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Carbon and nutrient burial within Peruvian coastal marsh driven by anthropogenic activities

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Abstract

This research assessed carbon and nutrient burial during the past ~ 60 years within a Peruvian coastal marsh ecosystem affected by anthropogenic activities, by examining total organic carbon (TOC), total nitrogen (TN) and isotopes ($\delta^{13}\text{C}$ and $\delta^{15}\text{N}$) tracers in two dated sediment cores. Significantly higher TOC and TN burial, up to 416.4 ± 65.0 and $0.7 \pm 0.1 \text{ g m}^{-2} \text{ year}^{-1}$ respectively, were observed after an uncontrolled urban expansion starting in the early 1970's to the 1990's. The TOC and TN burial rates were up to twofold higher than those observed for preserved coastal marshes. Furthermore, the decreased $\delta^{13}\text{C}$ values ($-16.1 \pm 0.6 \text{ ‰}$) and increasing $\delta^{15}\text{N}$ values ($+10.6 \pm 2.6 \text{ ‰}$) indicate higher deposition of algal material and urban sewage during the same period. The higher burial rates during 1970's - 1990's and reduced rates thereafter evidenced the role of coastal marsh ecosystems plays in sequestering carbon and nutrients.

Keywords:

Peruvian coastal marsh, carbon burial, organic matter source, urban expansion, eutrophication.

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4 32 There is a growing global interest in coastal wetland zones since they are recognized as highly
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6 33 productive ecosystems, as well as being able to support a vast biological diversity and provide
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8 34 important ecosystem services (Kathiresan and Alikunhi, 2011; Ouyang and Lee, 2014; Alongi,
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10 35 2014). One of the most important ecosystem services of coastal wetlands is the sequestration of
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12 36 terrestrial, marine, and atmospheric carbon as biomass and within their sedimentary environments
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14 37 (Donato et al., 2011; Breithaupt et al., 2012; Pérez et al., 2017a). Despite their high ecosystem
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16 38 value, coastal wetlands are threatened by anthropogenic pressures (e.g. urban expansion, industrial
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18 39 activities and land use) which may result in habitat degradation, eutrophication, increase in CO₂
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20 40 emissions and general loss of ecosystem services (Pendleton et al. 2012; Luisetti et al. 2014; Pérez
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22 41 et al., 2018a). Among coastal wetland zones, coastal marshes have a high carbon burial capacity,
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24 42 with a global average comparable with vegetated coastal forests such as mangrove ecosystems
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26 43 (Luisetti et al. 2013; Alongi et al., 2014; Wang et al., 2021). Also, due to their sedimentary
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28 44 characteristics and carbon burial capacity, salt marsh may mitigate local environmental
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30 45 contamination (e.g. sewage effluents, and eutrophication.) (Hughes et al., 2009; Livesley and
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32 46 Andrusiak, 2012; Lovelock et al., 2014). Anthropogenic impacts on coastal marsh may result in
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34 47 changes in primary production, organic matter sources and sediment accretion rates, and may
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36 48 modify significantly the carbon burial rates (Kristensen et al. 2008; Alongi 2014; Pérez et al.
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38 49 2018a).

34 50 The coastal marsh ecosystem “Refugio de Vida Silvestre Pantanos de Villa” (PV) is a “wetland
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36 51 of international importance, as “waterfowl habitat”. This location is considered as one of the most
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38 52 important wetland zones within the central Peruvian coast, being a biodiversity refuge along the
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40 53 South American Pacific desert and providing a variety of ecosystem services (e.g. supporting,
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42 54 provisioning, regulating and cultural services) (Lizarzaburu, 1992; INRENA, 1998; RAMSAR,
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44 55 2019). During the colonial period this coastal marsh area was used for agricultural and livestock
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46 56 farming. In the early 1960’s the area was designated for touristic recreation. In early 1970’s, PV has
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48 57 been affected by several anthropogenic activities (e.g. uncontrolled urban expansion, increased
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50 58 sewage loading, and oxidation ponds discharge) resulting in higher inputs of nutrients and organic
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52 59 matter into the sedimentary environments (FOVIDA, 2006; Aponte et al., 2018; Pulido and
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54 60 Bermudez, 2018). In 1989 this coastal area was designated as a reserved area (Aponte et al., 2018).
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56 61 Studies of the PV ecosystem have evaluated the carbon content in marsh vegetation (Aldave and
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58 62 Aponte 2019), the carbon burial capacity of aquatic plants (Aponte, 2017), carbon stock losses
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60 63 within areas affected by fires (e.g. ~210.40 Mg ha⁻¹) (Ramírez et al. 2022) as well as carbon stocks
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62 64 in natural wetland reserves (e.g. 2,21 x 10⁶ MgC for Los Pantanos de Villa wetland reserve)
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64 65 (Aponte et al. 2021). However, the sediment burial rates and the recent geochemical evolution of

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4 66 this coastal marsh ecosystem were previously unknown. Understanding carbon dynamics within
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6 67 eutrophic systems in coastal zones is a worldwide concern (Alongi 2014; Pérez et al., 2018a; Pérez
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8 68 et al. 2020a). This study aims to evaluate anthropogenic impacts on carbon and nitrogen burial
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10 69 within sedimentary environments of the PV coastal marsh ecosystem.
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13 71 The PV coastal marsh ecosystem (Fig. 1) is located within the densely urbanized Lima region
14 72 on the central coast of Peru, where approximately 10 million people live (Aponte and Ramirez,
15 73 2011; Aponte et al., 2018). This ecosystem presents enclosed geomorphology as it is not connected
16 74 directly to the ocean, and characterized by a 1-5% slope topography and contains approximately 276
17 75 hectares of marsh vegetation dominated by *Distichlis spicata*, *Salicornia fruticosa* and
18 76 *Schoenoplectus americanus* (Ramírez and Cano, 2010). The waterbodies located within this
19 77 ecosystem ranged between 1 and 2 meters in depth and exhibited water temperature of 25.5 ± 1.2
20 78 $^{\circ}\text{C}$, pH values of 8.3 ± 0.3 and dissolved oxygen values of $13.4 \pm 4.1 \text{mg L}^{-1}$ (Huaman-Vilca et al.,
21 79 2020). Also, during the past few decades, this coastal marsh zone has been affected by different
22 80 types of anthropogenic disturbances (e.g., uncontrolled urban expansion, land use, extractive
23 81 activities and sewage releases) (Lizarzaburu, 1992; Aponte and Ramirez, 2011; Aponte et al.,
24 82 2018).
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33 83 In 2019, two 30 cm-depth sediment cores were collected 70 meters apart in PV (Fig. 1).
34 84 Sediment compaction during the collection was minimized by using 10cm inner diameter PVC
35 85 tubes. The coring sites were selected avoiding plant roots to minimize the sediment mixture (Pérez
36 86 et al., 2020a). After collection, each sediment column was sectioned on site in 1-cm intervals from
37 87 the top to the base. Subsamples were stored at 4°C before being air dried at 40°C . Triplicate
38 88 samples of *Distichlis spicata*, *Salicornia fruticosa*, *Schoenoplectus americanus* were collected in
39 89 the study area. In addition, three samples of phytoplankton-derived organic matter were collected
40 90 from near water bodies using a phytoplankton net (Ingebrigtsen et al., 2017).
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46 91 The dry bulk sediment density was determined by dividing the dry sediment weight over the
47 92 initial sediment volume for each subsample (Ravichandran et al., 1995; Pérez et al., 2020). The
48 93 determination of ^{210}Pb activity in sediments was performed by direct measurement of 46.5 KeV
49 94 gamma-ray peaks, whereas ^{226}Ra activity was calculated averaging its daughters' peaks ^{214}Pb and
50 95 ^{214}Bi (295.2 KeV) (351.9 KeV) (609.3 KeV); finally, the ^{210}Pb -ex activity was estimated by
51 96 subtracting the ^{226}Ra from the total ^{210}Pb activity (Moore, 1984; Sanders et al., 2010; Pérez et al.,
52 97 2020a). These measurements were made in a semi-planar intrinsic germanium high purity coaxial
53 98 detector coupled to a multichannel analyzer (Sanders et al., 2010; Pérez et al. 2020a). The sediment
54 99 accumulation rates (SAR) were calculated using the constant initial concentration (CIC) method
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100 (Appleby and Oldfield, 1992). The sediment age was calculated using the equation: Sediment age =
101 [Year of sampling] – [Average depth of sediment interval / SAR] (Pérez et al., 2020b; Conrad et al.,
102 2019). Sediment sub-samples were decarbonated for each interval to measure total organic carbon
103 content (TOC) and carbon isotopic ratio ($\delta^{13}\text{C}$), while non-decarbonated sub-samples were used to
104 measure total nitrogen content (TN) and nitrogen isotopic ratio ($\delta^{15}\text{N}$) (Kennedy et al. 2005;
105 Schlacher and Connolly 2014; Peng et al. 2018). These analyses were performed using an elemental
106 analyzer coupled to an isotope ratio mass spectrometer Thermo Finnigan Model Delta Plus XP
107 (Naidu et al., 2000; Sanders et al. 2014; Pérez et al. 2020b). The TOC and TN burial rates were
108 estimated following the equations: Burial rates ($\text{g m}^{-2} \text{ year}^{-1}$) = [SAR; cm yr^{-1}] \times [DBD; g cm^{-3}] \times
109 [TOC or TN; g g^{-1}] (Sanders et al., 2014; Pérez et al., 2020b). According to the American Society
110 for Testing Materials (1963), the textural composition of sediments was determined using sieves.
111 Sedimentary phases were compared using arithmetic means for each geochemical variable.
112 Statistical differences were determined by the post hoc Tukey's Honestly Significant Difference test
113 (ANOVA + Tukey HSD) ($\alpha = 0.05$) with the previous testing of data normality by a Shapiro–Wilk
114 analysis.

