

Runnels Reverse Mega-pool Expansion and Improve Marsh Resiliency in the Great Marsh, Massachusetts (USA)

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Research Article

Keywords: Runnel, salt marsh, pool, restoration, vegetation classification

Posted Date: July 18th, 2022

DOI: <https://doi.org/10.21203/rs.3.rs-1610002/v1>

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Version of Record: A version of this preprint was published at Wetlands on April 6th, 2023. See the published version at <https://doi.org/10.1007/s13157-023-01683-6>.

Abstract

One of the main mechanisms for salt marsh decline across the United States is the inability of the marsh surface to keep pace with sea level rise. The interior platform is especially vulnerable, leading to the encroachment of short form *Spartina alterniflora* pannes, pool formation, and ultimately runaway pool expansion if recovery is not possible. Coastal ecologists in New England have been implementing a restoration strategy of runnels, or shallow ditches, to enhance drainage of oversaturated and ponding interior marshes. In 2015, runnels were constructed to drain two large and expanding pools in the Great Marsh System of Massachusetts, USA. Vegetation, elevation, and hydrology were monitored using field sampling and remote sensing analysis pre- and post-restoration over seven growing seasons to document the restoration trajectory of the pools and adjacent salt marsh platforms. Pool drainage improved reflecting tidal cycles after three growing seasons. Substantial colonization of *S. alterniflora* and *S. patens* into the previously unvegetated pools required three growing seasons. In the adjacent marsh platform, *S. patens* and *Distichlis spicata* increased in abundance with substantial declines in *S. alterniflora*. The runnel for one pool became partially blocked by vegetation after 2018 and inhibited drainage and recovery of the vegetation community in the pool yet not the platform. Runnels may be a viable solution for restoring interior marshes following vegetation loss yet substantial improvements in vegetation and hydrology may require 3–5 years and complete recovery of the vegetation community in the pool system at least a decade or more.

Introduction

Historically, salt marshes experienced a host of direct anthropogenic stressors ranging from modification of hydrology through mosquito ditching and ditch-plugging practices (Meredith et al. 1985, Wolfe 1996, Vincent et al. 2014, Burdick et al. 2020), fill-in for urbanization and infrastructure (Broomberg and Bertness 2005, Gedan et al. 2009), and development at the immediate upland edge (Bozek and Burdick 2005, Pontee 2013). Over the past thirty years, sea level rise (SLR) has been recognized to pose the largest threat to salt marsh systems on the East Coast in the United States. Recent estimates of rapid SLR rates, however, may outpace salt marsh elevation gain (Crosby et al. 2016), which can further disrupt marsh building by increasing plant mortality and decreasing belowground biomass production (Payne et al. 2019). Increased flooding can also be exacerbated by limited sediment supplies (Ganju et al. 2017) and legacy effects of prior agricultural alterations (Mora and Burdick 2013, Adamowicz et al. 2020, Smith et al. 2021). Large loss of coastal wetlands from SLR has been documented in Maryland (Schepers et al. 2020), Mississippi Delta (Day et al. 2011), New England (Watson et al. 2017), New York (Smith et al. 2021), and Virginia (Burns et al. 2021).

One of the main pathways for marsh loss to SLR is oversaturation, short-form *Spartina alterniflora* panne formation, and subsequent creation and expansion of pools on the interior marsh platform (Redfield 1972, Vincent et al. 2013, and Raposa et al. 2017). Compared to low marsh areas immediately along tidal creeks and shorelines, the higher elevation marsh platform is especially vulnerable to deterioration associated with reduced inorganic sediment inputs away from creeks (Temmerman et al. 2003, 2005, and Fagherazzi et al. 2013) and declines of belowground biomass inputs from dominant graminoids caused by increased flooding (Watson et al. 2016, Payne et al. 2019). Oversaturation of the interior marsh platform increases stressors on the vegetation community including anoxia, increased salinity, and build-up of hydrogen sulfide concentrations in the root zone (Van Huissteden and Van De Plassche 1998, Mendelssohn and Morris 2000, Berkowitz et al. 2018, Himmelstein

et al. 2021). Over time high marsh graminoids (*S. patens*, *Distichlis spicata*, *Juncus gerardii*) die off and are replaced by more stress-tolerant short-form *S. alterniflora* and *Salicornia* spp. or open water (Warren and Niering 1993, Raposa et al. 2017) with less consolidated soils and shallow standing water (Berkowitz et al. 2018, Burdick et al. 2020). Conversion of high marsh meadows to short form *S. alterniflora* pannes and pools has been attributed to population declines of endemic species such as *Ammospiza caudacauta* (Saltmarsh Sparrow) as well as *A. nelsoni* (Nelson's Sparrow) (Gjerdum et al. 2005, Shriver et al. 2016).

Small pools with stable, vertical banks are natural features on the marsh interior (Redfield 1972, Adamowicz and Roman 2005), however, coastal ecologists have recently been documenting large swaths of interior platform being lost to pool expansion, especially at lower elevations and flat slopes (Wilson et al. 2014, Ganju et al. 2020). Continued vegetation die-off and decrease in elevation (e.g., surface erosion, subsidence, and organic matter decomposition) may “push” the platform over a tipping point to vegetation collapse and pool formation (DeLaune et al. 1994, Day et al. 2011, Chambers et al. 2019). Runaway pool collapse occurs when the inorganic sediment supply is insufficient for the marsh to keep pace with SLR and stressful biogeochemical conditions at pool edges causes further vegetation die-off and bank collapse (Mariotti 2016, Mariotti et al. 2020, Himmelstein et al. 2021). The interior marsh can convert to large areas of standing water as smaller pools expand and merge over time (Himmelstein et al. 2021). Although pool recovery has been well-documented after reconnection with a tidal creek (Wilson et al. 2014, Smith and Pellew 2021), coastal ecologists have identified insufficient sediment inputs, microtidal ranges (< 1 m), and high tidal prisms as factors that further exacerbate pool expansion after tidal reconnection (Ganju et al. 2017, Schepers et al. 2017, 2020, Vinent et al. 2021).

Coastal ecologists have implemented a restoration strategy, termed runnels, to improve drainage of oversaturated soils without the impacts of over-aeration of the peat soil column (Wigand et al. 2017). Runnels are shallow vegetated swales (15–30 cm wide, 20–30 cm deep) constructed through areas of standing water or at the edge of pools which connect to the nearest hydrologic channel (ditch or creek). Improvements in drainage across the salt marsh platform is expected to decrease biogeochemical stressors and allow for eventual recolonization of high marsh graminoids. Besterman et al. (2022) documented a pattern of vegetation recovery over seven years after runnel construction in Rhode Island: (1) initial drainage of standing water and exposing bare ground, (2) colonization of *S. alterniflora* and *Salicornia* spp., and (3) replacement of low marsh vegetation with high marsh graminoids of *S. patens*, *Distichlis spicata*, and *Juncus gerardii*. Additionally, Perry et al. (2021) observed revegetation across salt marsh platforms as well as no net increase in peat oxidation after runnel construction. On longer time scales, revegetation and drainage enhancement of the salt marsh surface may lead to gains in elevation.

Runnelling is an additional tool for coastal ecologists and land managers to conserve and improve the resiliency of salt marsh systems. Adapted from Open Marsh Water Management methods to enhance tidal flow of mosquito breeding depressions (Wolfe et al. 2021), runnels have only been recently implemented and monitored for the purpose of rebuilding and improving high marsh vegetation communities. Questions remain about the efficacy across different tidal ranges, sediment inputs, and tidal prisms as well as maintenance requirements over time (Besterman et al. 2022). Recent applications of runnels have focused on recently converted low marsh pannes, where elevation and vegetation losses are less severe than marshes converting to pools (see Perry et al. 2021, Besterman et al. 2022). We constructed runnels in Spring 2015 at two large pools in the Parker River National Wildlife Refuge (Newbury, MA) to reconnect tidal exchange and in doing so, arrest pool

expansion and enhance revegetation of the salt marsh surface. Both pools have been present on the landscape as early as 1965 based on historic aerial imagery (United States Geologic Survey - EarthExplorer), however evidence of recent expansion spurred restoration efforts. By 2014, one year before runnel construction, the north and south pools had expanded to sizes of 0.60 ha and 3.52 ha and perimeters of 327 m and 1536 m, respectively. We monitored hydrology, vegetation, and elevation over seven growing seasons to document the efficacy of the runnels and the trajectory of the pool and adjacent marsh platform recovery. Additionally, we used remote sensing analysis of publicly available aerial imagery to document system-wide vegetation recovery over time.

