Blue Carbon Stocks and Exchanges Along the California Coast

Melissa A. Ward[1,2], Tessa M. Hill[1], Chelsey Souza[1], Tessa Filipczyk[1], Aurora M. Ricart[1,3], Sarah Merolla[1], Lena R. Capece[1], Brady C. O'Donnell[1], Kristen Elsmore[1], Walter C. Oechel[2], and Kathryn M. Beheshti[4].

Correspondence to: Melissa A. Ward (maward@ucdavis.edu)

[1]Bodega Marine Laboratory, University of California, Davis, 95616, USA
[2]San Diego State University, 92182, USA
[3]Bigelow Laboratory for Ocean Sciences, 04544, USA
[4]University of California, Santa Cruz, 95064, USA

Abstract. Salt marshes and seagrass meadows can sequester and store high quantities of organic carbon (OC) in their sediments relative to other marine and terrestrial habitats. Assessing carbon stocks, carbon sources, and the transfer of carbon between habitats within coastal seascapes are each integral in identifying the role of blue carbon habitats in coastal carbon cycling. Here, we quantified carbon stocks, sources, and exchanges in seagrass meadows, salt marshes, and unvegetated sediments in six bays along the California coast. In the top 20 cm of sediment, the salt marshes studied here contained approximately twice as much OC as did seagrass meadows, 4.92 ± 0.36 kg OC m^{-2} compared to 2.20 ± 0.24 kg OC m^{-2}, respectively. Both salt marsh and seagrass sediment carbon stocks were higher than previous estimates from this region but lower than global and U.S.-wide averages, respectively. Seagrass-derived carbon was deposited annually into adjacent marshes during fall seagrass senescence. However, isotope mixing models estimate that negligible amounts of this seagrass material were ultimately buried in underlying sediment. Rather, the vast majority of OC in sediment across sites was likely derived from planktonic/benthic diatoms and/or C3 salt marsh plants.

Keywords: Carbon cycling, lateral fluxes, seagrass meadow, salt marsh, carbon sequestration

1 Introduction

As carbon dioxide (CO_2) concentrations in the oceans and atmosphere continue to rise,

interest in measuring the relative quantities of carbon stored within natural ecosystems has

increased. These assessments can help improve global and regional climate models, the

prediction of future CO_2 concentrations related to sources and sinks, and our broader

understanding of nature-based climate change solutions (Serrano et al., 2019). Coastal habitats

including seagrasses, salt marshes, and mangroves have earned the moniker 'blue carbon'

habitats for their ability to store disproportionally high levels of organic carbon (OC) in their

39 sediments relative to other habitat types, thereby potentially serving in a management context to

40 provide carbon mitigation (Lovelock and Duarte, 2019; McLeod et al., 2011). This ability can be

41 largely attributed to the tendency for these habitats to exhibit high sediment accretion rates and

42 low decomposition rates (Peck et al., 2020; Serrano et al., 2019). This value has led to increasing

43 interest in blue carbon habitats, given their conservation can prevent significant emissions of

44 carbon (Lovelock et al., 2017; Pendleton et al., 2012) and their restoration can lead to increased

45 drawdown of atmospheric CO_2 (Freedman et al., 2009; Greiner et al., 2013).

46

47 **1.1 Carbon stock assessments and their significance**

48   Despite global interest, many regions are still lacking basic information on carbon stocks

49 and burial rates in blue carbon habitats. This information forms the foundation for more

50 advanced scientific research and can be extremely valuable within management contexts to

51 develop informed local and regional climate assessments. Salt marshes and seagrass meadows

52 cover extensive portions of North America's west coast, yet their carbon stocks have been

53 relatively understudied compared to other habitats in North America and other blue carbon

54 habitats in different regions of the world (Ouyang and Lee, 2014; Postlethwaite et al., 2018).

55   Eelgrass (*Zostera marina*) is the dominant seagrass species in North America and

56 occupies coastal waterways from Alaska to Mexico (Green & Short, 2003). Until recently,

57 seagrass carbon stock data were almost entirely absent along the west coast of North America

58 (see Capece et al., 2019; Kauffman et al., 2020; O'Donnell, 2017; Poppe & Rybczyk, 2018;

59 Postlethwaite et al., 2018; Röhr et al., 2018). Of these studies, very few data came from Central

60 or Southern California seagrass meadows, despite the fact that this region represents a key

61 temperate to subtropical transition in *Z. marina*'s range (Cabello-Pasini et al., 2003). As a result

62   of the previous paucity of data from the west coast of North America, this region was not

63   represented in previous global syntheses of seagrass carbon stocks (Duarte et al., 2010;

64   Fourqurean et al., 2012; Mazarrasa et al., 2015). Existing work on global seagrass carbon storage

65   identifies that two species of seagrasses in the Mediterranean (*Posidonia oceanica)* and Australia

66   (*Posidonia australis*) store significantly more carbon than other seagrass species, including *Z.*

67   *marina* (Fourqurean et al., 2012; Lavery et al., 2013; Prentice et al., 2020). These *Posidonia*

68   species were overrepresented in some early assessments of total global seagrass carbon storage –

69   making these global estimates unreliable when applied to management decisions or climate

70   models (Johannessen & Macdonald, 2016; Kennedy et al., 2010). In reality, the geographic and

71   interspecies variability in carbon stocks is likely greater than was initially anticipated (Macreadie

72   et al., 2018; Postlethwaite et al., 2018).

73   Salt marsh carbon stocks are similarly understudied in western North America, with

74   published carbon stock data from only four estuaries in the region and very minimal spatial

75   coverage and analyses performed in three of these locations (Brevik and Homburg, 2004;

76   Callaway et al., 2012; Kauffman et al., 2020; Patrick and DeLaune, 1990). Existing analyses of

77   North American freshwater wetlands and salt marshes are typically dominated by studies along

78   the east coast (Nahlik & Fennessy, 2016; Wilkinson et al., 2018; with the exception of Holmquist

79   et al. 2018). Within west coast studies, considerable variation in carbon stocks is observed

80   (Callaway et al., 2012; Chmura et al., 2003). Similar to seagrass meadows, these regions have

81   distinctly different oceanographic and geomorphological regimes, which can drive differences in

82   sediment carbon storage. Thus, seagrass and salt marsh carbon storage data collected from

83   understudied regions and across varying environmental gradients are necessary for understanding

84   carbon stock variability and its drivers.

85

1.2 Blue carbon sources and drivers

Despite an increasing breadth of information on blue carbon habitats in the last decade, a

number of questions remain before we can fully understand their role in regional carbon cycling

and climate adaptation (Macreadie et al., 2019). We do not yet fully understand what drives

variation in carbon stocks – a key aspect of ensuring the protection and enhancement of these

carbon services in the future. Previous work demonstrates that numerous factors can control

carbon accumulation in coastal sediments including overlying biomass, topography, hydrology,

mineralogy, and remineralization rates (Kelleway et al., 2016; Lima et al., 2020; Mazarrasa et

al., 2018; Prentice et al., 2019). In particular, sediment grain size has been demonstrated as a

significant predictor of carbon stocks (Dahl et al., 2016; O'Donnell, 2017; Serrano et al., 2016),

as it affects decomposition rates, which is likely related to the deposition of small particles and

low resuspension from the attenuation of water flow by seagrasses (Bos et al., 2007; Conley et

al., 2017; Gambi et al., 1990; Hendriks et al., 2008). Through similar pathways, fine, silty

sediments trapped in tidal salt marshes can also increase carbon storage (e.g. Zhou et al., 2007).

The relative importance of each potential driver of carbon stock variability may be highly

dependent on the environmental setting, species under study, and the interactive effects of these

drivers. For example, in Australian salt marshes, overlying vegetation was shown to only

significantly affect carbon stocks in sandy and mixed grain size sediments, having no effect on

stocks in fine sediments (Kelleway et al., 2016). However, in other cases vegetation has been the

primary predictor of carbon stocks in salt marshes, irrespective of grain size (Lovelock et al.,

2014; Saintilan et al., 2013). Similar region and species-specific complexities between carbon

stocks, overlying vegetation, and mineralogy have been described in seagrass meadows (e.g.

108   Lima et al., 2020; Serrano et al., 2016), meriting further investigation to understand these

109   complex interactions.

110   Knowing the relative contributions of locally produced (autochthonous) and imported

111   (allochthonous) carbon also elucidates the underlying mechanisms by which blue carbon habitats

112   store and accumulate carbon. Overlying vegetation can significantly impact sediment carbon

113   stocks; however, it is rarely the dominant source of carbon buried within blue carbon habitats

114   (Kennedy et al., 2010; Lewis et al., 2020; Mazarrasa et al., 2015; Prentice et al., 2019). In many

115   cases, contributions from terrestrial habitats, macroalgae, and suspended particulate organic

116   material contribute as much or more to carbon buried in blue carbon sediments than do

117   autochthonous sources (Drexler et al. 2020; Kennedy et al., 2010; Leorri et al., 2018; Ricart et

118   al., 2020). Thus, local primary production could significantly contribute to net annual carbon

119   drawdown within a given habitat yet play a minor role in carbon burial due to lateral export or

120   remineralization of this carbon. Understanding this phenomenon elucidates the role of blue

121   carbon habitats in broader, system-wide energy flows and carbon cycling (Hyndes et al., 2014;

122   Ricart et al., 2015). For instance, much of the carbon from laterally exported biomass may be

123   remineralized in the water column or as wrack in nearby habitats (Attard et al., 2019; Liu et al.,

124   2019), serving an important ecological role and altering the production and biogeochemical

125   cycles of recipient systems (Hyndes et al., 2014; Ince et al., 2007; Valiela and Cole, 2002).

126   Given that blue carbon habitats are highly productive (Duarte & Cebrián, 1996), support high

127   carbon burial (e.g. McLeod et al., 2011), and can co-occur within small geographic ranges

128   (Alongi, 2018; Bouillon & Connolly, 2009), laterally exported carbon from one blue carbon

129   habitat may be entrapped and buried in a neighboring habitat. From this landscape-scale

130   perspective, exchanges between blue carbon habitats could increase their capacity for carbon

131   burial or alter the sources of buried carbon. While some coastal studies have estimated lateral

132   carbon fluxes (Jiménez et al., 2017; Liu et al., 2019), very few studies place these lateral fluxes

133   into landscape-scale contexts that also address carbon burial in recipient habitats (Bouillon &

134   Connolly, 2009; Duarte & Krause-Jensen, 2017; Ricart et al., 2017).

135

136   The study presented here addresses this globally relevant topic of research in a relatively

137   understudied region by answering the following questions:

138   1) How much organic carbon is stored in seagrass and salt marsh sediments across a

139   California latitudinal gradient?

140   2) What are the sources of carbon buried within these sediments?

141   3) Is carbon exchanged between blue carbon habitats within the coastal landscape?

142

143   **2   Methods**

144   **2.1 Study Sites**

145   Sediment cores from salt marshes and seagrass meadows, along with neighboring

146   unvegetated sediments (hereafter 'bare sediment' near seagrass meadows, and 'pan' near salt

147   marshes) were collected from six bays across a latitudinal gradient in California (Fig. 1a). A total

148   of 82 sediment cores were collected, 30 of which have been discussed previously (O'Donnell,

149   2017) and are included here for comparison. The number of cores collected in each site and

150   general site characteristics are described in Table 1. While all sampled seagrass meadows were

Figure 1: Sediment cores were collected from six estuaries across a latitudinal gradient in California, where N = number of cores (A). Seagrass wrack deposition into Walker salt marsh from surrounding seagrass meadows (Merkel & Associates, 2017) was estimated, while sediment cores were taken from all four labeled habitats (B). Three sediment cores were collected beneath persistent wrack lines in Walker salt marsh, while three were collected from areas that do not receive regular, annual tidal deposition of wrack (C).

dominated by a single seagrass species (*Zostera marina)*, salt marshes contained a mixed community of halophytes, predominantly composed of pickleweed (*Sarcocornia pacifica*), and to a lesser extent salt grass (*Distichlis spicata*) and marsh jaumea (*Jaumea carnosa)*. Bare sediment cores were collected in unvegetated areas nearby each seagrass meadow at a minimum of 20 meters away from the meadows and at similar depths. Pan cores

171 were collected from patches of unvegetated sediment (2-4m diameter) found within the salt

172 marsh interior, a natural and semi-permanent feature of salt marsh habitats formed by elevational

173 depressions (Escapa et al., 2015). None of our sampling sites were actively restored and, to our

174 knowledge, respective vegetation has persisted through time.

