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Widespread Expansion of Salt Marsh Pools Observed on Maine Marshes Since 2009

K. DeWater[1], W. Kochtitzky[1], R. Ellis[1], P. Merrill[1], M. Pittsley[1], P. Morgan[1], C. Burns[2], A. Campbell[3,4], and S. Adamowicz[5]

[1]School of Marine and Environmental Programs, University of New England, Biddeford, ME, USA, [2]Chesapeake Bay National Estuarine Research Reserve - MD, Annapolis, MD, USA, [3]Goddard Space Flight Center, NASA, Rockville, MD, USA, [4]Goddard Earth Sciences Technology and Research II, University of Maryland, Baltimore County, MD, USA, [5]Rachel Carson National Wildlife Refuge, Wells, ME, USA

Key Points:

- From 2009 to 2021, salt marshes in Maine have experienced a 16% increase in pool cover
- Mega-pools are larger at lower elevations of the marsh and expand the most when merging with other pools
- Pool expansion can adversely impact a marshes ability to sequester carbon, supporting the need for restoration efforts on Maine's marshes

Supporting Information:

Supporting Information may be found in the online version of this article.

Correspondence to:

K. DeWater,
kdewater@une.edu

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Author Contributions:

Conceptualization: K. DeWater, W. Kochtitzky, P. Morgan
Formal analysis: K. DeWater, W. Kochtitzky, C. Burns
Funding acquisition: W. Kochtitzky
Investigation: K. DeWater
Methodology: K. DeWater, W. Kochtitzky
Software: K. DeWater, W. Kochtitzky
Supervision: W. Kochtitzky, P. Morgan, A. Campbell, S. Adamowicz
Validation: R. Ellis, P. Merrill, M. Pittsley
Visualization: K. DeWater
Writing – original draft: K. DeWater
Writing – review & editing: W. Kochtitzky, P. Morgan, C. Burns, A. Campbell, S. Adamowicz

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Abstract Salt marshes provide critical habitats, protect coastal infrastructure, and are increasingly exposed to sea level rise, with many having a history of agricultural use and ditching over the centuries. Pool area coverage can be considered an indicator of marsh health but is rarely quantified. In this study, we digitized marsh pools using aerial imagery to quantify changes in pool area and density on 12 salt marshes in Maine from 2009 to 2021 as a case study of marsh response to environmental changes. We categorized pools into three types: mega-pools, individual pools, and perimeter pools, based on morphology and examined whether pools remained singular, split, or combined. We found a 15.7% increase in pool area from 2009 to 2021 on all marshes, primarily driven by mega-pool expansion, whereas individual and perimeter pools remained relatively constant. The rate of pool expansion across all marshes was 49,000 m^2 a^{-1} with mean mega-pool size 6,400 ± 400 m^2. There was an increase in pool count per km^2 on all marshes except for the York River marsh, which still experienced area expansion. Pools primarily increase in cover through merging or being engulfed by mega-pools. Area cover change was not substantial when pools remained singular, split into many pools, or were only present in 2009 or 2021. Mega-pools were larger on lower marsh elevations and expanded at a greater rate when overlapping ditches, suggesting influence by sea level rise and historic agriculture. Marsh restoration projects that promote the drainage and re-vegetation of mega-pools may reverse this trend.

Plain Language Summary Salt marshes are habitats for a variety of species, many of which are endangered or threatened. Marshes also provide a buffer to coastal infrastructure and reduce damage from coastal flooding. Continued expansion of standing water, or pools (also called ponds), on marshes could cause habitat loss and limit their ability to perform ecosystem services. To quantify pool change, we mapped salt marsh pools present in 2009 and 2021 for 12 salt marshes in Maine. Maine's salt marshes experienced notable pool development, having increased standing water on marsh surfaces and reduced grass cover. We grouped pools based on shape: mega-pool (large, disconnected pools, associated with expansion), individual pools (small, singular pools, defined borders), and perimeter pools (long, thin pools, along embankments). We found an increase in pool area from 2009 to 2021 in all marshes driven by mega-pool expansion. The primary way in which pool cover increased was through pools combining to form mega-pools or being overtaken by existing mega-pools. Expansion could be occurring because of increased flooding due to sea level rise, prompting erosion, and lasting effects of historic agricultural practices on the marsh surface. Future marsh restoration projects promoting the drainage and re-vegetation of mega-pools may reverse this trend.

1. Introduction

1.1. Marshes, Sea Level Rise, and Human Impacts

Salt marshes are dynamic ecosystems that buffer coastlines from intense storms (Temmerman et al., 2012), sequester blue carbon (Brewer et al., 2023), and provide critical habitat for endangered and rare species (Wigand et al., 2017). Northeast U.S. salt marshes have experienced degradation due to human activities such as agricultural drainage, tidal restrictions, and ditching (Adamowicz et al., 2020; Burdick et al., 2020; J. A. M. Smith et al., 2022), along with climate change-driven sea level rise (Hartig et al., 2002; Kearney et al., 1988; Watson et al., 2016). In the 19th and 20th centuries, ditches were constructed for mosquito control, and farmers used ditches, dikes, and embankments to drain sections of marshes for hay cultivation, which now has lasting effects on pool development (Burdick et al., 2020; Sebold, 1998; D. C. Smith & Bridges, 1982; D. C. Smith et al., 1989). Sea level rise contributes to increased inundation and changes in tidal dynamics, while historic ditching disrupts natural drainage patterns and can lead to altered flow and sediment deposition (Corman et al., 2012; J. A. M. Smith et al., 2022; S. M. Smith et al., 2017). Hydrologic alteration can prompt the transition of vegetated marsh to pools and mudflats as marshes erode due to frequent and sustained inundation (Schepers et al., 2020). Pool expansion and formation may be an indication of increased inundation, though pools may also result from other mechanisms (Cavorta et al., 2003; Kearney et al., 1988). Given these environmental changes, Northeast salt marshes are an exemplary location to better understand changing morphology and marsh response. There is a growing need to document pool changes to support marsh restoration and to monitor the efficacy of future restoration projects globally (McKown et al., 2023; Watson et al., 2016; Wigand et al., 2017).

Threshold sea level rise rates for marsh survival depend on sediment supply, tide range and marsh inundation, and site-specific vegetation. Studies throughout the Northeast have indicated marsh vulnerability to submergence due to accelerated sea level rise using measurements and modeling (Raposa et al., 2017; Watson et al., 2017). For Maine's marshes, vertical accretion rates were measured to be 2.8 mm a^{-1} from 1986 to 2003, and most marshes could keep pace with sea level rise at that time (Goodman et al., 2007). However, accretion rates may vary among locations within a marsh, particularly high versus low marsh, at a single location within a marsh over time, and between nearby marshes (Yellen et al., 2022). Therefore, it is difficult to draw conclusions about marsh response to sea level rise based on accretion rates. However, with sea levels rising at 3 mm a^{-1} since 1990 in Portland, Maine and accelerating (Fernandez et al., 2020), Maine marshes may not be keeping pace, leading to inundation and vegetation dieback. Here, we provide observational evidence that thresholds may have been crossed for many of Maine's marshes by quantifying pool expansion, which can be indicative of increased inundation (Cavorta et al., 2003).

