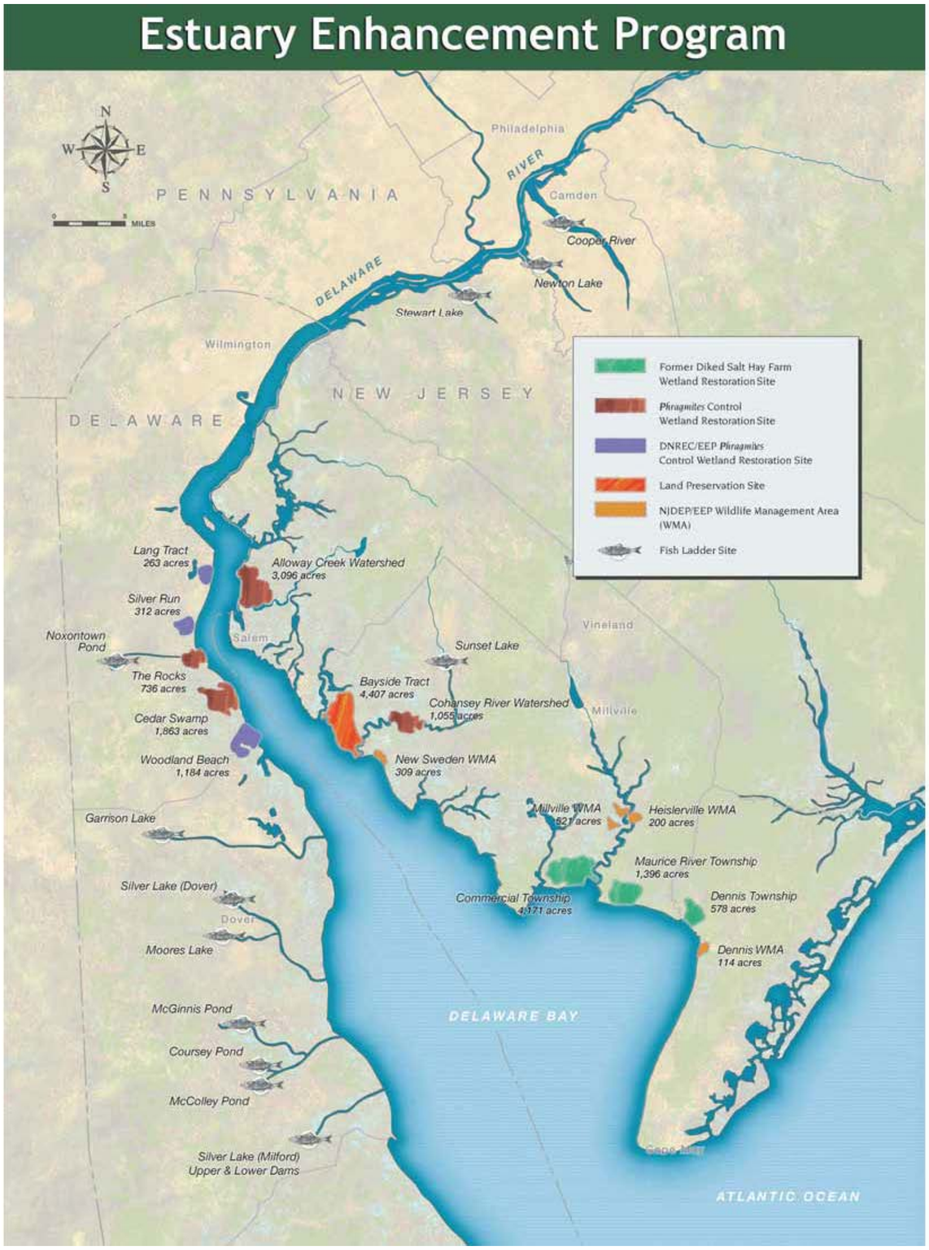


Figure 1. Location of Estuary Enhancement Program (EEP) sites along the Delaware Bay. Map posted by PSEG on the EEP website.[2]

Meta-Analysis Methods:

The meta-analysis utilized MetaWin 2.0 (Rosenberg et al., 2000; https://metawin.software.informer.com/) software with an increasing complexity of relevant key words over six effect sizes. Keyword searches run against selected research databases including: Science Direct; JStor, Google Scholar, Research Gate, and Aquatic Science and Fisheries Abstracts. Keywords in each effect are listed in Table 1.

[2] https://corporate.pseg.com/corporatecitizenship/environmentalpolicyandinitiatives/estuaryenhancementprogram

Table 1. Keywords used in each meta-analysis effect evaluation.

Effect 1 keywords:	Secondary production in undisturbed tidal marshes or reference tidal marshes; Secondary production in restored tidal marshes
Effect 2 keywords	Fish and shellfish growth, density and production in salt marshes
Effect 3 keywords	Fish, shellfish numbers, density, growth, survival in tidal salt marshes
Effect 4 keywords	Population dynamics of nekton in tidal salt marshes
Effect 5 keywords	Population dynamics of finfish and crabs in tidal salt marshes
Effect 6 keywords	Fish, blue crabs, numbers, abundance, density, growth and mortality in natural or reference tidal salt marshes versus restored tidal salt marshes; Fish, blue crabs, numbers, abundance, density, growth and morality in restored tidal salt marshes

Initially, more than 500 papers were identified by the six keyword searches on selected databases. From this initial list, numerous papers were excluded because of geographic location, unrelated tidal cycles, species considered, absence of reported data on population dynamics, etc. After the first screening, 335 papers remained, which were further evaluated for relevancy based on the species reported (e.g., benthos vs fish; terrestrial taxa); absence of quantitative information on density, growth, mortality; and emigration for relevant Atlantic coast taxa (Table 2). After the initial sorts, roughly half the papers had value for further review.

Table 2. Meta-analysis results

	Total Papers	Papers Kept
Effect 1	75	28
Effect 2	21	17
Effect 3	60	34
Effect 4	55	36
Effect 5	58	49
Effect 6	66	31
Totals	**335**	**146**

Meta-Analysis Site Characterization:

A Phragmites-Dominated Marsh:

The Delaware Bay estuary shoreline is fringed by approximately 81,000 ha (200,000 ac) of nearly contiguous tidal salt marshes, but marshes in the oligohaline-tidal freshwater portions of the estuary below Philadelphia, PA are dominated by an introduced variety of *P. australis*, the m-haplotype (Saltonstall, 2002), comprising ~16,000 ha (40,000 acres) (Weinstein and Balletto, 1999; Weinstein et al., 2000a).

Most of the 2,145 ha (5,298 ac) of *P. australis* dominated tidal marshes addressed in this study were diked early in the 20th century, but the times of dike breaching at these locations are largely unknown. *P. australis* is believed to negatively influence the habitat value of the marsh by elevating the marsh planform and filling in the microtopography of the marsh surface (Windham, 1995; Able et al., 2003; Rooth et al., 2003). The former alters the hydroperiod, the latter influences access to the marsh plain by resident and transient organisms (Boesch and Turner, 1984; Rozas et al., 1988; Kneib, 1997; Able and Hagen, 2000). The steep banks of *P. australis* lined tidal creeks may also negatively influences the survival of small fishes by exposing them to increased predation (McIvor and Odum, 1988).

Salt Hay Farms:

The nearly 1780 hectares (4,397 ac) of polyhaline salt hay farms acquired for restoration were originally contained by perimeter dikes, at ~ 1.5 m North American Vertical Datum (NAVD). Most of these farms had been in continuous agriculture since the mid-twentieth century, but some had been diked in the early 1700s (Weinstein et al., 2000a, b). As a result, many of the dikes had subsided with time and needed to be routinely maintained to continue production of salt hay (*Spartina patens* and *Distichlis spicata*). At one of the restoration locations, dikes were breached by a 1992 storm resulting in the extirpation of the salt hay grasses. It was anticipated that similar loss of high marsh grasses would occur when dikes at the remaining low-lying restoration sites were breached.

Data Organization and Compilation:

A seventeen-year period, 1999 through 2015, comprised the temporal framework for this study, including fish and shellfish abundance and length frequency distributions at seven locations (Figure 2). Reference sites included Mad Horse Creek (1564 ha) located in the oligo-mesohaline waters of the upper Bay, and Moore's Beach (504 ha), a polyhaline site, in the lower Bay (Figure 2). "Experimental" (hereinafter restoration) sites included Alloway Creek (85 ha), Mill Creek (518 ha) and Brown's Run (196 ha); all dominated by *P. australis* (from 42%, to near monocultures at 97%; Table 3). The *Phragmites*-dominated sites were treated by aerial application of the herbicide, Rodeo® containing a surfactant during the peak growing season in 1996, followed by prescribed burns in 1997 (Figure 3 a, b, c) (Able et al., 2003). By Spring 1998, the treated areas were predominantly unvegetated mud flats (Able et al., 2003). In the lower Bay, the experimental sites, Dennis Township (227 ha) and Commercial Township (1,504 ha) were former salt hay farms where the dominant high marsh vegetation, consisting mainly of *Spartina patens* and *Distichlis spicata*, were "artificially" maintained as high marsh by enclosure dikes that prevented daily tidal inundation. Restoration activities included breaching the dikes to establish entrance channels that were also dredged 300-4,000 m into the marsh (Weinstein et al., 2000a, b). Most of the construction activities were completed prior to 1999, and, for this reason, 1999 was chosen as the departure point for all comparisons in this study. Only two of the sites evaluated met the permit success criteria during the study period, Dennis Township in 2000, and Brown's Run in 2004 (Figure 4).

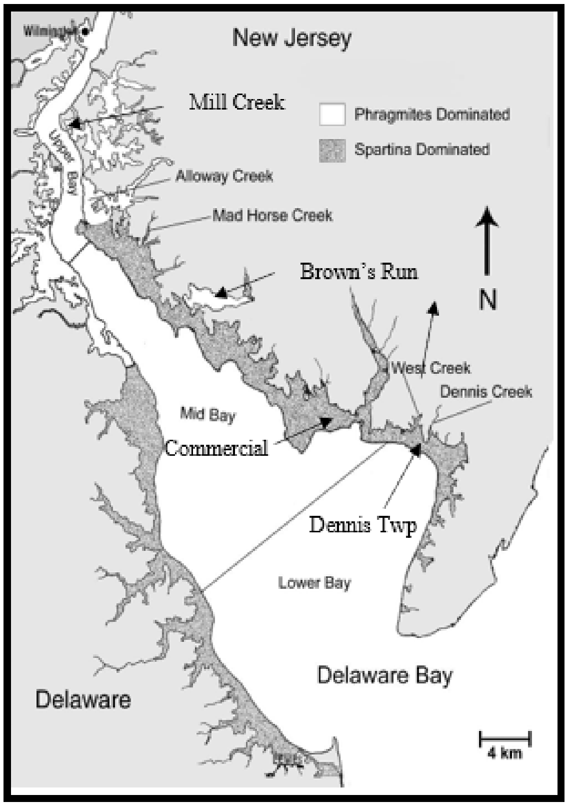


Figure 2. Reference (Mad Horse Creek in upper Bay; and Moore's Beach in lower Bay) and restoration sites (Mill Creek, Alloway Creek and Brown's Run in upper Delaware Bay; and Commercial Township and Dennis Township in lower Delaware Bay.

Table 3. Baseline area (ha), vegetation and geomorphic cover (creeks, ponds mud flats) at Delaware Bay reference and restoration sites.

Upper Bay		Total Area (ha)	Dominant Vegetation	Vegetation Cover (%)	Creeks/ Ponds/ Mud/ Flats (%)
Mad Horse Creek	Reference Site	1564	*Spartina alterniflora*	83	14
			Phragmites australis	3	
Alloway Creek	Restoration Site	85	*Spartina alterniflora*	36	4
			Phragmites australis	60	
Mill Creek	Restoration Site	518	*Spartina alterniflora*	2	1
			Phragmites australis	97	
Brown's Run	Restoration Site	196	*Spartina alterniflora*	47	11
			Phragmites australis	42	

Table 3. Continued

Lower Bay		Total Area (ha)	Dominant Vegetation	Vegetation Cover (%)	Creeks/ Ponds/ Mud/ Flats (%)
Moore's Beach	Reference Site	504	*Spartina alterniflora*	84	11
			Phragmites australis	5	
Dennis Twp.	Restoration Site	227	*Spartina spp.*	82	12
			Phragmites australis	6	
Commercial Twp.	Restoration Site	1504	*Spartina spp.*	63	2
			Phragmites australis	35	

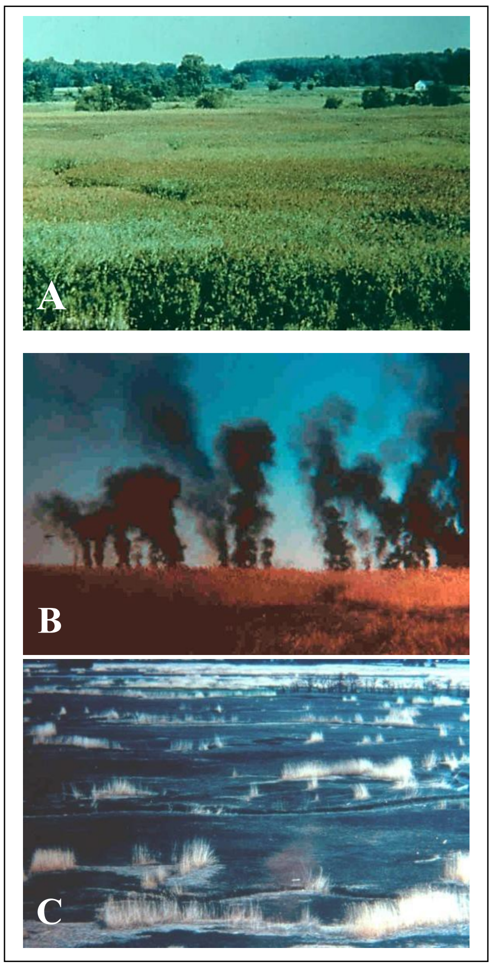


Figure 3. *Phragmites australis* monoculture at Alloway Creek Site sprayed by helicopter with Rodeo® and a surfactant during the peak growing season (A), dead stalks burned in the fall (B) leaving a marsh surface devoid of and most vegetation and microtopographic relief (C).