175   We estimated OC stocks in seagrass, salt marsh, pan, and bare sediments (see details below).

176 In addition, we analyzed sediment OC sources for two individual salt marshes in Elkhorn Slough

177 ('Elkhorn salt marsh') and Tomales Bay ('Walker salt marsh'), and we examined carbon

178 exchange between a single seagrass meadow and the neighboring Walker salt marsh. Walker salt

179 marsh is located approximately 5.5 km from the mouth of Tomales Bay and is in close proximity

180 to extensive meadows of *Z. marina* – the dominant seagrass in each of the meadows sampled

181 (Fig. 1b). This marsh lies where the mouth of Walker Creek meets bay waters, and thus can

182 receive terrestrial and riverine inputs, while simultaneously receiving marine inputs from tidal

183 exchange.

Bay	Coordinates	Number of cores	Site description
Humboldt Bay	40°48'12 N, 124°10'52 W	Seagrass: 4 Bare sediment: 3	A large semi-enclosed bay with a narrow mouth, stabilized by jetties. Primary freshwater sources come from the Elk River and its tributaries, with Freshwater Creek inputs entering the bay near the sampling site. Freshwater input is small relative to tidal flushing but can locally effect sedimentation rates within the bay (Schlosser and Eicher, 2012)
Bodega Harbor	38°19'60 N, 123°02'53 W	Seagrass: 15 Bare sediment: 3	There are no significant rivers or creeks that provide freshwater or sediment inputs. Sediment input in Bodega Bay is largely controlled by jetties at the mouth of the harbor and is influenced by its history of routine dredging to maintain shipping channels and control wind-blown sand entering from nearby dunes

Tomales Bay	38°10'17 N, 122°54'46 W	Seagrass: 15 Bare sediment: 9 Salt Marsh: 6	A long, narrow bay that receives periodic freshwater inputs from two primary tributaries, Walker Creek and Lagunitas Creek, which peaks seasonally during winter storms.
Elkhorn Slough	36°49'15 N, 121°45'06 W	Salt Marsh: 11 Pan: 3	Receives periodic freshwater inputs from the Old Salinas River and Bennett and Moro Coho Sloughs. After a long history of increasing sediment deposition and slowing tidal flow to the bay, levee breaches and removals in 1953 rapidly converted it to a higher tidal energy, erosional system, as it remains today (Van Dyke and Wasson, 2005). As a result, the extent of 'high-quality' salt marsh in 2000 was 23% of what it had been a century earlier (Van Dyke and Wasson, 2005)
Newport Bay	33°38'07 N, 117°53'17 W	Seagrass: 4 Bare sediment: 4	An urban bay, that receives terrestrial and urban inputs from the San Diego Creek Watershed. After a long history of high sedimentation and water quality nutrient impairments from urban and agricultural sources, a program was implemented in 1998 to meet water quality targets. Since its commencement, both sedimentation and nutrients in the bay have significantly decreased (Trimble, 2003).
Mission Bay	32°47'24 N, 117°13'40 W	Seagrass: 3 Bare sediment: 2	An urban, enclosed bay, that receives terrestrial and urban input from the San Diego River watershed and the Rose Creek tributary. Experiences long residence times and is primarily controlled by tidal flushing processes rather than the minimal freshwater inputs (Cyronak et al., 2018; Largier et al., 1997)

Table 1: Location and brief description of each bay sampled in this study, along with the total number of cores collected from each habitat type in each of these bays.

2.2 Carbon stocks

All sediment cores were sampled by manually inserting transparent, open-barrel PVC pipes (20 cm length, 5.08 cm diameter). Compaction occurred in 19% of cores, and a compaction

191 factor was applied when calculating carbon stocks according to Howard et al. (2014). Once

192 extracted, cores were capped and transported to the laboratory upright to prevent mixing of

193 sediment layers. Cores were then immediately extruded into sections at 2 cm intervals. Coarse

194 living plant material (> 1cm) was manually removed. Each section was dried at 60°C and

195 weighed, and dry bulk density (DBD) was determined by dividing dry bulk mass by the volume

196 of sampling interval.

197 Each section was then homogenized and divided into three subsamples of 10 g each, and the

198 remaining sample was archived. One of the three subsamples was acidified using 1.12 M HCl to

199 remove and measure total inorganic carbon (Milliman, 1974). The second subsample was

200 analyzed for total organic material (TOM) by loss-on-ignition method for 4 hours at 550° C

201 (Dean, 1974). Total C content (%) and $\delta^{13}C$ were additionally determined on a set of 44 of the

202 acidified subsamples randomly selected across each habitat type using an elemental analyzer

203 (PDZ Europa ANCA-GSL, SD ± 0.25 %) integrated with a continuous flow isotope ratio mass

204 spectrometer (PDZ Europa 20-20, SD ± 0.2 ‰) at the UC Davis Stable Isotope Facility. $\delta^{13}C$

205 ratios are expressed in parts per thousand (‰) relative to VPDB (Vienna Pee Dee Belemnite)

206 according to standard notation ($\delta^{13}C = [(R_{sample} / R_{standard}) - 1] \times 1000$, where R is the ratio

207 $^{13}C/^{12}C$). Organic carbon in each core section was then estimated using a power model developed

208 between measured TOM and measured OC in this set of subsamples (as in Craft et al., 1991).

209 Specifically, the equation $y = 0.22x^{1.1}$ was applied ($r^2 = 0.84$; Fig. S1). A power model was

210 selected over a linear model for these data to avoid negative estimates of carbon stocks at low

211 levels of TOM (Fig. S1). Carbon stocks were determined by multiplying DBD (g cm^{-3}) by %

212 OC.

In each core section, the proportion of fine sediments – the silt and clay fraction (<63 μm), hereafter "mud" – was quantified. The grain size analysis was conducted by mass loss in the third subsample from each section in 70 of the 82 total cores. The subsample was rinsed through a 63 μm sieve with deionized water, and the remaining sediments were dried at 60°C and re-weighed.

Comparable to previous studies, a core depth of 20 cm was selected (Fourqurean et al. 2012; Prentice et al., 2020; Röhr et al., 2018). As such, stock estimates are presented in kg OC m^{-2} in the top 20 cm of sediment. However, given the selected depths for stock estimates vary throughout the literature, we extrapolate all data to 1 m when comparing across studies. Some research suggests that OC content remains relatively constant below 10 cm to depths up to 1 m, making this extrapolation appropriate (Callaway et al., 2012; Prentice et al., 2020; Fig. S3), while evidence of downcore variability in other studies makes extrapolation less appropriate (e.g., St. Laurent et al., 2020; Serrano et al., 2012). When comparing the carbon stocks estimated here to those in previous studies, we include each studies' sampled core depths for clarity.

2.3 Carbon sources and exchange

We applied mixing models to assess salt marsh sediment carbon sources to understand within-estuary exchange of carbon among the sampled habitats. Two separate mixing models were produced for 1) Elkhorn salt marsh in Elkhorn Slough and Walker salt marsh in Tomales Bay, and 2) salt marsh sediments under seagrass wrack, versus not under wrack in Walker salt marsh. At Walker salt marsh, a total of six sediment cores were collected. Three of these sediment cores were collected from the interior marsh, while three were collected from underneath a seagrass wrack line along the tidal edge of the marsh (Fig. 1c). *Z. marina* in this

236   region is known to undergo a period of senescence as photoperiod shortens in the fall, as is

237   common in all temperate seagrasses (Fourqurean et al., 1997). Historical imagery of the site

238   shows persistent seagrass wrack concentrated along these tide lines that consistently appears in

239   early fall as seagrass senesces (Fig. S2; Google Earth, 2020). The biomass of seagrass wrack

240   along this tide line at the time of core collection was quantified along an 80 m transect within the

241   marsh by collecting all seagrass present in a 1 m^2 quadrat every 10 meters. This material was

242   taken back to the lab, sorted into above ground biomass (AGB) and below ground biomass

243   (BGB), rinsed, dried (60°C), and weighed. Biomass data from seagrass in nearby meadows was

244   previously published (see O'Donnell, 2017) and is used here for reference. Sediment cores

245   collected from beneath wrack lines were sectioned at 2 cm intervals, analyzed for total OC and

246   δ^{13}C content according to the methods described above. Total nitrogen content was also

247   determined on an unacidified portion of the same 44 sediment subsamples used in carbon

248   analyses (ThermoFinnigan Flash 1112 Series elemental analyzer, SD ± 0.02%). Data from

249   Walker salt marsh are also displayed as 'shallow' (surface sediment to 10 cm deep) and 'deep'

250   (10 cm and deeper), to facilitate interpretation of changes with depth.

251   The contributions of carbon sources to each core section were then estimated with a mixing

252   model using δ^{13}C and N:C ratios as tracers. Given δ^{15}N can be altered during early diagenesis

253   (e.g. Benner et al., 1991), we selected δ^{13}C and C:N ratios as the primary geochemical tracers,

254   and therefore did not include fractionation factors into the model (Craven et al., 2017). N:C

255   ratios are utilized rather than C:N ratios because mixed fractions returned by the model are based

256   on the denominator, and thus ^{13}C /^{12}C and N:C both estimate the fractional contribution of ^{12}C

257   (Craven et al., 2017; Perdue and Koprivnjak, 2007). However, C:N ratios are used in text and

258   figures given this format is more typically presented in the literature. We used a Bayesian mixing

259   model SIAR 4.2 (Parnell and Jackson, 2013) to estimate the contributions of several source

260   groups selected according to the dominant plant and algal species observed within the selected

261   salt marshes. The sources included C3 salt marsh plants, C4 salt marsh plants (*Distichlis*

262   *spicata)*, seagrass *(Z. marina)*, and plankton/benthic diatoms. Seagrass geochemistry was

263   estimated from seagrass leaves collected from meadows near the selected salt marshes in

264   Newport Bay, Tomales Bay, and Bodega Bay (Fig. 1a; Capece et al., 2019). Representative

265   geochemistry for all other sources has been published previously and was used here, from

266   samples collected in San Francisco Bay, approximately 60 km south of Walker salt marsh (see

267   Cloern et al., 2002 for full methods). All salt marsh sediment cores were collected from areas of

268   the marsh that were dominated by the aforementioned species, and as such, other marsh species

269   were not considered (e.g., *Spartina sp.).* While lower densities of other C3 marsh plants may

270   have been present in some cored sites, the variability demonstrated by the selected source

271   samples likely encompasses much of this variability given the utilization of the same

272   photosynthetic pathway (Cloern et al., 2002). Plankton samples were collected from estuarine

273   water samples while benthic diatom samples were collected from both salt marsh surface

274   sediments and neighboring mudflats (Cloern et al., 2002). These sources have similar isotopic

275   values and are pooled here and referred to as "diatoms" for simplicity, acknowledging that this

276   encompasses contributions from multiple planktonic and benthic sources. Given the overlapping

277   isotopic values of C3 plants and diatoms, these two sources were pooled in mixing models,

278   allowing an estimate of diatoms and/or C3 plants contributions to marsh sediments, in addition to

279   contributions from seagrass and C4 plants.

280

281   **2.4 Statistical Analyses**

282   To evaluate decay of carbon through time, we tested for significant changes in OC in each

283   habitat type with core section depth (a proxy for time) by fitting data to a generalized linear

284   mixed model (GLMM) using maximum likelihood with 'depth' and 'site' as fixed effects and

285   'core' as a random effect, using a gamma distribution and log link function to account for non-

286   normality. In sites and habitats that demonstrated significant OC changes with depth, the rate of

287   decay was estimated from the slope of its associated model. In all analyses to follow, OC across

288   all sections in each core were averaged, and statistics were performed on these core averages.