115 The decline of the ^{210}Pb -ex activity as a function of depth for cores V1 and V2 reflected a
116 consistent SAR (ESM-Fig. 1), whereas the textural composition of both cores presented less than
117 50% of sand content along with the sediment profile (ESM-Fig. 2), suggesting that significant
118 disturbance or mixing did not occur along with the sediment profile (Nittrouer et al., 1984; Appleby
119 and Oldfield, 1992). The similarity between ^{210}Pb -ex activity profiles in the replicate cores support
120 the reliability of our SAR estimates (ESM-Fig. 1). The SAR values obtained in this study are 0.51
121 and 0.58 cm yr^{-1} for V1 and V2 (ESM-Fig. 1), resulting in an average SAR of $0.55 \pm 0.04 \text{ cm yr}^{-1}$ for
122 this study area, which was used to calculate TOC and TN burial rates in both sediment cores. These
123 results are in agreement with the global mean determined for coastal marsh ecosystems (0.6 cm yr^{-1} ;
124 including both preserved and impacted ecosystems (Chmura et al., 2003; Ouyang and Lee,
125 2014). Studies have described that coastal marsh ecosystems affected by anthropogenic activities
126 (e.g. urban expansion, land use and industrial purposes) may present SAR values up to 0.90 cm yr^{-1} ,
127 whereas preserved marshes values may reach 0.4 cm yr^{-1} (Cahoon, 1994; Callaway et al., 1997;
128 Dittman et al., 2016). These higher SAR values within impacted systems may result from the input
129 derived from anthropogenic wastes (e.g. sediments, organic matter and nutrients) (Anisfeld et al.,
130 1999; Atwood et al., 2017; Pérez et al., 2018a). Higher SAR values may also be observed near the
131 water source (e.g. ocean, creeks, or rivers), which may increase the probability of sediment and
132 particulate material input into the sedimentary environments (Oenema and Delaune, 1988; Chmura

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133 and Hung, 2004; Zhou et al., 2007). Under this context, our results for the PV coastal marsh area
134 (0.55 cm yr⁻¹) are higher than the range of previous studies in preserved systems (0.3 cm yr⁻¹)
135 (Chmura et al., 2003) and local sea level rise (0.2 cm yr⁻¹) (Takahashi et al., 2014). These higher
136 rates could be related to the sewage releases derived from the nearby urban area, and to the enclosed
137 geomorphology which protects the system from ocean erosion (Aponte et al., 2018; Pulido and
138 Bermudez, 2018).

139 Variations of TOC, TN, δ¹³C and δ¹⁵N are strongly correlated and remarkably similar in both
140 cores, confirming the reliability of the records (Fig. 2). From these well-defined records, three
141 sedimentary phases, P1 to P3, were determined based on these records (P1: 1960's -1970's; P2:
142 1970's -1990's and P3:1990's -2010's). The TOC and TN contents were significantly higher in P2
143 in comparison with P3, whereas the lowest values were observed during P1 (Fig. 2A), also trends in
144 terms of burial rates were similar (Fig. 3). These observations suggest that a higher organic matter
145 burial started during the 1970's as a result of intensified fertilization of primary production driven by
146 the beginning of an uncontrolled urban expansion near to this coastal marsh area (Aponte et al.,
147 2018; Pulido and Bermudez, 2018). The transition from higher carbon and nutrient burial rates
148 during P2 to lower rates during P3 (Fig. 3A-B) agrees with this area's environmental history. During
149 the early 1970's a large and uncontrolled urban expansion occurred near to the PV coastal marsh
150 area, resulting in the input of higher nutrients and organic matter contents. Then, in the early
151 1990's, the urban expansion was controlled and its impacts were mitigated by the construction of a
152 sewage system which resulted in a likely decrease of these loads into the coastal marsh
153 environments (FOVIDA, 2006; Aponte et al., 2018). The TOC burial rates reached 400 g m⁻² yr⁻¹
154 during P2 (Fig. 3A), which is about twofold higher than the global average described for preserved
155 coastal marsh ecosystems (240 g m⁻² yr⁻¹) (Chmura et al., 2003; Ouyang and Lee, 2014; Wang et
156 al., 2021), and threefold higher than the burial rates registered for preserved mangrove forests
157 (Alongi, 2014; Pérez et al., 2018a). However, this is consistent with higher values observed in
158 coastal marsh ecosystems affected by natural (e.g. storms and flooding) and anthropogenic factors
159 (e.g. urban expansion, industrial activities and land use), which may present carbon burial rates
160 between 500 and 1000 g m⁻² yr⁻¹ (Andrews et al., 2008; Palomo and Niell, 2009; Day et al., 2011).
161 It is also comparable to burial rates measured in mangroves impacted by anthropogenic factors (e.g.
162 sewage releases, aquaculture, and urban expansion) (Pérez et al., 2018a; Pérez et al 2020a; Pérez et
163 al., 2020b). The TN burial rates during P2 were comparable to that observed in preserved mangrove
164 forest and sixfold lower than that observed within impacted mangroves (Pérez et al 2020a; Pérez et
165 al., 2020b). This comparison suggests that coastal marshes may be able to burial carbon within

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166 sedimentary environments at comparable rates to mangrove ecosystems although the nutrient
167 filtering capability may be lower than that observed within mangrove forests due to difference in
168 these vegetation types (Alongi, 2014).

169 During P1, the $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values (Fig. 2B) are close to those obtained from fresh marsh
170 vegetation (ESM-Table 1). These values suggest that terrestrial organic matter (e.g. local marsh
171 vegetation) was the dominant source of organic matter and nutrients during this period (Peterson et
172 al., 1985; Meyers 2003; Hewitt et al., 2020). The significantly lower TOC and TN percentage
173 contents during this sedimentary phase in comparison with the other phases (Table 1) may reflect
174 that lower productivity due to high organic matter remineralization rates of which could have been
175 occurring within the sedimentary environments (Alongi et al., 2005; Pérez et al., 2020a). During P2,
176 $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ exhibit higher values than P1 (Fig. 2B), comparable with the values derived from
177 phytoplankton material (ESM-Table 1). This result evidenced a higher contribution of algal material
178 deposition within the sedimentary environments (e.g. phytoplankton, cyanobacteria-derived organic
179 matter) (Meyers 2003; Chappuis et al., 2017; Hewitt et al., 2020) and likely reflects the beginning
180 of isotopically heavy nitrogen influx derived from anthropogenic activities during early 1970's
181 (Aponte et al., 2018). The high nutrient influx likely triggered the dominance of phytoplankton and
182 cyanobacteria material during this period, derived from the sewage release due to the urban
183 expansion near to the study area (Diebel et al., 2012; Paredes et al., 2019; Pérez et al., 2020a).
184 Finally, during P3, the $\delta^{13}\text{C}$ values are higher in comparison with P2 (Fig. 2B), and again consistent
185 with the $\delta^{13}\text{C}$ values obtained from fresh marsh vegetation (ESM-Table 1), which suggests the
186 deposition of higher proportions of terrestrial or saltmarsh organic matter (Peterson et al., 1985;
187 Hewitt et al., 2020). These results may reflect the expansion or greater contribution of saltmarsh
188 vegetation (Sanders et al., 2014; Pérez et al., 2020a). The $\delta^{15}\text{N}$ values reflect the deposition of
189 significantly lighter nitrogen in sediments (Table 1), suggesting the decrease in the influx of heavy
190 anthropogenic nitrogen during P3 (Dawson et al., 2002; Kendall et al., 2008; Paredes et al., 2019).
191 The lighter $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values were associated during this phase to lower percentage contents and
192 burial rates of TOC and TN in comparison with P2 (ESM-Fig. 3), which may reflect the decrease in
193 production and deposition of algal organic matter (e.g. phytoplankton and cyanobacteria) (Chappuis
194 et al., 2017; Hewitt et al., 2020; Pérez et al., 2020a). These changes occurred when public policies
195 were established to regulate urbanization during the early 1990's. Sewage systems and sanitary
196 controls were established, mitigating the discharge of nutrients and organic material into the
197 sedimentary environments of PV coastal marsh area (INRENA, 1998; FOVIDA, 2006; Aponte et
198 al., 2018).

