Analysis of Organic Matter Decomposition in the Salt Marshes of the Venice Lagoon (Italy) Using Standard Litter Bags

A. Puppin[1] , **M. Roner[1]**, **A. Finotello[1]** , **M. Ghinassi[1]**, **L. Tommasini[1]**, **M. Marani[2]** , and **A. D'Alpaos[1]**

[1]Department of Geosciences, University of Padova, Padova, Italy, [2]Department of Civil, Environmental and Architectural Engineering, University of Padova, Padova, Italy

Abstract Tidal salt marshes are widespread along the World's coasts, and are ecologically and economically important as they provide several valuable ecosystem services. In particular, their significant primary production, coupled with sustained vertical accretion rates, enables marshes to sequester and store large amounts of organic carbon and makes them one of the most carbon-rich ecosystems on Earth. Organic carbon accumulation results from the balance between inputs, that is, organic matter produced by local plants or imported, and outputs through decomposition and erosion. Additionally, organic matter deposition actively contributes to marsh vertical accretion, thus critically affecting the resilience of marsh ecosystems to rising relative sea levels. A better understanding of organic-matter dynamics in salt marshes is key to address salt-marsh conservation issues and to elucidate marsh importance within the global carbon cycle. Toward this goal, we empirically derived rates of organic matter decomposition by burying 712 commercially available tea bags at different marshes in the microtidal Venice Lagoon (Italy), and by analyzing them following the Tea Bag Index protocol. We find values of the decomposition rate (k) and stabilization factor (S) equal to 0.012 ± 0.003 days^{-1} and 0.15 ± 0.063, respectively. Water temperature critically affects organic matter decomposition, enhancing decomposition rates by 8% per °C on average. We argue that, at least in the short term, the amount of undecomposed organic matter that actively contributes to carbon sequestration and marsh vertical accretion strongly depends on the initial organic matter quality, which is a function of marsh and vegetation characteristics.

Plain Language Summary Salt marshes are important coastal environments regularly flooded by the tide and dominated by herbaceous plants, providing several valuable ecosystem services. They are, however, threatened by the effects of climate changes and human interferences. As organic matter accumulated in salt-marsh soil importantly contribute to surface elevation necessary for marshes to keep up with sea level rise and to store atmospheric carbon, this project aims to improve our understanding of decomposition processes affecting organic matter preservation and their controls in salt-marsh environment. Toward this goal, following the so-called Tea Bag Index protocol, we buried 712 commercially available tea bags in salt-marsh soils of the Venice Lagoon (Italy) measuring the reduction of their organic content due to decomposition processes after 3 months. Our results confirm that salt marshes are among biomes with the slowest decomposition rates. However, we observed a loss of about two-thirds of the initial labile organic mass after 90 days and that initial litter quality, depending on litter and vegetation characteristics, exerts a primary control on the amount of preserved organic matter contributing to carbon sequestration and marsh accretion.

1. Introduction

Salt marshes are transitional ecosystems found between permanently submerged and emerged coastal environments. They are the prevailing landscape in the intertidal zone of many low-energy temperate coasts (Adam, 1990; Allen & Pye, 1992; Mcowen et al., 2017), with vegetation dominated mostly by herbaceous halophytes adapted to regular inundation by saltwater (Perillo et al., 2009).

Salt marshes are crucial ecomorphodynamic structures in tidal environments as they offer valuable ecosystem services (Barbier et al., 2011) by attenuating waves (Möller et al., 2014; Temmerman et al., 2013), acting as sediment trapping zones (Allen, 2000; Mudd et al., 2009), boosting the production of economically and ecologically important fishery species (Boesch & Turner, 1984; MacKenzie & Dionne, 2008), and filtering both nutrients and pollutants (Costanza et al., 1997).

In addition, marshes serve as highly efficient sinks for blue carbon, that is, the organic Carbon (C) captured and stored by coastal ecosystems and oceans (Chmura et al., 2003; Duarte et al., 2005; Macreadie et al., 2019; McLeod et al., 2011). Such a potential for carbon sequestration derives from the high rates characterizing marsh primary production, coupled with sustained vertical accretion rates driven by the deposition of both Organic Matter (OM) and clastic sediments (e.g., A. D'Alpaos et al., 2007; Kirwan & Murray, 2007; Marani et al., 2007; Morris et al., 2002; Mudd et al., 2009), as well as from the fact that carbon storage occurs predominantly in belowground biomass and that soil is typically in anaerobic conditions (McLeod et al., 2011; Ouyang & Lee, 2020; Rogers et al., 2019; Roner et al., 2016; Wang et al., 2019). This allows marshes to sustain some of the highest rates of carbon sequestration per unit area of all ecosystems, making them an important player in the global carbon cycle and in strategies aimed at reducing CO_2 emissions (Macreadie et al., 2019; McLeod et al., 2011).

Despite their importance, coastal ecosystems are some of the most heavily used and threatened natural systems globally, experiencing widespread loss or degradation (Barbier et al., 2011; Duarte et al., 2005; Valiela et al., 2001). Marsh ecosystems are seriously exposed to the effects of climate changes and human interferences, sea level rise and lack of available sediments being the key factors in determining salt-marsh drowning and disappearance (A. D'Alpaos et al., 2011; Kirwan et al., 2010; Marani et al., 2007; Morris et al., 2002; Mudd, 2011; Tognin et al., 2021, 2022). Therefore, understanding of the processes regulating OM accumulation and C sequestration in salt marshes is critical to better estimate their resilience to climate change, as well as their contribution to the global carbon cycle, providing valuable information to improve management and conservation strategies (Cahoon & Guntenspergen, 2010; Couvillion et al., 2013; C. D'Alpaos & D'Alpaos, 2021; McLeod et al., 2011; Morris et al., 2016; Mueller et al., 2019; Nyman et al., 2006; Pendleton et al., 2012; Turner et al., 2002). However, the exact dimensions of the salt-marsh role on the global carbon cycle and of their contribution to offset CO_2 emissions, are still uncertain, also because of our current limited knowledge on the decomposition of organic matter in marsh soil and on how decomposition rates can be affected by climate changes and human disturbances (Djukic et al., 2018; Macreadie et al., 2019; Sifleet et al., 2011). This work deals with this important issue.

More in detail, salt-marsh blue carbon is sequestered over the short term in living biomass aboveground (leaves, stems, branches) and belowground (roots) and over longer time scales within soil organic matter (SOM) in sediments (McLeod et al., 2011). Local plant biomass and the OM imported by tidal currents and waves (Tognin et al., 2021) are the main sources of the SOM in marshes (Morris et al., 2016; Yousefi Lalimi et al., 2018). Organic inputs, together with inorganic sediments deposited over the marsh platform by tidal flooding and waves and captured by vegetation, are the main processes contributing to marsh accretion and forming marsh soil where carbon is stored (e.g., Kirwan et al., 2013; Mudd et al., 2009; Roner et al., 2016). Accumulation of SOM is strongly influenced by decomposition processes, which determine the fraction of the OM produced or imported that effectively contributes to soil formation and carbon burial (Kirwan et al., 2013; Mudd et al., 2009; Rybczyk & Cahoon, 2002).

Decomposition of organic matter is the process of physical breakdown and biochemical transformation of complex organic molecules which compose dead OM (litter) into simpler organic and inorganic molecules. Specifically, decomposition consists of three concurrent processes, namely, communition (or fragmentation), leaching of water-soluble compounds, and microbial catabolism (Sanderman & Amundson, 2013). Decomposition results in the release of CO_2 and other organic gases (e.g., methane, nitrogen, sulfides), water, mineral nutrients, and energy, playing a fundamental role in biogeochemical cycles (Pavao-Zuckerman, 2008). Decomposition is controlled by the interactions between the physical environment, the quality of the substrate, and the community of decomposer organisms, and it is therefore characterized by widely varying rates and extents. Organic matter is composed of a complex mixture of biopolymers. Some of these compounds, such as proteins, carbohydrates, and lipids are easily degraded by microorganisms (i.e., labile), while other compounds, such as lignin and hemicellulose, are resistant to decomposition (i.e., recalcitrant) (Megonigal et al., 2004). Given enough time and the proper environmental conditions, most naturally occurring labile compounds can be completely mineralized to inorganic forms, except for a fraction of the organic residues, which is transformed into stable SOM (i.e., stabilization; Sanderman & Amundson, 2013).

Studies on litter decay have often emphasized the importance of climate on decomposition processes (Sanderman & Amundson, 2013), and temperature has long been thought to be an important control of decay rates in wetlands (Montagna & Ruber, 1980; White et al., 1978). Recent experimental results on a global scale suggest that the decomposition rate and the stabilization factor are sensitive to temperature, the first increasing and the second

decreasing with warming (Keuskamp et al., 2013; Mueller et al., 2018). The results of a short cellulose decay experiment, conducted along the East Coast of the United States, showed a significant relationship between mean daily temperature and decay rate equivalent to a 19% increase in mass loss for each degree warming (Kirwan et al., 2014).

Nevertheless, large variations in litter mass loss were observed within biomes, across marshes and across different sites (e.g., Kirwan et al., 2014; Morris et al., 2002; Mueller et al., 2018; Reents et al., 2021), suggesting that local-scale factors other than climate have strong controls on decomposition dynamics (Cornwell et al., 2008; Djukic et al., 2018). Soil characteristics, in terms, for example, of aeration and drainage, and flooding conditions (hydroperiod), which are influenced by local topography and tidal levels, affect both biomass production and decomposition rates (Kirwan et al., 2013; Marani, Belluco, et al., 2006; Mudd et al., 2009; Yousefi Lalimi et al., 2018). Several studies suggest that the relationship between the decomposition rate and elevation can be highly variable (Yousefi Lalimi et al., 2018). Halupa and Howes (1995), for instance, found that the rates of carbon loss measured in a New Jersey salt marsh through litterbag experiments and laboratory incubation were related to flooding frequency and duration. However, Kirwan et al. (2013) found an uncertain and weak relationship between decomposition rates and flooding duration in the Chesapeake Bay brackish tidal marshes. Mueller et al. (2018), who examined the results of standardized litterbag experiments on 30 tidal-wetland sites worldwide, found no relation between decomposition rates and elevation, but showed that the stabilization factor was reduced in low-elevation and more frequently flooded zones compared to high-elevation zones. Yousefi Lalimi et al. (2018) found no clear trends of decomposition rates and stabilization factors related to elevation and flooding along three transects on coastal marshes located in North Carolina (USA).

Furthermore, the characteristics of the organic matter itself are also known to strongly influence decomposition processes, and some authors indicate chemical and structural composition of the plant material, which can vary among marsh species (Scarton et al., 2002; Stagg et al., 2018), as the main driver for decomposition rates (Freschet et al., 2012; Lang et al., 2009; Wang et al., 2017). In addition, vegetation composition can influence both soil physicochemical conditions (e.g., altering the pH and water-holding capacity of soil) (e.g., Marani, Belluco, et al., 2006; Rydin & Jeglum, 2006; Ursino et al., 2004) and decomposer community characteristics (e.g., affecting nutrient inputs) (Ward et al., 2015).

Climate change and anthropogenic disturbances may importantly affect OM decomposition, altering its biotic and abiotic drivers. For instance, warming temperatures are indicated to accelerate decomposition (Keuskamp et al., 2013; Kirwan et al., 2014; Mueller et al., 2018), increased atmospheric CO_2 and rising sea level could increase net primary production (Nguyen et al., 2015; Ratliff et al., 2015), however, changing climate and flooding conditions have the potential to affect species distribution and nutrient cycles (Perillo et al., 2009). Thus, the effects of climate change may alter decomposition processes in wetland environments in ways that are still unclear. These uncertainties in unraveling the governing processes for OM decomposition call for further analyses.

Here we aim at providing new insights on OM fate in salt-marsh soils, clarifying which are the physical and biological factors that control OM decomposition, and how decomposition affects the amount of OM effectively contributing to soil formation and carbon burial in tidal marshes. Toward these goals, we utilized the Tea Bags Index (TBI) approach (Keuskamp et al., 2013) to estimate the rates of OM decomposition and stabilization in salt-marsh soils within the microtidal Venice Lagoon (Italy), relating them to different environmental conditions measured directly in the field. By contributing to a better understanding of OM fate in tidal marshes, our results will be useful to inform models describing marsh ecosystem functioning and evolution (e.g., A. D'Alpaos & Marani, 2016; Fagherazzi et al., 2012; Marani et al., 2010; Mudd et al., 2009). An improved representation of SOM dynamics in biomorphodynamic models may be fundamental in evaluating marsh resilience in the face of climate changes and anthropogenic disturbances. In addition, the description of SOM decomposition is a key element for assessing carbon sequestration potential of salt-marshes.