289   After inspecting data for normality and homogeneity of variance, differences in OC and grain

290   size between habitat types and between sites were analyzed using simple linear models (SLM;

291   significance defined by $\alpha = 0.05$). Specifically, these data were fit to a linear model using

292   ordinary least squares with 'site' and 'habitat' (i.e., seagrass, salt marsh, bare sediment, pan) as

293   fixed effects, including their interaction. The relationships between TOM (%) and grain size

294   were analyzed using simple linear regressions, whereby a grain size filter was selectively applied

295   to determine the point at which the relationship between the two was no longer significant (p <

296   0.05). Differences in $\delta^{13}C$ (‰) or C:N ratios between sediment depth ('surface' versus 'deep'),

297   and between sediments collected under wrack versus not under wrack, were also tested with

298   SLMs using ordinary least squares with 'depth' or 'under wrack' as fixed effects. When

299   necessary, data were log transformed. Tukey's post-hoc analyses were conducted for multiple

300   comparisons. All statistical analyses were performed in R software (R Core Team, 2018).

301

302   **3   Results**

303   **3.1 Carbon Stock Assessments**

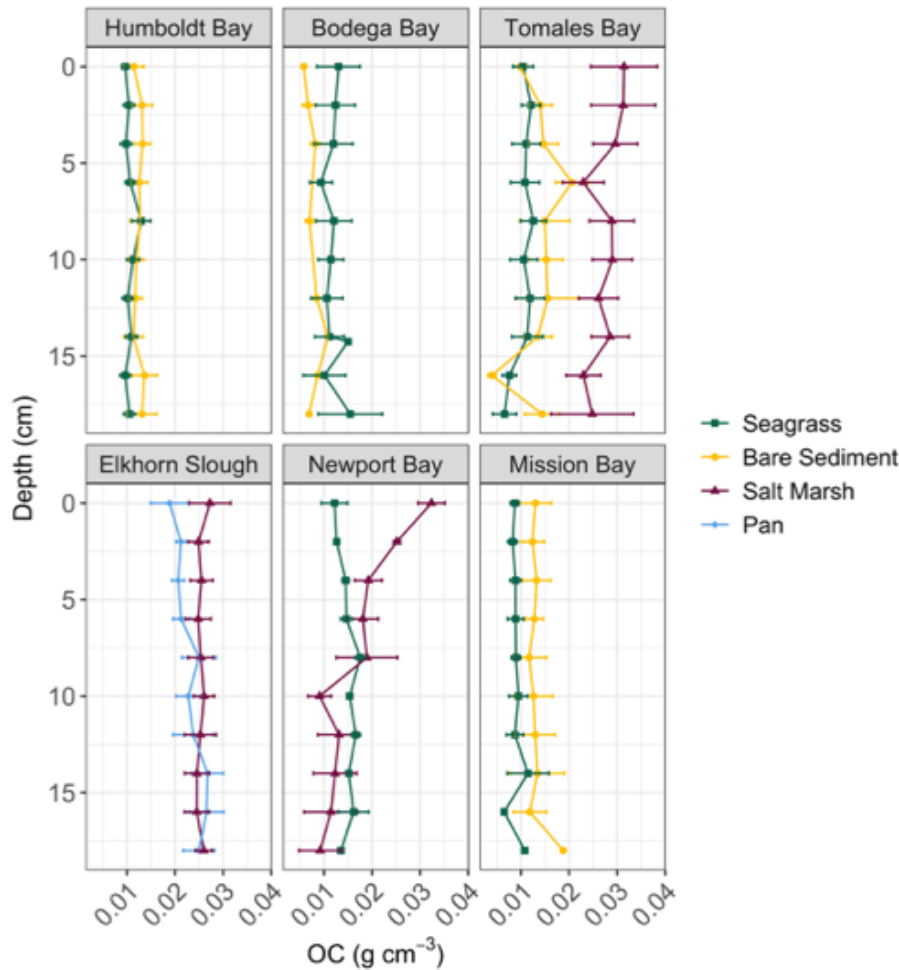



Figure 2 (left): Down core trends (surface = 0 cm) in average (± SE) OC (g cm^{-3}) for each habitat type and site. Data show an insignificant decline in OC through time (downcore) in all sites except Newport Bay salt marshes.

326

327   Down core OC demonstrates high variability, resulting in few significant differences in

328   OC with depth (Fig. 2). Specifically, only Newport Bay salt marsh sediments exhibited

329   significant loss of OC down core, which declined at a rate of 0.001 g C cm^{-3} per cm depth (Fig

330   2e; GLMM, T-value = -4.7, SE = 0.01, p < 0.05). Significant differences between OC stocks

331   emerged when cores were compared between habitats, with salt marshes containing significantly

332   more carbon than both bare sediment and seagrass meadows (Fig. 3b; SLM, p < 0.01, F-statistic

333   = 13.3, DF =3). Specifically, within the top 20 cm of sediment, salt marsh sediments contained

334   4.92 ± 0. kg OC m^{-2}, while seagrass meadows contained 2.20 ± 0.24 kg OC m^{-2} (Fig. 3b; mean ±

335   SE). Seagrass meadow carbon stocks were not significantly different than those of nearby bare

336   sediments (Tukey's post-hoc analysis, $p > 0.05$), which contained an average of 2.47 ± 0.32 kg

337   OC m^{-2}. Pan sediments also contained higher carbon than did seagrass meadows (Tukey's post-

338   hoc analysis, $p > 0.05$). A full list of carbon stocks is displayed in Table S1. Tukey's post-hoc

339   analysis indicates that in Tomales Bay, salt marshes contained significantly more carbon than

340   seagrass meadows (Fig. 3a; $p < 0.05$) while bare sediment and seagrass carbon stocks did not

341   significantly differ from one another ($p > 0.05$). We did not detect any other significant

342   differences when comparing carbon stocks across habitat types within each individual site

343   (Tukey's post-hoc analysis, $p > 0.05$).

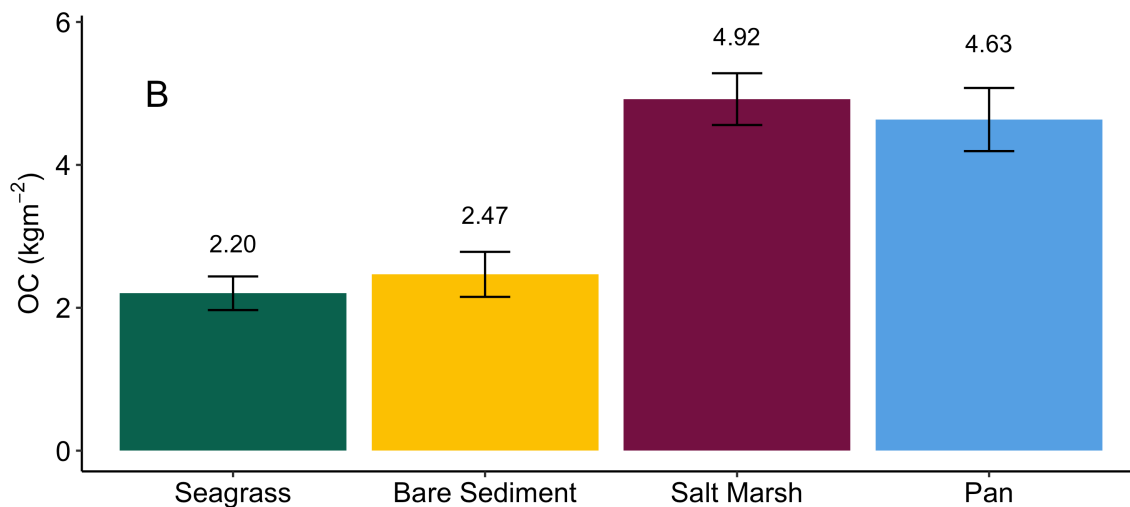
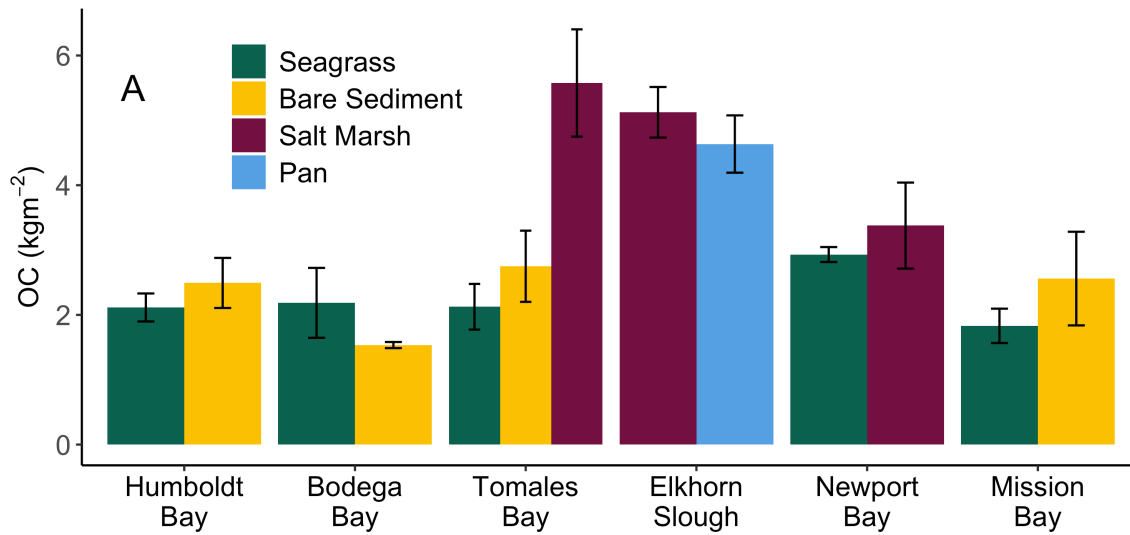



Figure 3: Average (± SE) total organic carbon (OC) stocks (kg m^{-2}) across all cores collected from each habitat type within each site in this study (A). Average (± SE) total organic carbon stocks (kg m^{-2}) across all cores collected within each habitat type in this study (B). Stocks are representative of the top 20 cm of sediment.

We observed a very strong relationship between grain size and storage of organic material, especially at lower TOM (%) values (Fig. 4). Specifically, the observed linear relationship between grain size and TOM is strongest when sediment is 8.5% TOM or less (linear model, r^2 =

354   0.95, p <0.05, F-statistic = 4957, DF = 269). However, as the proportion of fine sediments in

355   each sample increases, the relationship between grain size and TOM weakens rapidly (from r^2 =

356   0.73 to 0.53) after sediments consisting of more than 82% mud are included (linear model, p

357   <0.05, r^2 = 0.53, F-statistic = 233.1, DF = 201). Our results indicate that grain size was similar

358   between sites (SLM, p > 0.05, F-statistic = 2.7, DF = 3) but differed between habitat types (Fig.

359   5a; SLM, p < 0.05, F-statistic = 5.7, DF = 3), with salt marsh sediments demonstrating

360   significantly greater % mud than both seagrass and bare sediment (Tukey's post-hoc analysis, p

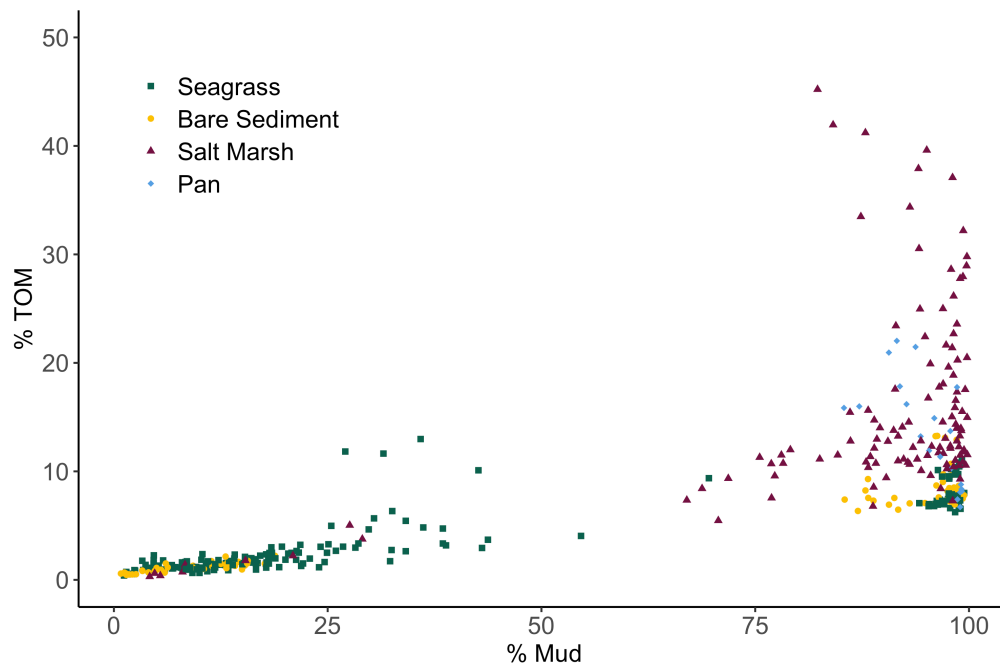
361   < 0.05). Recognizing that the sample size of pan cores was low, pan sediments displayed

362   comparable % mud to salt marsh sediments but there were no significant differences between

363   pan grain size and the other habitat types' grain sizes (Tukey's post-hoc analysis, p > 0.05).