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199 Aquatic or terrestrial environments able to sequester terrestrial, marine and atmospheric carbon,
200 accumulating it as biomass or within soils, sediments and the water column, are sometimes referred
201 to as "blue carbon ecosystems" (Donato et al., 2011; Breithaupt et al., 2012; Alongi, 2014). Due to
202 the current global environmental context, many non-governmental organizations and governments
203 are funding activities focused on conservation, expansion and restoration of blue carbon ecosystems
204 to enhance the carbon burial capacity of these areas to mitigate climate change (Alongi, 2014;
205 Dittman et al., 2016; Hiederer and Viñas, 2018). The coastal marsh ecosystems cover an area of
206 approximately 6 million hectares at a global scale presenting a carbon burial rate of 240 g m⁻²
207 yr⁻¹, including impacted and preserved systems (Chmura et al., 2003; Ouyang and Lee, 2014;
208 Dittmann et al., 2016). It is important to consider that within these coastal marsh ecosystems, more
209 than 70% of the carbon is retained in form of sedimentary organic matter (Batjes, 1996; Murray et
210 al., 2011; Alongi, 2014). We extrapolate the average carbon burial rates obtained in the sediment
211 cores to the complete PV coastal marsh surface area of 276 hectares to estimate the total carbon
212 burial and CO₂ equivalents of the ecosystem (Aponte et al., 2018; Pulido and Bermudez, 2018).
213 Thus, using the average carbon burial rate of 275 g m² yr⁻¹ obtained in this study and, 276 hectares
214 from PV coastal marsh ecosystems sequester approximately 759 tons of organic carbon per year in
215 sediments which may represent about 70% of the total amount (Batjes, 1996; Murray et al., 2011;
216 Alongi, 2014). Then, converting the calculated organic carbon to CO₂ equivalents by using the
217 converting factor of 3.67 (Dittmann et al., 2016; Hiederer and Viñas, 2018) results in an estimation
218 of 2790 tons of CO₂ buried per year within PV coastal marsh. If a value of USD 30 per ton of CO₂
219 is used for carbon credits calculation (Garnaut, 2008; Dittmann et al., 2016; Hiederer and Viñas,
220 2018), the carbon burial of the whole PV coastal marsh area represents approximately USD 83600
221 per year in carbon credits or USD 300 per hectare. Our study demonstrates the value of the PV
222 coastal marsh systems to reduce net anthropogenic carbon emissions.

223

224 In conclusion, two sediment cores analyzed from the PV coastal marsh wetland located within
225 the densely urbanized area of Lima - Peru show how the evolution of the urbanization conditions
226 since the 1960's has impacted the sedimentary environment and carbon burial within this important
227 ecosystem. During 1960's, the lower TOC and TN contents reflected conditions prior to
228 anthropogenic activities, while the isotopic values evidenced the deposition of terrestrial organic
229 matter. Starting in the early 1970's, the urban expansion and the associated sewage discharge
230 caused the eutrophication of the coastal wetland, strong algal productivity, and an increases in TOC
231 and TN burial rates. In the early 1990's, public policies on urbanization, sewage, and sanitary
232 system were introduced, inducing a decrease in algal production in PV, accompanied by lower TOC

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233 and TN contents and burial rates. We show that the PV coastal marsh presents a high capacity for
234 carbon and nutrient burial, and should therefore be considered as an important blue carbon sink.
235 This study provides the first evaluation of carbon burial in Peruvian coastal marshes in relation to
236 their environmental condition and site specific history. Further studies in preserved and impacted
237 coastal marshes are necessary for a better understanding of the dynamics of carbon and nutrient
238 burial capacity within these blue carbon systems.

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250 **Compliance with Ethical Standards**

251 **Author’s contributions:** All authors designed the study. AP wrote the manuscript with input from all other
252 authors. All authors approve the final version of the manuscript and agree to be accountable for all aspects of
253 the manuscript.

254 **Disclosure of potential conflicts of interest:** Non-applicable

255 **Research involving Human Participants and/or Animals:** Non-applicable

256 **Informed consent:** Non-applicable

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Carbon and nutrient burial within Peruvian coastal marsh driven by anthropogenic activities

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Abstract

This research assessed carbon and nutrient burial during the past ~ 60 years within a Peruvian coastal marsh ecosystem affected by anthropogenic activities, by examining total organic carbon (TOC), total nitrogen (TN) and isotopes ($\delta^{13}\text{C}$ and $\delta^{15}\text{N}$) tracers in two dated sediment cores. Significantly higher TOC and TN burial, up to 416.4 ± 65.0 and $0.7 \pm 0.1 \text{ g m}^{-2} \text{ year}^{-1}$ respectively, were observed after an uncontrolled urban expansion starting in the early 1970's to the 1990's. The TOC and TN burial rates were up to twofold higher than those observed for preserved coastal marshes. Furthermore, the decreased $\delta^{13}\text{C}$ values ($-16.1 \pm 0.6 \text{ ‰}$) and increasing $\delta^{15}\text{N}$ values ($+10.6 \pm 2.6 \text{ ‰}$) indicate higher deposition of algal material and urban sewage during the same period. The higher burial rates during 1970's - 1990's and reduced rates thereafter evidenced the role of coastal marsh ecosystems **plays** in sequestering carbon and nutrients.

Keywords:

Peruvian coastal marsh, carbon burial, organic matter source, urban expansion, eutrophication.

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4 32 There is a growing global interest in coastal wetland zones since they are recognized as highly
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6 33 productive ecosystems, as well as being able to support a vast biological diversity and provide
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8 34 important ecosystem services (Kathiresan and Alikunhi, 2011; Ouyang and Lee, 2014; Alongi,
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10 35 2014). One of the most important ecosystem services of coastal wetlands is the sequestration of
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12 36 terrestrial, marine, and atmospheric carbon as biomass and within their sedimentary environments
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14 37 (Donato et al., 2011; Breithaupt et al., 2012; Pérez et al., 2017a). Despite their high ecosystem
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16 38 value, coastal wetlands are threatened by anthropogenic pressures (e.g. urban expansion, industrial
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18 39 activities and land use) which may result in habitat degradation, eutrophication, increase in CO₂
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20 40 emissions and general loss of ecosystem services (Pendleton et al. 2012; Luisetti et al. 2014; Pérez
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22 41 et al., 2018a). Among coastal wetland zones, coastal marshes have a high carbon burial capacity,
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24 42 with a global average comparable with vegetated coastal forests such as mangrove ecosystems
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26 43 (Luisetti et al. 2013; Alongi et al., 2014; Wang et al., 2021). Also, due to their sedimentary
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28 44 characteristics and carbon burial capacity, salt marsh may mitigate local environmental
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30 45 contamination (e.g. sewage effluents, and eutrophication.) (Hughes et al., 2009; Livesley and
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32 46 Andrusiak, 2012; Lovelock et al., 2014). Anthropogenic impacts on coastal marsh may result in
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34 47 changes in primary production, organic matter sources and sediment accretion rates, and may
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36 48 modify significantly the carbon burial rates (Kristensen et al. 2008; Alongi 2014; Pérez et al.
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38 49 2018a).