Raw data collected in the EEP Monitoring Program were used as a baseline to extract information on numbers of fishes and blue crabs captured at reference and restoration sites during the study period (Table 4). As noted, monthly sampling effort varied among sites and years, but was generally extensive. Nekton were captured during the period April or May through November of each year by otter trawl at three locations in each marsh, and block nets ("weirs") were also set at three additional locations**.** Trawls consisted of four 2-min tows per station, against the tidal current at a constant engine speed (2,500 rpm). At the "weir" stations, block nets 2.0 m x 5.0 m, with 5.0 m by 1.5 m wings all with 6 mm meshes were set across intertidal creeks at high tide and retrieved 6h later at low tide. Although there is some overlap, the two gears have different selectivity and capture efficiencies; therefore, only two species from weir collections were analyzed in this study, *Fundulus heteroclitus*, and *Menidia menidia*. Both were far more abundant in weir than in trawl collections, and as important forage species that are preyed upon by resident and marine transient taxa, were used in the catch per unit effort estimates (Litvin and Weinstein, 2003).

Table 4. Sampling effort for sites and years used in the meta-analysis. Symbol (-) indicates data not included in comparisons among years (Figures 4 and 5) or no longer collected after restoration criteria were met.

		1999	2000	2001	2003	2004	2005	2014	2015	Total
Upper Bay										
Reference Site										
Mad Horse Creek	Trawl	220	223	189	-	187	192	126	126	1263
	Weir	16	16	14	-	12	14	14	14	100
Restoration Site										
Alloway Creek (P)										
	Trawl	109	143	126	-	126	126	-	-	630
	Weir	14	16	12	-	14	8	-	-	64
Mill Creek (P)	Trawl	187	190	146	-	166	160	126	126	1101
	Weir	16	16	14	-	14	14	14	14	
Brown's Run (P)	Trawl	190	192	168	-	168	168	-	-	886
	Weir	16	16	14	-	14	14	-	-	74
Lower Bay										
Reference Site										
Moore's Beach	Trawl	224	218	189	186	194	188	126	126	1451
	Weir	16	16	14	14	14	14	14	14	116
Restoration Site										
Dennis Township	Trawl	224	210	187	140	191	187	-	-	1139
	Weir	16	16	14	14	14	14	-	-	88
Commercial Township	Trawl	192	191	168	-	168	168	126	126	1139
	Weir	16	16	14	-	14	14	14	14	102

Dominant Nekton:

To provide for consistent comparisons among sites, a sub-set of data comprised of those fishes and blue crabs that made up 90% or more of the individuals captured at a given location was established and then examined for consistency across years and locations (Table 5). The individuals evaluated in Figures 4 and 5 for catch per unit effort and mean length were those that occurred at all locations for each comparison. Other species that were occasionally abundant such as Atlantic menhaden (*Brevoortia tyrannus*), black drum (*Pogonias chromis*) and spot (*Leiostomus xanthurus*), were not consistently captured on the dates and locations that form the data base and so were eliminated from the comparisons. The single exception, young-of-year Atlantic croaker (*Microponias undulatus*), were generally recruited to the sites in fall, and thus, were not compared among early seasonal dates. Changes in mean length were evaluated seasonally, during the spring-summer period (April/May–August), and, in the fall prior to emigration (September-November) when most species leave the marsh creeks to over winter in deeper portions of the estuary or offshore (Able and Fahay, 1998). By examining the length distribution histograms appearing in the monitoring reports required by the utility's permit, supplemented by published data by Able and Fahay (1998), an attempt was made to reduce the influence of "new" recruits and "older" individuals (1+) in the population, especially those taxa that use tidal salt marshes as nursery habitat during their first year of life (Table 6). By establishing these age/length cutoffs, "outliers" have less influence on the length frequency distributions and growth estimates.

Table 5. Reference and restoration marshes with the same dominant nekton captured across all years shown.

	1999	2000	2001	2003	2004	2005	2014	2015
Upper Bay								
Reference Site								
Mad Horse Creek	X	X	X		X	X	X	X
Restoration Sites								
Alloway Creek	X		X		X	X		
Mill Creek	X	X			X		X	X
Brown's Run	X				X			
Lower Bay								
Reference Site								
Moore's Beach	X	X	X	X		X	X	
Restoration Sites								
Dennis Township	X	X		X		X		
Commercial Township		X	X				X	

Table 6. Length frequency distribution cutoffs used to reduce effect of ongoing monthly recruitment to individual species mean lengths. For finfish, individuals that were 1+ or older were also removed from the data base.

Species	Length Cutoff	Removed +1
Cynoscion regalis	< 50 mm	Yes
Morone americana	< 40 mm	Yes
Morone saxatilis	< 25 mm	Yes
Callinectes sapidus	< 30 mm	No

Project Success Criteria:

For both wetland types, the goal was to reintroduce tidal flow to 1) the former salt hay farms by opening the dikes at several locations and enhancing drainage by re-excavating the highest order channel(s) that had filled in during the diked period (Lower Bay sites), and 2) at *Phragmites*-dominated locations by a program of herbicide spraying and controlled burning along with selected hydromodifications to substantially reduce the dominance of this bio-pollutant and allow *Spartina* spp. to recolonize the sites (Upper Bay sites; Weinstein and Balletto, 1999; Teal and Weinstein, 2002). The permit goal was to reduce *P. australis* coverage to ≤ 4% of the vegetated marsh surface, and concomitantly raise *S. alterniflora* and other naturally occurring plant species coverage to ≥ 76% while the restored drainage system would cover about 20% of the remaining surface area. By restoring a natural hydroperiod and the mosaic of interactive structural elements of marsh habitat - intertidal and subtidal marsh creeks, vegetated marsh surface, ponds and pannes - "quality" fish habitat would be provided (Teal and Weinstein, 2002).

Evaluation criteria:

Hedges' *d* (Hedges & Olkin 1985) was used to calculate effect size (*d*). Calculation requires a mean, standard deviation, and sample size for control and experimental populations. In this context, *d* is expressed as the difference between a control and experimental group, reference and restoration sites in this assessment, in terms of standard deviation units, and a negative value of *d* indicates relatively greater values for a given parameter in the control group. The cumulative effect size represents the overall magnitude of the effect present in the study; this value is considered to be significantly different from zero if its confidence limits do not bracket zero (Rosenberg et al., 2000). In addition, Q_{total} is calculated as a measure of total heterogeneity in effect sizes, or a weighted sum of squares analogous to the total sum of squares in analysis of variance (Rosenberg et al., 2000).

Results:

Mean Hedges' *d* and their 95% confidence intervals (Figures 4 and 5) were used to summarize and compare the overall effects of restoration on catch per unit effort (CPUE) and length frequency distributions across marsh types; *Phragmite*s- dominated sites and salt hay farms, and their paired reference marshes. Across the 17-year interval (1999-2015), five to eight species dominated the catches, and four to six species dominated the length frequency distributions. Additionally, composite summaries for mean effects and 95% confidence intervals for all sites combined within Bay regions are also shown and are also calculated as a "grand effects mean" effect for all sites in the upper and lower Bay combined (Figures 4c and 5c).

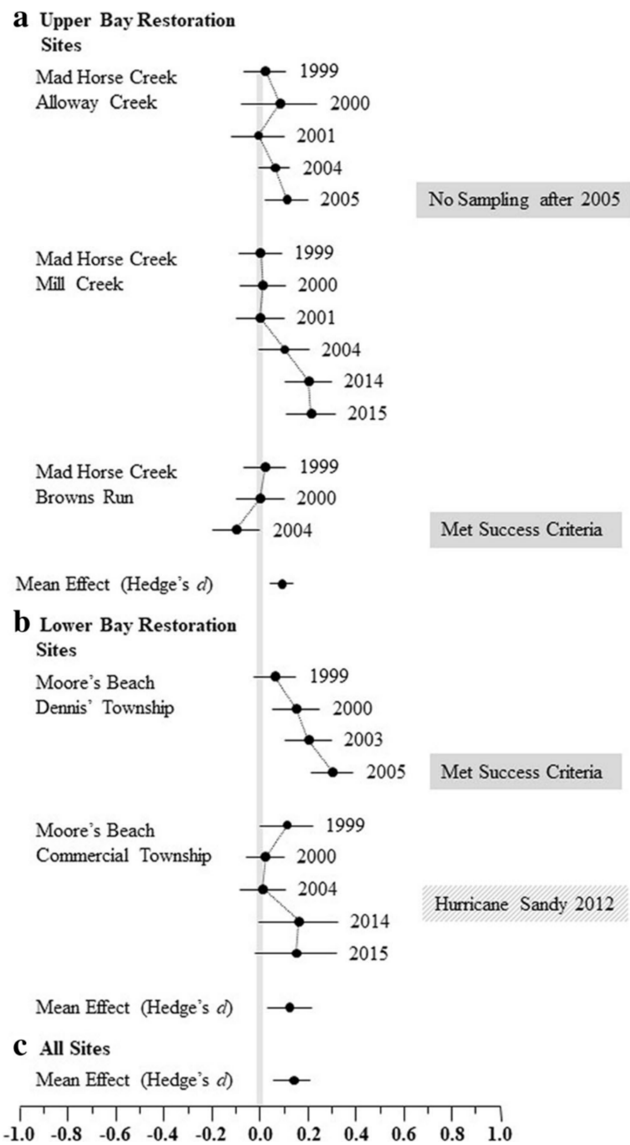


Figure 4. Catch Per Unit Effort (CPUE) for mean effect Hedges' *d* values and 95% confidence intervals (CI), for upper Bay (a) and lower Bay (b) restoration and reference sites. Means are also shown for all upper Bay (A) and lower Bay sites combined, as well as, a grand mean for all upper and lower Bay sites taken together (c). Figures from Weinstein et al., 2019.

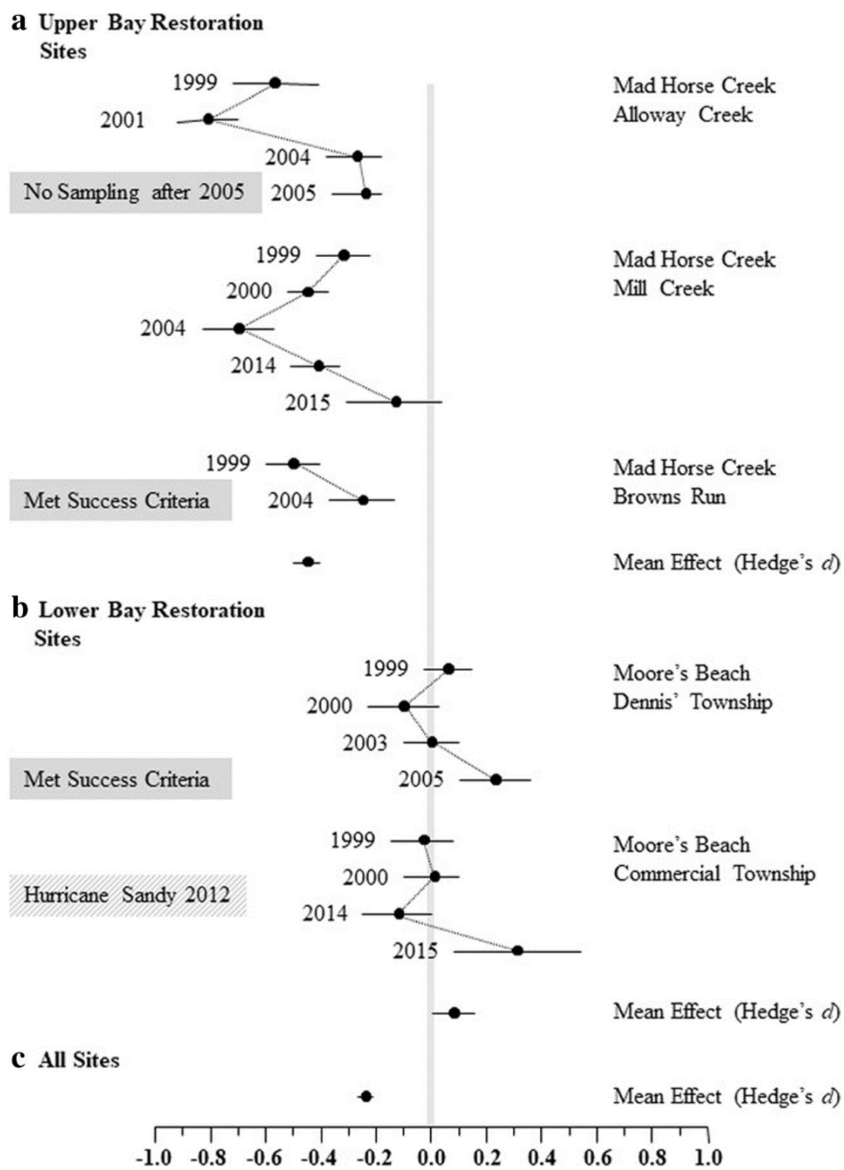


Figure 5. Mean species lengths for mean effects Hedges' *d* values and 95% confidence intervals (CI), for upper Bay (a) and lower Bay (b) restoration and reference sites. Means are also shown for all upper Bay and lower Bay sites combined, as well as, a grand mean for all upper and lower Bay sites taken together (c). Figures from Weinstein et al., 2019.