1.2. Salt Marsh Pools

The underlying mechanism for new pool formation is initial peat collapse due to the loss of living root network structures with subsequent pool expansion through erosion (Chambers et al., 2019; DeLaune et al., 1994; Himmelstein et al., 2021). Root networks are lost due to vegetation dieback, which can be caused by wrack deposition, ice rafting, and inundation from changing drainage channels (Erwin, 1996; Pethick, 1974; C. A. Wilson et al., 2014). Sustained inundation can also be caused by sea level rise (Temmerman et al., 2012) or agricultural embankments interfering with hydrology (Adamowicz et al., 2020; Burdick et al., 2020), making marsh grasses such as *Spartina alterniflora* and *Spartina patens* increasingly vulnerable to mortality due to saline stress, low oxygen, and high sulfide exposure (Alber et al., 2008; DeLaune et al., 1994; Kirwan & Guntenspergen, 2012). Pools can be evidence of a history of agriculture in marsh systems, as typified on the Webhannet marsh in Maine (Adamowicz et al., 2020; K. R. Wilson et al., 2010). It has been supported that most pools are secondary in origin or formed after the development of the marsh platform due to disturbances (K. R. Wilson et al., 2010). Due to the variety of disturbances to marsh surfaces, including the longstanding legacy of agriculture, it is difficult to discern the origin of existing pools.

Pools have been observed to form and drain to maintain an overall stable area cover on marshes in dynamic equilibrium. After a pool has formed, branches can develop from the main tidal channel to drain the pool, reducing inundation and saline stress and allowing for subsequent vegetation recovery (K. R. Wilson et al., 2009, 2014). However, it has more recently been shown that pools can undergo continual erosion following connection to drainage without vegetation recovery. Continued erosion after drainage is attributed to increased inundation from sea level rise and the exposure of soft-substrate pool bottoms to wave and tide action (Himmelstein et al., 2021; Mariotti et al., 2020). Unvegetated soil exposure can lead to widened drainage channels and runaway pool expansion, resulting in a loss of vegetated marsh surface and sediment export (Burns, Alber, & Alexander, 2021; Burns, Alexander, & Alber, 2021; Schepers et al., 2020). On intermediate and large sized pools, expansion is also caused by wind wave-driven edge erosion, whereas erosion of smaller pools is driven by biogeochemical processes (Himmelstein et al., 2021; Mariotti & Fagherazzi, 2013). As vegetation dieback occurs without recovery and pools expand, there may be a phase shift between the alternative stable states of vegetated marsh to unvegetated mudflat (Himmelstein et al., 2021; Kirwan & Murray, 2007; Morris et al., 2002).

The surface layer of vegetated marsh soils is notably efficient at sequestering carbon from the atmosphere, and states in New England have estimated the amount of carbon stored in marshes to count toward carbon neutrality goals (Alongi, 2020; Brewer et al., 2023). Though soil characteristics unique to pools are shallow and do not

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extend deep into the marsh vertically, it has been shown that soil surrounding expanding pools and inundated pannes store significantly less carbon and have decreased sediment stability compared to soils surrounding pools that are not expanding (Berkowitz et al., 2018; Kirwan & Mudd, 2012; Luk et al., 2023; Vincent et al., 2013, 2014). Pool expansion and development can shift the carbon balance of the marsh to be a carbon source instead of a sink as vegetation loss and sediment export are accelerated (Chambers et al., 2019; Himmelstein et al., 2021; Schepers et al., 2020). Therefore, understanding the patterns of changing pools on Maine marshes may help to inform carbon estimates and goals.

On the Webhannet marsh in Maine, there was an overall decrease in pool area from 1962 to 2003 (K. R. Wilson et al., 2009). However, pool development and expansion has since been anecdotally noted on the Webhannet, Biddeford Pool, and Goosefare Brook marshes in Maine and provide the basis for our analysis here. In Massachusetts, this pool expansion was described as mega-pooling, or runaway pool development in which pools continually expand (McKown et al., 2023).

Restoration efforts to develop shallow runnels to connect these mega-pools to drainage and promote re-vegetation have been effective (McKown et al., 2023) but may not be a long-term solution to assist marshes in adapting to sea level rise (Besterman et al., 2022). Additionally, the ability of marshes to sequester carbon and remain carbon sinks may be hindered by pool development that leaves soil exposed to erosion and decreases vegetation cover (Brewer et al., 2023). Therefore, it is necessary to monitor changes in marsh pool dynamics for the management and evaluation of future restoration efforts. Pool coverage has yet to be quantified on many of New England's salt marshes, including those in Maine, which is fundamental to understanding the impacts of sea level rise and historic agricultural practices on marsh hydrology and overall resilience.

1.3. Study Objectives

In this study, we aimed to understand how salt marsh pools have changed in area based on morphology, elevation, and their coincidence with ditches. We focused on 12 of the largest salt marshes in Maine (Figure 1), using them as a case study to understand pool dynamics and changes in the region. We hypothesized that there would be an increase in pool area, particularly of mega-pool area, across the 12-year period on all marshes. Therefore, the purpose of this research is to

1. Quantify changes in pool size, pool count, and pool type from 2009 to 2021
2. Categorize pools based on morphologies and expansion patterns (i.e., Mega-pools)
3. Compare pool changes to marsh elevation and relative rates of sea level rise
4. Generate a census of digitized pools for the largest salt marshes in southern Maine as a baseline for future restoration projects and impacts of climate change
5. Inform management decisions about which marshes need to be prioritized for restoration

2. Materials and Methods

2.1. Imagery

We selected every marsh described as "salt or brackish marsh" with *Spartina* as the dominant community in Southern Maine larger than 0.5 km^2 as classified by the Maine Natural Areas Program Current Tidal Marshes layer (Gawler & Cutko, 2015; Odum, 1988). The 12 marshes in Maine that were selected (in order of decreasing area) were Scarborough, Webhannet River, Spurwink, Marshall Point, York River, Ogunquit River and Stephens Harbor, Brave Boat Harbor, Cousins River, Biddeford Pool, Little River, Goosefare Brook, and Gooch's Beach marshes (Figure 1; see Table S2 and S3 for site descriptions). Ten of the marshes are back barrier marshes, separated from the coastline by sandy beaches and dunes; the York River and Cousins River marshes are along tidal rivers (Kelley et al., 1988).

We downloaded aerial photos from the National Agricultural Imagery Program (NAIP) through the United States Geological Survey Earth Explorer for 2009 and 2021. NAIP provides high spatial resolution and open-access imagery every four years, which is necessary for sufficiently accurate digitization. We selected the year 2021 because it is the most recent imagery available. The year 2009 was chosen because it is the earliest imagery available with RGB values, ensuring consistency in digitization by using color imagery rather than black and white. All aerial photos for 2009 were collected in July, and the 2021 aerial photos were taken in July and October (Tables S1–S3). Although some marshes were compared between summer and fall due to data availability, our