34 50 The coastal marsh ecosystem “Refugio de Vida Silvestre Pantanos de Villa” (PV) is a “wetland
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36 51 of international importance, as “waterfowl habitat”. This location is considered as one of the most
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38 52 important wetland zones within the central Peruvian coast, being a biodiversity refuge along the
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40 53 South American Pacific desert and providing a variety of ecosystem services (e.g. supporting,
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42 54 provisioning, regulating and cultural services) (Lizarzaburu, 1992; INRENA, 1998; RAMSAR,
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44 55 2019). During the colonial period this coastal marsh area was used for agricultural and livestock
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46 56 farming. In the early 1960’s the area was designated for touristic recreation. In early 1970’s, PV has
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48 57 been affected by several anthropogenic activities (e.g. uncontrolled urban expansion, increased
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50 58 sewage loading, and oxidation ponds discharge) resulting in higher inputs of nutrients and organic
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52 59 matter into the sedimentary environments (FOVIDA, 2006; Aponte et al., 2018; Pulido and
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54 60 Bermudez, 2018). In 1989 this coastal area was designated as a reserved area (Aponte et al., 2018).
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56 61 Studies of the PV ecosystem have evaluated the carbon content in marsh vegetation (Aldave and
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58 62 Aponte 2019), the carbon burial capacity of aquatic plants (Aponte, 2017), carbon stock losses
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60 63 within areas affected by fires (e.g. ~210.40 Mg ha⁻¹) (Ramírez et al. 2022) as well as carbon stocks
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62 64 in natural wetland reserves (e.g. 2,21 x 10⁶ MgC for Los Pantanos de Villa wetland reserve)
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64 65 (Aponte et al. 2021). However, the sediment burial rates and the recent geochemical evolution of

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4 66 this coastal marsh ecosystem were previously unknown. Understanding carbon dynamics within
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6 67 eutrophic systems in coastal zones is a worldwide concern (Alongi 2014; Pérez et al., 2018a; Pérez
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8 68 et al. 2020a). This study aims to evaluate anthropogenic impacts on carbon and nitrogen burial
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10 69 within sedimentary environments of the PV coastal marsh ecosystem.
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13 71 The PV coastal marsh ecosystem (Fig. 1) is located within the densely urbanized Lima region
14 72 on the central coast of Peru, where approximately 10 million people live (Aponte and Ramirez,
15 73 2011; Aponte et al., 2018). This ecosystem presents enclosed geomorphology as it is not connected
16 74 directly to the ocean, and characterized by a 1-5% slope topography and contains approximately 276
17 75 hectares of marsh vegetation dominated by *Distichlis spicata*, *Salicornia fruticosa* and
18 76 *Schoenoplectus americanus* (Ramírez and Cano, 2010). The waterbodies located within this
19 77 ecosystem ranged between 1 and 2 meters in depth and exhibited water temperature of 25.5 ± 1.2
20 78 $^{\circ}\text{C}$, pH values of 8.3 ± 0.3 and dissolved oxygen values of $13.4 \pm 4.1 \text{mg L}^{-1}$ (Huaman-Vilca et al.,
21 79 2020). Also, during the past few decades, this coastal marsh zone has been affected by different
22 80 types of anthropogenic disturbances (e.g., uncontrolled urban expansion, land use, extractive
23 81 activities and sewage releases) (Lizarzaburu, 1992; Aponte and Ramirez, 2011; Aponte et al.,
24 82 2018).
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32 83 In 2019, two 30 cm-depth sediment cores were collected 70 meters apart in PV (Fig. 1).
33 84 Sediment compaction during the collection was minimized by using 10cm inner diameter PVC
34 85 tubes. The coring sites were selected avoiding plant roots to minimize the sediment mixture (Pérez
35 86 et al., 2020a). After collection, each sediment column was sectioned on site in 1-cm intervals from
36 87 the top to the base. Subsamples were stored at 4°C before being air dried at 40°C . Triplicate
37 88 samples of *Distichlis spicata*, *Salicornia fruticosa*, *Schoenoplectus americanus* were collected in
38 89 the study area. In addition, three samples of phytoplankton-derived organic matter were collected
39 90 from near water bodies using a phytoplankton net (Ingebrigtsen et al., 2017).
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46 91 The dry bulk sediment density was determined by dividing the dry sediment weight over the
47 92 initial sediment volume for each subsample (Ravichandran et al., 1995; Pérez et al., 2020). The
48 93 determination of ^{210}Pb activity in sediments was performed by direct measurement of 46.5 KeV
49 94 gamma-ray peaks, whereas ^{226}Ra activity was calculated averaging its daughters' peaks ^{214}Pb and
50 95 ^{214}Bi (295.2 KeV) (351.9 KeV) (609.3 KeV); finally, the ^{210}Pb -ex activity was estimated by
51 96 subtracting the ^{226}Ra from the total ^{210}Pb activity (Moore, 1984; Sanders et al., 2010; Pérez et al.,
52 97 2020a). These measurements were made in a semi-planar intrinsic germanium high purity coaxial
53 98 detector coupled to a multichannel analyzer (Sanders et al., 2010; Pérez et al. 2020a). The sediment
54 99 accumulation rates (SAR) were calculated using the constant initial concentration (CIC) method
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100 (Appleby and Oldfield, 1992). The sediment age was calculated using the equation: Sediment age =
101 [Year of sampling] – [Average depth of sediment interval / SAR] (Pérez et al., 2020b; Conrad et al.,
102 2019). Sediment sub-samples were decarbonated for each interval to measure total organic carbon
103 content (TOC) and carbon isotopic ratio ($\delta^{13}\text{C}$), while non-decarbonated sub-samples were used to
104 measure total nitrogen content (TN) and nitrogen isotopic ratio ($\delta^{15}\text{N}$) (Kennedy et al. 2005;
105 Schlacher and Connolly 2014; Peng et al. 2018). These analyses were performed using an elemental
106 analyzer coupled to an isotope ratio mass spectrometer Thermo Finnigan Model Delta Plus XP
107 (Naidu et al., 2000; Sanders et al. 2014; Pérez et al. 2020b). The TOC and TN burial rates were
108 estimated following the equations: Burial rates ($\text{g m}^{-2} \text{ year}^{-1}$) = [SAR; cm yr^{-1}] \times [DBD; g cm^{-3}] \times
109 [TOC or TN; g g^{-1}] (Sanders et al., 2014; Pérez et al., 2020b). According to the American Society
110 for Testing Materials (1963), the textural composition of sediments was determined using sieves.
111 Sedimentary phases were compared using arithmetic means for each geochemical variable.
112 Statistical differences were determined by the post hoc Tukey's Honestly Significant Difference test
113 (ANOVA + Tukey HSD) ($\alpha = 0.05$) with the previous testing of data normality by a Shapiro–Wilk
114 analysis.

115 The decline of the ^{210}Pb -ex activity as a function of depth for cores V1 and V2 reflected a
116 consistent SAR (ESM-Fig. 1), whereas the textural composition of both cores presented less than
117 50% of sand content along with the sediment profile (ESM-Fig. 2), suggesting that significant
118 disturbance or mixing did not occur along with the sediment profile (Nittrouer et al., 1984; Appleby
119 and Oldfield, 1992). The similarity between ^{210}Pb -ex activity profiles in the replicate cores support
120 the reliability of our SAR estimates (ESM-Fig. 1). The SAR values obtained in this study are 0.51
121 and 0.58 cm yr^{-1} for V1 and V2 (ESM-Fig. 1), resulting in an average SAR of $0.55 \pm 0.04 \text{ cm yr}^{-1}$ for
122 this study area, which was used to calculate TOC and TN burial rates in both sediment cores. These
123 results are in agreement with the global mean determined for coastal marsh ecosystems (0.6 cm yr^{-1} ,
124 including both preserved and impacted ecosystems (Chmura et al., 2003; Ouyang and Lee,
125 2014). Studies have described that coastal marsh ecosystems affected by anthropogenic activities
126 (e.g. urban expansion, land use and industrial purposes) may present SAR values up to 0.90 cm yr^{-1} ,
127 whereas preserved marshes values may reach 0.4 cm yr^{-1} (Cahoon, 1994; Callaway et al., 1997;
128 Dittman et al., 2016). These higher SAR values within impacted systems may result from the input
129 derived from anthropogenic wastes (e.g. sediments, organic matter and nutrients) (Anisfeld et al.,
130 1999; Atwood et al., 2017; Pérez et al., 2018a). Higher SAR values may also be observed near the
131 water source (e.g. ocean, creeks, or rivers), which may increase the probability of sediment and
132 particulate material input into the sedimentary environments (Oenema and Delaune, 1988; Chmura

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133 and Hung, 2004; Zhou et al., 2007). Under this context, our results for the PV coastal marsh area
134 (0.55 cm yr⁻¹) are higher than the range of previous studies in preserved systems (0.3 cm yr⁻¹)
135 (Chmura et al., 2003) and local sea level rise (0.2 cm yr⁻¹) (Takahashi et al., 2014). These higher
136 rates could be related to the sewage releases derived from the nearby urban area, and to the enclosed
137 geomorphology which protects the system from ocean erosion (Aponte et al., 2018; Pulido and
138 Bermudez, 2018).