2. Materials and Methods

2.1. Study Site

The litterbag experiments were conducted in the Venice Lagoon (Figure 1), in the north-western Adriatic Sea (Italy). With an area of about 550 km^2 and an average water depth of 1.5 m, the Venice Lagoon is the largest lagoon in the Mediterranean and is characterized by a semidiurnal tidal regime with a mean tidal range of about 1.0 m. The lagoon is connected to the sea by three inlets: Lido, Malamocco, and Chioggia.
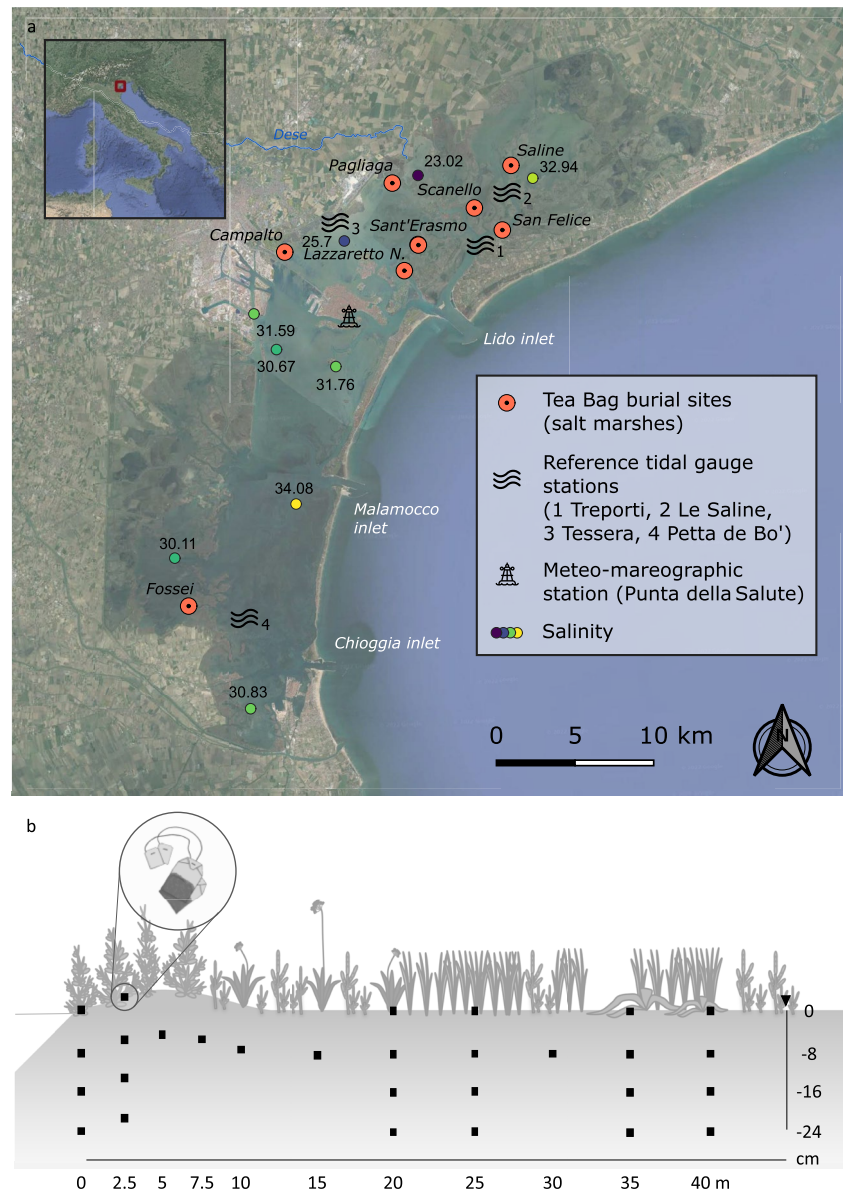
Figure 1. Study area and configuration of the burial experiment. Location of the study sites in the Venice lagoon, Italy (a), and a sketch of the scheme used for the burial experiments conducted along one of the analyzed marsh transects (b). Point 0 of each transect is sited at the edge of the marsh. Salinity indicates average values during the incubation periods from SAMANET network data.

Tea bags were deployed at 7 different marsh sites within the lagoon (Figure 1). Most of the studied marshes - namely S. Felice, Saline, Scanello, S. Erasmo, Lazzaretto Nuovo, Pagliaga and Campalto (see Figure 1) are located in the northern and most naturally-preserved part of the lagoon, and are found at variable distances from the Lido inlet, whereas the Fossei salt marsh is located in the southern lagoon (Figure 1).

The salt marshes in S. Felice, Saline, Scanello, S. Erasmo, and Lazzaretto Nuovo are adjacent to large tidal channels departing from the Lido inlet, and their origin has been ascribed to marine sediment inputs transported by flood tides (Bonometto, 2005; Silvestri et al., 2005). These marshes show typically concave-up profiles, with raised edges and lower elevations toward the inner portion of the marsh (Bonometto, 2005), and host exclusively halophytic vegetation, mainly constituted by *Salicornia veneta*, *Limonium narbonense*, and *Sarcocornia fruticosa*, associated with *Spartina maritima*, *Juncus maritimus*, *Puccinellia palustris*, *Inula crithmoides*, *Suaeda maritima*, and *Arthrocnemum macrostachyum* (Silvestri, 2000; Silvestri et al., 2005).

In contrast, the Pagliaga salt marsh is located at the landward boundary of the lagoon, close to the estuary of the springwater Dese River which debouches into the Lagoon carrying an average freshwater discharge of about 6.5 m³/s and negligible sediment supply (Figure 1). Freshwater inputs maintain a near-freshwater plant community dominated by *Phragmithes australis*, together with the halophytic species *Juncus maritimum*, *Salicornia veneta*, and *Halimione portulacoides* (Silvestri, 2000). Similarly, the Campalto salt marsh is found along the lagoon-mainland boundary, though it was originated on continental grounds of the coastal plains that were reached and permeated by brackish waters (Bonometto, 2005). The Campalto marsh hosts halophytic species dominated by *Limonium narbonense*, associated with *Sarcocornia fruticosa*, *Spartina maritima*, *Salicornia veneta*, and scarce *Suaeda maritima*.

Finally, the Fossei East and Fossei West salt marshes are located in the southern part of the Venice Lagoon, about 7 km NW of the Chioggia inlet, within a wetland area originated from pre-existing brackish environments or salinized freshwaters arising from river diversions (Bonometto, 2005; Roner et al., 2021). These marshes were artificially established between 1994 and 1996 to limit the lateral erosion of wind-exposed natural marshes, and they have been colonized by the typical halophytic vegetation species mainly constituted by *Salicornia veneta*, *Limonium narbonense*, *Sarcocornia fruticosa*, *Suaeda maritima*, and *Puccinellia palustris* (Cecconi et al., 1998).

2.2. Tea Bag Index

To estimate decomposition rates and organic matter stabilization in salt marsh soils, the Tea Bag Index (TBI) protocol proposed by Keuskamp et al. (2013) was adopted. The protocol consists of a simplified litter-bag experiment carried out utilizing standard litter in the form of commercially available tea bags (Lipton, Unilever). Two types of tea material with distinct qualities were used: green tea, with high cellulose content that is expected to undergo fast decomposition, and rooibos tea, characterized by high lignin content and expected to decompose slowly. Each tetrahedron-shaped synthetic tea bag contains about 2 g of tea and has a mesh size of 0.25 mm, allowing microorganisms and mesofauna to enter the bag and decompose the organic matter contained therein (Keuskamp et al., 2013).

Green and rooibos tea bags were buried pairwise in salt-marsh soils along linear transects and retrieved after about 90 days, with four transects having an incubation period <80 days (San Felice 1 and 2, Sant'Erasmo1 and 2 Jun 17–Sept 2 2015) and one transect >100 days (San Felice - Tavolini 24 Jun–11 Oct 2016) (see Table 1). Litter bag experiments were carried out between April and October, when mean temperature are usually above 10°C (see Figure S4 Supporting Information S1), which is the suggested threshold beyond which decomposition is inhibited (Prescott, 2010). We buried 712 tea bags in total (356 green tea; 356 rooibos tea) along transects oriented perpendicularly to the marsh margin (Table 1). According to Keuskamp et al. (2013), the standard burial depth corresponds to 8 cm below the marsh surface. However, at some sites, tea bags were also placed at different depths, corresponding to 0, 16, 24, and 40 cm relative to the marsh surface (Table 1) to analyze the effects of burial depths on decomposition dynamics. The tea bags were deployed between April and October in 2015, 2016, and 2017. In some cases, experiments were repeated in different periods along the same transect (Table 1).

Following Mueller et al. (2018) and Yousefi Lalimi et al. (2018), the initial weight of the tea bag content was determined by subtracting the mean weight of 10 empty bags (bag + string + label) (0.2459 and 0.2473 g, for green and rooibos tea respectively) from the weight of the intact tea bag prior to deployment (tea + bag + string + label) (the scientific protocol suggested by Keuskamp et al. (2013) available at http://www.teatime4science.org/wp-content/uploads/scientific.pdf recommends to determine the air dry weight of at least 5 empty bags). Once retrieved, the content of tea bags was extracted, oven-dried for 48 hr at 70°C, and weighted. To eliminate the effects of small amounts of inorganic sediments that could have entered the bags during the experiments, a Loss On Ignition procedure (LOI, by heating samples in a muffle oven at 550°C for 3 hr) was conducted on the contents of the retrieved bags after drying and recording the weights. The material remaining after combustion includes inorganic sediments that entered the bags after burial, as well as tea material that is not easily burned. We estimated this latter term by measuring the remaining material after the application of a LOI procedure to a group of 10 intact tea bags prior to deployment. We subtracted the weights of uncombusted material from the weights of the material remaining after combustion of the retrieved bags, to estimate the weight of inorganic sediment/salt that had entered each bag after burial. We finally used the weight of the extraneous mass to correct the weight of the tea bag organic content after incubation. The results were finally used to estimate the Tea Bag Index according to the methodology proposed by Keuskamp et al. (2013), that we briefly recall in the following.

The Tea Bag Index consists of two parameters describing the decomposition rate (k) and the litter stabilization factor (S). By using two distinct tea types with contrasting decomposability, it is possible to estimate k and S

Table 1
Burial Experiment Design

	2015			2016				2017		
	Apr 29 - Jul 29 23.75°C	Jun 11 - Sept 9 26.35°C	Jun 17 - Sept 2 26.55°C	Jun 24 - Sept 23 25.91°C	Jun 24 - Oct 11 25.07°C	Jun 30 - Sept 28 25.69°C	Jul 13 - Oct 11 24.76°C	Apr 12 - Jul 10 21.43°C	Apr 12 - Jul 19 21.91°C	Jul 10 - Oct 12 24.34°C
S. Felice - Spaccotralo	−8 cm									
S. Felice - Spaccotralo 1		−8 cm						−8, −16 cm		0, −8, −16, −24 cm
S. Felice - Spaccotralo 2			−8 cm	−8 cm				−8, −16 cm		0, −8, −16, −24 cm
S. Felice - Tavolini	−8 cm				−8 cm					
Saline				−8 cm				−8, −16 cm		0, −8, −16, −40 cm
Sant'Erasmo 1			−8 cm							
Sant'Erasmo 2			−8 cm							
Scanello		−8 cm								
Pagliaga		−8 cm								
Campalto									−8, −16 cm	
Lazzaretto Nuovo							−8 cm			
Fossei E						−8 cm				
Fossei W						−8 cm				

Note. Incubation sites (Figure 1) with relative incubation periods, mean water temperature during the incubation period, and burial depth.

using a single measurement in time. Specifically, the fractional weight, $W(t)$, of a buried bag content is assumed to change through time as a result of decomposition processes according to an exponential decay function with constant decomposition rate k:

$$W(t) = ae^{-kt} + (1 - a) \tag{1}$$

where t denotes time, and (a) and $(1 - a)$ are the labile and the recalcitrant fraction of the substrate, respectively. This approach assumes that decomposition of non-hydrolyzable materials is negligible for burial periods shorter than 3 months.

During decomposition, parts of the labile compounds stabilize and become recalcitrant (Prescott, 2010), resulting in a deviation of the actual decomposed fraction a (i.e., limit value) from the hydrolyzable fraction H (i.e., chemically labile). This deviation represents the stabilization factor S, which, following Keuskamp et al. (2013) can be computed as:

$$S = 1 - \frac{a}{H} \tag{2}$$

As to the hydrolyzable fractions of green and rooibos tea (H_g and H_r, respectively) we used values determined in Keuskamp et al. (2013) by sequential extraction of nonpolar extractives (NPE), water solubles (WS), acid solubles (AS), and acid insolubles (AIS). The hydrolyzable fractions for green and rooibos tea, defined as the sum of the NPE, WS, and AS fractions, were $H_g = 0.842 \pm 0.023$ g g^{-1} and $H_r = 0.552 \pm 0.050$ g g^{-1} (Keuskamp et al., 2013).