364   Tukey's post-hoc analyses did not indicate any significant differences in grain size between

365   habitat types within each site (Fig. 5a; p > 0.05).

366

367
368   **Figure 4:** Biplot showing the relationship among total organic material (TOM) (%) of each 2-cm
369   core section from each habitat type plotted against sediment grain size (% Mud).

370

371

372

373

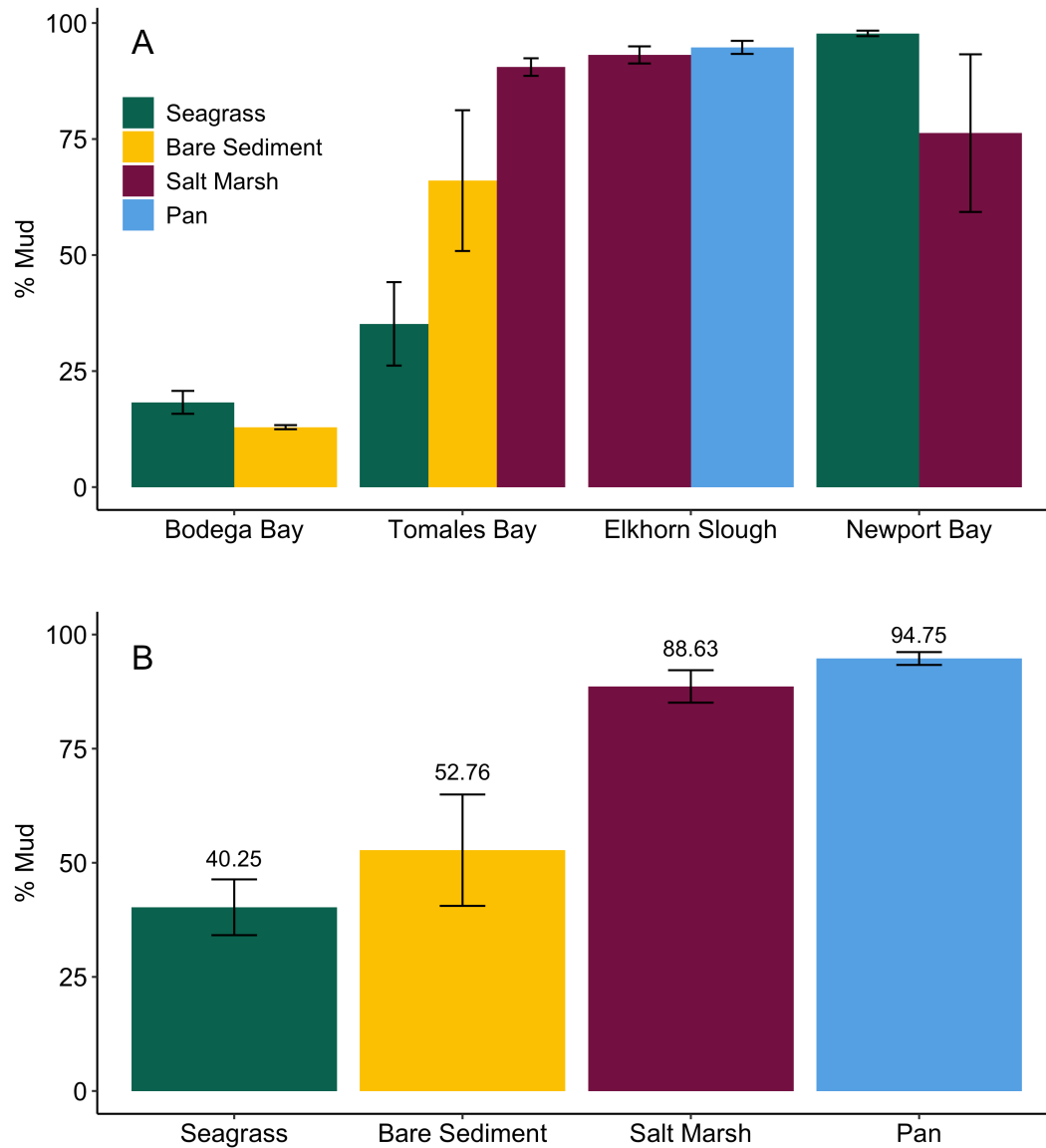



Figure 5: Average (± SE) % mud of all sediments quantified in each site within each habitat type (A), average (± SE) % mud in each habitat type across all sites measured (B).

3.2 Carbon Sources and Exchange

3.2.1   Wrack Deposition

379   While we conducted only one ground survey of seagrass wrack lines within Walker salt

380   marsh in October 2019, historical aerial imagery was utilized to verify that these wrack lines

381   appear reliably in this season in similar locations each year (Fig. S2). This phenomenon can be

382   viewed as far back as 2002, before which images are of low quality or unavailable (Google

383   Earth, 2020). These wrack lines typically persist throughout the winter, becoming indiscernible

384   via aerial imagery by spring. Previous data collected by the authors (O'Donnell, 2017) quantified

385   seagrass senescence within the nearby Tom's Point seagrass meadow (Fig. 1b), where average

386   summer seagrass biomass was 440 ± 59.4 g m^{-2} and winter seagrass biomass was 115 ± 16.5 g m^{-2}

387   (AGB and BGB, mean $\pm$ SE; Table 2). Within the neighboring Walker salt marsh, fall estimates

388   of wrack demonstrated that 106 ± 24.6 g m^{-2} of seagrass (dry weight) was deposited along tide

389   lines (Table 2). While both seagrass AGB and BGB are included in this value, seagrass BGB

390   only accounted for 3.5% of total seagrass biomass measured (Table 2). Analysis of seagrass

391   leaves collected from Tomales Bay demonstrated that seagrass material was composed of 31.6%

392   of OC. Thus, we estimate Walker salt marsh receives 33.4 ± 7.6 g OC m^{-2} in the form of seagrass

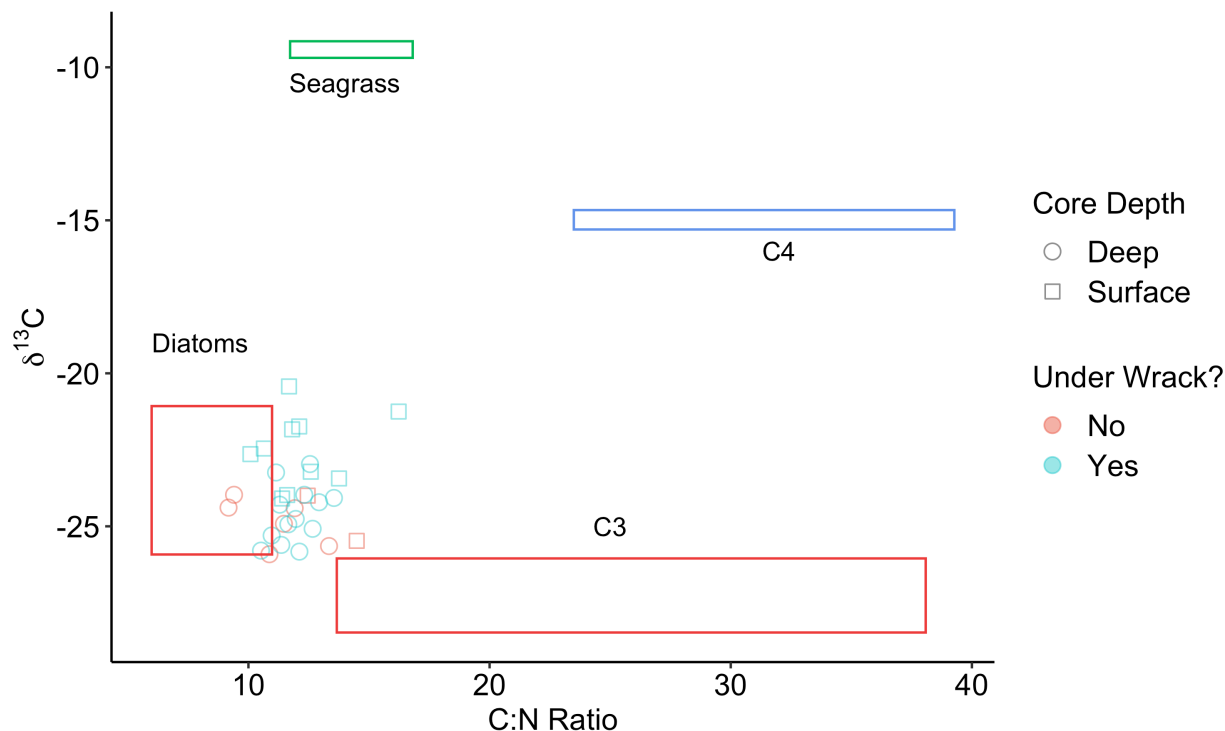
393   along wrack lines each year.

	Seagrass (winter) (g/m2)	Seagrass (summer) (g/m2)	Wrack (g/m2)	Wrack carbon (gC/m2)
AGB	58.6 ± 14	361.3 ± 47.8	102.8 ± 24	32.5 ± 7.6
BGB	57.1 ± 4.6	79 ± 15.5	3.7 ± 2.6	1.2 ± 0.8
Total	115.7 ± 16.5	440.3 ± 59.4	105.7 ± 24.6	33.4 ± 7.8

394

395 **Table 2:** Living seagrass above ground biomass (AGB) and below ground biomass (BGB) was
396 collected in Tom's Point seagrass meadow (Fig. 1b) in both summer and winter (g m^{-2}). Total
397 seagrass wrack delivered to Walker salt marsh was quantified as AGB and BGB (g m^{-2}) and
398 converted to carbon (g C m^{-2}) using a 31.6% carbon conversion rate.
399

3.2.2 Sediment Carbon Sources

In Walker salt marsh, surface (< 10 cm) wrack sediments had higher $\delta^{13}C$ values than sediments collected from the interior of the marsh (non-wrack sediments) and thus were more similar to the $\delta^{13}C$ values of seagrass (Fig. 6; SLM, $p < 0.05$; F-statistic =27.3; DF = 18). Shallow wrack sediments had an average $\delta^{13}C$ of -22.5 ± 0.38 ‰, while non-wrack sediments had an average $\delta^{13}C$ of -24.9 ± 0.26 ‰ (mean ± SE). However, when both shallow and deep (>10cm) sediments were included, wrack sediment $\delta^{13}C$ did not significantly differ from non-wrack sediments (SLM, $p > 0.05$, F-statistic = 0.34, DF = 31). C:N sediment ratios did not significantly differ from one another regardless of collection depth or location (SLMs, $p > 0.05$). When data were applied to a mixing model, apparent discrepancies in seagrass-derived carbon contributions between shallow and deep, and wrack and non-wrack sediments were insignificant (SIAR mixing model, Table S2). Similarly, no significant quantity of seagrass-derived carbon was detected in Elkhorn salt marsh sediments (SIAR mixing model, Table S2). Rather, models estimate that sediments were derived almost entirely from either diatoms or C3 plants – 83% in Elkhorn Slough and 88% in Walker salt marsh (Fig. 7; Table S2). Thus, regardless of site, depth, or the presence of wrack, model results estimate no significant storage of seagrass-derived carbon in sediment (Table S2). Instead, C3 plants and/or diatoms are the primary significant contributors to underlying carbon storage across salt marshes in all locations.

