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Research article

Distribution and disturbances of ditches across salt marshes of the Northeast U.S. with implications for management and restoration

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ABSTRACT

Effective management of valuable coastal systems, such as salt marshes requires an understanding of the complex stressors influencing their continued threat of drowning. However, efforts to determine the effects of one potential stressor, ditches, have produced diverging results complicating management efforts. Ditches (linear trenches dug to drain salt marshes for agriculture and mosquito control) alter salt marsh hydrology, but their effects on widescale marsh function and degradation are poorly understood. We created a dataset of visible ditches and summarized ditch densities (length of ditches over area) for salt marshes of the Northeast U.S. to evaluate ditching against vulnerability metrics, including elevation and the unvegetated to vegetated marsh ratio (UVVR). We identified a scale dependency in which the larger/coarser the spatial scale of analysis, the greater the fraction of ditched salt marshes. Scale dependence explains discrepancies between previously determined ditch indices. In terms of effects on marsh vulnerability, relative elevation was not influenced by visible ditch presence. Ditch densities affected UVVR, exhibiting a multiple threshold behavior. When present at low densities, ditches have little effect on ponding; yet as ditch densities increase, UVVR (i.e., ponding) increases. The relationship between ditching and UVVR reverses at the highest ditch densities, with ponding substantially decreasing. The multiple threshold vulnerability response of Northeast salt marshes to the hydrologic influences imposed by ditching suggests restoration strategies should consider the degree of ditching rather than simply ditching presence.

1. Introduction

Climate change and ongoing coastal land alteration are degrading salt marshes globally (Fagherazzi et al., 2020). Salt marshes of the Northeastern U.S. (Maine to Virginia; Fig. 1a) are especially vulnerable to loss given their history of intensive anthropogenic alteration. Estimates of salt marsh loss over the last two centuries are as high as 50% in at least one Northeast state (RI; Bromberg and Bertness, 2005), and

Abbreviations: UVVR, Unvegetated to Vegetated Marsh Ratio.
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marshes that remain are subject to increasing pressures, including direct modifications (e.g., ditching, tidal restrictions, filling) and larger-scale stressors (e.g., relative sea level rise, invasive species, pollution; (Bertness et al., 2014; Chambers et al., 1999; Deegan et al., 2012; Kennish, 2001; Kirwan et al., 2010). Given the variety of stressors and diversity of physical drivers (i.e., tidal regimes, sediment supplies, and storm exposures; Wiberg et al., 2020), Northeast salt marshes can serve as an ideal region to provide management guidelines applicable to coastlines globally with the aim of preserving intact salt marshes and restoring those that are degraded.

One direct human modification to salt marshes – ditching – was commonly practiced across the Northeast to improve yields for salt hay farming (*Spartina patens*, *Juncus gerardii*, and *Distichlis spicata*) during European colonization (Rozsa, 1995) and to drain standing water for mosquito control in the first half of the 20th century (Crain et al., 2009; Daiber, 1986). Ditches, especially those created for mosquito control, are regularly spaced, narrow, linear channels and are organized into patterns including parallel, grid or checkerboard, and herringbone (dug at acute angles to a main channel stem) that are easily distinguishable from wider, sinuous, multi-ordered tidal creeks (Tonjes, 2013). Agricultural ditches, while still numerous on the marsh platform, can be difficult to distinguish on the landscape because many have aged and infilled, while the more recently installed mosquito ditches may be more apparent and often overlay prior agricultural alterations (Adamowicz et al., 2020).

Hydrological changes caused by ditching may also alter the morphology of salt marshes given that both ditch types aim to increase drainage of the marsh platform. However, previous work to assess the effects of ditching on salt marsh morphodynamics have returned mixed results (Aerni et al., 2023; Smith et al., 2021; Tonjes, 2013). Some studies observed that ditches increase drainage, which may result in lowered water tables, increased sediment oxygenation and organic matter decomposition, and sediment compaction, ultimately resulting in elevation loss and increased ponding (Bourn and Cottam, 1950; Brain, 2016; Lesser, 1982; Smith et al., 2021). Others found that ditch levees (formed from dredge spoil during ditch construction) can restrict the delivery of sediment to the interior of the marsh during tidal inundation while simultaneously preventing normal drainage during ebb tide

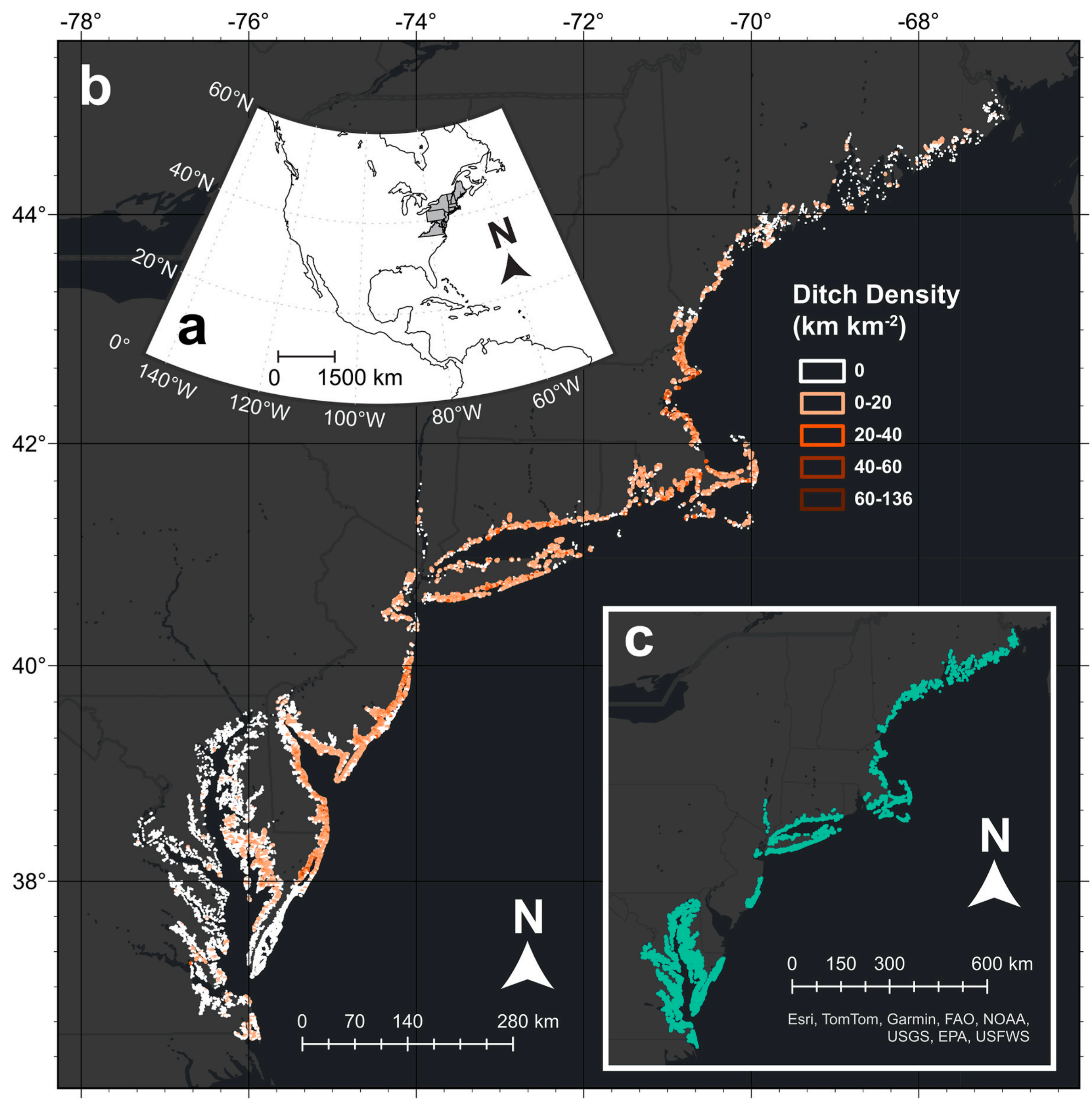

Fig. 1. a: Inset map with Northeastern U.S. study region in gray. b: Map of visible ditch densities within 165-m salt marsh pixels for the Northeast. c: Inset map of marsh units in the Northeast region used for this study.