The calculation of both the decomposition rate (k) and the stabilization factor (S) is carried out following Keuskamp et al. (2013), who found that, after 3 months, the labile fraction of green tea, a_g, was almost completely decomposed (see Keuskamp et al. (2013) for the results of the laboratory incubation experiments with multiple harvests). Because after 3 months, the first term on the right-hand side of Equation 1 vanishes, the labile fraction of green tea, a_g, can be determined at the end of the incubation experiment. Using Equation 2, given a_g and considering the green tea hydrolyzable fraction H_g taken from Keuskamp et al. (2013) on the basis of chemical analysis, the stabilization factor S_g for green tea can be determined. According to Keuskamp et al. (2013), the value of S is assumed to be equal for both tea types, being dependent only on environmental conditions (i.e., $S_g = S_r$). We recall that this assumption was successfully tested by Keuskamp et al. (2013). Consequently, the labile fraction a_r of rooibos tea can be estimated using Equation 3, which reads

$$a_r = H_r(1 - S) \tag{3}$$

with $W(t)_r$ and a_r known, k is calculated using the exponential decay function given in Equation 1.

2.3. Environmental Parameters

In order to separate the role of different environmental conditions on the decomposition rate and stabilization factor, a number of environmental parameters were determined.

Mean water temperature (T) was determined for each incubation period by calculating the mean value of the water temperature measured every 10 min at Punta della Salute station by the CPSM - Centro Previsioni e Segnalazioni Maree of the Venice Municipality (for temperature data representativity see Figures S3 and S4 Supporting Information S1).

Elevation above Mean Sea Level (MSL) of each burial site was determined using two TOPCON GR-3 GPS receivers (dual frequency - L1/L2 - and dual constellation - NavStar/Glonass - with integrated Tx/Rx UHF radio). The real tea bag burial elevation (βz) was then calculated by subtracting the burial depth (ranging from 0 to 40 cm) to the elevation of the marsh surface.

The distance of each burial site from the edge of the nearest channel (ζ) was determined during field surveys. In order to collect information on flooding frequency and duration, based on the elevation of the burial sites, we calculated the percentage of the burial period during which each site was flooded by water (φ) (except for Sant'Erasmo, Lazzaretto Nuovo and Campalto transects, where elevation data were not available), comparing local marsh elevations with tidal levels measured at 10-min frequency by the nearest tidal gauge station of the CPSM - Centro Previsioni e Segnalazioni Maree monitoring network (Figure 1) (distances to nearest tide gauge are reported in Table S1 in Supporting Information S1, for water level data representativity see Figures S1 and S2 in Supporting Information S1).

2.4. Statistical Analyses

To test for the effects of environmental parameters on k and S, the Kendall correlation coefficient (τ) (Kendall, 1938) was determined. The Kendall test is a non-parametric, rank-based method, aimed at determining whether there exists a monotonic relationship between two variables of interest. The value of the τ coefficient ranges from 1 to −1, indicating a positive or a negative association, respectively. Because of the large number of ties in our study, a modified equation that considers the number of ties was used (Soliani, 2019; Sprent & Smeeton, 2000).

In addition, we also employed the Wilcoxon signed-rank test to analyze the difference between decomposition metrics measured at different depths. The Wilcoxon signed-rank test is a non-parametric statistical hypothesis test used to compare two populations when the observations are paired, and it is appropriate for a repeated measure design where the same variables are evaluated under two different conditions.

3. Results

3.1. Tea Bag Index Metrics for the Venice Lagoon and Organic Matter Mass Loss Results

Decomposition rates for the tea bag pair samples retrieved in the Venice Lagoon vary between 0.003 and 0.027 days^{-1}, with a median value of 0.012 ± 0.003 days^{-1} (Figures 2a, 2b), whereas the stabilization factors range between 0.016 and 0.544, their median value being equal to 0.15 ± 0.063 (Figures 2c, 2d).
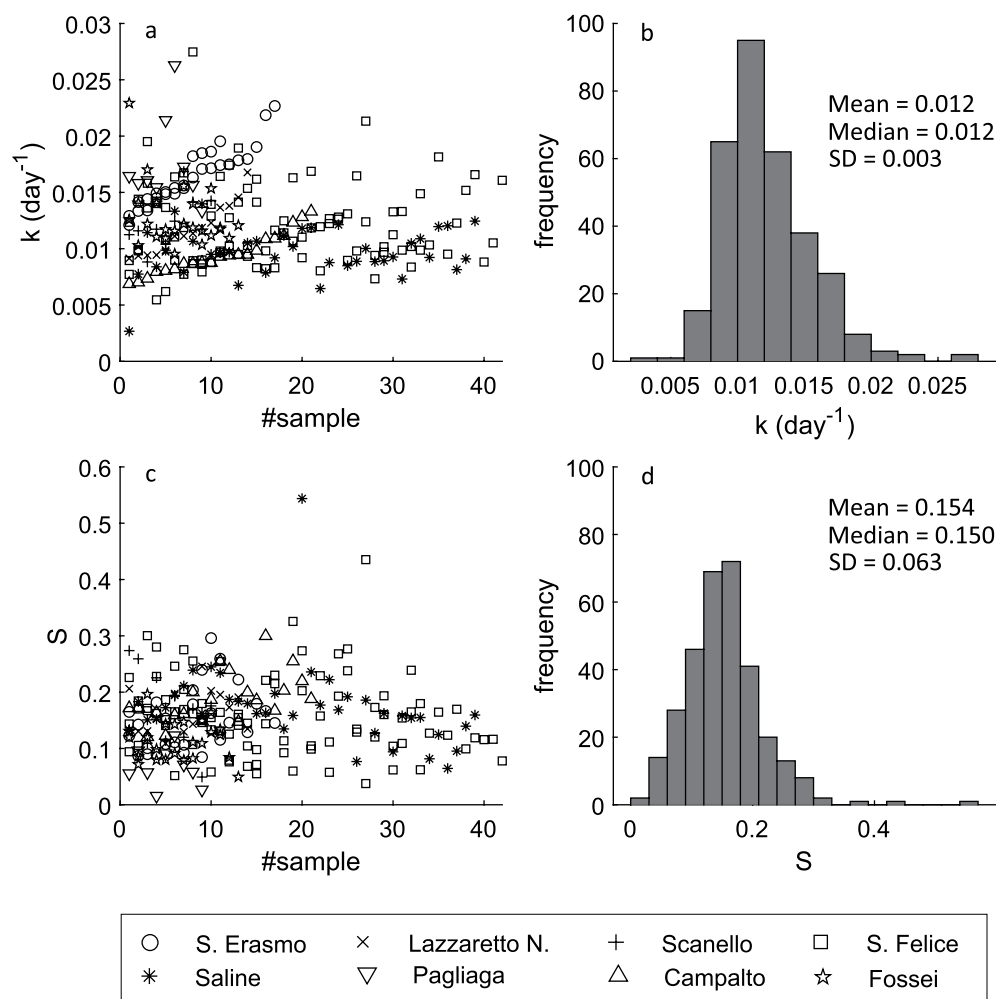

Figure 2. The Tea Bag Index metrics for the analyzed samples in the Venice Lagoon. (a) Decomposition rates (k) for all the samples; (b) Frequency distribution of the decomposition rates (k); (c) Stabilization factors (S) for all the samples; (d) Frequency distribution of the stabilization factors (S).
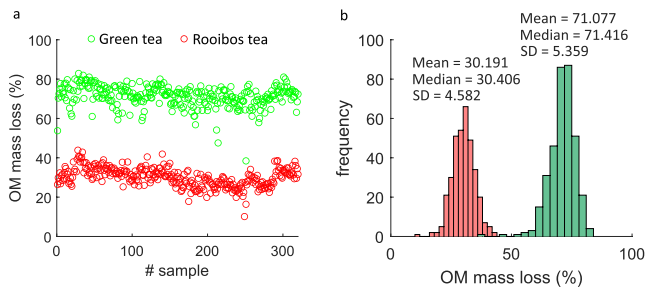
Figure 3. Organic matter mass loss results. Percent tea mass loss after 90 days of field incubation for the analyzed samples of green tea (green empty circles) and rooibos tea (red empty circles) are reported in panel a, with the related frequency distributions shown in panel b.

As expected, tea mass loss after 3 months of field incubation (Figure 3) was higher for green tea (71.08 ± 5.36%) than for rooibos tea (30.19 ± 4.58%).

To further inspect differences between green tea and rooibos tea decomposition, we also computed the decay coefficient k^* as suggested by Kirwan et al. (2014) and Kirwan and Blum (2011), who did not account for the proportion of recalcitrant and labile material ($k^* = \left[-ln(C_t/C_o)\right]/t$, where C_t and C_o are the mass of organic matter at the end and at the beginning of the experiment, respectively). The decay coefficients k^* obtained for our tea bag experiments are 0.014 ± 0.002 days^{-1} for the green tea and 0.004 ± 0.001 days^{-1} for the rooibos tea.

3.2. Relationships Between Environmental Parameters and Decomposition Processes

The analysis of the effects of environmental variables (i.e., temperature, elevation, distance from the marsh edge, and flooding period) on decomposition metrics computing Kendall correlation coefficient (τ) (Figures 4 and 5), show that the decomposition rate, k, increases with T, whereas the stabilization factor, S, decreases as temperature increases (Kendall's tau test, p-value <0.01).

Furthermore, we observed a statistically-significant relationship between the decomposition rate (k) and both the burial elevation (z_β) (Figure 5a; Kendall's tau test, p-value = 0.0025; k increasing with elevation) and the distance from the closest marsh edge (ζ) (Figure 5c; Kendall's tau test, p-value = 0.0010; k decreasing with increasing distance). Conversely, no significant relation was found between the stabilization factor (S) and either burial elevation (z_β) or the distance from the marsh edge (ζ) (Figures 5b and 5d). The stabilization factor S is also negatively correlated to the duration of marsh flooding (φ) (Figure 5f), whereas flooding (φ) displays no significant correlation with the decomposition rate k (Figure 5e).

When considering decomposition rate values, k, measured along single transects during different incubation periods, temperature showed to have the strongest influence on decomposition processes, as observed in the analyses presented above, whereas elevation and distance from the marsh edge did not significantly affect decomposition rates along single transects (Figure 6).

3.3. Temperature Sensitivity of SOM Decomposition

In order to examine the effects of temperature on decomposition processes, we considered the average values of k and S for every surveyed temperature value (Figure 4). Our results suggest an exponential increase of SOM decomposition rates with temperature (Figure 7a) ($k = a\ e^{bT}$), whereas the stabilization factor, S, is shown to linearly decrease with temperature from about $S = 19\%$ at 21°C to approximately $S = 12\%$ at 27°C (Figure 7b).

In a complex system such as a community of soil organisms, where the total decomposition activity is determined by the combined activity of a wide range of different organisms that presumably have different individual responses to temperature, the temperature-sensitivity indicator Q_{10} may represent a useful tool to summarize observed responses (Kirschbaum, 1995). Q_{10} is defined as the factor by which a reaction rate increases in response to a 10°C increase in temperature (Davidson & Janssens, 2006) (if $k = a\ e^{bT}$, $Q_{10} = e^{(10\ b)}$). According to the previously-recalled relation between k and temperature (Figure 8a), a Q_{10} value of about 2.22 ($Q_{10} = e^{10 \times 0.08} = 2.22$, see Figure 8 for the value of $b = 0.08$) was computed based on our data (Langley et al., 2005). When considering the Q_{10} values for the decomposition rates k^* of both green tea and rooibos in our tea bag experiments, we obtain $Q_{10} = 1.4$ and $Q_{10} = 1.9$ for green and rooibos tea ($b = 0.033$ and 0.066, respectively), respectively. As to the stabilization factor, S, we observed that it declined by about 30% over the observed range of temperatures ($21 < T < 27$°C).