139 Variations of TOC, TN, δ¹³C and δ¹⁵N are strongly correlated and remarkably similar in both
140 cores, confirming the reliability of the records (Fig. 2). From these well-defined records, three
141 sedimentary phases, P1 to P3, were determined based on these records (P1: 1960's -1970's; P2:
142 1970's -1990's and P3:1990's -2010's). The TOC and TN contents were significantly higher in P2
143 in comparison with P3, whereas the lowest values were observed during P1 (Fig. 2A), also trends in
144 terms of burial rates were similar (Fig. 3). These observations suggest that a higher organic matter
145 burial started during the 1970's as a result of intensified fertilization of primary production driven by
146 the beginning of an uncontrolled urban expansion near to this coastal marsh area (Aponte et al.,
147 2018; Pulido and Bermudez, 2018). The transition from higher carbon and nutrient burial rates
148 during P2 to lower rates during P3 (Fig. 3A-B) agrees with this area's environmental history. During
149 the early 1970's a large and uncontrolled urban expansion occurred near to the PV coastal marsh
150 area, resulting in the input of higher nutrients and organic matter contents. Then, in the early
151 1990's, the urban expansion was controlled and its impacts were mitigated by the construction of a
152 sewage system which resulted in a likely decrease of these loads into the coastal marsh
153 environments (FOVIDA, 2006; Aponte et al., 2018). The TOC burial rates reached 400 g m⁻² yr⁻¹
154 during P2 (Fig. 3A), which is about twofold higher than the global average described for preserved
155 coastal marsh ecosystems (240 g m⁻² yr⁻¹) (Chmura et al., 2003; Ouyang and Lee, 2014; Wang et
156 al., 2021), and threefold higher than the burial rates registered for preserved mangrove forests
157 (Alongi, 2014; Pérez et al., 2018a). However, this is consistent with higher values observed in
158 coastal marsh ecosystems affected by natural (e.g. storms and flooding) and anthropogenic factors
159 (e.g. urban expansion, industrial activities and land use), which may present carbon burial rates
160 between 500 and 1000 g m⁻² yr⁻¹ (Andrews et al., 2008; Palomo and Niell, 2009; Day et al., 2011).
161 It is also comparable to burial rates measured in mangroves impacted by anthropogenic factors (e.g.
162 sewage releases, aquaculture, and urban expansion) (Pérez et al., 2018a; Pérez et al 2020a; Pérez et
163 al., 2020b). The TN burial rates during P2 were comparable to that observed in preserved mangrove
164 forest and sixfold lower than that observed within impacted mangroves (Pérez et al 2020a; Pérez et
165 al., 2020b). This comparison suggests that coastal marshes may be able to burial carbon within

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166 sedimentary environments at comparable rates to mangrove ecosystems although the nutrient
167 filtering capability may be lower than that observed within mangrove forests due to difference in
168 these vegetation types (Alongi, 2014).

169 During P1, the $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values (Fig. 2B) are close to those obtained from fresh marsh
170 vegetation (ESM-Table 1). These values suggest that terrestrial organic matter (e.g. local marsh
171 vegetation) was the dominant source of organic matter and nutrients during this period (Peterson et
172 al., 1985; Meyers 2003; Hewitt et al., 2020). The significantly lower TOC and TN percentage
173 contents during this sedimentary phase in comparison with the other phases (Table 1) may reflect
174 that lower productivity due to high organic matter remineralization rates of which could have been
175 occurring within the sedimentary environments (Alongi et al., 2005; Pérez et al., 2020a). During P2,
176 $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ exhibit higher values than P1 (Fig. 2B), comparable with the values derived from
177 phytoplankton material (ESM-Table 1). This result evidenced a higher contribution of algal material
178 deposition within the sedimentary environments (e.g. phytoplankton, cyanobacteria-derived organic
179 matter) (Meyers 2003; Chappuis et al., 2017; Hewitt et al., 2020) and likely reflects the beginning
180 of isotopically heavy nitrogen influx derived from anthropogenic activities during early 1970's
181 (Aponte et al., 2018). The high nutrient influx likely triggered the dominance of phytoplankton and
182 cyanobacteria material during this period, derived from the sewage release due to the urban
183 expansion near to the study area (Diebel et al., 2012; Paredes et al., 2019; Pérez et al., 2020a).
184 Finally, during P3, the $\delta^{13}\text{C}$ values are higher in comparison with P2 (Fig. 2B), and again consistent
185 with the $\delta^{13}\text{C}$ values obtained from fresh marsh vegetation (ESM-Table 1), which suggests the
186 deposition of higher proportions of terrestrial or saltmarsh organic matter (Peterson et al., 1985;
187 Hewitt et al., 2020). These results may reflect the expansion or greater contribution of saltmarsh
188 vegetation (Sanders et al., 2014; Pérez et al., 2020a). The $\delta^{15}\text{N}$ values reflect the deposition of
189 significantly lighter nitrogen in sediments (Table 1), suggesting the decrease in the influx of heavy
190 anthropogenic nitrogen during P3 (Dawson et al., 2002; Kendall et al., 2008; Paredes et al., 2019).
191 The lighter $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values were associated during this phase to lower percentage contents and
192 burial rates of TOC and TN in comparison with P2 (ESM-Fig. 3), which may reflect the decrease in
193 production and deposition of algal organic matter (e.g. phytoplankton and cyanobacteria) (Chappuis
194 et al., 2017; Hewitt et al., 2020; Pérez et al., 2020a). These changes occurred when public policies
195 were established to regulate urbanization during the early 1990's. Sewage systems and sanitary
196 controls were established, mitigating the discharge of nutrients and organic material into the
197 sedimentary environments of PV coastal marsh area (INRENA, 1998; FOVIDA, 2006; Aponte et
198 al., 2018).

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199 Aquatic or terrestrial environments able to sequester terrestrial, marine and atmospheric carbon,
200 accumulating it as biomass or within soils, sediments and the water column, are sometimes referred
201 to as "blue carbon ecosystems" (Donato et al., 2011; Breithaupt et al., 2012; Alongi, 2014). Due to
202 the current global environmental context, many non-governmental organizations and governments
203 are funding activities focused on conservation, expansion and restoration of blue carbon ecosystems
204 to enhance the carbon burial capacity of these areas to mitigate climate change (Alongi, 2014;
205 Dittman et al., 2016; Hiederer and Viñas, 2018). The coastal marsh ecosystems cover an area of
206 approximately 6 million hectares at a global scale presenting a carbon burial rate of 240 g m⁻²
207 yr⁻¹, including impacted and preserved systems (Chmura et al., 2003; Ouyang and Lee, 2014;
208 Dittmann et al., 2016). It is important to consider that within these coastal marsh ecosystems, more
209 than 70% of the carbon is retained in form of sedimentary organic matter (Batjes, 1996; Murray et
210 al., 2011; Alongi, 2014). We extrapolate the average carbon burial rates obtained in the sediment
211 cores to the complete PV coastal marsh surface area of 276 hectares to estimate the total carbon
212 burial and CO₂ equivalents of the ecosystem (Aponte et al., 2018; Pulido and Bermudez, 2018).
213 Thus, using the average carbon burial rate of 275 g m² yr⁻¹ obtained in this study and, 276 hectares
214 from PV coastal marsh ecosystems sequester approximately 759 tons of organic carbon per year in
215 sediments which may represent about 70% of the total amount (Batjes, 1996; Murray et al., 2011;
216 Alongi, 2014). Then, converting the calculated organic carbon to CO₂ equivalents by using the
217 converting factor of 3.67 (Dittmann et al., 2016; Hiederer and Viñas, 2018) results in an estimation
218 of 2790 tons of CO₂ buried per year within PV coastal marsh. If a value of USD 30 per ton of CO₂
219 is used for carbon credits calculation (Garnaut, 2008; Dittmann et al., 2016; Hiederer and Viñas,
220 2018), the carbon burial of the whole PV coastal marsh area represents approximately USD 83600
221 per year in carbon credits or USD 300 per hectare. Our study demonstrates the value of the PV
222 coastal marsh systems to reduce net anthropogenic carbon emissions.

