Drift macroalgal distribution in northern Gulf of Mexico seagrass meadows

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Drift macroalgae, often found in clumps or mats adjacent to or within seagrass beds, can increase the value of seagrass beds as habitat for nekton via added food resources and structural complexity. But, as algal biomass increases, it can also decrease light availability, inhibit faunal movements, smother benthic communities, and contribute to hypoxia, all of which can reduce nekton abundance. We quantified the abundance and distribution of drift macroalgae within seagrass meadows dominated by turtle grass *Thalassia testudinum* across the northern Gulf of Mexico and compared seagrass characteristics to macroalgal biomass and distribution. Drift macroalgae were most abundant in areas with higher seagrass shoot densities and intermediate canopy heights. We did not find significant relationships between algal biomass and point measures of salinity, temperature, or depth. The macroalgal genera *Laurencia* and *Gracilaria* were present across the study region, *Agardhiella* and *Digenia* were collected in the western Gulf of Mexico, and *Acanthophora* was collected in the eastern Gulf of Mexico. Our survey revealed drift algae to be abundant and widespread throughout seagrass meadows in the northern Gulf of Mexico, with likely influences the habitat value of seagrass ecosystems.

1 **Drift macroalgal distribution in northern Gulf of Mexico seagrass meadows**

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23 **ABSTRACT**

24   Drift macroalgae, often found in clumps or mats adjacent to or within seagrass beds, can

25 increase the value of seagrass beds as habitat for nekton via added food resources and structural

26 complexity. But, as algal biomass increases, it can also decrease light availability, inhibit faunal

27 movements, smother benthic communities, and contribute to hypoxia, all of which can reduce

28 nekton abundance. We quantified the abundance and distribution of drift macroalgae within

29 seagrass meadows dominated by turtle grass *Thalassia testudinum* across the northern Gulf of

30 Mexico and compared seagrass characteristics to macroalgal biomass and distribution. Drift

31 macroalgae were most abundant in areas with higher seagrass shoot densities and intermediate

32 canopy heights. We did not find significant relationships between algal biomass and point

33 measures of salinity, temperature, or depth. The macroalgal genera *Laurencia* and *Gracilaria*

34 were present across the study region, *Agardhiella* and *Digenia* were collected in the western Gulf

35 of Mexico, and *Acanthophora* was collected in the eastern Gulf of Mexico. Our survey revealed

36 drift algae to be abundant and widespread throughout seagrass meadows in the northern Gulf of

37 Mexico, with likely influences the habitat value of seagrass ecosystems.

38

39 **INTRODUCTION**

40   Drift macroalgae often originate as attached algae on seagrass leaves and other hard

41 substratum before becoming uprooted by various physical disturbances (e.g., currents, waves)

42 (Norton & Mathieson, 1983; Bell & Hall, 1997; Biber, 2002; Lirman et al., 2003). They are

43 commonly found in small patches within and around the calm, coastal seagrass meadows from

44 early spring to mid-summer (Norton & Mathieson, 1983), with distributions influenced by water

45 currents as well as the roughness of the surrounding substrate (Bell et al., 1995; Bell & Hall,

46 1997; Biber, 2007; Fonseca & Koehl, 2006). The ecosystem benefits of seagrass habitats are

47 strongly correlated with their structural complexity, and macroalgal communities often further

48 increase this complexity 3-100-fold (Morris & Hall, 2001; Kingsford, 1995). Seagrass and

49 macroalgae are major constituents in some of the most productive coastal ecosystems and they

50 enhance fisheries by providing valuable nursery habitat for a variety of finfish and invertebrate

51 fauna (e.g., Carr, 1991; Jackson et al., 2001; Heck, Hays & Orth, 2003; Guido et al., 2004; Bos et

52 al., 2007). In the Mediterranean, seagrass and macroalgal habitats support 30-40% of the

53 commercial fish and 29% of recreational fish during their juvenile life stages (Jackson, Wilding

54 & Attrill, 2015). The structure provided by seagrass and macroalgal communities can enhance

55 feeding and growth rates, while lowering predation rates for many shrimp, crab, and fish species

56 (Orth, Heck & van Montfrans, 1984; Kingsford & Choat, 1985; Bax, 1998; Rooker, Holt & Holt,

57 1998; Nagelkerken et al., 2002). The detached nature of drift macroalgal communities can also

58 aid in dispersal of many small fish and invertebrates utilizing this structure (Astill & Lavery,

59 2001; Holmquist, 1994).