(Kennish, 2001; Raposa et al., 2017). The resulting waterlogging and salinization through evaporation leads to vegetation stress and eventual root collapse (DeLaune et al., 1994; Miller and Egler, 1950). Nonetheless, many studies of Northeast salt marshes found little or limited influence of ditches on elevation loss and ponding when compared to apparently unditched sites (Redfield, 1972; Vincent et al., 2013), and even observed reductions in salt marsh pool size and density as a result of ditching (Adamowicz and Roman, 2005; Lathrop Jr and Bognar, 2001; Lathrop et al., 2000). It therefore remains unclear if the presence of ditches contributes to widespread elevation loss, ponding, or other degradation of vegetated salt marsh platforms.

Cross-system and regional-scale comparisons to address this question across the Northeast have traditionally been limited by a lack of readily available data on ditch presence. Bourn and Cottam (1950) estimated that roughly 90% of salt marshes from Maine to Virginia were ditched by 1938, and since then important strides have been made to better identify and quantify patterns of ditch presence in the Northeast (Adamowicz and Roman, 2005; Aerni et al., 2023; Correll et al., 2017; Crain et al., 2009; McGarigal et al., 2017; Smith et al., 2021). Still, accurate maps of ditches and quantitative information on ditch distribution for all Northeast salt marshes are needed both for researchers to assess the effects of ditches on coastal morphodynamics and vulnerability, and for land managers in developing restoration strategies.

Restoration of ditches has been proposed as a tool to improve salt marsh condition (Adamowicz et al., 2020; Burdick et al., 2020), but clear evidence of the effects of ditches on salt marsh vulnerability is needed to better understand the extent and dynamics of salt marsh alteration to guide effective methods for mitigating impacts and improving salt marsh health. The high prevalence of ditches across the coastal Northeast makes restoration of all ditches unrealistic, therefore, identifying conditions under which remediation of these systems is feasible and likely to succeed is crucial to maximize the effective use of resources. The objective of our study was to develop a comprehensive map using standardized methods of ditches visible in modern aerial imagery for salt marshes from Maine to Virginia in the Northeastern U.S. and use this map to quantify ditch densities within the region. We used this dataset to analyze and understand scale dependency on the patterns of ditch occurrence and the influence of ditches on salt marsh vulnerability. In addition to highlighting the importance of geospatial scale of analysis, we assessed the relationship between the presence/absence of visible ditches and ditch densities with two continuous and commonly used metrics of salt marsh vulnerability: elevation and unvegetated to vegetated marsh ratio, hereafter UVVR. Elevation is an indicator of salt marsh position relative to sea level, with higher elevations inferring greater resilience to drowning (Raposa et al., 2016; Reed, 1995), while UVVR is a useful vulnerability metric due to its wide geographic extent and adoption as a proxy for salt marsh health (i.e., vulnerability to drowning; Ganju et al., 2022). Collectively, our study provides a foundational scientific assessment of ditch density, distribution, and ecological impact necessary to guide salt marsh restoration decisions, and we outline potential management actions in response to various intensities of ditching.

2. Methods

2.1. Ditch digitization & densities

Ditches visible on the World Imagery Basemap (0.3 m resolution supplied by DigitalGlobe) were hand digitized in ESRI ArcGIS Pro Version 3.1.3 at a scale of 1:2500 within the bounds of the U.S. Fish and Wildlife Service National Wetland Inventory's (NWI) Estuarine Intertidal Emergent Wetland class (E2EM; U. S. Fish and Wildlife Service. Publication date, 2019) cropped to the state borders of Maine through Virginia (Peck and Walker et al., 2024, Fig. 1b for footprint). Hand digitizing ditches is limited by the size of the ditches relative to the spatial resolution of the aerial imagery (0.3-m). We recognize that we were only able to digitize ditches that were visible in the 2023 World Imagery Basemap and that there may be not-visible, buried ditches (i.e., zombie ditches) affecting hydrology and also that the hydrology of visible ditches may have altered since their original construction due to lack of maintenance (discussed in detail in the discussion section). Precision of hand digitizing was assessed by comparing total ditch length determined by seven separate independent digitizers of the same 0.8 km^2 marsh area within the Barnstable, MA salt marsh; however, all marsh areas had 2–4 digitizers check that the digitized ditch polylines were complete.

Ditch densities (km km^{-2}) were calculated as the summed geodesic length of ditches within a wetland area divided by the geodesic wetland area (NAD83 Albers geographic coordinate system was used for all area and length measurements). First, we calculated ditch densities using the NWI E2EM polygon areas to compare the quality of our hand digitized ditch dataset against a previously calculated ditch metric created by McGarigal et al. (2017), who performed image analysis of 1-m resolution lidar DEMs for 64% of the Northeast salt marshes. We compared ditch densities between the two datasets when both methods identified at least one ditch within NWI E2EM polygons by calculating a linear best fit between the datasets and a total difference in summed ditch length.

To assess the effects of salt marsh area on estimates of ditch presence and density, NWI E2EM polygons were converted to rasters for a standardized set of varying pixel sizes (30-, 165-, 300-, 1650-, and 3000-m), which were converted back to square polygons. The larger pixel sizes were selected for comparison to previous studies, which tended to be completed at a much coarser scale. Ditch densities were also calculated within conceptual marsh units, when available (Maine; Ackerman et al., 2024; Massachusetts; Ackerman et al., 2021; Connecticut; Ackerman et al., 2023a, New York; Defne et al., 2024, Edwin B. Forsythe National Wildlife Refuge (NWR) in New Jersey Defne and Ganju, 2016; Chincoteague Bay in Maryland and Virginia; Defne and Ganju, 2018a,b; Eastern Shore of Virginia; Ackerman et al., 2023b, and Chesapeake Bay in Maryland and Virginia; Ackerman et al., 2022) (Fig. 1c for footprint). Conceptual marsh units, which are based on the NWI E2EM footprint, were calculated as the area within high-elevation ridge lines separating surface flow from other tidal basins based on flow accumulation and surface slopes modeled using relative elevations at 1-m resolution digital elevation maps (median area = 0.027 ± 0.059 km^2 or 115 ± 240-m resolution; Fig. S1; see Defne and Ganju (2018a,b) for details).