419
420
421 **Figure 6**: Points represent the δ^{13}C and C:N ratios of Walker Salt Marsh sediment data, collected
422 from underneath persistent wrack lines (blue) and from areas free of wrack (red) (see Fig. 1c).
423 Sediment subsamples collected from the surface to 10cm deep are labelled as surface samples
424 (square) and below 10cm are labelled as deep samples (circle). Colored boxes represent sources
425 (means ± SD) from diatoms (red), seagrass (green), C4 salt marsh plants (blue), and C3 salt
426 marsh plants (red).
427

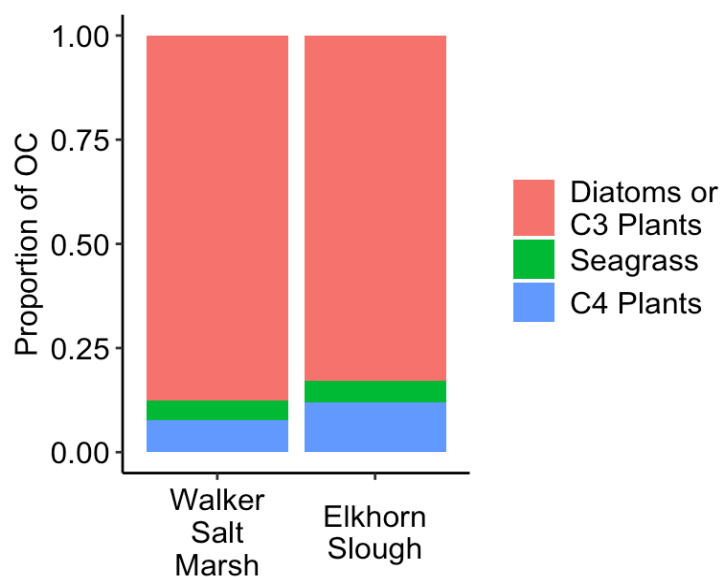



Figure 7 (left)**:** Mixing model results displaying the average source contributions to salt marsh sediment organic carbon (OC) for two salt marshes. Exact values and significance are shown in Table S2.

4   Discussion

4.1   Carbon Stocks

The lack of clear downcore trends (Fig. 2) could suggest relative OC stability with depth and through time or that there have been minimal changes to factors such as vegetation and grain size through time. The persistence of this pattern is supported by the longer cores sampled in seagrass and bare sediment sites (Fig. S3). One possible explanation for significant downcore OC losses in Newport Bay salt marsh is change in historic hydrography. Specifically, previous sediment profiles from this region also found finer surface sediments above coarser materials – associated with the conversion from an erosional to a depositional system in the early 20th century due to urbanization of the surrounding area (Trimble, 2003). This change in grain size down core may drive the associated OC loss down core, discussed further below. Although we did not quantify sediment carbon accumulation rates here, previous work in Tomales Bay seagrass meadows indicate rates of 11.37 - 15.16 g C m^{-2} yr^{-1} via ^{210}Pb dating (O'Donnell, 2017). By these accumulation rate estimates, we estimate the top 20 cm of sediment sampled here accumulated over approximately 100-130 years. Given the relatively slow accumulation rates and lack of down core trends, we interpret that our data represent realistic stock assessments for each respective habitat and location, rather than being reflective of shallow surface sediment carbon stocks.

Determining drivers of OC storage variation across habitats is notably complex, where a mix of factors such as grain size, elevation, hydrodynamic energy, and vegetation type may influence underlying sediment OC storage (Kelleway et al., 2016; Lima et al., 2020; Miyajima et al., 2015). Here, sediment grain size and its associated drivers allow us to further examine variation in sediment OC storage variation. Specifically, grain size likely contributed to the observed inter-

464   habitat differences in OC storage, given its strong correlation with TOM. Despite no significant

465   differences in grain size between habitat types within each site (likely due to high grain size

466   variability and low relative sample sizes), apparent qualitative differences elude to possible

467   explanations of stock differences. For example, seagrass sediments had lower % mud than salt

468   marsh sediments in Tomales Bay, a trend that was reversed in Newport Bay (Fig. 5a; Table S1).

469   Similarly, seagrass sediments did not always contain a higher % mud than neighboring bare

470   sediments, contrary to what is typically reported (Bos et al., 2007; Conley et al., 2017; Mazarassa

471   et al., 2015), which may explain their apparent lack of sediment OC differences. Overall, salt

472   marshes had greater OC storage and smaller average grain size than seagrass meadows. The

473   periodic, lower energy inundation of the sampled marsh and pan habitats may facilitate the

474   observed smaller grain sizes – by preventing the resuspension of small particles that might occur

475   in higher hydrodynamic energy or submerged systems (Christiansen et al., 2000; Yang et al.,

476   2008).

477   In marsh and pan habitats, biological factors such as primary production and decomposition

478   rates may become more important drivers of carbon burial than would be true in dominantly

479   sandy sediments (Miyajima et al., 2017). It is possible that root systems of overlying vegetation

480   add bulk organic material into available sediment space, contributing to increased carbon

481   deposition as well as decreased space for mineral accumulation (e.g., Rogers et al., 2019). Yet

482   despite the absence of vegetation in pan sites, we did not observe significant differences in

483   carbon stocks between pan sediments and surrounding salt marsh sediments. This could be

484   attributed to 1) significant carbon contributions from surrounding salt marsh biomass to pan sites

485   overtime, 2) historical marsh recovery along pan edges and subsequent burial of vegetation

486   captured in pan cores, 3) variable decomposition rates in both pan and salt marsh sediments, or

487   4) because the expected effect of canopy-vegetation on salt marsh sediment is too small relative

488   to other drivers, producing statistically insignificant results. In sum, although seagrasses and salt

489   marshes can facilitate carbon storage by altering grain size distributions, hydrodynamics and

490   geomorphology play a critical role in grain size distributions and therefore, carbon storage. The

491   hydrographic changes in Newport Bay salt marshes associated with decreased grain sizes further

492   emphasize this point – drawing attention to the importance of watershed and sediment

493   management to regional carbon storage. Furthermore, although grain size is clearly a key driver

494   in carbon storage, our data demonstrate that it becomes of limited use as a predictor of carbon

495   stocks after the proportion of mud exceeds 82% (Fig. 4). This demonstrates that using grain size

496   as a cost-effective way to estimate carbon stocks (as has been suggested, e.g. Serrano et al.,

497   2016) only appears possible in sandy and mixed grain size sediments within the sites studied

498   here. Carbon stocks in fine sediment sites above this threshold cannot be estimated using grain

499   size alone – a management-relevant finding for efforts to incorporate habitat-specific carbon

500   storage into regional climate plans.

501

502   **4.2 Global and Regional Comparisons**

503   With recent reviews on coastal carbon stocks in salt marshes and seagrass meadows, we can

504   compare our data to global and regional averages (no sufficient syntheses of pan or bare

505   sediment carbons stocks were identified). We find that carbon storage in California seagrass

506   sediments studied here is lower than global estimates yet is higher than regional estimates of Z.

507

Study	Habitat Type/Species	Location(s)	Value (Mg C_{org} ha^{-1})	Methods (core length and extrapolation depth)
Seagrass				

This study	Seagrass *(Z. marina)*	California	110 ± 11.8	20 cm cores extrapolated to 1 m
Kauffman et al. (2020)	Seagrass *(Z. marina)*	*U.S. west coast	80 ± 12	1 m cores collected (no extrapolation)
Prentice et al. (2020)	Seagrass *(Z. marina)*	U.S. west coast	65.12	25 cm cores extrapolated to 1 m
Röhr et al. (2018)	Seagrass *(Z. marina)*	*U.S. west coast *Temperate Northern Hemisphere	69.4 ± 8.4 108.9 ± 39.56	25 cm cores extrapolated to 1 m
Fourqurean et al. (2012)	Seagrass (many species)	Global	[1]139.7	A combined estimate from short cores (≥ 20cm & < 1 m) extrapolated to 1 m and full cores (1 m)
Salt Marsh				
This study	Salt marsh dominated by *Sarcocornia pacifica*, *Distichlis spicata* and *Jaumea carnosa*	California	235 ± 17.7	20 cm cores extrapolated to 1m
Kauffman et al. (2020)	Salt marsh dominated by *Distichlis spicata* and *Sarcocornia perennis*	*U.S. west coast	190 ± 16	1 m cores collected (no extrapolation)
Holmquist et al. (2018)	Tidal wetlands (mixed, dominated by estuarine emergent wetlands)	*Conterminous U.S. tidal wetlands	270 ± 1.4	A combined estimate from short cores (≥ 10cm & < 1 m) extrapolated to 1 m and full cores (1 m)

508
509 **Table 3:** Summary of previously reported carbon stocks in seagrass meadows and tidal wetlands,
510 all normalized to the top 1m of sediment. Studies that included any data from California are
511 denoted by an asterisk. Values are reported as mean ± SE, unless otherwise noted. [1]Fourqurean et
512 al. (2012) value represents median OC storage.
513

514 *marina* from the U.S. west coast (Table 3). Early estimates from the U.S. west coast were

515 markedly low, with one study of *Z. marina* from British Columbia measuring average stocks of

516   1.34 kg C m^{-3} with a maximum OC of 1.3%, compared to the 11.01 ± 1.18 kg C m^{-3} and a

517   maximum OC of 7.0 % estimated here (Postlethwaite et al., 2018). While estimates from studies

518   averaging over broader U.S. west coast regions are higher than this Canadian case study (Table

519   3), the data presented here are still higher than regional estimates, suggesting possible latitudinal

520   difference in carbon storage between the northern and southern regions of the North American

521   west coast. This is not surprising given the variation in environmental and geomorphic

522   conditions over this large stretch of coastline, which likely influence the carbon storage capacity

523   along with the factors discussed here such as hydrodynamics or grain size conditions.

524   Recent studies of the U.S. west coast and of conterminous U.S. tidal wetlands allow for

525   comparison with salt marsh data collected here. Much like with seagrass meadows, California

526   salt marsh carbon storage was also higher than previous estimates from the U.S. west coast

527   (Kauffman et al., 2020; Table 3). Both the data presented here as well as those from Kauffman et

528   al. (2020) (Table 3) were collected from similar marsh elevation zones and species compositions,

529   reducing some potential for these factors to result in the observed differences in carbon stocks.

530   Previous work has found that high-marsh zone sediments are likely to contain greater OC than

531   mid- and lower-marsh zones, and that less inundation can facilitate increased root productivity

532   and increased OC (the sites presented here would be considered 'lower-marsh zones') (Blum,

533   1993; Connor et al., 2001; Zhou et al., 2007). Moreover, sediments beneath other common marsh

534   species in this region (e.g. *Spartina sp.*) were not sampled, yet may have differing productivity

535   levels, resulting in OC stock differences. This may contribute to the variation between our

536   estimated salt marsh carbon stock and that of conterminous U.S. tidal wetlands (Holmquist et al.,

537   2018), which includes a variety of habitat types including forested and shrub-dominated tidal

538   wetlands.

539

4.3 Carbon Sources and Exchanges

541   Across both salt marshes assessed in this study, diatoms and/or C3 plants were the

542 dominant carbon sources in sediment. Although the isotopic values resulting from C4 and

543 seagrass sources are poorly resolved, lack of a significant values from either source allows for

544 interpretation. Specifically, seagrasses do not significantly contribute to any OC sources in

545 sediment, while C4 plants may contribute minimally to OC in sediment in Elkhorn Slough (1.1-

546 11.9%; Table S2). Thus, despite similar isotopic values between seagrass and C4 plants, if

547 seagrass-derived organic matter was buried under wrack sediments, we would expect to see

548 higher $\delta^{13}C$ values in wrack sediments with depth. The insignificant differences in $\delta^{13}C$ between

549 wrack and non-wrack sediments suggest that seagrass is not ultimately being buried, but rather,

550 carbon derived from C3 plants and diatoms are ultimately retained in sediment.