60   While macroalgae can enhance the ecosystem services of seagrasses, at sufficiently high

61 biomass, macroalgae can lead to declines in organismal abundance and species diversity (e.g.,

62 Hull, 1987; Bonsdorff, 1992; Zajac & McCarthy, 2015). The degree of this change may be tied

63 to species-specific morphological traits (Bartholomew et al., 2000). Green filamentous

64 macroalgae in the Baltic Sea, for example, is problematic at high concentrations, causing

65 hypoxia and altering the resident benthic communities (Vahteri et al., 2000). Massive influxes of

66 the brown alga *Sargassum* to coastal systems have also led to similar declines in flora and fauna

67 in the Caribbean, causing benthic mortality and decreasing habitat value (Chávez et al., 2020).

68 However, brown algae in New Zealand led to an increase in fish and invertebrate abundance,

69 with higher species abundances relative to attached vegetation and open water areas (Kingsford

70 & Coat, 1985). Understanding the algal composition throughout the northern Gulf of Mexico and

71 seasonal changes in biomass may allow us to better understand the species-specific effects that

72 algae have on seagrass and their associated nekton communities.

73 Algae are classified into three evolutionarily distinct lineages based on variations in their

74 morphological characteristics and tissue pigment composition, and consist of brown algae

75 (Phaeophyceae), green algae (Chlorophyta), and red algae (Rhodophyta), with rhodophytes being

76 the dominant taxa in the northern Gulf of Mexico (Virnstein & Carbonara, 1985; Holmquist,

77 1997). Over the last few decades, seagrass areal coverage has declined in many areas, including

78 the northern Gulf of Mexico and southern Atlantic estuaries of North America (Hall et al., 1999;

79 Peneva, Griffith & Carter, 2008; Carter et al., 2011). Macroalgal blooms have increased in

80 frequency and intensity (e.g., Benz, Eiseman & Gallaher, 1979; Virnstein & Carbonara, 1985;

81 Zieman, Fourqurean & Iverson, 1989; Kopecky & Dunton, 2006; Fredericq et al., 2009) and are

82 predicted to proliferate under future scenarios of warmer sea surface temperatures and ocean

83 acidification (Brodie et al., 2014). Consequently, the functional role of drift algae may increase

84 in importance in regions where seagrasses have declined. However, drift algae are known for

85 being highly variable across space and time (Benz, Eiseman & Gallaher, 1979; Bell & Hall,

86 1997), with their abundance and movement within estuaries rarely quantified and difficult to

87 track, complicating our understanding of drift algae in seagrass ecosystems. Although

88 underappreciated for their habitat value, drift macroalgae likely augment the value of seagrass

89 beds as habitat for nekton by increasing the physical structure within and adjacent to seagrass

90 beds. We surveyed seagrass meadows across the northern Gulf of Mexico and quantified (1) the

91   abundance of drift macroalgae in early and late summer and (2) the relationship between algal

92   abundance and environmental and seagrass metrics.

93

94   **MATERIALS & METHODS**

95   **Study regions.** Five estuaries in the northern Gulf of Mexico, each containing at least 20

96   sites within seagrass meadows, were surveyed twice during the early (May–June) and late

97   (August–September) summer of 2018 (Fig. 1). Sites were selected by overlaying a tessellated

98   hexagonal grid (500 m edge length) on each estuary in ArcGIS (Moore, 2009; Neckles et al.,

99   2012; Wilson & Dunton, 2012). A randomly generated site within each of 20 to 25 grid cells that

100   contained more than 50% seagrass coverage and a minimum of 500 m separation were selected