2.2. Quantifying the effects of ditches on salt marsh vulnerability

We used an existing map with a spatial resolution of 3-m developed by Correll et al. (2019) to determine the percentage of vegetation community type (high marsh, low marsh, salt pools/pannes, terrestrial border, *Phragmites australis*, mudflat, open water, and upland) for each marsh unit. This map extended from Maine to Virginia but did not include salt marsh for much of the northern reaches and western shore of the Chesapeake. Terrestrial and upland vegetation can be found in marsh polygons for a number of reasons, including mismatch in scale and that these systems are spatially dynamic, and may have transitioned between vegetation communities since the classification of the NWI polygons. Elevation, tidal range, and UVVR were also reported with the marsh units from the above sources. We calculated a de-meaned, de-tided marsh unit elevation (here after, referred to as normalized elevation) by first subtracting the regional (i.e., Maine, Massachusetts, Connecticut, New York, Edwin B. Forsythe National Wildlife Refuge, Chincoteague Bay, Eastern Shore, and Chesapeake Bay) mean marsh elevation and then dividing by the system-wide mean tidal range (sensu Ganju et al., 2020). Marsh unit UVVR values are aggregates of unvegetated and vegetated pixels within the defined marsh unit area (see Ganju et al., 2022, 2020). We restricted our analysis to UVVR values between 0 and 2, as UVVRs above 2 (i.e., 67% unvegetated) indicate that the majority of the marsh unit area is functioning as open water, rather than salt marsh (Ganju et al., 2020).

To test whether visible ditch presence affects salt marsh vulnerability, normalized elevation and UVVR distributions were compared across marsh units with and without visible ditches. To account for potential differences in the vulnerability metrics across high and low marshes, we classified marsh units into high marsh and low marsh based on proportion, and then used Kruskal-Wallis tests to assess significant differences between normalized elevation and UVVR for marsh units with and without ditches.

We also assessed how marsh unit normalized elevation and UVVR changed with increasing visible ditch densities. Curve fits were iteratively tested for each data series on raw and median data, and exponential fits ($y = a\,e^{bx}$) were selected. To better investigate a relationship between increasing visible ditching intensity and marsh unit UVVR, marsh units were characterized into three different categories of salt marsh quality: (1) UVVR >0.15, (2) UVVR = 0.1–0.15, and (3) UVVR <0.1. The percent of marsh unit area within each of these categories of salt marsh quality was determined for salt marshes lacking visible ditches to demonstrate the expected range of salt marsh conditions under little to no influence from ditches. The percent of marsh unit area within each of these categories of salt marsh quality was determined for salt marshes lacking visible ditches to demonstrate the expected range of salt marsh conditions under little to no influence from ditches (i.e., the vulnerability is controlled by other stressors). The remaining salt marshes were binned by ditch densities in increments of 5-km km^{-2} and percent salt marsh area was calculated for each quality category, as well as the total salt marsh area. Assessment of percent area trends in salt marsh quality with increasing ditch density in comparison to the distribution of salt marsh quality under no visible ditches helped us identify a multiple threshold effect of ditching on salt marsh ponding.

3. Results

3.1. Ditch digitization

Within the total 3,564 km^2 of NWI E2EM salt marsh assessed, we digitized 14,239 km of ditches. This dataset represents more assessed salt marsh area than other previously published datasets for the Northeast (Correll et al. (2017) mapped ditches for about 25 km^2 (0.7% of total NWI area), Smith et al. (2021) mapped ditches for about 236 km^2 (6%), Aerni et al. (2023) mapped ditches for about 1550 km^2 (43%), McGarigal et al. (2017) mapped ditches for about 2281 km^2 (64%)).

Precision of hand digitizing was assessed based on seven separate independent digitizations of the same 0.8 km^2 Barnstable polygon, where the sum of ditch length ranged from 13 to 15 km, with a mean of 15 ± 1 km, representing a 13% standard error. We interpret this 13% error as a conservative estimate of error precision, as all polygons had 2–4 digitizers check that the digitized ditch polylines were complete. We compared our dataset with a geographically similar, available ditch polyline dataset provided by McGarigal et al. (2017), who used image classification of lidar DEMs to identify ditches >75 m in length. When conservatively comparing only regions where both datasets had ditches, we identified >2.5x the total length of ditches. As such, our mapped ditch densities tended to be higher than those of McGarigal et al. (2017) for the same regions (Fig. S2a). McGarigal et al. (2017) only assessed ditches within ~64% of Northeast NWI E2EM polygons and frequently their algorithm mistakenly classified sinuous tidal channels as ditches (e.g., Fig. S2b).

3.2. Distribution of ditch densities and the effect of scale

Visible ditch presence and densities were highest in New England states, except Maine, though densities were also high between New York and Maryland (Fig. 1b). Lowest densities were observed along the Eastern Shore of Virginia and much of the Chesapeake Bay. Overall, the majority of marsh units were classified as not ditched, likely an artifact of the disproportionate area of salt marsh in the Chesapeake where ditching densities were often at or near 0 km km^{-2}. Vegetation community distributions, which were not available for much of the northern reaches and western shore of the Chesapeake (Fig. S3), indicated that visible ditches were generally more present in high marshes compared to low marshes, though no clear trend was observable in habitat type with increasing visible ditch densities (Fig. 2a).

When assessed over standardized salt marsh areas, the fraction of visibly ditched salt marshes ranged from 7.3% at 30-m resolution (900 m^2 salt marsh) to 64% at 3000-m resolution (9 km^2 salt marsh), following an increasing logarithmic trend (Fig. 3). Visible ditch presence was observed in 15% of the available marsh units from Maine to Virginia provided by Defne and Ganju (2018a,b), which was less than expected based on the median area of these units (115-m resolution). However, this estimate does not include marsh units from New Hampshire, Rhode Island, parts of New Jersey, and the Delaware Bay—regions known for intensive ditching (Aerni et al., 2023). Including these areas would likely increase the overall estimate of visible ditch presence. Our visible ditch presence results tended to be lower than those previously estimated (Aerni et al., 2023; Bourn and Cottam, 1950; Correll et al., 2017; Crain et al., 2009; Smith et al., 2021). Median visible ditch densities declined with greater standardized salt marsh areas (Fig. S4). Fewer literature values of ditch density are available, and none span the entire Northeast (Adamowicz and Roman, 2005; Corkran, 1938; Corman, 2009).

3.3. Ditches and vulnerability

Comparison of marsh units with and without visible ditch presence reveals little observable difference in normalized marsh unit elevations (Fig. 4a; $p < 0.001$, df = 48,725) or marsh unit UVVR (Fig. 4b; $p = 0.06$, df = 47,667). Similarly, when these vulnerability metrics are tested for marsh units with a greater proportion of high marsh (Figs. S5a–b) or low marsh (Figs. S5c–d), there is little observable difference between salt marshes with and without visible ditch presence (high > low marsh: elevation: $p < 0.001$, df = 16,384, UVVR: $p < 0.001$, df = 15,954; high < low marsh: elevation: $p = 0.7$, df = 3,672, UVVR: $p = 0.05$, df = 3569). Though many of these comparisons produced statistically significant results, likely as a result of large sample sizes, the observable differences are not obvious enough to conclude an effect of visible ditch presence on salt marsh vulnerability. Similarly, marsh unit normalized elevation showed no obvious trend with increasing visible ditch densities (Fig. 5a); however, marsh unit UVVR values decreased significantly with increasing ditch density, especially when assessed across binned data to account for differences in sample size across the full range of ditch densities (Fig. 5b, $R^2 = 0.56$, df = 18, sse = 0.05, rmse = 0.016).

When no visible ditches were present in marsh units, 41%, 16%, and 43% of salt marsh area fell into the UVVR quality categories of >0.15, 0.1–0.15, and <0.1, respectively (Fig. 6a). As visible ditch densities increased, little change in this distribution of UVVR quality categories is observable until ~30 km km^{-2} ditch density, at which point the fraction of salt marsh area with extensive ponding begins to increase, peaking at a value of 75% at a ditch density of between 45 and 50 km km^{-2}. At visible ditch densities higher than ~55 km km^{-2}, the fraction of marsh units with extensive ponding sharply decreases to 0% at the highest ditch density of 65 km km^{-2}, and the fraction of marsh units with little to no ponding (i.e., UVVR <0.1) increases to 100%, though the total salt marsh area with this extremity of ditching is minor (~0.5 km^2). This thresholding effect is observable in the aerial photography from regions with no visible ditches to those with a moderate, high, and extreme density of visible ditches (Fig. 6b–e).