$\tau = -0.4835$; p-value = 2×10^{-24} (n = 223) $\tau = -0.2313$; p-value = 1×10^{-6} (n = 223)
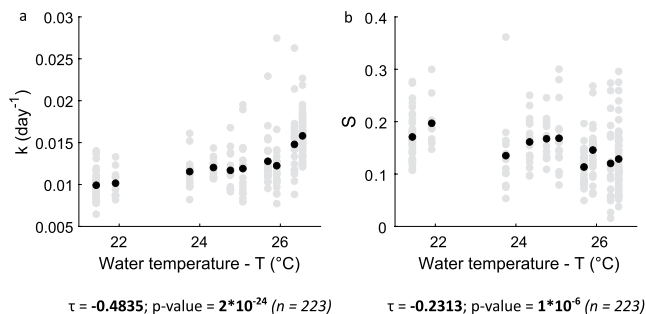
Figure 4. Effect of temperature on decomposition metrics. Decomposition rate (k) (a) and stabilization factor (S) (b) as a function of the mean water temperature (T) measured during the incubation period. Black closed circles represent averages for a given temperature value. In the lower inset, results of Kendall's tau test are reported. Bold text indicates p-values lower than the chosen significance level ($p < 0.01$). The number of samples ($n = 223$) is also reported.
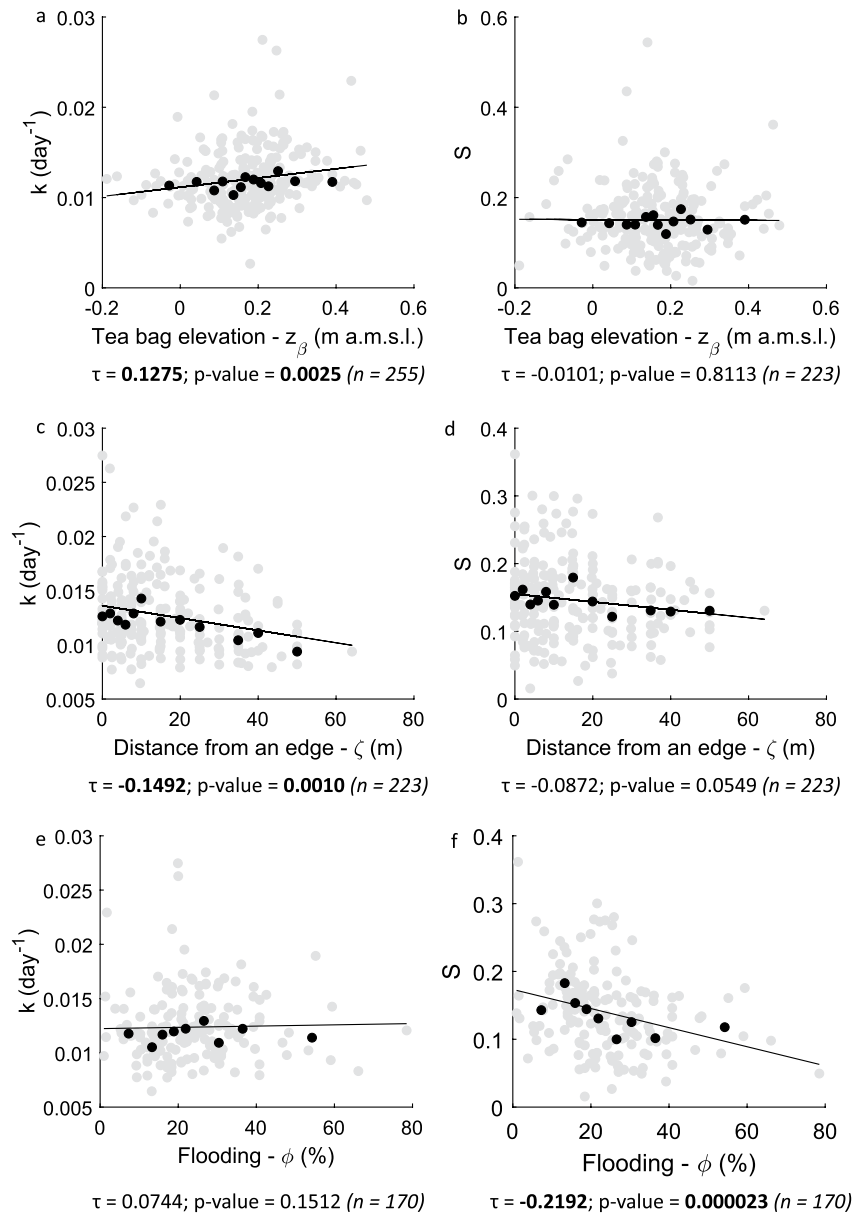
Figure 5. Effects of environmental conditions on organic matter decomposition. Decomposition rate (k) and stabilization factor (S) as a function of (a), (b) burial elevation (z_β) (all samples $n = 255$ considered); (c), (d) distance from the edge (ζ) (only samples buried at 8 cm depth considered, $n = 223$); (e), (f) salt-marsh flooding (φ), computed as the percentage of the burial period during which each site was submerged by water, (only samples buried at 8 cm depth considered, $n = 170$). Black closed circles represent binned averaged values obtained by averaging sets of 20 values. In each panel, the results of the Kendall's tau test are reported in the lower inset, with bold text indicating p-values lower than the chosen significance level ($p < 0.01$).

3.4. SOM Decomposition at Different Depths Below the Marsh Surface

At some sites during the 2017 campaign some tea bags were buried at different depths, from the surface to a few tens of centimeters, to analyze depth-related decomposition variability. For 17 burial sites along three transects (S. Felice-Spaccotralo 1 and 2, and Saline) k and S values were measured at the same time on the marsh surface and at depths of 8, 16, 24, and 40 cm below the surface (Table 2).

The analysis of depth-related decomposition variability (Table 2) showed that decomposition rates measured on the marsh surface are in general significantly different from those measured belowground. As an example, the Wilcoxon matched pairs signed rank test revealed a significant difference in k between the surface and the 8-cm
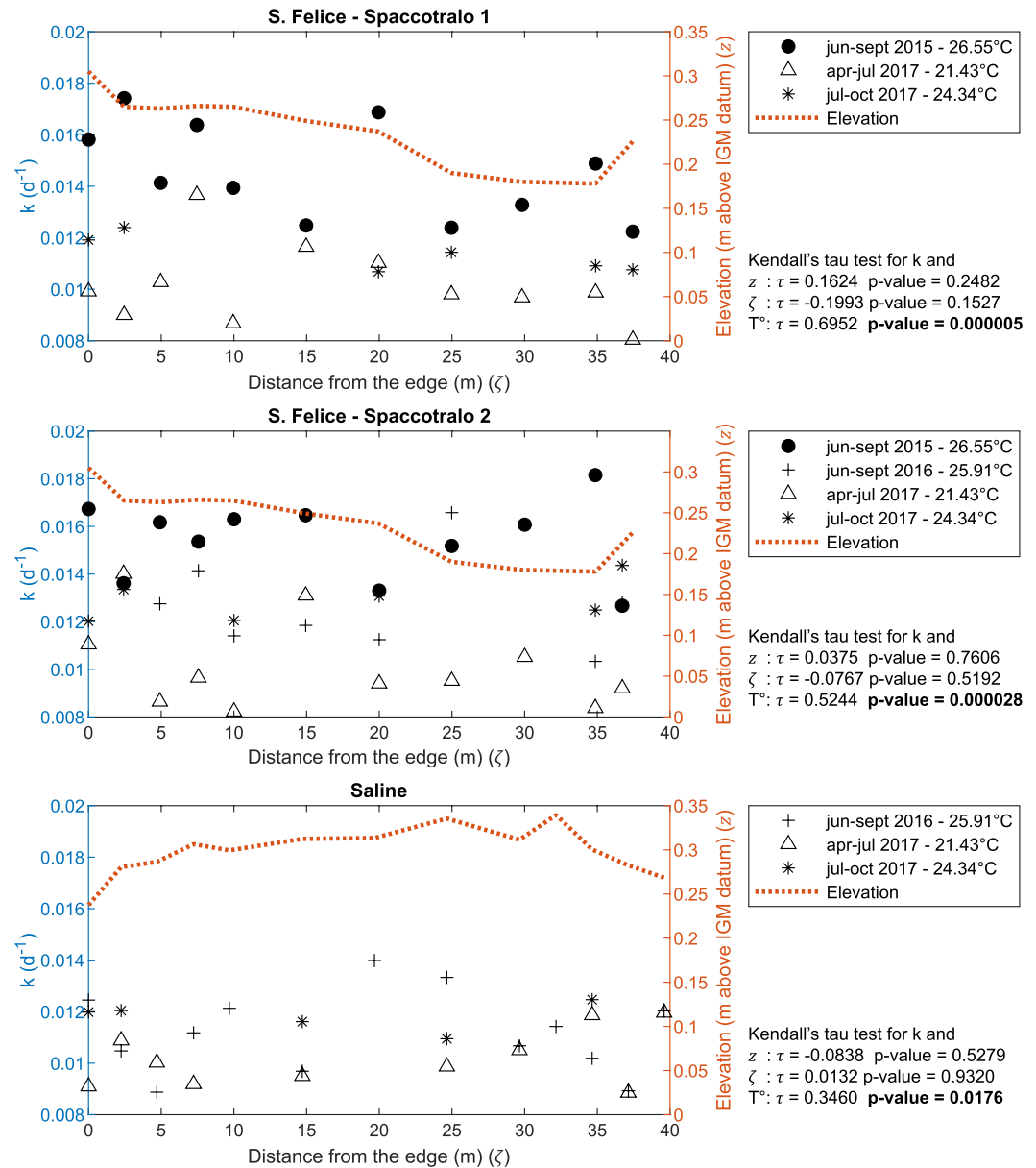
Figure 6. Decomposition variability at the site scale. Decomposition rates (k) for different periods with different mean water temperature along single transects, S. Felice – Spaccotralo 1 (a), S. Felice – Spaccotralo 2 (b), Saline (c), with respect to marsh elevation. In the lower right inset, results of Kendall's tau test are reported.

depth ($n = 17$, $Z = 2.0119$, $p = 0.0442 < 0.05$) and between the surface and the deepest tested layers (depths of 24 and 40 cm) ($n = 17$, $Z = -1.9645$, $p = 0.0495 < 0.05$), whereas no significant differences were observed between the surface and the 16-cm depth. The difference in decomposition rates, k, among different belowground layers (8, 16, 24, 40 cm depth), on the contrary, were not significant. Differences in stabilization factors, S, turned out to be significant only between the surface and the deepest layers tested (24 and 40 cm depth) ($n = 17$, $Z = -2.8166$, $p = 0.0049 < 0.05$).

Regardless of the statistical significance, for about 70% of burial sites along our study transects, decomposition rates decrease with depth, whereas stabilization factors increase with depth. In general, deeper soil layers showed lower decomposition rates and higher stabilization factors compared to more surficial layers (Figures 8a and 8b). The differences between decomposition metrics measured at different depths are in any case limited and in most cases do not exceed 20% (Figures 8c and 8d).
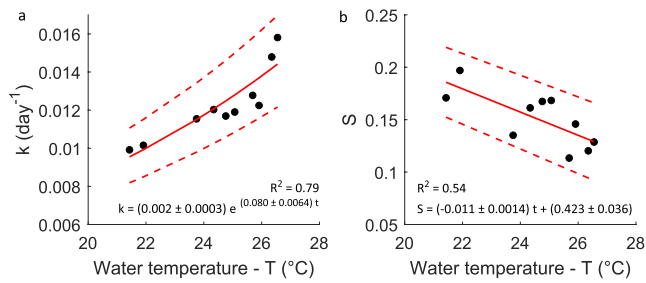
Figure 7. Correlation between temperature and decomposition metrics. (a) Exponential regression of decomposition rate (k) (means of the values measured for each incubation period or temperature classes) as a function of the mean water temperature (p-value = 4.32E−09). (b) Linear regression of the stabilization factor (S) as a function of the mean water temperature (p-value = 0.0274). Regression coefficients, standard deviations, significance and R^2 values were calculated by a standard jackknife resampling method.

4. Discussion

We first computed the decomposition metrics, namely the decomposition rate (k) and the stabilization factor (S), for all the tea bag pair samples retrieved in the Venice lagoon (Figure 2) ($k = 0.012 \pm 0.003$ days^{-1}, $S = 0.15 \pm 0.063$).

A comparison between the values of k and S obtained for the Venice lagoon with those reported by Keuskamp et al. (2013) for different ecosystems is reported in Figure 9, together with data derived for different salt-marsh ecosystems by Mueller et al. (2018) and Yousefi Lalimi et al. (2018). Our results confirm that salt marshes exhibit some of the lowest decomposition rates among different biomes, although they are also characterized by reduced stabilization factors (Keuskamp et al., 2013; Mueller et al., 2018; Yousefi Lalimi et al., 2018). Indeed, Mueller et al. (2018) and Yousefi Lalimi et al. (2018) obtained average k values of 0.012 ± 0.007 and 0.011 ± 0.002 for their salt-marsh sites, respectively, which nicely meet our findings. These decomposition rates are smaller than those obtained for temperate, mixed and tropical forests and for mangroves (Figure 9) (Keuskamp et al., 2013). As far as stabilization factors are concerned, our average value is slightly larger than Yousefi Lalimi et al. (2018)'s one and close to Mueller et al. (2018)'s one.

Assuming the labile OM fraction to decompose according to an exponential-decay model $W(t) = e^{-kt}$ (where $W(t)$ is the remaining fraction of the organic mass at time t and k is the decomposition rate), the computed value of $k = 0.012$ days^{-1} for the Venice lagoon implies that 66% of the initial organic mass would decompose after 90 days. Conversely, a mean stabilization factor of $S = 0.15$ leads, during an incubation timespan of 90 days, to the stabilization of about 15% of the labile material, the latter being therefore not affected by decomposition processes on the short term.

