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5 **Sea Level Rise in the Everglades: Plant-Soil-Microbial Feedbacks in Response to**
6 **Changing Physical Conditions**

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16 **Introduction**

17 Coastal wetlands occupy the intertidal zone between the freshwater Everglades and the Gulf of
18 Mexico. At the interface with the marine environment, these wetlands are dominated by mangrove
19 forests, tidal creeks, and mudflats. Behind the coastal fringe zone is an ecotone that transitions into a
20 freshwater/upland community characterized by an increasing diversity of halophytic to glycophytic trees,
21 shrubs, and herbaceous vegetation. These productive coastal wetlands provide numerous, important
22 ecosystem services. Directly benefiting humans, coastal wetlands function as habitat and nursery grounds
23 for shellfish and other commercially important fisheries, stabilize the coastline, and lessen the impact of
24 storm surges (Aburto-Oropeza et al. 2008, Costanza et al. 2008, Gedan et al. 2011). Moreover, some of
25 the most important functions of coastal wetlands are less tangible processes, including nutrient cycling,
26 pollution removal, and carbon (C) storage (Craft et al. 2009, Gedan et al. 2009, Barbier 2013). Because
27 coastal wetlands are a transitional ecosystem between the land and the ocean, sea level rise represents a
28 significant threat to their global distribution and extent. In order to persist, the soil platform of a coastal
29 wetland must maintain a specific elevation niche relative to the ocean, normally between mean sea level
30 and mean high tide (Morris et al. 2002).

31 This chapter addresses the connections between sea level rise and Everglades soil microbial
32 processes, with a focus on soil C dynamics and the relationship between vegetation, soil, and microbial
33 ecology. Specifically, salinity and inundation are expected to increase under projected sea level rise,
34 hydrologic drivers that are known to alter the structure and productivity of coastal wetland plant
35 communities (e.g., Williams et al 1999, Donnelly and Bertness 2001, Teh et al. 2008, Smith 2009). Plant
36 community change can thus affect the quality and quantity of organic C inputs to the system, which in
37 turn, can influence soil microorganisms and biogeochemical processes (Neubauer et al. 2013, Morrissey
38 et al. 2014). The feedback between plant and soil microbial dynamics, vegetation change, and evidence
39 of such shifts in the Everglades will be discussed. Because soil microbes play a key role in regulating
40 how much organic matter is buried or stored in coastal wetlands, which promotes vertical accretion and
41 resilience to rising sea levels, we will also describe the current understanding of how sea level drivers
42 may directly impact soil microbial activity and diversity. Finally, the chapter will conclude with a
43 discussion of how sea level rise impacts are assessed and the current evidence of sea level rise in the
44 Everglades.

45 **South Florida and Sea Level Rise Throughout History**

46 Changes in global sea level are a natural part of the geologic history of the earth and have always
47 been a key abiotic driver of the ecology of Florida. South Florida, in particular, is considered as being
48 “land from sea” because this region has undergone a repeated history of submergence and emergence
49 from the ocean that is recorded in a stratigraphy of alternating freshwater and marine sediments (Willard
50 and Bernhardt 2011). For instance, during the Eemian interglacial period, which began about 120,000
51 years ago, sea levels were at least 6.6 meters higher than present and south Florida was a shallow bay
52 (Muhs et al. 2011). During this period, the coral reefs formed that would later become the Florida Keys
53 and sediments began to collect on the Florida platform. Approximately 18,000 years ago, at the height of
54 the last glacial period, sea levels dropped to approximately 120 m lower than today, exposing the
55 carbonate platform that includes present-day south Florida (Wanless et al. 1994). This was followed by a
56 period of step-wise rises in sea level that deposited and recycled sediments, allowed for organic matter

57 accumulation, and eventually led to the evolution of the Everglades during a period of relatively stable
58 sea levels, approximately 3,200 years before present (Locker 1996). Sea level began to rise again in
59 approximately 1850 at a rate of $\sim 1.7 \text{ mm y}^{-1}$, and has accelerated since 1993 to a rate of between 2.8 and
60 $3.1 (\pm 0.7) \text{ mm y}^{-1}$. Half of the current rate of sea level rise is attributed to thermal expansion of the ocean
61 and half to melting land ice (IPCC 2007).

62 The flat, emergent wetland-dominated landscape of the Everglades has earned it the nickname,
63 The River of Grass. Over its recent geologic history, the size and shape of the Everglades have changed
64 in concert with sea levels. Currently, approximately 1/3 of the greater Everglades are within 1.5 m of sea
65 level, and half of Everglades National Park lies below 0.6 m of sea level (Titus and Richman 2001).
66 Further, the slope of the Everglades averages just 5-8 cm (2-3 inches) per mile (Lodge 2010, McVoy et al.
67 2011). These geomorphologic conditions make south Florida's natural environments highly vulnerable to
68 increased sea levels. When considering the population that lives immediately adjacent to the Everglades
69 (approximately 6 million people, U.S. Census Bureau), the flat, gently sloping landscape renders the
70 Everglades a virtual "canary in a coal mine" for how both society and the natural environment will
71 respond to sea level rise.

72 As sea level rises, the coastal zone along the Everglades will be increasingly exposed to salinity
73 and inundation. The general thinking is coastal wetlands may respond by: 1) keeping pace with sea level
74 through vertical accretion, 2) migrating landward to maintain an optimal elevation relative to sea level, or
75 3) submerging. It has been predicted that if sea level rises between 0.18-0.59 m before the end of this
76 century, 30% of the world's coastal wetlands will be lost, either to submergence or the inability to migrate
77 landward (IPCC 2007). An increase in mean sea level of 1 m will inundate approximately $4,050 \text{ km}^3$ (4
78 million acres) of wetlands in coastal areas (Neumann et al. 2000). Therefore, understanding the controls
79 over the fate of C stored in coastal soils will be critical in predicting the fate of our coastlines—not to
80 mention the vast reservoir of nutrients (particularly nitrogen (N) and phosphorus (P)) stored in those soils.

81 In south Florida, sea levels are conservatively predicted to rise 0.60 m by 2060 (Zhang et al.
82 2011). This will cause salinity and inundation to increase in fresh and brackish water areas of the Florida

83 coastal zone and will increase the risk of storm surge-induced flooding and saltwater exposure in
84 oligohaline areas of the Everglades at the top of the estuarine ecotone (Teh et al. 2008, Pearlstine et al.
85 2010). The Everglades also has the added risk factor of being a highly modified and managed system
86 with a large number of canals, ditches, and dams used to divert water for flood control, agricultural water
87 supply, and human consumption. For example, the construction of canals on the east and west sides of
88 Lake Okeechobee in the northern Everglades caused a 10-fold increase in the portion of freshwater
89 flowing directly into the Atlantic Ocean and Gulf of Mexico, bypassing the Everglades. Much of this
90 freshwater had previously flowed south through the Everglades and into Florida Bay, effectively
91 countering saltwater along the coastal interface (Nuttle et al. 2000). The change in the quantity, timing,
92 and distribution of freshwater delivery to the coastal zone by this and many other hydrologic alterations
93 throughout the Everglades are believed to be amplifying saltwater intrusion and the rate of the landward
94 migration of coastal habitats, especially in the eastern Everglades (Ross et al. 2000).