Methods

Study Site & Study Design

The study site, hereafter referred to as Pine Island Marsh, is located within the Great Marsh system and Parker River National Wildlife Refuge (Newbury, MA) between the Parker River and Pine Island Creek (Fig. 1 - Left). Pine Island Marsh is a 191-ha salt marsh platform dissected by three major tidal creeks, numerous smaller order creeks, and artificial drainage ditches (ditch density = 149 m ha⁻¹). The system is macrotidal with a mean tidal range of 2.64 m (NOAA Gauge 8441241). The three major rivers of the system (Rowley, Ipswich, and Parker Rivers) are dammed upstream, limiting the sediment supply. The system is considered sediment deficient for long-term salt marsh stability (Cavatorta et al. 2003, Langston et al. 2020) with median suspended sediment concentrations of 15 mg L⁻¹ and approaching 40 mg L⁻¹ near the mouth of the estuary (Hopkinson et al. 2018). Average salt marsh elevation gain rates ranged from 3.5 mm yr⁻¹ in the high marsh platform to 9.5 mm yr⁻¹ in the low marsh compared to recorded SLR rate of 4.8 mm yr⁻¹ (Langston et al. 2020). The vegetation community is comprised of low marsh on tidal creek and ditch banks of tall form *S. alterniflora* (> 35 cm height), high marsh platform dominated by *S. patens*, *D. spicata*, and *J. gerardii* mixed with pannes of low form *S. alterniflora* (< 35 cm) and *Salicornia depressa* (Anderson and Treshow 1980, Millette et al. 2010).

Two pools (north and south pool) were selected for restoration using runnels based on vestiges of agricultural features, recent dieback of halophyte graminoids, presence of drifted algal mat, and proximity to hydrologic pathways for runnel construction (Fig. 1 – Right). Runnels approximately 30 cm width and 15 cm depth were constructed in the Spring of 2015 with a post driver and angle iron (30 cm wide and 15 cm depth) from the edge of the pool to the nearest hydrologic pathway with lengths ranging 60–200 cm (Fig. 2). One runnel was constructed at the north pool and two at the south pool to compensate for its size and tidal prism (Fig. 3). We cut the runnels to set the pools on a drainage trajectory whereby the size of the tidal prism would expand to eventually become balanced with the dimensions of the runnel.

Field Sampling

Restoration progress was monitored based on a before-after study design for vegetation and salt marsh surface elevation (Neckles et al. 2002). Pre-restoration data were collected for vegetation in Spring 2015 and follow-up monitoring was completed in Summer 2015, Summer 2016, Fall 2018, and Fall 2021. Elevation was monitored in Spring 2015, Summer 2016, and Fall 2021. Transects parallel to the pool edge were established in Spring of 2015 on four sides of each pool to monitor the recovery of the vegetation and elevation following runnel construction (Fig. 3). Five pairs of permanent vegetation plots were randomly located along each 30 m transect

with at least 3 m between plots. Each location consisted of a pair of 0.5 m² plots with one on the salt marsh platform edge and the second immediately adjacent in the pool (n = 40 per pool site).

All vascular and non-vascular species as well as abiotic categories (e.g., bare mud, dead vegetation, desiccated algal mat, etc.) were identified and estimated for visual cover to the nearest 1% by ocular method within two hours of low tide (Neckles et al. 2002). Species were classified as halophytes or submerged aquatic vegetation (i.e., *Ruppia maritima*) according to Tiner (2009). To better understand the encroachment of *S. alterniflora* in the high marsh zone, the High Marsh Health Ratio (HMH ratio) was calculated as:

$$\frac{\text{HighMarshGraminoids}}{\text{HighMarshGraminoids} + \textit{Spartinaalterniflora}}$$

where dominant high marsh graminoids is the aggregate cover of *S. patens*, *D. spicata*, and *J. gerardii*. The ratio ranges from zero to one, where values closer to one indicate a healthy high marsh dominated by high marsh graminoids. Values closer to zero indicate salt marsh platforms converting to low marsh or *S. alterniflora* pannes.

Surface elevations of the salt marsh platform and pool plots were monitored to estimate how runnel drainage would affect accretion rates. In 2015, the relative elevation of the plots was measured with a laser level (± 0.5 cm accuracy; Johnson Rotary Laser Level, Burnsville, MN). In 2016 and 2021, true plot elevations (NAVD88 m) were measured with an RTK GPS (± 3.0 cm accuracy; Leica GSSN Rover model GS14, St. Gallen, Switzerland). Elevations throughout the study were measured within the center of the vegetation plot. The difference in elevation between the salt marsh platform and pool plot was calculated for each pair for all three years.

Hydrology was monitored post-restoration (July & September 2015, July 2016, October 2018, and October 2021) in the runnelled pools and adjacent natural hydrologic channels, which served to indicate the effectiveness of the hydrologic path. Odyssey capacitance water level loggers (2015–2018; Christchurch, New Zealand) and Hobo pressure transducers (2021; Onset, Inc., Bourne, MA) were deployed in each pool and one nearby channel, located downstream of the runnel, to record every 10 minutes for one lunar tidal cycle (~ 30 days). Due to the clogged runnel, the channel water level recorder for the north pool was moved to Jericho Creek in 2021 (110 m from the pool). Water level recorders were stored in PVC casings with 6.4 mm holes and attached to stakes with clamps. Stakes were installed into the sediment until the PVC casing touched the sediment surface. In 2021, water level elevations for pools were measured by placing water level recorders in perforated PVC tubes (6.4 mm holes every 10 cm for 0.6 m) set 60 cm below ground in the pools to monitor groundwater levels (Roman et al. 2001). Elevations of water level recorders were measured with an RTK-GPS (NAVD88 m). An additional pressure transducer was attached to a nearby tree to compensate for atmospheric pressure.

Statistical Analysis

Water level recorder and atmospheric pressure data were processed in Odyssey or Hoboware software. The average high tide, low tide, higher high tide, maximum water elevation, flooding duration, and flood frequency for the salt marsh platform and pool elevations was calculated for each water level recorder with the *VulnToolKit* package in R (R Core Team 2013, Hill and Anisfield 2021). The average elevation of all platform and pool plots for each pool and season were used for the flooding duration and flooding frequency analyses. The

2016 elevations were used for the 2015 and 2018 water level analyses since the true elevation was not recorded.

To assess broader community shifts in the vegetation community, non-metric dimensional scaling (NMDS) ordination was conducted for each pool separately (Bray-Curtis dissimilarity, square-root transformation). Rare species (< 5% occurrence) were removed prior to multivariate analysis. Species that were highly and significantly correlated ($r^2 > 20\%$, $p < 0.05$) were overlain on the ordination. Pairwise analysis of similarity (ANOSIM; Bray-Curtis dissimilarity) and Similarity Percentages (SIMPER) was conducted on the vegetation community to observe the trajectory of the platform and pool habitats (McCune and Grace 2002). Cover types of similar functional groups were aggregated: (1) Bare, dead, desiccated algal mat, wrack, and *Vaucheria* algae as Non-live Cover, (2) Floating green algae and *Fucus* sp. as Algae. Multivariate analyses were conducted with the *vegan* package (Oksanen et al. 2020) and data visualization with *ggplot2* package (Wickham 2016).

Remote Sensing of Vegetation Recovery

The proportion of vegetated marsh surface before and after runnel construction was measured using publicly available aerial imagery of the National Agriculture Imagery Program (NAIP) through the United States Department of Agriculture of 2012–2018. Aerial imagery (RGB – Infrared) was acquired in the summer (July – August) with resolutions of 1 m (2012, 2014) improving to 0.6 m (2016, 2018). The normalized difference vegetation index (NDVI) was calculated to provide a high-resolution measurement of plant health across the salt marsh surface. Vegetated and non-vegetated areas (*i.e.*, water, wrack, bare mud, floating algae, and roads) were classified through pixel classification with the Classification Wizard Tool in ArcGIS Pro (Esri, Redlands, CA). The outline of each pool was manually drawn based on 2012 imagery, and a 5 m buffer was created around the pool outline to capture potential pool expansion. The vegetated and non-vegetated surface areas were tabulated from classified pixels within each pool outline.