551   Several underlying causes may lead to this high proportion of diatom and/or C3-derived

552 carbon (Fig. 7). The recalcitrance of both autochthonous and allochthonous material in sediment

553 can vary depending on the material's composition (e.g. Burdige, 2007), and thus knowing the

554 sources and composition of deposited material aids in understanding its chance of long-term

555 burial. In seagrass meadows, previous work demonstrated that seagrass BGB likely contributes

556 most to autochthonous carbon burial when compared to AGB, due to higher proportions of

557 refractory compounds and decreased grazing pressure (Trevathan-Tackett et al., 2017). This may

558 serve to explain our lack of seagrass signal in salt marsh sediments under wrack. Seagrass wrack

559 material deposited on top of the salt marsh is predominantly AGB, likely breaking down or being

560 transported elsewhere before any appreciable portion reaches long-term sediment carbon pools.

561 Instead, the dominant C3 signal is likely driven by the presence of pickleweed species

562   (*Sarcocornia sp.)*, which have a considerable portion of below ground biomass. Despite the

563   presence of salt grass (*Distichilis spicata)*, the relative lack of C4-derived carbon in sediment

564   may be due to a few potential factors: 1) salt grass is less common relative to pickleweed within

565   our sites, 2) based on our field observations, root systems of salt grass do not seem to penetrate

566   as deeply or contain as much BGB as do those of pickleweed, and 3) salt grass contains less

567   woody tissue than pickleweed, making it less refractory (Jepson Flora Project, 2020). In the case

568   of diatom and planktonic sources, their significant presence in sediment may be due to a greater

569   abundance overall, or due to complex preservation pathways such as the facilitation of carbon

570   burial by binding sediments through extracellular polymeric substances (Drexler et al., 2020;

571   Macreadie et al., 2019; Oakes and Eyre, 2013).

572   From the seasonal senescence indicated by seagrass meadow biomass data (Table 2), we

573   infer that a significant amount of seagrass AGB is either degraded within the meadow or

574   exported for remineralization or deposition elsewhere. Given that strong tidal flows can occur

575   within meadows and that high densities of seagrass wrack were observed in a neighboring salt

576   marsh, lateral export of AGB from the meadow is likely. Nonetheless, the majority of OC

577   deposited into the marsh in the form of seagrass wrack is likely remineralized over the course of

578   the year – with wrack decomposition outpacing a sediment accumulation rate that might support

579   carbon preservation. While wrack remineralization can support local estuarine food web

580   metabolism, it can also produce emissions (Jiménez et al., 2017; Liu et al., 2019). However,

581   these emissions are small when compared to the levels of carbon sequestration within the marsh

582   – if all of the seagrass wrack along the wrack lines was remineralized annually, this degradation

583   would only contribute 33 ± 7.8 g C m^{-2}yr^{-1} to the atmosphere (Table 2). For comparison, dating

584   (primarily ^{137}Cs) from salt marshes nearby estimate carbon accumulation as 174 ± 45 g C m^{-2}yr^{-1}

585   (Ouyang & Lee, 2014). Thus, even along wrack lines, carbon accumulation far outpaces carbon

586   release from breakdown of allochthonous material. Although estimates of decomposition and

587   accumulation rates in this site could further inform these concepts, the slow annual cycle of

588   seagrass wrack deposition and disappearance observed in the aerial imagery (Fig. S2), and the

589   lack of seagrass-derived carbon in underlying sediment make a compelling case that little of this

590   material is ultimately buried.

591

592   **5   Conclusions**

593   We find that California salt marshes can store approximately twice as much carbon than do

594   seagrass meadows within this region. Grain size – an easier metric to quantify than carbon stock

595   – can be used to estimate regional carbon storage in sandy and mixed-grain size sediments. This

596   information can serve to inform local and regional management plans in efforts to prioritize and

597   quantify carbon storage across these habitat types. While seagrass meadows may act as local

598   sinks for carbon, they also export substantial amounts of AGB annually, which can be

599   remineralized and converted to CO_2, rather than buried in neighboring blue carbon habitats. C3

600   salt marsh plants and/or diatoms contributed to the OC pools in all three salt marshes under study

601   – likely due to their prevalence within our sites and their resistance to degradation. This

602   comprehensive study assesses several key research needs in blue carbon science informing

603   current efforts to prioritize and quantify carbon storage across these habitat types.

604

605   **6   Data Availability**

606   All data included in this manuscript will be made publicly available on the Dryad data

607   repository.

608

7   Competing Interests

All data included in this manuscript will be made publicly available on the Dryad data

repository.

612

8   Author Contributions

Ward – conceptualization, investigation, methodology, writing (original draft preparation),

formal analysis; Hill – conceptualization, funding support, writing (review & editing); Souza –

investigation (field and lab support), writing (review & editing); Filipczyk – investigation (field

and lab support), writing (review & editing); Ricart – methodology, investigation, writing

(review & editing); Merolla – investigation (field and lab support), writing (review & editing);

Capece – writing (review & editing); O'Donnell – investigation (field and lab support),

methodology; writing (review & editing);); Elsmore – investigation (field and lab support),

writing (review & editing); Oechel – writing (review & editing); Beheshti – investigation (field

and lab support), writing (review & editing).

623

9   Acknowledgements

We acknowledge Al Carranza, Grant Susner, Amanda Nordstrom, Siena Watson, Camille Frias,

Daphne Bradley, Anya Morrill, Natalie Rossi, and Jezebel Powers for laboratory and field

assistance. This work was supported by California Sea Grant (R/HCME-03, support to Tessa

Hill, Brian Gaylord, Eric Sanford, and Kristy Kroeker), the California Ocean Protection Council

(support to Tessa Hill, Brian Gaylord, Eric Sanford, and Kristy Kroeker), the CSU Council on

Ocean Affairs, Science & Technology (support to Melissa Ward), and the San Diego ARCS

631   foundation (support to Melissa Ward). Dr. Jim Cloern also generously provided data to support

632   isotopic mixing models. We also thank Audubon Canyon Ranch for site access from their

633   Cypress Grove Research Center. Lastly, we would like to thank Dr. Peter Bowler, whose

634   mentorship and legacy in wetland conservation has sparked lifelong collaborations and scientific

635   careers.

636
10 References
638
639   Alongi, D. M. Blue carbon coastal sequestration for climate change mitigation.
640   Springer International Publishing, Briefs in Climate Studies, doi: 10.1007/978-3-319-91698-9,
641   2018.

643   Attard, K. M., Rodil, I. F., Berg, P., Norkko, J., Norkko, A. and Glud, R. N.: Seasonal
644   metabolism and carbon export potential of a key coastal habitat: The perennial canopy-forming
645   macroalga *Fucus vesiculosus*, Limnology and Oceanography, 64(1), 149–164,
646   doi:10.1002/lno.11026, 2019.

647   Benner, R., Fogel, M. L. and Sprague, E. K.: Diagenesis of belowground biomass of *Spartina*
648   *alterniflora* in salt-marsh sediments, Limnology and Oceanography, 36(7), 1358–1374,
649   doi:10.4319/lo.1991.36.7.1358, 1991.

650   Blum, L. K.: Spartina alterniflora root dynamics in a Virginia marsh, Marine Ecology Progress
651   Series, 102, 169–178, 1993.

652   Bos, A. R., Bouma, T. J., de Kort, G. L. J. and van Katwijk, M. M.: Ecosystem engineering by
653   annual intertidal seagrass beds: Sediment accretion and modification, Estuarine, Coastal and
654   Shelf Science, 74(1–2), 344–348, doi:10.1016/j.ecss.2007.04.006, 2007.

655   Bouillon, S. and Connolly, R. M.: Carbon Exchange Among Tropical Coastal Ecosystems, in
656   Ecological Connectivity among Tropical Coastal Ecosystems, edited by I. Nagelkerken, pp. 45–
657   70, Springer Netherlands, Dordrecht., 2009.

658   Brevik, E. C. and Homburg, J. A.: A 5000 year record of carbon sequestration from a coastal
659   lagoon and wetland complex, Southern California, USA, CATENA, 57(3), 221–232,
660   doi:10.1016/j.catena.2003.12.001, 2004.

661   Burdige, D. J.: Preservation of Organic Matter in Marine Sediments: Controls, Mechanisms, and
662   an Imbalance in Sediment Organic Carbon Budgets?, Chem. Rev., 107(2), 467–485,
663   doi:10.1021/cr050347q, 2007.

Cabello-Pasini, A., Muñiz-Salazar, R. and Ward, D. H.: Annual variations of biomass and photosynthesis in *Zostera marina* at its southern end of distribution in the North Pacific, Aquatic Botany, 76(1), 31–47, doi:10.1016/S0304-3770(03)00012-3, 2003.

Callaway, J. C., Borgnis, E. L., Turner, R. E. and Milan, C. S.: Carbon Sequestration and Sediment Accretion in San Francisco Bay Tidal Wetlands, Estuaries and Coasts, 35(5), 1163–1181, doi:10.1007/s12237-012-9508-9, 2012.

Capece, L. The origin of sedimentary organic carbon in temperate seagrass meadows in California estuaries. Thesis 22619435, University of California, Davis, ProQuest Dissertations Publishing, 2019.

Chmura, G. L., Anisfeld, S. C., Cahoon, D. R. and Lynch, J. C.: Global carbon sequestration in tidal, saline wetland soils, Global Biogeochemical Cycles, 17(4), doi:10.1029/2002GB001917, 2003.

Christiansen, T., Wiberg, P. L. and Milligan, T. G.: Flow and sediment transport on a tidal salt marsh surface, Estuarine, Coastal and Shelf science, 50(3), 315–331, https://doi.org/10.1006/ecss.2000.0548, 2000.

Cloern, J. E., Canuel, E. A. and Harris, D.: Stable carbon and nitrogen isotope composition of aquatic and terrestrial plants of the San Francisco Bay estuarine system, Limnology and Oceanography, 47(3), 713–729, doi:10.4319/lo.2002.47.3.0713, 2002.

Conley, D. C., Austin, M., Davidson, I., Buscombe, D. and Masselink, G.: Grain size selection in seagrass beds, coastal dynamics, (200), 11, 2017.

Connor, R. F., Chmura, G. L. and Beecher, C. B.: Carbon accumulation in bay of fundy salt marshes: Implications for restoration of reclaimed marshes, Global Biogeochemical Cycles, 15(4), 943–954, doi:10.1029/2000GB001346, 2001.

Craft, C. B., Seneca, E. D. and Broome, S. W.: Loss on Ignition and Kjeldahl Digestion for Estimating Organic Carbon and Total Nitrogen in Estuarine Marsh Soils: Calibration with Dry Combustion, Estuaries, 14(2), 175, doi:10.2307/1351691, 1991.

Craven, K. F., Edwards, R. J. and Flood, R. P.: Source organic matter analysis of saltmarsh sediments using SIAR and its application in relative sea-level studies in regions of C_4 plant invasion, Boreas, 46(4), 642–654, doi:10.1111/bor.12245, 2017.

Cyronak, T., Andersson, A. J., D'Angelo, S., Bresnahan, P., Davidson, C., Griffin, A., Kindeberg, T., Pennise, J., Takeshita, Y. and White, M.: Short-Term Spatial and Temporal Carbonate Chemistry Variability in Two Contrasting Seagrass Meadows: Implications for pH Buffering Capacities, Estuaries and Coasts, 41(5), 1282–1296, doi:10.1007/s12237-017-0356-5, 2018.

Dahl, M., Deyanova, D., Gütschow, S., Asplund, M. E., Lyimo, L. D., Karamfilov, V., Santos, R., Björk, M. and Gullström, M.: Sediment properties as important predictors of carbon storage

in *Zostera marina* meadows: A Comparison of four European areas, PLoS ONE, 11(12), e0167493, https://doi.org/10.1371/journal.pone.0167493, 2016.

Dean, W. E.: Determination of carbonate and organic matter in calcareous sediments and sedimentary rocks by loss on ignition; comparison with other methods, Journal of Sedimentary Research, 44, 242–248, https://doi.org/10.1306/74D729D2-2B21-11D7-8648000102C1865D, 1974.

Drexler, J. Z., Davis, M. J., Woo, I. and De La Cruz, S.: Carbon sources in the sediments of a restoring vs. historically unaltered salt marsh, Estuaries and Coasts, 43(6), 1345–1360, https://doi.org/10.1007/s12237-020-00748-7, 2020.