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224 In conclusion, two sediment cores analyzed from the PV coastal marsh wetland located within
225 the densely urbanized area of Lima - Peru show how the evolution of the urbanization conditions
226 since the 1960's has impacted the sedimentary environment and carbon burial within this important
227 ecosystem. During 1960's, the lower TOC and TN contents reflected conditions prior to
228 anthropogenic activities, while the isotopic values evidenced the deposition of terrestrial organic
229 matter. Starting in the early 1970's, the urban expansion and the associated sewage discharge
230 caused the eutrophication of the coastal wetland, strong algal productivity, and an increases in TOC
231 and TN burial rates. In the early 1990's, public policies on urbanization, sewage, and sanitary
232 system were introduced, inducing a decrease in algal production in PV, accompanied by lower TOC

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233 and TN contents and burial rates. We show that the PV coastal marsh presents a high capacity for
234 carbon and nutrient burial, and should therefore be considered as an important blue carbon sink.
235 This study provides the first evaluation of carbon burial in Peruvian coastal marshes in relation to
236 their environmental condition and site specific history. Further studies in preserved and impacted
237 coastal marshes are necessary for a better understanding of the dynamics of carbon and nutrient
238 burial capacity within these blue carbon systems.

239
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250 **Compliance with Ethical Standards**

251 **Author’s contributions:** All authors designed the study. AP wrote the manuscript with input from all other
252 authors. All authors approve the final version of the manuscript and agree to be accountable for all aspects of
253 the manuscript.

254 **Disclosure of potential conflicts of interest:** Non-applicable

255 **Research involving Human Participants and/or Animals:** Non-applicable

256 **Informed consent:** Non-applicable

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Table 1. Post-hoc Tukey's Honestly Significant Difference test (ANOVA + Tukey HSD) results for the temporal variations of the geochemical proxies between sedimentary phases within the “RVSPV” saltmarsh ecosystem. Three sedimentary phases were delineated by considering the main temporal changes **in the carbon nutrient burial trends for both sediment cores**: Phase 1 (P1), during 1960's – 1970's ($n = 10$); Phase 2 (P2), during 1970's – 1990's ($n = 20$); and Phase 3 (P3), during 1990's – 2010's ($n = 26$). “X” symbol represents arithmetic mean for the whole sediment column. * $p < 0.05$. *Underlined acronyms present non-statistical differences among them.*

Variables	n	Average \pm SE	Interactions
TOC (%)	10	P1: 2.1 ± 1.1	* P1 < P3 < P2
	20	P2: 8.3 ± 1.3	
	26	P3: 5.2 ± 0.5	
	56	X: 5.8 ± 2.2	
TN (%)	10	P1: 0.4 ± 0.1	* P1 < P3 < P2
	20	P2: 0.8 ± 0.1	
	26	P3: 0.5 ± 0.1	
	56	X: 0.5 ± 0.2	
TOC ($\text{g m}^{-2} \text{ yr}^{-1}$)	10	P1: 117.6 ± 63.2	* <u>P1 P3</u> < P2
	20	P2: 406.5 ± 61.0	
	26	P3: 225.4 ± 22.3	
	56	X: 263.2 ± 137.7	
TN ($\text{g m}^{-2} \text{ yr}^{-1}$)	10	P1: 1.7 ± 0.3	* <u>P1 P3</u> < P2
	20	P2: 3.8 ± 0.6	
	26	P3: 2.1 ± 0.4	
	56	X: 2.6 ± 1.0	
TOC (g m^{-2})	10	P1: 237.3 ± 133.2	* P1 < P3 < P2
	20	P2: 852.5 ± 130.3	
	26	P3: 464.4 ± 107.0	
	56	X: 562.1 ± 260.8	
TN (g m^{-2})	10	P1: 34.3 ± 11.2	* <u>P1 P3</u> < P2
	20	P2: 74.2 ± 12.4	
	26	P3: 42.19 ± 8.4	
	56	X: 52.2 ± 19.8	
$\delta^{13}\text{C}$ (‰)	10	P1: -21.3 ± 0.4	* <u>P3 P1</u> < P2
	20	P2: -16.1 ± 0.6	
	26	P3: -21.1 ± 1.1	
	56	X: -19.6 ± 2.0	
$\delta^{15}\text{N}$ (‰)	10	P1: $+5.4 \pm 2.3$	* <u>P3 P1</u> < P2
	20	P2: $+10.6 \pm 2.6$	
	26	P3: $+3.8 \pm 0.6$	
	56	X: $+6.4 \pm 3.2$	

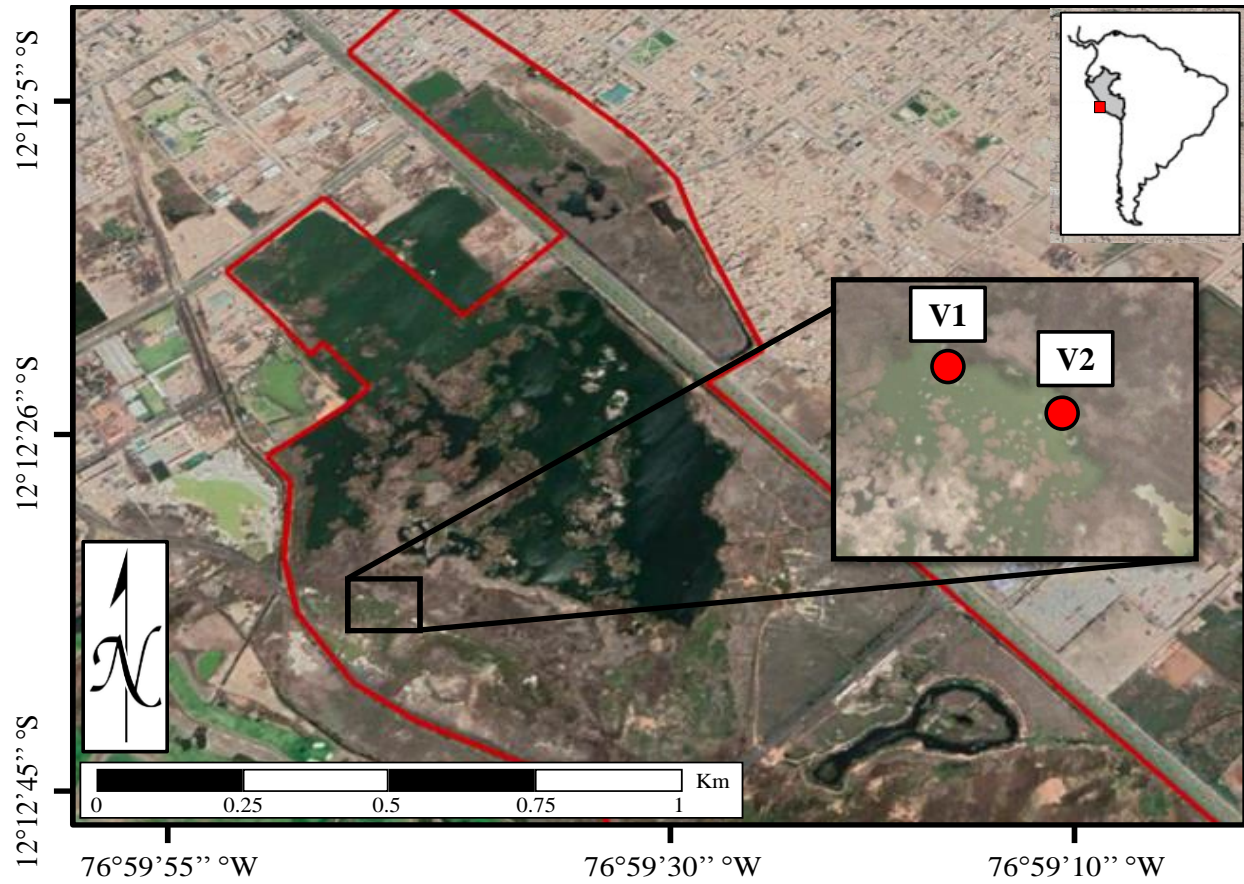


Fig. 1 - Map of the sampling stations within the “RVSPV” saltmarsh ecosystem (Red circles). Saltmarsh bounds are outlined in red. *Satellite image from US Department of State Geographer. Data SIO, NOAA, U.S. Navy, NGA, GEBCO and published by Google Earth 2019.*