Results

Hydrology

Runnels improved drainage for both pools between 2015–2016, despite interannual variation in tidal cycle. The impact of the runnels on pool hydrology was more pronounced in the south pool than the north (Table 1). The western runnel in the south pool occupied the location of an original farming ditch and expanded over the course of several years whereas the runnel for the north pool became clogged with vigorous *S. alterniflora* growth and sediment sometime after 2018. Pool hydrology matched natural creek hydrology in both timing and tidal elevations in 2018 when it was draining every tidal cycle (Fig. 4). Improvement in drainage at north pool was observed by 2018, three years after restoration, however conditions declined by 2021, likely due to the runnel filling with sediment and vegetation. Drainage in the north pool did not improve immediately based on > 90% flooding duration and 100% high tide flooding frequency in 2015 and 2016 (Table 1). The lack of rapid drainage throughout the study was apparent based on tapered decreases in water elevation (*i.e.*, lagging and incomplete drainage) after high tides (Fig. 4). The salt marsh platform surrounding the north pool experienced greater drainage of flood waters with flooding duration decreasing from 10% in 2015 to 5% in 2021 with similar high tide flooding frequencies.

Table 1

Hydrology metrics from water level recorders in the pools from Fall 2015–2021 at north and south pool. Water elevations were only recorded after runnel construction. Elevations for 2015–2018 are based on 2016, since true elevation (NAVD88 m) was not recorded in 2015 and 2018. Values are reported as mean +/- standard error.

Site	Season	High Tide (NAVD88 m)	Higher High Tide (NAVD88 m)	Max Tide (NAVD88 m)	Pool High Tide Flooding Frequency (%)	Pool Flooding Duration (%)	Platform High Tide Flooding Frequency (%)	Platform Flooding Duration (%)
North	2015 (Spring)	1.39 +/- 0.02	1.44 +/- 0.02	1.66	100	92.6	31.5	10.3
	2016	1.44 +/- 0.01	1.48 +/- 0.02	1.75	100	97.1	55.2	26.1
	2018	1.41 +/- 0.02	1.47 +/- 0.04	2.12	76.3	74.8	33.9	11.3
	2021	1.40 +/- 0.02	1.45 +/- 0.03	1.78	100	100	34.5	5.1
South	2015 (Spring)	1.32 +/- 0.02	1.35 +/- 0.03	1.89	33.3	26.0	15.8	3.0
	2016	1.45 +/- 0.02	1.50 +/- 0.03	1.80	75.4	63.6	49.1	25.5
	2018	1.44 +/- 0.03	1.51 +/- 0.04	2.06	62.7	23.5	37.3	8.5
	2021	1.37 +/- 0.02	1.42 +/- 0.03	1.77	56.9	13.2	31.0	4.3

Vegetation

The vegetation community required at least two growing seasons to respond to increased drainage and at least four growing seasons for cover to increase significantly with the recovery more pronounced at the south site (Table 2, Fig. 5). In the pool habitat, percent cover of *S. alterniflora* cover increased by 31% north and 37% south and high marsh graminoid cover, the aggregate of *S. patens*, *D. spicata*, and *J. gerardii*, increased 7% north and 27% south while the drained bare areas declined by 21% north and 63% south from pre-restoration conditions to 2021. *S. alterniflora* cover peaked in Fall 2018 for the north pool and subsequently was reduced in 2021. In the south pool habitat, *S. alterniflora* cover declined by half between 2018–2021, however, it was predominantly replaced by high marsh graminoid cover. The non-live cover continued to decline in the same timeframe at the south pool habitat.

Table 2

Vegetation metrics of North and South Pool 2015–2021. Spring 2015 is baseline data before runnel construction. Metrics are percent cover except otherwise stated in the table. Values reported as mean $\pm$ standard error.

Site	Habitat Zone	Year	<i>Spartina alterniflora</i>	High Marsh Graminoid	Non-live	High Marsh Health Ratio	Halophyte Richness (species per plot)
North	Pool	2015 (Spring)	9.9 $\pm$ 2.3	2.4 $\pm$ 1.2	87.6 $\pm$ 2.5	0.22 $\pm$ 0.08	1.5 $\pm$ 0.1
		2015 (Summer)	24.4 $\pm$ 4.4	4.7 $\pm$ 1.7	79.7 $\pm$ 4.4	0.18 $\pm$ 0.07	1.8 $\pm$ 0.2
		2016	16.4 $\pm$ 4.2	2.9 $\pm$ 1.2	70.4 $\pm$ 4.6	0.18 $\pm$ 0.06	1.9 $\pm$ 0.2
		2018	41.1 $\pm$ 5.4	7.2 $\pm$ 1.8	44.4 $\pm$ 5.0	0.16 $\pm$ 0.04	2.2 $\pm$ 0.2
		2021	25.4 $\pm$ 4.8	9.5 $\pm$ 2.6	65.9 $\pm$ 5.7	0.29 $\pm$ 0.06	2.0 $\pm$ 0.1
	Platform	2015 (Spring)	21.4 $\pm$ 4.3	44.3 $\pm$ 5.9	34.3 $\pm$ 2.9	0.64 $\pm$ 0.07	2.2 $\pm$ 0.1
		2015 (Summer)	23.9 $\pm$ 3.5	43.1 $\pm$ 5.7	30.8 $\pm$ 3.1	0.60 $\pm$ 0.06	2.8 $\pm$ 0.2
		2016	23.6 $\pm$ 4.2	50.5 $\pm$ 5.5	24.5 $\pm$ 2.2	0.66 $\pm$ 0.06	2.5 $\pm$ 0.2
		2018	13.6 $\pm$ 2.5	67.3 $\pm$ 4.6	9.1 $\pm$ 1.9	0.82 $\pm$ 0.03	3.3 $\pm$ 0.3
		2021	7.8 $\pm$ 1.6	79.1 $\pm$ 2.7	11.4 $\pm$ 1.4	0.91 $\pm$ 0.02	2.6 $\pm$ 0.3
South	Pool	2015 (Spring)	3.8 $\pm$ 1.5	3.2 $\pm$ 1.9	91.3 $\pm$ 2.7	0.28 $\pm$ 0.11	1.4 $\pm$ 0.2
		2015 (Summer)	9.5 $\pm$ 3.7	2.3 $\pm$ 1.3	86.0 $\pm$ 3.9	0.35 $\pm$ 0.12	1.6 $\pm$ 0.2
		2016	14.9 $\pm$ 3.8	3.3 $\pm$ 1.2	78.8 $\pm$ 3.6	0.32 $\pm$ 0.10	1.8 $\pm$ 0.2

Site	Habitat Zone	Year	<i>Spartina alterniflora</i>	High Marsh Graminoid	Non-live	High Marsh Health Ratio	Halophyte Richness (species per plot)
		2018	47.3 +/- 5.2	18.6 +/- 3.4	30.6 +/- 3.8	0.31 +/- 0.07	2.9 +/- 0.3
		2021	41.3 +/- 4.1	30.3 +/- 3.7	27.6 +/- 1.9	0.43 +/- 0.05	3.6 +/- 0.3
	Platform	2015 (Spring)	18.6 +/- 5.4	27.4 +/- 6.1	53.2 +/- 6.2	0.67 +/- 0.09	2.2 +/- 0.2
		2015 (Summer)	19.3 +/- 5.2	22.8 +/- 5.8	56.7 +/- 6.3	0.59 +/- 0.09	2.6 +/- 0.2
		2016	21.9 +/- 6.3	39.5 +/- 6.0	36.0 +/- 5.0	0.69 +/- 0.08	2.5 +/- 0.2
		2018	11.7 +/- 3.4	73.5 +/- 4.3	9.4 +/- 1.6	0.86 +/- 0.04	3.4 +/- 0.3
		2021	7.3 +/- 1.7	76.4 +/- 3.6	14.2 +/- 3.1	0.91 +/- 0.02	3.8 +/- 0.4

The adjacent platform also trended toward a less saturated environment with the recovery of high marsh graminoids and declines of *S. alterniflora*. The percent cover of low marsh grass initially increased and peaked in 2016 to 2% north and 3% south and then subsequently declined by 13% north, 7% south in 2021. In contrast, high marsh graminoid cover continually increased throughout monitoring (+ 35% north, + 49% south). The HMH ratio reflected the divergent trajectories with large increases at both sites over time (+ 0.27 north, + 0.21 south). At both sites, non-live cover declined through 2018 and increased slightly as *S. alterniflora* declined and high marsh graminoids were unable to fully colonize all of the bare patches. Plot-level richness of the vegetation community increased in both habitats over time with noticeable increases of high marsh forbs on the salt marsh platform including *Agalinis maritima*, *Limonium nashii*, *Plantago maritima*, and *Solidago sempervirens*.