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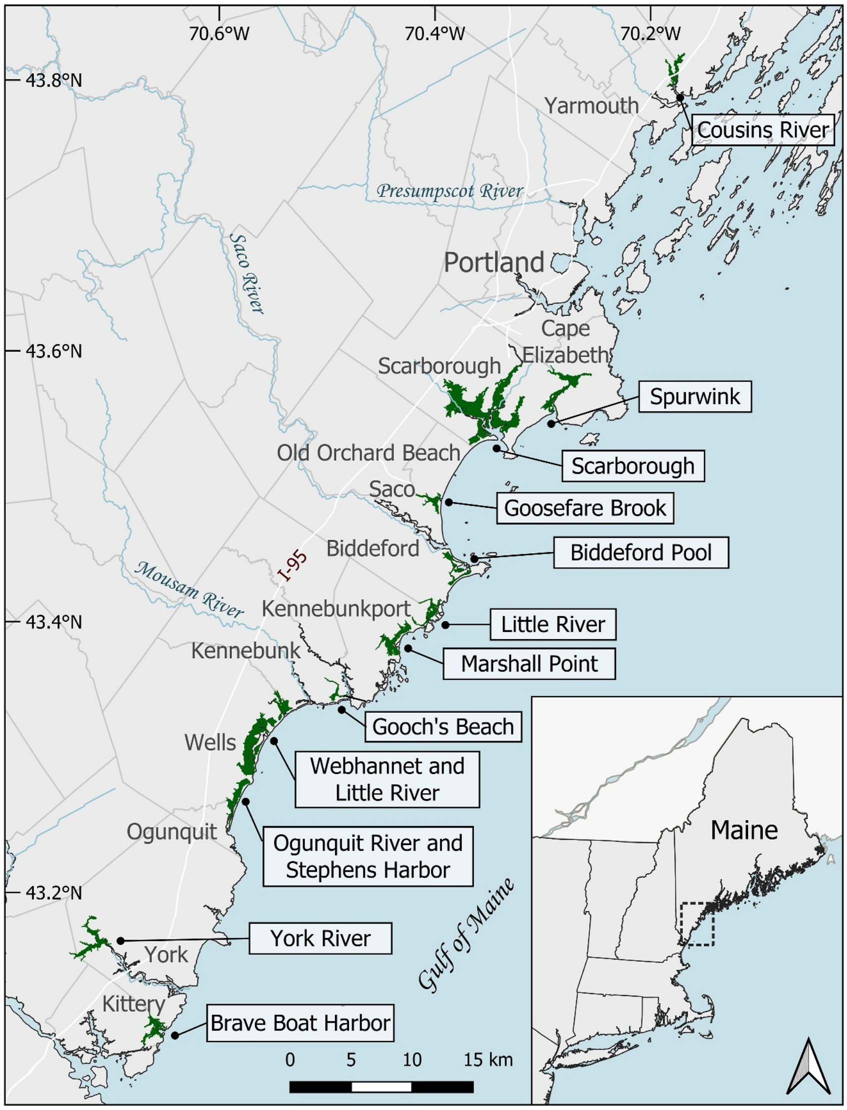

Figure 1. Map of the selected marshes for which pools were digitized. Marshes are shown in green. The marsh polygons were retrieved from the Maine Natural Areas Program Current Tidal Marshes layer.

data set aims to capture long-term trends rather than short-term seasonal cycles. The spatial resolution of the aerial photos is 1 m for 2009 and 0.6 m for 2021. All aerial photos were taken during comparable low tidal stages and were not collected during periods of extreme precipitation events (Table S2 and S3).

2.2. Digitization

We define salt marsh pools following Adamowicz & Roman (2005)as depressions on a marsh that retain water throughout a tidal cycle and are not connected to tidal channels. Salt marsh pools as we define them are also referred to as ponds (Schepers et al., 2020) or interior pools (Watson et al., 2016). For this study, we did not include pannes, defined as depressions on the marsh surface that do not retain standing water (Adamowicz & Roman, 2005), since they have not been linked to the feedback loop of salt marsh erosion (Schepers et al., 2020). We further defined pools into three categories as their features were apparent and consistent in all observed marshes (Figure 2):

- Mega-pool: Defined as disjointed borders, typically in higher marsh, and characterized by being surrounded by smaller fragmented pools (described by Himmelstein et al. (2021); named by McKown et al. (2023)), typically have areas greater than hundreds of square meters.

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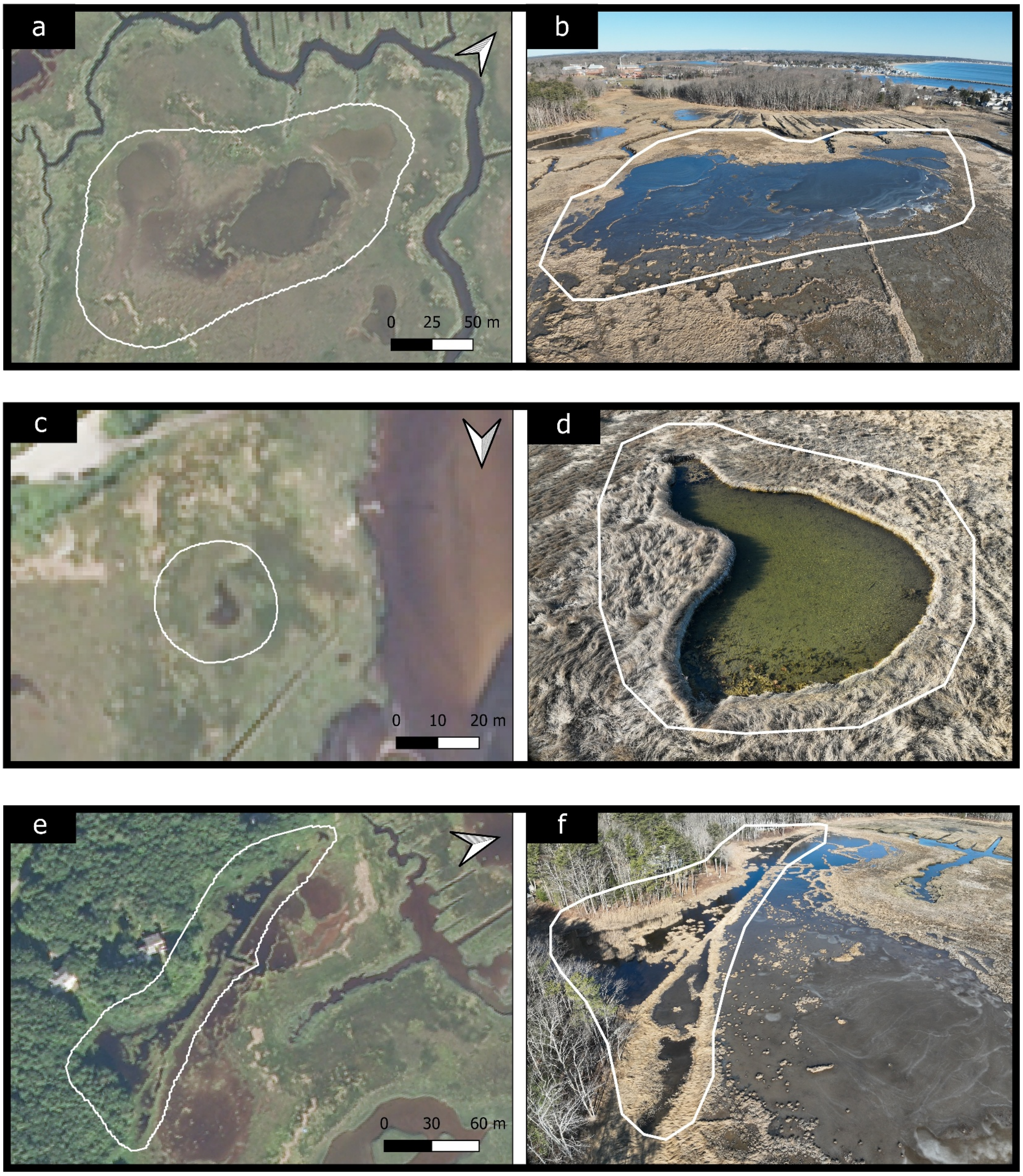

Figure 2. Examples of the three pool types. (a) Mega-pool on the Biddeford Pool marsh from a nadir view; (b) the same mega-pool from an oblique view; (c) individual pool on the Little River marsh from a nadir view; (d) the same individual pool from an oblique view; (e) perimeter pool on the Marshall Point marsh from a nadir view; (f) the same perimeter pool from an oblique view. Nadir images on the left show pools as they appear in 5 July 2009 NAIP imagery; oblique images on the right were collected with a drone on 23 March 2024.