101   for assessment (Belgrad et al., 2021). Across all regions, turtle grass (*Thalassia testudinum*) was

102   the dominant macrophytic taxon; however, manatee grass (*Syringodium filiforme*) and shoal

103   grass (*Halodule wrightii*) were also common. Star grass (*Halophila engelmannii*) and widgeon

104   grass (*Ruppia maritima*) were present but occurred in <0.01% of surveys and are not considered

105   in this study. *In situ* measurements of seagrass coverage ranged from 0-100%. Measurements

106   were collected across twenty sites within Laguna Madre, TX (LM; 26°08'N, 97°14'W), Corpus

107   Christi Bay, TX (CB; 27°51'N, 97°08'W), and the Chandeleur Islands, LA (LA; 29°54'N,

108   88°50'W). In Florida, measurements were collected from 25 sites at both Cedar Key (CK;

109   29°05'N, 83°01'W) and Charlotte Harbor (CH; 26°04'N, 82°14'W). At each sampling location,

110   abiotic conditions (i.e., temperature, salinity, DO, depth) were recorded, drift macroalgae

111   biomass measured, and seagrass cover/abundance and morphometrics assessed. Because we

112   sampled synoptically using the same methods, we were able to assess both algal biomass and

113   distribution within and among locations across the northern Gulf of Mexico (Table 1; Belgrad et

114   al., 2021; Correia, 2021).

115   **Drift algal abundance assessment.** Within each hexagon, drift algal abundance within

116   seagrass meadows was measured using a flat otter trawl, an epibenthic sled, and 1-m² quadrats.

117   The use of these three sampling techniques provided valuable information about broad and fine

118   scale macroalgal distributions within seagrass habitats. While the trawl covers a larger area than

119   the epibenthic sled, it can quickly become fouled by high biomass of drift algae, making it

120   difficult to standardize trawl lengths. Epibenthic sleds allow for a more standardized comparison

121   between seagrass and macroalgal habitat within a given area. Each sled tow was pulled within

122   seagrass habitats for the same distance allowing for a precise representation of macroalgae

123   within each seagrass bed. Quadrats were used in the same vicinity as the epibenthic sled and

124   trawl to assess seagrass and algal percent cover.

125   A 4.85-m flat otter trawl with a 3.8-cm stretch mesh body and 1.3-cm stretch mesh bag

126   was towed through seagrass beds at an average speed of 3.7–5.6 km h^{-1} for approximately 2

127   minutes, resulting in linear distances of approximately 116.7 m ± 0.12 SE per trawl. Latitude and

128   longitude were recorded at the beginning, midpoint, and end of each trawl to record the trawl

129   path to obtain accurate distance calculations and provide coordinates for sled and quadrat

130   surveys. Macroalgal abundance was determined as the wet weight of algae present in the trawl,

131   measured using a spring scale. Samples of drift algae were taken from the trawl, bagged, frozen,

132   and later identified to genus using a dissecting microscope (Littler & Littler, 2000). Due to

133   variations in the trawl sampling distance, drift algal weight was standardized to trawl area (g·m^{-}

134   ^{2}). At the center of each trawl path, environmental parameters including salinity, temperature

135   (°C), and dissolved oxygen (mg L^{-1}) were measured using a YSI Pro 2030 containing a galvanic

136   DO sensor (Model 2002) immediately following the trawl. Water depth (cm) was also measured

137   at the center of the trawl path.

138   We returned to each site to sample using the epibenthic sled, which consisting of an

139   aluminum frame (0.75-m wide and 0.6-m high), with two skids on either side (0.8 m in length),

140   and a 2-mm stretch mesh net. Sled samples were collected near the midpoint of the trawl path.

141   The sled was towed for 13.3 m at approximately 0.5 m sec^{-1}, covering an area of 10 m^2. Algae

142   from benthic sled samples were bagged, frozen, and transported to the lab where they were later

143   identified to genus and weighed.