Multivariate analyses were used to better understand the trajectory of the vegetation community recovery and which species drove the shifts over time. Two-dimensional NMDS ordinations found divergent recoveries of the vegetation community between the north (stress = 7.2) and south pool (stress = 12.8). At the north pool site (Fig. 6), the salt marsh platform plots progressed towards a vegetation community with higher *S. patens* cover ($r^2 = 0.95$) whereas the pool plots remained relatively stagnant with high non-live ($r^2 = 0.90$) and *S. alterniflora* cover ($r^2 = 0.92$). At the south pool site (Fig. 6), both the pool and salt marsh platform plots progressed towards a vegetation community with greater *S. patens* cover ($r^2 = 0.71$) and less non-live cover ($r^2 = 0.87$). The revegetation trajectories of the two habitats also diverged as seen on Axis 2 with platform plots progressing towards a vegetation community with greater *D. spicata* and the pool habitats towards greater *S. alterniflora*.

ANOSIM comparisons echoed the univariate analyses in identifying that the vegetation communities of both the pool and platform habitats required at least 3–4 years to recover significantly from pre-restoration conditions (Table 3). Dissimilarity from pre-restoration vegetation communities continually increased across both platforms and the south pool site. Dissimilarity in the north pool site declined after 2018 and became similar to pre-restoration conditions in 2021. SIMPER analyses indicated that *S. alterniflora* drove dissimilarity in the pools over time (Table 4), indicating the vegetation community mainly improved through the colonization of the low marsh grass. In the platform habitats, the contribution of *S. alterniflora* declined after 2016 and was replaced by *S. patens* as the largest contributor of dissimilarity at both sites. At the platform adjacent to the south pool, *S. alterniflora* was not in the top three contributions to dissimilarity in 2021, demonstrating the continued increase in high marsh grasses of *S. patens* and *D. spicata*.

Table 3

Pairwise analysis of similarity comparisons of the vegetation community over time. For the pool and platform habitats, the vegetation community for each year was compared to the pre-restoration community of Spring 2015. Additionally, the vegetation communities between the habitats were compared for each year. Communities that are significantly different (Bonferroni Corrected $p < 0.004$ per site) are highlighted.

		Pool		Salt Marsh Platform	
Site	Year	Global R	p	Global R	p
North	2015 (Summer)	0.003	0.350	0.039	0.117
	2016	0.119	0.006	0.003	0.338
	2018	0.501	< 0.001	0.289	< 0.001
	2021	0.199	0.001	0.413	< 0.001
South	2015 (Summer)	-0.006	0.491	-0.033	0.839
	2016	0.147	0.003	0.035	0.144
	2018	0.822	< 0.001	0.406	< 0.001
	2021	0.939	< 0.001	0.455	< 0.001

Table 4

Similarity Percentage (SIMPER) analysis for the vegetation community for each pool site and habitat. Pairwise SIMPERs were compared for each year to the pre-restoration vegetation community (Spring 2015). The three species that explained the most dissimilarity between vegetation communities are provided. The vegetation species are: SPALT = *Spartina alterniflora*, SPPAT = *Spartina patens*, DISPI = *Distichlis spicata*, RUMAR = *Ruppia maritima*, and Non-live = Bare + Dead + Wrack + Algal Mat + *Vaucheria* algae.

Site	Habitat	Year	Species 1	Dissimilarity	Species 2	Dissimilarity	Species 3	Dissimilarity
North	Pool	2015	SPALT	44.2%	SPPAT	26.2%	Non-live	19.7%
		2016	SPALT	45.2%	SPPAT	25.8%	Non-live	22.8%
		2018	SPALT	41.6%	Non-live	33.6%	SPPAT	21.1%
		2021	SPALT	39.6%	SPPAT	31.3%	Non-live	25.5%
	Platform	2015	SPALT	31.8%	SPPAT	30.5%	Non-live	16.9%
		2016	SPALT	36.0%	SPPAT	33.1%	Non-live	19.0%
		2018	Non-live	35.3%	SPPAT	25.5%	SPALT	24.6%
		2021	SPPAT	31.1%	Non-live	30.5%	SPALT	28.5%
South	Pool	2015	SPALT	37.0%	RUMAR	17.4%	SPPAT	16.6%
		2016	SPALT	38.1%	SPPAT	15.0%	Non-live	14.6%
		2018	SPALT	34.3%	Non-live	27.0%	DISPI	14.1%
		2021	SPALT	31.0%	Non-live	26.4%	SPPAT	19.8%
	Platform	2015	SPALT	28.6%	SPPAT	27.6%	Non-live	19.0%
		2016	SPPAT	28.5%	SPALT	27.9%	Non-live	19.2%
		2018	Non-live	25.8%	SPPAT	25.6%	DISPI	20.2%
		2021	SPPAT	27.9%	Non-live	23.8%	DISPI	20.5%

Elevation

Elevations of each habitat zone across the pool sites remained relatively unchanged and within the accuracy of the RTK-GPS (± 3 cm) after runnel construction (Table 5). The difference in elevation between the pool and

immediate platform increased by 3 cm at the north site and decreased by 1.5 cm at the south site.

Table 5

Elevation measurements across pool site and habitat zone. Elevations were not tied into NAVD88 m datum with an RTK-GPS in Spring 2015. Difference in elevation between habitat zones are reported with the pool habitat. Values reported as mean +/- standard error.

Site	Year	Pool Mean Elevation (NAVD88 m)	Platform Mean Elevation (NAVD88 m)	Difference in Elevation (m)
North	Spring 2015	-	-	0.172 +/- 0.010
	2016	1.26 +/- 0.01	1.41 +/- 0.01	0.153 +/- 0.008
	2021	1.22 +/- 0.01	1.44 +/- 0.01	0.223 +/- 0.015
South	Spring 2015	-	-	0.160 +/- 0.013
	2016	1.31 +/- 0.01	1.47 +/- 0.01	0.158 +/- 0.011
	2021	1.28 +/- 0.01	1.43 +/- 0.01	0.145 +/- 0.012

Remote Sensing of Vegetation Recovery

The remote sensing analysis provided a site-level understanding of the recovery of the vegetation community as the vegetation expanded beyond the scope of the pool edge monitoring. Remote sensing analysis showed a trend of pool expansion and high marsh soil saturation before runnel construction and then vegetation recovery and expansion into the pools after restoration activities (Fig. 7, Table 6). Following runnel construction, vegetation expanded along the entire perimeter of the south pool with considerable increases along the northern and southern perimeter as well as southeastern wet areas. Vegetation expanded 0.37 ha within four growing seasons after runnel construction. In the north pool, vegetation expansion was minor (0.1 ha) yet consistent along the entire pool perimeter including the emergence of several patches within the pool in the south by 2018. It should be noted that two non-hydrologically connected pools just north of the north pool continued to expand and these unvegetated areas were measured within the 5 m buffer, interfering with the estimate of vegetation recovery.

Table 6

Unvegetated - Vegetated Ratio Analysis results. The pool outline area as manually drawn based on 2012 imagery and remained constant throughout the analysis.