95 **Soil Elevation and the Carbon Cycle**

96 Carbon storage and accumulation is vital to the health of the coastal Everglades because it is a
97 major component of the soils and sediments that serve as the wetland platform. Everglades soils are
98 predominately histosols and entisols. Histosols, or peat soils, found in the Everglades coastal zone can be
99 up to 5.5 m deep with total C contents of 15-42% and organic matter contents of 32-89% (Whelan et al.
100 2005, Castaneda 2010, Chambers et al. 2014). These areas often support riverine and fringe mangroves.
101 Everglades entisols are typically either of a marl-type, derived from dry-down and deposition of calcitic
102 periphyton mats, or derived from marine sediments resulting from storm and tidal influences in the
103 coastal zone. These inorganic C- dominated soils are typically shallower and support “dwarf” mangroves,
104 sawgrass, or may be unvegetated.

105 Many coastal wetlands have persisted through centuries of sea level fluctuations due to the
106 natural feedback mechanism of vertical marsh accretion: the accumulation of soil C that leads to an
107 increase in the elevation of the wetland platform (Morris et al. 2002, McKee et al. 2007). This process is
108 a dynamic interplay between sea level and primary production governed by both autochthonous

109 contributions to soil elevation (i.e., belowground production and litter accumulation) and deposition of
110 allochthonous mineral sediments on the soil surface (Morris et al. 2002). The vegetation directly adds
111 organic C to the soil, while also enhancing further sediment deposition (containing both organic and
112 inorganic C) through the slowing of water velocities by aboveground biomass. Studies indicate both
113 organic and inorganic C are important to vertical marsh accretion (Day et al. 2000), with the latter
114 generally comprising a greater percentage of the soil as the proximity to the ocean or rivers increases.
115 Soil C accumulation in coastal wetlands is a balance between the C inputs (i.e., imports and CO₂ fixation)
116 and outputs (i.e., exports, CO₂, and CH₄ flux) from the system. Major C reservoirs include plant biomass,
117 detritus, peat, microbial biomass, and dissolved C (Fig. 1). In order for coastal wetlands to “keep pace”
118 with sea level rise, production and sediment input/retention must exceed losses on a magnitude that
119 mimics or exceeds the current rate of sea level rise (Morris et al. 2002, Day et al. 2011).

120 Several lines of evidence suggest that conditions are generally met for coastal wetland
121 elevations to be sustainable in the context of current rates of sea-level rise. For example, a meta-analysis
122 of the accretion rates from 15 mangrove forests throughout the world indicates approximately 80% of the
123 systems studied are currently accreting at, or faster than, the rate of global sea level rise (Alongi 2008).
124 At least one area of the Everglades is responding in a similar manner. In the coastal fringe mangroves of
125 Shark River Slough, average accretion rates between 1924 and 2009 were 2.5 to 3.6 mm y⁻¹, while sea
126 level was rising at a rate of 2.2 mm y⁻¹ (Smoak et al. 2013). However, rates of sea level rise are not
127 uniform across our coasts, coastal vegetation is not restricted to mangrove forests, and human impacts in
128 the coastal zone, such as freshwater diversion, alter dynamics that may otherwise lead to accretion rates
129 that exceeds rates of sea level rise.

130 The relative importance of organic versus inorganic sediment accumulation necessary to combat
131 coastal submergence has been studied extensively elsewhere (e.g., Nyman et al. 1990, Day et al. 2011),
132 but little information is available for the Everglades. Often, allochthonous deposition is the critical factor
133 for wetland accretion in meso-tidal systems characterized by high sediment loads; these types of coastal
134 wetlands are thought to be the most stable during periods of sea level rise (Morris et al. 2002). Where

135 sediment supplies are low, as is the case in many areas of the coastal Everglades, root biomass and
136 productivity likely become the major contributor to soil accretion. A recent study of mangroves in Belize
137 and Florida showed approximately 50% of production occurs belowground and mangrove roots can
138 account for 1-12 mm of surface elevation increase per year—equivalent to as much as 55% of the annual
139 vertical change in soil elevation (McKee 2011). Furthermore, mangrove roots are fairly refractory and
140 have slow rates of decomposition in the field, especially relative to mangrove leaves (Middleton and
141 McKee 2001). Root decomposition studies by Huxham et al. (2010) and McKee et al. (2007) found about
142 24-60% of root material remained after 1-year of decomposition in the field, depending on species and
143 tidal position. However, root accumulation rates differ by mangrove forest types (e.g., fringe, dwarf, etc.)
144 and are sensitive to variations in the supply of N or P (McKee et al. 2007). A study conducted in Taylor
145 Slough mangroves found that both resource limitations and greater inundation were correlated with
146 increased fine root production, which was a primary driver of soil formation and accretion (Castaneda-
147 Moya et al. 2011). This suggests that differential plant growth responses to nutrient availability or sea
148 level rise stressors, such as inundation, may be key determinants of coastal wetland resilience. The
149 response of mangrove root productivity to nutrient addition is especially important in the coastal
150 Everglades because Florida Bay tends to be the primary source of nutrients (Childers et al. 2006), making
151 increased nutrient supply a possible indirect effect of sea level rise.

152 While root dynamics are thought to be important drivers of soil elevation in the coastal
153 Everglades, storm events that deliver large quantities of inorganic sediment can be significant in localized
154 areas (Castañeda-Moya et al. 2010), as well as seasonal groundwater dynamics that may affect the shrink-
155 swell dynamics of the peat soils (Whelan et al., 2005). For example, a single hurricane (Wilma, 2005)
156 deposited between 0.5 and 4.5 cm of sediment across Shark River Slough and exceeded and annual
157 vertical accretion rate by 8-17 times (Castañeda-Moya et al. 2010). Benthic mats (dominated by algae
158 and cyanobacteria) can also contribute as much as 0.5-5 mm in soil elevation each year, accounting for 9-
159 35% of the total increase in soil elevation (McKee 2011). In addition to the direct contributions benthic

160 mats and inorganic sediments make to soil accretion, their contributions may also increase nutrient
161 availability in the wetland (Castañeda-Moya et al. 2010), which will feed back into primary production.

