Bulletin of the Southern California Academy of Sciences

Volume 111 | Issue 2 Article 5

8-1-2012

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Cover Page Footnote
Acknowledgements We thank the School of Earth and Environmental Sciences within the Schmid College of Science and Technology at Chapman University for funding this project as the laboratory component of the Fall 2011 Ecosystems Ecology course. Kody Cabreros, Lauren Cruz, Jessica Jung, and Elizabeth Malcolm contributed to the field and laboratory aspects of this project. Dr. J. Patrick Megonigal at the Smithsonian Environmental Research Center generously provided the static chambers used to measure net ecosystem respiration. The Board of the Huntington Beach Wetlands Conservancy under the leadership of Dr. Gordon Smith provided access to field sites and valuable insights into the history and ecology of these ecosystems. Comments from 2 anonymous reviewers greatly improved this manuscript. Drs. Jason Keller and Kimberly Takagi were supported by a grant from the National Science Foundation (DEB#0816743).

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Bull. Southern California Acad. Sci.
111(2), 2012, pp. 153–161

Soil Organic Carbon Storage in Restored Salt Marshes in Huntington Beach, California

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Abstract.—There is a growing interest in managing wetland restoration and conservation projects to maximize carbon sequestration. We measured soil organic carbon storage and methane flux from two southern California salt marshes which had been restored for 2 and 22 years. We hypothesized that organic carbon would accumulate following restoration and that methane flux would be negligible in both sites. While methane flux was minimal, soil organic carbon content was generally higher in the more recently restored site. Although there is a potential for carbon sequestration in salt marshes, tracking this process through time may be complicated by initial site conditions.

Introduction

As a result of anthropogenic activities, atmospheric concentrations of carbon dioxide ($[CO_2]$) have increased from pre-industrial levels of ~280 ppm to current levels exceeding 379 ppm (Forster et al. 2007), the highest values seen for at least the past 650,000 years (Jansen et al. 2007). This rise in atmospheric $[CO_2]$ has led to an acceleration of sea level rise as well as changes in ecosystem carbon cycling (Denman et al. 2007). Among the many options for mitigating these impacts is the capture and storage of atmospheric CO_2 in long-lived carbon pools (Metz et al. 2007).

Long-term carbon storage in ecosystems is possible through the accumulation of biomass or soil organic matter. Wetland ecosystems may be particularly well suited for carbon storage because a lack of oxygen in flooded or saturated soils imposes a fundamental constraint on microbial decomposition, frequently resulting in the accumulation of soil organic matter (Megonigal et al. 2004). Indeed, wetland ecosystems have accumulated ~500 Pg of carbon in their soils, approximately one-third of the total terrestrial soil carbon on a global scale (Bridgham et al. 2006).

There is thus a growing interest in linking the carbon storage potential of wetland ecosystems to ongoing management efforts through the sale of carbon credits on emerging carbon markets (e.g., Galatowitsch 2009; Hansen 2009). Given the continued loss of wetlands from the landscape (Dahl 2011) and the high cost of wetland mitigations (Environmental Law Institute 2007), the potential opportunity for wetland projects to become partially "self-funded" is an intriguing possibility. The prevalence of ecosystem-based biosequestration projects on existing, voluntary carbon markets (Galatowitsch 2009) further highlights the appeal of this approach. Presumably, carbon credits could be awarded to a number of project types, including: (i) wetland restoration or creation projects that result in an increase in soil carbon and (ii) wetland conservation projects

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that result in the preservation of existing soil carbon pools. Crediting these project types implicitly assumes that carbon accumulates through time following restoration and that wetland degradation leads to a loss of soil carbon stocks. However, it should be noted that there are a number of policy and scientific uncertainties surrounding this approach (Galatowitsch 2009; Hansen 2009; Palmer and Filoso 2009; Crooks et al. 2010).