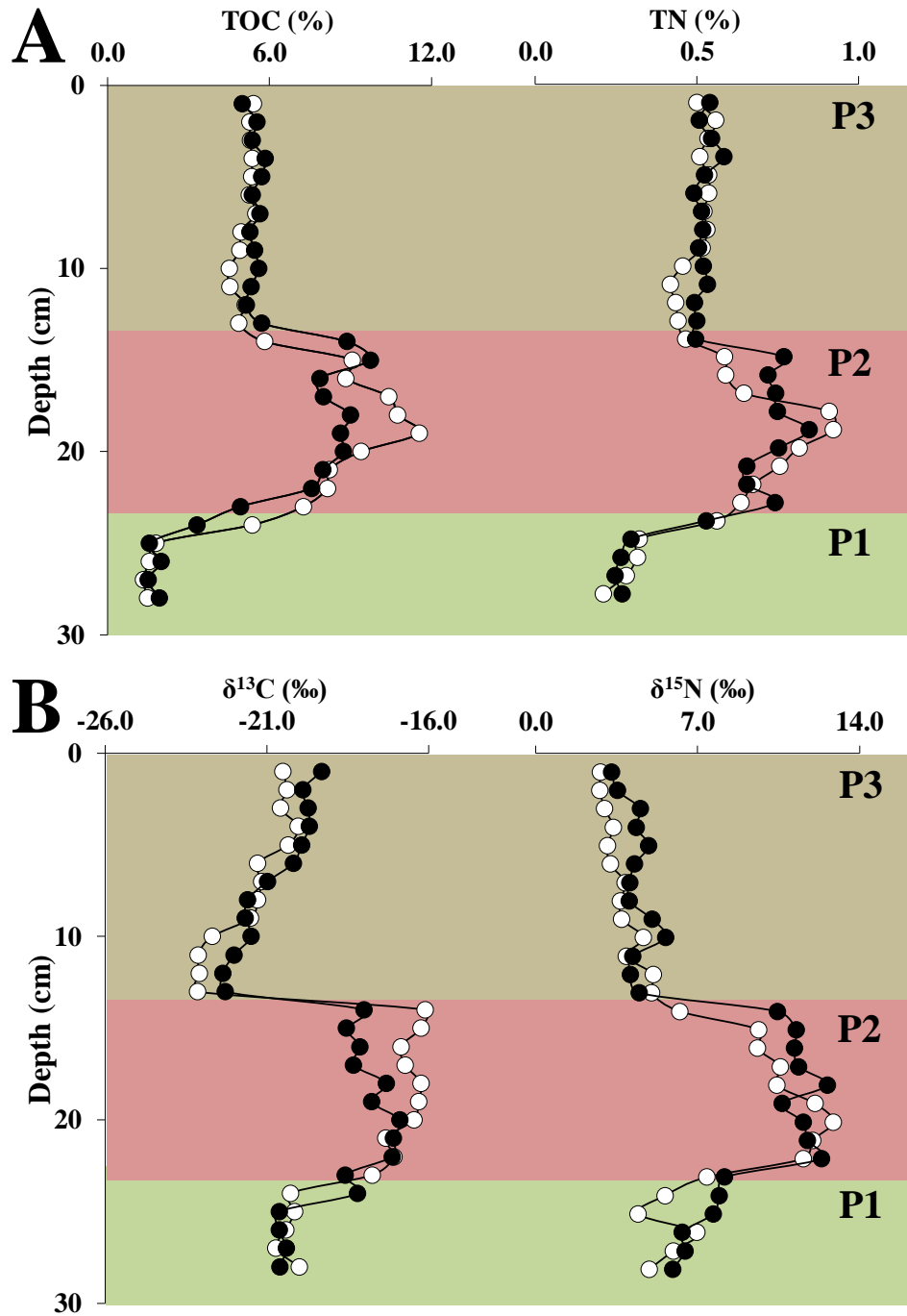


Fig. 2 - (A) Percentage content of total organic carbon (TOC) and total nitrogen (TN) (B) Isotopic signatures of $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$. Different phases are presented: P1 (Green shaded area), during 1960's – 1970's; P2 (Red shaded area), during 1970's – late 1990's; and P3 (Brown shaded area), during 1990's – 2010's. White circles (Core V1), Black Circles (Core V2).

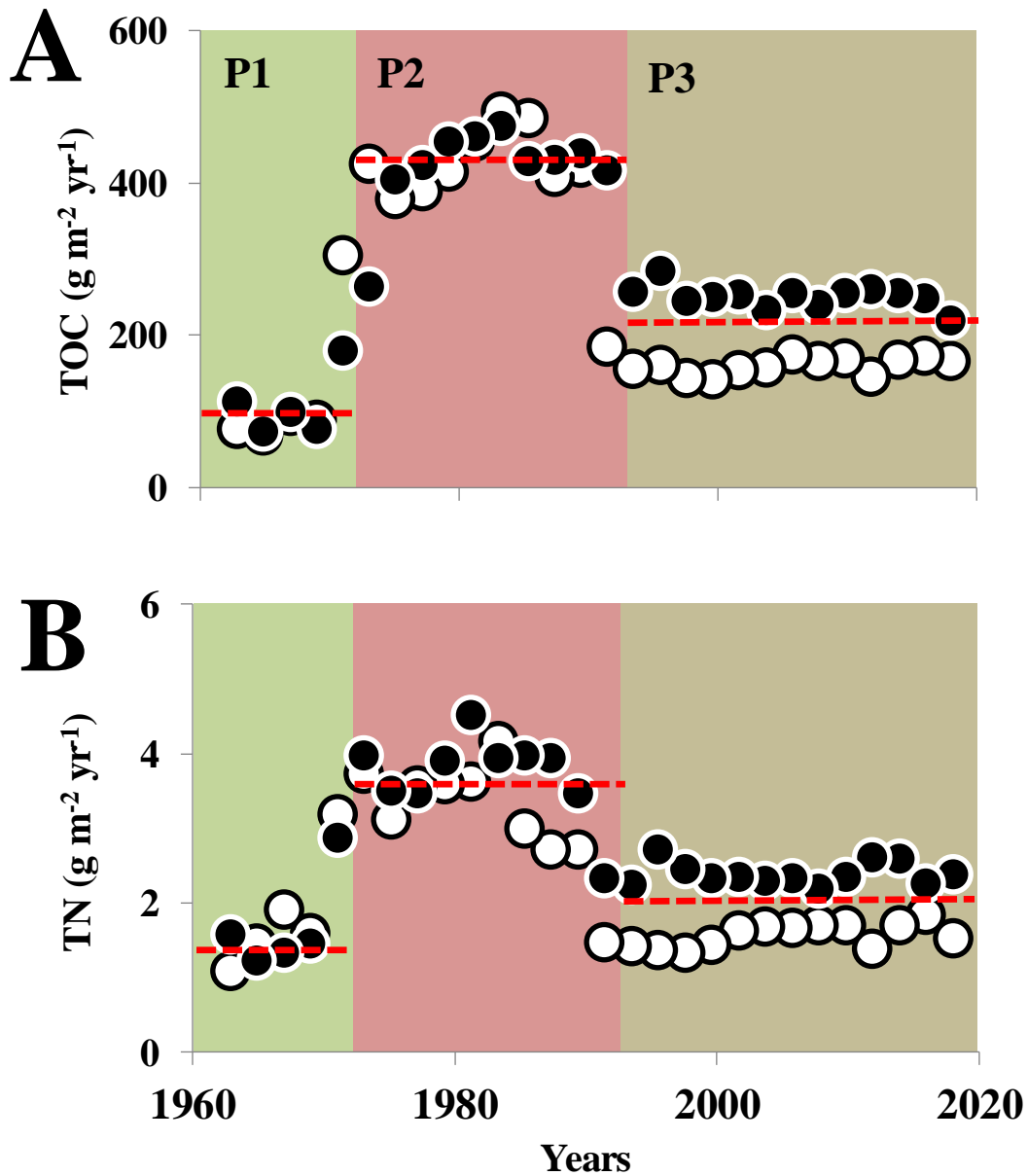
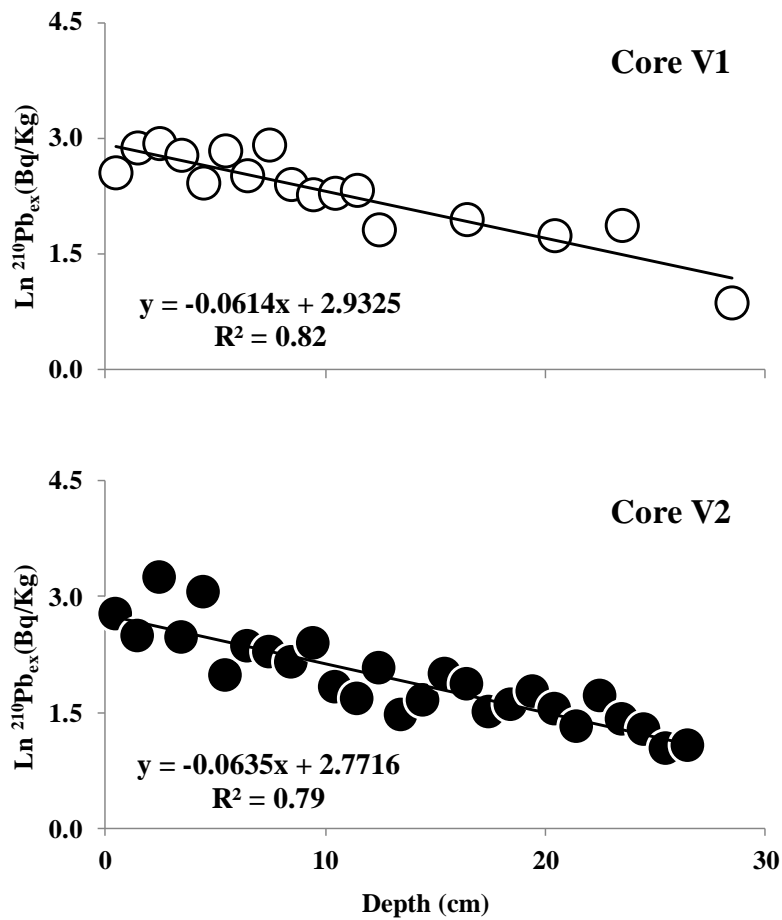
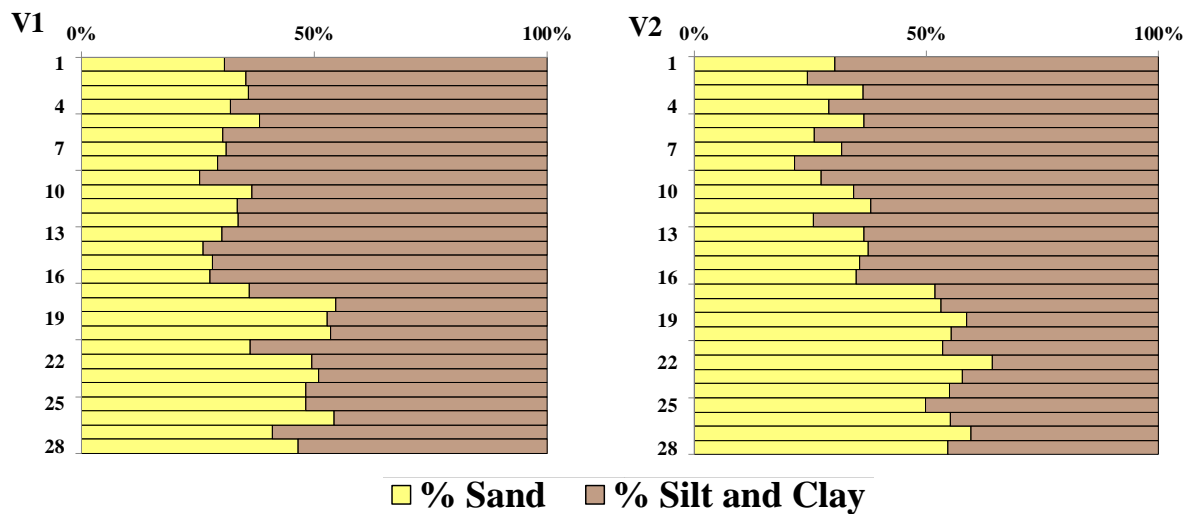