162 Heterotrophic soil microbes, on the other hand, continually act in opposition to the accumulation
163 of organic matter by utilizing C in senescing leaves and roots, as well as organic exudates from roots, as
164 energy sources. These organisms assist in the break-down of organic polymers into monomers, which can
165 then be assimilated by bacteria and function as electron donors during respiration. The process of organic
166 matter mineralization can result in the release of organic C in a particulate (POC) or dissolved (DOC)
167 form, or as dissolved carbon dioxide (CO₂) or methane (CH₄) gas (Schlesinger 1997). This C may be
168 exported via aquatic transport during the ebb tide, or lost to the atmosphere via soil efflux or diffusion and
169 ebullition through the water column (Dittmar et al. 2006, Bouillon et al. 2008). Laboratory intact soil
170 core studies (excluding vegetation) indicate organic C loss from coastal wetland soils predominately
171 occurs through microbial respiration and the subsequent release of CO₂ back to the atmosphere. Across a
172 salinity gradient of tidal freshwater, brackish, and salt marsh soils, an average of >96% of C lost was
173 emitted as CO₂, ≤3% as CH₄, and <1% as DOC (Chambers et al. 2013). At the ecosystem scale,
174 mangrove research has found a significant portion of C (up to 50% of litter production) is exported as
175 leaves, detrital material, and dissolved C, with the contribution of C to the coastal zone dependent upon
176 the tidal amplitude, season, geomorphology, and productivity (Jennerjahn and Ittekkot 2002, and
177 references therein). Dissolved inorganic C (DIC) export appears to be important in mangrove systems
178 and may exceed DOC export by a factor of 3 to 10 (Bouillon et al. 2008). Finally, a large portion of
179 mangrove organic C remains buried in the soil, with a global average burial rate of 26.1 Tg OC
180 (Breithaupt et al. 2012).

181 Depending on the influence of tides and presence of litter consumers (e.g., crabs), other internal
182 pathways contributing to C loss may prevail. Early work by Robertson et al. (1992) showed that
183 sesarmid crabs can account for the consumption or burial of up to 28% of the total annual litterfall in a
184 mangrove forest, making it easier for by-products to be mineralized by microorganisms or to be exported
185 tidally. Other studies tracking the fate of leaf litter have shown that >30% of the dry mass of leaf litter is

186 leachable, contributing to C loss but also representing a source of nutrients that can facilitate bacterial
187 colonization and increased palatability of the leaf detritus (Davis et al. 2003, Davis et al. 2006). Increased
188 exposure of coastal wetlands to tidal flushing due to sea level rise could affect the relative importance of
189 this type of internal cycling, thus affecting net soil accretion.

190 Considering all of this, it remains unclear if the necessary soil accretion rates can be maintained
191 under the accelerated rate of sea level rise scientists are predicting (e.g., Church and White 2006, Haigh et
192 al. 2014), and what the consequences will be of the water management activities and human development
193 within the greater Everglades watershed. Despite pole-ward expansion of mangroves with global
194 warming and the ability to migrate landward, Alongi (2008) predicts a 10-15% global loss of mangroves
195 under accelerated rates of sea level rise by 2100. Some research forecasts accretion in many coastal
196 wetlands world-wide will be out-paced by rising sea levels, leading to submergence and land loss. This
197 fate can be driven by physical constraints, such as a small tidal range or low sediment supply in the
198 coastal zone (Nicholls et al. 1999), or a biological feedback. For example, submergence is often observed
199 when areas of salt-sensitive coastal vegetation are not colonized by more salinity tolerant species quickly
200 enough. The existing vegetation becomes stunted or dies as a result of water-logging, osmotic stress, the
201 accumulation of toxic hydrogen sulfide (HS^-) from intruding seawater, or salinity induced nutrient
202 inhibition (Koch et al. 1990, Bradley and Morris 1991, Batzer and Shartz, 2006). Because live plants
203 play a vital function in maintaining soil structure, slowing water velocities to allow for sediment
204 deposition, and providing a source of soil organic matter, a decrease in vegetation health can rapidly
205 cascade into peat collapse, ponding, and accelerated submergence (Nyman et al. 1990, DeLaune et al.
206 1994). There is also speculation that rising sea levels could exacerbate erosion along the coastline of the
207 Everglades, which also contributes to submergence (Wanless et al. 1994).

208 *Peat Collapse in the Everglades?*

209 Much of the coastal Everglades include peat soils (Craft and Richardson 2008), which have a
210 very low bulk density (often exceeding 85% pore space by volume; Nyman et al. 1990), making them
211 susceptible to collapse. Peat collapse can occur in a variety of systems under varying circumstances (Day

212 et al. 2011), but is typically initiated by a loss of soil structure, such as by de-watering, accelerated
213 decomposition, or root death. This reduces the strength of the soil matrix and causes the surface soil
214 material to cave-in upon the subsurface soil, resulting in a rapid loss of elevation (DeLaune et al. 1994).
215 Recent geological research in the Everglades has shed light on the potential for collapse of freshwater
216 organic peat soils as they are exposed to increasingly saline conditions (Fig. 2). In a report for Everglades
217 National Park, Wanless and Vlaswinkel (2005) describe a series of “collapse” events that occurred in
218 Cape Sable and Whitewater Bay that they attributed to disturbance caused by channelization, storm surge,
219 sea level rise, and freshwater diversion. These collapse events have led to the exposure of previously
220 freshwater wetlands to more inundated and saline conditions, often converting emergent freshwater marsh
221 areas to open water before mangroves have time to become established and stabilize the soil. Although
222 similar to general “submergence,” in which the depth of inundation increases without a comparable
223 response in vertical accretion and the system slowly converts to open water, peat collapse results in a
224 more rapid transition to open water conditions and can occur away from the aquatic edge, such as near the
225 inland ecotonal boundary of salt-tolerant vegetation, as observed in the Everglades.

226 Freshwater peat collapse could limit the landward colonization of mangrove propagules in
227 response to sea level rise by reducing the soil elevation to a point that it is too deep for new vegetation to
228 become established; this occurrence may be spatially heterogeneous, based on the location of freshwater
229 peat deposits throughout the Everglades. Shallow open water habitat or embayments may develop
230 instead, potentially interrupting the dispersal mechanisms for more salt-tolerant mangrove vegetation to
231 migrate landward. Alternatively, successful colonization of peat collapse areas would hinge upon the
232 ability of vegetation to rapidly accrete new peat soil or mineral sediment, as described earlier, to keep
233 pace with increasing sea level. Areas of south Florida where freshwater diversion have been extensive
234 may be especially vulnerable to wetland loss via peat collapse as sea level rises.

235 Wanless and Vlaswinkel (2005) describe a few plausible explanations for the observed peat
236 collapse in the Everglades. First, over-drainage of some areas may lead to soil drying, organic matter
237 oxidation, and compaction. Next, hurricanes, which occur frequently in south Florida on geological

238 timescales, physically damage vegetation, leading to soil exposure. These soils are then vulnerable to
239 oxidation and collapse until vegetation is reestablished and organic soils are re-stabilized. Finally,
240 Wanless and Vlaswinkel (2005) describe a second scenario where increasing salinity (from sea-level rise
241 or storm surge) stresses, and then kills obligate freshwater vegetation at the top of the estuarine ecotone.
242 Until more salt-tolerant vegetation is reestablished, the organic peats are also vulnerable to oxidation and
243 collapse.

