

1 **Title:**

2 Widespread seagrass die-off has no legacy effect on basal resource use of seagrass food webs in
3 Florida Bay, USA

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37 **Keywords:** Habitat loss, stable isotopes, epiphytes, mixing models, foundation species
38

39 **Abstract**

40 Macrophyte foundation species provide both habitat structure and primary production, and
41 loss of these habitats can alter species interactions and lead to changes energy flow in food webs.
42 Formerly extensive seagrass meadows in Florida Bay recently experienced a widespread loss of
43 seagrass habitat due to a *Thalassia testudinum* mass mortality event in 2015 associated with
44 prolonged hypersalinity and bottom-water anoxia. Using stable isotope analysis paired with
45 Bayesian mixing models, we investigated the basal resource use of seven species of seagrass-
46 associated consumers across Florida Bay in areas affected by the 2015 seagrass die-off. Three
47 years after the die-off, basal resource use did not differ for species collected inside and outside
48 the die-off affected areas. Instead, consumers showed seasonal patterns in basal resource use
49 with seagrass the most important in the wet season (58%), while epiphytes were the most
50 important in the dry season (44%). Additionally, intraspecific spatial variability in resource use
51 was lower in the wet season compared to the dry season. We were unable to detect a legacy
52 effect of a major disturbance on the basal resource use of the most common seagrass-associated
53 consumers in Florida Bay.

54 **Introduction**

55 Disturbance events can drastically alter habitat composition (Carlson *et al.*, 2010). Both
56 press (e.g. sea level rise) and pulse (e.g. drought) disturbances from both natural and
57 anthropogenic sources are becoming more common, especially in coastal ecosystems, leading to
58 drastic alterations in coastal seascapes (Carlson *et al.*, 2010; Santos *et al.*, 2011). For example,
59 the combined effects of long-term stressors such as sea-level rise and decreased sediment supply
60 via river flow modification, as well as pulsed disturbances such as hurricanes and tropical
61 storms, have led to widespread loss of coastal habitats in Louisiana (Couvillion *et al.*, 2016).
62 This is true in other coastal ecosystems as well, such as seagrass beds, which have seen habitat
63 modification through altered light and salinity regimes, as well as by eutrophication from coastal
64 development, human population growth, and water management practices (Waycott *et al.*, 2009;
65 Santos *et al.*, 2011; Hall *et al.*, 2016). Importantly, coastal habitats support many different
66 species, and habitat loss from disturbance events has led to shifts and losses in consumer biomass
67 (Boström *et al.*, 2011; Yeager *et al.*, 2016; Santos *et al.*, 2018).

68 Macrophyte foundation species for many coastal habitats, serving not only as structural
69 habitat, but also contributing directly or indirectly as primary production sources to the food web
70 (Jones *et al.*, 1994; Boström *et al.*, 2011; James *et al.*, 2019). Any loss in production of
71 foundation species (e.g. seagrass) leads to a loss in secondary production unless consumers
72 compensate by altering basal resource use (Deegan and Garritt, 1997; Smith *et al.*, 2008; Lesser
73 *et al.*, 2020). For example, seagrass omnivores increase variation in basal resource use to
74 compensate for lower production (Lesser *et al.*, 2020; Santos *et al.*, 2022). Habitat loss can also
75 alter species interactions by increasing competition for space and resources and changing prey
76 capture efficiency (Hovel and Lipcius, 2001; Fahrig, 2003; Canion and Heck, 2009). Changes in

77 species interactions can alter energy flow and lead to changes in trophic structure in the food web
78 (Hovel and Lipcius, 2001; Robinson *et al.*, 2015). For example, the loss of seagrass habitat
79 through the invasion of macroalgae leads to increased competition due to altered basal resources
80 compared to food webs in continuous seagrass meadows (Deudero *et al.*, 2014). Therefore,
81 disturbance events that result in losses of foundation species have the potential to significantly
82 alter coastal food webs, and there is a critical need to document and understand the connection
83 between disturbance of macrophyte foundation species and changes to the food webs they
84 support.

