Pond Excavation Reduces Coastal Wetland Carbon Dioxide Assimilation

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Abstract Coastal wetlands comprise important global carbon sinks; however, anthropogenic disturbance accompanied with accelerating sea level rise threaten their continued survival. In this study, we quantified habitat disturbance to salt marshes in Barnegat Bay, New Jersey, resulting from the construction of ponds for mosquito control. Geographic object-based image analysis of high-resolution four-band aerial imagery revealed that over 7,000 ponds were constructed in the marsh complex with pond densities as high as 290 ponds per km². Physical disturbance from pond creation and sediment dispersal extended to over 17% of the bay's tidal wetlands. By tracking recolonization of vegetation, we estimated that it took 5 years for 51% vegetation recovery and 10 years for 69% recovery, with complete recovery (100%) not expected for more than 50 years. This suggests that efforts to extend the lifespan of drowning coastal wetlands through sediment additions might disrupt carbon dioxide assimilation, as effects of disturbance persist. Focusing on greenhouse gas exchange, our work found that areas of marsh vegetation contribute to carbon assimilation (−42 g C · m⁻² · year⁻¹), while ponds and areas of bare peat created by pond excavation were associated with carbon emissions (44 and 125 g C · m⁻² · year⁻¹, respectively). These results suggest that the conversion of wetlands to ponds—which is a significant driver of coastal wetland loss worldwide—may convert coastal wetlands from greenhouse gas sinks to sources. Additionally, quantifying the area of vegetation within a marsh (vs. bare ground or open water) is important for quantifying their greenhouse gas mitigation function.

Plain Language Summary Coastal wetlands are important for our planet as they remove carbon dioxide from the atmosphere. Our goal was to determine how the construction of ponds in wetlands—which is common worldwide to support expanding aquaculture—affects the ability of marshes to take up carbon dioxide. We estimated the number and extent of ponds constructed for mosquito control in coastal wetlands in Barnegat Bay, New Jersey. Additionally, we measured greenhouse gas exchange from the ponds, surrounding intact marsh, and in bare areas at several sites across the growing season to estimate the total carbon uptake of the whole marsh. We found that over 7,000 ponds have been dug for mosquito control in New Jersey and that pond density is as high as 290 ponds per km². We also found that the ponds constructed in wetlands reduce the ability of wetlands to remove carbon dioxide from the atmosphere.

1. Introduction

Coastal wetlands form a transition zone between marine and terrestrial ecosystems. They provide crucial ecosystem services, which include acting as fish nurseries, improving coastal water quality through the sequestration and removal of sediment and nutrients, providing shoreline protection through wave attenuation and storm surge reduction, and sequestering globally significant volumes of carbon (C; Rochlin et al., 2012). These coastal ecosystems store C as aboveground biomass (i.e., leaves, stems, and branches), as belowground biomass (i.e., roots and rhizomes), and within C-dense organic soils, where anoxic conditions prevent or slow the remineralization of organic matter. Salt marshes additionally trap allochthonous sediment via emergent stems and accrete vertically; thus, they do not have a C saturation point (Chmura et al., 2003; Mcleod et al., 2011). Because of this vertical accretion capacity, vegetated coastal wetlands sequester organic C at rates of 20 to 2,000 g of C · m⁻² · year⁻¹, which has been scaled to 100 Tg of C per year globally (Hopkinson et al., 2012; Mcleod et al., 2011).
Recently, intense scientific attention has been focused on the C sequestration capacity of vegetated coastal ecosystems (termed “blue carbon”). This is because the conservation and restoration of such environments can provide environmental benefits through the mitigation of greenhouse gas (GHG) emissions as well as other ecosystem services valuable to coastal communities facing climate change (Carr et al., 2018; Narayan et al., 2017). In addition, C finance is being used to subsidize restoration projects, expanding and diversifying the sources of funding available for coastal conservation and restoration (Emmett-Mattox et al., 2010). However, both climate change and anthropogenic disturbances threaten the continued survival of vegetated coastal ecosystems (Watson & Hinojosa Corona, 2017).

Previous studies have shown that anthropogenic disturbances often shift wetland GHG budgets from sinks to sources, transitioning the net radiative forcing of the habitat from negative to positive. As wetlands are converted to lands managed for agriculture or timber harvest, CH4 emissions tend to decrease in association with soil drying, but CO2 emissions increase as soil C is remineralized (Ciais et al., 2014). Where ecosystems are affected by upstream fertilizer applications or untreated wastewater, N2O emissions can be elevated due to enhanced nitrogen cycling (Petrescu et al., 2015). However, most studies of disturbance impacts to wetland GHG balance have focused on freshwater wetlands, where the climate footprint is a function of the stoichiometry between CO2 uptake via bioassimilation and the CH4 emissions that result from the remineralization of fixed C in flooded anoxic soils (Van Der Nat & Middelburg, 2000). In contrast, for saline tidal wetlands, the abundance of marine sulfate tends to retard methanogenesis as sulfate-reducing bacteria outcompete methanogens in the presence of abundant sulfate ions (Poffenbarger et al., 2011). As such, studies of disturbance to the GHG balance of salt marshes have focused on the impacts of wetland clearing/drainage or tidal restrictions to GHG exchange. These impacts can remineralize soil C (Lovelock et al., 2011) or increase CH4 emissions due to salinity reductions (Kroeger et al., 2017). Previous work has demonstrated the value of coastal wetland restoration for reducing GHG emissions (Connor et al., 2001; Cornell et al., 2007; Matsui et al., 2012), thereby building support for the finance of coupled C sequestration-restoration projects (Duarte et al., 2013). However, there are still few field studies that have quantified the impacts of disturbances to wetland C-sequestration capacity (Sidik & Lovelock, 2013).

Alteration of U.S. coastal marshes began with European colonization (Sebold, 1992; James-Pirri et al., 2012), and initially included modifications for grazing, agriculture, salt hay farming, and waterfowl impoundments (Sebold, 1992). Salt marshes provide a suitable breeding habitat for salt marsh mosquito species (Rochlin et al., 2012) and have been physically modified to control mosquito populations since the early 1900s (Lathrop et al., 2000). Mosquitos are a public health concern, as they are vectors for diseases like encephalitis and West Nile virus. Efforts to control mosquitos via drainage led to installation of parallel grid ditching in almost 95% of the tidal marshes between Maine and Virginia, mostly dating to the Works Progress Administration of the 1930s (Elsey-Quirk & Adamowicz, 2016; Lathrop et al., 2000; Rochlin et al., 2012).