244 The scenarios posed by Wanless and Vlaswinkel (2005) rely on vegetation reestablishment in
245 order to protect peat soils from oxidizing and “collapsing”, as it is the establishment of plants and their
246 investment of belowground biomass (i.e., roots) that serve to stabilize the soils. Studies of peat collapse
247 events in coastal Louisiana also indicate death of belowground biomass is a key instigator and warning
248 sign of impending collapse (Turner et al. 2004). However, accelerated soil mineralization may already be
249 underway while the vegetation is still in place. Additionally, some areas of the coastal Everglades are
250 composed of mineral sediments and soils that developed under the influence of marl-forming periphyton.
251 The response of these types of soils to sea level rise, de-watering, and accelerated mineralization are not
252 understood. In this chapter, we propose a slightly different alternative that considers changes in both the
253 plant-mediated control over peat soil formation and maintenance, as well as the underlying
254 biogeochemical mechanisms behind peat degradation, compaction, and collapse. Recent experimental
255 data combined with concepts in wetland soil biogeochemistry (described later in this chapter) may shed
256 more light on the validity of the peat collapse concept and provide us with better predictive capability as
257 to how Everglades peat soils will respond to sea-level rise.

258 **Vegetation Change**

259 The distribution of vegetation communities across the Everglades is a function of salinity, depth,
260 hydroperiod, and nutrient availability (Daoust and Childers 2004, Barr et al. 2010, Castaneda-Moya et al
261 2013, Troxler et al. 2013). In the coastal zone, vegetation species typically orient in identifiable zones
262 parallel to the shoreline or tidal creeks; these zones are dictated mainly by each species’ tolerance to salt
263 and water-logging (Fig. 3). In general, the Everglades coastal fringe is dominated by red mangrove

264 (*Rhizophora mangle*), followed by black mangrove (*Avicennia germinans*) and white mangroves
265 (*Laguncularia racemosa*) as you move landward. Mangroves are typically able to tolerate high salinities
266 (~30+ ppt) and are adapted to wide fluctuations in both water level and salinity. A sharp ecotone often
267 separates mangroves from the less salinity tolerant communities, such as hardwood hammocks, that are
268 located further inland. This boundary is maintained by a combination of the salinity gradient and a
269 positive feedback mechanism by which each species modifies its' own environment to promote its
270 permanence (Jiang et al. 2011). In some areas of the coastal Everglades, particularly the southeastern
271 region near Taylor Slough, the mangroves have a less abrupt inland ecotone, first becoming interspersed
272 with herbaceous and succulent halophytes (e.g., *Batis maritima*, *Borrchia frutescens* and *Juncus*
273 *romoerianus*), then giving way to Gulf Coast spikerush (*Elecocharis cellulosa*) and less salt tolerant
274 sawgrass (*Cladium jamaicense*) communities (Ross et al. 2000). There are also areas of unvegetated mud
275 flats and limestone pinnacles (Smith et al. 2013). If the coastal Everglades follow the typical model for
276 landward migration, the seaward edge of the mangrove forests will submerge and give-way to subtidal
277 habitats such as tidal flats and seagrass beds. Upstream, oligohaline and freshwater marshes will give
278 way to mangroves. However, the risk of soil collapse in predominantly peat-based oligohaline or
279 freshwater areas needs to be considered; this could lead to the establishment of open water conditions
280 prior to mangrove establishment (Wanless and Vlaswinkel 2005).

281 *Salinity as a driver of vegetation change*

282 Across the coastal Everglades landscape, salinity is a key driver of primary productivity and
283 composition (Castañeda-Moya et al. 2011, 2013, Troxler et al. 2013). However, the salinity gradient is
284 highly dynamic, varying daily (with tides), seasonally (wet and dry season), and over the longer term
285 (e.g., drought periods). During the wet season (May-November), freshwater flow from the northern
286 Everglades penetrates well into the mangrove zone, while reduced flows during the dry season
287 (December-April) allow tidal water to penetrate further upstream. For example, under dry season
288 conditions in lower Taylor Slough, surface water salinity can exceed 40 ppt, and porewater salinity levels
289 in some sawgrass (*C. jamaicense*) marshes can exceed 30 ppt (McIvor et al. 1994, Troxler et al. 2012).

290 During the wet season, the surface water of these same areas is typically fresh (<0.5 ppt). Sawgrass is
291 only weakly salt-tolerant and can be negatively affected by even small increases in salinity, showing signs
292 of physiological stress at salinities as low as 5 ppt (Rejmankova and Macek 2008). During periods of
293 drought in Taylor Slough, the replacement of sawgrass by more salt-loving species such as mangroves has
294 been observed (Ross et al. 2000). Mangrove expansion in other areas of the Everglades has also been
295 noted and thought to be correlated with increased water levels associated with sea level rise (Smith et al.
296 2013). Extended droughts, managed flow reductions, and the construction of canals and ditches can all
297 initiate inland mangrove expansion by allowing the tidal prism to migrate upstream and carry with it
298 mangrove propagules to facilitate mangrove expansion (Fig. 4; Ross et al. 2000).

299 The observed expansion of mangroves can have numerous implications for soil microbiology,
300 although, to our knowledge, this area of research has received little attention in the literature. For
301 example, while both mangroves and sawgrass conservatively store nutrients in tissues of leaves, resulting
302 in high C to nutrient ratios, the quantity and quality of organic matter provided by mangrove wood and
303 leaf litter may be less labile than sawgrass litter. This consideration led Bianchi et al. (2013) to
304 hypothesize that an increased C storage capacity of wetlands would ensue with continued pole-ward
305 expansion of mangroves into coastal zones previously dominated by graminoids (e.g., *Spartina*
306 *alterniflora*). The landward migration of mangroves in the Everglades could alter the impact of
307 disturbance events such as fire, lightning strikes, and hurricanes on light penetration and soil temperature,
308 and could also alter the rate of vegetation-induced sediment deposition. Finally, soil redox potential may
309 be differ under cover of these different plant communities, which affects the utilization of various electron
310 acceptors by soil microbes and influences the overall rate of C mineralization (Reddy and DeLaune
311 2008). Within mangrove forests themselves, there can be species zonation and spatial heterogeneity that
312 produce differences in rhizosphere oxidation, which can affect C mineralization pathways, the availability
313 of nutrients, rates of soil respiration, and other physio-chemical conditions in the soil (Alongi et al. 2000).
314 In contrast to mangroves, soil redox potential in monotypic sawgrass communities seem to vary with

315 water level, but otherwise remain fairly consistent throughout the surficial (0-25 cm) soil and are not
316 significantly affected by nutrient gradients (Qualls et al. 2001).

317 In addition to shifting species composition, salinity can also have a direct physiological effect on
318 vegetation and conditions in the soil. In the Everglades, studies have found a linear decrease in the light-
319 use efficiency of mangroves as salinity increases, suggesting decreased productivity with saltwater
320 intrusion (Barr et al. 2010). Sawgrass (*C. jamaicense*) aboveground net primary production has also been
321 reported to be negatively correlated with surface water salinity, especially periods of high maximum
322 salinities, which primary production rates seem slow to recover from (Childers et al. 2006, Troxler et al.
323 2013). Increasing root phosphatase activity has been detected for sawgrass plants associated with
324 relatively low salinity levels (0.5-5ppt), indicating a strong demand for P (Rejmankova and Macek 2008).
325 Field and laboratory studies conducted elsewhere on marsh vegetation have documented plant mortality
326 and reduced growth in several other common Gulf of Mexico species (e.g., *Sagittaria lancifolia*, *Panicum*
327 *hemitomon*, *Leersia oryzoides*) as salinity increases, with each species having slightly different levels of
328 salt-sensitivity (McKee and Mendelsohn 1989). Wetland vegetation that is not adapted to saltwater often
329 suffers from osmotic stress (Batzer and Shartz 2006) and an accumulation of hydrogen sulfide (HS⁻) in
330 the soil porewater. Hydrogen sulfide, a by-product of sulfate reduction, acts as a phytotoxin and
331 suppresses plant metabolism, reduces growth, and inhibits nutrient uptake (Koch et al. 1990).