85 Florida Bay is a shallow estuary, located between the Florida Keys and the Everglades, and
86 is home to vast seagrass beds comprised mainly of *Thalassia testudinum* (Fourqurean and
87 Robblee, 1999). These meadows are vital habitat for many ecologically and economically
88 important species and support a lucrative recreational fishery (Kelble *et al.*, 2013; Brown *et al.*,
89 2018; Stainback *et al.*, 2019). In the Everglades watershed, hydrology and water management
90 have been engineered to allow for urban development, to control flooding, and to support
91 agriculture, resulting in reduced freshwater flow and seasonal hypersalinity (due to seasonality of
92 freshwater flows) in portions of Florida Bay (Marshall *et al.*, 2020). When combined with
93 increasing temperatures, prolonged hypersalinity can lead to hypoxic stress, sulfide intrusion,
94 and mortality of seagrasses (Kelble *et al.*, 2007; Koch *et al.*, 2007). The first well-documented
95 example of widespread seagrass die-off in Florida Bay began in 1987 and continued for nearly
96 four years (Robblee *et al.*, 1991; Fourqurean and Robblee, 1999). Seagrasses had recovered by
97 2010 (Hall *et al.*, 2021), but in 2015, a second mass mortality event, caused by the same set of
98 stressors, occurred in the same location, resulting in comparable losses of *T. testudinum* (Hall *et*
99 *al.*, 2016).

100 Although benthic macrophyte communities in Florida Bay have high ecological and
101 economic importance and have received considerable attention (Zieman *et al.*, 1989; Fourqurean
102 *et al.*, 2001), not much is known about the food webs they support (Chasar *et al.*, 2005; Rezek *et*
103 *al.*, 2020). The two goals of this study were to: 1) characterize the energy flow of seagrass
104 consumers in Florida Bay and 2) determine if energy flow differed in areas that were and were
105 not affected by the 2015 seagrass die-off. Seagrass production is seasonal within Florida Bay
106 (Fourqurean *et al.*, 2001), but we hypothesized that seagrass food webs would be supported
107 mainly from epiphyte production regardless of season (Kitting *et al.*, 1984; Moncreiff and
108 Sullivan, 2001). Additionally we hypothesized that consumers collected in areas affected by the
109 2015 seagrass die-off would have shifted resource use away from seagrass-associated basal
110 resources because of spatial differences in resource availability (Deegan and Garritt, 1997; Olin
111 *et al.*, 2012), based on the spatial clustering of the die-off impacts and the degree of recovery
112 thereafter. Seven consumer species were collected from stations inside and outside the die-off
113 boundary throughout Florida Bay, and stable isotope analysis paired with Bayesian mixing
114 models were run to determine resource use of dominant consumer species across sites (Stock *et*
115 *al.*, 2018).

116 **Methods**

117 Consumer species were collected for stable isotope analysis from 26 sites within the
118 seagrass die-off boundary and 15 sites located in unaffected areas at NOAA long-term Juvenile
119 Sportfish monitoring sites (Figure 1; Kelble *et al.*, 2013; Kearney *et al.*, 2015). The die-off
120 boundary was based on in situ observations described in Hall *et al.* (2016). Areas within the
121 seagrass die-off boundary experienced heavy loss of *Thalassia testudinum* resulting in low
122 seagrass cover compared to areas outside of the boundary (Furman *et al.*, 2021). To represent

123 consumers species abundant in seagrass habitats, bay anchovy (*Anchoa mitchilli*), Silver Jenny
124 mojarra (*Eucinostomus gula*), pigfish (*Orthopristis chrysoptera*), pinfish (*Lagodon rhomboides*),
125 pink shrimp (*Farfantepenaeus duorarum*), rainwater killifish (*Lucania parva*), and silver perch
126 (*Bairdiella chrysoura*) were collected via otter trawl as described in Kelble *et al.* (2013) during
127 the wet (September 2018) and dry seasons (May 2019) both inside and outside the seagrass die-
128 off boundary. Based on salinity trends, the central portion of Florida Bay likely received more
129 freshwater inflow during the wet season of 2018 than average, while dry season 2019 was similar
130 to historic trends (Figure S1). Species collected represent various feeding guilds (e.g. benthic
131 omnivore, pelagic omnivore) and make up the majority of the biomass of lower trophic level
132 consumers in Florida Bay (Sogard *et al.*, 1989; Chasar *et al.*, 2005; Flaherty *et al.*, 2013). Five
133 individuals were pooled for each species collected in a trawl at each sampling location to
134 generate one composite sample for stable isotope analysis (e.g. n = 1 pooled sample/species/site;
135 Table 1, S1, Figure S2). Pooling samples is common for isotope samples to control for between-
136 individual variation, as well as ensure that enough tissue is collected for analysis (Phillips *et al.*,
137 2014; Nelson *et al.*, 2019). Pooling individuals allowed for a better representation of isotope
138 values of species at a given site to compare spatial trends in resource use. Whole individuals
139 were dried at 50 °C for 48 hours and ground together to make a composite sample that was sent
140 for stable isotope analysis.