Both forage and predatory species were present in the dominant groups. The former included bay anchovy (*Anchoa mitchilli*), Atlantic silversides (*Menidia menidia*), hogchokers (*Trinectes maculatus*) and the common mummichog (*Fundulus heteroclitus*). Silversides and mummichogs were not included in the length distribution measurements. Predatory finfish included the tertiary predators striped bass (*Morone saxatilis*) and weakfish (*Cynoscion regalis*), and white perch (*M. americana*) and Atlantic croaker (*Micropogonias undulatus*). Blue crabs (*Callinectes sapidus*) were also abundant in the catches.

In only three instances in the upper Bay, at Alloway Creek in 2005; the last year of trawl sampling, and at Mill Creek in 2014 and 2015, did 95% confidence interval not overlap zero, suggesting that Alloway and Mill Creek CPUE means in those years exceeded those of the reference site, Mad Horse Creek (Figure 4a). The exception to the overall pattern, Brown's Run had a negative mean *d* in 2004 (-0.08; 95% CI, -0.19 to 0.03; $Q_{total} = 41.07$, $p < 0.001$), the last year of sampling when the vegetation/geomorphology criteria were satisfied. The overall mean Hedges' *d* for all upper Bay restoration sites was also positive and the 95% confidence interval did not overlap zero (+0.08; 95% CI, +0.05 to +0.10; $Q_{total} = 1111.16$, $p < 0.001$) (Figure 4a).

CPUE values at lower Bay sites were all positive and different than 0 from the onset year, 1999, through 2015 at Dennis Township, except in 1999 when the 95% CI overlapped zero (Figure 4b). The pattern at the much larger Commercial Township site differed, with all but the 1999 confidence intervals overlapping zero. However, the mean Hedges' *d* for all lower Bay sites combined was positive (+0.15; 95% CI,+0.13 to +0.17**;** $Q_{total} = 1456.73$, $p < 0.001$) (Figure 4b), as was the overall "grand" mean *d* value for upper and lower Bay sites combined (+0.10; 95% CI, +0.08 to +0.11**;** $Q_{total} = 2308.16$, $p < 0.001$) (Figure 4c).

Length frequency distributions at Alloway Creek displayed minimum convergence toward similarity with the reference site Mad Horse Creek; recording a mean negative *d* value in 2005 (7 years later) of -0.55 (95% CI -0.67 to -0.43; Q = 307.85, p<0.001) (Figure 5a). Although, Mill Creek lagged the reference site through 2014; and while conditions appeared to improve somewhat in 2015, the mean *d* value was still negative by the end of the year (Figure 5a). When sampling at Brown's Run terminated in 2004, after the vegetation/geomorphic success criteria were met, the site still had not reached parity in mean length distribution profiles with Mad Horse Creek in 2004. The mean effects value for all upper Bay sites combined for lengths was also negative relative to that of the reference marsh through all years of the study (-0.46; 95% CI -0.49 to –0.43; $Q_{total} = 2916.52$, $p < 0.001$) (Figure 5a).

Restoration trajectories in mean length distributions in the lower Bay at both salt hay farms "out-performed" those of the upper Bay with mean *d* values and their confidence intervals positive at Dennis Creek in the 1999 and 2005 comparisons with Moore's Beach. Although the mean Hedges' *d* value at Commercial Township exceeded that of the Moore's Beach reference site in 2014, all yearly 95% CI overlapped zero indicating that the differences were not statistically significant, and length frequency distributions were essentially the same (Figure 5b). However, the mean effects value for lengths at both lower Bay sites combined also exceeded that of the reference marsh, Moore's Beach, and was positive (+0.06; 95% CI +0.02 to +0.11; $Q_{total} = 434.13$, $p < 0.001$) (Figure 5b). Finally, the grand mean effect for all upper and lower Bay sites combined was negative (-0.27; 95% CI -0.29 to – 0.24; $Q_{total} = 3734.57$, $p < 0.001$) (Figure 5c) suggesting the conditions at the restoration sites did not converge with those of the reference sites by the end of the study.

Discussion:

Scientists engaged in the Estuary Enhancement Program have adopted both restoration ecology and ecological restoration principles; how best to design restoration trajectories that approximate natural ecosystems to the extent practical, and to how to include design characteristics that support and likely enhance secondary production of species that contribute to the economy of Delaware Bay. The latter is summarized in Executive Order 13547 - issued from President Obama's office on July 2010, which, in part, sought to advance policy for improved coastal resiliency to aid coastal communities that rely on natural resources for their livelihood and economic prosperity. The preservation and restoration of coastal wetlands were a centerpiece of EO 13547. Nearly four decades ago drainage features of tidal marshes, especially the smallest tidal creeks, were recognized as critically important to nekton production. Weigert & Pomeroy (1981) commented "our present view of the food web of the marsh and estuary suggests that the preservation of fisheries depends as much on the protection of the smallest tidal creeks as upon production of the marsh and its *Spartina* production." Similarly, Haines noted in 1979 that the 'true' nursery-ground of the estuary is "perhaps not so much the large open water rivers and sounds as the salt marshes and [their] narrow tidal creeks." She noted further that export of marsh plant production may occur "not [necessarily] as particulate detritus but as living organisms." The thesis that salt marshes, with their extensive drainage networks and edge habitat contribute to the production of nekton has since been well established (Gosselink, 1984; Zimmerman and Minello, 1984; Browder et al., 1989; Baltz et al., 1993; Teal and Howes, 2001; Minello et al., 2003).

Thus, in addition to restoring desirable vegetation at the sites, critical focus was placed on restoring to the extent possible, natural drainage features. Prior to construction, published literature was reviewed, and a set of metrics established to describe baseline conditions that would enhance the extent of interface ("edge") between the vegetated marsh surface and the adjoining drainage system. These were adopted by the engineers engaged in the project and have been described in great detail elsewhere (Balletto et al., 2005; Teal and Weishar, 2005; Weishar et al., 2005a, b). These natural drainage features are described in Weinstein et al. (2001) and noted here:

- Tidal creek drainage characterized by fourth or fifth order stream systems, high drainage density, bifurcation ratios, sinuosity and stream length;
- Subtidal refugia for nekton in the highest order streams;
- A wetting/drying cycle, ~ 4.5 h, characterized by sufficient intertidal periods to aerate surficial sediments on the marsh, especially stream bank locations;
- Natural stream bank slopes; and
- Vegetation:open water ratios of about four to one.

It should also be emphasized that only the highest order channels were excavated at the salt hay farms, and the next order "notched" a relatively short distance into the marsh (Weinstein et al., 2000a). No channels were excavated at the *Phragmites*-dominated sites, as these were already opened to tide. It was anticipated that, with adaptive management assistance (Weinstein et al., 1997), the sites would "self-engineer" to a natural stable state (Teal and Weinstein, 2002). The process has been monitored throughout the course of the EEP and summarized in annual progress reports and published literature (Weishar et al., 2005a, b).

Summary:

Monitoring of restoration trajectories at 4,049 ha of tidal wetlands were required to offset nekton losses at a power facility on Delaware Bay. Catch per unit effort (CPUE) and seasonal growth of dominant species were examined over a 17-year period at two reference and five restoration sites. Overall, the restoration process at the EEP sites proceeded fairly rapidly with nekton abundances at restored marshes converging with, or exceeding those of the reference sites, especially in the lower Bay.

Mean Hedges' *d* determined for CPUE in the upper Bay were generally indistinguishable from those of the reference site, except for one restored location. The overall mean *d* for upper Bay restoration sites combined, however, was positive. Lower Bay CPUE values were also positive but several 95% confidence intervals overlapped zero. The CPUE mean effects value for all lower Bay sites combined was also positive.

Length frequency distributions, on the other hand, minimally converged with reference site values in the upper Bay. The mean *d* value for all upper Bay restored sites was negative throughout all years of study, as was the overall upper Bay *d.* Although mean effects for length frequency distributions in the lower Bay were generally higher than those up-Bay, most 95% confidence intervals overlapped zero with few exceptions, suggesting no statistical difference. The mean *d* for lengths at all lower-Bay sites combined was positive. Finally, the grand mean effect for lengths at all sites combined was negative suggesting that nekton growth at the restoration sites did not merge with those of the reference sites by the end of the study.

Other Relevant Studies Conducted on Delaware Bay:

Benayas et al. (2009) conducted a meta-analysis on 89 studies, including several derived from the EEP restoration program. The authors sought to link the enhancement of biodiversity and ecosystem services by ecological restoration. They surveyed the Millennium Ecosystem Assessment (MEA) (2005) classification for ecosystem goods/services that included supporting, provisioning, regulating, and cultural values, but the latter were not included in the analysis. Their findings suggested that supporting/regulation and biodiversity ecosystem services in restored sites was generally higher in restored versus degraded systems, but lower than in reference systems. Median response ratios for biodiversity and combined ecosystem goods/services were 86 and 80% of that in reference systems across a wide variety of biomes including several salt marshes that are the subject of this study. Benayas et al. (2009) concluded that even studies at local scales (such as the EEP) are "likely to lead to large increases in biodiversity and provision of ecosystem [goods] services, offering the potential of a win-win solution in terms of combining biodiversity conservation with socio-economic development objectives." Further, the authors commented "ecological restoration can be effective in restoring natural capital." A similar framework has been adopted for ecosystem valuation of selected Delaware Bay tidal marshes but will be published separately as a site-specific community resiliency case-study.

While tracking the progress of the tidal marsh restoration process over two decades, beginning with engineering designs and in-ground construction, and then monitoring vegetation and geomorphic success criteria, we simultaneously monitored the use of the site by nekton, principally finfish and blue crabs. Several findings from the published results of the nekton monitoring study, principally, the presence/absence of individual species, their abundance and seasonal growth rates formed the basis for the meta-analysis reported herein. It should be stated, however, that a plethora of other studies, funded principally by external resource agencies, mostly federal, have expanded our understanding of the role of the invasive m-haplotype of *Phragmites australis* (Saltonstall, 2002) in tidal marsh ecology of Delaware Bay. Our earliest observations, based on field visits to sites that previously were virtual monocultures of this invasive variety, and a review of the literature (Weinstein and Balletto, 1999) all suggested the *P. australis* was elevating the marsh surface (Rooth et al., 2003;) and dramatically reducing surface microtopography to virtual "billiard table" status (Figure 3C. Larval fish traps placed flush with the marsh surface at Alloway Creek *P. australis* – dominated sites, captured significantly fewer larval and early juvenile common mummichogs (*Fundulus heteroclitus*) than traps placed at recovering sites dominated by *Spartina alterniflora* (Able et al., 2003).

The current study, however, builds upon the efforts of other investigators working on the EEP. In their nine year study (1996-2004) in the lower Bay, which began soon after the dikes were breached (in 1996 and 1997, at the Dennis Township and Commercial Township sites, respectively), Able et al. (2008) noted that the "fish assemblages in subtidal creeks [Moore's Beach, Commercial and Dennis Townships] responded in a highly variable fashion to changes in the restored marshes relative to the reference marsh" They also described the influence of both environmental and larval supply to the mouths of these creeks as independent variables potentially influencing annual recruitment patterns. They did, however, observe more consistent results with their weir samples, commenting that "total abundance started out greater at restored marshes and then gradually diminished through time to converge with reference marsh abundance levels".

At upper Bay sites, patterns were somewhat different. In a three-year study commencing approximately two years after herbicide treatment at Alloway Creek, Grothues and Able **(**2003a, b) reported lack of a trend toward convergence (or divergence) for the fish assemblage at treatment sites compared to control site (areas of *Spartina* dominance within the Alloway Creek system). Although they concluded that the "response to treatment (as a dynamic concept) was small or slow during this 3-yr study", they also noted that the site was still in transition to full *Spartina* coverage. In a second study, encompassing the five-year period 1996 through 2000, Grothues & Able (2003b) noted relatively poor correlations between CPUE and *Spartina* vegetation cover at all sites (Mad Horse Creek, Mill Creek and Brown's Run). They concluded that "species composition of subtidal marsh creeks, a first indicator of habitat quality is weakly influenced by an induced change [herbicide treatment] in the composition of upper Delaware marsh surface vegetation...". But, while noting negative effects of *Phragmites* dominated marsh on larval and small juvenile fishes, there appeared to be little or no effect on the presence of larger fish and decapod crustaceans (Able and Hagen, 2000), as has been previously reported by others (Meyer et al., 2001; Warren et al., 2001; Fell et al., 2003; Jivoff and Able, 2003; Yozzo et al., 2003).

The findings of the present study extend these earlier efforts, and in some cases include sampling out to 2015. We also noted those years when, at a given site, the permitted vegetation and geomorphic criteria were met (Figures 4 and 5). Prior to this period, edaphic algal growth was extensive over most of the open flats (M.P. Weinstein, personal observations), and in vegetated areas where initial sparse growth allowed more sunlight to reach the marsh surface. Edaphic algae, as a source of nutrition, is not only believed to be highly palatable for resident nekton but may be a preferred food source for some (Sullivan and Moncrieff, 1990; Currin et al., 2003). We focused on those species that comprised 90% or more of the individuals collected at each site, and that were mainly comprised of young-of-year. We observed time-dependent trends in both the lower Bay, and for CPUE a tendency toward rapid convergence ($d \geq 0.0$) with the reference sites (Figure 4). The restoration trajectory in CPUE at Brown's Run (Figure 4a), however, was not as distinct as that for all other sites, and although d was negative, the 95% CI (-0.19 to +0.03) overlapped zero in 2004. We even seemed to detect a short-term "setback" in CPUE at Commercial Township in 2014-2015 in the aftermath of Hurricane Sandy when surge created considerable erosion and damage to vegetation on the recovering marsh surface. This did not seem to have as severe, if any, effect at the up-Bay Mill Creek site.