Site	Pool Outline Area (ha)	Year	Unvegetated Pixels	Vegetated Pixels	Unvegetated Area (ha)	Vegetated Area (ha)
North	0.77	2012	5915	1800	0.59	0.18
		2014	5950	1765	0.59	0.18
		2016	5972	1733	0.60	0.17
		2018	5756	1948	0.58	0.19
South	4.28	2012	32721	10070	3.27	1.01
		2014	33965	8826	3.40	0.88
		2016	33412	9371	3.34	0.94
		2018	30277	12507	3.03	1.25

Discussion

Connection of Pools to Hydrologic Pathways

Despite the limited hydrologic data (1 month) which can yield highly variable results year-to-year, the data clearly demonstrates that runnels altered existing pool hydrology through reconnection with tidal channels and enhanced drainage of the surrounding platform marsh. When put into the context of increasing high tide flooding frequency during the monitoring period, the declines in flooding duration in the pool and on the platform are more impressive. For example, the south pool experienced 50% less time flooded in 2021 despite a 70% increase in flooding frequency since 2015 (Table 1). Each system required at least 2–3 growing seasons to match tidal cycles and drain based on lagging water level elevations following each high tide in 2016. An immediate improvement of hydrology is well-documented for landscape-scale salt marsh restoration projects that improve tidal hydrology (Buchsbaum et al. 2006, Smith et al. 2009, Raposa et al. 2019), however, long-term monitoring (> 5 years) of water elevations provides additional insight into the gradual recovery of the tidal cycle unique to drained pool systems (see Smith et al. 2009, Karberg et al. 2018)

Hydrologic gains were stronger in the south pool than in the north pool throughout the monitoring based on water level recorders. Contrasting runnel and hydrology trajectories between the south and north pools after 2018 impacted the recoveries of the vegetation community. Enhanced drainage and tidal flushing, especially 5–10 cm belowground, can ameliorate biogeochemical stress on the vegetation community including reducing salinity, sulfides, and anoxia (Flynn et al. 1999). The inability of the system to fully drain each tidal cycle restricts the colonization of *S. alterniflora* to the outer edges of the pool and may prevent high marsh graminoids from becoming dominant on the adjacent marsh platform. The salt marsh platform of the north site continued to see improvements in drainage in 2021 (Table 1), despite less drainage in the pool, suggesting the runnels, though partially clogged, still improved biogeochemical conditions of the adjacent marsh platform.

Recovery of the Vegetation Community

Runnels facilitated significant, yet divergent, recoveries of the vegetation community of the pool and adjacent high marsh platform. Besterman et al. (2021) outlined the recovery of the vegetation community from runnels in three phases: drainage of standing water and increase in bare peat, establishment of early colonizers like *S. alterniflora* and *Salicornia* spp., and incremental increase in high marsh graminoids species. The south pool followed the recovery model well as it underwent loss of standing water, increase of *S. alterniflora* in 2015–2018 followed by replacement of high marsh graminoids in 2021. In the north pool, *S. alterniflora* had variable increases in 2015–2018 followed by a decline in 2021, attributed to the runnel becoming clogged.

The vegetation community of platform sites had an initial minor peak in *S. alterniflora* within 2 growing seasons followed by the replacement of HMGs as observed by Perry et al. (2021). Despite limited drainage by 2021 in the north pool, the marsh platform had similar HMG cover and HMH ratio to the south site. Drainage may have beneficially impacted the marsh platform as much as 5 m away from the pool perimeter (Raposa et al. 2019) without increased rates of decomposition and subsidence (Perry et al. 2021). Remote sensing analysis confirmed this observation with the revegetation of pannes around the edges of the south pool and expansion of vegetation along the entire perimeters of both pools. Vegetation classification from aerial imagery or field plots extending further into the marsh platform should be considered in future monitoring plans for evaluating pool drainage as traditional monitoring techniques may not capture the full extent of the recovery (see Ganju et al. 2017, Wasson et al. 2019).

Trajectory of Pool Recovery

The main restoration goal for runnels is to “push” the expanding pool or oversaturated area back into the negative feedback loop of self-maintenance by reducing waterlogging on the salt marsh surface (Burdick and Roman 2012). Ideally, connecting the pools to tidal exchange will set the system on a long-term trajectory to recover elevation loss and sustain a high marsh vegetation community. Previous drainage studies (Raposa 2008, Raposa et al. 2019, Perry et al. 2021, Besterman et al. 2022) have documented immediate improvement of tidal exchange in oversaturated areas within years and gradual recolonization by halophyte vegetation over 4–7 growing seasons. A lag of 1–2 growing seasons after restoration activities was noted for substantial *S. alterniflora* growth in the pools, which was driven by incremental amelioration of soil and porewater chemistry (Raposa 2008, Smith et al. 2009). Recovery timeframes observed in runnel projects have also been documented in other tidal restoration efforts (i.e., culvert replacements, impoundment removals, and living shorelines) with immediate recoveries of hydrology and porewater chemistry and gradual recovery of vegetation within 5–10 years (Konisky et al. 2006, Smith and Warren 2012, Bilkovic and Mitchell 2017).

A crucial question that still remains about runnel projects is the ability of the pool to build in elevation fast enough to close the gap with the marsh platform. Continued gains of HMGs throughout the drained pool depends on elevations gains in the long-term. In this study, the difference in pool edge and platform elevation at the south site would require at least 70 years based on the observed elevation gain rate of 2.1 mm yr⁻¹ from 2015 to 2021. Through surface elevation table (SET) monitoring and modeling, revegetating pools in New England had increased accretion rates of 2–8 mm yr⁻¹ greater than the adjacent marsh interior (Wilson et al. 2014, Mariotti et al. 2020). Historic aerial photography has shown that complete pool recovery requires multi-decade timeframes (Wilson et al. 2009, 2014, Mariotti 2016, Smith and Pellew 2021). Long-term field monitoring of surface elevation (via surveying and/or SETs) at future drainage projects should be conducted to better understand pool recovery trajectories.

Management Considerations of Pool Drainage

Runnelling is an effective tool for drainage of large, ponded areas resulting from historic marsh manipulations but should not be widely applied to all pools on the landscape. Pools are natural features of stable interior marshes and provide nesting, foraging, and refugia habitat for nekton and avian species (Adamowicz and Roman 2005, Raposa and Roman 2006, Smith and Niles 2016). Through interpretation of stratigraphic coring and historic aerial photography, coastal ecologists better understand that pools undergo natural cycles of birth, expansion, merging, and recovery (Wilson et al. 2009, 2010, Smith and Pewell 2021). However, the very large pools that developed in the 20th century following abandonment of embankments and ditches (Adamowicz et al. 2020) are unnatural legacies of past agricultural practices. Rather than allow these pools to expand and degrade the underlying peat for decades (with losses in marsh elevation and stored carbon), runnels can be used to drain shallower portions and revegetate, rehabilitating the marsh platform. Projects that use runnels should include a long-term monitoring program that reflects project goals to gauge performance. Hydrology, vegetation, surface elevation, and avian monitoring are common metrics that may be used to evaluate the trajectory of the pool recovery (see Neckles et al. 2002, Ganju et al. 2017, Adamowicz et al. 2020). During post-restoration monitoring, maintenance of runnels or site-specific adaptive management should be considered.

Conclusions

Runnelling is a valuable tool in the restoration toolbox for coastal managers and ecologists to prevent runaway marsh collapse and pool expansion in the marsh interior, improve the health of the high marsh, and promote endemic sparrow conservation. Although tidal exchange and drainage are achieved relatively quickly, the vegetation community may require at least several years for substantial colonization of *S. alterniflora* and HMGs into the pool edge. Complete pool recovery to high marsh with small, interspersed pools may take several decades as the elevation of the basins reaches parity with the marsh platform (Wilson et al. 2014), though increasing rates of SLR (> 5 mm/yr) may prevent complete recovery. As mega-pools revegetate, tidal hydrology improves the health of the vegetation community of the surrounding high marsh community and likely the nesting habitat of salt marsh sparrows. Although not an immediate or permanent fix to marsh degradation from the interaction of SLR with relict agricultural features, runnelling has the potential to stimulate the recovery of valuable high marsh habitats from large and expanding unvegetated pools. Such efforts may increase resiliency for decades or at least until new high marsh is formed from upland migration (Besterman et al. 2022).

Declarations

Acknowledgments:

We would like to thank the Rachel Carson and Parker River National Wildlife Refuges for their cooperation and support of the research. The Refuge staff including Susan Adamowicz, Nancy Pau, Frank Drauszewski, Bob Springfield, and Katie Bland were incredibly helpful and supportive of field work. Funding for this research was provided by Hurricane Sandy funds (DOI FWS/R5/CGS #F14AP00522 and DOI FWS/CGS #F15 AC01066) administered through the United States Fish and Wildlife Service and by the Atlantic Coast Joint Venture, grant number F21AP00933. Contribution number XXX of the Jackson Estuarine Laboratory, University of New Hampshire.

Funding

Funding for this research was provided by Hurricane Sandy funds (DOI FWS/R5/CGS #F14AP00522 and DOI FWS/CGS #F15 AC01066) administered through the United States Fish and Wildlife Service and by the Atlantic Coast Joint Venture, grant number F21AP00933.

Competing Interests

The authors have no relevant financial or non-financial interests to disclose.