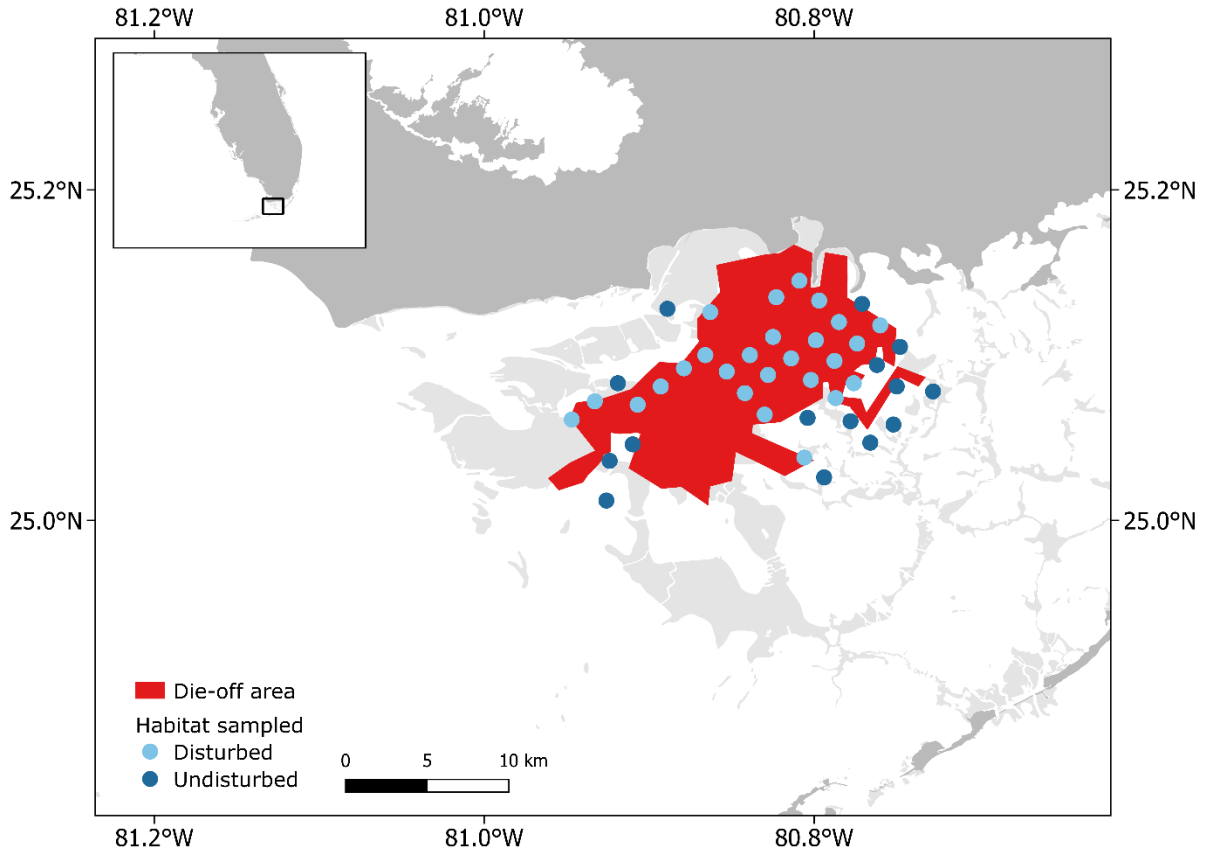
141 Basal resources (i.e. primary producers) were collected at each of the trawl sampling
142 locations in both the wet and dry seasons by hand, with the addition of mangrove leaves
143 collected by hand at mangrove islands throughout the sampling area. Seagrass (*Halodule wrightii*
144 and *Thalassia testudinum*) were rinsed with deionized (DI) water and scraped with the back of a
145 scalpel to remove sediment and epiphytes. Seagrass leaves were then acid washed with 10% HCl