The potential to sequester carbon in vegetated coastal ecosystems (i.e., "blue carbon"), including salt marshes, is particularly appealing in this context for several reasons (Irving et al. 2011; Mcleod et al. 2011). First, salt marsh ecosystems must accrete new soil to keep pace with sea level rise (Mudd et al. 2009; Kirwan et al. 2010). This results in a continual increase in the volume of soil where organic carbon can be sequestered and buries existing soil at deeper depths where persistent anaerobic conditions limit further decomposition. In a recent review, Chmura et al. (2003) estimated a carbon sequestration rate of 210 g CO_2 m^{-2} y^{-1} for tidal saline wetlands including salt marshes, a per area value which is an order of magnitude higher than sequestration rates in northern peatland ecosystems. Second, the presence of sulfate in sea water is assumed to suppress methane (CH_4) production due to the dominance of competitively superior sulfate reducing microbes (Megonigal et al. 2004; Poffenbarger et al. 2011) in salt marsh ecosystems. This is critical because CH_4 is a potent greenhouse gas with 25-times the global warming potential of CO_2 (Forster et al. 2007), and has been shown to offset the decreased radiative forcing due to carbon sequestration in freshwater wetland environments (e.g., Bastviken et al. 2011). While it is generally assumed that CH_4 fluxes are negligible in salt marsh ecosystems, these fluxes have rarely been quantified (but see Chmura et al. 2011; Poffenbarger et al. 2011).

Here, we measure the soil organic carbon storage and CH_4 flux in two adjacent salt marshes in southern California which have been restored for 2 and 22 years. We hypothesized that (i) the soil organic carbon pool would be larger in the older restoration site and (ii) CH_4 fluxes would be minimal at both sites. This work provides an important baseline for evaluating the potential for carbon sequestration in similar ecosystems and is unique for salt marshes in Southern California, where soil carbon storage has not been as well studied as in other locations such as the East and Gulf Coasts and the San Francisco Bay region.

Methods

Site Description

The Huntington Beach Wetlands utilized in this study are located in northern Orange County, California (Figure 1). These sites are relics of a larger, approximately 3000-acre, marsh complex that historically existed at the mouth of the Santa Ana River but has decreased in size as a result of anthropogenic activities (Grossinger et al. 2011). By the mid-1940s, these systems had been isolated from tidal exchange due to channelization of the Santa Ana River and flood control measures associated with local development as well as the construction of the adjacent Pacific Coast Highway (Jones & Stokes Associates 1997).

The current restoration of the 25-acre Talbert Marsh began in 1988, following a short-term, temporary reconnection to tidal exchange in 1979. Tidal flow was fully restored to Talbert in February 1989 following the removal of a levee from an adjacent flood control channel (the site remains disconnected from the Santa Ana River mouth). In 1991, a new tidal outlet was constructed and the old channel was closed (Jones & Stokes Associates 1997).