Alterations of the salt marsh caused changes to the physical and chemical soil environment over short time scales (Elsey-Quirk & Adamowicz, 2016); ditching lowered water tables, reduced sedimentation, and increased soil aeration, decomposition, and compaction (Elsey-Quirk & Adamowicz, 2016). Water table dynamics and topography in the marsh setting influence surface sedimentation, soil development, porewater chemistry, vegetation community, and plant productivity; therefore, altering these systems has had strong impacts to ecosystem structure and function (Elsey-Quirk & Adamowicz, 2016). Because of concern with the negative impacts to wildlife, such as fish and water fowl, which utilized ponded areas, grid ditching for mosquito control was phased out in the 1960s (James-Pirri et al., 2012; Lathrop et al., 2000).

In an attempt to minimize the use of pesticides and reduce the negative impacts on hydrology and vegetation of salt marshes from ditching, an alternative practice to control mosquitos was developed in New Jersey in the 1960s (Ferringo & Jobbins, 1968; James-Pirri et al., 2012; Lathrop et al., 2000). Termed open marsh water management (OMWM), its objective is to reduce mosquito breeding from a given area of marsh while enhancing wildlife habitat (Ferringo & Jobbins, 1968). This practice has been used extensively in the mid-Atlantic and in sparse areas of Massachusetts, Connecticut, New York, Florida, and Louisiana (Elsey-Quirk & Adamowicz, 2016). The techniques used in OMWM are regionally dependent. In New England, ponds have been created through ditch plugging, while in the Mid-Atlantic region, radial ditches and isolated ponds have been constructed (Elsey-Quirk & Adamowicz, 2016; James-Pirri et al., 2009). Ditch plugging involves plugging the seaward end of present ditches to create pools (Vincent et al., 2012). Radial
ditching involves the connection of present ditches to isolated ponds (Figure 1; Ferringo & Jobbins, 1968; James-Pirri et al., 2009). These plugs and radial ditches connect to the historic ditches, allowing high tides to bring mosquito larvae-eating fish to the ponds to reduce the larval population in a given area (Elsey-Quirk & Adamowicz, 2016; Ferringo & Jobbins, 1968).

Although pond features in the salt marsh landscape are understudied, current research is focusing on their role in contributing to overall marsh drowning (Hartig et al., 2002; Mariotti, 2016). There are many ways natural ponds can form on the marsh platform—waterlogging from poor drainage, plant loss associated with disturbance by macroalgae, reduced sediment accretion, and vegetation die-back (Mariotti, 2016; Wilson et al., 2009; Wilson et al., 2010). Although ponds are deemed natural features of the marsh; their ability to assimilate CO2 in terms of the entire marsh complex is not well understood.

In this study, we focused on quantifying changes in C-assimilation potential associated with disturbance by measuring GHG exchange in vegetated marsh referred to as intact marsh (IM), from ponds excavated for mosquito control within these vegetated wetlands, and from bare peat (BP) areas that were created by sediment side-casts during pond construction. While changes in C storage resulting from OMWM in wetlands have not previously been reported, we expect that this disturbance could reduce wetland C-assimilation through two mechanisms. First, we expected that C previously sequestered in vegetated wetlands that is side-cast during pond excavation will be remineralized. Second, we expected that conversion of vegetated wetlands (a GHG sink) to an open water pond (a GHG source; Moseman-Valtierra et al., 2016) would decrease the potential C sequestration of the landscape. In addition to obtaining information about the impacts of this disturbance that can inform U.S. coastal wetland management (i.e., the suitability of continued pond construction), the excavation of similar aquaculture ponds within coastal wetlands is a major driver of coastal landscape change globally and is suspected to be a significant driver of emissions (Sidik & Lovelock, 2013). For instance, over 50% of salt marshes have been lost in just one region area of Sri Lanka due to shrimp farming and shrimp farming is projected to expand globally. Because this land conversion is happening in tidal wetlands, which are well-known C-sinks, it is important to quantify the effects of this land conversion on GHG emissions to permit informed management (Thornton et al., 2003).

The objective of our study was twofold: first, to quantify the area of marsh that had been disturbed through pond excavation, and, second, to use measures of GHG exchange among vegetated and disturbed habitat types to estimate the impacts of this disturbance to coastal wetland GHG balance. Image classification was conducted using geographic object-based image analysis (GEOBIA) of four-band high-resolution aerial photography, using object-image metrics for classification. Cavity ring-down spectroscopy and infrared gas analysis were used to examine in situ GHG fluxes in three disturbed salt marshes across the growing season, and measures were scaled to construct a yearly budget. These data were then used to show the overall effect of disturbance to coastal wetland GHG balance, highlighting the importance of considering the area of vegetation, rather than the overall marsh area—which typically includes pools, channels, and bare areas—in estimates of potential GHG emission mitigation by coastal wetlands.

2. Materials and Methods

2.1. Study Site

Three study sites were chosen in the Barnegat Bay-Little Egg Harbor coastal lagoon system, New Jersey, USA (Figure 2). The lagoon is shallow (~1–3 m), microtidal (tides of 0.15–0.95 m), and strongly affected by the land-sea breeze. There are currently 10,900 ha of salt marsh in Barnegat Bay; twentieth century coastal development resulted in the loss of approximately 28% of the bay’s historic salt marshes (Kennish, 2001). Study sites were chosen in areas that had been subjected to extensive pond excavation: Catts Island in the northern portion of Barnegat Bay (39.9824°F, 74.1236°W), Upper Barnegat Bay Wildlife Management Area in central Barnegat Bay (39.8490°N, 74.1431°W), and the Tuckerton peninsula in south Barnegat Bay (39.5210°N, 74.2996°W (Figure 2). Some additional measures were conducted at the Lighthouse Center for Environmental Education (39.7705°N, 74.11925°W; supporting information).

Study sites were arranged to capture the north to south salinity gradient (15–35‰) found in Barnegat Bay (Kennish, 2001). The vegetation of these sites consisted of Spartina alterniflora (mixed tall and short form) and Spartina patens. Mosquito control ponds were constructed at Catts Island between 1984 and 1988, at
the Upper Barnegat Bay Wildlife Management Area in 1989, and along the Tuckerton peninsula in 1999 (J. Schmidt, Ocean County Mosquito Extermination Commission, personal communication Friday, February 2, 2018). For Barnegat Bay tidal wetlands, peak aboveground biomass has been previously

![Figure 1. Aerial photography of New Jersey coastal marsh (39.6265°N, 74.2702°W) (a) before (2005) and (b) after implementation of open marsh water management (2014).](image)

![Figure 2. Study sites in Barnegat Bay, Ocean County: Cattus Island, Upper Barnegat Bay Wildlife Management Area, the Lighthouse Center, and the Tuckerton peninsula.](image)
Table 1

| Segmentation Type | Scale parameter | NIR weight | Shape heterogeneity |
|-------------------|-----------------|------------|---------------------|
| MRS               | 15              | 2          | 0.5                 |
| SDS               | 15              | 5          | 0.5                 |

Abbreviations: MRS: multiresolution segmentation; NIR: near infrared; SDS: spectral-difference segmentation.