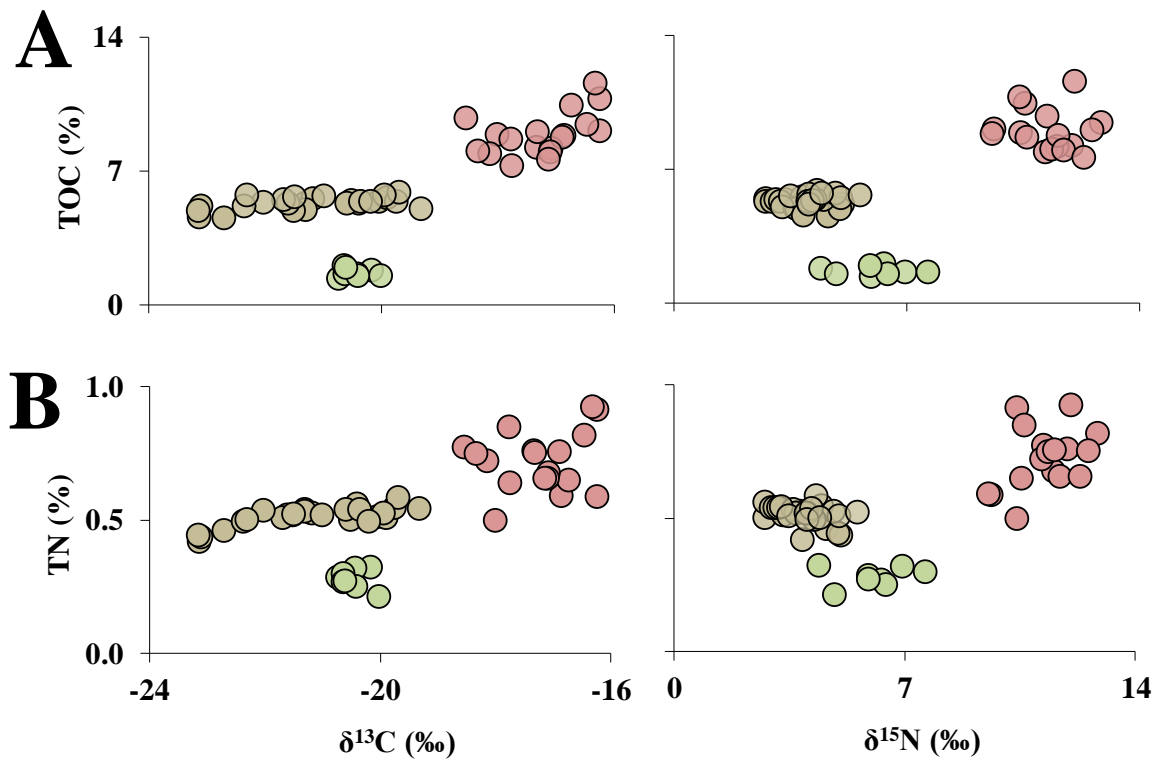
Fig. 3 - (A) **Burial rates** of total organic carbon (TOC) (B) Fluxes of total nitrogen (TN). Different phases are presented: P1 (Green shaded area), during 1960's – 1970's; P2 (Red shaded area), during 1970's – late 1990's; and P3 (Brown shaded area), during 1990's – 2010's. White circles (Core V1), Black Circles (Core V2). Dashed red lines represent average **burial rates**.



ESM-Fig. 1 - Depth variability in $\text{Ln } ^{210}\text{Pb}_{\text{excess}}$ (Bq kg^{-1}) activity for V1 and V2 sediment cores.



ESM-Fig. 2 - Textural composition in terms of sand, silt and clay content for each sediment core.



ESM-Fig. 3 - Relationships of (A) percentage contents of TOC versus $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$, (B) percentage contents of TN versus $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$. All data from V1 and V2 was plotted. Different phases are presented: Phase 1 (Green circles), during 1960's – 1970's (n = 15); Phase 2 (Red circles), during 1970's – 1990's (n = 39); and Phase 3 (Brown circles), during 1990's – 2010's (n = 40).

1 **ESM-Table 1** - Isotopic values \pm SD for carbon ($\delta^{13}\text{C}$) and nitrogen ($\delta^{15}\text{N}$) from fresh *Distichlis spicata*, *Salicornia fructicosa*, *Schoenoplectus americanus*
 2 structures and *phytoplankton*. The samples were collected by triplicate within the study area. "R1" and "R2" represent samples replicates; "X" symbol
 3 represents arithmetic mean from both replicates.

		$\delta^{13}\text{C}$ (‰)	$\delta^{15}\text{N}$ (‰)
Phytoplankton	R1	-18.46	10.79
	R2	-18.79	11.36
	X	-18.6 ± 0.2	11.1 ± 0.4
<i>Distichlis spicata</i>	R1	-24.84	4.14
	R2	-25.47	4.76
	X	-25.2 ± 0.4	4.5 ± 0.4
<i>Salicornia fructicosa</i>	R1	-26.3	3.74
	R2	-26.5	3.68
	X	-26.4 ± 0.1	3.7 ± 0.1
<i>Schoenoplectus americanus</i>	R1	-28.4	3.20
	R2	-27.7	2.90
	X	-28.5 ± 0.4	3.1 ± 0.2

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Declaration of interests

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests:

Credit Author Statement

Carbon and nutrient burial within Peruvian coastal marsh driven by anthropogenic activities

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AP, HA, CJS and MC designed the study and all co-authors approved the final version of the manuscript and agree to be accountable for all aspects of this document.