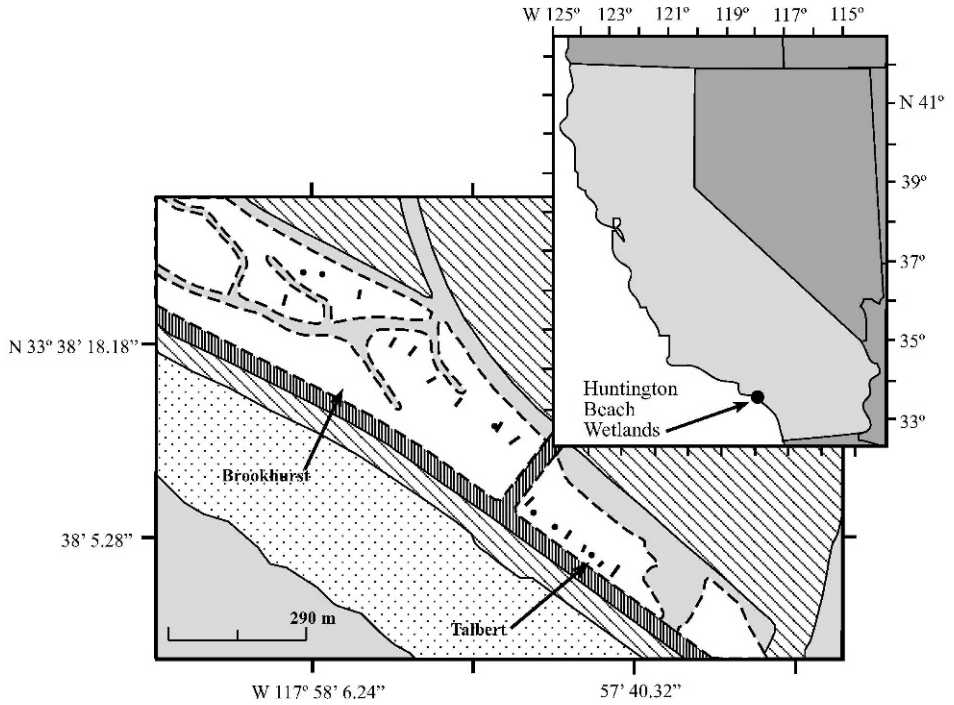

Fig. 1. Map of Talbert Marsh and Brookhurst Marsh in Huntington Beach, California. Talbert Marsh was restored in 1989 and Brookhust Marsh was restored in 2009. Marshes boundaries are indicated by dashed lines. Soil organic carbon was measured in 3 soil cores collected in each marsh (soil black circles). Net ecosystem respiration, including net CH_4 flux, was measured using 2 static chambers adjacent to each soil core. Belowground biomass was measured in 3 evenly-spaced root cores collected along existing vegetation monitoring transects (solid black lines) within each marsh.

Restoration of the adjacent, 67-acre Brookhurst Marsh began in 2008 and tidal flow was restored in July 2009 following the removal of a levee from the flood control channel (Gordon Smith, Huntington Beach Wetlands Conservancy, personal communication). Sites utilized for the current project were located in the mid/high marsh which is dominated by vegetation characteristic of southern California salt marsh ecosystems, including: *Sarcocornia pacifica* (Standl.) A.J. Scott (perennial pickleweed), *Frankenia salina* (Molina) I.M. Johnst (alkali seaheath), *Batis maritima* L. (saltwort), and *Distichlis spicata* (L.) Greene (saltgrass).

Soil Organic Carbon

To quantify soil organic carbon content, 3 soil cores were collected from both Talbert and Brookhurst Marshes in September-October of 2011 (Figure 1). Cores were collected to depths of ~42 cm (Talbert) and ~46 cm (Brookhurst) using a 15.3-cm diameter stainless steel tube equipped with a sharpened bottom edge. Despite the twisting and downward pressure required to collect soil cores, compaction of soils was minimal and depths to the soil surface measured from the inside and outside of the soil core were within 1.5 cm. Upon extraction from the core, soils were sliced into 2-cm depth increments using a serrated knife and returned to the laboratory at Chapman University for processing. Depth increments were dried to a constant mass at 60 °C for 2 weeks and then weighed to determine bulk density (g dry mass cm^{-3}). Subsamples of dried soils

from each depth were homogenized using a mortar and pestle and subsequently ground to a fine powder using coffee grinders. Organic matter content of ground soils was determined as loss on ignition (LOI) at 400 °C for 10 hours and converted to organic carbon content on a percent mass basis using the quadratic relationship described by Craft et al. (1991).

Net Ecosystem Respiration

Net ecosystem respiration (NER) was measured as the net flux of CO_2 and CH_4 using ~15 L dark, static chambers constructed from 29.8-cm diameter sewer PVC pipe. The bottom of the chambers had a sharpened, beveled edge and the top was routed and lined with a 2.4-mm diameter o-ring. NER was measured concomitantly with soil core collection using 2 chambers adjacent to each soil core (i.e., 6 flux measurements from each site). For each measurement, chambers were inserted to a depth of 2–3 cm below the soil surface and capped with a thick PVC cover equipped with a two-way stopcock to allow for gas sampling. Chamber covers were painted silver to minimize heating effects. Five headspace samples were collected at 10–15 min intervals using 10-mL syringes (also equipped with stopcocks). Samples were analyzed on the same day they were collected for CO_2 and CH_4 using a gas chromatograph equipped with a flame ionization detector and an in-line methanizer to convert CO_2 to CH_4 (SRI 8610C, SRI Instruments, Torrance, CA).