By 2015, *P. australis*' ongoing dominance at the upper Bay restoration appeared to be reflected in relatively poor growth ($d < 0$) in young-of-year nekton captured at Alloway Creek, Mill Creek and Brown's Run, although the 2015 value at Mill Creek tended toward convergence with mean growth of individuals at the reference site (Figure 5) . Of course, this is only one summary datum and the general variability at all sites suggests that additional data will be required for further verification. More so, the overall mean effect (for Hedges d) for the upper Bay sites was negative and non-overlapping zero (Figure 5c) suggesting the potential for growth for nekton at the restoration sites overall, still lagged that of the reference site, Mad Horse Creek. Interestingly, our previous, and rather extensive, studies of the flow of nutrients from dominant vegetation in the marsh, including not only from *P. australis* and *S. alterniflora*, but also microphytobenthos and phytoplankton, not only attests to the value of tidal salt marshes in supporting the estuarine food web and secondary production, but that this support *includes* nutrients from the C3 plant, *P. australis* as a source of C, N and S in the trophic spectrum of the upper Bay (Currin et al., 2003; Wainright et al., 2000; Litvin and Weinstein, 2003, 2004; Litvin et al., 2014; Weinstein and Litvin, 2015; Weinstein et al., 2000a, 2005, 2014). In this same context, our previous work on condition of dominant finfish in *Phragmites* dominated marshes on the Hudson River suggested that although somatic criteria, length and weight at age, were virtually identical in the two populations, common mummichogs (*Fundulus heteroclitus*) and white perch (*Morone americana*) captured at the *Phragmites* dominated site were unable to lay down sufficient energy reserves in the form of triacyl glycerides (TAG) and free fatty acids for overwintering/migration at the end of the growing season (Weinstein et al., 2009, 2010). The latter effort was independently verified by Dibble & Meyerson (2012) who examined *Fundulus heteroclitus* captured in *Phragmites* dominated, tidally restricted marshes, and at undisturbed reference marshes, at several locations in New England. *F. heteroclitus* captured at the former sites also exhibited significantly reduced lipid reserves and increased lean biomass relative to fish collected at the reference locations.

Growth of individuals at the lower Bay restoration sites (Commercial and Dennis Townships) also converged relatively rapidly with that of the reference site (Moore's Beach), both soon after the vegetation/geomorphic criteria were met at Dennis Township and by 2014, at Commercial Township. Hedges' *d* values actually exceeded mean values at both locations (Figure 5B). Similarly, the overall mean effect for all sites examined together was positive (Figure 5C).

Our findings also parallel those summarized in Strange et al. (2002) who suggested that full recovery of restored marsh may take many years. In our work, transient species were reported at significantly higher densities in restored compared to the reference marshes in the lower Bay (Figure 4), which at the generally lower elevations made hydroperiod a potential driving variable for nekton abundance in the newly opened sites on Delaware Bay. The observation that lengths of seasonally resident species in restored marshes lagged those of reference marshes has also been reported elsewhere (Minello and Zimmerman, 1992; Minello and Webb, 1997) and has led to the suggestion that growth or condition of individuals may be a better proxy for restoration success than structural criteria, at least over the short term (Miller and Simenstad, 1997).

The observation that nekton were equally abundant across restored sites (mean CPUE $d \geq 0.0$) shortly after treatment for *Phragmites* in the upper Bay (Figure 4a), and soon after the dikes were opened in the lower Bay (Figure 4b) at all restoration sites except Brown's Run (when sampling was terminated in 2004) might at first suggest that the vegetation/geomorphic criteria established for the project as recorded in the EEP's 404 permit (*P. australis* coverage to ≤ 4% of the marsh plain, with concomitant increase in *S. alterniflora* coverage to ≥ 76%) was too stringent. Yet average size of nekton captured at *Phragmites*-dominated sites generally lagged those of comparable species at the reference site (Figure 5a), supporting earlier findings that *Phragmites*-dominated marshes limit the production of nekton that occupy them. It is also abundantly clear that monitoring should not only include structural (vegetation and geomorphic) criteria but should be extended out along a time axis that assesses secondary production and other functional criteria. That time axis should be long-term, likely a decadal time frame, when growth and secondary production estimates of nekton populations at restored sites converges with those at reference sites.

Conclusions for Meta-analysis of Wetland Literature:

As highlighted by the literature review and summary of restoration activities in the Delaware Bay tidal wetlands, it is increasingly evident that the criteria for wetland restoration success - especially for the return of ecosystem functions - requires long-term monitoring. Whether for simple structural characteristics like vegetation cover or especially faunal usage and secondary production, the time frame for restoration success should extend out to decadal, or more, time frames (Craft et al., 1999; Strange et al., 2002; Talley and Levin, 2001). Although more difficult to design for, 404 permit criteria must go beyond simple short-term growth and survival estimates for vegetation, clearly woefully inadequate with today's advances in restoration ecology research, to a more holistic and integrated program assessing not only wetland structure and function but the social values contributing to community resilience in a rapidly changing world.

Section 2. In-situ measures of Plant Community and Hydroperiod: The Wading River as a Regional Priority Area:

Project Leads: Meiyin Wu, Michael P. Weinstein, and Robert Hazen

Introduction:

The Wading River and estuary are recognized as regionally important due to the minimal development and impacts in the watershed. This section describes recognized metrics to evaluate baseline conditions within minimally disturbed estuarine wetland complexes. A "bound of expectation" (Weinstein et al., 1997) can be developed for a range of conditions that provide structural and functional success criteria for restoration projects. For example, criteria like vegetation coverage (type, biomass and density), hydroperiod, drainage density, bifurcation ratios, sinuosity, stream lengths and stream order, percent standing water, and invasive species coverage could all be tabulated, and central tendency and variance (ultimately from a series of reference sites) calculated. Restoration endpoints falling within these bounds would be considered successful. This section also describes ecological and hydrologic data that were collected to establish baseline conditions within this estuary system.

Study Area:

Facets of the following tasks involve the understanding of New Jersey tidal wetlands with emphasis on the Wading River tidal wetland complex. This study area is a minimally disturbed watershed located in the area between the mouth of the Mullica River and the headwaters of the Wading River and its tributaries, hereinafter the Wading River Complex (WRC), a total area of 590 km^2 (230 mi^2), (Figure 6). Here, land use is minimally impacted by humans, and the watershed is dominated by forest and wetlands. Suburban development, agriculture, and "barren" lands each make up about 1% of the uplands whereas ~ 15% of the lower Wading River basin consists of riverine reaches and wetlands. The latter comprise tidal salt, tidal fresh, and freshwater wetlands. Tidal salt marshes extend to ~ 4.5 miles upstream of the river mouth.

Because of its location in the Pinelands National Reserve, freshwaters in the system are typically characterized as acidic and with low nutrient levels. However, tributaries draining agricultural and developed lands may display elevated pH and total dissolved solids (Zampella et al., 2007). Furthermore, compounds such as nutrients and dissolved organic carbon are known to decrease with salinity and affect primary production in wetlands. The WRC was selected to minimize the confounding influence of urban development on baseline measurements. The physical and biologic characteristics of this wetland can be used as a benchmark baseline for informing preservation priorities and as guidance for tidal wetland restoration.

Figure 6. The Wading River watershed and estuary system.

Establishment of Quantitative Metrics Relating Geomorphological and Ecological Characteristics:

Several components of the geomorphic (planform) setting in Atlantic-coast riverine wetlands are believed to be critical to the transfer of primary production to consumers and in determining the rates at which secondary productivity occurs in downstream waters. For tidal salt marshes the "classic" marsh drainage is a fourth- or fifth-order system (*Metric 1*) characterized by high drainage density (*Metric 2*) and high bifurcation ratios (*Metric 3*), extensive sinuosity (*Metric 4*), long stream lengths (*Metric 5*), and open water to intertidal marsh ratios of about 4:1 (*Metric 6*) (Figure 7). The marsh surface is highly reticulated (*Metric 7*) with plant tussocks interspersed with first-order streams (National Research Council, 1995). The small streams function much like a capillary network in the efficient exchange of materials (detritus, nutrients, dissolved organic matter, phytoplankton, fauna, etc.). Additionally, small nekton have access to virtually the entire vegetated marsh surface via first-order rivulets (*Metric 8*) (Litvin and Weinstein, 2003; Minello et al., 2003). About 2% of the marsh surface in many Atlantic-coast marshes consists of standing water in pools and ponds (*Metric 9*) and provides important nursery areas for resident marsh species. Some of the pools and ponds include open marsh water management (OMWM) areas designed to control mosquito populations. Microtopography associated with high drainage density, sinuosity, and marsh surface reticulation is equated with extensive "edge," (*Metric 10*) thus enhancing material exchange and trophic relays (i.e., the role of nekton in the transport of production across marsh landscapes to downstream waters (Litvin and Weinstein, 2003). Depositional creek banks are common features of undisturbed marshes that may serve as predation refugia for the earliest life stages of nekton, substrates for benthic microalgal standing crop (*Metric 11*) and production, especially during the summer and early fall when shading by the macrophyte canopy is at a maximum.

Drainage Parameters

Channel Order
4^{th} or 5^{th} for Most Marshes

Sinuosity
$S = L_{sin}/L_{str}$

L_{sin} = Sinuous length;
L_{str} = Straight-line length

Bifurcation Ratio
$R_b = N_0/N_{0+1}$

N_0 = channels in a given order;
N_{0+1} = # next order

Drainage Density
$D = L_T/A$

L_T = Total length of channels;
A = Marsh Area

Stream Length Ratio
$R_L = L_0/L_{0+1}$

L_0 = Length in 1^{st} order;
L_{0+1} = Length in next order

"CLASSIC" SALT MARSH

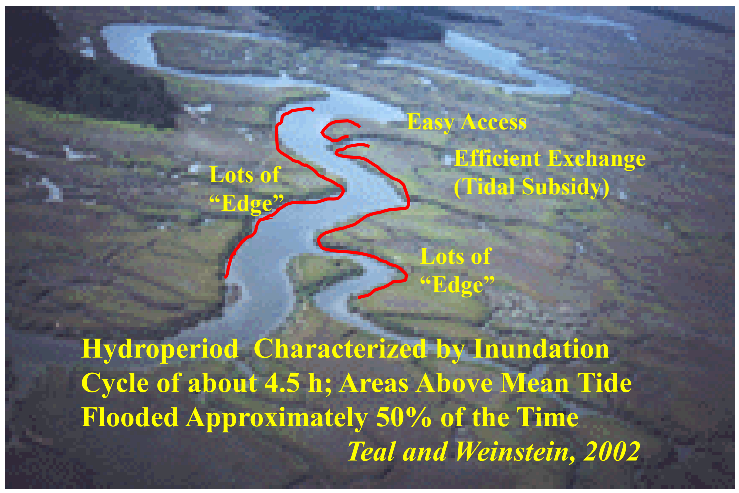


Figure 7. Salt marsh hydro-geomorphic metrics that can be used to assess the integrity of a wetland complex.

Typically, the intertidal low marsh is flooded for approximately a 4.5 hour hydroperiod (*Metric 12*) during mean tides, thus allowing for extended access to the marsh plain by nekton and other consumers, and for the effective removal of toxins and the exchange of nutrients to support macrophyte growth (Able et al., 2000; Kneib, 1997). During low tides, the surface of the marsh sediments is exposed and aerated. Additionally, in a "healthy" marsh with unrestricted tidal exchange, marsh plants decompose relatively quickly and may be available to consumers by the end of the first growing season (Kneib, 2003). Benthic microalgae and many phytoplankton with their high palatability and food chain efficiency, are also readily assimilated by many consumers in the marsh (Weinstein, 2007; Minello et al., 2012; Beck et al., 2003).

Each of these traits describing the intertidal marsh surface and the associated drainage network are critical to the efficient exchange of nutrients and organisms, ultimately to the estuary, where the preponderance of nekton reside during their first year of life. In addition to metrics 1-12 for tidally influenced sections of the WCR, disturbance history (*Metric 13*), vegetation composition (areal coverage) to the lowest practical taxon, stem density and biomass, and "vigor" (with chlorophyll-a content as a surrogate) (*Metrics 14, 15, 16, 17*) will also be critical to "functional" assessments.

Five transects in the WCR were established as part of this study (Figure 8) and data were collected to determine hydroperiod (Metric 12) and evaluate plant species characterization (Metric 13). Ecological plant data were collected along transects A, B, and C (Figure 8, Tables 7 - 11) and hydrological data were collected along Transects D and E (Table 15) where the hydroperiod was determined.