4. Discussion

4.1. Where are ditches found?

Visible ditch presence in salt marshes varies across the Northeastern

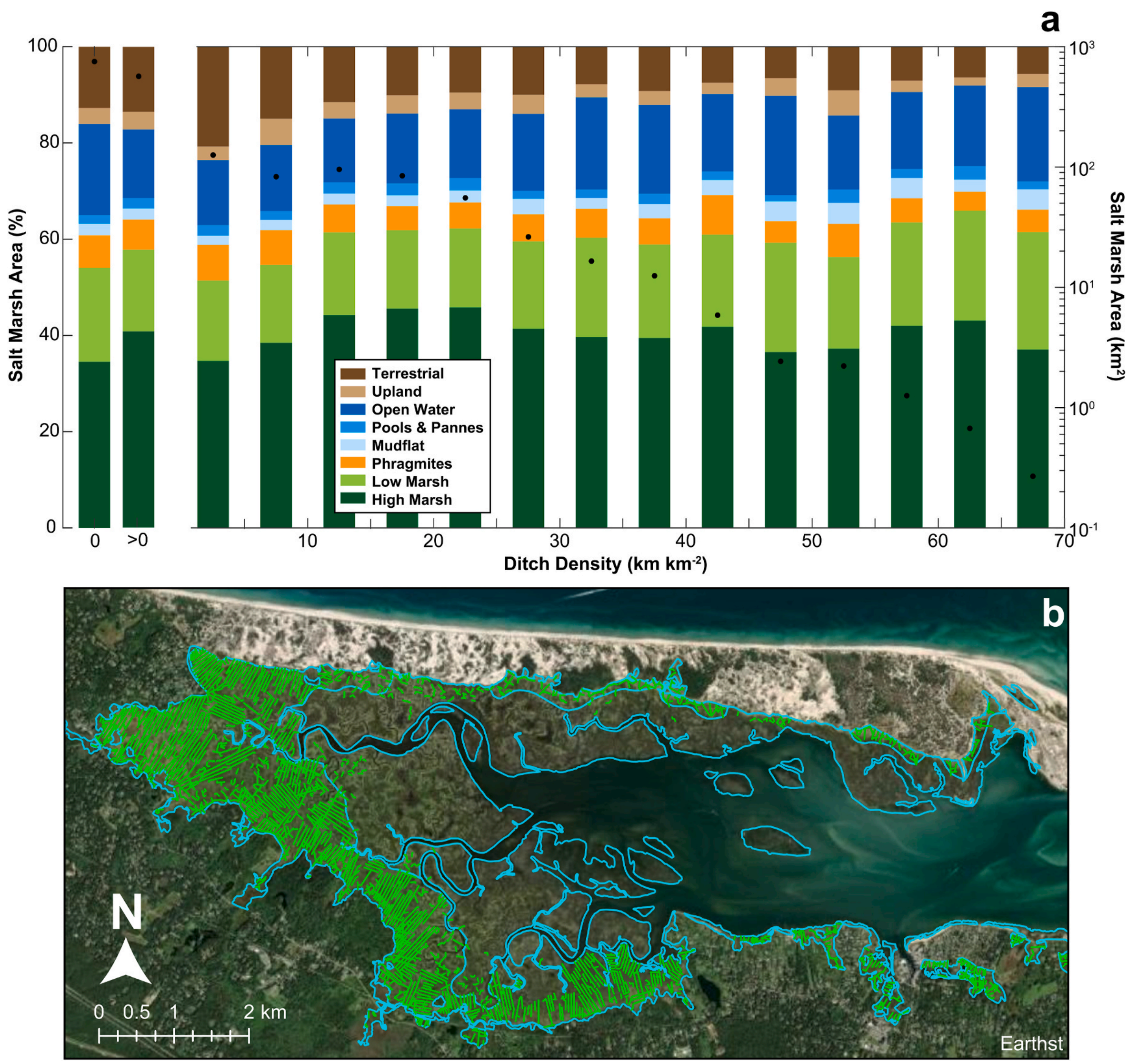

Fig. 2. a: Distribution of tidal marsh vegetation communities (derived from Correll et al., 2019) within marshes of differing ditch densities (binned by 5 km km^{-2}). Total salt marsh area (right log y axis) in each bin is plotted as black dots. b: Aerial photograph example of Barnstable, MA salt marsh (cyan polygon) where visible ditches (green lines) tend to be clustered in more landward portions of the marsh.

U.S. with highest densities found in New England (with the exception of Maine), and lowest densities around the Chesapeake Bay and Eastern Shore of Virginia (Fig. 1b), as others have previously observed (Aerni et al., 2023; Correll et al., 2017; Smith et al., 2021). Visible ditches also tend to be more present in salt marshes with a higher proportion of high marsh to low marsh than those without visible ditches, as well as in salt marshes with more terrestrial/upland vegetation (Fig. 2a). We offer three possible explanations: (1) ditching could change the elevation of the marsh platform through its effects on hydrology; (2) ditching may result in a shift in vegetation species without a change in elevation; and/or (3) ditches may be more often installed in high marsh than low marsh because higher elevations would be easier to drain. We test the effects of ditching on elevation (first hypothesis) below; however, we expect the opposite trend where ditches should decrease marsh elevation as others have also observed (e.g., Smith et al., 2021). While the scale of our analysis does not allow us to assess ditch effects on the distribution of specific vegetation species, previous studies have noted that ditches can alter salt marsh vegetation, favoring high marsh grasses/forbs (Bourn and Cottam, 1950; Vincent et al., 2014). Alternately, ditches may have been more often constructed in higher elevation, landward portions of salt marsh (e.g., Fig. 2b) that would naturally be composed of less salt-tolerant species. In the latter case, it does seem likely that both mosquito control efforts and the creation of farmable/grazable land would focus on regions of the salt marsh closest to human development that would naturally be higher elevation marsh. Future studies should seek to test this hypothesis specifically.

4.2. How many salt marshes are ditched?

The evaluation of salt marsh ecological and morphodynamic features is often complicated by varying methodologies and metrics used by different researchers. In addition to different detection methods and metrics, authors rarely assess salt marshes using the same area or footprint. In the case of presence/absence studies, the area of marsh surveyed will affect results with the chance for ditch detection increasing logarithmically with increasing area surveyed. This scale dependency is apparent in our assessment where the proportion of salt marshes ditched increases from 7.3% at 30-m resolution (900 m^2 salt marsh) to 64% at 3000-m resolution (9 km^2 salt marsh) (Fig. 3a). Despite dissimilar methods, this phenomenon is also apparent when comparing ditch presence/absence quantified by other authors (though scale of analysis is often not reported). For instance, with increasing scale of analysis,