144   **Seagrass and algal vegetative sampling (quadrats).** The structural complexity of

145   seagrass meadows and percent cover of drift algae were assessed using a 1-m^2 quadrat divided

146   into 100, 10-cm x 10-cm cells. Twelve quadrats were haphazardly thrown along each trawl: four

147   quadrats at the beginning, middle, and end of each trawl path. The percent cover, shoot count,

148   and canopy height of each seagrass species present in a quadrat was recorded. Seagrass percent

149   cover by species, as well as the cover of drift algae, were measured by counting the number of

150   grid cells within each quadrat that contained a particular vegetation type (0–100 grids quadrat^{-1}).

151   The shoot count was calculated for each seagrass species by counting the number of shoots

152   within a random quadrat grid cell. Canopy height was defined as the mean of three randomly

153   selected canopy height measurements.

154   **Statistical analysis.** SAS© was used for all statistical analyses. When comparing algal

155   biomass across the northern Gulf of Mexico, analyses were performed using General Linear

156   Models (GLM) in SAS© with region (CB, CH, CK, LA, and LM) and sampling period (early or

157   late summer) as fixed factors. The algal biomass within the trawl and epibenthic sled samples

158   were log-transformed to mitigate skewness and achieve normality. Comparisons of algal weight

159   across early and late summer were then performed using procedure GENMOD with link function

160   gamma to analyze continuous variables and $\alpha = 0.05$ was maintained in all *post hoc* testing.

161   To compare algal percent cover to environmental and seagrass variables within the

162   quadrats, a multiple linear regression with backward elimination model selection was performed

163   using procedure REG in SAS©, maintaining $\alpha = 0.05$ during model selection. Variables included

164   in this regression were shoot count, average canopy height, salinity, temperature, dissolved

165   oxygen, and the percent cover of *T. testudinum, H. wrightii,* and *S. filiforme* from the quadrats.

166

167   **RESULTS**

168   **Macroalgal biomass across the northern Gulf of Mexico.** Drift macroalgae were

169   present in both the early and late summer sampling times, reaching biomasses of over 50 g m^{-2} in

170   one site in Charlotte Harbor, FL (Fig. 2). Regional comparisons of trawl samples showed no

171   significant changes in the overall algal biomass in Corpus Christi Bay, TX or Cedar Key, FL

172   between early and late sampling times ($p = 0.49$ and 0.29, respectively; Table 2). Charlotte

173   Harbor, FL had significantly higher biomass of drift macroalgae in the late summer when

174   compared to the early summer months ($p < 0.001$; Table 2). Meanwhile, Laguna Madre, TX and

175   Chandeleur Islands, LA both had significantly less macroalgae in the late summer months

176   compared to the early summer ($p = 0.05$ and 0.01, respectively; Table 2). Macroalgal biomass

177   collected in the epibenthic sled followed a similar pattern among regions and sampling times

178   (Fig. 2, Table 2).

179   **Macroalgal species composition across the northern Gulf of Mexico.** Drift algal

180   community composition varied across region. Macroalgae identified in Laguna Madre, TX

181   consisted of the genera *Agardhiella, Amphiroa, Dictyota, Digenia, Gracilaria, Hypnea,* and

182   *Laurencia*. Corpus Christi Bay, TX had similar algal genera comprised of *Agardhiella*,

183   *Chondria, Dictyota, Digenia, Gracilaria*, and *Laurencia*. Macroalgae found in Chandeleur

184   Islands, LA consisted of *Agardhiella, Chondria, Gracilaria, Laurencia,* and *Spyridia*. Cedar

185   Key, FL consisted of *Acanthophora, Dictyota, Digenia, Gracilaria, Laurencia, Polysiphonia,*

186   and *Ulva*. Charlotte Harbor, FL were mainly comprised of *Acanthophora, Cladophora,*

187   *Gracilaria*, and *Spyridia* genera, and *Hypnea, Laurencia*, and *Ulva* were also present.