146 and rinsed with DI water, before being dried 50 °C for 48 hours and ground to be sent for stable
147 isotope analysis. Mangrove leaves (*Rhizophora mangle*) were rinsed with DI water, dried 50 °C
148 for 48 hours, ground, and sent for stable isotope analysis. Macroalgal thalli (e.g. *Halimeda* spp.,
149 *Caulerpa* spp., *Penicillus* spp., *Batophora oerstedii*) were acid washed with 10% HCl, and rinsed
150 with DI water before being dried 50 °C for 48 hours, ground, and sent for stable isotope analysis.
151 Each macroalgal species was sent for isotope analysis separately. Particulate organic matter
152 (POM) was not collected in this study and not included as a potential basal resource. However,
153 POM stable isotope values from other studies collected in Florida Bay overlap with those of the
154 macroalgae collected for carbon, nitrogen, and sulfur, and therefore, not isotopically
155 distinguishable from the macroalgal sources collected in this study (Figure 2; Chasar *et al.*, 2005;
156 Eggenberger *et al.*, 2019). Gastropod grazers that specialize in epiphyte consumption (*Turbo*
157 *castanea*) were collected by hand at the same time as the other primary producers (Frankovich
158 and Zieman, 2005). Grazers were removed from their shells, rinsed with DI water, and dried at
159 50 °C for 48 hours before being ground. Each species of primary producer collected at a given
160 site was pooled so that there was one replicate per site. Each potential basal resource was
161 collected from a minimum of seven sampling locations. All samples were shipped to the
162 Washington State University Stable Isotope Core Facility for analysis of C, N, and S stable
163 isotopes.

164 Bayesian mixing models were run in R version 4.0.5 (R Core Team, 2021) using the
165 package MixSIAR (v 3.1.10, Stock et al. 2018) to determine the relative basal resource
166 contributions to each consumer species. Models were run for each season with consumer species
167 as a fixed factor and sampling location as a random factor nested within species. Macroalgae
168 species could not be distinguished isotopically from one another, and thus were combined for

169 statistical analyses (hereafter, referred to as algae). Grazers were used as proxies for epiphyte
170 stable isotopes values by trophically-correcting isotope values one trophic level (Frankovich and
171 Zieman, 2005). Using trophically-corrected primary consumers that specialize on a specific
172 primary producer is a common way to obtain stable isotope values of that primary producer,
173 especially when the primary producer is difficult to collect enough organic material for stable
174 isotope analysis (Fry, 2006; Layman *et al.*, 2012; Phillips *et al.*, 2014). Basal resources were
175 averaged across space and season to serve as the source values for consumers in the mixing
176 models. Basal resources were averaged across time and space because, due to tissue turnover
177 times, consumers would likely be assimilating basal resources that from across the year (not just
178 the seasonal time point when sampled), and by averaging our two time points, we would best
179 represent the potential source isotope values. Corrections were made for the elemental
180 concentration in each source to account for differences between sources, and multiplicative error
181 (residual error x process error) was used (Stock and Semmens, 2016). The trophic enrichment for
182 each element was $C = 2.5 \pm 0.3$ (mean \pm SD), $N = 7.25 \pm 1$, and $S = 1 \pm 0.2$. We chose these
183 values because this enrichment enclosed the consumer isotope values within the mixing polygon
184 and was ecologically relevant to the consumers studied (Figure 2; Phillips *et al.*, 2014; Nelson *et*
185 *al.*, 2015, 2019). Each model was run with a Markov chain Monte Carlo algorithm that consisted
186 of three chains, chain length of 1,000,000, burn-in of 500,000, and thinning interval of 500 to
187 ensure model convergence.

188



189

190 **Figure 1.** Seagrass die-off area (red) and the trawl sampling locations (blue) in Florida Bay. Die-

191 off footprint is based on initial observations from the beginning of the die-off (Hall *et al.* 2016).