332 In some cases, vegetation shifts are thought to be initiated by extreme salinity events, such as
333 storm surges. Even if the intensity and frequency of storm surges is not affected by climate change, sea
334 level rise will still result in an increase in the height of the surge, and thus increase the area of land
335 inundated by a high water event. However, studies suggest episodes of coastal flooding will increase in
336 the coming decades (Najjar et al. 2000). Based on historic data and predictive models, the return period of
337 storm surges throughout Florida is expected to be condensed, such that a 1-in-50 year surge will be
338 experienced roughly every 5 years (Park et al. 2011). This coincides with evidence that the number of
339 maximum water level events (meteorological and storm related) has increased in frequency in south
340 Florida since 1961 (Obeysekera et al. 2011).

341 It is thought that much of the current distribution of mangroves is a product of past hurricanes
342 (Doyle et al. 2003). Storm surges, like droughts, can accelerate the landward migration of the mangrove
343 ecotone by carrying and depositing propagules further inland. The sharp vegetation boundary between
344 the coastal mangroves and hardwood hammocks seems especially vulnerable to saltwater intrusion
345 events. Models predict that just a 1-day saltwater intrusion event of salinities >15 ppt could initiate a
346 transition to a mangrove dominated system within a previously hammock community as a result of
347 salinity stress (Teh et al. 2008). Meanwhile, other research in the Everglades has documented
348 catastrophic damage to mangroves as a result of hurricanes (e.g., wind throws, defoliation, smothering by
349 sediments), leading to a transition to mudflats near the coastline (Smith et al. 2009).

350 *Phosphorus as a driver of vegetation change*

351 Because of the Everglades' legacy as a nutrient-limited system, differentiating between the
352 impacts of sea level rise (salinity and inundation), and the associated increase in nutrient supply
353 (especially P) accompanying saltwater intrusion, can be challenging. Most estuarine wetlands tend to be
354 N-limited, meaning plant productivity is constrained by the availability of N needed for biomass synthesis
355 (Vitousek and Howarth 1991). While N is still an important element for growth and production in the
356 Everglades, P typically regulates the species composition and trophic state of the greater ecosystem (Noe
357 et al. 2001). In its pristine, pre-drainage state, P in the Everglades was naturally low (<10 ug L⁻¹), with
358 much of it being derived from the atmosphere (Belanger et al. 1989). The P limitation was a key driver in
359 evolution of Everglades ecology, favoring the establishment of a unique assemblage of species with low P
360 requirements (e.g., periphyton, sawgrass). Today, the northern Everglades are subject to P loading, mostly
361 from agricultural sources, which has led to a shift in plant communities. This is especially evident in the
362 Water Conservation Areas where periphyton biomass has declined and areas previously occupied almost
363 exclusively by *C. jamaicense* are now monotypic stands of *Typha domingensis* (Davis 1991, McCormick et
364 al. 1998).

365 Understanding the implications of P loading in the northern Everglades is relevant to a discussion
366 of sea level rise because P concentrations in Florida Bay are naturally higher than the un-impacted areas

367 of the Everglades, making the ocean the primary source of P in the coastal zone (Childers et al. 2006,
368 Rivera-Monroy et al. 2007). There is increasing evidence of the importance of marine-derived P in
369 shaping mangrove forest structure and productivity in the Everglades. For example, the reestablishment
370 of a mangrove community damaged by Hurricane Donna (1960) was evaluated to determine the driving
371 forces of structural development. The study found that neither soil salinity nor sulfide concentrations
372 reached levels known to influence species composition, but concentrations of N and P mirrored
373 productivity rates. Both basal area and wood production were highest at the coast (where N and P
374 availability was greatest) and decreased further inland (Chen and Twilley 1999). Research also indicates
375 the importance of marine-derived P in the partitioning of C within mangroves. Trees growing in upstream
376 portions of the estuary and regions with low tidal exchange allocated more biomass belowground, rather
377 than aboveground, in response to the P gradient (Castañeda-Moya et al. 2013). Belowground productivity
378 contributes significantly to soil accretion and preserves soil structure to combat peat collapse (DeLaune et
379 al. 1994, Turner et al. 2004). How an influx of P with saltwater intrusion might affect accretion rates,
380 belowground productivity, and soil stability in the coastal Everglades has not been investigated.

381 *Vegetation-Microbial Interactions*

382 Vegetation change directly impacts soil microbiology by altering the amount and timing of C
383 available to microbes, the lability of the C substrate, and the structural habitat for microbes. This is in
384 addition to the direct impact of the physical changes caused by sea level rise (e.g., increasing salinity,
385 altered nutrient availability, and changing depth and duration of flooding). In general, plant species
386 richness and diversity tend to decrease with increasing salinity (e.g., Odum 1988, Wieski et al. 2010,
387 Sharpe and Baldwin 2012). In the Everglades, sea level rise threatens the future of as many as 21 species
388 of rare, low-lying coastal plants that lack adjacent suitable habitat for species migration (Saha et al. 2011).
389 The decrease in wetland structural complexity caused by salinity also reduces the variety of terrestrial
390 organic matter to serve as a microbial substrate, and could prompt a similar decline in the diversity of the
391 microorganisms that rely on them. Indeed, a variety of studies have shown different plant species and
392 functional guilds support unique microbial assemblages (e.g., Grayston et al. 1998, Troxler et al. 2012),

393 suggesting changes in coastal plant community distribution driven by sea level rise will likely cascade
394 into an alteration of soil microbial ecology. However, no studies to date have directly studied the effect of
395 reduced plant species diversity due to sea level rise on soil microbial diversity.