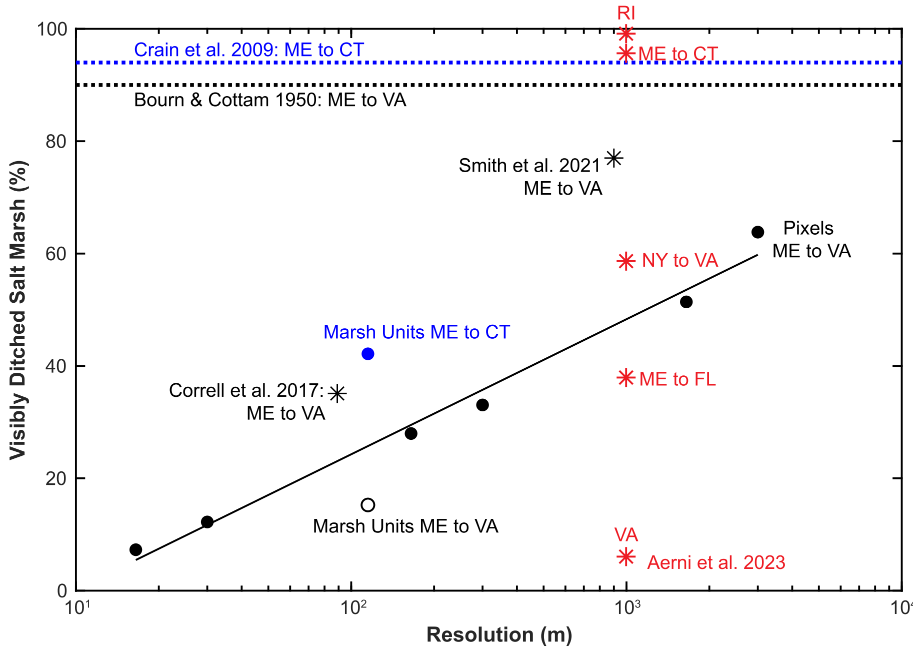

Fig. 3. The effect of spatial resolution of salt marsh areas on the percent of salt marshes that are ditched (i.e., total number of marshes that with one or more visible ditches divided by the total number). Black points are the total percents for each marsh pixel (30- to 3000-m) with a best fit logarithmic line to show trend. Blue and white points are total percents for marsh units for New England (Maine (ME) to Connecticut (CT)) and the entire Northeast (ME to Virginia (VA)). Previously published estimates of percent ditch presence are indicated as dashed lines when no analysis resolution was specified (Bourn and Cottam, 1950; Crain et al., 2009), and as stars when specified (Correll et al., 2017; Smith et al., 2021; all red points are from Aerni et al., 2023).

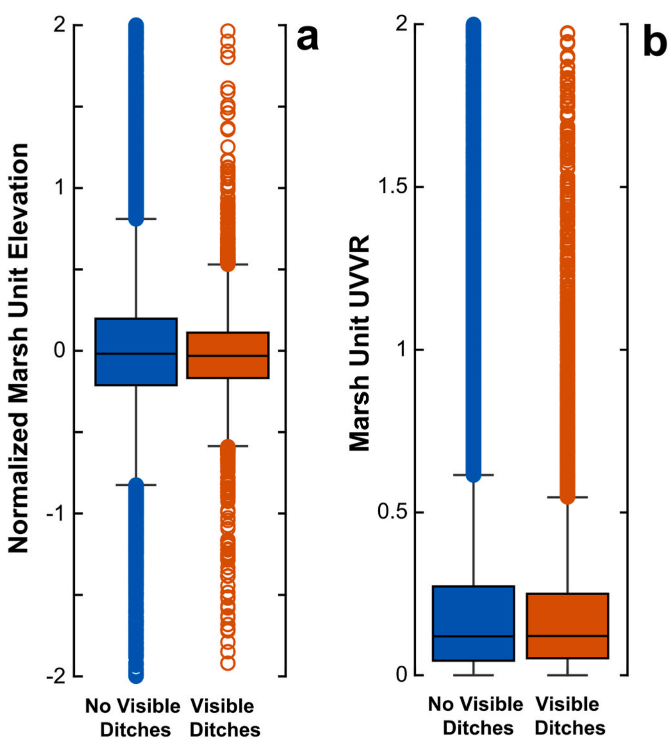

Fig. 4. Box plots of a: marsh unit elevation relative to regional mean and normalized regional tidal range (normalized elevation) and b: marsh unit unvegetated to vegetated ratio (UVVR) for salt marshes with (orange) and without (blue) visible ditches. Boxes represent the first and third quartile, center lines indicate medians, whiskers are minimums and maximums, and circles are outliers. Kruskal-Wallis tests revealed a: a significant difference ($p < 0.001$, df = 48,725), and b: a weak difference ($p = 0.06$, df = 47,667).

Correll et al. (2017) and Smith et al. (2021) identified ditching in 35.1% and 77% of salt marshes from Maine to Virginia, respectively. Analysis of ditching intensity is also scale dependent, with a trend of decreasing visible ditch density with increased salt marsh area (Fig. S4).

Overall, both the percent of marshes that are visibly ditched (Fig. 3a) and visible ditch density (Fig. S4) are both lower in our analysis for the Northeast than that reported in past studies. One explanation could be the timing of our analysis. Our digitization was based on 2023 basemap imagery, whereas most ditching occurred in the first half of the 20th century or earlier in the Northeast. Bourn and Cottam (1950) report that 90% of salt marshes from Maine to Virginia are ditched. This figure may be greater than our highest estimate of ditch density at a scale of 3000-m because many ditches have been infilled over the more than 70 years since their assessment (suggested by Weinstein et al., 2000). Indeed, Corman et al. (2012) observed that ~17% of ditches on Long Island, NY had naturally filled since installation in the 1930s. Additionally, since we restricted our geographic extent to salt marshes as classified by the NWI E2EM designation, we do not include ditched areas in our inventory for any salt marsh that completely converted to another habitat (transitioned to upland, reclaimed for development, or drowned to mudflat/open water) in the time between the initial analysis of Bourn and Cottam (1950) and the ~2000–2010 publication of the NWI.

When compared to other contemporary inventories (Aerni et al., 2023; Correll et al., 2017; Crain et al., 2009; Smith et al., 2021), our relatively lower ditch presence estimates are likely due to the larger area of salt marshes that we assessed. There is a tendency of previous ditch assessments to focus on salt marshes above a certain size, excluding small marshes. For instance, Aerni et al. (2023) selected marshes greater than 1 km^2; however, amongst the NWI E2EM classified salt marshes for the Northeast, areas >1 km^2 account for only 64% of the total area. Indeed, 84% of NWI E2EM salt marshes <1 km^2 have no visible ditches, whereas only 41% of salt marshes >1 km^2 have no ditches. As such, inclusion of more of these smaller salt marshes decreases the overall fraction of ditched salt marsh. This pattern might suggest that these smaller salt marsh complexes may have been less frequently targeted as regions where ditching is necessary for mosquito control or where