Duarte, C., Marbà, N., Gacia, E., Fourqurean, J. W., Beggins, J., Barrón, C. and Apostolaki, E. T.: Seagrass community metabolism: Assessing the carbon sink capacity of seagrass meadows, Global Biogeochemical Cycles, 24(4), doi:10.1029/2010GB003793, 2010.

Duarte, C. M. and Cebrián, J.: The fate of marine autotrophic production, Limnology and Oceanography, 41(8), 1758–1766, doi:10.4319/lo.1996.41.8.1758, 1996.

Duarte, C. M. and Krause-Jensen, D.: Export from seagrass meadows contributes to marine carbon sequestration, Front. Mar. Sci., 4, doi:10.3389/fmars.2017.00013, 2017.

Escapa, M., Perillo, G. M. E. and Iribarne, O.: Biogeomorphically driven salt pan formation in Sarcocornia-dominated salt-marshes, Geomorphology, 228, 147–157, doi:10.1016/j.geomorph.2014.08.032, 2015.

Fourqurean, J. W., Duarte, C. M., Kennedy, H., Marbà, N., Holmer, M., Mateo, M. A., Apostolaki, E. T., Kendrick, G. A., Krause-Jensen, D., McGlathery, K. J. and Serrano, O.: Seagrass ecosystems as a globally significant carbon stock, Nature Geoscience, 5(7), 505–509, doi:10.1038/ngeo1477, 2012.

Fourqurean, J. W., Moore, T. O., Fry, B., and Hollibaugh, J. T.: Spatial and temporal variation in C:N:P ratios, δ15N, and δ13C of eelgrass *Zostera marina* as indicators of ecosystem processes, Tomales Bay, California, USA, 157, 147–157, https://doi.org/10.3354/meps157147, 1997.

Freedman, B., Stinson, G. and Lacoul, P.: Carbon credits and the conservation of natural areas, Environ. Rev., 17(NA), 1–19, doi:10.1139/A08-007, 2009.

Gambi, M., Nowell, A. and Jumars, P.: Flume observations on flow dynamics in *Zostera marina* (eelgrass) beds, Mar. Ecol. Prog. Ser., 61, 159–169, doi:10.3354/meps061159, 1990.

Google Earth. Tomales Bay, California, USA. 38°12'42.4"N 122°55'39.7"W. 2020.

Greiner, J. T., McGlathery, K. J., Gunnell, J. and McKee, B. A.: Seagrass restoration enhances "Blue Carbon" sequestration in coastal waters, PLoS ONE, 8(8), e72469, doi:10.1371/journal.pone.0072469, 2013.

Green, E.P. and Short, F.T. (Eds.) (2003). World Atlas of Seagrasses. *University of California*

738   *Press, Berkeley, USA*, 324 pp.

739

740   Hendriks, I., Sintes, T., Bouma, T. and Duarte, C.: Experimental assessment and modeling
741   evaluation of the effects of the seagrass *Posidonia oceanica* on flow and particle trapping,
742   Marine Ecology Progress Series, 356, 163–173, doi:10.3354/meps07316, 2008.

743   Holmquist, J. R., Windham-Myers, L., Bliss, N., Crooks, S., Morris, J. T., Megonigal, J. P.,
744   Troxler, T., Weller, D., Callaway, J., Drexler, J., Ferner, M. C., Gonneea, M. E., Kroeger, K. D.,
745   Schile-Beers, L., Woo, I., Buffington, K., Breithaupt, J., Boyd, B. M., Brown, L. N., Dix, N.,
746   Hice, L., Horton, B. P., MacDonald, G. M., Moyer, R. P., Reay, W., Shaw, T., Smith, E., Smoak,
747   J. M., Sommerfield, C., Thorne, K., Velinsky, D., Watson, E., Grimes, K. W. and Woodrey, M.:
748   accuracy and precision of tidal wetland soil carbon mapping in the conterminous United States,
749   Scientific Reports, 8(1), 9478, doi:10.1038/s41598-018-26948-7, 2018.

750   Howard, J., Hoyt, S., Isensee, K., Pidgeon, E., Telszewski, M. (eds.) (2014). Coastal Blue
751   Carbon: Methods for assessing carbon stocks and emissions factors in mangroves, tidal salt
752   marshes, and seagrass meadows. Conservation International, Intergovernmental Oceanographic
753   Commission of UNESCO, International Union for Conservation of Nature. Arlington, Virginia,
754   USA.

755   Hyndes, G. A., Nagelkerken, I., McLeod, R. J., Connolly, R. M., Lavery, P. S. and Vanderklift,
756   M. A.: Mechanisms and ecological role of carbon transfer within coastal seascapes, Biological
757   Reviews, 89(1), 232–254, doi:10.1111/brv.12055, 2014.

758   Ince, R., Hyndes, G. A., Lavery, P. S. and Vanderklift, M. A.: Marine macrophytes directly
759   enhance abundances of sandy beach fauna through provision of food and habitat, Estuarine,
760   Coastal and Shelf Science, 74(1–2), 77–86, doi:10.1016/j.ecss.2007.03.029, 2007.

761   Jepson Flora Project (eds.) 2020, Jepson eFlora, https://ucjeps.berkeley.edu/eflora/, accessed on
762   February 5th, 2020.

763   Jiménez, M. A., Beltran, R., Traveset, A., Calleja, M. L., Delgado-Huertas, A. and Marbà, N.:
764   Aeolian transport of seagrass (*Posidonia oceanica*) beach-cast to terrestrial systems, Estuarine,
765   Coastal and Shelf Science, 196, 31–44, doi:10.1016/j.ecss.2017.06.035, 2017.

766   Johannessen, S. C. and Macdonald, R. W.: Geoengineering with seagrasses: is credit due where
767   credit is given?, Environmental Research Letters, 11, 113001, https://doi.org/10.1088/1748-
768   9326/11/11/113001, 2016.

769   Kauffman, J. B., Giovanonni, L., Kelly, J., Dunstan, N., Borde, A., Diefenderfer, H., Cornu, C.,
770   Janousek, C., Apple, J. and Brophy, L.: Total ecosystem carbon stocks at the marine-terrestrial
771   interface: Blue carbon of the Pacific Northwest Coast, United States, Glob. Change Biol., 26(10),
772   5679–5692, doi:10.1111/gcb.15248, 2020.

773   Kelleway, J. J., Saintilan, N., Macreadie, P. I. and Ralph, P. J.: Sedimentary factors are key
774   predictors of carbon storage in SE australian saltmarshes, Ecosystems, 19(5), 865–880,
775   doi:10.1007/s10021-016-9972-3, 2016.

776   Kennedy, H., Beggins, J., Duarte, C. M., Fourqurean, J. W., Holmer, M., Marbà, N. and
777   Middelburg, J. J.: Seagrass sediments as a global carbon sink: Isotopic constraints, Global
778   Biogeochem. Cycles, 24(4), GB4026, doi:10.1029/2010GB003848, 2010.

779   Largier, J. L., Hollibaugh, J. T. and Smith, S. V.: Seasonally hypersaline estuaries in
780   mediterranean-climate regions, Estuarine, Coastal and Shelf Science, 45(6), 789–797,
781   doi:10.1006/ecss.1997.0279, 1997.

782   Lavery, P. S., Mateo, M.-Á., Serrano, O. and Rozaimi, M.: Variability in the carbon storage of
783   seagrass habitats and its implications for global estimates of Blue Carbon ecosystem service,
784   PLoS ONE, 8(9), e73748, doi:10.1371/journal.pone.0073748, 2013.

785   Leorri, E., Zimmerman, A. R., Mitra, S., Christian, R. R., Fatela, F. and Mallinson, D. J.:
786   Refractory organic matter in coastal salt marshes-effect on C sequestration calculations, Science
787   of the Total Environment, 633, 391–398, doi:10.1016/j.scitotenv.2018.03.120, 2018.

788   Lewis, C. J., Young, M. A., Ierodiaconou, D., Baldock, J. A., Hawke, B., Sanderman, J., Carnell,
789   P. E. and Macreadie, P. I.: Drivers and modelling of blue carbon stock variability in sediments of
790   southeastern Australia, Biogeosciences, 17(7), 2041–2059, doi:10.5194/bg-17-2041-2020, 2020.

791   Lima, M. do A. C., Ward, R. D. and Joyce, C. B.: Environmental drivers of sediment carbon
792   storage in temperate seagrass meadows, Hydrobiologia, doi:10.1007/s10750-019-04153-5, 2020.

793   Liu, S., Trevathan-Tackett, S. M., Ewers Lewis, C. J., Ollivier, Q. R., Jiang, Z., Huang, X. and
794   Macreadie, P. I.: Beach-cast seagrass wrack contributes substantially to global greenhouse gas
795   emissions, Journal of Environmental Management, 231, 329–335,
796   doi:10.1016/j.jenvman.2018.10.047, 2019.

797   Lovelock, C. E., Adame, M. F., Bennion, V., Hayes, M., O'Mara, J., Reef, R. and Santini, N. S.:
798   Contemporary rates of carbon sequestration through vertical accretion of sediments in mangrove
799   forests and saltmarshes of South East Queensland, Australia, Estuaries and Coasts, 37(3), 763–
800   771, doi:10.1007/s12237-013-9702-4, 2014.

801   Lovelock, C. E., Atwood, T., Baldock, J., Duarte, C. M., Hickey, S., Lavery, P. S., Masque, P.,
802   Macreadie, P. I., Ricart, A. M., Serrano, O. and Steven, A.: Assessing the risk of carbon dioxide
803   emissions from blue carbon ecosystems, Frontiers in Ecology and the Environment, 15(5), 257–
804   265, doi:10.1002/fee.1491, 2017.

805   Lovelock, C. E. and Duarte, C. M.: Dimensions of Blue Carbon and emerging perspectives,
806   Biology Letters, 15, 20180781, https://doi.org/10.1098/rsbl.2018.0781, 2019.

807   Macreadie, P. I., Ewers-Lewis, C. J., Whitt, A. A., Ollivier, Q., Trevathan-Tackett, S. M.,
808   Carnell, P., and Serrano, O.: Comment on `Geoengineering with seagrasses: is credit due where
809   credit is given?', Environ. Res. Lett., 13, 028002, https://doi.org/10.1088/1748-9326/aaa7ad,
810   2018.

811   Macreadie, P. I., Anton, A., Raven, J. A., Beaumont, N., Connolly, R. M., Friess, D. A.,
812   Kelleway, J. J., Kennedy, H., Kuwae, T., Lavery, P. S., Lovelock, C. E., Smale, D. A.,

813   Apostolaki, E. T., Atwood, T. B., Baldock, J., Bianchi, T. S., Chmura, G. L., Eyre, B. D.,
814   Fourqurean, J. W., Hall-Spencer, J. M., Huxham, M., Hendriks, I. E., Krause-Jensen, D.,
815   Laffoley, D., Luisetti, T., Marbà, N., Masque, P., McGlathery, K. J., Megonigal, J. P.,
816   Murdiyarso, D., Russell, B. D., Santos, R., Serrano, O., Silliman, B. R., Watanabe, K. and
817   Duarte, C. M.: The future of Blue Carbon science, Nature Communications, 10(1), 1–13,
818   doi:10.1038/s41467-019-11693-w, 2019.

819   Mazarrasa, I., Lovelock, C. E., Serrano, O., Lavery, P. S., Fourqurean, J. W., Kennedy, H.,
820   Mateo, M. A., Krause-Jensen, D., Steven, A. D. L. and Duarte, C. M.: Seagrass meadows as a
821   globally significant carbonate reservoir, Biogeosciences, 12(16), 4993–5003,
822   doi:https://doi.org/10.5194/bg-12-4993-2015, 2015.

823   Mazarrasa, I., Samper-Villarreal, J., Serrano, O., Lavery, P. S., Lovelock, C. E., Marbà, N.,
824   Duarte, C. M. and Cortés, J.: Habitat characteristics provide insights of carbon storage in
825   seagrass meadows, Marine Pollution Bulletin, 134, 106–117,
826   doi:10.1016/j.marpolbul.2018.01.059, 2018.