188   **Macroalgal percent cover in relation to seagrass and abiotic parameters.** Within the

189   quadrat surveys, *T. testudinum* percent cover and average canopy height both significantly

190   related to drift algal percent cover ($p = 0.03$ and $p < 0.001$, respectively; Fig. 3), whereas the

191   seagrass shoot counts, salinity, temperature, and percent cover of *H. wrightii* and *S. filiforme* did

192   not significantly contribute to the drift algae cover in these areas ($p > 0.05$; Table 3). Drift algal

193   cover increased with increasing *T. testudinum* cover, and algae were most dense in areas with

194   intermediate seagrass canopy heights around 400 mm tall (Fig. 3).

195

196   **DISCUSSION**

197   Drift macroalgae were found throughout the study region in both the early and late

198   summer months, with lower biomasses observed in the late summer months across Laguna

199   Madre, TX, Cedar Key, FL, and Chandeleur Islands, LA, consistent with previous findings (e.g.,

200   Benz, Eiseman & Gallaher, 1979; Virnstein & Carbonara, 1985). Conversely, macroalgal

201   abundance was higher in the late summer in Charlotte Harbor, FL, and did not significantly

202   change from early to late summer in Corpus Christi Bay, TX. The inconsistent patterns observed

203   in these two estuaries may be the result of tidal and freshwater variations that affect delivery

204   rates of nutrients and/or flushing of drift macroalgae. Corpus Christi Bay, TX is an enclosed

205   system with little direct influence from the Gulf of Mexico and long water residence times (Solis

206   & Powell, 1999; Pulich, 2007). Since macroalgae are commonly flushed by tidal currents, the

207   long water residence time and protection from tidal flushing in Corpus Christi Bay may reduce

208   seasonal algae decline observed in other locations. The high percentage of clay and silt within

209   the benthic sediments of Corpus Christi Bay further indicates differences in the local

210   hydrodynamic regime (Shideler, Stelting &McGowen, 1981), with previous studies showing a

211   direct positive link between algae biomass and the amount of silt-clay in the system (Bell & Hall,

212   1997). In contrast, Charlotte Harbor, FL, has experienced an increase in macroalgal blooms in

213   recent years, particularly on the eastern shore (BTT, 2021). The increase in nutrient

214   concentrations from creeks and streams, as well as the limited water circulation on the east side

215   of Charlotte Harbor, appears to be a major driver of high accumulation of drift algae in these

216   locations. Nutrient inputs from leaking septic systems, fertilizer and agricultural runoff, untreated

217   stormwater, ineffective sewage treatment systems, and altered freshwater inflow have been

218   deemed the primary cause of these algae blooms (Lapointe et al., 2016, BTT, 2021). While

219   seasonal fluctuations often show a decline in algal abundances in the late summer months,

220   localized anthropogenic and hydrodynamic differences may cause localized variations.

221   Drift macroalgae were most dense in areas with higher percent cover of *T. testudinum* but

222   with an intermediate canopy height (~400 mm). Perhaps unsurprisingly, algae are more likely to

223   be entrained within denser seagrass beds (Virnstein & Carbonara, 1985; Bell & Hall, 1997), but

224   this varies depending on location and scale. For example, at smaller spatial scales (m), shoot

225   count and blade length were associated with algal density patterns in Tampa Bay, FL (Bell, Hall

226   & Robbins, 1995). However, when this same location was studied at a larger scale (km), seagrass

227   cover explained 57% of the variation in algal cover, suggesting that spatial scale is important

228   when comparing algal to seagrass communities (Bell & Hall, 1997). The relationship between

229   seagrass canopy height and macroalgal density could be related to light-limiting growth

230   restrictions. When macroalgae reach sufficient biomass, they can restrict the light available to

231   seagrasses, decreasing productivity (Hauxwell et al., 2001; Huntington & Boyer, 2008). This

232   may be why areas with the tallest seagrass canopies also have less macroalgae. Because light is

233   less of a limiting factor, seagrass communities can grow at a faster rate when macroalgae are not

234   present. Conversely, entrapment of algae at intermediate canopy heights could be related to the

235   interaction between the algae and flow conditions. Taller canopies, that extend closer to the

236   surface of the water may be influenced by higher water velocities and turbulence, increasing the

237   likelihood of macroalgae dislodgement.