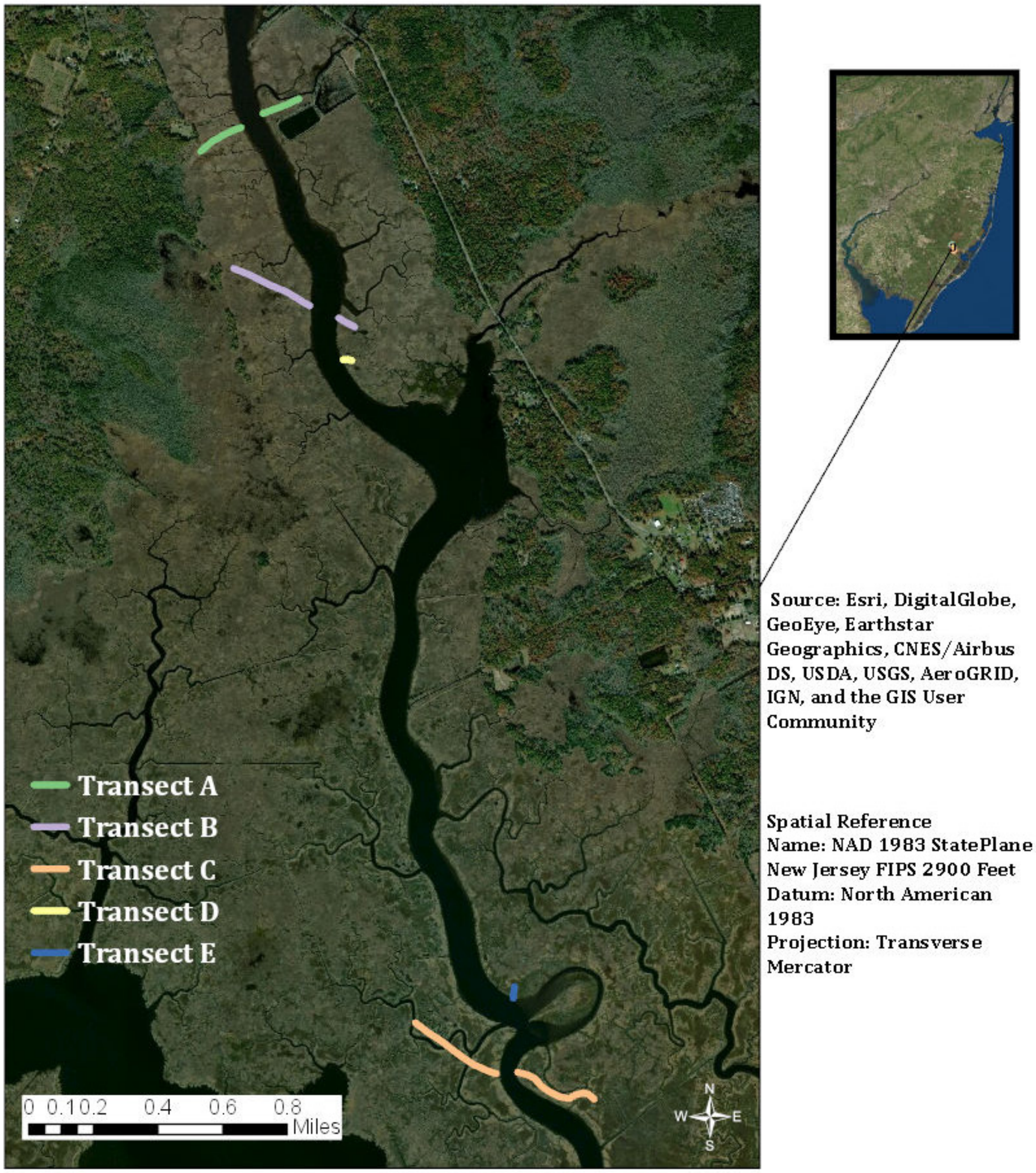


Figure 8. Transects for ecological data collection (A, B, C) and hydrographic data collection (D, E).

Ecological Data:

A series of 1-meter quadrats were established along three transects extending upgradient from the Wading River on both sides of the bank. The transects ranged in length due to vegetative conditions. Two transects (Transects A and B), were in freshwater reaches of the system while Transect C was established in more saline reaches. Due to very similar community characteristics, data from transects A and B were aggregated and summarized together. Data from the individual transects are reported in Tables 9 – 11 and 12 – 14.

Nine different vegetative species were identified (Table 7) within the quadrats along the three transects. All ecological data are summarized in Figure 9, which represent a reduction of the detailed species composition data in Tables 9 – 11 and cover composition data in Tables 12 – 14.

Table 7. Key to Species Identification

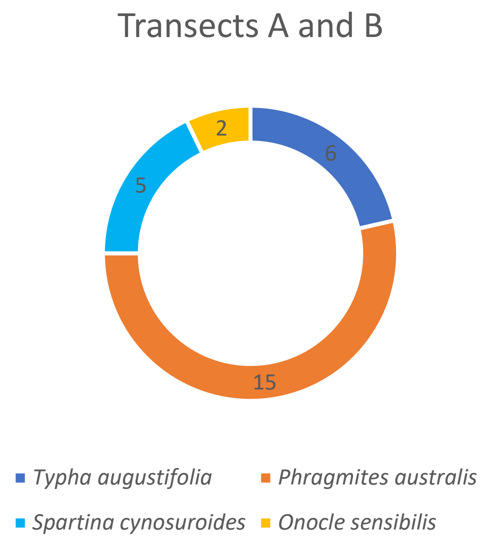
Abbreviation	Scientific Name	Common Name
SA	*Spartina alterniflora*	Cordgrass
OS	*Onocle sensibilis*	Sensitive Fern
AA	*Peltandra virginica*	Arrow arum
SC	*Spartina cynosuroides*	Big cordgrass
SP	*Spartina patens*	Salt hay
DS	*Distichlis spicata*	Spike grass
PA	*Phragmites australis*	Common reed
TA	*Typha augustifolia*	Narrow-leaf cattail
SO	*Scirpus olneyi*	Three square
UNK		Unknown tree species

Table 8 shows the average total plant cover found in quadrats grouped by the salinity categories (A, B, fresh water, C saline water). The coverage, as indicated by percent cover and wet weight are greater in the freshwater area. The mean plant coverage for transects A and B is 89.6% and 68.4% for C.

Table 8. Plant coverage of the dominant species and average wet weight within each quadrat grouped by transect and salinity category.

Transect	Percent Cover	Wet WT. per Quadrat (g)
A & B - Fresh water	89.6	45.6
C - Saline	68.4	22.0

Figure 9 (data summarized from Tables 9 - 11) shows the differences in plant species composition from the combined Transects A and B and the individual transect C. The composition and cover within Transect A and B were found to be highly redundant and thus combined for assessments. The species composition reflects this difference in habitat conditions. Quadrats along Transects A and B, situated in the upper part of the estuary and reflecting lower saline conditions, were dominated by *Phragmites australis, Spartina cynosuroides,* and *Typha augustifolia.* The quadrats along the lower Transect C were dominated by *Spartina alterniflora* and *Spartina patens.*



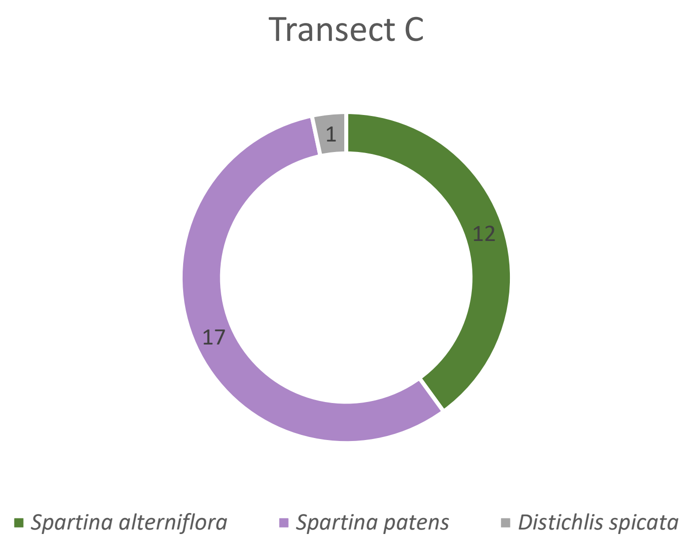


Figure 9. Dominant species characterization from quadrat surveys. Transects A and B summarize data from Tables 9 and 10, respectively. Transect C summarized from Table 11.

Table 9. Species Composition of Transect A (Data summarized in Figure 9)

Quadrat	Plant Cover (%)	Dominant Species	Cover of Dominant Species (%)	Other Species
1	70	TA	93	AA
2	64	PA	100	N/A
3	76	TA	100	N/A
4	76	TA	87	AA
5	96	PA	100	N/A
6	100	PA	100	N/A
7	70	TA	96	AA
8	70	SC	92	AA
9	72	SC	92	AA
10	35	SC	59	AA
11	50	SC	98	TA
12	40	PA	100	N/A
13	60	PA	59	OS

Table 10. Species Composition of Transect B. (Data summarized in Figure 9)

Quadrat	Plant Cover (%)	Dominant Species	Cover of Dominant Species (%)	Other Species
9	40	PA	100	N/A
10	40	PA	100	N/A
11	70	PA	100	N/A
12	44	PA	100	N/A
13	56	PA	100	N/A
14	48	PA	100	N/A
15	48	PA	100	N/A
16	56	PA	76	TA
17	75	TA	100	N/A
18	36	PA	100	N/A
19	24	SC	100	N/A
20	44	TA	100	N/A
21	50	PA	37	OS, Cedar, TP
22	69	OS	66	PA, TA
23	71	OS	55	PA, TA

Table 11. Species Composition of Transect C. (Data summarized in Figure 9)

Quadrat	Plant Cover (%)	Dominant Species	Cover of Dominant Species (%)	Other Species
1	70	SA	65	SP
2	60	SA	60	SP
3	60	SA	50	SP
4	70	SA	55	SP
5	60	SA	100	N/A
6	65	SP	70	SA
7	75	SP	70	--
8	85	SP	100	N/A
9	90	SP	50	--
10	80	SP	70	SA
11	95	SP	100	N/A
12	85	SP	70	SA
13	50	SA	100	N/A
14	40	SP	50	SA
15	90	SP	70	SA
16	100	SP	47	DS
17	45	SA	45	SP, DS
18	55	DS	55	SA
19	83	SP	38	SA, SO, DS
20	84	SP	60	SA
21	86	SP	55	SA
22	90	SA	70	SP
23	65	SA	63	SP
24	65	SA	100	N/A
25	98	SP	69	DS
26	69	SA	61	PA, DS
27	100	SP	95	DS
28	100	SP	85	SO
29	78	SA	58	DS, SP
30	80	SP	72	DS, SA

Table 12. Plant cover characteristics of Transect A. Summarized in Table 8.

Quadrat	Species	Plant Cover (%)	Stem Density (stem/m^2)	Height (cm)	Flowering (Y or N)	Live Plant Weight (g)	Dead Plant Weight (g)
1	AA	7	4	70	N	13.9	0
1	TA	93	56	215	Y	61.2	45
2	PA	100	41	153.92	N	85.8	58
3	TA	100	24	150.88	N	79	0
4	AA	13	1	41.15	N	0.6	0
4	TA	87	17	153.92	N	0	45.4
5	PA	100	43	204.22	Y	61	0
6	PA	100	40	300	Y	67	0
7	AA	4	3	40	N	16.9	0
7	TA	96	57	110.16	Y	37	21.17
8	AA	8	12	70	N	24.8	17.1
8	SA	92	93	130	Y	44.5	23.6
9	AA	8	5	100	N	21.1	0
9	SS	92	52	170	Y	61.7	0
10	AA	41	14	50	N	21.1	0
10	SS	59	44	140	Y	17.2	0
11	TA	2	1	180	Y	0	0
11	SS	98	47	150	Y	51	28.9
12	PA	100	150	150	Y	41.4	48.6
13	OS	20	4	70	N	3.8	0
13	TP	21	4	60	N	13.2	0
13	PA	59	103	200	Y	177.6	0

Table 13. Plant cover characteristics of Transect B. Summarized in Table 8.

Quadrat	Species	Plant Cover (%)	Stem Density	Height (cm)	Flowering (Y or N)	Live Plant Weight (g)	Dead Plant Weight (g)
9	PA	100	125	170	Y	101.5	40.6
10	PA	100	94	220	Y	122	0
11	PA	100	114	220	Y	83.6	0
12	PA	100	66	82	Y	72.7	21.7
13	PA	100	110	130	Y	70.7	66.7
14	PA	100	141	125	Y	98.3	33.5
15	PA	100	156	160	Y	59.1	0
16	TA	24	2	175	N	23.5	0
16	PA	76	99	250	Y	24.2	0
17	TA	100	44	160	Y	23.6	29.5
18	PA	100	119	170	Y	34.1	0
19	SC	100	52	180	N	56.3	16.1
20	TA	100	140	160	N	36.4	53.6
21	TP	14	2	70	N	12.6	0
21	Cedar	19	1	N/A	N	0	0
21	OS	30	51	50	N	18.2	0
21	PA	37	125	170	Y	63.6	33.7
22	OS	14	2	40	N	13.7	0
22	TA	20	6	160	N	68.1	0
22	PA	66	111	130	N	71.5	76.8
23	OS	19	22	50	N	13	0
23	TA	26	11	140	N	47.5	23.8
23	PA	55	117	210	Y	37.8	64.1

Table 14. Plant cover characteristics of Transect C. Summarized in Table 8.

Quadrat	Species	Plant Cover (%)	Stem Density	Height (cm)	Flowering (Y or N)	Live Plant Weight (g)	Dead Plant Weight (g)
1	SA	65%	1952	40	Y	23.8	14.4
1	SP	35%	720	40	Y	0	13
2	SA	60%	896	40	N	23	17.1
2	SP	40%	1120	40	Y	3.1	17.4
3	SA	50%	1968	40	N	23.1	12.1
3	SP	50%	1008	30	Y	13.7	16.8
4	SA	55%	1440	20	N	25.1	0
4	SP	45%	2416	30	N	5	23.2
5	SA	100%	2384	50	Y	33.1	25.4
6	SA	30%	768	70	Y	37.5	12.9

Table 14. Plant cover characteristics of Transect C. Summarized in Table 8.