- Individual pool: Defined borders, typically circular and primarily naturally occurring (described by Redfield (1972), Adamowicz and Roman (2005), and McKown et al. (2023)), typically have areas less than hundreds of square meters.
- Perimeter pool: Defined as disjointed borders, long, thin pools noted for formation along embankments on a marsh's border or roads

Stream digitizing was used in conjunction with a One by Wacom tablet in QGIS version 3.28.0 to manually polygonize the extent of salt marsh pools and pool systems (following K. R. Wilson et al. (2009), Watson et al. (2016), Burns, Alber, and Alexander (2021), and Burns, Alexander, and Alber (2021)). While digitizing, pools were given an attribute of either mega-pool, individual pool, or perimeter pool. Mega-pools and perimeter pools were delineated by the extent of their disjointed edges, which in some cases resulted in small areas of vegetation being included within the pool boundaries. However, our goal was to identify the full extent of the marsh surface encompassed by the mega-pools; therefore, the edge of each pool polygon was placed at the outermost extent of standing water associated with each pool. All pools were digitized on a scale of 1:400 to 1:1000 in UTM 19N.

After pools were digitized on all marshes for both years, each pool was attributed a unique identification based on the pool type (i.e., mega, individual, perimeter). Based on this unique identification, we used spatial joins and geoprocessing intersection tools to identify changes in pools categorized as: remained, combined, split, absent in 2021, or formed before 2021. Indexing pools by morphology and change type allowed us to identify pools that remained singular in both years, pools that merged, pools that separated, pools only present in 2009, and pools only present in 2021.

Imagery from 2009 had poor location accuracy, which caused georeferencing errors between digitized pools in 2009 and 2021. While this discrepancy was not prevalent throughout the data set, and offset pools were typically within 2 m of their correct location, it still affected less than 850 small individual pools, each measuring less than 5 m wide. As a result, we were unable to spatially join the two data sets for these pools and track them across our study. To correct this error, all pools with georeferencing errors were identified and manually moved to overlap with the 2021 air photos. We therefore provide two different layers for 2009, one with the corrections to make the layers internally consistent with 2021 and one without the correction to align with the 2009 air photos. The area of the pools is the same, but the position of the pool differs.

2.3. Quantification

After georeferencing errors were corrected, pool size, percent cover, and type were analyzed, and pool areas were calculated. Each marsh area as defined by the Maine Natural Areas Program Current Tidal Marshes layer was calculated. These calculations were used to determine pool cover, type, and changes between 2009 and 2021. Using intersections, we identified which pools overlapped with ditches in the digitized ditch layer from the U.S. Geological Survey and compared changes in pool area based on these intersections (Peck et al., 2024).

2.4. Elevation Analysis

For this analysis, we used the best available digital elevation model (DEM) from the United States Geological Survey (OCM Partners, 2024) derived from 2020 lidar data, downloaded from the NOAA Data Access Viewer with a vertical datum of NAVD88. The DEM has a reported vertical accuracy of 4.21 cm and a horizontal accuracy of 0.37 cm, with 0.7 m spatial resolution. The DEM was processed using custom python scripts to calculate the mean marsh elevation within the marsh polygons provided by the Maine Natural Areas Program Current Tidal Marshes Layer (DeWater et al., 2024). The mean elevation of each pool was calculated by determining the mean elevation of the marsh within the digitized pool polygons using the DEM. Because the DEM was hydro-flattened during processing, the elevation of pools appears flat; therefore, the calculated mean elevation for each pool represents the elevation at which the pool would be flooded by tides (OCM Partners, 2024). Each pool was then categorized into tidal datum bins: Lowest Astronomical Tide (LAT), Mean Lower Low Water (MLLW), Mean Low Water (MLW), Mean High Water (MHW), Mean Higher High Water (MHHW), or Highest Astronomical Tide (HAT). For example, a pool in the MHW bin would be flooded by tides at that level and higher but not by tides less than MHW. The tidal datum values were determined in NAVD88 based on NOAA's reported tidal data for the Portland tide gauge. Differences in tidal datums across our survey region fall within the stated uncertainty of conversion between NAVD88 and tidal datums along with DEM uncertainty, so we rely on the Portland datum conversion to identify regional trends.

2.5. Uncertainty Analysis

To minimize inconsistencies from digitizer bias, only one person digitized all 24,712 pools in this study. All digitized layers were checked by at least two independent reviewers for accuracy of digitization and completeness. Layers were reviewed for potential discrepancies caused by tides and shadows. Any pool covered by a tide or shadowed in 1 year but not the other was removed from the data set. This was infrequent, occurring fewer than 50 times. Because pools are difficult to discretely define, our results have better relative than absolute accuracy.

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Table 1
Marsh Area, Pool Area for 2009 and 2021, and Number of Pools in 2009 and 2021

Name	Marsh area (km^2)	Pool area 2009 (m^2)	Pool area 2021 (m^2)	Number of pools 2009	Number of pools 2021
Scarborough	10.1	1,615,000 ± 9,100	1,900,000 ± 6,700	4,812	7,345
Webhannet River	4.88	689,200 ± 5,400	727,800 ± 3,500	2,719	2,970
Spurwink	1.91	265,800 ± 3,400	302,900 ± 2,400	950	1,064
Marshall Point	1.81	250,700 ± 3,600	273,900 ± 2,500	439	582
York River	1.59	201,100 ± 2,700	232,600 ± 2,000	82	75
Ogunquit and Stephens Harbor	1.31	220,300 ± 2,800	277,800 ± 2,100	566	678
Brave Boat Harbor	1.06	91,700 ± 1,700	115,600 ± 1,300	196	297
Cousins River	0.86	54,200 ± 1,400	71,500 ± 1,000	155	218
Biddeford Pool	0.81	63,300 ± 1,400	75,200 ± 980	85	123
Little River	0.80	186,600 ± 2,800	204,900 ± 1,900	355	507
Goosefare Brook	0.71	68,600 ± 2,200	112,800 ± 1,300	115	213
Gooch's Beach	0.44	41,100 ± 1,400	50,600 ± 870	82	84
Total	27.3	3,748,000 ± 13,200	4,335,000 ± 9,400	10,556	14,156

Note. Pool area is provided as pool area ± uncertainty from Equation 1.

We considered digitizer accuracy (D) and pixel resolution (P) as our primary sources of uncertainty and the only uncertainties we can quantify (Burns, Alber, & Alexander, 2021; Burns, Alexander, & Alber, 2021; Crowell et al., 1991; Fletcher et al., 2003). Other sources of uncertainty could include environmental variables (i.e., lighting, tide stage, rain, etc.), which we note for each image/marsh (Table S2). Seasonal variations in pools are also a source of error but are unable to be quantified due to a lack of data availability (Tables S1–S3). Digitizer accuracy depends on the digitizer's interpretation of varying imagery and their ability to consistently identify pool extents. Digitizer accuracy may vary based on pool type as individual pools, typically smaller than mega-pools, were easier to digitize than larger mega-pools. To quantify this error, digitizer consistency was calculated for each time frame of imagery and for each pool type (mega, individual, and perimeter pool). Three mega, individual, and perimeter pools were digitized 10 times for 2009 and 2021. The standard deviation of area was used to quantify digitizer accuracy by pool type and year. Uncertainty due to pixel (spatial) resolution of the aerial imagery impacted the data set in that it determined the accuracy of pool extent delineation. Only pools greater than the pixel size could be identified and digitized. These two uncertainties can be expressed as a single value of uncertainty (U) since they are random and uncorrelated. This uncertainty value is the square root of the sum of both error sources (Equation 1) (Burns, Alber, & Alexander, 2021; Burns, Alexander, & Alber, 2021; Crowell et al., 1991; Fletcher et al., 2003).

$$U = \pm\sqrt{D^2 + P^2} \tag{1}$$

3. Results

Pool area increased on all 12 marshes from a total of 3,748,000 ± 13,200 m^2 to 4,335,000 ± 9,400 m^2 between 2009 and 2021 at an overall rate of 49,000 $m^2\ a^{-1}$. The number of pools on all marshes increased from 10,566 to 14,156 between 2009 and 2021 at an overall rate of 300 pools a^{-1} (Table 1). The mean pool size (±Uncertainty) across all marshes in 2021 for mega-pools was 6,400 ± 400 m^2; individual pools was 50 ± 27 m^2; and perimeter pools was 300 ± 100 m^2. There is no notable trend in the data between marsh size, marsh location (north vs. south), or type of marsh (back barrier vs. along a tidal river).