NER is expressed as μmol CO_2 m^{-2} min^{-1} based on the linear ($r^2>0.90$) accumulation of CO_2 over the measurement period. Because chambers enclosed intact vegetation, CO_2 produced was the combination of heterotrophic microbial respiration as well as autotrophic plant and algal respiration. Gas concentrations were corrected for the actual volume of the static chambers based upon measurements of the depth from the top of the chamber to the soil surface and for air temperatures obtained from a nearby weather station (http://www.wunderground.com/weatherstation/WXDailyHistory.asp?ID=KCAHUNTI3). Two chambers from Talbert and one chamber from Brookhurst which did not exhibit a linear accumulation of CO_2 (suggesting a faulty chamber seal) were excluded from subsequent analysis. We did not observe linear accumulation (or decline) of CH_4 in any chamber.

Belowground Biomass

Belowground biomass was measured using root cores collected using 5.08-cm diameter aluminum cores to a depth of 20 cm. Cores were collected along transects previously established for monitoring vegetation cover in both sites (Christine Whitcraft et al., California State University, Long Beach, unpublished data). There were 5 transects in the mid/high marsh in Talbert and 9 in Brookhurst. Soil cores used to measure organic carbon content were bracketed by at least 2 adjacent transects (Figure 1). Root cores were stored at 4 °C for up to 10 d prior to analysis. Cores were subsequently washed over a 1 mm soil sieve and living roots were collected and dried at 60 °C for 48 h. Additionally, two belowground biomass cores were taken adjacent (within 2 m) to each of the soil cores. Belowground biomass was expressed as kg dry biomass per m^3 soil volume.

Statistical Analysis

Differences in soil organic carbon content between Talbert and Brookhurst were analyzed using a repeated measures ANOVA with depth as a repeated, within-subject factor and site as a fixed, between-subject factor. Across all depths, soil organic carbon content data were not normally distributed even following common transformations

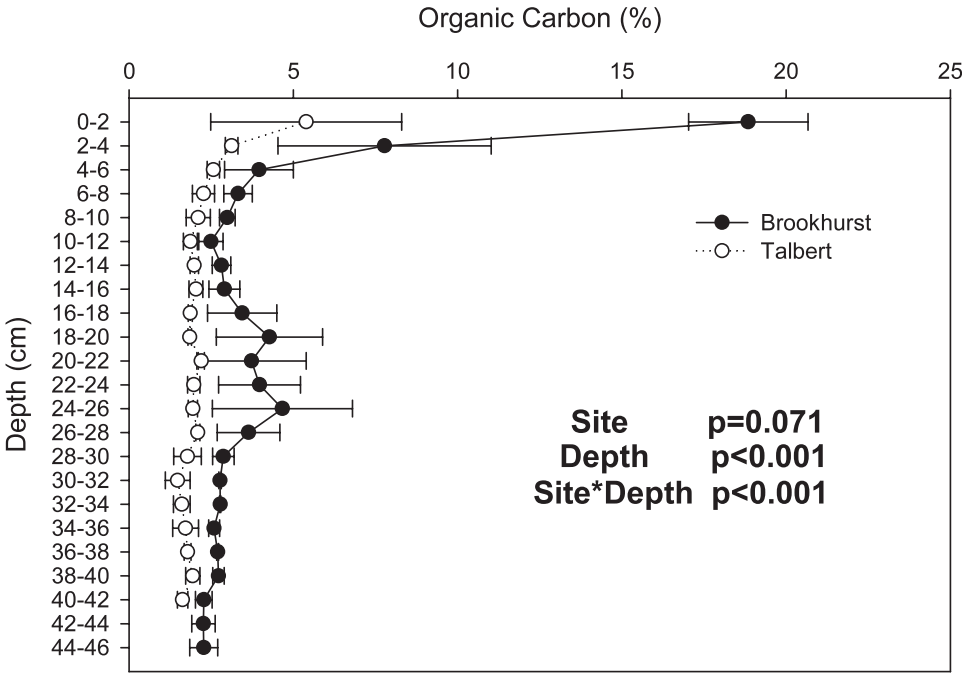

Fig. 2. Depth profiles of soil organic carbon content (mean ± 1 SE) in Talbert (restored in 1989) and Brookhurst (restored in 2009) Marshes. Results of a repeated measures ANOVA with site as a fixed factor and depth as a repeated factor are shown.