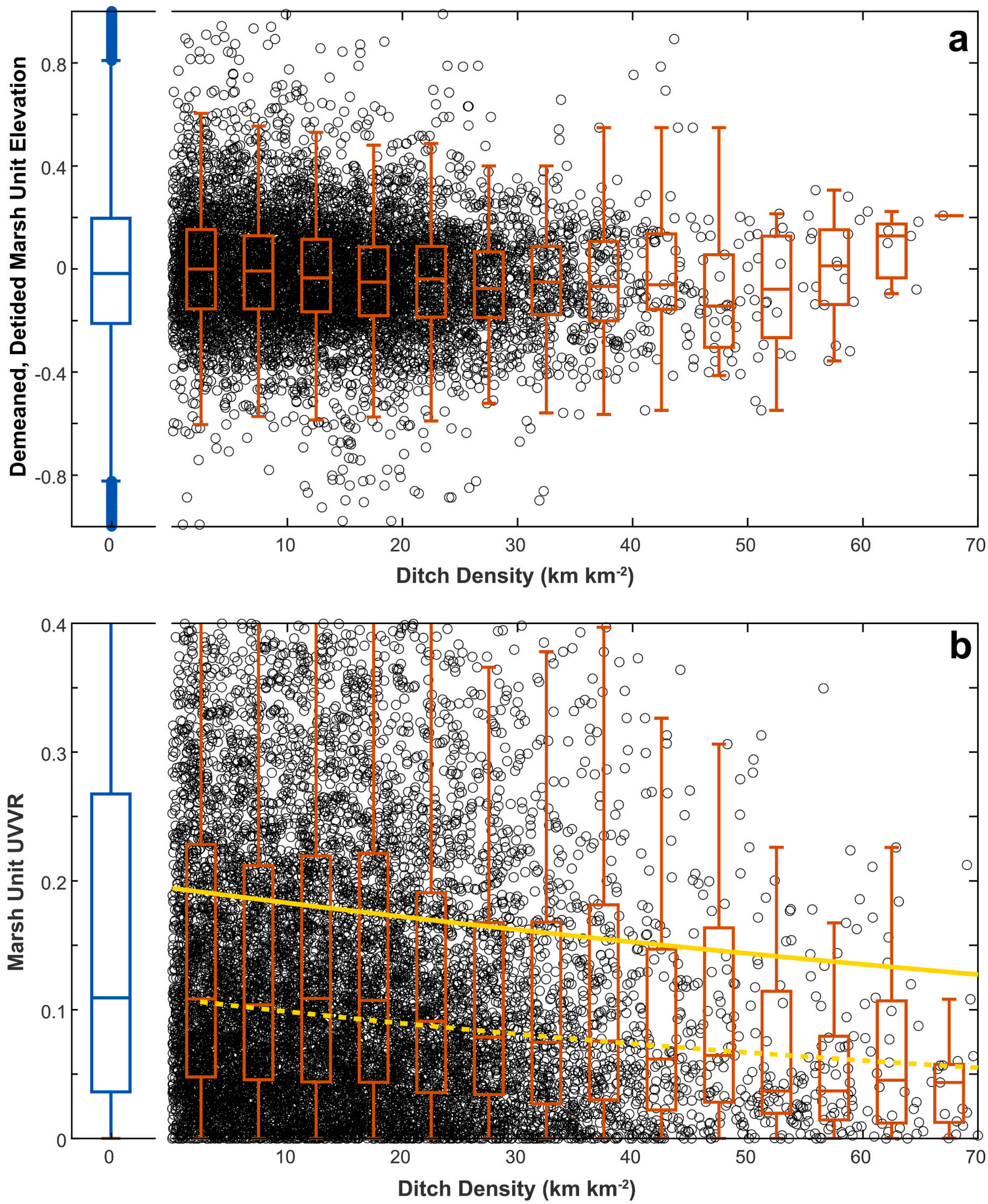

Fig. 5. a: Elevation relative to regional mean and normalized regional tidal range (normalized elevation) and b: Unvegetated to vegetated ratio (UVVR) for marsh units without (left blue box plot) and with (right black individual circles and orange box plots) visible ditches (binned by 5 km km^{-2}). See Fig. 4 for an explanation of box plots. Best fit exponential relationships are plotted as a solid yellow line for continuous data ($R^2 = 0$, df = 12,190, sse = 712, rmse = 0.24) and as a dashed yellow line for median binned data ($R^2 = 0.56$, df = 18, sse = 0.05, rmse = 0.016).

ditching would produce enough arable land, or earlier agricultural ditching was more modest for smaller marshes and subsequent infilling was more likely. Alternatively, these smaller salt marshes may have been larger ecosystems that lost area due to ditching. Future studies should seek to test the latter possibility using historical aerial photographs in regions where significant salt marsh area has convertedto mudflat or subtidal area.

In coastal research and management in the Northeast there is a common trend of focusing on larger salt marsh systems. This is evident when comparing the large marsh complexes found in the National Estuarine Research Reserve systems to the many small and narrow salt marshes in this region (Morgan et al., 2009; Roman et al., 2000). We argue that better inclusion of smaller marshes improves the accuracy and representativeness of assessments of the impacts of various stressors on salt marsh morphodynamics, including ours. Further, smaller salt marshes may be targeted for their research value, as in the case of ditch

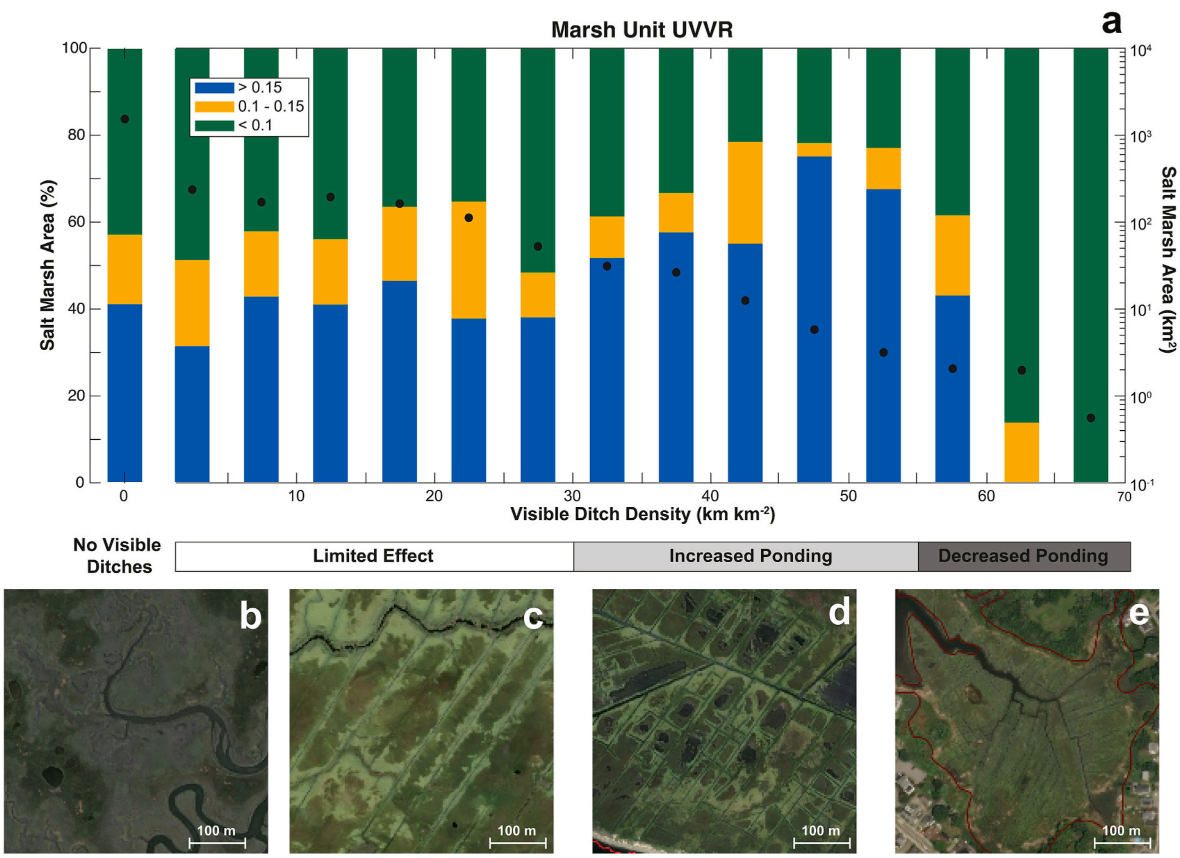

Fig. 6. a: Bar plots of percent salt marsh area (left y-axis) that falls within three unvegetated vegetated ratio (UVVR) classes for marsh units without (left bar) and with (right bars) visible ditches (binned by 5 km km^{-2}). Total salt marsh area (right log y-axis) in each bin is plotted as black dots. Aerial photograph examples of salt marshes with b: no visible ditches (Eastern Shore, Virginia), c: moderate density of ditches (Barnstable, Massachusetts), d: high density of ditches (West Sayville, New York), and e: extreme density of ditches (Great Marsh, Massachusetts). All photographs are at the same scale.

presence where they may provide important control sites.