Little River marsh had the greatest pool area cover at 23.62% ± 0.01%, followed by Ogunquit marsh and Scarborough marsh at 21.18% ± 0.01% and 18.75% ± 0.01%, respectively, in 2021. The Cousins River marsh had the least pool area cover at 8.34% ± 0.01% in 2021. On all marshes, an increase in pool percent cover was observed between 2009 and 2021 with the average increase being +18% ± 7%. The largest increase in pool percent cover occurred on Goosefare Brook marsh at 6.18% ± 0.01%. Ogunquit marsh had the second greatest

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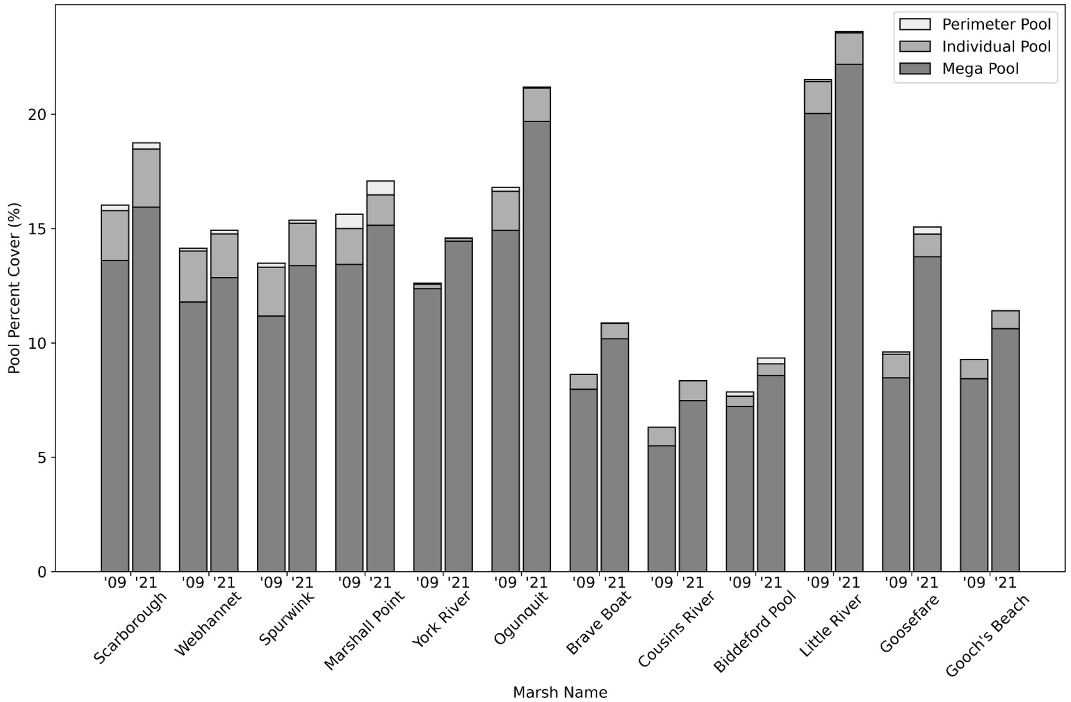

Figure 3. Percent cover of pools (%) on each marsh in 2009 and 2021. Bars are separated into the proportion of mega-pools in dark gray, individual pools in light gray, and perimeter pools in white. Uncertainty error bars represent the thickness of the bar outlines.

change in percent cover at 4.38% ± 0.01%. The smallest increase in pool percent cover was observed on Webhannet marsh at 0.76% ± 0.01% (Figure 3).

Most pool expansion was driven by an increase in mega-pool area cover. Goosefare Brook marsh experienced a 6.01% ± 0.01% increase in mega-pool cover with a −0.032% ± 0.01% change in individual pool cover from 2009 to 2021 and only a 0.21% ± 0.01% increase in perimeter pool cover. On the Biddeford Pool marsh, there was a 1.35% ± 0.01% increase in mega-pool cover with only a 0.06% ± 0.01% increase for both individual pools and perimeter pools. A 2.33% ± 0.01% increase in mega-pool cover occurred on the Scarborough marsh with a 0.36% ± 0.01% and a 0.04% ± 0.01% increase in individual pool and perimeter pool cover, respectively (Figure 3).

On all marshes, there was an increase in mega-pool density (Figure 4). Little River and Ogunquit marshes show the greatest density of mega-pools at 47 mega-pools per km^2 and 31 mega-pools per km^2, respectively, in 2021. The greatest increase in pool density was observed on Goosefare Brook marsh from 18 mega-pools per km^2 in 2009 to 28 mega-pools per km^2 in 2021. The York River marsh experienced a decrease in total pool count per km^2 with a decrease in the number of individual pools but an increase in the number and area cover of mega-pools (Table 1, Figures 3 and 4).

The majority of the observed increase in pool cover occurred when pools combined (Figure 5). Pools that were only present in 2009 or 2021 and pools that remained, were present and remained singular in both years, did not substantially contribute to the increase in area cover. Pools that split from being singular in 2009 to many pools in 2021 experienced an overall decrease in area cover; Biddeford Pool marsh was an exception, however, with a 0.5% increase in area of pools that split (Figure S1 in Supporting Information S1). Goosefare Brook marsh experienced the greatest increase in pool cover (Figure 5), driven by pools combining, which had over a 1,000% increase in area (Figure S1 in Supporting Information S1). Goosefare Brook was followed by the Scarborough and Little River marshes, which had a 600% and 300% increase in the area of combined pools, respectively (Figure S1 in Supporting Information S1). Brave Boat, Goosefare, Ogunquit, and Cousins River marshes experienced the greatest increase in the area of remaining pools at over 20% increases (Figure 5; Figure S1 in Supporting Information S1).

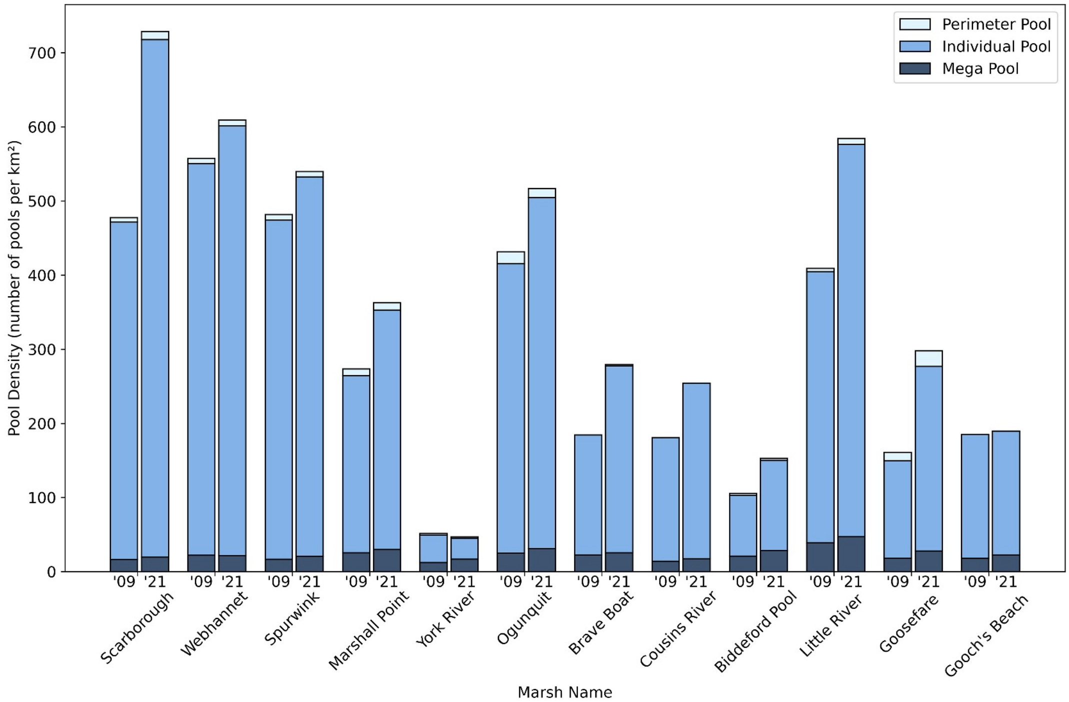

Figure 4. Pool density (number of pools per km^2) on each marsh in 2009 and 2021. Bars are broken into the number of mega-pools, individual pools, and perimeter pools per km^2.