Measured at 700±200 g/m² and 580±55 g/m² at two locations over monitoring periods that have exceeded 5 years (K. Raper, Academy of Natural Sciences, unpublished results), and belowground production at those same locations has been estimated at 180±70 g/m² and 260±55 g/m², yielding an overall standing peak biomass estimate of 880±210 g/m² and 840±90 g/m², respectively (Elsey-Quirk & Unger, 2018).

2.2. Imagery and Processing

The areal extent of wetlands disturbed by the construction of mosquito control ponds was quantified using GEOBIA of 2014 leaf-off high-resolution (0.3-m) four-band (near infrared [NIR]; red, green, and blue) digital imagery (Office for Coastal Management, 2018). We cross-referenced aerial tiles to maps obtained by the Ocean County Mosquito Extermination Commission where OMWM practices took place to ensure analysis was restricted to OMWM impacted areas only. Using eCognition Developer (Version 9.3, Trimble, Westminster, CO, USA), a workflow of segmentation, classification, and manual post editing was followed. Imagery was segmented using multiresolution segmentation followed by spectral-difference segmentation. Segmentation parameters were adjusted in an iterative process to optimize the segmentation of image-object primitives for classification (Dronova, 2015; Table 1) but were the same for all tiles. Once appropriately segmented, a simple binary classification into the categories of OMWM pond vs non-OMWM pond was performed on the image-object primitives by supervised rule-based methods using NIR spectral intensity values, which distinguished vegetation (>100) from open water (55–80) and object-image size and shape, which were used to distinguish other water features from anthropogenically constructed ponds. The mean NIR values of OMWM ponds ranged from 55 to 80 in most of the tiles with some discrepancies in some of the aerial tiles in Barnegat Bay. This mean NIR value is significantly lower than wetland vegetation with a mean NIR above 100; thus, vegetation can be easily delineated from open water. To distinguish OMWM ponds from other water bodies, a combination of mean NIR and object-image size and shape were used. Defining spatial characteristics included elongated oval shape, inner vegetated islands within, and radial arms (supporting information). Shapes identified as anthropogenically constructed ponds were exported as shapefiles and postedited in ArcGIS 10.5 (ESRI, Redlands, CA, USA).

Pond shapes were merged within a radius of 5 m to produce a count of pond number and mean area. A binary confusion matrix was constructed by overlaying 250 random points (125 in suspected OMWM ponds and 125 in suspected wetlands) and classifying these points manually into one of four categories: (1) points intersecting OMWM ponds that were classified correctly, (2) points not intersecting OMWM ponds that were classified correctly, (3) points that fell in OMWM ponds that were incorrectly classified, and (4) points that did not fall in OMWM ponds that were incorrectly classified. An OMWM pond was defined as a round pond, not connected to the channel network, which had inner islands. Producer’s and user’s accuracy and Kappa (a statistical measure of agreement) were computed.

Ages were assigned to ponds based on maps provided by the mosquito commission (J. Schmidt, Ocean County Mosquito Extermination Commission, personal communication Friday, February 2, 2018) as well as inspection of historical imagery. To estimate the area of marsh that was disturbed by sediment side-casts, the extent of bare-ground surrounding the ponds in the spring following pond construction was digitized manually for 10 haphazardly chosen ponds, and for these ponds revegetation was tracked over time using Google Earth Imagery. Years since pond construction were regressed against re-colonization rate using a rational function, and this equation was used to estimate the time required for macrophyte re-colonization, as well as the areal extent of bare ground created by sediment side-casts for the entire estuary. To permit the application of this revegetation function to other ponds, the ratio of BP to pond was calculated. The cumulative area disturbed by pond excavation (habitat conversion, as well as sediment dispersal) was tracked, including areas that had revegetated.

2.3. GHG Exchange

We measured in situ GHG exchange (including net ecosystem exchange (NEE), ecosystem photosynthesis, and ecosystem respiration) across the 2017–2018 growing season at three locations extensively disturbed by construction of ponds for mosquito control (Figure 2 and Table 2). Some additional measures were conducted to investigate the timing of collar placement (day of measure vs. permanent placement) and to
compare gas exchange in marsh vegetation completely unimpacted by OMWM with areas of marsh revegetated after disturbance by OMWM. No significant differences were observed between collars placed the day of measurement vs. earlier in the growing season nor between disturbed and pristine vegetation communities (supporting information).

Methane and CO₂ exchange were measured at several OMWM ponds per site per visit, including measurements for IM vegetation, in areas of BP created by sediment sidecasts during pond construction, and in ponds (Figure 3 and Table 2). Measures of CH₄ and CO₂ exchange were made using an LGR Ultra-Portable Greenhouse Gas Analyzer, and CO₂ was measured using a LI-COR LI-8100. Analyzers were connected to transparent acrylic static chambers (30 × 20 cm) attached to bases installed in the marsh soil (for ponds, a floating chamber was used). A backflow-prevention valve controlled pressure equilibration within the static chambers, and a small fan was attached inside the chamber for proper air mixing during the measurements (Martin & Moseman-Valtierra, 2015). One measurement was made per habitat per pond, and GHG measurement sites were generally 50 m+ distant from each other. Gas fluxes were measured during light conditions and dark conditions for the BP and IM habitats, but only light conditions were measured for the pond habitat due to difficulties with shading of large areas of pond. Although the same ponds at each of the sites were measured for GHG exchange, the plots were in different areas of the pond during each sampling period. Measurements were made within 3 hr of low tide during daylight hours between 9 a.m. and 2 p.m. Diel measures were made (including in marsh, an OMWM pond, and in bare ground) during spring and summer in 2018 to build light-response curves for empirically modeling yearly CO₂ exchange.