396 Shifts in vegetation composition can also impact microbes due to differences in the
397 bioavailability of their litter material. For example, polyphenolic compounds such as condensed tannins
398 and lignins, reduce the ability of microbes to degrade those tissues, creating the need for expensive
399 exoenzyme production to break-down these refractory materials (Field and Lettinga 1992, Berg et al.
400 1996). Leached, plant-derived polyphenolic compounds may also have an inhibitory effect on microbial
401 activity (Field and Lettinga 1992). In general, species with high lignin content, high leaf dry matter
402 content, and greater specific leaf area (common characteristics of woody species) have slower decay rates
403 (Prescott 2011). Inputs of organic matter from plants occur as leaf and root litter, and dissolved organic
404 matter (DOM) from litter leaching. While components of freshly leached DOM can be easily degraded
405 and stimulate microbial respiration, litter DOM can also contain less labile components that reflect those
406 chemically-complex compounds found in litter tissue (see Cornwell et al. 2008 for overview). While both
407 the complexity and nutritional composition (structure or C-C bonds and C:N:P) of plants either in leaves
408 or roots are important for soil organic matter (SOM) dynamics, microbial community composition also
409 has a proximate control on SOM (Melillo et al. 1982). For instance, fungal and actinomycete bacterial
410 species are among the most efficient in degrading more complex C compounds, but are associated with
411 degradation in aerobic, low nutrient environments (Goodfellow and Williams 1983, McCarthy et al. 1987,
412 Güsewell and Gessner 2009, Peltoniemi et al. 2009). With adequate nutrient supply, microbes may also
413 synthesize metabolically expensive enzymes to acquire complex C molecules of recalcitrant tissues
414 (Moorehead and Sinsabaugh 2006). Recent research suggests SOM quantity and quality (as indicated by
415 C:N) is inversely related to salinity in oligohaline wetlands (Morrissey et al. 2014b). In the Everglades,
416 the expansion of mangroves into areas previously dominated by hardwood hammocks or graminoid is
417 expected to result in significant changes in the quality of liter material and alter SOM dynamics in the
418 coastal zone.

419 **Microbial Ecology and Biogeochemistry**

420 The connection between sea level rise and microbiology is an emerging area of research with
421 broad implications given the strong connection between hydrology, plant production, soil microbes, and
422 soil C storage. Soil microbes are often the first organisms in a wetland to respond to environmental
423 changes due to their large surface-to-volume ratio and rapid turnover rate. Although microbial changes
424 are less visible than vegetation shifts, they often occur within a matter of hours or days following an event
425 and could involve alterations in community function, composition, and diversity. For example,
426 periphyton mats in the Everglades (a mixture of cyanobacteria, algae, and microinvertebrates), are highly
427 sensitive to changes in P concentrations and begin to show changes in species composition within a few
428 weeks—long before changes can be observed in the soil or vegetation (McCormick et al. 2001). In
429 coastal wetlands, saltwater intrusion, increased inundation, and storm surge events linked to sea level rise
430 could all directly alter soil microbial community structure, activity, and subsequently the balance of soil C
431 that controls how resilient a wetland is to sea level rise (e.g., Chambers et al. 2011, 2013, Neubauer et al.
432 2013).

433 From the most fundamental perspective, salinity increases the ionic strength and conductivity of
434 the microbial environment in coastal wetlands. In order to survive in saline conditions, microorganisms
435 must be capable of osmoregulation, which can involve either the accumulation of potassium chloride in
436 the cytoplasm, or the biosynthesis and/or accumulation of compatible solutes (Oren 2008). Salt intolerant
437 species without these capabilities will experience osmotic stress, disruptions in metabolic function, or
438 even cell lysis upon exposure to salinity (e.g., Reitz and Haynes 2003, Wichern et al. 2006). There is a
439 long history of research pertaining to the effects of salt accumulation in upland soils that demonstrates salt
440 can reduce the size of the soil microbial community and microbial activity, as indicated by lower rates of
441 CO₂ and CH₄ production (Pattnaik et al. 2000, Muhammad et al. 2006, Gennari et al. 2007). It is believed
442 the higher conductivity in the soil-water environment causes osmotic/ionic stress to the organisms and
443 leads to an overall decrease in the rate of C cycling in these systems (Frankenberger and Bingham 1982,
444 Gennari et al. 2007). Few studies have investigated the direct impacts of ionic strength on soil

445 microbiology in wetlands or aquatic systems. A laboratory study where a freshwater wetland soil was
446 exposed to an increase in conductivity from 0 to ~28 mS cm⁻¹ through the addition of NaCl demonstrated
447 a 30 % decline in microbial respiration over a period of 3 weeks (Chambers et al. 2011). Whether this
448 decline resulted from a general reduction in microbial activity or a shift in community structure remains
449 unclear.

450 When addressing the impact of sea level rise on wetland soil microbiology, another
451 environmental change may be even more critical than the increase in ionic strength—an increase in the
452 sulfate, SO₄²⁻, concentration. Unlike inland salinity discussed above, seawater contains a consistent ratio
453 of ions, of which SO₄²⁻ is the third most abundant. Sulfate functions as a terminal electron acceptor
454 (TEA) that soil microbes can utilize for anaerobic respiration. In most terrestrial ecosystems, the
455 presence of SO₄²⁻ is of little consequence to the microbial biota because TEAs are plentiful. The most
456 commonly used TEA by heterotrophic bacteria is oxygen, which utilizes C as an energy source (electron
457 donor) and produces energy in a relatively efficient manner. The presence of water reduces the diffusion
458 of oxygen into the soil by 10,000 times, requiring microbes in wetlands and coastal ecosystems to rely on
459 alternative TEAs, which they utilize in a specific sequence based on their availability and potential energy
460 yield (Patrick and DeLaune 1977; Table 1). Sulfate is near the bottom of the energy cascade and is used
461 only when the environment becomes sufficiently reduced (as indicated by an oxidation reduction potential
462 (Eh) of <-100 mV) and produces only -0.7 x 10⁻³ kJ mol⁻¹ of energy.

463 Sulfate reduction is typically the dominant pathway of microbial respiration in brackish and
464 saline marshes and mangroves (Howarth 1984, Weston et al. 2006, Kristensen et al. 2008). This suggests
465 that although an increase in ionic strength from seawater intrusion may slightly suppress microbial
466 activity by causing osmotic stress, there will also be a stimulatory effect from the influx of SO₄²⁻
467 providing an abundance of new TEAs for anaerobic respiration. This was demonstrated under laboratory
468 conditions where CO₂ production rates showed a short-term increase (20-32%) in proportion to the
469 concentration of sulfate added to a freshwater wetland soil (Chambers et al. 2011). Additions of 10 ppt
470 seawater to freshwater sediment cores can cause SO₄²⁻ reduction to become the dominant pathway for

471 microbial respiration after only 12 days, and account for 95% of all organic C oxidation after 35 days of
472 exposure (Weston et al. 2006). However, the stimulation in the overall rate of C loss through respiration
473 may be short-lived as other factors, such as the availability of labile C substrates or nutrients, become
474 limiting (Chambers et al. 2011). A longer-term manipulative field study in a freshwater tidal marsh found
475 CO₂ flux actually declined in treatments exposed to increased salinity for 3.5 years; this decline was
476 correlated with a reduction in the quality of the SOM, further demonstrating the importance of plant-
477 microbial interactions (Neubauer et al. 2013). Interestingly, while the stimulatory effect of SO₄²⁻ on
478 respiration appears to be somewhat transient, the suppression of methanogenesis through competitive
479 inhibition by sulfate reducers appears to persist over time (Chambers et al. 2011, Neubauer et al. 2013).
480 In the laboratory, a pulse of brackish water (13 ppt) in a freshwater wetland soil reduced CH₄ flux by 97%
481 in just 5 days (Chambers et al. 2013). In the field, oligohaline water additions to a freshwater wetland soil
482 caused a 2 to 3-fold decrease in CH₄ production that persisted for 3.5 y (Neubauer et al. 2013).