238   This study was limited in its ability to determine species-specific algal effects across

239   varying seagrass characteristics. During sampling, macroalgal species were not separated,

240   identified, and weighed in the field, but were rather weighed collectively and a sample of each

241   species brought back to the lab for later identification. Preliminary site selection was also chosen

242   based on the presence of *T. testudinum* across all site locations, which could have obscured less

243   obvious patterns when comparing algae biomass to *S. filiforme* and *H. wrightii*. Another potential

244   reason for the seagrass species-specific differences are the varying morphological characteristics

245   across *T. testudinum*, *S. filiforme*, and *H. wrightii*, including differences in blade width,

246   thickness, and shoot height (Loria, 2019). Future research should further investigate

247   concentrations of drift macroalgae in *S. filiforme* and *H. wrightii* dominated seagrass meadows to

248   corroborate our nonsignificant findings.

249   Macroalgae proliferate when sufficient light, nutrient availability, and warm temperatures

250   co-occur (EPA, 2013) and can become entrained in seagrass beds in large quantities when

251   seagrasses are dense and water flow is low enough to prevent dislodgement (Bell & Hall, 1997).

252   Although algae can be beneficial by increasing habitat complexity and food resources within

253   seagrass meadows (e.g., Carr, 1994; Kingsford, 1995; Jones, Lawton & Shachak, 1997; Morris &

254   Hall, 2001; Guido et al., 2004), large blooms of macroalgae can displace other benthic habitats

255   and reduce the health of the seagrass meadows (Valiela et al., 1997; Tagliapieta et al., 1998).

256   This study identified several patterns in algal density and composition on a Gulf-wide scale. Data

257   collected during this study also found large abundances of small fish and invertebrates living

258   within these seagrass beds, indicating that the biomass of algae present in the northern Gulf of

259   Mexico do not appear to be detrimental (Belgrad et al., 2021; Correia, 2021). Given the extent

260   that drift algae spatiotemporally vary, research to quantify links between algal genera and faunal

261   community composition could resolve much of the uncertainty surrounding this relationship. As

262   the climate continues to change, macroalgal blooms may become more variable and

263   understanding the interaction between algal-fauna relationships becomes increasingly important.

264   Understanding the role of macroalgae within Gulf of Mexico seagrass beds will allow us to

265   better manage the fisheries and other coastal resources in the future.

266

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424   **Figure 1. Location of the 5 study estuaries (stars) throughout the northern Gulf of Mexico.**

425   Regions include Laguna Madre, TX (LM), Corpus Christi Bay, TX (CB), the northern extent of

426   the Chandeleur Islands (LA), Cedar Key, FL (CK), and Charlotte Harbor, FL (CH). n represents

427   the number of sites that were sampled within each estuary during the early and late summer

428   2018. Map data © 2022 Google.

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431   **Figure 2. Average trawl and sled macroalgal weight in each region.** (A) Average trawl

432   macroalgae weight (g m^{-2}) + SE and (B) benthic sled macroalgae weight (g m^{-2}) + SE sampling

433   across each region during the early (black) and late (grey) summer months. The regions include

434   Laguna Madre, TX (LM, n = 20), Corpus Christi Bay, TX (CB, n = 20), Chandeleur Islands, LA

435   (LA, n = 20), Cedar Key (CK, n = 25), and Charlotte Harbor, FL (CH, n = 25). The asterisk (*)

436   indicates that there is a significant difference between early (May–June 2018) and late (August–

437   September 2018) sampling within that region.

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440   **Figure 3. Vegetation comparisons.** (A) Scatterplots comparing the drift macroalgae percent

441   cover to *T. testudinum* percent cover and (B) average seagrass canopy height from the quadrat

442   dataset. Panel (A) displays drift algae cover increasing with increased *T. testudinum* cover and

443   (B) shows algae percent cover was highest at intermediate canopy heights (~400 mm).

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Figure 1

Location of the 5 study estuaries (stars) throughout the northern Gulf of Mexico.