Gas exchange measurements were 5 min per habitat type. GHG fluxes were calculated using the volume of chamber and area of the footprint (Martin & Moseman-Valtierra, 2015). The ideal gas law (PV = nRT) was used to calculate the gas concentration rate of change over time with the measured air temperature and

### Table 2

| Marsh System       | Spring          | Summer          | Fall            | Winter          |
|--------------------|-----------------|-----------------|-----------------|-----------------|
| Cattus Island      | 15 May 2017 (6/6) | 22 June 2017 (8/0) | 11 September 2017 (8/4) | 19 January 2018 (4/2) |
| UB Wildlife Mgmt   | 20 April 2018 (6/3) | 1 August 2017 (8/4) | 26 October 2017 (8/4) | 12 September 2017 (8/4) |
| Tuckerton Peninsula| 26 April 2018 (6/3) | 31 July 2017 (8/4) | 14 November 2017 (7/4) | 13 September 2017 (8/3) |
| Lighthouse Center  | 17 April 2018 (6/3) | 3 August 2017 (8/4) | 15 November 2017 (6/3) | 17 July 2017 (2/0) |

Abbreviation: GHG: greenhouse gas.

**Figure 3.** Measurements characterized carbon dioxide and methane exchange in (a) pond, (b) vegetated marsh, and (c) bare peat habitats created by sediment side casts. Figure was adapted from IAN 2017.
atmospheric pressure (Martin & Moseman-Valtierra, 2015). Porewater salinity was collected from a depth of 20–30 cm using a sipper and measured in the field using a calibrated refractometer. Soil temperature was measured with a probe inserted 5 cm into the soil.

2.4. Data Analysis

Linear mixed effects models (lme) were used to examine the dependence of GHG exchange, soil temperature, and salinity on the categorical variables of site and habitat, as well as season (expressed as a continuous variable as the number of days from the summer solstice, Julian date). To detect differences between GHG, soil temperature, salinity, and site, we constructed lme models using GHG, soil temperature, salinity as fixed variables, and either habitat type or Julian date as the random effects. Julian date was included as a random effect when testing the effect of habitat type on the fixed variables and then habitat type was made the random effect when testing the effect of Julian date on the fixed variables. As there were no significant differences in GHG exchange between sites, this factor was excluded from lme models. Statistically significant differences were determined using \( \alpha = 0.05 \), using multiple comparisons of means of general linear hypotheses (glht) as a post-hoc test. To detect differences between light and dark measures for CH4 fluxes, we constructed lme models using CH4 fluxes and chamber exposure (clear vs. opaque) as the fixed variables and date as a random effect.

To estimate yearly CO2 and CH4 fluxes for marshes of Barnegat Bay, including ponded and BP habitats, models were generated based on seasonal and diel sampling data. The three habitat categories—IM, BP, and ponds (P)—were modeled separately and scaled to their spatial extent at Barnegat Bay, where NEE represents net ecosystem exchange, GPP represents gross ecosystem production, ER represents ecosystem respiration, and \( A \) represents area:

\[
\text{NEE} = \frac{[\text{GPP}_{\text{IM}} - \text{ER}_{\text{IM}}]}{A_{\text{IM}}} + \frac{[\text{NEE}_{\text{BP}} \times A_{\text{BP}}]}{A_{\text{BP}}} + \frac{[\text{NEE}_{\text{P}} + A_{P}]}{A_{P}}
\]

For CO2 for the IM habitat, GPP and ER were modeled separately employing partial least squares regression (PLSR). The PLSR regression and principal component analysis identified the environmental parameters that drove GPP and ER. Both datasets were preprocessed to exclude entries with missing response variable, and predictor variables were imputed via K-nearest neighbors, Box-Cox-transformed, centered, and scaled (Abdul-Aziz et al., 2018; Kuhn & Johnson, 2013). The resulting datasets contained 117 observations for photosynthesis rates and 126 observations for respiration rates. PLSR models were computed using the SIMPLS algorithm with tenfold cross validation. To estimate gross ecosystem production and ecosystem respiration across the growing season for 2017, environmental parameters (i.e., temperature and photosynthetically active radiation) were obtained from Jacques Cousteau Marine Field Station (National Oceanic and Atmospheric Administration National Estuarine Research Reserve System [NOAA NERRS], 2018). NEE of the bare ground and P habitats (as well as methane for all habitats) were factored in the above model as constants, since they displayed positive fluxes throughout the day and across the year and did not show significant correlations with the environmental parameters measured during this study. In our diel studies, we did not find methane or carbon dioxide emissions that varied with tidal stage (Figure 7), so we did not incorporate water level variability into our analysis. Analyses were performed using R open-source software (R Core Team 2016). The lme4 package (Bates et al., 2015) was used for linear mixed effects analyses. The multcomp package (Hothorn et al., 2008) was used for simultaneous tests for general linear hypotheses (glht). The pls package (Wehrens & Mevik, 2007) was used for scaling.

3. Results

3.1. Spatial Analysis

The results of the GEOBIA show that the total area directly converted from marsh to open water as part of OMWM practices was approximately 3.0 km² (308 ha). The spatial extent of salt marshes in Barnegat Bay is 109 km² (10,900 ha), not accounting for the disturbance associated with OMWM. Therefore, the area directly disturbed by OMWM equates to almost 3% of the total area of salt marsh in Barnegat Bay. Based on expert classification of random points, our GEOBIA classification was 96.5% accurate (Table 3). Errors of commission and errors of omission were symmetrical, indicating no systematic under or over prediction. By aggregating ponds within a 5-m radius, we estimated that the number of ponds constructed for mosquito control
Figure 4. Open marsh water management pond density within the footprint of the salt marshes in Barnegat Bay, NJ. The pond densities range from approximately 20.8 ponds per km² in the southern regions to the highest densities and 293 ponds per km² in the central part of the bay in Edwin B. Forsythe National Wildlife Refuge.

Table 3

|                  | OMWM present | OWWM absent | Users accuracy (%) |
|------------------|--------------|-------------|--------------------|
| OMWM classified  | 126          | 4           | 97                 |
| present          |              |             |                    |
| OMWM classified  | 5            | 115         | 96                 |
| absent           |              |             |                    |
| Producers accuracy (%) | 96          | 97          | 96.5               |

Note. The user accuracy is the percent accuracy with commission error subtracted, while producer accuracy is the percent accuracy with omission error subtracted. Kappa = 0.928.
Abbreviation: OMWM: open marsh water management.
summed to 7,030 with pond densities up to 290 km$^{-2}$ in some areas (Figure 4); while mean pond area was estimated as 440 ± 23 m$^2$.