Following the above-recalled asymptotic model proposed by Keuskamp et al. (2013) described by Equation 1, our results suggest that, in the short term, the remaining mass of organic matter strongly depends on the initial recalcitrant fraction $(1 − a)$. Indeed, after the complete decomposition of the labile fraction, the remaining mass is equal to the recalcitrant fraction summed to the stabilized labile fraction.

This was confirmed by the observed difference in mass loss between green and rooibos tea (mean mass loss of about 70% and 30% in 90 days, respectively). Faster initial decomposition of green tea was expected, and it is due to its higher fraction of water-soluble compounds in contrast to the low content of soluble or hydrolyzable compounds in rooibos tea (Keuskamp et al., 2013). Numerous authors suggested that substrate quality, as a function of chemical and structural characteristics, is a primary control on organic litter decomposition rates, indicating C, N, P, lignin content and their ratios as good predictors for plant decomposability (Enríquez et al., 1993; Kazakou et al., 2009; Li et al., 2012; Prieto et al., 2016; Silver & Miya, 2001). This is confirmed also at a local scale and for coastal marshes (Scarton et al., 2002; Stagg et al., 2018). Therefore, potential shifts in species relative abundance and primary production in the future caused by climate change and anthropogenic disturbances could have large effects on carbon cycle, affecting litter quality, biotic and abiotic soil conditions, and consequently decomposition rates (Cornelissen et al., 2007; Cornwell et al., 2008; Djukic et al., 2018).

Our results on the decay coefficient $k*$ are overall consistent with findings by Kirwan et al. (2014) and Kirwan and Blum (2011), who measured the organic mass loss in litter bag decomposition experiments in the North American Atlantic coasts by computing the decay coefficient $k*$ not accounting for the proportion of recalcitrant and labile material ($k* = \left[-ln(C_t/C_o) \right]/t$, where C_t and C_o are the mass of organic matter at the end and at the beginning of the experiment, respectively). As expected, the decay coefficients $k*$ obtained for our tea bag experiments are different for the green tea (0.014 ± 0.002 days^{-1}) and the rooibos tea (0.004 ± 0.001 days^{-1}), confirming that green tea is

Table 2
Decomposition Variations in Matched Paired Samples at Different Depth

Matched pairs	n	Z value	p-value
k 0/−8	17	2.0119	0.0442*
S 0/−8	17	−1.8699	0.0615
k 0/−16	17	1.0651	0.2868
S 0/−16	17	−1.6805	0.0929
k 0/max depth (−24 or −40)	17	1.9645	0.0495*
S 0/max depth (−24 or −40)	17	−2.8166	0.0049*
k −8/−16	17	−0.4971	0.6192
S −8/−16	17	0.4971	0.6192
k −8/max depth (−24 or −40)	17	0.3550	0.7226
S −8/max depth (−24 or −40)	17	−0.3550	0.7226
k −16/max depth (−24 or −40)	17	1.0651	0.2868
S −16/max depth (−24 or −40)	17	−1.0951	0.2868

Note. Results of the Wilcoxon matched pairs signed rank test on the values of k and S measured at different depths. Asterisks indicate p-values being less than the chosen significance level ($p < 0.05$).
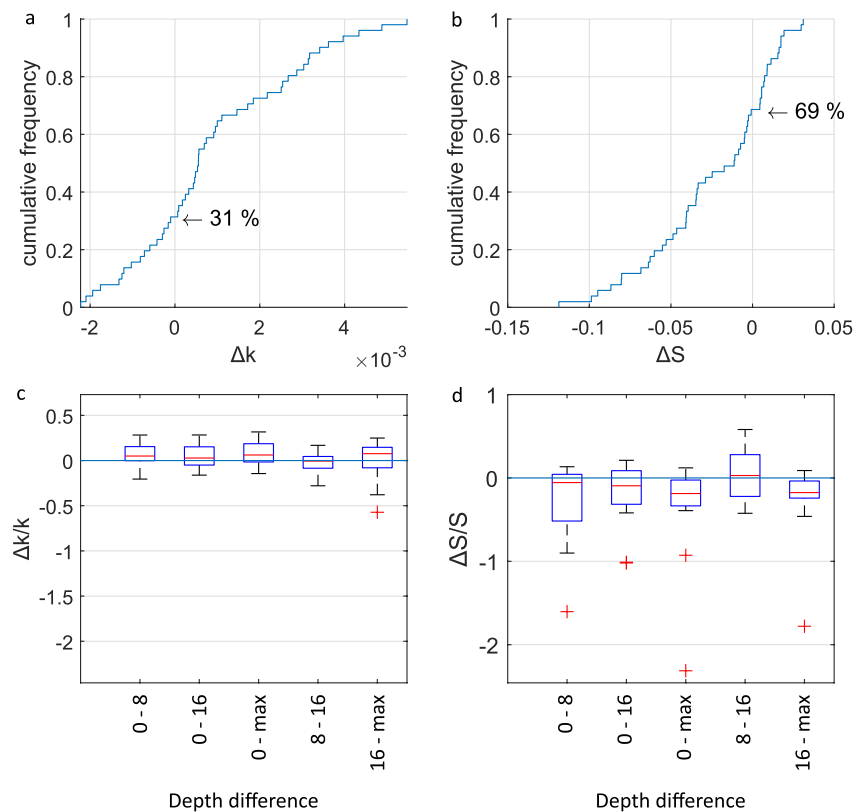
Figure 8. Differences of decomposition metrics at different depth. Cumulative distribution function of difference in k (a) and S (b) between the surface and all other deeper layers and box-plots representing the relative difference between k (c) and S (d) values at different depths. (The central mark indicates the median, and the bottom and top edges of the box indicate the 25th and 75th percentiles, respectively. Whiskers extend to the most extreme data points not considered outliers and the outliers are plotted individually using the '+' symbol).

more labile, whereas rooibos is more recalcitrant (Figure 3). Moreover, values from our samples nicely match the decay coefficients k^* found by Kirwan and Blum (2011) for *Spartina alterniflora* stem and leaf material (0.004 ÷ 0.016 days^{-1}) and by Kirwan et al. (2014) for *Schoenoplectus americanus* root and rhizome material (0.003 ÷ 0.007 days^{-1}). In the case of salt-marsh vegetation, aboveground litter was observed to generally decompose faster than belowground litter (Scarton et al., 2002; Stagg et al., 2018). Therefore, green and rooibos tea showed consistent decay coefficients with respect to aboveground and belowground salt-marsh plant litter, respectively.

Although the aim of Tea Bag Index analysis is to elucidate climate and environmental effects on decomposition processes, by use of a standard organic material (thereby removing any influence by litter quality), this comparison further confirms that tea material can be considered representative of organic matter in salt-marsh environments and further highlights the influence of litter quality on organic matter conservation.

Secondly, we analyzed the effects of environmental variables such as temperature, elevation, distance from the marsh edge, and flooding period on both the decomposition coefficient (k) and on the stabilization factor (S) (Figures 4 and 5). We show that the average water temperature during the incubation period (T) holds the most statistically significant relation with both k and S, with the decomposition rate, k, increasing with T, and the stabilization factor, S, decreasing as temperature increases (Kendall's tau test, p-value <0.01, Figure 4). Notably, these relationships hold in spite of the relatively narrow range of temperature variability observed in our dataset, with $21< T < 27°C$ for all the experiments carried out in this study.

Such significant control of T on both S and k is consistent with previous findings highlighting the critical role of temperature on OM decay rates in tidal wetlands (Keuskamp et al., 2013; Kirwan & Blum, 2011; Kirwan et al., 2014; Montagna & Ruber, 1980; Mueller et al., 2018; White et al., 1978). Being a chemical process
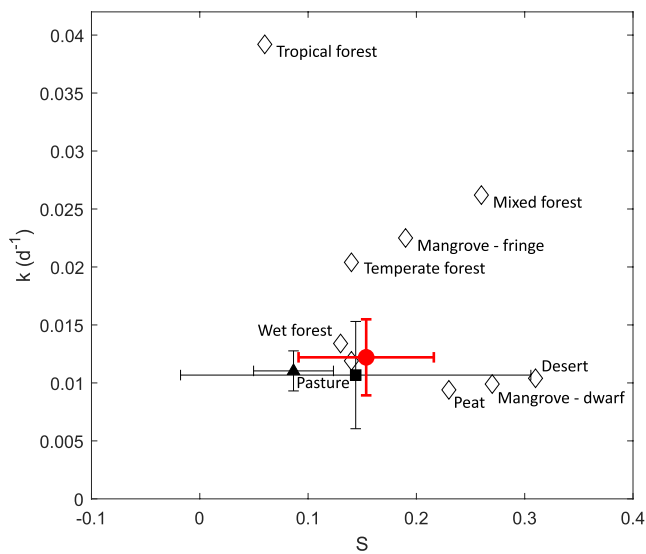
Figure 9. Decomposition metrics in major biomes. The mean values of the decomposition rate (k) and of the stabilization factor (S) found in this study (red dot), compared with data from different ecosystems according to Keuskamp et al. (2013) (empty diamonds) and for different marshes around the world, based on data from Yousefi Lalimi et al. (2018) (solid triangle, from three transects in North Carolina (USA) marshes) and Mueller et al. (2018) (solid square, data from 30 tidal-wetland sites worldwide, most of which were salt marshes). Error bars indicate standard deviations.

mediated by microbial enzymes, the decomposition of SOM is known to be strongly affected by temperature (e.g., Davidson & Janssens, 2006; Moinet et al., 2020). Indeed, temperature can affect litter decomposition both directly, by regulating the activity of decomposers, and indirectly, through changes in other temperature-sensitive conditions influencing decomposition processes, such as soil moisture and oxygen diffusion. Oxygen diffusion from the atmosphere, plant aerenchyma, or aerated water, which increases with increasing temperature, affects oxygen availability for decomposition reactions. In saturated or partially saturated soils, oxygen availability represents a rate-limiting factor for decomposition processes (Kirwan et al., 2014).

Furthermore, we found a statistically-significant relationship between the decomposition rate (k) and both the burial elevation (z_β) (Figure 5a; Kendall's tau test, p-value = 0.0025; k increasing with elevation) and the distance from the closest marsh edge (ζ) (Figure 5c; Kendall's tau test, p-value = 0.0010; k decreasing with increasing distance), the latter being a proxy for the distance to the nearest source of water and mineral sediments. Conversely, no significant relation was found between the stabilization factor (S) and either burial elevation (z_β) or the distance from the marsh edge (ζ) (Figures 5b and 5d). The stabilization factor S is also negatively correlated to the duration of marsh flooding (φ) (Figure 5f), whereas flooding (φ) displays no significant correlation with the decomposition rate k (Figure 5e).

Flooding is known to slow oxygen diffusion by suppressing microbial respiration and allowing only anaerobic decomposition, which includes fewer and generally slower degradative enzymatic pathways (Davidson & Janssens, 2006; Morris et al., 2016). Differences in flooding frequency along elevational gradients in tidal wetlands have proved to induce sharp gradients in oxygen availability and redox conditions (Kirwan & Megonigal, 2013; Morris et al., 2016). Therefore, the intrinsic spatial variability in salt-marsh topography is likely to influence microclimatic conditions affecting decomposition processes.

The dependence of k on both z_β and ζ is somehow expected in view of the typical concave-up morphology characterizing most of the marsh platforms, where surface elevation decreases progressively moving away from the marsh edge towards the inner marsh. Clearly, marsh elevation also exerts a primary control on flooding frequency and duration (Chmura & Hung, 2004; A. D'Alpaos et al., 2007; Marani, Belluco, et al., 2006). Thus, these interdependencies may suggest that larger oxygen availability in more elevated, which are less frequently flooded sites, promotes the decomposition of organic matter, enhancing microbial respiration. Other factors linked to these morphological features that could potentially influence decomposition processes are vegetation characteristics (i.e., above and belowground production, C-N ratio) and nutrient availability, which may affect microbial community (Kirwan et al., 2014; Mueller et al., 2018; Yarwood, 2018). However, the control of marsh flooding on OM decomposition is not as clear as in the case of elevation and distance from the edge, with φ not being significantly correlated to k. Nevertheless, we should highlight that φ was derived indirectly by comparing site-specific measurements of marsh elevation with tide level data retrieved from nearby tidal gauge stations. Therefore, a degree of uncertainty is to be expected for φ, which might not be an optimal representation of the actual time during which the marsh soil is submerged, the latter process depending also on local marsh microtopography and on the characteristics of the extensive networks of tidal creeks that typically cut through the marsh platform.