Quadrat	Species	Plant Cover (%)	Stem Density	Height (cm)	Flowering (Y or N)	Live Plant Weight (g)	Dead Plant Weight (g)
6	SP	70%	1920	40	Y	37	14.2
7	SA	30%	896	40	Y	27.2	0
7	SP	70%	2848	30	N	16.2	24.3
8	SP	100%	3200	40	Y	24.1	33.7
9	SP	50%	12800	30	Y	16.7	23.7
9	SA	50%	12800	30	Y	30.1	0
10	SP	70%	1840	40	N	30.4	35
10	SA	30%	528	40	Y	26.4	0
11	SP	100%	2928	30	Y	23.7	38.3
12	SA	30%	992	50	Y	32.9	0
12	SP	70%	1344	40	N	29	24.2
13	SA	100%	2032	100	Y	60.6	0
14	SA	50%	1136	100	Y	35.2	17
14	SP	50%	896	40	Y	4.8	0.6
15	SA	30%	480	180	N	33.7	0
15	SP	70%	7200	50	Y	31.2	5
16	Unk Tree	6%			N	0	0
16	DS	47%	464	35	Y	12	0
16	SP	47%	2592	40	Y	31.3	0
17	SA	45%	224	40	N	38.5	0
17	DS	30%	992	35	N	13.3	0
17	SP	25%	800	33	N	38.5	0
18	DS	55%	1440	25	N	15.7	0
18	SA	45%	176	35	Y	15.9	0
19	SP	38%	560	30	N	7	0
19	SA	20%	288	56	N	19.7	0
19	DS	25%	480	33	N	11.6	0
19	SO	17%	400	60	N	25.8	0
20	SA	24%	496	75	Y	32	0
20	SP	60%	1072	25	Y	5.4	6.7
21	SA	16%	192	45	N	30.7	0
21	SP	100%	5584	55	N	49.3	0
22	SA	70%	368	40	N	26.1	0
22	SP	30%	768	35	N	16.7	0
23	SA	63%	272	45	N	24.7	0
23	SP	37%	560	40	N	13.9	0
24	SA	100%	832	80	N	24.5	0
25	SP	69%	1312	26	Y	6.1	0
25	DS	31%	256	30	Y	26.5	0
26	PA	14%	15	90	N	-	-
26	SA	61%	640	80	N	10.7	17.9

Table 14. Plant cover characteristics of Transect C. Summarized in Table 8.

Quadrat	Species	Plant Cover (%)	Stem Density	Height (cm)	Flowering (Y or N)	Live Plant Weight (g)	Dead Plant Weight (g)
26	DS	25%	240	30	Y	0	1
27	SP	95%	2192	20	Y	17.2	3.6
27	DS	5%	48	20	Y	19.1	0
28	SO	15%	160	40	N	0.3	0.7
28	SP	85%	1248	30	Y	14.5	1.8
29	DS	15%	528	40	Y	0	0.7
29	SA	58%	592	50	N	6.3	0
29	SP	27%	1168	45	Y	0	5.3
30	DS	10%	64	25	Y	1	0
30	SA	18%	880	50	N	13.8	2.6
30	SP	72%	1536	50	Y	14.1	0.8

Water-Level Data:

During marsh flooding small fish can swim among the plant stems and feed on the many resident invertebrates. This essential characteristic of tidal marshes is vital to the ecosystem and survival for juveniles of the commercial fishery species including striped bass, bluefish, and others as it offers safe foraging habitat. The two significant habitat areas of the studied marsh were *Spartina* and *Phragmites* dominated. *Phragmites* dominated along transect D in the low salinity/freshwaters of the upper reaches of the estuary while *Spartina* species dominated the more saline areas of transect E in the lower reaches, as shown in Figure 8.

As the tide rises it fills the channel and eventually spills over the marsh plain and pools over the vegetation. Water-level data collection was conducted to determine the marsh hydroperiod, or the time the marsh plain is inundated, and plant stems are covered with water. Water level was determined by setting eight stakes along two transects with color changing tape. The tape recorded the maximum water level elevation and was readable after the tide went out. The flood depth at each stake is shown in Table 15.

Along this stretch of the Wading River overbank flooding occurs approximately 2 hours and 10 minutes following low tide. The marsh is then flooded by up to 4.75 and 4.5 inches at Transects D and E, respectively (Table 15). The flood depths on the marsh plain ranged due to the varying elevation of the marsh. Flood stage over the marsh can last upwards of 7 hours before waters recede back into the channels as tides ebb.

The most proximate continuous tide gage is located at the mouth of Great Bay at Shooting Thorofare (Figure 10). The tidal range on 6/1/2017 was 3.08 ft (Figure 11).

Table 15. Flooding depth and hydroperiod determination along Transect D and E. Data reflect conditions on 6/1/2017.

Transect D Phragmites Dominated		Transect E Spartina Dominated	
Stake Number	**Maximum Flood Depth (inches)**	**Stake Number**	**Maximum Flood Depth (inches)**
1	4.75	1	4.50
2	1.50	2	0.00
3	1.50	3	0.00
4	2.75	4	0.50
5	2.00	5	0.50
6	2.75	6	1.75
7	2.75	7	2.50
8	3.00	8	3.50

Comments:
Low Tide time: 10:49
Bank full time: 12:58
High tide time: 16:24
Hydroperiod: (6 h 52 min) 3 h 26 min time between bank full and high tide

Conditions:
Half-moon, winds at 15 knots against tide

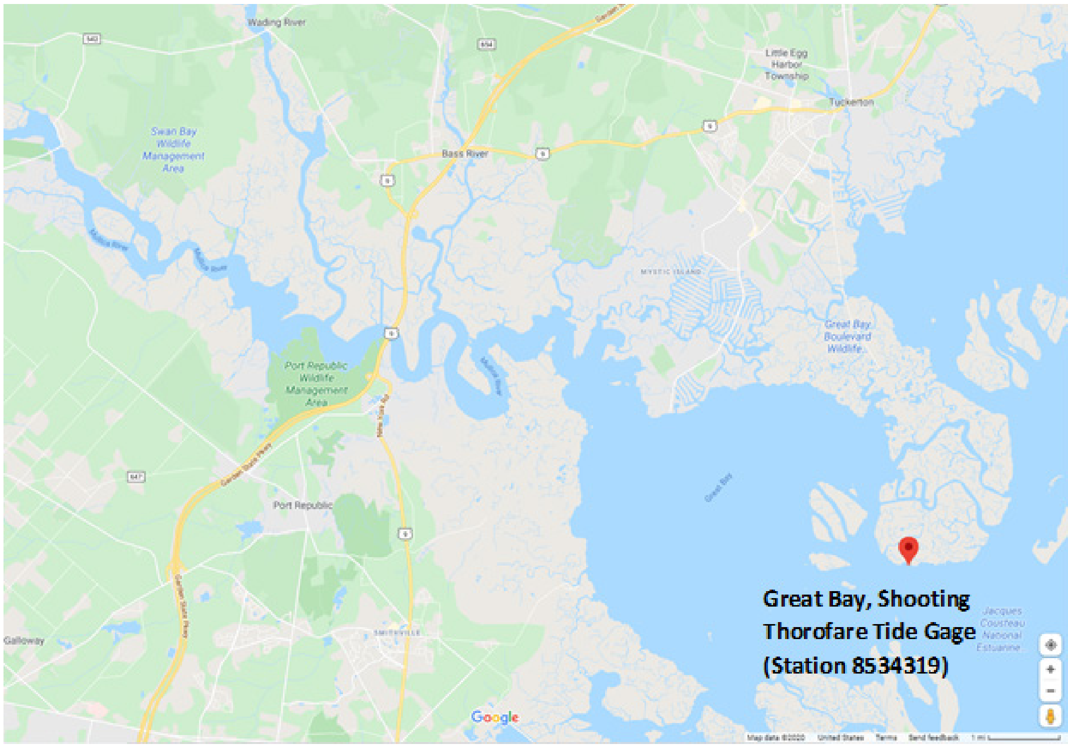


Figure 10. Location of the study area along the Wading River tidal estuary and tide gage at Great Bay, Shooting Thorofare, NJ.

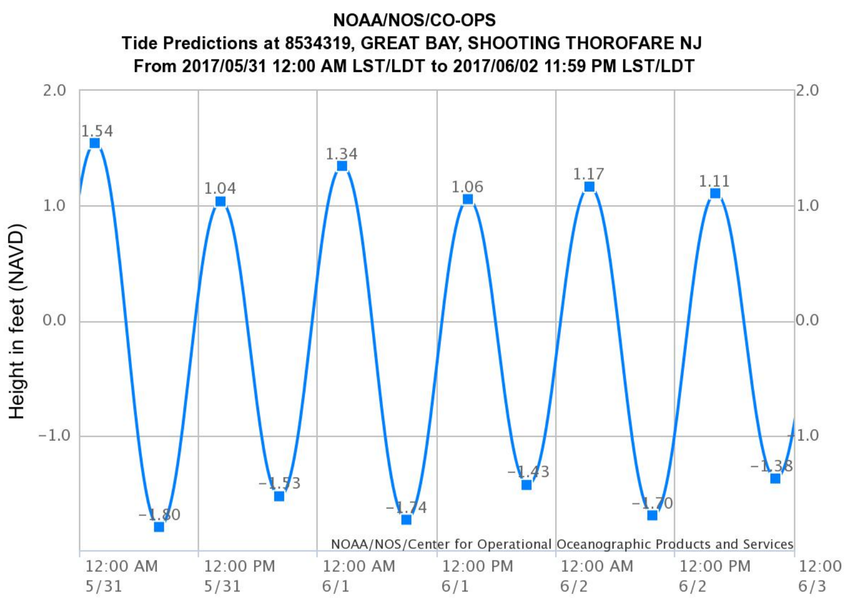


Figure 11. Tide cycle for the day previous and day following the 6/1/2017 sample event. Historic data for this site can be found at NOAA's Tides and Currents page.[3]

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[3] https://tidesandcurrents.noaa.gov/noaatidepredictions.html?id=8534319&units=standard&bdate=20170601&edate=20170601&timezone=LST/LDT&clock=12hour&datum=MLLW&interval=hilo&action=dailychart

Conclusions for In-situ measures of Plant Community and Hydroperiod: The Wading River as a Regional Priority Area:

Plant community data was collected along three transects. Two transects in the upper part of the estuary were found to be very similar and data were thus evaluated together. In-situ ecological data collected along Transects A and B show that quadrats along the transects reflect lower salinity conditions and were dominated by *Phragmites australis, Spartina cynosuroides*, and *Typha augustifolia*. The third transect was situated in the lower part of the estuary. The quadrats along Transect C were in more saline conditions and were dominated by *Spartina alterniflora* and *Spartina patens*. Evaluation of water levels along two transects on a single day provided sufficient information to determine that the marsh may flood up to 4.75 inches and flood stage over the marsh can last upwards of 7 hours before the tides ebb.

Section 3. Establishment of Quantitative Metrics that Relate Geomorphological and Ecological Characteristics:

Project Lead: Danlin Yu

Using High Resolution Multi-Spectral Imaging to Characterize Vegetation Composition, Biomass Stem Density, and Stream Characteristics:

High resolution satellite images for a 20 km^2 area of the Wading River Complex were acquired from WordView 2 (https://www.usgs.gov/centers/eros/science/usgs-eros-archive-commercial-satellites-cdp-imagery-worldview-2) and Pleiades (https://eos.com/pleiades-1/) satellites for summer and winter from 2011 – 2018 (2012 summer image is unavailable). Imagery from both sources has a 50-centimeter resolution. All the images were geo-rectified prior to being acquired. The images were projected with a UTM 18N coordinate system and WGS 84 datum. The images were not re-projected to the New Jersey State Plane coordinate system to ensure accurate digitization and ensuing data extraction as well as to avoid potential information loss during the resampling and assessment processes (Figure 12).

The Wading River Complex was digitized from the satellite imagery in two formats: one in polygon, and the other in polyline format using ArcGIS software (Version 10.8). The polygon layer was developed to measure the total water surface area of the river in the complex. This layer was carefully digitized for winter and summer periods for all years (2011 – 2018) and compared. Analysis of the digitized polygons show that no detectable water surface area changes are observed during the eight years. The polyline layer was used to measure the complexity of the river system in the study area. Similarly, the digitized polyline shows minimal to no detectable changes to stream channel orientation over the eight years.

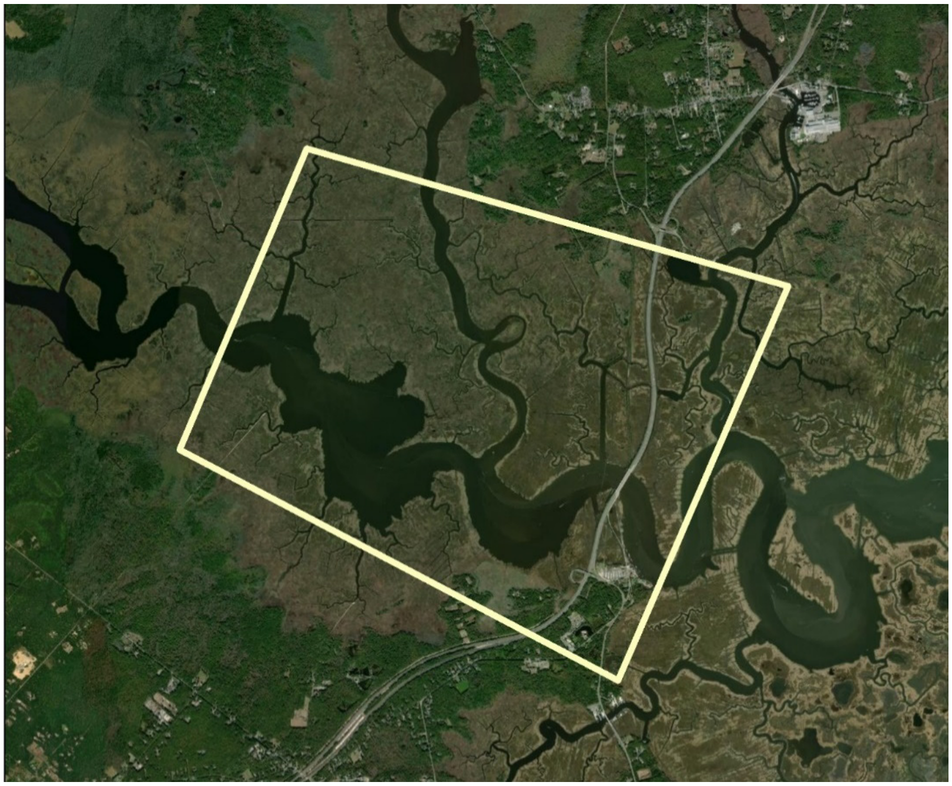

Figure 12. Regional location within the WRC where satellite imagery was evaluated. Aerial imagery shown is courtesy of ESRI services.