Author Contributions

McKown, Burdick, Moore, Peter, Payne, and Gibson made substantial contributions to the data collection in the field and review and editing of the manuscript. Burdick and Moore were responsible for the conception, funding acquisition, experimental design, and supervision of the study. McKown, Peter, Payne, and Gibson were responsible for the data analysis, data visualization, and data interpretation. McKown was responsible for the original draft of the manuscript. All authors agree to be accountable for all aspects of the work in ensuring that questions related to the accuracy or integrity of any part of the work are appropriately investigated and resolved. All authors read, reviewed, and approved the final manuscript.

Ethics Committee Approval

Approval from ethics committees at the University of New Hampshire (IACUC, IRP) were not required for this study.

Data Availability

The datasets generated during the current study are available in the Figshare Repository 'Pine Island Runnel, Great Marsh, Massachusetts 2014 – 2021 Dataset', doi: 10.6084/m9.figshare.19666242. The dataset includes vegetation community, hydrology, surface elevation, plot locations, and geospatial analysis results. R-script for water level elevation and multi-variate analyses are available from the corresponding author on reasonable request.

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Figures

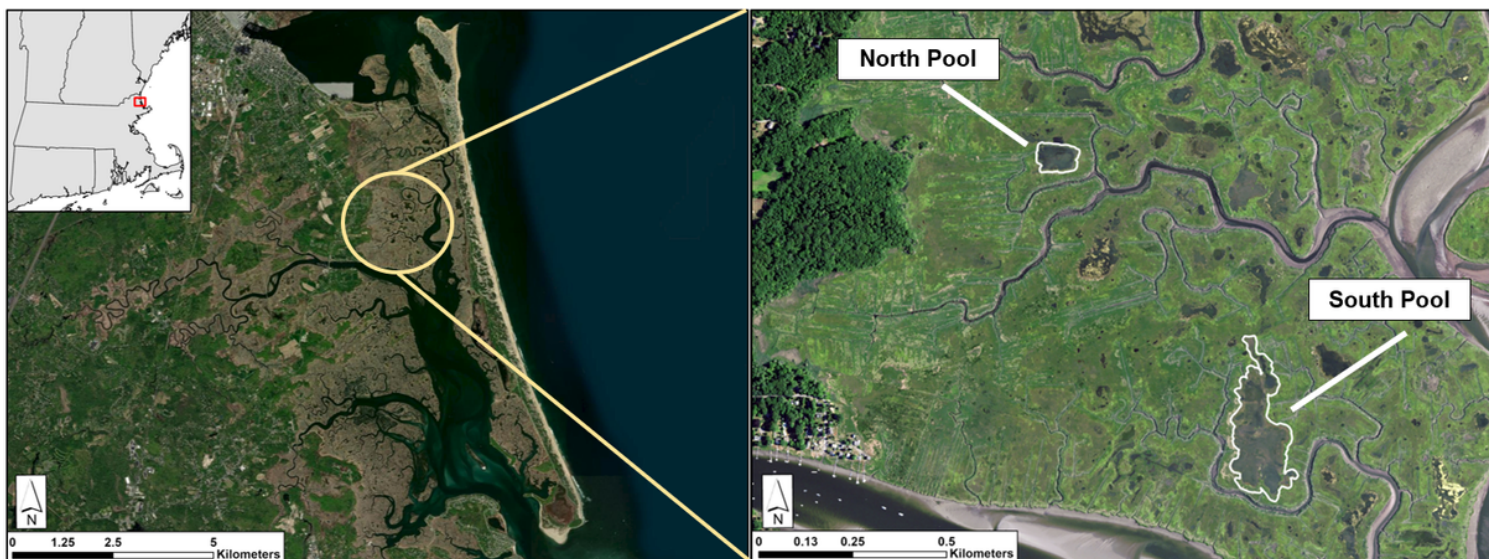


Figure 1

Left: Site map of study area in the Parker River National Wildlife Refuge (Newbury, Massachusetts). **Right** North and south pool are outlined in white

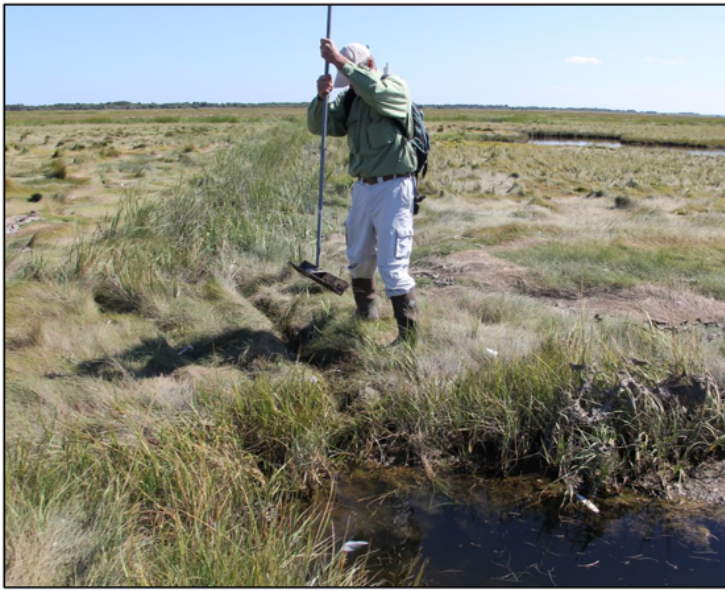


Figure 2

Left: Creation of runnel by compacting the marsh surface using a handheld tamping tool at the north pool site in 2015. Runnel connected an existing ditch to the expanding pool. **Right:** Runnel at the south pool site in 2015 after construction. Runnels were created with dimensions of approximately 30 cm wide - 15 cm depth and lengths of 60 – 200 cm

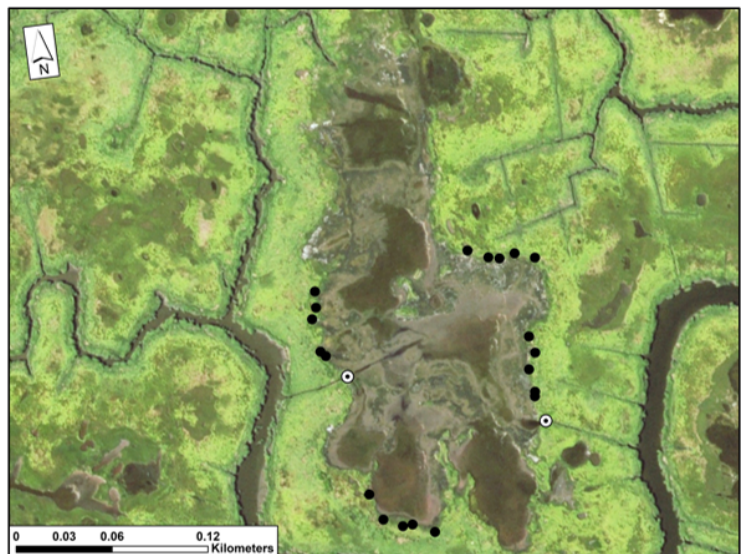
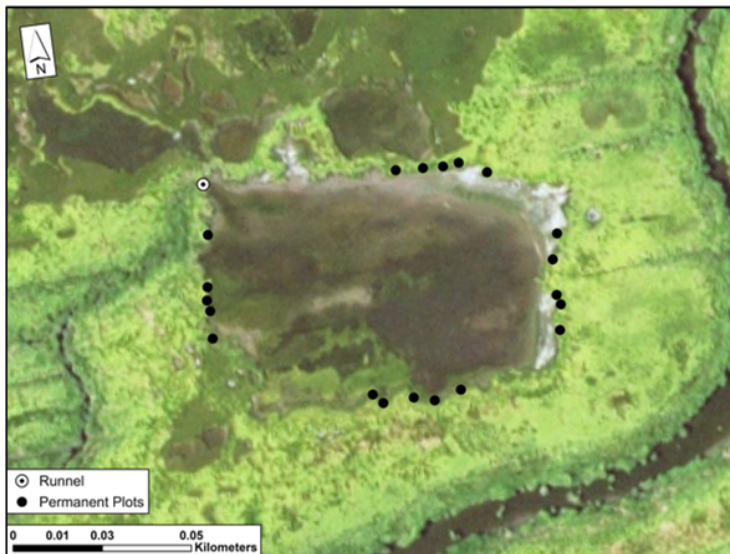


Figure 3

Field sampling design and runnel construction (white) for north and south pools. Permanent plots (black) consist of paired salt marsh platform edge and pool plots