Digitization of areas disturbed by pond construction found that sediment was dispersed over an area that averaged 5.9 ± 0.55 times greater than the pond area. Considering pond area in concert with disturbance estimates, these two figures together suggest that disturbance from pond construction—either through direct excavation (308 ha) or sediment sidecasts (1510 ha)—reached 1,820 ha, or 17% of Barnegat Bay tidal wetlands by 2015. Using recolonization rates estimated by study of focal ponds, we estimated changes over time in the extent of bare ground that summed to as high as 500 ha and as low as 15 ha. Using a rational function to approximate revegetation from disturbance related to sediment dispersal, we estimated that after 5 years, there was a recolonization of approximately 51% of the vegetation, while after 10 years 69% of vegetation recolonized. Based on the function we developed, the time required for complete recolonization would exceed 50 years, and there are clearly bare spots remaining even in locations where ponds were constructed in the 1970s (Figure 5).

3.2. Carbon Dioxide Exchange

Carbon dioxide fluxes varied by habitat and by season (Table 4). We found that from spring 2017 to 2018, NEE of CO$_2$ in the IM habitat was generally negative (i.e., CO$_2$ taken up), averaging $-2.3$ μmol·m$^{-2}$·h$^{-1}$ during the growing season to as low as $-0.04$ μmol·m$^{-2}$·s$^{-1}$ in winter and early spring. In contrast, BP and P habitats generally were net emitters of CO$_2$ and the seasonality trend was both reversed and less pronounced (Figure 6). NEE for BP and ponds was highest in the summer (0.35 and 0.14 μmol·m$^{-2}$·s$^{-1}$, respectively) and declined in the fall and winter months to 0.07 and 0.05 μmol·m$^{-2}$·s$^{-1}$, respectively (Figure 6). While IM supported significantly greater rates of CO$_2$ uptake than BP and P habitats, there were no significant differences between BP and P habitats (Table 4). There were significant differences in CO$_2$ fluxes between light and dark conditions for both IM and BP in May ($F_{1,42} = 44.1, p<0.001$) intact ($F_{1,42} = 40.3, p<0.001$) bare, and July ($F_{1,30} = 64.35, p<0.001$) intact ($F_{1,28} = 25.1, p<0.001$) bare, suggesting that both photosynthesis and respiration were occurring in areas of BP (in addition to IM). We observed strong diel cycles in CO$_2$ only in the IM (Figure 7), although in July the BP areas followed IM, with slight uptake during the day and emissions at night.

3.3. Methane Exchange

Although values for CH$_4$ emissions varied much more widely than CO$_2$ exchange measurements, methane emissions were generally positive and yearly average values ranged from 4.5 μmol·m$^{-2}$·h$^{-1}$ from BP to 6.4 μmol·m$^{-2}$·h$^{-1}$ from IM vegetation, and 3.5 μmol·m$^{-2}$·h$^{-1}$ from P (Figure 6 and Table 4). Methane emissions varied significantly throughout the growing season, with greater emissions closer to the summer solstice (Table 4). There were no significant differences in methane emission rates measured in vegetation and in BP under light and dark conditions although on one of the two dates on which we conducted diel measures (the Lighthouse Center, during summer), there were slight diel trends in methane emissions from the P and BP, with greater methane being emitted during daylight hours. There were much higher methane emission values from the IM, although these emissions had no relation with temperature or daylight. The IM habitat emitted the highest levels of methane, and there were no significant differences in methane emissions between the P and BP habitats (Table 4). A few ebullition events were observed, characterized by step increases in methane levels (Van Der Nat & Middelburg, 2000). A comparison of methane emissions and salinity suggested that overall methane emissions were quite low and similar to what would be expected based on the polyhaline conditions (Figure 8).

3.4. Edaphic Variables

Mean porewater salinity values ranged from 18 to 36 ppt, and salinity varied significantly by site and season (Table 5). Tuckerton had the highest porewater salinity, while Cattus Island had the lowest porewater salinity, reflecting the north to south salinity gradient found in Barnegat Bay. Salinity significantly varied by habitat type at Tuckerton (although not at other sites), with highest values in bare ground, and the lowest values in the ponds. Salinity was found to vary significantly by season (analyzed as a continuous variable) at two of the three sites, with lower values during the summer growing season. There was no effect of habitat type on soil temperature at the three sites (Table 5). Soil temperature in at Tuckerton was significantly greater than found at Cattus Island and Upper Barnegat Bay Wildlife Management area.
Figure 5. (a) Percent revegetation of areas disturbed by sediment dispersal as a function of time and (b) change over time in pond area, bare area, and cumulative area disturbed by pond construction or sediment dispersal in Barnegat Bay.
3.5. Impacts of Disturbance on GHG Exchange

Our analysis identified the parameter photosynthetically active radiation, soil temperature, and day from summer solstice as relevant for the PLSR model of GPP of IM. The same variables were incorporated into the respiration model, with the addition of atmospheric temperature. The predictor variables loaded on three components for the GPP model and on two components for ER. Coefficients of determination for

Table 4
Greenhouse Gas Exchange Varied Across Seasons, Between Habitats, and Between Sites

| Growing Season | Fall/Winter | Habitat Type | Julian Date | Site |
|----------------|-------------|--------------|-------------|------|
|                | Average CO₂ Exchange (μmols s⁻¹ m⁻²) ± SE | Results of linear mixed effects model | | |
| Cattus Island  |                                          |                                          | | |
| Intact Marsh   | -2.20 ± 1.43 <br> 0.103 ± 0.291 <br> 0.377 ± 0.245 | F₂,128 = 8.68, p < 0.001*<br>F₁,125 = 2.068, p= 0.15 <br>F₂,346 = 0.345, p = 0.71 | < Bare ground, Pond |
| Bare Ground    | -0.527 ± 0.371 <br> 0.577 ± 0.136 <br> 0.419 ± 0.029 | | |
| Pond           |                                          |                                          | | |
| Upper BB Magnt | -2.03 ± 1.32 <br> 0.036 ± 0.194 <br> 0.109 ± 0.130 | F₂,115 = 10.38, p < 0.001*<br>F₁,115= 1.866, p = 0.18 | < Bare ground, Pond |
| Intact Marsh   | -0.183 ± 0.123 <br>-0.067 ± 0.071 <br> 0.030 ± 0.015 | | |
| Bare Ground    |                                          |                                          | | |
| Pond           |                                          |                                          | | |
| Tuckerton      | -2.11 ± 1.07 <br> 0.873 ± 0.188 <br> 0.186 ± 0.105 | F₂,106= 14.51, p < 0.001*<br>F₁,106 = 2.725, p = 0.10 | < Bare ground, Pond |
| Intact Marsh   | -0.830 ± 0.180 <br> 0.081 ± 0.081 <br> 0.304 ± 0.152 | | |
| Bare Ground    |                                          |                                          | | |
| Pond           |                                          |                                          | | |