The digitization and evaluation from high-resolution remote sensing data suggests that the hydrologic system in the Wading River wetland complex area has sustained little to no significant changes over the eight years. Since the wetland setting and hydrologic structure and the associated complexity is the foundation for the secondary production of commercial and recreation fish species, especially nekton and their forage base, the digitization exercise suggests that any changes in fish species in the WRC are likely not related with changes in the structure of the river system.[4]

Coupling Between In-Situ Data and Remote Sensing Data:

Based on image inspection and land-use classification from the 2016 National Land Cover Database (NLCD) (Jin et al., 2019), five primary cover classes in the study area were identified. The land cover codes, and land cover types are: 11 – Water, 20 – Developed (contains 21 – 24 subclasses of the NLCD), 40 – Forest (contains 41 – 43 subclasses of the NLCD), 90 – Woody Wetlands, and 95 – Emergent Herbaceous Wetlands. A training data set (polygon layer) for supervised image classification was generated utilizing the Semi-automatic Classification Plug-in tool for QGIS (Version 3.12) (https://www.qgis.org/).

From the training data set and the satellite images, three primary machine learning algorithms for supervised image classification, namely, the random forest algorithm (RF), the support vector machine algorithm (SVM), and the neural network algorithm (NN) were tested using the “raster,” “RStoolbox,” and “caret” packages in R (Version 3.6)[5]. This assessment does not

[4] The two digitized shapefiles are included with this report as supplemental material.
[5] Detailed R scripts used to acquire the information with relevant packages and comments are included with the report as supplemental material.

include the summer of 2012 since only a panchromatic image was able to be obtained and did not provide the level of detail needed for this assessment. The average accuracy is calculated from the 5-fold cross-validation confusion matrix on each year, by determining the correspondence of the classified pixels to the original values at the training sites.

Cross-validation measures suggest that the random forest algorithm produced the best classification accuracy with an overall accuracy of approximately 94% (ranging from 84% - 96%) and Kappa index ranging from 0.80 – 0.96. Detailed classification accuracy for each year is reported in Table 16.

Using the random forest algorithm, an image classification for all years (2011 – 2018) for both the summer (excluding 2012) and winter timeseries was produced. The resultant images are used to conduct land cover change analysis and summarize the change of each land cover types year by year. The information was used to generate the system dynamic simulation. Although the accuracy statistics show the supervised classification with random forest algorithm are acceptable, clear misclassification between high reflective water surface and developed land-cover types was observable from the final classified images, especially in summer. The total areas based on the classified images for the five identified land-cover types from 2011 – 2018 are summarized in Tables 17 (summertime) and 18 (wintertime).

Table 16. Random Forest supervised classification accuracy report for the summer and winter evaluations for 2011 – 2018.

Year and Season	Average Accuracy	Kappa
2011 summer	0.954	0.943
2011 winter	0.933	0.916
2012 summer	NA	NA
2012 winter	0.94	0.925
2013 summer	0.955	0.944
2013 winter	0.838	0.798
2014 summer	0.948	0.935
2014 winter	0.915	0.894
2015 summer	0.95	0.937
2015 winter	0.941	0.926
2016 summer	0.929	0.911
2016 winter	0.946	0.932
2017 summer	0.943	0.929
2017 winter	0.967	0.959
2018 summer	0.946	0.932
2018 winter	0.901	0.876

Table 17. Summertime areas of land-cover types (square kilometers) based on Random Forest supervised classifications.

Land cover	2011	2013	2014	2015	2016	2017	2018
Water	5.10	5.50	5.13	5.14	5.34	5.56	5.86
Developed	1.28	1.07	1.88	2.54	1.80	1.99	2.27
Forest (reclassified to *Phragmites*)	1.47	1.63	0.80	1.53	1.02	0.86	0.80
Woody Wetlands	2.97	5.28	3.94	4.00	4.43	3.68	4.37
Herbaceous Wetlands	9.90	7.24	8.97	7.52	8.12	8.64	7.42

Table 18. Wintertime areas of land-cover types (square kilometers) based on Random Forest supervised classifications.

land use	2011	2012	2013	2014	2015	2016	2017	2018
Water	5.25	4.95	4.11	5.23	4.57	5.26	5.30	6.32
Developed	1.66	1.20	3.63	1.48	2.01	0.94	0.82	4.09
Forest (reclassified to *Phragmites*)	2.01	1.48	1.62	1.37	1.54	2.04	1.31	1.31
Woody Wetlands	3.82	5.67	4.64	3.92	4.29	3.56	5.18	2.10
Herbaceous Wetlands	7.98	7.42	6.71	8.66	8.32	8.92	8.11	6.91

From analyzing the remote sensing images, it can be seen that the Wading River Complex is a relatively pristine wetland area, dominated by woody and herbaceous wetlands and water. The developed areas where human activities are present is largely restricted in the north and southeast corner of the evaluation area. Visual inspection of the images suggests that the presence of *Phragmites australis* and human activities are closely related.

From Tables 17 and 18, we can see rather inconsistent classification of the developed land-cover types, especially in the wintertime since developed land (road surface, rooftops, and other impervious surfaces) and frozen wetland have rather similar surface reflectance signatures. As a matter of fact, comparing Tables 17 and 18, we see that the summertime provides more consistent classification results than during the wintertime for different land-use types. Our ensuing analysis, regarding land-cover changes, will hence use the classified images for the summertime only for better modeling effects.

Building a System Dynamic Model to Evaluate the Potential for Secondary Nekton Productivity in the Wading River Complex:

By observing the Tables 9 – 14 from the first half of the study regarding the plant cover across the three transects in the Wading River Complex (where A and B are freshwater, C is saline water), we can see that the dominant species in the freshwater portion is *Phragmites australis*, which is believed to negatively influence the habitat value of the marsh by elevating the marsh planform and filling in the microtopography of the marsh surface (Catling et al., 2011). In the more saline region of the Wading River Complex, however, *Phragmites australis* appears in much lower coverage.

It was determined that the image classification process incorrectly interpreted the "forest" cover type. What was classified as forest was in fact most often *Phragmites australis,* located mainly in the northern part and southeastern corner of the evaluation area where relatively intense human activities (i.e. road and buildings) are present. After relabeling the land-cover type, the changing trends of all five different land-cover types from 2011 – 2018 in both summer and winter times were further examined. In addition, pair-wise scatterplots using the summertime data were produced to identify the potential co-variation among different land-cover types (Figure 13).

Examination of these outputs is the foundation for building a strong system dynamic simulative model. The pairwise scatterplots are simple tools to identify potential co-variations among different variables so that dynamic relationships among different variables can be explored. The lack of longer time-series data, however, suggests that any such exploration resulted co-variation must be treated with caution and should only be used as a guidance to explore actual co-variation among variables that will be used to construct the system dynamic model.

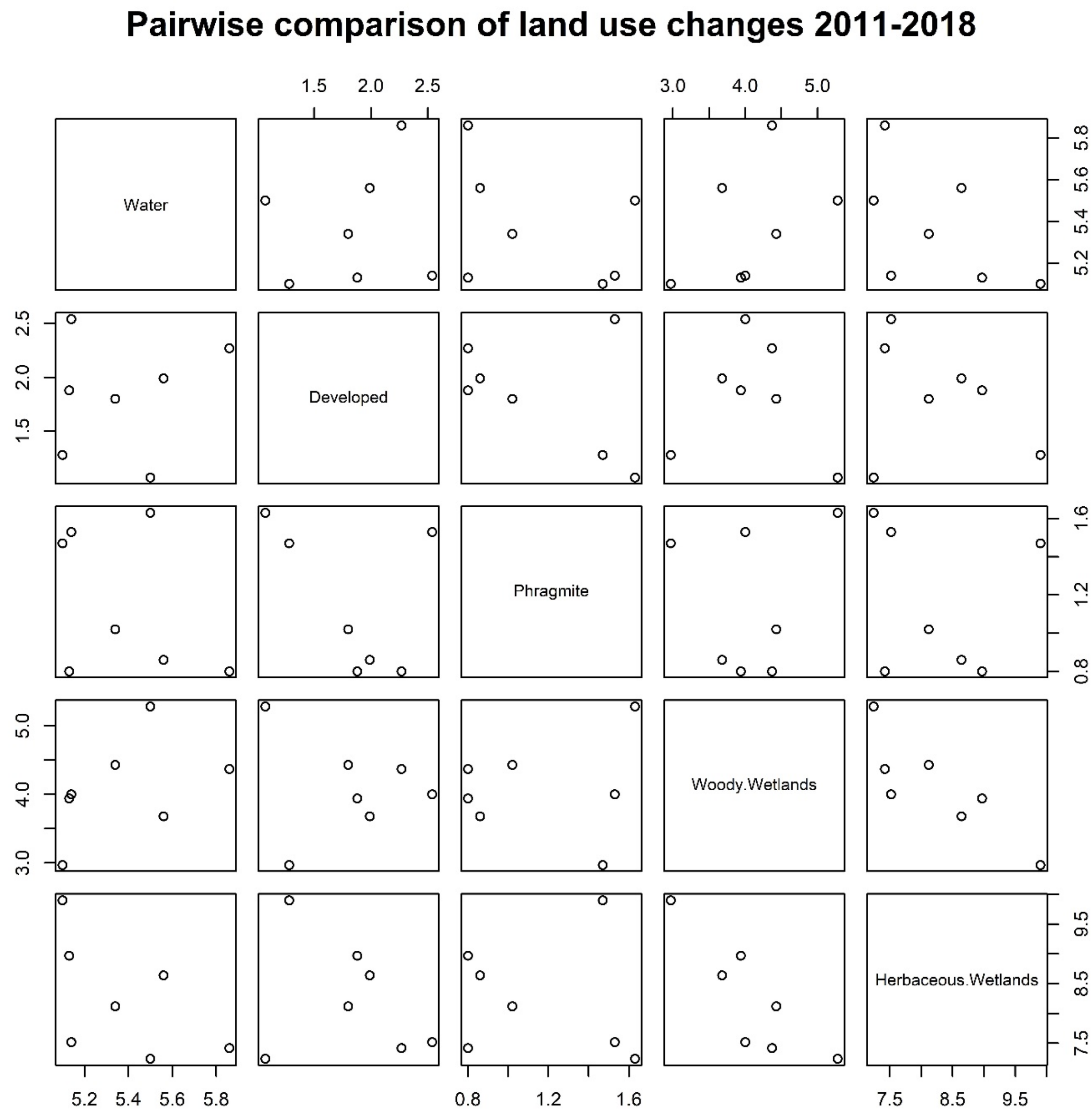

Figure 13. Pair-wise scatterplots among the five different land cover types from 2011 – 2018 summer seasons, excluding 2012. Units are km^2.

Co-variation Relationships Among the Different Land-use Land-cover Types:

By examining the *Phragmites* scatterplots (rows and columns), we can see that the changes over the years between *Phragmites* and developed and herbaceous wetland land-cover types show clear co-variation. Although the short time series might prevent fully disclosing co-variation from being established, the scatterplots provide a start to construct potentially meaningful co-variation relationships. Indeed, further exploration among the three land-cover types suggests that the relationship between the rate of change of *Phragmites* land cover and the herbaceous wetlands follows a quadratic relationship (after removing the outliers in 2015):

Annual change rate of *Phragmites* = 6.0624 – 0.2049 * Herbaceous Wetlands + 0.0016 * Herbaceous Wetlands^2 (R^2 = 0.7598) [1]

After removing one outlier between the *Phragmites* and developed pairs in 2015 due to relatively low classification accuracy of developed land, the relationship shows as:

Developed = 19.023 – 1.2159 * *Phragmites* (R^2 = 0.934) [2]

In addition, there is a relatively obvious negative relationship between the woody wetlands and herbaceous wetlands. The exploratory analysis suggests that a negative exponential relationship describes the changing trends well between the two land-cover types:

Woody wetlands = 117.4 * e – 0.029 * Herbaceous wetlands (R^2 = 0.7248) [3]

After removing the relatively irregular water data for 2013 and 2014, a negative co-variation between *Phragmites* and water is also identified:

Water = 38.263 - 0.9254 * Phragmites (R^2 = 0.854) [4]

After removing an outlier in 2011, a quadratic relationship between herbaceous wetlands and developed land cover is identified:

Herbaceous wetlands = 5.0001 + 8.4013 * Developed – 0.3684 * Developed^2 (R^2 = 0.693) [5]

Further Refinement of the Co-variation Relationships: Grey System Series Simulation:

The major task of this phase of the study was to establish a system dynamic model that is able to simulate the dynamic changes of the land cover in the Wading River Complex. To build an effective system dynamic model, it is necessary to identify the different components in the system. In this relatively pristine wetland complex, the land cover system components are the five different types of land-cover previously evaluated. Because of the limited human activities in the complex, and relatively undisturbed biogeographical conditions, the system is relatively simple. The interactive co-variation among the five different types of land cover are explored starting with Figure 13 and fitted through exponential, linear, logarithmic, quadratic (polynomial), and power relationships. We were able to identify the co-variations among all five land-use types after a series of exploratory analyses as explained in equations [1-5].

Although we explore the co-variations among the components based on a balance between maximum fit and least complexity, all the above identified relationships have rather rigid linear or non-linear relationships. To ensure the simulations remain meaningful, we also introduced a Grey System Series Simulation model (Deng, 1982; Deng, 2002; Liu and Forrest, 2010; Yu and Fang, 2017) to control for unrealistic data generation. A brief description of the Grey System Series Simulation model follows (Yu and Fang, 2017).