Regions include Laguna Madre, TX (LM), Corpus Christi Bay, TX (CB), the northern extent of the Chandeleur Islands (LA), Cedar Key, FL (CK), and Charlotte Harbor, FL (CH). n represents the number of sites that were sampled within each estuary during the early and late summer 2018. Map data © 2021 Google.

Figure 1.

Figure 2

Average trawl and sled macroalgal weight in each region.

(A) Average trawl macroalgae weight (g m^{-2}) + SE and (B) benthic sled macroalgae weight (g m^{-2}) + SE sampling across each region during the early (black) and late (grey) summer months. The regions include Laguna Madre, TX (LM, n = 20), Corpus Christi Bay, TX (CB, n = 20), Chandeleur Islands, LA (LA, n = 20), Cedar Key (CK, n = 25), and Charlotte Harbor, FL (CH, n = 25). The asterisk (*) indicates that there is a significant difference between early (May–June 2018) and late (August–September 2018) sampling within that region.
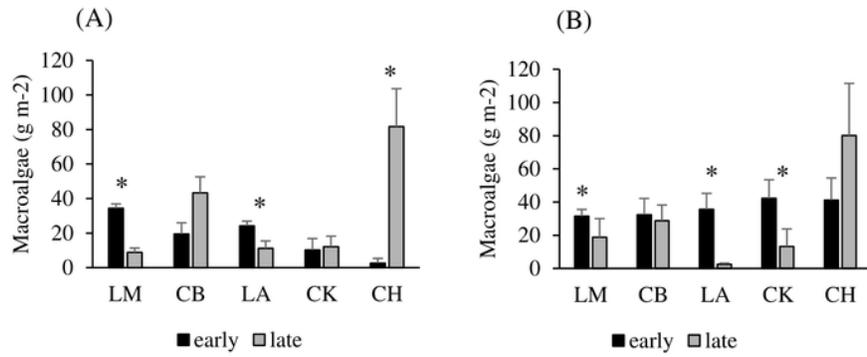
Figure 2.

Figure 3

Vegetation comparisons .

(A) Scatterplots comparing the drift macroalgae percent cover to *T. testudinum* percent cover and (B) average seagrass canopy height from the quadrat dataset. Panel (A) displays drift algae cover increasing with increased *T. testudinum* cover and (B) shows algae percent cover was highest at intermediate canopy heights (~400 mm).
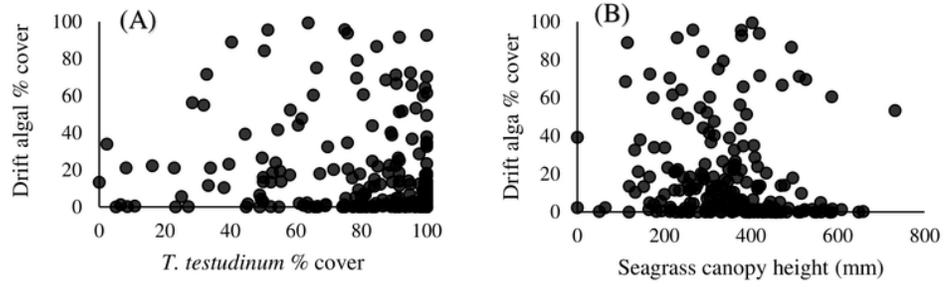
Figure 3.

Abiotic variables from each region.

Abiotic variables (Mean ± SE) measured during the early (May–June) and late
(August–September) summer months 2018.

1 **Table 1. Abiotic variables from each region**. Abiotic variables (Mean ± SE) measured during the early (May–June) and late

2 (August–September) summer months 2018.