Average CH₄ Exchange (μmols hr⁻¹ m⁻²) ± SE

| Growing Season | Fall/Winter | Habitat Type | Julian Date | Site |
|----------------|-------------|--------------|-------------|------|
| Cattus Island  |                                          | Results of linear mixed effects model | | |
| Intact Marsh   | 1.68 ± 1.11 <br> 1.63 ± 1.01 <br> 3.68 ± 3.53 | F₂,67 = 1.56, p = 0.21 <br>F₁,67 = 1.98, p = 0.14 | | |
| Bare Ground    | 0.99 ± 0.43 <br> 1.97 ± 0.99 <br> 1.58 ± 1.60 | | |
| Pond           |                                          |                                          | | |
| Upper BB Magnt | 9.43 ± 7.22 <br> 8.73 ± 6.84 <br> 4.12 ± 1.47 | F₂,57 = 1.20, p = 0.31 <br>F₁,57 = 1.182, p =0.33 | | |
| Intact Marsh   | 13.62 ± 9.65 <br> 10.53 ± 5.11 <br> 1.92 ± 0.56 | | |
| Bare Ground    |                                          |                                          | | |
| Pond           |                                          |                                          | | |
| Tuckerton      | 10.06 ± 6.20 <br> 4.58 ± 2.33 <br> 4.80 ± 2.10 | F₂,55 = 2.56, p = 0.09 <br>F₁,55= 1.91, p = 0.17 | | |
| Intact Marsh   | 1.14 ± 0.28 <br> 0.156 ± 0.62 <br> 1.56 ± 0.51 | | | Tuckerton, Upper BB > Cattus |
| Bare Ground    |                                          |                                          | | |
| Pond           |                                          |                                          | | |

Note: Simultaneous tests for general linear hypotheses (glht) was used for a multiple comparison post hoc test.

Figure 6. Environmental conditions and net ecosystem exchange of carbon dioxide and methane across spring of 2017 to spring of 2018. Environmental data are from the JCNERR (NOAA, 2018); temperature is averaged daily; photosynthetically active radiation is daily maximum values. Net ecosystem exchange of frozen ponds in January of 2018 was assigned a value of 0. Negative values for net ecosystem exchange represent uptake, while positive values represent emissions. Line fits for gas fluxes are third-order polynomial curves; uncertainty values are standard error.
the GPP and ER models were 0.45 and 0.48, respectively. The resulting PLSR model estimated a net assimilation of 3,370 t C per year from IM habitat at Barnegat Bay, based on 2017 meteorological data (Figure 9 and Table 6). The bare ground habitat was estimated to emit 217 t C per year, and the ponds were estimated to emit an additional 137 t C per year. In total, the amount of organic C assimilated by Barnegat Bay marshes in 2017, including ponds, was estimated at −3,020 tC (Figure 9).

4. Discussion

Our goal was to determine the areal extent of Barnegat Bay coastal wetlands that had been disturbed through direct excavation to open water ponds for mosquito control, and indirectly converted to BP by sediment side-casts, as well how this has shifted the balance of GHG exchange. Here, we compare the disturbance posed by OMWM with other disturbances present in coastal wetlands, in terms of both extent and magnitude. We discuss the accuracy of our image classification technique relative to other methods of coastal habitat classification. We then focus on GHG exchange and implications of OMWM on GHG emissions and implications for net C assimilation.

Coastal marshes have been subject to historic land use conversion for row crop agriculture, pasture land, aquaculture, residential, and industrial development (Sebold, 1992; Mcleod et al., 2011). Their simple topography and coastal access have made them good candidates for land use conversion. Fertile wetland soils
make these ecosystems attractive for farming and grazing, and many of the world’s notable agricultural areas are coincident with former coastal wetlands, such as the Sacramento Delta in California, the coastal plain of the Netherlands, and the Sanjiang Plain in China (Liu et al., 2004; Verhoeven & Setter, 2009). Construction of shrimp ponds for aquaculture has disturbed many of the world’s mangroves (Naylor et al., 2000). Where land for coastal development has been at a premium (such as the urban areas of Boston, the San Francisco Bay Area, and Washington DC), much of the urban development lies on former coastal wetlands and up to 60–90% wetlands have been lost (Gedan et al., 2009; Goals Project, 1999).

### 4.1. Spatial Analysis

In this study we found that 3% of the wetlands in Barnegat Bay have been directly disturbed by the construction of ponds for mosquito control, creating over 7,000 anthropogenically excavated pools. Further, we found that an additional 14% of the wetlands have been disturbed by sediment sidecasts. One potential analog for considering this disturbance is the construction of shrimp ponds in coastal wetlands for aquaculture. In comparison with the water management areas in New Jersey, shrimp farm ponds tended to be larger, more spatially aggregated, constructed more rapidly, and associated with negative environmental impacts such as eutrophication (Dahdouh-Guebas et al., 2002; Muttitanon & Tripathi, 2005; Verhoeven & Setter, 2009). In comparison with shrimp ponds, the disturbance posed by water management areas in Barnegat Bay has