483 Hydroperiod is another environmental driver of soil microbial processes. In the coastal
484 Everglades, hydroperiod fluctuates seasonally based on rainfall, and daily, based on semi-diurnal tides.
485 During low water (low tide) conditions, more oxygen can diffuse into the soil to promote aerobic
486 respiration. For this reason, low tide CO₂ production rates can be between 50-300% higher in coastal
487 wetlands than high tide CO₂ production rates, with the variability attributed to differences in the hydraulic
488 conductivity of the soil (Chambers et al. 2013). However, as sea level rises, we can expect deeper, more
489 prolonged periods of inundation. Even wetlands that are accreting vertically at a pace comparable to sea
490 level rise tend to do so in a step-wise manner, creating a lag phase in which inundation is greater than
491 under static sea level conditions (Kirwan and Temmerman 2009). Longer periods of water-logging
492 generally slow down microbial activity because organisms must rely exclusively on anaerobic pathways,
493 which tend to be slower and less efficient than aerobic respiration. In a mesocosm study that simulated
494 sea level rise in an Everglades mangrove peat soil, soil organic C loss was 90% higher under control
495 water levels, as compared increased inundation, when combined with elevated salinities (Chambers et al.
496 2014). This may reduce the amount of organic C lost through the microbial pathway as hydroperiod

497 increases, promoting soil C storage and accretion. However, the same study also demonstrated that
498 prolonged inundation may actually result in a loss of soil material, as seen by a decrease in surface (0-5
499 cm) soil bulk density (Chambers et al. 2014). While the mechanism for this reduction in bulk density is
500 not known, it was correlated with an increase in porewater dissolved organic C, causing speculation it
501 may be a product of excessive leaching during water-logging or increased shear stress due to the deeper
502 water column above the soil (Chambers et al. 2014).

503 It is generally thought microbial density and diversity is comparable in freshwater and saltwater
504 systems, but the identity of the individual organisms themselves differs with salinity (Capone and Kiene
505 1988). However, new evidence contradicts this axiom, finding higher microbial biomass-associated C in
506 salt marsh soils, compared to freshwater and brackish marsh soils (Chambers et al. 2013). Other research
507 has found a direct correlation between bacterial abundance and salinity in freshwater tidal marshes
508 (Morrissey et al. 2014b). Only one study of microbial community composition has been performed in the
509 Everglades along a salinity transect from 0 to 49 ppt. Here, the diversity of the microbial community
510 remained similar, but the identity of the microbes diverged significantly, based primarily on salinity, and
511 secondarily on P availability (Ikenaga et al. 2010).

512 Ultimately, while soil microbial communities may be rapid indicators of wetland ecosystem
513 response, the feedback between plant and microbial communities will modulate this response. For
514 example, the presence or absence of plants will have significant influences on soil redox potential (i.e.,
515 the size of the oxidized rhizosphere), quantity and quality of organic C, and will interact with enzyme
516 synthesis. Under conditions in which salinity drives a decline of plant productivity, vegetation death, or
517 vegetation community shifts, diversity and function of soil microbial communities will be fundamentally
518 altered. Unfortunately, plant-microbial interactions are not well understood in wetlands, especially in
519 coastal peatlands.

520 **The Future of the Coastal Everglades**

521 As salinity and inundation patterns in the Everglades change in response to sea level rise and
522 human-driven alterations in hydrology, coastal zone ecology is changing as well. The movement of

523 ecotones (regions bridging two distinct community types) is often a reliable way of monitoring
524 environmental change because ecotones develop and migrate in response to specific environmental
525 gradients. Several studies have used historical aerial photography and various bio-indicators to document
526 shifts in the location of coastal ecotones in the Everglades over time. This provides a glimpse of how the
527 ecosystem has responded to past sea level changes and a basis for predicting future ecosystem responses.

528 The land boundary of the Everglades, and all of Florida, has changed significantly throughout
529 geologic time in response to sea level. Soil cores indicate the current seaward edge of the Everglades
530 formed from red mangrove (*R. mangle*) derived peats that began accumulating approximately 3,500 years
531 B.P. (Parkinson et al. 1989). As sea level rose, the mangrove soil platform accreted vertically and
532 expanded landward. Meanwhile, the establishment and growth of oyster reefs off the coast also allowed
533 mangroves to expand in the seaward direction, creating mangrove islands (Parkinson et al. 1989). The
534 distribution of mangroves, and peat accumulation from relic mangrove forests, are considered good
535 indicators of historic sea level because they always occupy the upper portion of the tidal range (Scholl
536 1964).

537 Current research shows that mangroves within the Everglades are continuing to respond to
538 changing sea levels, mainly through the expansion of their coverage at the expense of inland marsh
539 habitat (Doyle et al. 2003). A look at aerial photographs of the Ten Thousand Islands National Wildlife
540 Refuge on the western edge of the Everglades has shown a 35% increase in mangrove coverage in the
541 past 78 years. The construction of canals near the coast is a driving force in the expansion of mangroves
542 within previously low salinity marshes because they provide a conduit for saltwater intrusion and
543 propagule dispersal (Krauss et al. 2011). In a region near Taylor Slough, known as the Southeast Saline
544 Everglades, researchers have documented the movement of inland ecotones since the 1940s. Here the
545 boundary between the mangrove-graminoid community and the interior sawgrass marsh has shifted inland
546 3.3 km, which is believed to be in response to a combination of reduced freshwater flows and encroaching
547 seas (Ross et al. 2000, Troxler 2012). In the southeast Everglades, extensive water diversions and flow
548 alterations are accelerating saltwater intrusion and may be a useful model for predicting sea level rise

549 effects elsewhere. According to mollusk records in the coastal soils near Biscayne Bay, prior to local
550 drainage efforts that began ~70 years ago, the marsh/mangrove ecotone was migrating landward at a rate
551 of 0.14 m y^{-1} , but since drainage, the rate has increased to 3.1 m y^{-1} (Gaiser et al. 2006). This migration is
552 correlated with an increase in salinity from 2 ppt to 13.2 ppt. In addition to general mangrove expansion,
553 models also predict the mangroves themselves will have reduced height and contain a greater proportion
554 of red mangroves as sea level rises (Doyle et al. 2003).

555 While there has been significant research demonstrating that the inland ecotones of the coastal
556 Everglades are migrating landward with rising sea levels, there are large uncertainties about how the
557 balance of coastal erosion and soil accretion will determine the position of the land boundary as sea level
558 rises. Some studies suggest that coastal erosion directly resulting from sea level rise is low, with most
559 documented erosional events accompanying hurricanes (Doyle et al. 2003, Wanless et al. 1994). At
560 present, mangrove soil accretion rates are exceeding sea level rise rates in at least one area of the
561 Everglades (Smoak et al. 2012), but more research is needed to understand how accretion may vary
562 spatially. The fact that seawater serves as the primary source of nutrients to the coastal zone (Childers et
563 al. 2006) suggests that saltwater intrusion could increase productivity (Chen and Twilley 1999), and
564 subsequently soil accretion. However, increasing salinity may have the opposite effect due to the
565 complicated relationship with concomitant physical forcings such as inundation depth, sediment supply,
566 and disturbances (e.g., fire, wind, storm surge). The interaction of salinity-induced collapse of freshwater
567 peats and mangrove transition is also worth considering as a driver shaping the coastal Everglades of the
568 future.