Moreover, it is worthwhile noting that soil aeration is affected also by plant evapotranspiration and water table dynamics. Groundwater flow and evapotranspiration are known to promote the formation of an aerated layer below the soil surface, thus allowing for a prolonged presence of oxygen for aerobic respiration even when the marsh is flooded (Boaga et al., 2014; Marani, Silvertri, et al., 2006; Ursino et al., 2004). Ursino et al. (2004) and Marani, Silvertri, et al. (2006) showed that in the absence of evapotranspiration, and when water infiltration dominates over root uptake, more aerated soil conditions are found near the channels, whereas the inner marsh portions are more oxygenated when plant transpiration balances infiltration from the flooded surface. This inferences clearly highlight how soil and vegetation characteristics (e.g., hydraulic conductivity and vegetation density), which are in turn affected by local elevation and distance from the nearest marsh margin (e.g.,

Roner et al., 2016), might crucially affect soil aeration. Therefore, oxygen dynamics in marsh soil are quite complex and may cause difficulties in the detection and interpretation of the relationship between flooding conditions and decomposition processes. Furthermore, even if the effect of vegetation on litter decay was not directly inspected in the present work, the observed relationships between decomposition and physical factors may embody the influence of biotic component, due to the mutual interactions between hydrodynamic, morphological and biological factors characterizing salt-marsh environments. Salt-marsh live vegetation plays an important role in decomposition processes, with direct and indirect influences. Vegetation composition can influence soil physical and chemical conditions (e.g., altering the pH and water-holding capacity of soil) (e.g., Marani, Belluco, et al., 2006; Rydin & Jeglum, 2006; Ursino et al., 2004). In addition, different plant species affect nutrient inputs and, consequently, the activity and composition of the decomposer community (Ward et al., 2015).

As to the temperature sensitivity of SOM decomposition, our results suggest an exponential increase of SOM decomposition rates with temperature (Figure 7a) ($k = 0.002 \, e^{0.08 \, T}$), whereas the stabilization factor, S, is shown to linearly decrease with temperature (Figure 7b) ($S = -0.011 \, T + 0.423$). Temperature sensitivity of SOM decomposition is usually described using Arrhenius' (1889) kinetic theory according to which, when substrates are abundant at enzymes reaction sites, decomposition rates increase exponentially with temperature (Davidson & Janssens, 2006; Moinet et al., 2020). In addition, the Arrhenius equation suggests that the sensitivity of the reaction rate on temperature increases with increasing availability of the recalcitrant fraction in the substrates (i.e., "intrinsic temperature sensitivity"). However, environmental constraints also influence the observed decomposition response to temperature (i.e., the "apparent temperature sensitivity"), affecting the accessibility to organic carbon substrates (Davidson & Janssens, 2006).

The Q_{10} value computed based on our data (2.22) is coherent with the typical values found for reaction rates in biological systems as well as for organic matter decomposition at ambient temperature (Davidson & Janssens, 2006; Kirschbaum, 1995; Singh & Gupta, 1977). The sensitivity of decomposition on temperature obtained in our analyses is lower than the sensitivity reported for litter bag experiments carried out directly on the soil surface in a rarely flooded salt marsh on the Virginia's coast ($Q_{10} = 3.44$) (Kirwan & Blum, 2011). In addition, our Q_{10} value for k agrees with the estimated temperature sensitivities of CO_2 emissions from freshwater wetland ($Q_{10} = 1.3–2.5$) (Inglett et al., 2012), slightly exceeding those reported by Morris and Whiting (1986) for salt marsh soils ($Q_{10} = 1.5–1.8$). In addition, they appear slightly larger than those reported by Kirwan et al. (2014), when estimating the range of temperature sensitivity of decomposition rate ($k*$) ($Q_{10} = 1.3–1.5$), without accounting for the proportion of recalcitrant and labile material.

Interestingly, when considering the Q_{10} values for the decomposition rates $k*$ of green tea and rooibos ($Q_{10} = 1.4$ and $Q_{10} = 1.9$ for green and rooibos tea), they are closer to those reported by Kirwan et al. (2014). Our analysis on the decay coefficient $k*$ highlighted a lower temperature sensitivity for more labile substrates (e.g., green tea in our TBI analysis) than for recalcitrant substrate (rooibos tea), consistently with the substrate-quality dependence of temperature sensitivity suggested by Arrhenius' (1889) kinetic theory (Davidson & Janssens, 2006). Therefore, our results show the dependency of decomposition processes on temperature and further confirm the need to consider the influence of initial litter quality.

In agreement with Yousefi Lalimi et al. (2018), who found that decomposition rates and stabilization factors of shallower soil layers (3-cm depth) and deeper layers (8-cm depth) were significantly different, our results from the analysis of depth-related decomposition variability (Table 2) suggest that there is a significant difference between the decomposition process on the marsh surface, probably consistent within the first few centimetres of soil, with respect to belowground conditions. Differences in k between different deeper layers within the soil are instead reduced, which may indicate either conditions becoming more uniform or a possible saturation of the effects due to changing controlling factors. The lower decomposition rates and higher stabilisation factors found in deeper soil layers (Figures 8a and 8b) are consistent with the reduced oxygen availability in those layers.

Even though our findings show that deeper soil layers are, on average, characterized by lower decomposition rates and higher stabilization factors, this difference appears limited (Figures 8c and 8d) and decomposition metrics from the study area indicate that the organic matter decay is rapid enough to consume all the labile material before it can be buried and permanently stabilized.

Our results highlight how global warming may strongly affect decomposition processes in salt marsh soil and provide further elements to elucidate decomposition controls at the marsh scale, where ecomorphodynamic processes importantly affect both biotic and abiotic factors.

The decomposition experiment of the present work covers a period of 3 month, providing insight into the short-term mass loss dynamics of plant litter. According to Valiela et al. (1985), the decomposition of fresh litter normally occurs in three phases. The first phase includes leaching and rapid microbial assimilation of the water-soluble fraction (sugars and starch, etc.), the second phase involves the loss of structural components (cellulose), and the final phase, described as the metastable phase, consists of a slow decrease in mass that occurs due to the gradual loss of lignin (Valiela et al., 1985; Marley et al., 2019). Marley et al. (2019) suggest that the 3-month tea-bag decomposition experiment only covers the first and second phase of decomposition and Djukic et al. (2018) advise that, as long-term decomposition involves different litter components, the effect and relative importance of multiple drivers are likely to vary at spatial and temporal scales. However, under the exponential decay hypothesis, the first phases of the decomposition process are responsible for most of the organic mass loss, therefore, even if caution should be paid when extrapolating from short-term to long-term rates, the Tea Bag Index approach still provide important insights into the decomposition process of organic matter. Therefore, despite the need for caution, the TBI method, with a standardized protocol, remains a useful and cost-effective tool in understanding and comparing decomposition dynamics of organic litter within and between intertidal environments (Marley et al., 2019)

5. Conclusions

To better understand the dynamics of organic matter which contributes to soil formation and carbon sequestration in salt marshes, we analyzed decomposition processes in salt-marsh ecosystems in the microtidal lagoon of Venice and investigated how they are influenced by environmental factors. We carried out decomposition experiments using standardized organic material according to the Tea Bag Index protocol (Keuskamp et al., 2013), focusing only on the initial phases of the decay (i.e., 3 months).

Our results highlight that temperature plays a significant role in OM decomposition and that it is indeed responsible for an increase in the decomposition rate of about 8% per °C. Our results indicate a faster OM decay on the marsh edge and at higher elevations, supporting the idea of slower organic matter degradation toward inner and lower marsh portions (Roner et al., 2016). Moreover, our results highlight how oxygen availability importantly affects decomposition, even if the complexity of soil aeration dynamics prevented the detection of any clear trends in our dataset. We also suggest that the variability in decomposition metrics, namely the decomposition rate k and the stabilisation factor S, reflects site specific micro-scale variability of the environmental forcing affecting decomposition processes.

Although actual values of S and k for salt-marsh organic matter may differ from those obtained using standard litter, the latter resulted to be comparable with the decay rates calculated using autochthonous plant litter in other studies (e.g., Kirwan & Blum, 2011; Kirwan et al., 2014; Yousefi Lalimi et al., 2018). Specifically, the decomposition rates measured in our study display a mean value of 0.012 ± 0.003 days^{-1} (ranging between 0.003 and 0.027 days^{-1}), which would lead to a loss of about two-thirds of the initial organic mass after 90 days. Nevertheless, salt-marsh decomposition rates are confirmed to be some of the lowest with respect to other biomes (see Figure 9). As to the stabilization processes, our measurements highlight a high variability in stabilization factors: on average 15% of the organic matter was stabilised during the initial stage of decay, with values ranging between 2% and 54%.

Within this scenario, in terms of future climate conditions, the initial litter quality appears to be a primary constraint for the amount of preserved organic matter contributing to carbon sequestration and marsh accretion, as also confirmed by the observed difference in mass loss between green and rooibos tea (mean mass loss of about 70% and 30% in 90 days, respectively). This suggests the need to accounting for both direct and indirect effects of climate change as possibly affecting decomposition processes. Hence, we highlight the need for further analyses on autochthonous plant litter dynamics, also on longer time scales, considering their variability.

Although caution is needed in the interpretation of the decomposition metric measured, our data provide additional insights to improve biogeomorphological model ability to describe marsh response to the effects of climate change and anthropogenic perturbation and further elucidate marsh importance within the global carbon cycle.

Conflict of Interest

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

Data Availability Statement

All data needed to evaluate the conclusions in the paper are available at https://researchdata.cab.unipd.it/id/eprint/871 (Puppin et al., 2022). Meteorological and tide level data for the Venice Lagoon are also freely available at www.comune.venezia.it/content/dati-dalle-stazioni-rilevamento and www.venezia.isprambiente.it/rete-meteo-mareografica.

References

Adam, P. (1990). *Saltmarsh ecology*. Cambridge University Press. https://doi.org/10.1017/CBO9780511565328

Allen, J. R. L. & Pye, K. (Eds.). (1992)., *Saltmarshes. Morphodynamics, conservation and engineering significance*. Cambridge University Press.

Allen, J. R. L. (2000). Morphodynamics of Holocene salt marshes: A review sketch from the Atlantic and southern north sea coasts of Europe. *Quaternary Science Reviews*, *19*(17–18), 1255–1840. https://doi.org/10.1016/S0277-3791(00)00157-8

Arrhenius, S. (1889). Über die Reaktionsgeschwindigkeit bei der Inversion von Rohrzucker durch Säuren. *Zeitschrift für Physikalische Chemie*, *4*(1), 226–248. https://doi.org/10.1515/zpch-1889-0416

Barbier, E. B., Hacker, S. D., Kennedy, C., Koch, E. W., Stier, A. C., & Silliman, B. R. (2011). The value of estuarine and coastal ecosystem services. *Ecological Monographs*, *81*(2), 169–193. https://doi.org/10.1890/10-1510.1

Boaga, J., D'Alpaos, A., Cassiani, G., Marani, M., & Putti, M. (2014). Plant-Soil interactions in salt marsh environments: Experimental evidence from electrical resistivity tomography in the Venice Lagoon. *Geophysical Research Letters*, *41*(17), 6160–6166. https://doi.org/10.1002/2014GL060983

Boesch, D. F., & Turner, R. E. (1984). Dependence of fishery species on salt marshes: The role of Food and Refuge. *Estuaries*, *7*(4), 460–468. https://doi.org/10.2307/1351627

Bonometto, L. (2005). Functional characteristics of salt marshes (barene) in the Venice Lagoon, and environmental restoration scenarios. Workshop on Venice lagoon. In C. A. Fletcher & T. Spencer (Eds.), *Flooding and environmental challenges for Venice and its lagoon: State of knowledge* (pp. 473–486). Cambridge University Press.

Cahoon, D. R., & Guntenspergen, G. R. (2010). Climate change, sea-level rise, and coastal wetlands. *National Wetlands Newsletter*, *32*(1), 8–12.

Cecconi, G., Codato, F., Nascimbeni, P., & Mattarolo, F. (1998). Valore ambientale delle barene artificiali. *Quaderni Trimestrali Consorzio Venezia Nuova*, *IV*, 27–49.