#### Table 5

**Porewater Salinity and Soil Temperature Varied Across Seasons, Between Habitats, and Between Sites**

| Growing Season | Fall/Winter | Habitat Type | Julian Date | Site |
|----------------|-------------|--------------|-------------|------|
| Cattus Island  |             |              |             |      |
| Intact Marsh   | 18.58 ± 2.78| 24.30 ± 1.57 | F<sub>2,100</sub> = 2.335, p = 0.10 | F<sub>1,101</sub> = 45.96, p < 0.01*  
fall/winter > summer |
| Bare Ground    | 19.81 ± 2.79| 24.00 ± 2.38 |             |      |
| Pond           | 17.94 ± 1.07| 20.89 ± 1.53 |             |      |
| Upper BB Magmt|             |              |             |      |
| Intact Marsh   | 24.70 ± 3.57| 28.51 ± 1.77 | F<sub>2,94</sub> = 2.431, p = 0.09 | F<sub>1,95</sub> = 5.83, p < 0.01*  
fall/winter > summer |
| Bare Ground    | 26.68 ± 4.81| 29.5 ± 5.01  |             |      |
| Pond           | 25.25 ± 1.26| 26.33 ± 0.52 |             |      |
| Tuckerton      |             |              |             |      |
| Intact Marsh   | 35.81 ± 4.93| 36.8 ± 4.76  | F<sub>2,92</sub> = 30.015, p < 0.01*  
Intact Marsh, Bare ground > Pond |
| Bare Ground    | 36.41 ± 8.52| 39.21 ± 5.93 | F<sub>1,95</sub> = 4.14, p < 0.01*  
fall/winter > summer |
| Pond           | 27.29 ± 3.27| 31.60 ± 1.14 |             |      |
| Upper BB Magmt|             |              |             |      |
| Intact Marsh   | 24.70 ± 3.57| 28.51 ± 1.77 | F<sub>2,77</sub> = 3.98, p = 0.02*  
Pond, Bare ground > Intact Marsh |
| Bare Ground    | 26.68 ± 4.81| 29.5 ± 5.01  | F<sub>1,78</sub> = 120.4, p < 0.01*  
summer > fall/winter |
| Pond           | 25.25 ± 1.26| 26.33 ± 0.52 |             |      |
| Tuckerton      |             |              |             |      |
| Intact Marsh   | 25.76 ± 1.74| 10.48 ± 0.92 | F<sub>2,93</sub> = 12.235, p < 0.01*  
Bare ground > Intact Marsh, Pond |
| Bare Ground    | 24.07 ± 3.76| 12.76 ± 1.07 | F<sub>1,94</sub> = 126.14, p < 0.01*  
summer > fall/winter |
| Pond           | 27.13 ± 1.82| 9.3 ± 0.87   |             |      |

**Note.** Simultaneous test for general linear hypothesis (glht) was used for a multiple comparison post hoc test.
affected less overall area. However, this practice was put in place at least in part as a habitat enhancement technique. How then do the excavated ponds compare with natural ponds found in coastal wetlands that have been eliminated by past drainage practices? In a study of salt marsh pools in the northeastern United States from Long Island to Maine, Adamowicz and Roman (2005) found that natural pools in

![Figure 9.](image)

Table 6

| Model term         | CO₂ flux (μmol CO₂ m⁻² s⁻¹) | C- CO₂ flux (g C m⁻² year⁻¹) | C-CH₄ flux (g C m⁻² year⁻¹) | Area (10⁶ m²) | Total flux by habitat (t C/year) | Total fluxa by habitat (t C/year) |
|--------------------|------------------------------|------------------------------|-----------------------------|---------------|---------------------------------|----------------------------------|
| Live marsh GEP     | $-1.91 ± 2.6$                | $-726 ± 985$                 |                             | 80.8          | $-58.700$                       |                                  |
| Live marsh ER      | $1.8 ± 1.9$                  | $680 ± 719$                  |                             | 80.8          | 55,300                          |                                  |
| Live marsh NEE     | $-42 ± 1.212$                | $0.53 ± 0.11$                |                             | 80.8          | $-3,370 ± 1,212$               | $-3,330 ± 1,212$                |
| Bare peat          | $0.33 ± 0.2$                 | $125 ± 7.6$                  | $0.33 ± 0.2$                | 1.74          | 217 ± 7.6                      | 216 ± 7.6                       |
| OMWM pond          | $0.11 ± 0.04$                | $44 ± 1.5$                   | $0.32 ± 0.17$               | 3.08          | 137 ± 1.5                      | 136 ± 1.5                       |
| Total Barnegat Bay | $8.56$                       | $-3,020 ± 1,212$             |                             |               | $-2,978 ± 1,212$               |                                  |

aIncluding CO₂ equivalent offsets from methane emissions.
unditched marshes covered an average area of $\sim 200 \pm 100$ m², at a density of 13±7 pools per hectare, and comprised 9±4% of habitat cover. Although this study focused on New England where marsh pools are possibly more common as they include kettle ponds in glacially deposited outwash sediments (Sorrie, 1994), our study identified some differences between anthropogenic ponds and natural ponds (Lathrop et al., 2000; Adamowicz & Roman, 2005). Barnegat Bay OMWM ponds tended to be larger, more homogenously sized, and fewer in number than the natural ponds, which may alter the balance of habitat provision. Although the overall area of pond construction summed to only 3% of Barnegat Bay’s wetlands, in the areas where ponds were constructed, they covered 24% of the marsh (Lathrop et al., 2000), far larger than the ~10% estimated for natural ponds (Adamowicz & Roman 2005).

Not only did our work identify ~7,000 anthropogenic ponds on the marsh complex, our work also provides some perspective on the timescale at which it can expect recolonization of vegetation after thin-layer placement of dredge material, a tactic that is being used increasingly to extend the lifespan of drowning wetlands (Wigand et al., 2017). Our work estimated 51% recolonization after 5 years and nearly 70% recolonization after 10 years. However, based on bare areas remaining after several decades, and a regression function that does not suggest complete recolonization within 30 years, our work suggests relatively prolonged impacts to vegetation cover from dredge spoil casts. If marsh platforms never are completely recolonized by macrophytes, sediment additions may not provide enough positive benefits in terms of flood mitigation, C sequestration, and wildlife habitat, all of which depend on a vegetated marsh platform, to outweigh the impact of the short-term disturbance. Therefore, our work highlights the importance of understanding the features of the wetland landscape prior to intervention as the varying geomorphic features can alter the success of thin layer treatment.

Application of GEOBIA resulted in a highly accurate classification of water management areas (~95% accurate). Gao and Mas (2008) concluded that GEOBIA provides an advantage over pixel-based approaches with high-resolution imagery as pixel-based classifiers confuse the spectral variability in high-resolution imagery. Additionally, previous studies have documented greater accuracy in delineation of wetland classes using GEOBIA than pixel-based classification methods (Dronova, 2015). For example, Myint et al. (2008) found 31% greater accuracy with GEOBIA than with pixel-based classifiers, and a review of other studies has suggested improvements in accuracy ranging from 7 to 28% (Dronova, 2015). However, interpreting the accuracy in such comparisons can be problematic as different studies rely on different methods to estimate the final classification accuracy (Dronova, 2015).