4.3. How do ditches affect salt marsh quality?

We demonstrate the importance of spatial scale on ditch presence and densities, and our work builds on the growing appreciation for the need to apply metrics of vulnerability at appropriate scaling (e.g., Mariotti, 2020; Smith et al., 2024; Yando et al., 2023). Because of this, we have placed substantial focus on identifying the appropriate spatial scale to assess the effects of ditching on salt marsh vulnerability. Ditches are low elevation and unvegetated features within the salt marsh platform, and as such, we must look at a scale that is coarse enough to allow for absorption of ditch pixels into vegetated pixels. Marsh units provide a standardized area that is defined by an inundation/drainage basin for tidal flow (Defne and Ganju, 2018a,b). Vulnerability metrics determined across these marsh units are representative of the geomorphic state of the system, rather than individual components of a complex, heterogeneous landscape composed of ditches, small tidal creeks, ponds, pannes, mudflats, and vegetation. Additionally, marsh units are determined at fine enough resolution to still provide sufficient sample size for our comparisons between vulnerability metrics and visible ditch densities. Of the 3,564 km^2 of NWI E2EM salt marsh in our study's Northeastern footprint, we assessed publicly available marsh unit areas, totaling 2,229 km^2 (63% of total, n = 79,945 marsh units).

We found no obvious influence of visible ditching on salt marsh normalized elevation (Fig. 4a). Though statistically significant, the differences in normalized elevation were negligible and therefore likely do not reflect a true effect on the vulnerability of salt marshes. We did find that visible ditches are more present in marsh units with a higher fraction of high to low marsh vegetation. However, when queried by marsh units with a greater fraction of high marsh compared to low, and vice versa, salt marsh units with and without visible ditches still do not have obviously different relative elevations.

Our scale of analysis may be inappropriate, explaining the absence of an observable influence of ditching on salt marsh normalized elevation. When elevation is averaged over the entire marsh unit, microtopography caused by ditches — high elevation levees on ditch edges due to ditch spoil and low elevation ponding further from ditch channels (Smith et al., 2021) – may obscure any differences between marsh units with and without ditches. Previous studies that tested the effects of ditches *in situ* do find an effect on elevation. Burdick et al. (2020), for instance, measured 9 cm of subsidence between tightly spaced (14 m) ditches. Still, in the absence of control sites, it is hard to know whether similar elevation loss is observable across all salt marshes in Northeastern U.S. affected by ditches and whether ditches are the main driver of change. Future studies should seek to further assess the effects of ditches on elevation at a range of scales across the entirety of the Northeastern U.S.

The UVVR is a metric of ponding extent that is correlated with elevation (Ganju et al., 2020). UVVR may be more nuanced than normalized elevation and may therefore be a more appropriate metric to assess ditching impact. While we see no difference in UVVR ranges between marsh units with and without visible ditch presence, mean UVVR does exponentially decrease with increasing intensity of ditching. Some previous studies observed similar trends in ponding (Smith et al., 2021), while others found that ditching produced limited effects or even decreased pond sizes and densities (Redfield, 1972; Vincent et al., 2013). Thus, the impact of ditching on ponding remains unclear.

The issue with assessing observational data in a bivariate fashion (marsh unit visible ditch density compared to UVVR), is that neither estimate is standardized by area (which does vary across marsh units;

Fig. S1). Further, this approach does not account for other potential stressors on salt marsh health that may cofound with increasing ditch density, such as population (Bromberg and Bertness, 2005), crab herbivory (Smith, 2024), and/or relative sea level rise, which displays a latitudinal trend partly due to glacial isostatic adjustment (Ohenhen et al., 2023). Indeed, the wide distribution of marsh unit UVVR at 0 km km^{-2} ditch density clearly highlights the occurrence of cofounding influences on salt marsh vulnerability (Fig. 6a). When visible ditch density is 0 km km^{-2}, only 16% of salt marsh area is categorized as having a UVVR between 0.1 and 0.15, considered resilient to drowning under sea level rise (Ganju et al., 2020, 2022; Wasson et al., 2019). In fact, a much larger percentage of salt marsh exhibits a UVVR greater than 0.15 (41%), considered a sign of runaway ponding (Duran Vinent et al., 2021; Mariotti, 2020), or a UVVR less than 0.1 (43%) with no/limited ponding. By visualizing the area distributions of the three UVVR categories with increasing ditching intensity, we can address how visible ditches affect salt marsh quality without needing to directly account for any additional stressors.

With increased ditch densities, the proportion of salt marsh area with extensive ponding only begins to increase at densities greater than ~30 km km^{-2} (Fig. 6a), indicating that drainage from ditches below this density may approximate the hydrologic function performed by sinuous tidal creeks. Above this ditch density, the deleterious effects of ditching that result in greater ponding, become more pronounced. Surprisingly however, this trend of increased ponding only continues until visible ditch densities reach ~55 km km^{-2}, at which threshold the proportion of salt marsh with ponding sharply decreases to be replaced with salt marsh without extensive ponding. Perhaps this ditch density can more effectively drain the salt marsh, preventing pond formation. These multiple thresholds may explain the complicated and often contradictory prior assessments of ditching on salt marsh pond formation, as noted by others (e.g., Aerni et al., 2023; Smith et al., 2021; Tonjes, 2013).

Parallel ditching may only cause an increase in ponding under a certain range of visible ditch densities, but these ponds are often distinctive based on their morphology. In particular, "waffle" pools, square or rectangular ponds of standing water (resembling maple syrup), form between the ridges adjacent to parallel ditches (Adamowicz et al., 2020; Smith et al., 2021). Based on our results, we suggest that while the waffle pool morphology may directly result from parallel ditches, increased ponding results from ditches only within a ditch density range of 30–55 km km^{-2}. However, it is important to note that this study is only accounting for the aerial footprint of ponds, and other authors have observed that ponds are deeper in areas affected by ditches (Adamowicz et al., 2020; Smith et al., 2021), and these deeper ponds may be difficult to restore.

4.4. *What are the implications for salt marsh management and restoration?*

Our geospatial analyses across all salt marshes of the Northeastern U. S. (defined by the NWI E2EM) provide more robust information to help inform management decisions related to ditches, especially the trends we observed in marsh unit UVVR in response to increases in visible ditch density. From our identification of an ecological threshold effect imposed by increasing ditch densities on the distribution of marsh unit UVVR, we identify three degrees of ditching intensity – moderate (0–30 km km^{-2}), high (30–55 km km^{-2}), and extreme (>55 km km^{-2}). These thresholds can provide important information to include in models used to predict system responses to restoration decisions. Such models may be useful for identifying restoration actions that may produce the largest management benefit relative to input (i.e., cost, time) – i.e., relate ecological thresholds to specified management utility thresholds – which should guide prioritizing the scale and location of restoration efforts (Nichols et al., 2014).