Chmura, G. L., Anisfeld, S. C., Cahoon, D. R., & Lynch, J. C. (2003). Global carbon sequestration in tidal, saline wetland soils. *Global Biogeochemical Cycles*, *17*(4), 22–31. https://doi.org/10.1029/2002gb001917

Chmura, G. L., & Hung, G. A. (2004). Controls on salt marsh accretion: A test in salt marshes of Eastern Canada. *Estuaries*, *27*(1), 70–81. https://doi.org/10.1007/BF02803561

Cornelissen, J. H. C., Van Bodegom, P. M., Aerts, R., Callaghan, T. V., Van Logtestijn, R. S. P., Alatalo, J., et al. (2007). Global negative vegetation feedback to climate warming responses of leaf litter decomposition rates in cold biomes. *Ecology Letters*, *10*(7), 619–627. https://doi.org/10.1111/j.1461-0248.2007.01051.x

Cornwell, W. K., Cornelissen, J. H. C., Amatangelo, K., Dorrepaal, E., Eviner, V. T., Godoy, O., et al. (2008). Plant species traits are the predominant control on litter decomposition rates within biomes worldwide. *Ecology Letters*, *11*(10), 1065–1071. https://doi.org/10.1111/j.1461-0248.2008.01219.x

Costanza, R., D'Arge, R., De Groot, R., Farber, S., Grasso, M., Hannon, B., et al. (1997). The value of the world's ecosystem services and natural capital. *Nature*, *387*(6630), 253–260. https://doi.org/10.1038/387253a0

Couvillion, B. R., Steyer, G. D., Wang, H., Beck, H. J., & Rybczyk, J. M. (2013). Forecasting the effects of coastal protection and restoration projects on wetland morphology in coastal Louisiana under multiple environmental uncertainty scenarios. *Journal of Coastal Research*, *67*, 29–50. SPEC. ISSUE). https://doi.org/10.2112/SI_67_3

D'Alpaos, A., Lanzoni, S., Marani, M., & Rinaldo, A. (2007). Landscape evolution in tidal embayments: Modeling the interplay of erosion, sedimentation, and vegetation dynamics. *Journal of Geophysical Research*, *112*(F1), F01008. https://doi.org/10.1029/2006JF000537

D'Alpaos, A., & Marani, M. (2016). Reading the signatures of biologic–geomorphic feedbacks in salt-marsh landscapes. *Advances in Water Resources*, *93*, 265–275. https://doi.org/10.1016/j.advwatres.2015.09.004

D'Alpaos, A., Mudd, S. M., & Carniello, L. (2011). Dynamic response of marshes to perturbations in suspended sediment concentrations and rates of relative sea level rise. *Journal of Geophysical Research*, *116*(F4), F04020. https://doi.org/10.1029/2011JF002093

D'Alpaos, C., & D'Alpaos, A. (2021). The valuation of ecosystem services in the venice lagoon: A multicriteria approach. *Sustainability*, *13*(17), 9485. https://doi.org/10.3390/su13179485

Davidson, E. A., & Janssens, I. A. (2006). Temperature sensitivity of soil carbon decomposition and feedbacks to climate change. *Nature*, *440*(7081), 165–173. https://doi.org/10.1038/nature04514

Djukic, I., Kepfer-Rojas, S., Schmidt, I. K., Larsen, K. S., Beier, C., Berg, B., et al. (2018). Early stage litter decomposition across biomes. *Science of the Total Environment*, *628–629*, 1369–1394. https://doi.org/10.1016/j.scitotenv.2018.01.012

Duarte, C. M., Middelburg, J. J., & Caraco, N. (2005). Major role of marine vegetation on the oceanic carbon cycle. *Biogeosciences*, *2*(1), 1–8. https://doi.org/10.5194/bg-2-1-2005

Enríquez, S., Duarte, C. M., & Sand-Jensen, K. (1993). Patterns in decomposition rates among photosynthetic organisms: The importance of detritus C:N:P content. *Oecologia*, *94*(4), 457–471. https://doi.org/10.1007/BF00566960

Fagherazzi, S., Kirwan, M. L., Mudd, S. M., Guntenspergen, G. R. G. R., Temmerman, S., D'Alpaos, A., et al. (2012). Numerical models of salt marsh evolution: Ecological, geomorphic, and climatic factors. *Reviews of Geophysics*, *50*(1). https://doi.org/10.1029/2011RG000359

Freschet, G. T., Aerts, R., & Cornelissen, J. H. C. (2012). A plant economics spectrum of litter decomposability. *Functional Ecology*, *26*(1), 56–65. https://doi.org/10.1111/j.1365-2435.2011.01913.x

Halupa, P. J., & Howes, B. L. (1995). Effects of tidally mediated litter moisture content on decomposition of Spartina alterniflora and S. patens. *Marine Biology*, *123*(2), 379–391. https://doi.org/10.1007/BF00353629

Inglett, K., Inglett, P., Reddy, K., & Osborne, T. (2012). Temperature sensitivity of greenhouse gas production in wetland soils of different vegetation. *Biogeochemistry*, *108*(1–3), 77–90. https://doi.org/10.1007/s10533-011-9573-3

Kazakou, E., Violle, C., Roumet, C., Pintor, C., Gimenez, O., & Garnier, E. (2009). Litter quality and decomposability of species from a Mediterranean succession depend on leaf traits but not on nitrogen supply. *Annals of Botany*, *104*(6), 1151–1161. https://doi.org/10.1093/aob/mcp202

Kendall, M. G. (1938). A measure of rank correlation. *Biometrika*, *30*(1–2), 81–93. https://doi.org/10.1093/biomet/30.1-2.81

Keuskamp, J. A., Dingemans, B. J. J., Lehtinen, T., Sarneel, J. M., & Hefting, M. M. (2013). Tea bag Index: A novel approach to collect uniform decomposition data across ecosystems. *Methods in Ecology and Evolution*, *4*(11), 1070–1075. https://doi.org/10.1111/2041-210X.12097

Kirschbaum, M. U. F. (1995). The temperature dependence of soil organic matter decomposition, and the effect of global warming on soil organic C storage. *Soil Biology and Biochemistry*, *27*(6), 753–760. https://doi.org/10.1016/0038-0717(94)00242-S

Kirwan, M. L., & Blum, L. K. (2011). Enhanced decomposition offsets enhanced productivity and soil carbon accumulation in coastal wetlands responding to climate change. *Biogeosciences*, *8*(4), 987–993. https://doi.org/10.5194/bg-8-987-2011

Kirwan, M. L., Guntenspergen, G. R., D'Alpaos, A., Morris, J. T., Mudd, S. M., & Temmerman, S. (2010). Limits on the adaptability of coastal marshes to rising sea level. *Geophysical Research Letters*, *37*(23), a–n. https://doi.org/10.1029/2010GL045489

Kirwan, M. L., Guntenspergen, G. R., & Langley, J. A. (2014). Temperature sensitivity of organic-matter decay in tidal marshes. *Biogeosciences*, *11*(17), 4801–4808. https://doi.org/10.5194/bg-11-4801-2014

Kirwan, M. L., Langley, J. A., Guntenspergen, G. R., & Megonigal, J. P. (2013). The impact of sea-level rise on organic matter decay rates in Chesapeake Bay brackish tidal marshes. *Biogeosciences*, *10*(3), 1869–1876. https://doi.org/10.5194/bg-10-1869-2013

Kirwan, M. L., & Megonigal, J. P. (2013). Tidal wetland stability in the face of human impacts and sea-level rise. *Nature*, *504*(7478), 53–60. https://doi.org/10.1038/nature12856

Kirwan, M. L., & Murray, A. B. (2007). A coupled geomorphic and ecological model of tidal marsh evolution. *Proceedings of the National Academy of Sciences*, *104*(15), 6118–6122. https://doi.org/10.1073/PNAS.0700958104

Lang, S. I., Cornelissen, J. H. C., Klahn, T., Van Logtestijn, R. S. P., Broekman, R., Schweikert, W., & Aerts, R. (2009). An experimental comparison of chemical traits and litter decomposition rates in a diverse range of subarctic bryophyte, lichen and vascular plant species. *Journal of Ecology*, *97*(5), 886–900. https://doi.org/10.1111/j.1365-2745.2009.01538.x

Langley, J. A., Johnson, N. C., & Koch, G. W. (2005). Mycorrhizal status influences the rate but not the temperature sensitivity of soil respiration. *Plant and Soil*, *277*(1), 335–344. https://doi.org/10.1007/s11104-005-7932-3

Li, X., Cui, B., Yang, Q., Tian, H., Lan, Y., Wang, T., & Han, Z. (2012). Detritus quality controls macrophyte decomposition under different nutrient concentrations in a eutrophic shallow lake, North China. *PLoS One*, *7*(7), e42042. https://doi.org/10.1371/journal.pone.0042042

MacKenzie, R. A., & Dionne, M. (2008). Habitat heterogeneity: Importance of salt marsh pools and high marsh surfaces to fish production in two Gulf of Maine salt marshes. *Marine Ecology Progress Series*, *368*, 217–230. https://doi.org/10.3354/meps07560

Macreadie, P. I., Anton, A., Raven, J. A., Beaumont, N., Connolly, R. M. R. M., Friess, D. A. D. A., et al. (2019). The future of Blue Carbon science. *Nature Communications*, *10*(1), 1–13. https://doi.org/10.1038/s41467-019-11693-w

Marani, M., Belluco, E., Ferrari, S., Silvestri, S., D'Alpaos, A., Lanzoni, S., et al. (2006). Analysis, synthesis and modelling of high-resolution observations of salt-marsh eco-geomorphological patterns in the Venice lagoon. *Estuarine, Coastal and Shelf Science*, *69*(3–4), 414–426. https://doi.org/10.1016/j.ecss.2006.05.021

Marani, M., D'Alpaos, A., Lanzoni, S., Carniello, L., & Rinaldo, A. (2007). Biologically-controlled multiple equilibria of tidal landforms and the fate of the Venice lagoon. *Geophysical Research Letters*, *34*(11), L11402. https://doi.org/10.1029/2007GL030178

Marani, M., D'Alpaos, A., Lanzoni, S., Carniello, L., & Rinaldo, A. (2010). The importance of being coupled: Stable states and catastrophic shifts in tidal biomorphodynamics. *Journal of Geophysical Research*, *115*(F4), F04004. https://doi.org/10.1029/2009JF001600

Marani, M., Silvestri, S., Belluco, E., Ursino, N., Comerlati, A., Tosatto, O., & Putti, M. (2006). Spatial organization and ecohydrological interactions in oxygen-limited vegetation ecosystems. *Water Resources Research*, *42*(6). https://doi.org/10.1029/2005WR004582

Marley, A. C. R. G., Smeaton, C., & Austin, W. E. N. (2019). An Assessment of the tea bag Index method as a proxy for organic matter decomposition in intertidal environments. *Journal of Geophysical Research: Biogeosciences*, *124*(10), 2991–3004. https://doi.org/10.1029/2018JG004957

McLeod, E., Chmura, G. L., Bouillon, S., Salm, R., Björk, M., Duarte, C. M., et al. (2011). A blueprint for blue carbon: Toward an improved understanding of the role of vegetated coastal habitats in sequestering CO2. *Frontiers in Ecology and the Environment*, *9*(10), 552–560. https://doi.org/10.1890/110004

Mcowen, C. J., Weatherdon, L. V., Van Bochove, J.-W., Sullivan, E., Blyth, S., Zockler, C., et al. (2017). A global map of saltmarshes. *Biodiversity Data Journal*, *5*(1), e11764. https://doi.org/10.3897/BDJ.5.e11764

Megonigal, J. P., Hines, M. E., & Visscher, P. T. (2004). Anaerobic metabolism: Linkages to trace gases and aerobic processes. In W. H. Schlesinger (Ed.), *Biogeochemistry* (pp. 317–424). Elsevier-Pergamon.

Moinet, G. Y. K., Moinet, M., Hunt, J. E., Rumpel, C., Chabbi, A., & Millard, P. (2020). Temperature sensitivity of decomposition decreases with increasing soil organic matter stability. *Science of the Total Environment*, *704*, 704. https://doi.org/10.1016/j.scitotenv.2019.135460

Möller, I., Kudella, M., Rupprecht, F., Spencer, T., Paul, M., Van Wesenbeeck, B. K., et al. (2014). Wave attenuation over coastal salt marshes under storm surge conditions. *Nature Geoscience*, *7*(10), 727–731. https://doi.org/10.1038/NGEO2251

Montagna, P. A., & Ruber, E. (1980). Decomposition of Spartina alterniflora in different seasons and habitats of a Northern Massachusetts salt marsh, and a comparison with other Atlantic regions. *Estuaries*, *3*(1), 61–64. https://doi.org/10.2307/1351935

Morris, J. T., Barber, D. C., Callaway, J. C., Chambers, R., Hagen, S. C., Hopkinson, C. S., et al. (2016). Contributions of organic and inorganic matter to sediment volume and accretion in tidal wetlands at steady state. *Earth's Future*, *4*(4), 110–121. https://doi.org/10.1002/2015EF000334

Morris, J. T., Sundareshwar, P. V., Nietch, C. T., Kjerfve, B., & Cahoon, D. R. (2002). Responses of coastal wetlands to rising sea level. *Ecology*, *83*(10), 2869–2877. https://doi.org/10.1890/0012-9658(2002)083[2869:rocwtr]2.0.co;2

Morris, J. T., & Whiting, G. J. (1986). Emission of gaseous carbon dioxide from salt-marsh sediments and its relation to other carbon losses. *Estuaries*, *9*(1), 9–19. https://doi.org/10.2307/1352188

Mudd, S. M. (2011). The life and death of salt marshes in response to anthropogenic disturbance of sediment supply. *Geology*, *39*(5), 511–512. https://doi.org/10.1130/focus052011.1

Mudd, S. M., Howell, S. M., & Morris, J. T. (2009). Impact of dynamic feedbacks between sedimentation, sea-level rise, and biomass production on near-surface marsh stratigraphy and carbon accumulation. *Estuarine, Coastal and Shelf Science*, *82*(3), 377–389. https://doi.org/10.1016/j.ecss.2009.01.028