### 4.2. GHG Exchange

During the growing season, the rate of NEE in the IM habitat was similar to other studies measuring CO₂ fluxes from short-form Spartina alterniflora, where the high marsh (where our marsh measures were made) was found to take up only modest levels of CO₂ (Martin & Moseman-Valtierra, 2015; Moseman-Valtierra et al., 2016). As the growing season ended, the trend of uptake decreased until November and then we saw small emissions from the IM during winter. The bare ground habitats exhibited small emissions throughout the growing season, with magnitudes similar to those observed previously from unvegetated marsh (Martin & Moseman-Valtierra, 2015; Moseman-Valtierra et al., 2016). In bare areas, we saw small rates of photosynthetic uptake presumably due to the presence of photosynthetic microalgae (macroalgae was typically not present), but uptake values were outweighed by respiratory emissions, suggesting that BP placed on the marsh platform can cause C remineralization. From ponds, we saw small but variable CO₂ exchange values through the growing season. Because gas exchange from wetland ponds has rarely been measured (Moseman-Valtierra et al., 2016), and no relationships were found with environmental variables measured, it is difficult to attribute GHG exchange in ponds to specific factors measured in this study. Shallow ponds in salt marsh landscapes have been described as very heterogeneous in terms of their gas exchange by Spivak et al. (2017). Our data agree with the notion that these semi-isolated ponds either import or export organic material (Spivak et al., 2017). Therefore, ponds in wetland landscapes—either natural or anthropogenic—should not be included in C inventory calculations until properly researched, as these ponded features are still understudied in terms of C assimilation (Spivak et al., 2017; Berkowitz et al., 2018).

Although very episodic and highly variable, our study found generally higher rates of methane emissions from live vegetation than from ponds or BP. These values were nearly identical to past literature estimates of mean methane emissions by salt marshes (0.53 C/year from marsh, 0.33 C/year from BP, and 0.32...
C/year from ponds vs. 0.46 g C/year from mean literature values; Kroeger et al., 2017). Given the high salinity, these values did not significantly alter the global warming potential of the wetlands (Poffenbarger et al., 2011; Kroeger et al., 2017). However, this significant difference suggests that transport of methane to the atmosphere is mediated by the belowground structures of marsh vegetation and that this factor outweighs the higher water table relative to the sediment surface found in ponds (Colmer, 2003; Van Der Nat & Middelburg, 1998).

4.3. Scaling

The predictive model of annual CO₂ exchange from Barnegat Bay marshes estimated that the gross primary production (GPP) in vegetated marsh was equal to 726 g C · m⁻² · year⁻¹, of which 689 g was returned to the atmosphere via respiration. While previous measures of peak biomass (both above and belowground) at Barnegat Bay have summed to 840–880 g/m², of which we can estimate only 40% is C (Osgood & Zieman, 1993), studies suggest that the ratio of peak biomass to net annual aboveground production for Spartina spp. may range from about 1 to 5, as grasses continuously gain and lose leaves (Linthurst & Reimold, 1978; Valiela et al., 1975). These peak biomass values would thus infer rates of GPP equal to 340–1,600 g C/m², broadly in concordance with the GPP (726 g C/m²) we estimated from our scaling function.

The difference between GPP and ecosystem respiration yielded a net C assimilation estimate of 41.7 g C · m⁻² · year⁻¹, while in contrast, bare ground and P habitats were estimated to release 125.0 and 44.4 g C · m⁻² · year⁻¹, respectively. Accounting for the total extent of these three habitats in Barnegat Bay, we can project a net uptake of 3,020 t C/year.

The estimated error of both PLSR models is large relative to the total flux values (5.1 for GPP and 3.6 for ER), suggesting a poor robustness and model accuracy. This behavior can be expected for the comprehensive modelling of in situ observations with high spatio-temporal heterogeneity across seasons (Collier et al., 2014; Rosenstock et al., 2016) Nevertheless, we regard the model predictions to be realistic values when compared to other studies (Magenheimer et al., 1996; Morris & Whiting, 1986) and representative of a diverging trend in C fluxes between the three habitat types. Therefore, the model outcome allows us to estimate the GHG balance of Barnegat Bay without disturbances. We estimate that the yearly C uptake by the system would be 3,500 t C/year without OMWM ponds, which is 129 t C more than with ponds. In addition, the carbon emissions resulting from the remineralization of organic matter from bare peat also results from OMWM construction. If the corresponding area would still be IM, the total uptake would be 3,570 t C/year. We infer that OMWM practices led to a 5.6% reduction in C uptake by Barnegat Bay marshes in 2017. However, this estimate could be an overestimate as the preexisting conditions of the marsh were not evaluated, and as this study shows, the wetland landscape has varying levels of C assimilation.

While not considered by our study, two other sources of C exchange are clearly relevant for understanding the effects of disturbance on C sequestration: (1) lateral transfer of C in various form (dissolved inorganic carbon, dissolved organic carbon, and particulate organic carbon) via tidal advection, which likely causes net export of C, and (2) deposition of sediment containing allochthonous organic C, which likely contributes to C sequestration. Although lateral exchange of C from wetlands has been well studied across the eastern United States, observed values have varied quite widely from 5 to 500 g C · m⁻² · year⁻¹ (Mulholland, 2003; Shih et al., 2010; Herrmann et al., 2015), making it difficult to constrain how lateral exchange of dissolved and particulate C may impact landscape scale estimates of C sequestration. Previous studies have however focused on sediment deposition across marsh landscapes, including the consideration of pools as well as sediment sidecasts from OMWM pond construction (Elsey-Quirk & Adamowicz, 2016). A recent study focusing specifically on sediment accretion in areas disturbed by sidecasts has concluded that raising the marsh by 10 cm through these sidecasts decreased sediment deposition nearly fivefold, presumably due to the reduction in inundation time (Elsey-Quirk & Adamowicz, 2016). Comparison of sediment accumulation rates in pools vs. on the marsh platform has suggested alternatively that pools accrete more quickly (Erwin et al., 2006; Wilson et al., 2010), that accretion rates are similar (Morton et al., 2009), or that accretion rates are lower (Watson, 2011) than on the marsh platform. These various findings are related to the two competing drivers of sediment accumulation: inundation time, which tends to increase sediment deposition as there is more time for sediment to settle out vs. the sediment trapping and stabilizing functions of vegetation, which ponds lack. Considered jointly, we propose that pools and bare areas are both exporting stored C
latterly as well as supporting little C accumulation via sediment deposition and that the findings of our study—that disturbance decreases C storage at rates that exceed the disturbance footprint—would hold true if lateral C transport pathways were also explicitly considered.

5. Conclusions

Our work has addressed how wetland fragmentation or the replacement of marsh vegetation with open water—which occurs due to coastal development as well as due to sea level rise—is impacting the potential C assimilation of salt marshes. Our work suggests that to have accurate measurements of the C sequestration in our salt marshes for climate change mitigation, we first need have an accurate idea of the actual spatial extent of productive plants, because as our data show, open water and bare ground may not contribute to GHG assimilation. As policy moves toward expanding coastal wetland restoration to C markets, we first need to determine the actual area of productive plants—as they are the link to mitigate against anthropogenic C emissions and the resulting of climate forcing.

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