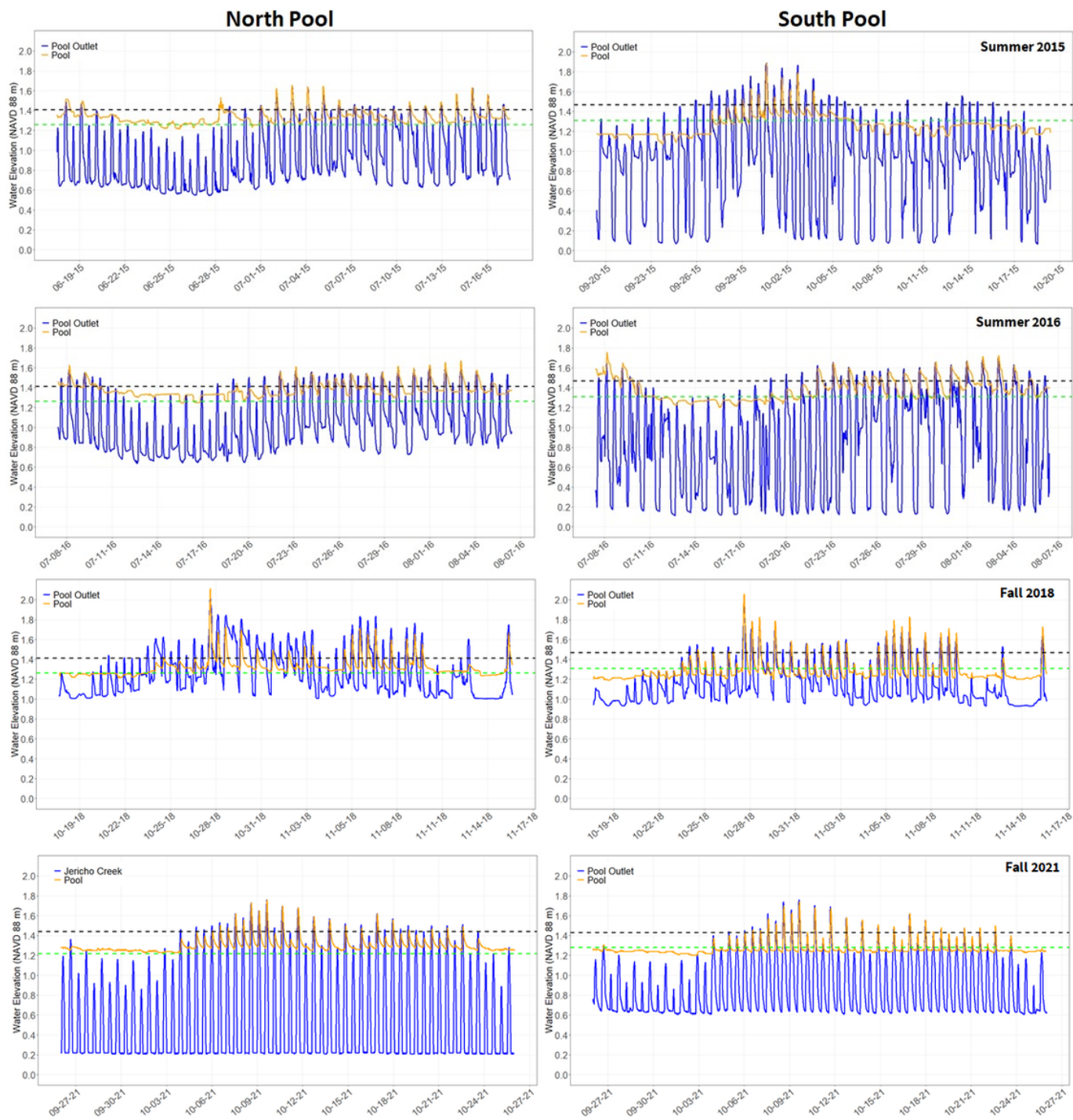


Figure 4

Water level elevations for hydrologic channel (blue) and pool (orange) water level recorders in Summer 2015 (after runnel construction), Summer 2016, Fall 2018, Fall 2019. The north pool is shown on left and south pool on the right. The average salt marsh platform elevation (black dashed line) and pool elevation (green dashed line) are shown for each site

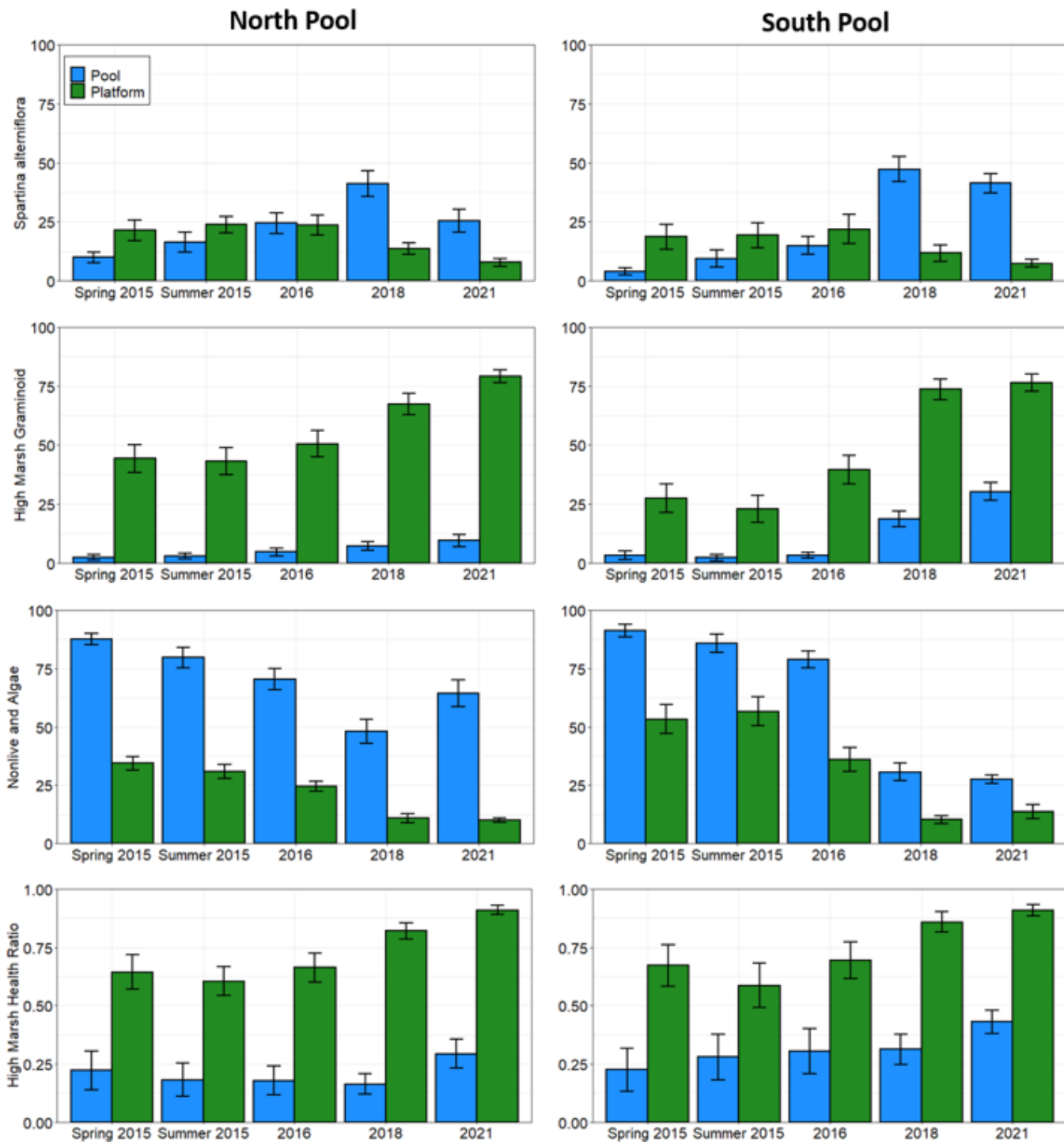


Figure 5

Vegetation metrics of *Spartina alterniflora* cover, high marsh graminoid cover, and non-live and algae cover in the North (**Left**) and South (**Right**) pool sites. Error bars are +/- standard error