Under moderate ditching, <30 km km^{-2} visible ditch density, the extent of ponding is comparable to that of marsh areas with no visible ditches indicating that visible ditches are not driving interior deterioration. As such, remediation of ditches alone is unlikely to improve the health of the marsh, and an assessment of other stressors should be undertaken before developing a comprehensive restoration plan. We recognize that some sites we categorized as "no visible ditches" or as low visible ditch densities likely have a history of more prevalent ditches that filled (Weinstein et al., 2000). We therefore cannot say with certainty that deterioration of those marshes was not a result of the immediate effect in the years to decades following ditch construction or the lingering hydrological impacts of buried, or "zombie" ditches (as detailed by Adamowicz et al., 2020). Future studies should seek to quantify the effects of "zombie" ditches on current salt marsh vulnerability at a local and regional scale. However, our use of the UVVR vulnerability metric, which is indicative of the open-water conversion process rather than a snap-shot in time (Wasson et al., 2019), gives confidence to our results as representative of contemporary trends.

With our findings that visible ditches are of greatest influence on marsh vulnerability within the 30–55 km km^{-2} ditch density range, we propose that sites within this range should be considered for restoration activities that may directly reduce ditch density and increase vegetation cover. For instance, ditch remediation outlined by Burdick et al. (2020) aims to fill ditches with salt hay, promoting the regeneration of *Spartina alterniflora* in the ditch footprints. Although these techniques are still being developed and tested, practitioners have found that their ditch remediation strategy increased plant cover and stem densities while depth of ditches decreased in the three years since treatment. One technique that may be effective for decreasing UVVR in all sites affected by excessive ponding is the installation of runnels, which are shallow channels that connect standing water with deeper ditches or creeks, improving hydrology and increasing vegetation cover (Adamowicz et al., 2020; Besterman et al., 2022; Burdick et al., 2020; McKown et al., 2023, 2024; Perry et al., 2022). In the seven years since runnels were installed in one Massachusetts salt marsh, Besterman et al. (2022) saw a 68% overall increase in vegetative cover, with the greatest improvement in high marsh species. In a Rhode Island salt marsh, Watson et al. (2022) demonstrated that in the four years since installation, runnels reduced the groundwater table, restoring tidal hydrology, increasing vegetative cover, and reducing ponding. Whether these short-term gains result in long-term restoration and long-term stability of Northeast salt marshes requires further investigation.

Marshes with extreme visible ditching density, >55 km km^{-2}, correlated with low UVVR, indicating that these salt marshes are experiencing less deterioration due to ponding than their less ditched counterparts. However, these apparently less vulnerable salt marshes have likely changed through ecogeomorphic processes as a result of the hydrodynamic shifts caused by extensive ditching. Therefore, from a management perspective, "vulnerability" should be evaluated from the decision context. For example, if the management objective is to restore historical marsh vegetation composition, managers may conclude that even with very low UVVR, extensive vegetation cover, and minimal ponding, the lack of natural habitat heterogeneity is undesirable. High vegetation cover observed in these highly ditched marshes indicates a shift in vegetation species coverage to higher percentage of forb species relative to natural tidal creeks, where *Spartina alterniflora* is more dominant (Vincent et al., 2014). If managing for ecosystem services, such as provisioning for wildlife, a lack of natural ponds may also negatively impact species such as waterfowl and nekton that utilize open water on the marsh surface (Erwin et al., 2006; Kneib, 1997). Restoring the diversity of natural features to this landscape may be difficult, with traditional methods of integrating ponds into ditched systems through Open Marsh Water Management resulting in mixed results for increasing habitat value (Erwin et al., 1994; Pepper and Shriver, 2010; Riepe, 2010). Resulting ponds are often smaller and shallower than those required for many bird species and may result in altered ecosystem biogeochemistry and reduced carbon storage, and often require leveling of marsh surface targeting vegetation such as *Spartina patens*, which grow in clumped formations raised above the surface elevation (Erwin et al., 1994; Pepper and Shriver, 2010; Riepe, 2010; Spivak et al., 2017). However, newer methodologies such as Integrated Marsh Management aimed at replacing grid ditch networks with naturalized tidal channels and ponds, have proven effective in increasing the abundance and diversity of nekton species, shorebird and waterfowl densities, avian species diversity, as well as decreasing forb vegetation cover (Rochlin et al., 2012).

These complexities highlight the importance of careful consideration of management goals for effective decision-making regarding marsh restoration or ditch remediation. By conducting geospatial analyses across a range of spatiotemporal scales and including empirical estimates of multiple ecological thresholds, our findings provide important inputs for advancing complex marsh dynamics models for predicting outcomes of management interventions at varying scales, and levels of ditch densities, which are critical for prioritizing investments and restoring function to coastal marsh ecosystems. We also note that restoration of ditches may be beneficial in many instances regardless of their local effects on hydrology and total vegetation coverage, as for instance ditches appear to provide conduits and habitats for invasive species (e.g., Ning et al., 2019) and increase carbon emissions (e.g., Xue et al., 2024).

5. Conclusion

Here, we aimed to address the degree of ditch influence on salt marsh degradation in the U.S. Northeast. To do this, we conducted a region-wide evaluation, incorporating multiple spatial resolutions and assessments in the context of several vulnerability indices. We were able to rectify discrepancies in previous studies' findings on both the fraction of ditched salt marshes and the influence of ditches on salt marsh vulnerability to interior loss through ponding. Our study identified a scale dependency in ditch presence and density and elucidated the relationship between the vulnerability metric UVVR and ditch density. Perhaps the most influential of our findings for management considerations was the non-linear relationship between marsh vulnerability and ditching density. We observed little influence of ditches below densities of 30 km km^{-2}, while above this threshold, UVVR increases indicating internal marsh deterioration until densities reach a higher secondary threshold of greater than 55 km km^{-2}. At this point, UVVR dramatically decreases indicating a decline in marsh ponding. Our study represents an advance in disentangling the complexities of the effects of anthropogenic modification on salt marsh dynamics and reducing the uncertainty of ditch impacts on marsh vulnerability, both of which may better inform restoration and other management interventions.

Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the U.S. Government. The findings and conclusions in this article are those of the author(s) and do not necessarily represent the views of the U.S. Fish and Wildlife Service.

CRediT authorship contribution statement

Erin K. Peck: Writing – original draft, Visualization, Validation, Resources, Project administration, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Julie E. Walker:** Writing – original draft, Validation, Project administration, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Katherine V. Ackerman:** Writing – original draft, Methodology, Formal analysis, Conceptualization. **Joel Carr:** Writing – original draft, Methodology, Conceptualization. **Maureen D. Correll:** Writing – original draft, Methodology, Investigation, Funding acquisition, Conceptualization. **Zafer Defne:** Writing – original draft, Methodology, Formal analysis, Conceptualization. **Linda A. Deegan:** Writing – original draft, Conceptualization. **Mitchell J. Eaton:** Writing – original draft, Conceptualization. **Neil K. Ganju:** Writing – original draft, Supervision, Methodology, Funding acquisition, Conceptualization. **Mitch Hartley:** Writing – original draft, Conceptualization. **Catherine Johnson:** Writing – original draft, Conceptualization. **Jason Mercer:** Writing – original draft, Methodology, Formal analysis, Conceptualization. **Katharine J. Ruskin:** Writing – original draft, Investigation, Data curation, Conceptualization. **Jonathan D. Woodruff:** Writing – original draft, Supervision, Funding acquisition, Conceptualization. **Brian Yellen:** Writing – original draft, Conceptualization.

Data availability

Ditch polylines are available at https://doi.org/10.5066/P13WF8ZT.

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Declaration of competing interest

The authors have nothing to declare.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jenvman.2025.124444.

Data availability

I have shared a link to my data.

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