Suppose we have a data series $X^{(0)} = (x^{(0)}(1), x^{(0)}(2), \ldots, x^{(0)}(n))$, a first order accumulative addition operation on this series will produce the first order accumulatively added (1-AA) series: $X^{(1)} = (x^{(1)}(1), x^{(1)}(2), \ldots, x^{(1)}(n))$, where $x^{(1)}(k) = \sum_{i=1}^{k} x^{(0)}(i)$. On the other hand, a first order accumulative deduction operation will produce the first order accumulatively deducted (1-AD) series: $X^{(1)d} = (x^{(1)d}(1), x^{(1)d}(2), \ldots, x^{(1)d}(n))$, where $x^{(1)d}(k) = x^{(0)}(k) - x^{(0)}(k\text{-}1)$. Often the data series produced via cumulative addition or deduction will exhibit stronger regularity than the original one. Except for cumulative addition and deduction, another important operation, the mean operation with consecutive neighbors, is often employed and critical in grey model simulation and prediction. This operation will generate a new data series: $Z = (z(2), z(3), \ldots, z(n))$, where $z(k) = 0.5\, x(k) - 0.5\, x(k\text{-}1)$, $z(1)$ is often omitted since there is no left neighbor for $x(1)$.

The primary model that is developed for data series simulation and prediction is called the GM(r, h) model, with r equal to the order of the cumulative operation (often cumulative addition), and h equal to the number of co-varying data series plus the target series. As pointed out by Deng (2002), GM (r, h) models are often well suited for data simulation and data series prediction. Here, a GM (1, 1) model is determined to be the most appropriate.

In this study, the primary purpose for introducing GM models into the system dynamic modeling scheme is to mitigate the potential rigidity and uncertainty that were generated via statistical analysis with relatively limited data points (ten years in the current study). For this purpose, and the argument by (Deng, 2002), we focus on the GM (1, 1) model.

The GM (1, 1) model is based on the target series, its 1-AA series, and the mean operation with consecutive neighbors generated from the 1-AA series. The basic form of the GM (1, 1) model is expressed as (Deng, 2002; Liu et al., 2010):

$$x^{(0)}(k) + az^{(1)}(k) = b \quad (k = 2, \ldots, n) \qquad [6]$$

where n is the number of data points, a is regarded as the development coefficient, and b the grey action measure. The development coefficient is a measure of the target series' changing trend and the inherent uncertainty, while the grey action measure reflects the internal relationships among data points (Liu et al. 2010). From equation (2) [(Liu et al., 2010), p 147] a and b can be derived via least squares operation:

$$\begin{cases} a = \dfrac{\dfrac{1}{n-1}\sum_{k=2}^{n} x^{(0)}(k)\sum_{k=2}^{n} z^{(1)}(k) - \sum_{k=2}^{n} x^{(0)}(k) z^{(1)}(k)}{\sum_{k=2}^{n}(z^{(1)}(k))^2 - \dfrac{1}{n-1}(\sum_{k=2}^{n} z^{(1)})^2} \\ b = \dfrac{1}{n-1}[\sum_{k=2}^{n} x^{(0)}(k) + a\sum_{k=2}^{n} z^{(1)}(k)] \end{cases} \qquad [7]$$

With a and b obtained, the series can be simulated and forecasted as (Liu et al., 2010):

$$x^{(0)}(k) = \left(\frac{b-0.5a}{b+0.5a}\right)^{k-2}\left(\frac{b-ax^{(0)}(1)}{b+0.5a}\right)$$

The model has been implemented in the Grey System Prediction software package developed by Liu et al. (2010). Since a measures the inherent uncertainty of the target data series, from experimental analyses, Liu et al. 2010 concluded that a GM (1, 1) model with an *a* value that is within the range of [-0.3, 0.3] can be used for relatively longer term (up to 10 – 15 temporal periods, years in our study) forecast.

By combining the conventionally obtained co-variation relationship with the Grey System Series Simulation approach, we built the relatively simple system dynamic model that functions to simulate the co-variation among the five different land use type. The general system dynamic model structure is presented in Figure 14. The model structure is relatively simple since the dynamic is built entirely on the observed co-variation among the five different land-cover types. As aforementioned, all the explorations among the variables pertain to the "co-variation" among them. It is recognized that there are many other agents that are at work to drive the dynamics of the system. By observing the co-variation, however, the system dynamic simulation might be able to capture intuitively the latent relationships that drive the dynamics of the system. Such system dynamic simulations do not assume any causality nor correlation, but co-variation only. The in-depth mechanism among different system components are assumed to be highly complex with high dimension eliminating any assumptions of causality. Figure 14 is a simple manifestation of how such dynamics might be presented in a feedback loop structure. Simply put, one land-cover type is regarded as co-varying with another land-cover type. While the land-cover type of *Phragmites australis* is our focus type, we use that land-cover type as the hub to connect all other land-cover types in the simultaneous dynamics.

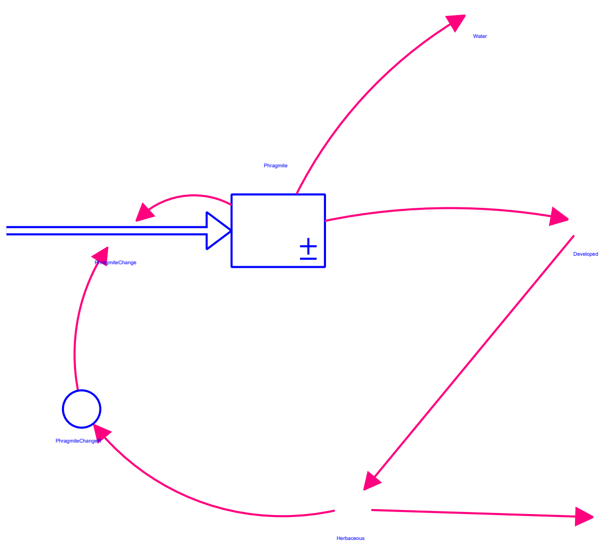

Figure 14. The conceptual system dynamic model structure of the five land use types in the Wading River Complex.

From Figure 14, we can see that the annual change rate of the *Phragmites* is at the center of the entire system since the amount of *Phragmites* distribution relates directly or indirectly to all other four types of land covers. Equation 1 suggests that the annual change of *Phragmites* cover can be best correlated with the change of herbaceous wetland in a quadratic form. *Phragmites australis* cover decreases as the herbaceous wetland cover increases to a point, then it starts to increase again.

It must be emphasized here that there is no causality assumed. The observation is simply the covariation between the two land-cover types. However, the quadratic form has the tendency to produce outputs that first become very small then become large. To prevent the quadratic equations to go beyond bounds, the grey system series simulative value for the annual change rate is added to ensure the change rate remains a reasonable amount. Equation 1 is then changed to:

MIN [(6.0624 - 0.2049 * Herbaceous +0.0016 * Herbaceous^2) , (-0.05638 * (-1.08556)^(TIME - 2))] [8]

Here, a logical selection is made from the minimum output of either the first or second portion of the determinant. The first part is the quadratic form from equation 1, the second part, namely, -0.05638 * (-1.08556)^(TIME - 2), where TIME is in years, is the grey system simulative equation with $a = 0.004$, and $b = -0.195$. Equations (2) – (5) remain unchanged. The model is then simulated in ISEE®'s Stella Architect software to produce the most reliable simulation of how different types of land-cover will evolve and change in the next decade to 15 years. We do not simulate beyond 15 years because the simulation will become increasingly unreliable due to increasing possibility of unforeseeable factors in the long run (Yu and Fang, 2017). The results produced from the simulation are reported in Table 19. The area of the simulated results is slightly less than the actual area because of loss of pixels during the simulation process.

Table 19. Results of the Grey system coupled system dynamic simulation of land-cover changes from 2019 – 2033. Units are in square kilometers.

Year	Water	*Phragmites*	Developed	Herbaceous Wetland	Woody Wetland
2019	4.76	1.47	1.25	7.71	4.64
2020	4.69	1.55	1.16	7.45	4.87
2021	4.68	1.56	1.14	7.40	4.92
2022	4.66	1.58	1.12	7.34	4.97
2023	4.62	1.63	1.06	7.13	5.16
2024	4.51	1.75	0.92	6.59	5.69
2025	4.38	1.88	0.76	5.83	6.53
2026	4.23	2.04	0.56	4.79	7.89
2027	4.22	2.06	0.54	4.69	8.03
2028	4.21	2.07	0.53	4.59	8.17
2029	4.20	2.08	0.51	4.50	8.32
2030	4.18	2.10	0.50	4.40	8.47
2031	4.17	2.11	0.48	4.29	8.63
2032	4.16	2.12	0.46	4.19	8.79
2033	4.14	2.14	0.44	4.08	8.96

One immediate impression from Table 19 is that developed land cover will continue to decrease, along with herbaceous wetland and water. Although the area after around five years (2023) will become less of a prediction but more of an indication of future trend, it is still alarming to see expected increases in *Phragmites australis* in the near future. More valuable than the numeric output is the inferred trend produced by this 25-year assessment. We see a predicted increase in *phragmites* and woody wetlands. It is important to note that this analysis is not be able to directly answer which land-cover type is replacing which.

As aforementioned, the existence and rapid spreading of *Phragmites australis* could pose a severe threat to local fauna and flora. This simulation model suggests that based on recent conditions, that it is likely that *Phragmites* cover can and will likely increase in the near future. The magnitude of such an increase is speculative though. While simulative model results should not be treated as actual prediction, the trends revealed by the simulative model, however, shall provide guidance for understanding the baseline conditions of the relatively pristine wetland complex in the WRC and similar areas with a well rooted P*hragmites* population.

Discussion and Conclusion for Establishment of Quantitative Metrics that Relate Geomorphological and Ecological Characteristics:

By classifying ten years of high-resolution remote sensing images of the WRC into five dominant land cover types, this research explores the covariation among five land cover types to predict how these land covers will likely change in the near future. A Grey System Series coupled system dynamic simulative model was employed to undertake the task. While the available information is rather limited for the simulation, the simulative results are still indicative that the invasive species *Phragmites australis* is likely to increase its dominance in the WRC in the near future, and could pose a severe threat to local fauna and flora.

Specifically, because of the negative impact of *Phragmites australis* on the secondary nekton species survival and food foraging capability, evaluating the changing trend of *Phragmites australis* in the salt marsh of Wading River Complex might serve as an indirect indicator for the potential productivity of the secondary nekton species. Although evidence suggests that a subspecies of *Phragmites* existed in North America prior to the European colonization, it is commonly believed that current populations represent an exotic/invasive subspecies of *Phragmites australis* (Saltonstall, 2002; Catling et al., 2011). Regardless of the origination of the population, the existence and rapid spreading of *Phragmites australis* could pose a severe threat to local fauna and flora.

Lacking traces of the invasive pathways of *Phragmites australis* in the Wading River Complex prevents any meaningful causality relationship between the coverage of *Phragmites australis* and other factors from being established. Direct secondary nekton surveys were not conducted as part of this study. Therefore, a direct prediction of how secondary production in the WRC using land-cover types and river network changes over the past decade is not possible. As such, potential impacts to nekton species can only be evaluated through a theoretical approach based of habitat preferences and potential changes to that habitat.

Project-wide Summary and Conclusions for Developing a Watershed-scale Baseline for Tidal Wetlands:

The outcomes from this work provide important perspectives in the context of wetland restoration, monitoring, and management. Components of this project directly complement the goals and vision outlined in the EPA-approved Wetland Program Plan in New Jersey (NJWPP) and provides valuable information in a minimally disturbed wetland complex for a wide range of stakeholders involved in ecological restoration and natural resource conservation.

In Section 1 we summarized results of a meta-analysis of restoration success primarily in the

Delaware Bay. Catch per unit effort (CPUE) and seasonal growth of dominant species were examined over a 17-year period at two reference and five restoration sites. Overall, the restoration process at the Estuary Enhancement Program (EEP) sites proceeded fairly rapidly with nekton abundances at restored marshes converging with, or exceeding those of the reference sites, especially in the lower part of the bay. Nekton were equally abundant across restored sites shortly after treatment for *Phragmites* in the upper Delaware Bay and soon after the dikes were opened in the lower Bay but the average size of nekton captured at *Phragmites*-dominated sites generally lagged those of comparable species at the reference site, supporting earlier findings that Phragmites-dominated marshes limit the production of nekton that occupy them. This work reinforces the need for monitoring to include structural criteria and continue for a sufficient period of time, perhaps a decadal time frame, that allows for the assessment of secondary production and other functional criteria.

Plant community and water level data were collected along five transects as part of Section 2. Transects in the upper part of the estuary were found to be very similar, as they both reflect lower saline conditions and were dominated by *Phragmites australis, Spartina cynosuroides*, and *Typha augustifolia*. The transect situated in the lower part of the estuary exhibited more saline conditions and was dominated by *Spartina alterniflora* and *Spartina patens*. Water level readings from two transects were used to determine that the marsh may flood up to 4.75 inches and flood stage over the marsh can last upwards of 7 hours.

Section 3 describes efforts to classify ten years of high-resolution remote sensing images of the WRC into five dominant land cover types. Through this modelling approach we were able to predict how land cover will likely change in the near future. While the available information is limited for this simulation in this estuary, the simulative results are indicative that the invasive species *Phragmites australis* is likely to increase its dominance in the estuary and could pose a severe threat to nekton productivity.

Future Work:

A potential future extension of the current study, which could be extremely beneficial for local wetland management and nekton species inventory, would be to connect surveys of secondary production with remotely-sensed landscape variables to establish an index that can accurately reflect potential changes in the relative abundance of local secondary nekton species. Recent studies that include monitoring water quality, including nutrients, and soil salinity through remote sensing techniques (Wang et al., 2019a; Wang et al., 2019b; Wang et al., 2019c) could be a potential direction to be pursued in future studies.

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