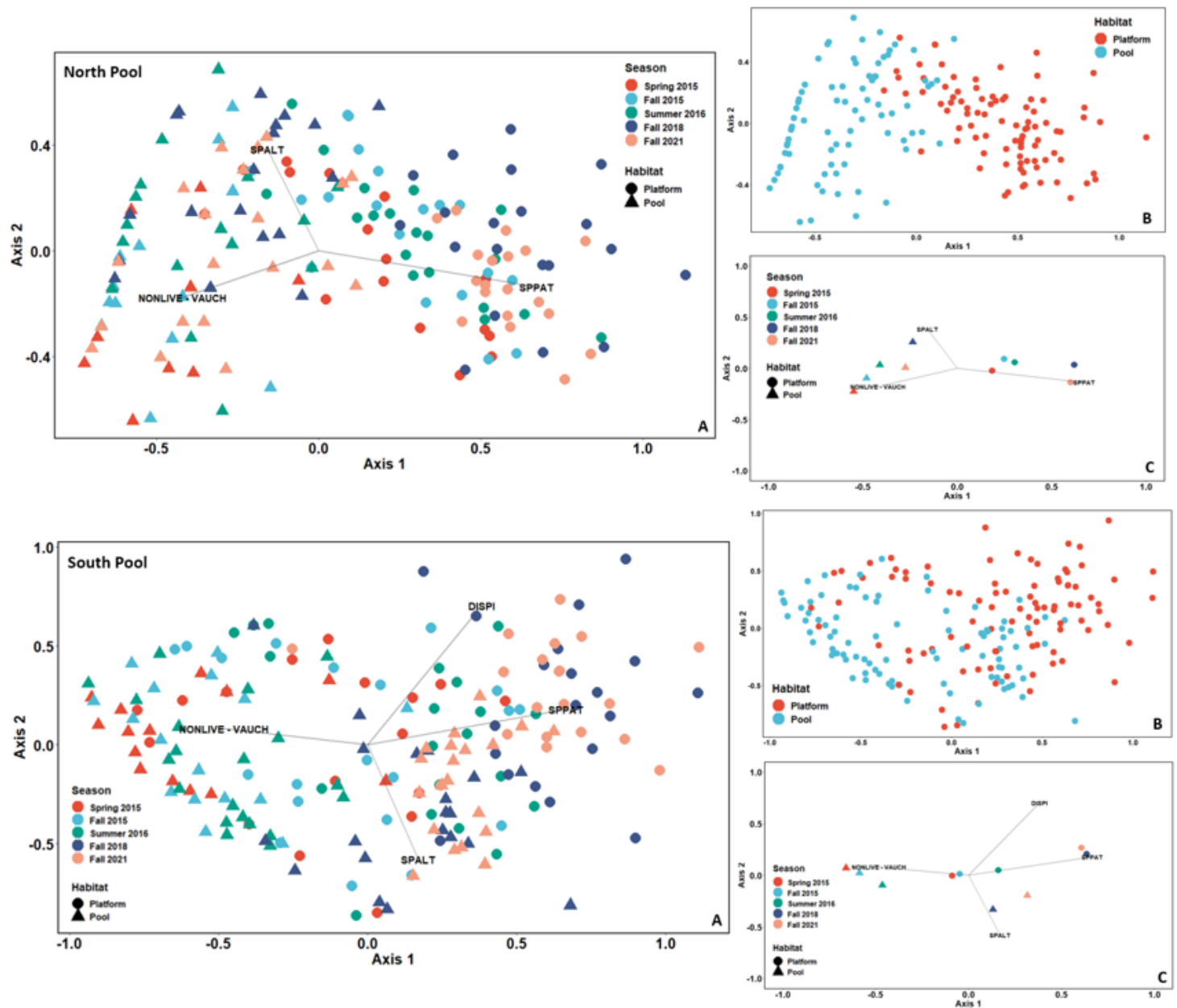


Figure 6

(A) Non-metric multidimensional scaling ordination (NMDS) of the north pool (**Top**) and south pool (**Bottom**). Vegetation species that were highly and significantly correlated ($r^2 > 20\%$, $p < 0.05$) are overlain in main NMDSs. **(B)** Main NMDS with plots symbolized by either pool or platform habitat. **(C)** The mean locations for each habitat and season from the main NMDS were calculated and visualized. Species abbreviations are DISPI = *Distichlis spicata*, SPALT = *Spartina alterniflora*, SPPAT = *S. patens*, NONLIVE – VAUCH = Non-live Cover

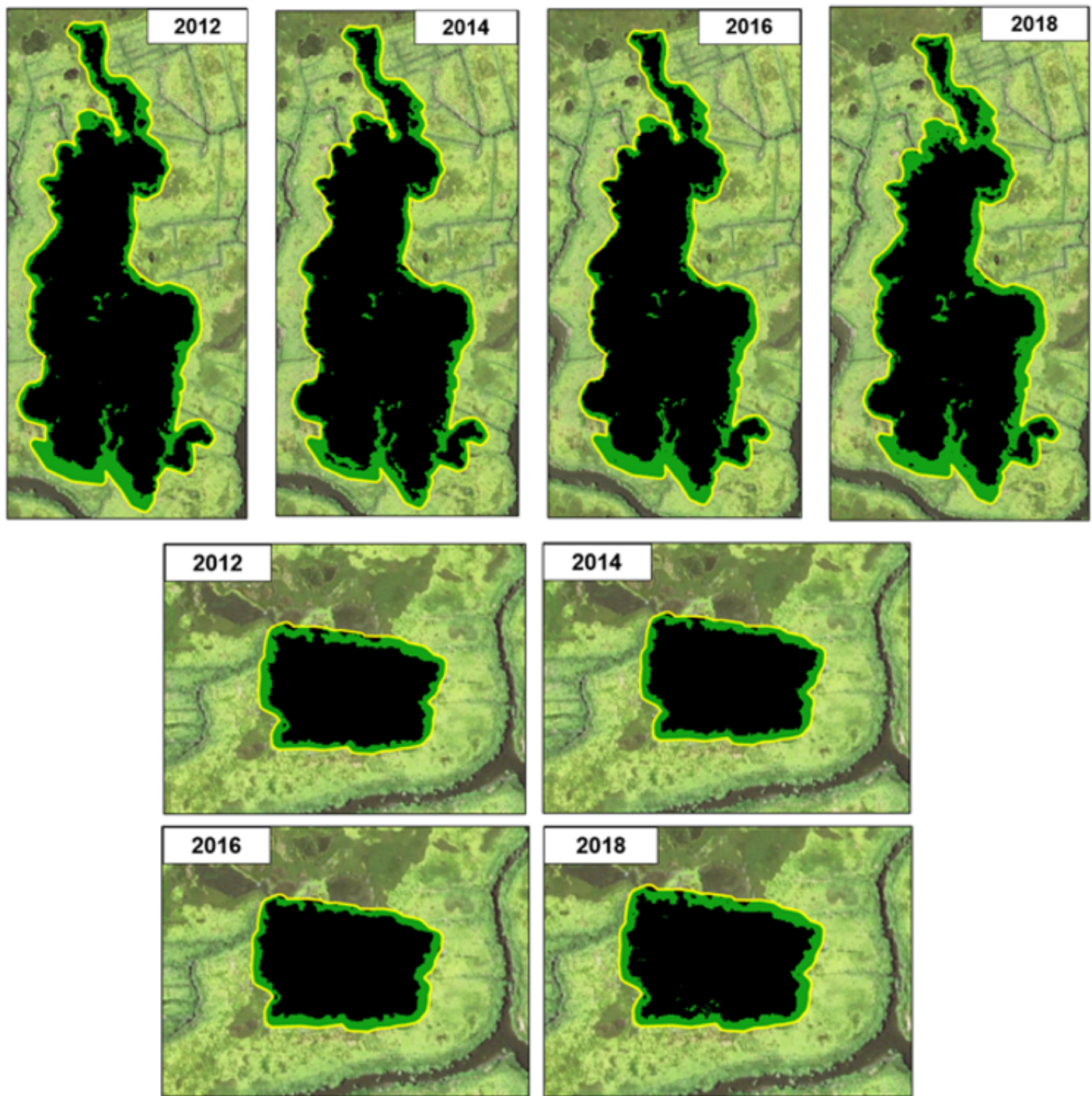


Figure 7

Remote sensing analysis of South (**Top**) and North Pools (**Bottom**). Runnel construction took place in Spring 2015. National Agriculture Imagery Program aerial imagery of 2018 is background for all panels. Vegetated areas are shown in green and unvegetated areas in black. The 5 m buffer of the pool outline is shown in yellow