(e.g., arcsin square root and logarithmic transformations). The repeated measures ANOVA was performed on un-transformed data. We used t-tests to analyze differences in NER, belowground biomass, and total carbon storage between Talbert and Brookhurst. Belowground biomass data were logged-transformed to meet assumptions of normality prior to analysis. Linear regressions were used to explore potential relationships between belowground biomass (collected adjacent to soil cores) and cumulative soil carbon storage in the upper 10 and 40 cm of the soil profile. All analyses were completed using PASW Statistics 18 (SPSS 2009).

Results

Mean percent soil organic carbon was higher at all depths in the more recently restored Brookhurst Marsh than in Talbert Marsh (Figure 2). However, differences between sites varied by depth (depth * site interaction $F(20,80)=6.338$, $p<0.001$; Figure 2). In both sites, the highest organic carbon content was seen in the surface 0–2 cm with 18.8 ± 1.8% and 5.4 ± 2.9% (mean ± 1 SE) in Brookhurst and Talbert, respectively. When corrected for bulk density, the total amount of organic carbon held in the upper 40 cm of the soil profile (mean ± 1 SE) did not differ between sites with 13.6 ± 0.3 kg carbon m^{-2} in Brookhurst and 9.3 ± 0.8 kg carbon m^{-2} in Talbert ($t(4)=1.58$, $p=0.19$).

NER as net CO_2 flux was similar between sites ($t(7)=0.18$, $p=0.87$) with values (mean ± 1 SE) of 113.5 ± 23.3 μmol CO_2 m^{-2} min^{-1} and 106.7 ± 32.2 μmol CO_2 m^{-2} min^{-1} in Brookhurst and Talbert, respectively. Given the lack of CH_4 accumulation in the chambers, net CH_4 flux was assumed to be negligible at both sites.

Belowground biomass (mean ± 1 SE) based on root cores to a depth of 20 cm was 7.0 ± 1.2 kg dry biomass m^{-3} in Brookhurst and 9.4 ± 3.2 kg dry biomass m^{-3} in Talbert, and did not differ between sites ($t(40)=0.25$, $p=0.81$). Across both sites, there were no relationships between belowground biomass (from root cores collected adjacent to the soil cores) and cumulative soil organic carbon storage in the upper 10-cm of the soil profile ($p=0.51$) or the upper 40-cm of the soil profile ($p=0.64$).

Discussion and Conclusions

Both Talbert and Brookhurst Marshes stored organic carbon in their soils. When averaged across all depths to 40 cm, organic carbon densities (± 1 SE) were 0.034 ± 0.006 and 0.023 ± 0.002 g cm^{-3} in Brookhurst and Talbert, respectively. These values are comparable to the 0.039 ± 0.003 g cm^{-3} soil carbon density of salt marsh soils reported by Chmura et al. (2003). Of the sites used by Chmura et al. to calculate this average, only 6 were from the Northeastern Pacific, including 3 data points from a single southern California marsh complex in Tijuana Slough, California (Chmura et al. 2003). The average soil carbon density from the Tijuana Slough site was 0.025 ± 0.008 g cm^{-3} which is comparable to the values measured in the current study.

In contrast to our initial hypothesis, mean percent organic carbon was higher at all depths in the more recently restored Brookhurst Marsh, although the magnitude of these differences depended on soil depth (Figure 2). This apparent lack of carbon accumulation through time differs from a number of studies which have shown that soil organic carbon content frequently increases following wetland restoration activities (e.g., Craft et al. 2003; Cornell et al. 2007). However, a recent meta-analysis suggests that soil carbon pools recover slowly following wetland restoration, and are generally well below values seen in reference marshes even many decades following restoration (Moreno-Mateos et al. 2012).