Marsh pools changed in a variety of ways from 2009 to 2021. Mega-pool expansion was prevalent across all marshes, commonly combining with individual pools (Figure 3; Figures 6a–6d). In some instances, individual mega-pools decreased in size from 2009 to 2021 due to shoreline erosion or connection to drainage channels (Figures 6b and 6e). Observed changes in individual pools include being engulfed by mega-pools (Figures 6a–6d) and re-vegetation, though infrequent (Figure 6f). Changes in perimeter pools were infrequent and minimal across all marshes.

All marsh pools occurred above the mean low water elevation but within the highest astronomical tide range, aside from less than 10 pools being above highest astronomical tide across all 12 marshes in 2009 and 2021. The greatest pool expansion was observed in pools between mean higher high water and highest astronomical tide at an increase of 21% ± 1%. Pools between mean high water and mean higher high water expanded at 8% ± 0.6%, and pools from mean low water to mean high water expanded at 13% ± 1%.

4. Discussion

4.1. Widespread Pool Expansion

Salt marshes are essential ecosystems that serve as habitats for many endangered and rare species and protect the coastline from intense storms and flooding. The expansion of marsh pools and subsequent erosion can reduce the marshes' ability to provide these ecological services. Marshes across the Northeast are facing challenging conditions due to the impacts of ditching and agriculture over the last several centuries (Adamowicz et al., 2020) and current rapid sea level rise (Maine Climate Council, 2020; Raposa et al., 2017). Although some evidence suggests that salt marshes can respond to sea level rise by increasing the rate of drainage channel formation and vertical accretion (C. A. Wilson et al., 2014), our results indicate that Maine's marshes are experiencing rapid expansion of mega-pools. This trend of expansion is consistent with observations across the Northeast and could indicate a diminished capacity of marshes to adapt to rising sea level and hinder their ability to provide important ecological functions (McKown et al., 2023; Raposa et al., 2017).

Marsh accretion and ability to adapt to sea level rise depend on various factors such as sediment availability, deposition, vegetation productivity, and nutrient input from surrounding watersheds. The continued expansion of

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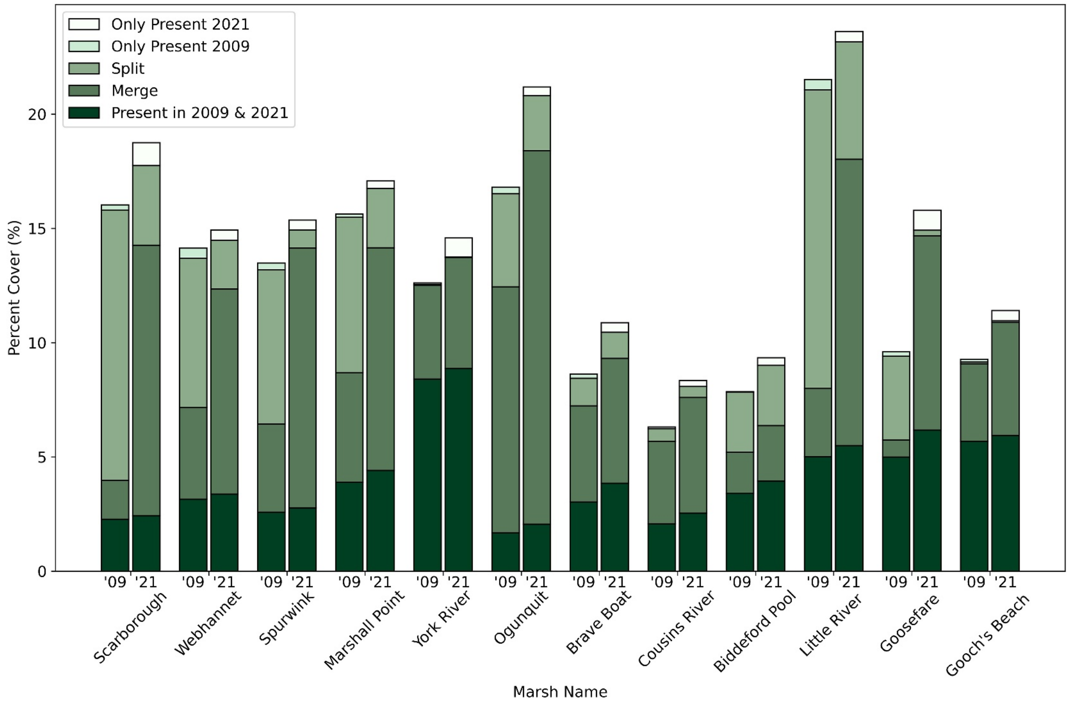

Figure 5. Percent cover (%) of pools categorized by type of change on each marsh in 2009 and 2021. Present in 2009 and 2021 represents pools that existed in 2009 and 2021 and did not split or merge with any other pools. Merge refers to pools that were separate in 2009 but combined into one pool by 2021. Split refers to pools that were one pool in 2009 but divided into more than one pool by 2021. Only Present 2009 refers to pools that existed in 2009 but not in 2021. Only Present 2021 refers to pools that did not exist in 2009 but did exist in 2021 (See K. R. Wilson et al., 2009). 2009 bars represent percent cover of pools as they existed in 2009; for example, merge in 2009 represents separate pools before they combined. 2021 bars represent percent cover of pools as they existed in 2021; for example, merge in 2021 represents singular pools after they combined. Uncertainty error bars are less than the thickness of the bar outlines.

pools may interact with these factors and compound marsh instability by promoting vegetation die-off (Himmelstein et al., 2021; Kirwan & Murray, 2007; Kirwan et al., 2010). As vegetation declines, there is less potential for sediment deposition to build up the marsh surface, and soils become more vulnerable to erosion and peat collapse (DeLaune et al., 1994). This could lead to a feedback loop of pool expansion, erosion, and marsh degradation, further limiting the marsh's ability to accrete and keep pace with rising sea levels (Schepers et al., 2020).

While some mega-pools or parts of mega-pools may connect to drainage channels (Figure 6b), they can remain un-vegetated, and their soft-substrate bottoms remain exposed to tidal erosion (Schepers et al., 2020). We observed continued mega-pool expansion despite connection to drainage on these marshes (Figure 6b). If marsh pools were forming, connecting to drainage, and re-vegetating in dynamic equilibrium as suggested by C. A. Wilson et al. (2014), we would expect to see minimal change in pool cover. A minimal change in pool cover was observed with individual and perimeter pools, and on some marshes, there was a decrease in individual pool cover from 2009 to 2021 (Figure 3). However, with mega-pools, there was significant expansion on all marshes, establishing the mega-pools as the primary contributor to the observed increased pool cover (Figure 3).