569 As discussed earlier, Wanless and Vlaswinkel (2005) suggest that collapsed areas of freshwater
570 peat—as observed in the Cape Sable area of Everglades National Park—may coalesce through time,
571 resulting in larger open water areas. In fact, it is believed that this phenomenon contributed to the
572 formation of Whitewater Bay in Everglades National Park (Wanless and Vlaswinkel 2005). One could
573 hypothesize that unless collapsed areas receive new sediment or are colonized by mangroves that can re-
574 stabilize the soil, they may continue to grow larger and transition directly into subtidal habitat as sea

575 levels continue to rise. With 8,744 km² of south and southwest Florida being located below the 1.5m
576 elevation contour (Titus and Richman 2001) and rates of sea level rise thought to be accelerating (Church
577 and White 2006), the fate of the seaward boundary of the Everglades remains highly uncertain.

578 **Conclusion**

579 Overall, there is ample evidence to support the idea that coastal ecology in the Everglades is
580 changing in concert with rising sea levels. In the southeast Everglades, the signature of sea level rise has
581 been blurred with significant hydrologic modifications that are decreasing freshwater flows and
582 accelerating saltwater intrusion (Ross et al. 2000). However, ecotone shifts and elevated salinities are
583 being documented across the entire ecosystem. Changes in vegetation communities are often the most
584 apparent manifestation of migrating environmental gradients and directly impact soil microbiology by
585 altering the physical and chemical environment. However, the structure and function of microbial
586 communities will likely respond far in advance of vegetation shifts. Few studies have addressed the
587 direct impacts of sea level rise on soil micro-biota, but evidence such as changes in soil respiration rates
588 (CO₂ production) suggest seawater intrusion can accelerate heterotrophic microbial activity for the short-
589 term and suppress methanogenesis for the long-term. The important question is how changes in
590 vegetation type, productivity, and microbial activity will affect the overall balance of C in coastal
591 wetlands. An increase in C inputs and net decrease in microbial respiration would create a positive
592 feedback to promote vertical accretion and increase wetland resilience. In the most likely scenario,
593 different regions of the Everglades will have unique responses to sea level based on the current health of
594 the ecosystem, the supply of inorganic sediments, nutrient availability, topography, and occurrence of
595 extreme events. Future research should focus on quantifying the impact to microbial populations and
596 vital microbial processes such as nutrient cycling, C storage, and the plant-soil interactions that modulate
597 the stability of coastal peatlands vulnerable to sea-level rise, such as the Everglades.

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918 **Tables**

919 Table 1. Theoretical energy yields (the more negative the value, the greater the net energy gain) calculated
 920 as reactions coupled with glucose oxidation ($C_6H_{12}O_6 \rightarrow CO_2$) and H_2 oxidation ($H_2 \rightarrow H^+$).

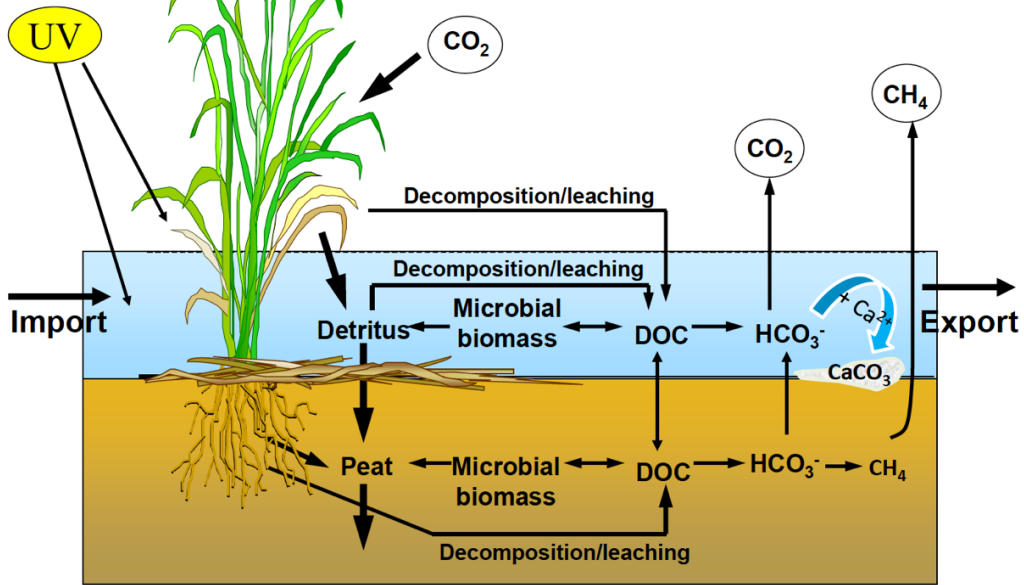
Alternative Electron Acceptor	Dominant C End-Product	Eh Range (@ pH 7)	Theoretical Energy Yield (ΔG°_R)
NO_3^-	CO_2	250 to 350 mV	$-14.5 \times 10^{-3} \text{ kJ mol}^{-1}$
Mn^{4+}	CO_2	220 to 300 mV	$-3.0 \times 10^{-3} \text{ kJ mol}^{-1}$
Fe^{3+}	CO_2	120 to 180 mV	$-1.7 \times 10^{-3} \text{ kJ mol}^{-1}$
SO_4^{2-}	CO_2	-100 to -250 mV	$-0.7 \times 10^{-3} \text{ kJ mol}^{-1}$
CO_2	CH_4	< -250 mV	$-0.1 \times 10^{-3} \text{ kJ mol}^{-1}$

921

922 **Figures**

923 Figure 1. A generalized depiction of the carbon cycle in wetlands, including the major storage reservoirs
 924 and transformations.

Carbon Cycle



Modified from: K. R. Reddy, and R. Delaune 2008

925

926 Figure 2. Photographs showing evidence of peat collapse in a sawgrass (*Cladium jamaicense*) marsh
 927 surrounded by an expanding mangrove forest in lower Shark River Slough, Everglades National Park.

928 [Photo credit: South Florida Water Management District.]



929

930

931 Figure 3. Typical vegetation zonation patterns observed in the coastal Everglades, with mangroves
932 occupying the land fringe, followed by one or more ecotonal boundaries into more salt tolerant vegetation
933 [Photo credit: Lisa G. Chambers].



934
935 Figure 4. Mangroves expanding landward along tidal creeks in the coastal Everglades, likely facilitated
936 by saltwater intrusion and propagule dispersal [Photo credit: Lisa G. Chambers].



937

938