Mueller, P., Ladiges, N., Jack, A., Schmiedl, G., Kutzbach, L., Jensen, K., & Nolte, S. (2019). Assessing the long-term carbon-sequestration potential of the semi-natural salt marshes in the European Wadden Sea. *Ecosphere*, *10*(1). https://doi.org/10.1002/ecs2.2556

Mueller, P., Schile-Beers, L. M., Mozdzer, T. J., Chmura, G. L., Dinter, T., Kuzyakov, Y., et al. (2018). Global-change effects on early-stage decomposition processes in tidal wetlands-implications from a global survey using standardized litter. *Biogeosciences*, *15*(10), 3189–3202. https://doi.org/10.5194/bg-15-3189-2018

Nguyen, T., Hoang, H., Thi, B., & Anh, K. (2015). The removal of heavy metals by iron mine drainage sludge and *Phragmites australis*. *Environmental Research Letters*, *10*(11), 115006. https://doi.org/10.1088/1748-9326/10/11/115006

Nyman, J. A., Walters, R. J., Delaune, R. D., & Patrick, W. H. (2006). Marsh vertical accretion via vegetative growth. *Estuarine, Coastal and Shelf Science*, *69*(3–4), 370–380. https://doi.org/10.1016/j.ecss.2006.05.041

Ouyang, X., & Lee, S. Y. (2020). Improved estimates on global carbon stock and carbon pools in tidal wetlands. *Nature Communications*, *11*(1), 1–7. https://doi.org/10.1038/s41467-019-14120-2

Pavao-Zuckerman, M. A. (2008). Soil ecology. In S. E. Jørgensen & B. D. Fath (Eds.), *Encyclopedia of ecology* (pp. 3277–3283). Elsevier Inc. https://doi.org/10.1016/B978-008045405-4.00850-8

Pendleton, L., Donato, D. C., Murray, B. C., Crooks, S., Jenkins, W. A., Sifleet, S., et al. (2012). Estimating global "blue carbon" emissions from conversion and degradation of vegetated coastal ecosystems. *PLoS One*, *7*(9), e43542. https://doi.org/10.1371/journal.pone.0043542

Perillo, G. M. E., Wolanski, E., Cahoon, D. R., & Hopkinson, C. S. (2009). In G. M. E. W. E. Perillo, D. R. Cahoon, & C. S. Hopkinson (Eds.), *Coastal wetlands: An integrated ecosystem approach, II)*. Elsevier.

Prescott, C. E. (2010). Litter decomposition: What controls it and how can we alter it to sequester more carbon in forest soils? *Biogeochemistry*, *101*(1), 133–149. https://doi.org/10.1007/s10533-010-9439-0

Prieto, I., Stokes, A., & Roumet, C. (2016). Root functional parameters predict fine root decomposability at the community level. *Journal of Ecology*, *104*(3), 725–733. https://doi.org/10.1111/1365-2745.12537

Puppin, A., Roner, M., Finotello, A., Ghinassi, M., Tommasini, L., Marani, M., & D'Alpaos, A. (2022). Analysis of organic matter decomposition in the salt marshes of the venice lagoon (Italy) using standard litter bags [Dataset]. Research Data Unipd. https://doi.org/10.25430/research-data.cab.unipd.it.00000762

Ratliff, K. M., Braswell, A. E., & Marani, M. (2015). Spatial response of coastal marshes to increased atmospheric CO2. *Proceedings of the National Academy of Sciences*, *112*(51), 15580–15584. https://doi.org/10.1073/pnas.1516286112

Reents, S., Mueller, P., Tang, H., Jensen, K., & Nolte, S. (2021). Plant genotype determines biomass response to flooding frequency in tidal wetlands. *Biogeosciences*, *18*(2), 403–411. https://doi.org/10.5194/bg-18-403-2021

Rogers, K., Kelleway, J. J., Saintilan, N., Megonigal, J. P., Adams, J. B., Holmquist, J. R., et al. (2019). Wetland carbon storage controlled by millennial-scale variation in relative sea-level rise. *Nature*, *567*(7746), 91–95. https://doi.org/10.1038/s41586-019-0951-7

Roner, M., D'Alpaos, A., Ghinassi, M., Marani, M., Silvestri, S., Franceschinis, E., & Realdon, N. (2016). Spatial variation of salt-marsh organic and inorganic deposition and organic carbon accumulation: Inferences from the Venice lagoon, Italy. *Advances in Water Resources*, *93*, 276–287. https://doi.org/10.1016/j.advwatres.2015.11.011

Roner, M., Ghinassi, M., Finotello, A., Bertini, A., Combourieu-nebout, N., Donnici, S., et al. (2021). Detecting the Delayed signatures of changing sediment supply in salt-marsh landscapes: The case of the venice lagoon (Italy). *Frontiers in Marine Science*, *8*. https://doi.org/10.3389/fmars.2021.742603

Rybczyk, J. M., & Cahoon, D. R. (2002). Estimating the potential for submergence for two wetlands in the Mississippi River delta. *Estuaries*, *25*(5), 985–998. https://doi.org/10.1007/BF02691346

Rydin, H., & Jeglum, J. (2006). In H. Rydin & J. Jeglum (Eds.), *The biology of peatlands*. Oxford University Press.

Sanderman, J., & Amundson, R. (2013). Biogeochemistry of decomposition and detrital processing. In H. D. Holland & K. K. Turekian (Eds.), *Treatise on geochemistry* (Second EditionVol. 10, pp. 217–272). Elsevier Inc. https://doi.org/10.1016/B978-0-08-095975-7.00807-X

Scarton, F., Day, J. W., & Rismondo, A. (2002). Primary production and decomposition of *Sarcocornia fruticosa* (L.) scott and *Phragmites australis* Trin. Ex Steudel in the Po delta, Italy. *Estuaries*, *25*(3), 325–336. https://doi.org/10.1007/BF02695977

Sifleet, S., Pendleton, L., & Murray, B. C. (2011). *State of the science on coastal blue carbon A summary for policy makers*. Nicholas Institute for Environmental Policy Solutions. Retrieved from http://scholar.google.com/scholar?hl=en&btnG=Search&q=intitle:State+of+the+Science+on+Coastal+Blue+Carbon+A+Summary+for+Policy+Makers#0

Silver, W. L., & Miya, R. K. (2001). Global patterns in root decomposition: Comparisons of climate and litter quality effects. *Oecologia*, *129*(3), 407–419. https://doi.org/10.1007/s004420100740

Silvestri, S. (2000). *La vegetazione alofila quale indicatore morfologico negli ambienti a marea*. University of Padova.

Silvestri, S., Defina, A., & Marani, M. (2005). Tidal regime, salinity and salt marsh plant zonation. *Estuarine, Coastal and Shelf Science*, *62*(1–2), 119–130. https://doi.org/10.1016/j.ecss.2004.08.010

Singh, J. S., & Gupta, S. R. (1977). Plant decomposition and soil respiration in terrestrial ecosystems. *The Botanical Review*, *43*(4), 449–528. https://doi.org/10.1007/BF02860844

Soliani, L. (2019). *Statistica non parametrica classica e moderna*. Piccin Nuova Libraria S.p.A.

Sprent, P., & Smeeton, N. C. (2000). *Applied nonparametric statistical methods* (3rd ed.). Chapman and Hall/CRC.

Stagg, C. L., Baustian, M. M., Perry, C. L., Carruthers, T. J. B., & Hall, C. T. (2018). Direct and indirect controls on organic matter decomposition in four coastal wetland communities along a landscape salinity gradient. *Journal of Ecology*, *106*(2), 655–670. https://doi.org/10.1111/1365-2745.12901

Temmerman, S., Meire, P., Bouma, T. J., Herman, P. M. J., Ysebaert, T., & De Vriend, H. J. (2013). Ecosystem-based coastal defence in the face of global change. *Nature*, *504*(7478), 79–83. https://doi.org/10.1038/nature12859

Tognin, D., D'Alpaos, A., Marani, M., & Carniello, L. (2021). Marsh resilience to sea-level rise reduced by storm-surge barriers in the Venice Lagoon. *Nature Geoscience*, *14*(12), 906–911. https://doi.org/10.1038/s41561-021-00853-7

Tognin, D., Finotello, A., D'Alpaos, A., Viero, D. P., Pivato, M., Mel, R. A., et al. (2022). Loss of geomorphic diversity in shallow tidal embayments promoted by storm-surge barriers. *Science Advances*, *8*(13). https://doi.org/10.1126/sciadv.abm8446

Turner, R., Swenson, E., & Milan, C. (2002). Organic and inorganic contributions to vertical accretion in salt marsh sediments. In M. P. Weinstein & D. A. Kreeger (Eds.), *Concepts and controversies in tidal marsh ecology* (pp. 583–595). Springer Netherlands. https://doi.org/10.1007/0-306-47534-0_27

Ursino, N., Silvestri, S., & Marani, M. (2004). Subsurface flow and vegetation patterns in tidal environments. *Water Resources Research*, *40*(5). https://doi.org/10.1029/2003WR002702

Valiela, I., Bowen, J. L., & York, J. K. (2001). Mangrove forests: One of the world's threatened major tropical environments. *BioScience*, *51*(10), 807–815. https://doi.org/10.1641/0006-3568(2001)051[0807:mfootw]2.0.co;2

Valiela, I., Teal, J. M., Allen, S. D., Van Etten, R., Goehringer, D., & Volkmann, S. (1985). Decomposition in salt marsh ecosystems: The phases and major factors affecting disappearance of above-ground organic matter. *Journal of Experimental Marine Biology and Ecology*, *89*(1), 29–54. https://doi.org/10.1016/0022-0981(85)90080-2

Wang, F., Kroeger, K. D., Gonneea, M. E., Pohlman, J. W., & Tang, J. (2019). Water salinity and inundation control soil carbon decomposition during salt marsh restoration: An incubation experiment. *Ecology and Evolution*, *9*(4), 1911–1921. https://doi.org/10.1002/ece3.4884

Wang, H., Piazza, S. C., Sharp, L. A., Stagg, C. L., Couvillion, B. R., Steyer, G. D., & McGinnis, T. E. (2017). Determining the spatial variability of wetland soil bulk density, organic matter, and the conversion factor between organic matter and organic carbon across coastal Louisiana, U.S.A. *Journal of Coastal Research*, *33*(3), 507–517. https://doi.org/10.2112/JCOASTRES-D-16-00014.1

Ward, S. E., Orwin, K. H., Ostle, N. J., Briones, M. J. I., Thomson, B. C., Griffiths, R. I., et al. (2015). Vegetation exerts a greater control on litter decomposition than climate warming in peatlands. *Ecology*, *96*(1), 113–123. https://doi.org/10.1890/14-0292.1

White, D. A., Weiss, T. E., Trapani, J. M., & Thien, L. B. (1978). Productivity and decomposition of the dominant salt marsh plants in Louisiana. *Ecology*, *59*(4), 751–759. https://doi.org/10.2307/1938779

Yarwood, S. A. (2018). The role of wetland microorganisms in plant-litter decomposition and soil organic matter formation: A critical review. *FEMS Microbiology Ecology*, *94*(11), 1–17. https://doi.org/10.1093/femsec/fiy175

Yousefi Lalimi, F., Silvestri, S., D'Alpaos, A., Roner, M., & Marani, M. (2018). The spatial variability of organic matter and decomposition processes at the Marsh Scale. *Journal of Geophysical Research: Biogeosciences*, *123*(12), 3713–3727. https://doi.org/10.1029/2017JG004211

References From the Supporting Information

Carniello, L., Defina, A., Fagherazzi, S., & D'Alpaos, L. (2005). A combined wind wave-tidal model for the Venice Lagoon, Italy. *J. Geophys. Res.Earth Surface*, *110*(F4), 1–15. https://doi.org/10.1029/2004JF000232

Carniello, L., D'Alpaos, A., & Defina, A. (2011). Modeling wind waves and tidal flows in shallow micro-tidal basins. *Estuarine, Coastal and Shelf Science*, *92*(2), 263–276. https://doi.org/10.1016/j.ecss.2011.01.001

Carniello, L., Defina, A., & D'Alpaos, L. (2012). Modeling sand-mud transport induced by tidal currents and wind waves in shallow microtidal basins: Application to the Venice Lagoon (Italy). *Estuarine, Coastal and Shelf Science*, *102–103*, 105–115. https://doi.org/10.1016/j.ecss.2012.03.016

Finotello, A., Marani, M., Carniello, L., Pivato, M., Roner, M., Tommasini, L., & D'Alpaos, A. (2020). Control of wind-wave power on morphological shape of salt marsh margins Water. *Science Engineering*, *13*(1), 45–56. https://doi.org/10.1016/j.wse.2020.03.006

Defina, A. (2000). Two-dimensional shallow flow equations for partially dry areas. *Water Resources Research*, *36*(11), 3251–3264. https://doi.org/10.1029/2000WR900167