K. R. Wilson et al. (2009) digitized a sample of 50 pools on the Webhannet marsh in 1962, 1977, 1991, and 2003. They found that most of the pools decreased in area over the 41-year period but that changes were dynamic, with pools increasing and decreasing in area over time. During that time frame, from 1962 to 2003, the rate of relative sea level rise, calculated from monthly mean sea level records was ~1.8 mm a^{-1} (Fernandez et al., 2020). In contrast, we found an increase in the area of pool cover and the number of mega-pools for all 12 marshes, including the Webhannet marsh, even though our study only covers a 12-year time period. Since 1990, the rate of relative sea level rise observed by the Portland, Maine tide gauge has been 3.05 mm a^{-1}, an increase from 1.8 mm a^{-1} since 1912 (Fernandez et al., 2020). Although data from the Portland tide gauge suggests minimal change in

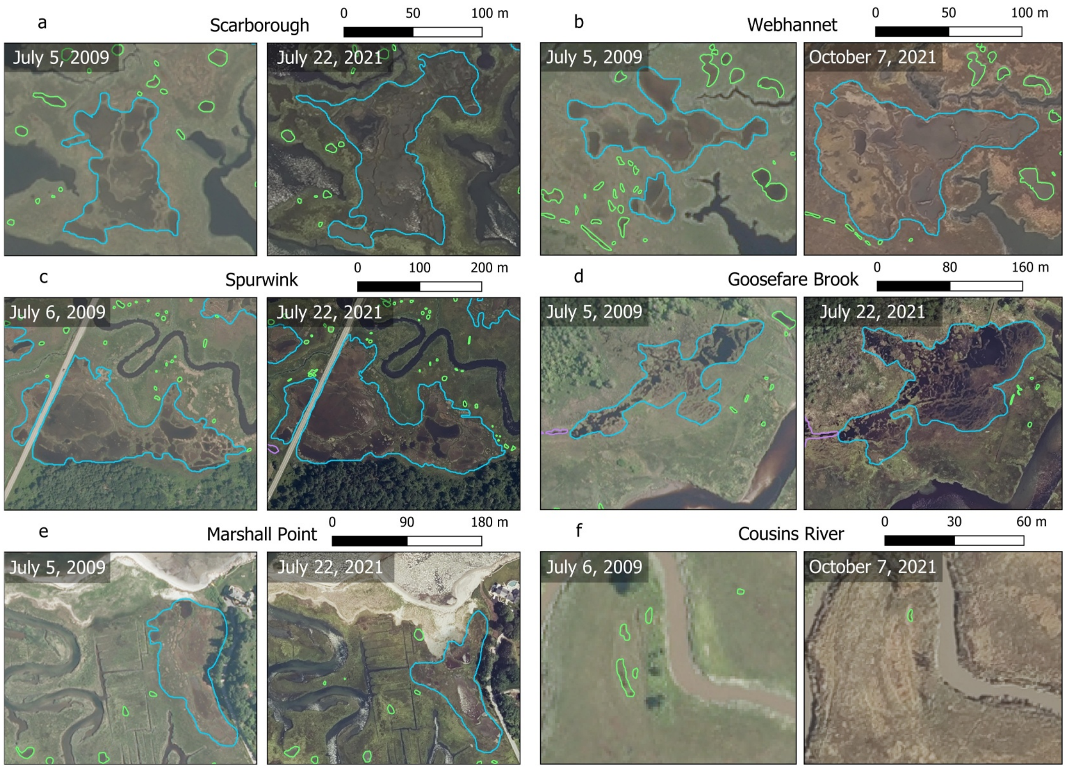

Figure 6. Examples of changes in salt marsh pools from 2009 to 2021. In all maps, the north is oriented upwards. Blue polygons outline the extent of mega-pools, green outlines individual pools, and purple outlines perimeter pools. (a) Scarborough marsh, where a mega-pool expanded and engulfed individual pools. (b) Webhannet marsh, where a portion of the mega-pool was connected to channel drainage but also expanded and engulfed individual pools. (c) Spurwink marsh, where a mega-pool expanded along Sawyer Road and engulfed individual pools. (d) Goosefare Brook marsh, where a mega-pool expanded and engulfed individual pools. Channels in the northeast corner are developing and extending from the main tidal channel toward the mega-pool. (e) Marshall Point marsh, where there was a decrease in the size of a mega-pool due to shoreline overwash. (f) Cousins River marsh, where individual pools re-vegetated.

sea level over the 12-year period of this study, due to its coincidence with the 18.5-year nodal cycle, marshes are likely responding to the decades of sea level rise that occurred prior to this specific period.

There are various ways in which pools can change over time (present in 2009 and 2021, split, merge, only present in 2009, only present in 2021 (Figure 5)). For example, the York River marsh was the only marsh in which we documented a decrease in pool density; however, we still observed an increase in pool area, suggesting pools merged and expanded (Figure 4; Table 1). On the Webhannet marsh, K. R. Wilson et al. (2009) documented 60% of pools in what they refer to as the "life" stage, or pools that remained singular, over four-time frames from 1962 to 2003 but did not quantify changes in the area of pools that correspond to different types of changes. Here, we quantified area changes of pools grouped by the type of change, and we refer to their life stage as present in 2009 and 2021. We found that pools present in 2009 and 2021 (remained singular) did not substantially undergo expansion (Figure 5). While our study only examined changes over a 12-year period, we found that pool merging is by far the main driver of pool expansion, whereas splitting often resulted in a decrease in area cover (Figure 5; Figure S1 in Supporting Information S1). Pools formed due to inundation typically develop in depressions on the marsh from past agricultural practices or because they are blocked by a tidal restriction. In these depressions or above tidal restrictions, water collects and inundates the surrounding vegetation, leading to die-off and erosion. Once one pool forms, due to this inundation, it is likely that other pools will form and combine as the inundation continues. Pool splitting suggests deposition of sediment within the pools, thereby increasing the elevation in the pool and lowering the extent of inundation. The formation of new pools was not a notable contributor to the increase in pool area because formation was infrequent as most depressions or areas above tidal restrictions on the marsh surfaces already had pools present. While pools only existed in 2009 and pools splitting led to a decrease in pool area, the decrease was not sufficient to balance the expansion caused by pools merging.

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Northeast marshes have a longstanding history of agricultural use and ditching, which have interfered with marsh hydrology patterns (Sebold, 1998; D. C. Smith & Bridges, 1982; D. C. Smith et al., 1989). Agricultural embankments on the marshes can prevent adequate drainage and ditches can cause subsidence, promoting the continued expansion of mega-pools as vegetation dieback occurs from inundation (Adamowicz et al., 2020; Burdick et al., 2020). We observed the presence and formation of perimeter pools along marsh borders adjacent to embankments, occurring infrequently on 10 of the 12 marshes and not at all on Cousins River and Gooch's Beach marshes. It is difficult to quantify the extent to which these underlying ditches and embankments impact the development of mega-pools due to the limited documentation of agriculture in Maine marshes that occurred since the 19th century (Burdick et al., 2020; D. C. Smith & Bridges, 1982; D. C. Smith et al., 1989). However, based on our analysis quantifying pools that intersect with the USGS marsh ditches layer (Peck et al., 2024), we found a greater rate of expansion of pools that coincide with ditches. Of pools that coincide with ditches, area cover increased by 30% ± 1%, whereas pools that do not coincide with ditches only increased in total area cover by 8% ± 0.5%. Though we find greater expansion with pools intersecting ditches, this could be explained by ditches preferentially occurring in higher marsh areas, which coincides with mega-pool development.