Abiotic parameter	Time	Laguna Madre, TX	Corpus Christi Bay, TX	Chandeleur Islands, LA	Cedar Key, FL	Charlotte Harbor, FL
Salinity (ppt)	Early	36.8 ± 0.06	33.9 ± 0.16	16.2 ± 0.49	28.3 ± 0.59	22.6 ± 1.22
	Late	37.1 ± 0.13	34.9 ± 0.14	27.6 ± 0.27	25.1 ± 0.70	19.9 ± 0.96
Temperature (^{o}C)	Early	28.1 ± 0.20	27.0 ± 0.15	29.8 ± 0.55	29.7 ± 0.16	30.1 ± 0.22
	Late	29.7 ± 0.38	30.2 ± 0.20	30.8 ± 0.25	30.1 ± 0.52	31.8 ± 0.29
Dissolved Oxygen (mg L^{-1})	Early	8.4 ± 0.43	6.2 ± 0.32	9.5 ± 0.58	7.6 ± 0.30	6.3 ± 0.24
	Late	6.1 ± 0.47	11.8 ± 0.46	8.9 ± 0.38	6.8 ± 0.35	7.7 ± 1.50

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Comparisons of trawl and sled algal weight across region and sampling time .

Multiple and general linear regression models for macroalgal biomass across region during early (May–June 2018) and late (August–September 2018) summer sampling. Each location was then separated and analyzed individually using generalized estimating equations with sample period as the fixed factor.

1 **Table 2. Comparisons of trawl and sled algal weight across region and sampling time.**

2 Multiple and general linear regression models for macroalgal biomass across region during early

3 (May–June 2018) and late (August–September 2018) summer sampling. Each location was then

4 separated and analyzed individually using generalized estimating equations with sample period

5 as the fixed factor.

	SS	df	F ratio	Prob > F
Trawl algae weight				
Region (LM, CB, LA, CK, CH)	1937715.58	4	6.87	< 0.0001
Sample period (early, late)	142138.139	1	2.52	0.1137
Region*sample period	2547045.91	4	9.03	< 0.0001
Sled algae weight				
Region (LM, CB, LA, CK, CH)	92.383	4	3.38	0.0104
Sample period (early, late)	197.888	1	28.99	< 0.0001
Region*sample period	65.214	4	2.39	0.0521

Algal biomass comparisons across sampling time	Mean estimate	Chi-Square	Pr > ChiSq
Trawl			
LM (early v late)	-479.345	6.51	0.0107
CB (early v late)	3773.820	0.49	0.4859
LA (early v late)	-527.146	3.86	0.0493
CK (early v late)	-1150.98	1.11	0.2921
CH (early v late)	299.2744	14.22	0.0002
Sled			
LM (early v late)	-26.07	4.97	0.7533
CB (early v late)	238.02	0.10	0.3665
LA (early v late)	-13.46	18.31	0.2010
CK (early v late)	-28.12	1.64	< 0.0001
CH (early v late)	88.59	0.82	0.0258

Backward regression to compare algae to abiotic and seagrass characteristic s.

Multiple linear regression with backward elimination model selection results comparing algal density across abiotic and seagrass variables. The variables included in the model, following selection, include *T. testudinum* density and the average canopy height. Variables that were determined to be insignificant to the model during the selection process were average shoot count, salinity, water temperature, *H. wrightii* density, *S. filiforme* density, and dissolved oxygen.

1 **Table 3. Backward regression to compare algae to abiotic and seagrass characteristics.**

2 Multiple linear regression with backward elimination model selection results comparing algal

3 density across abiotic and seagrass variables. The variables included in the model, following

4 selection, include *T. testudinum* density and the average canopy height. Variables that were

5 determined to be insignificant to the model during the selection process were average shoot

6 count, salinity, water temperature, *H. wrightii* density, *S. filiforme* density, and dissolved oxygen.

	SS	*df*	*F* ratio	Prob > *F*
Final model	4.87	2	7.10	0.0010
Variables in final model				
T. testudinum density	1.73		5.04	0.0258
Average canopy height	4.50		13.11	0.0004

	Model R^2	*C(p)*	*F* ratio	Prob > *F*
Variables removed from the model				
Average total shoot count	0.0760	7.02	0.02	0.8964
Salinity	0.0749	5.26	0.25	0.6182
Temperature	0.0737	3.53	0.27	0.6028
H. wrightii density	0.0702	2.36	0.83	0.3626
Dissolved oxygen	0.0652	1.52	1.18	0.2791
S. filiforme density	0.0604	0.65	1.14	0.2862