Streever (2000) cautioned against drawing too strong a conclusion from chronosequence approaches because the magnitude of inter-site differences in ecosystem properties are often greater than changes in those properties observed through time. We hypothesize that this was the case in the Huntington Beach wetlands used in this study, and that differences in soil organic carbon content were driven by initial site conditions in Talbert and Brookhurst, rather than by processes occurring during the additional 20 years of post-restoration development at Talbert. Pre-restoration vegetation surveys along 2 transects in Talbert Marsh suggest that total vegetation cover was less than 40% (Jones & Stokes Associates 1997), much lower than pre-restoration cover values of 88% measured along the 9 transects used to measure belowground biomass in Brookhurst Marsh (Christine Whitcraft et al., unpublished data). The lack of differences in standing belowground biomass and NER between sites despite a 20-year difference in age also suggests that carbon cycling may be similar at both sites despite their different ages.

Further, Maezumi (2010) collected a single core from Brookhurst prior to marsh restoration in 2008 and observed organic carbon content of ~9% in the surface 0–1 cm compared to a value of ~7% in the surface soil of a single core collected from Talbert in the same year. These values differ from the average organic carbon contents measured in our study (~19% and ~5% in the surface 2 cm for Brookhurst and Talbert, respectively); however, they suggest that higher organic carbon content in Brookhurst soil may have existed prior to restoration of this site. This highlights the importance of collecting baseline, pre-restoration data if increased carbon sequestration is a goal of wetland management activities. The presence of elevated soil organic carbon in Brookhurst despite being disconnected from tidal influence for nearly a century may also call into

question the assumption that soil carbon is rapidly lost following the introduction of aerobic soil conditions in wetland environments, and may have important implications for wetland conservation projects that hope to capitalize on maintaining existing soil carbon pools.

Our NER measurements suggest that net CH_4 flux was not significant during our samplings. The lack of significant CH_4 fluxes from this salt marsh site is consistent with the limited previous work on this topic. Poffenbarger et al. (2011) suggest that at soil salinities above 18 ppt, which are common at Brookhurst and Talbert, CH_4 flux is minimal due to competitive suppression by sulfate reducing microbial activities. Chmura et al. (2011) also demonstrated that small, but measurable, end of season fluxes of the greenhouse gases CH_4 and nitrous oxide (N_2O) did not offset CO_2 uptake and storage in Canadian salt marshes. While our data support the assertion that salt marshes may be ideal sites for carbon sequestration due to a lack of substantial CH_4 fluxes, we cannot rule out the possibility of measurable CH_4 fluxes from more brackish portions of these sites or following major rain events which can bring large amounts of freshwater into these systems. Previous work on soils from salt marshes in southern California demonstrates that CH_4 production is possible in anaerobic laboratory incubations within a few days (Jason Keller, unpublished data) suggesting a potential for this metabolic process in these soils.

Storage of organic carbon through the soil profile and a lack of CH_4 fluxes in two restored southern California salt marshes reinforce the potential for these systems to be managed to maximize carbon sequestration. Soil organic carbon content in these systems is comparable to marshes in other, better-studied, geographical regions. However, higher soil carbon content in the more recently restored Brookhurst Marsh demonstrate that soil organic carbon may persist even in the absence of tidal connectivity in these systems and highlight the importance of initial site conditions in tracking soil carbon storage in restoration projects. A lack of significant CH_4 fluxes from both sites is consistent with previous research; however, additional work is necessary to quantify potential spatial and temporal variability in this important process.

Acknowledgements

We thank the School of Earth and Environmental Sciences within the Schmid College of Science and Technology at Chapman University for funding this project as the laboratory component of the Fall 2011 Ecosystems Ecology course. Kody Cabreros, Lauren Cruz, Jessica Jung, and Elizabeth Malcolm contributed to the field and laboratory aspects of this project. Dr. J. Patrick Megonigal at the Smithsonian Environmental Research Center generously provided the static chambers used to measure net ecosystem respiration. The Board of the Huntington Beach Wetlands Conservancy under the leadership of Dr. Gordon Smith provided access to field sites and valuable insights into the history and ecology of these ecosystems. Comments from 2 anonymous reviewers greatly improved this manuscript. Drs. Jason Keller and Kimberly Takagi were supported by a grant from the National Science Foundation (DEB#0816743).

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