Mega-pool and perimeter pools were frequently observed forming along roads and railroads, some of which act as tidal restrictions by intersecting channels. Pool formation along roads occurred notably along Sawyer Rd in the Spurwink Marsh (Figure 6c; Figure S2a in Supporting Information S1) and Harbor Rd (previously named Lower Landing Rd) in the Webhannet marsh (Figure S2b in Supporting Information S1) (See Figure S2 in Supporting Information S1 for more examples of pool expansion along roads). Pool formation along embankments was documented along Harbor Rd on the Webhannet marsh from 1962 to 2003 (K. R. Wilson et al., 2009). We identified the pools reported by K. R. Wilson et al. (2009) along Harbor Rd and categorized them as mega-pools which expanded from 2009 to 2021. On the Spurwink marsh, there are plans to remove Sawyer Rd in the coming years because it is frequently flooded by high tides. Removing Sawyer Rd could restore the hydrology of the marsh similar to restoration strategies for correcting drainage (McKown et al., 2023) and allow the mega-pool to drain and re-vegetate. The response of the marsh to road removal should be monitored closely to determine if road and tidal restriction removal could be a potential strategy in improving marsh resilience through restored marsh hydrology.

To understand the impact of elevation on pool size and distribution, we examined the elevation of pools compared with overall marsh elevation. The Goosefare Brook marsh, which by far, had the greatest increase in pool percent cover at 64%, was the only marsh where the mean pool elevation was lower than the mean marsh elevation. Increased exposure to rising sea levels over the past decades likely contributes to the formation and expansion of pools on the marsh, as demonstrated by Goosefare Brook, in which the average pool is lower than the mean marsh elevation and is therefore more exposed to erosive forces from flooding. This may suggest that pool development is impacted by the frequency of inundation and flooding because pools at lower elevations are more often exposed to tides and rising sea levels.

We further found a moderate negative correlation ($R^2 = -0.2$) between pool elevation and mean mega-pool size, indicating that larger mega-pools tended to develop at lower elevations, whereas smaller mega-pools occurred at higher marsh elevations. An opposite trend was observed for individual pools in which there was a moderate positive correlation ($R^2 = 0.4$) between pool elevation and mean individual pool size, suggesting that higher elevations typically have larger individual pools, while lower elevations have smaller individual pools. Similarly, perimeter pools were found to be larger and more frequent at higher marsh elevations. Pools within the tide range of mean higher high water to highest astronomical tide experienced the greatest rate of expansion compared to pools between mean low water and mean high water, as well as pools between mean high water and mean higher high water. With previous sea level rise, lower elevations experienced more frequent and longer duration periods of inundation, where we found mega-pools to be larger in area and are likely still adjusting to higher sea levels. The greater expansion rate of pools in higher tide ranges may further indicate impacts of human modifications of the marsh but is in part confounded by elevation. Historic ditches, which are typically found in higher marshes, can cause elevation depressions, and we found a greater rate of expansion of pools intersecting these ditches, similar to the greater expansion of mega-pools found in higher tide ranges.

4.2. Sediment Erosion

Previously, marsh accretion rates have been compared to rates of sea level rise to predict marsh survival in Maine (Goodman et al., 2007); however, these measures can be difficult to apply broadly to marshes over regions and

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time (Yellen et al., 2022). The health and resilience of marshes can be indicated across a greater spatial and temporal range by quantifying the extent of pool expansion and recovery. On marshes with greater pool cover and infrequent pool recovery, biogeochemical indices of health such as biomass, soil shear strength, and porewater are lower than those on marshes with greater vegetation coverage (Himmelstein et al., 2021). Sediment export due to erosion is greater on marshes with greater pool coverage and unvegetated soils (Ganju et al., 2013, 2020) because of the positive feedback driven by soil exposure to wind wave-driven erosion (Himmelstein et al., 2021; Schepers et al., 2020). If the rate of sea level rise exceeds accretion in the pool, it will not recover and instead, will continue to expand, forming mega-pools (Mariotti, 2016). Our results indicate that the feedback loop of erosion, particularly of mega-pools eroding without recovery, occurs on all the marshes we surveyed (Figure 3; Figure S3 in Supporting Information S1). Based on conclusions from previous studies, the continued pool expansion documented here further suggests a potential increase in sediment export from the marshes in this study because unvegetated soils are more vulnerable to wind and wave erosion (Ganju et al., 2013, 2020; Himmelstein et al., 2021). With more unvegetated areas exposed in mega-pools, marshes could become more susceptible to increased erosion and sediment loss. However, the potential increase in sediment export requires further study, as it is influenced by factors other than pool expansion, such as marsh sediment supply, and could have implications for marsh stability and coastal processes.

4.3. Restoration

While pools naturally occur (Redfield, 1972; K. R. Wilson et al., 2009), they also form due to hydrological interference from agriculture (Adamowicz et al., 2020; Burdick et al., 2020); our study has found that mega-pools which typically form on depressions resulting from agriculture have expanded since 2009 in part due to sea level rise and lasting effects of human activity on the marshes. Therefore, the results of our study support the need for restoration efforts with the intent to drain mega-pools and support re-vegetation. One restoration strategy could be implementing runnelling, which has been shown to be effective in draining mega-pools by reconnecting them to tidal channels with shallow ditches, allowing for re-vegetation (McKown et al., 2023), though runnelling is not a long-term solution for marsh resilience to sea level rise (Besterman et al., 2022). While in some cases, we observed a lack of re-vegetation following connection to drainage of some pools (Figure 6b), this does not suggest that runnelling will be ineffective because the mechanism by which the pools have been connected to drainage and the depth of drainage channels are different. Given recent and upcoming efforts of marsh restoration, it is important to document the changes of marsh pools to inform these and future efforts and quantify their efficacy as strategies to limit pool expansion. We need to reverse recent inundation trends to support resilient and healthy marshes to buffer storms, sequester carbon, and provide habitat for rare and endangered species. Future studies should focus on understanding the efficacy of these restoration methods and monitoring for any unintended consequences.

5. Conclusions

Salt marshes provide essential habitat for many endangered and threatened species, including birds. Marshes also act as natural buffers for coastal infrastructure, reducing the impact of storm surges and high tides. However, the expansion of pools, or standing water, on the marsh surface can diminish their capacity to provide these ecological benefits. We found an increase in pool area from 2009 to 2021 across all 12 marshes in this study, consistent with trends observed throughout the Northeastern US region. There was an increase in pool count per km^2 on all marshes except for the York River marsh, which still experienced an increase in area cover. Individual pool and perimeter pool area and count remained about the same, whereas increases in count and area were driven by mega-pool expansion. Mega-pools were larger on lower marsh elevations, whereas individual pools were larger on higher marsh elevations, suggesting that sea level rise may be prompting mega-pool expansion. Of the various ways pools can change over time, pools merging was the primary contributor of expansion, whereas pools that were present in 2009 and 2021 (remained singular), split into many pools, only existed in 2009, or only existed in 2021 contributed minimally to expansion and even decreased pool area. Goosefare Brook and Ogunquit River marshes experienced the greatest increase in pool cover, while the Webhannet marsh experienced the smallest increase over the 12 years. The Little River, Ogunquit, and Scarborough marshes have the greatest pool percent cover. Primary drivers of pool expansion could include rising sea levels and historic agricultural ditching, which continues to interfere with underlying marsh hydrology. Therefore, our results support the advocacy and need for marsh restoration projects to promote drainage and re-vegetation of mega-pools. In the coming decades, sea level

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rise will significantly impact salt marshes worldwide and continued monitoring efforts are needed to ensure that marshes continue to provide numerous critical ecosystem services. Our work suggests that intervention is needed to reverse the expansion of pools and maintain marsh health.

Conflict of Interest

The authors declare no conflicts of interest relevant to this study.

Data Availability Statement

Data layers of pool polygons and python scripts have been made available at DUNE: DigitalUNE https://dune.une.edu/gis_data/1 (DeWater et al., 2024).

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