827   Mcleod, E., Chmura, G. L., Bouillon, S., Salm, R., Björk, M., Duarte, C. M., Lovelock, C. E.,
828   Schlesinger, W. H., and Silliman, B. R.: A blueprint for blue carbon: toward an improved
829   understanding of the role of vegetated coastal habitats in sequestering CO2, Frontiers in Ecology
830   and the Environment, 9, 552–560, https://doi.org/10.1890/110004, 2011.

831   Merkel & Associates, Inc. 2017 Tomales Bay Eelgrass Inventory, prepared the National Oceanic
832   Atmospheric Administration (NOAA) Greater Farallones National Marine Sanctuary. December
833   2017.
834
835   Milliman, J. D.: Precipitation and Cementation of Deep-Sea Carbonate Sediments, in Deep-Sea
836   Sediments: Physical and Mechanical Properties, edited by A. L. Inderbitzen, pp. 463–476,
837   Springer US, Boston, MA., 1974.

838   Miyajima, T., Hori, M., Hamaguchi, M., Shimabukuro, H., Adachi, H., Yamano, H. and
839   Nakaoka, M.: Geographic variability in organic carbon stock and accumulation rate in sediments
840   of East and Southeast Asian seagrass meadows, Global Biogeochemical Cycles, 29(4), 397–415,
841   doi:10.1002/2014GB004979, 2015.

842   Miyajima, T., Hori, M., Hamaguchi, M., Shimabukuro, H., and Yoshida, G.: Geophysical
843   constraints for organic carbon sequestration capacity of *Zostera marina* seagrass meadows and
844   surrounding habitats, 62, 954–972, https://doi.org/10.1002/lno.10478, 2017.

845   Nahlik, A. M. and Fennessy, M. S.: Carbon storage in US wetlands, Nat Commun, 7(1), 13835,
846   doi:10.1038/ncomms13835, 2016.

847   O'Donnell, B. C. Carbon sequestration within Northeastern Pacific seagrass meadows, Thesis
848   10607483, University of California, Davis, ProQuest Dissertations Publishing, 2017.
849

850 Oakes, J. M. and Eyre, B. D.: Transformation and fate of microphytobenthos carbon in
851 subtropical, intertidal sediments: potential for long-term carbon retention revealed by 13C-
852 labeling, Biogeosciences, 11(7), 1927–1940, https://doi.org/10.5194/bg-11-1927-2014, 2014.

853 Ouyang, X. and Lee, S. Y.: Updated estimates of carbon accumulation rates in coastal marsh
854 sediments, Biogeosciences, 11(18), 5057–5071, doi:10.5194/bg-11-5057-2014, 2014.

855 Parnell, A. C. and Jackson, A.L. SIAR: stable isotope analysis in R. R package version 4.2. http:
856 //CRAN.R-project.org/package=siar, 2013.
857
858 Patrick, Wm. H. and DeLaune, R. D.: Subsidence accretion and sea level rise in south San
859 Francisco Bay marshes, Limnology and Oceanography, 35(6), 1389–1395,
860 doi:10.4319/lo.1990.35.6.1389, 1990.

861 Peck, E. K., Wheatcroft, R. A. and Brophy, L. S.: Controls on Sediment Accretion and Blue
862 Carbon Burial in Tidal Saline Wetlands: Insights From the Oregon Coast, USA, J. Geophys. Res.
863 Biogeosci., 125(2), doi:10.1029/2019JG005464, 2020.

864 Pendleton, L., Donato, D. C., Murray, B. C., Crooks, S., Jenkins, W. A., Sifleet, S., Craft, C.,
865 Fourqurean, J. W., Kauffman, J. B., Marbà, N., Megonigal, P., Pidgeon, E., Herr, D., Gordon, D.
866 and Baldera, A.: Estimating global "Blue Carbon" emissions from conversion and degradation of
867 vegetated coastal ecosystems, edited by S. Thrush, PLoS ONE, 7(9), e43542,
868 doi:10.1371/journal.pone.0043542, 2012.

869 Perdue, E. M. and Koprivnjak, J. F.: Using the C/N ratio to estimate terrigenous inputs of organic
870 matter to aquatic environments, Estuarine, Coastal and Shelf Science, 73(1–2), 65–72,
871 doi:10.1016/j.ecss.2006.12.021, 2007.

872 Poppe, K. L. and Rybczyk, J. M.: Carbon Sequestration in a Pacific Northwest Eelgrass (*Zostera*
873 *marina*) Meadow, BioOne, 92(2), 80–91, doi:10.3955/046.092.0202, 2018.

874 Postlethwaite, V. R., McGowan, A. E., Kohfeld, K. E., Robinson, C. L. K. and Pellatt, M. G.:
875 Low blue carbon storage in eelgrass (*Zostera marina*) meadows on the Pacific Coast of Canada,
876 PLOS ONE, 13(6), e0198348, doi:10.1371/journal.pone.0198348, 2018.

877 Prentice, C., Hessing-Lewis, M., Sanders-Smith, R. and Salomon, A. K.: Reduced water motion
878 enhances organic carbon stocks in temperate eelgrass meadows, Limnology and Oceanography,
879 64(6), 2389–2404, doi:10.1002/lno.11191, 2019.

880 Prentice, C., Poppe, K. L., Lutz, M., Murray, E., Stephens, T. A., Spooner, A., Hessing-Lewis,
881 M., Sanders-Smith, R., Rybczyk, J. M., Apple, J., Short, F. T., Gaeckle, J., Helms, A., Mattson,
882 C., Raymond, W. W., and Klinger, T.: A Synthesis of Blue Carbon Stocks, Sources, and
883 Accumulation Rates in Eelgrass (*Zostera marina*) Meadows in the Northeast Pacific, Global
884 Biogeochem. Cycles, 34, https://doi.org/10.1029/2019GB006345, 2020.
885
886 R Core Team (2018). R: A language and environment for statistical computing. R Foundation for
887 Statistical Computing, Vienna, Austria. URL https://www.R-project.org/.
888

889   Ricart, A., Dalmau, A., Pérez, M. and Romero, J.: Effects of landscape configuration on the
890   exchange of materials in seagrass ecosystems, Mar. Ecol. Prog. Ser., 532, 89–100,
891   doi:10.3354/meps11384, 2015.

892   Ricart, A. M., Pérez, M. and Romero, J.: Landscape configuration modulates carbon storage in
893   seagrass sediments, Estuarine, Coastal and Shelf Science, 185, 69–76,
894   doi:10.1016/j.ecss.2016.12.011, 2017.

895   Ricart, A. M., York, P. H., Bryant, C. V., Rasheed, M. A., Ierodiaconou, D. and Macreadie, P. I.:
896   High variability of Blue Carbon storage in seagrass meadows at the estuary scale, Scientific
897   Reports, 10(1), 5865, doi:10.1038/s41598-020-62639-y, 2020.

898   Rogers, K., Kelleway, J. J., Saintilan, N., Megonigal, J. P., Adams, J. B., Holmquist, J. R., Lu,
899   M., Schile-Beers, L., Zawadzki, A., Mazumder, D., and Woodroffe, C. D.: Wetland carbon
900   storage controlled by millennial-scale variation in relative sea-level rise, 567, 91–95,
901   https://doi.org/10.1038/s41586-019-0951-7, 2019.

902   Röhr, M. E., Holmer, M., Baum, J. K., Björk, M., Boyer, K., Chin, D., Chalifour, L., Cimon, S.,
903   Cusson, M., Dahl, M., Deyanova, D., Duffy, J. E., Eklöf, J. S., Geyer, J. K., Griffin, J. N.,
904   Gullström, M., Hereu, C. M., Hori, M., Hovel, K. A., Hughes, A. R., Jorgensen, P.,
905   Kiriakopolos, S., Moksnes, P.-O., Nakaoka, M., O'Connor, M. I., Peterson, B., Reiss, K.,
906   Reynolds, P. L., Rossi, F., Ruesink, J., Santos, R., Stachowicz, J. J., Tomas, F., Lee, K.-S.,
907   Unsworth, R. K. F., and Boström, C.: Blue carbon storage capacity of temperate Eelgrass
908   (*Zostera marina*) meadows, Global Biogeochemical Cycles, 32, 1457–1475,
909   https://doi.org/10.1029/2018GB005941, 2018.

910   Saintilan, N., Rogers, K., Mazumder, D. and Woodroffe, C. Allochthonous and autochthonous
911   contributions to carbon accumulation and carbon store in southeastern Australian coastal
912   wetlands, Estuarine, Coastal and Shelf Science, 128, 84–92, doi:10.1016/j.ecss.2013.05.010,
913   2013.

914   Schlosser, S., and A. Eicher. The Humboldt Bay and Eel River Estuary Benthic Habitat Project.
915   California Sea Grant Publication T-075. 246 p, 2012.

916   Serrano, O., Mateo, M. A., Renom, P., and Julià, R.: Characterization of soils beneath a
917   Posidonia oceanica meadow, Geoderma, 185–186, 26–36,
918   https://doi.org/10.1016/j.geoderma.2012.03.020, 2012.

919   Serrano, O., Lavery, P. S., Duarte, C. M., Kendrick, G. A., Calafat, A., York, P. H., Steven, A.
920   and Macreadie, P. I.: Can mud (silt and clay) concentration be used to predict soil organic carbon
921   content within seagrass ecosystems?, Biogeosciences, 13(17), 4915–4926, doi:10.5194/bg-13-
922   4915-2016, 2016.

923   Serrano, O., Kelleway, J. J., Lovelock, C. and Lavery, P. S.: Conservation of Blue Carbon
924   Ecosystems for Climate Change Mitigation and Adaptation, Coastal Wetlands, pp. 965–996,
925   Elsevier., 2019.

926 St. Laurent, K. A., Hribar, D. J., Carlson, A. J., Crawford, C. M., and Siok, D.: Assessing coastal
927 carbon variability in two Delaware tidal marshes, Journal of Coast Conservation, 24, 65,
928 https://doi.org/10.1007/s11852-020-00783-3, 2020.

929 Trevathan-Tackett, S. M., Macreadie, P. I., Sanderman, J., Baldock, J., Howes, J. M. and Ralph,
930 P. J.: A Global Assessment of the Chemical Recalcitrance of Seagrass Tissues: Implications for
931 Long-Term Carbon Sequestration, Front. Plant Sci., 8, 925, doi:10.3389/fpls.2017.00925, 2017.

932 Trimble, S. W.: Historical hydrographic and hydrologic changes in the San Diego creek
933 watershed, Newport Bay, California, Journal of Historical Geography, 29(3), 422–444,
934 doi:10.1006/jhge.2002.0485, 2003.

935 Valiela, I. and Cole, M. L.: Comparative evidence that salt marshes and mangroves may protect
936 seagrass meadows from land-derived nitrogen loads, Ecosystems, 5(1), 92–102,
937 doi:10.1007/s10021-001-0058-4, 2002.

938 Van Dyke, E. and Wasson, K.: Historical ecology of a central California estuary: 150 years of
939 habitat change, Estuaries, 28(2), 173–189, doi:10.1007/BF02732853, 2005.

940 Wilkinson, G. M., Besterman, A., Buelo, C., Gephart, J. and Pace, M. L.: A synthesis of modern
941 organic carbon accumulation rates in coastal and aquatic inland ecosystems, Scientific Reports,
942 8(1), 15736, doi:10.1038/s41598-018-34126-y, 2018.

943 Yang, S. L., Li, H., Ysebaert, T., Bouma, T. J., Zhang, W. X., Wang, Y. Y., Li, P., Li, M. and
944 Ding, P. X.: Spatial and temporal variations in sediment grain size in tidal wetlands, Yangtze
945 Delta: On the role of physical and biotic controls, Estuarine, Coastal and Shelf Science, 77(4),
946 657–671, https://doi.org/10.1016/j.ecss.2007.10.024, 2008.

947 Zhou, J., Wu, Y., Kang, Q. and Zhang, J.: Spatial variations of carbon, nitrogen, phosphorous
948 and sulfur in the salt marsh sediments of the Yangtze Estuary in China, Estuarine, Coastal and
949 Shelf Science, 71(1–2), 47–59, doi:10.1016/j.ecss.2006.08.012, 2007