192 There were 26 sites within the die-off boundary (Disturbed = light blue) and 15 sites outside of

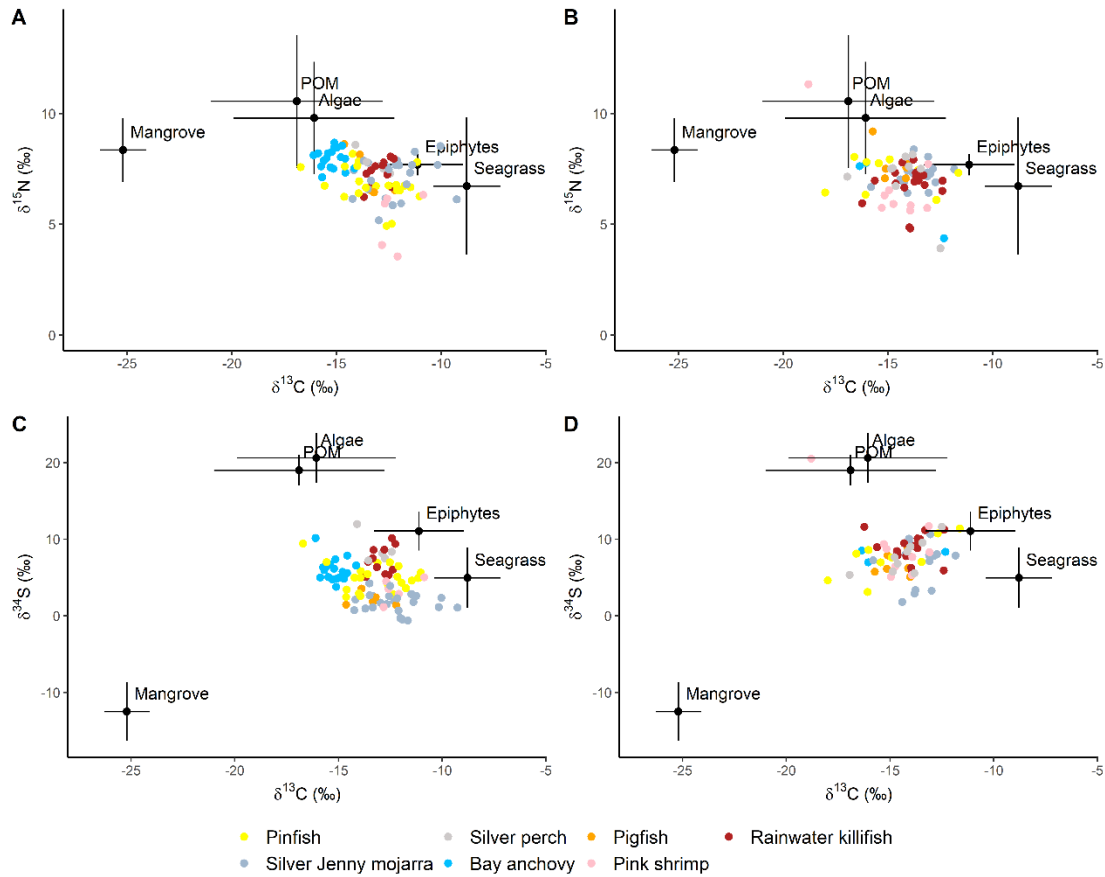
193 the die-off boundary (Undisturbed = dark blue).

194

195 **Table 1.** Isotope values (mean \pm SD) for each species and season collected inside (disturbed) and
 196 outside (undisturbed) the seagrass die-off area in Florida Bay.
 197

Species	Season	Habitat	$\delta^{13}\text{C}$	$\delta^{15}\text{N}$	$\delta^{34}\text{S}$	# of sites
Bay anchovy	Wet	Disturbed	-15.4 \pm 0.5	7.9 \pm 0.5	6.1 \pm 1.7	12
	Wet	Undisturbed	-14.8 \pm 0.5	8.2 \pm 0.4	5.8 \pm 0.8	6
	Dry	Disturbed	-14.9 \pm 2.2	6.6 \pm 1.9	7.9 \pm 0.8	3
Mojarra	Wet	Disturbed	-12.9 \pm 0.7	7.1 \pm 0.8	1.8 \pm 1.4	12
	Wet	Undisturbed	-11.6 \pm 1.6	7.1 \pm 1.1	1.8 \pm 1.0	10
	Dry	Disturbed	-13.7 \pm 0.8	7.2 \pm 0.6	7.3 \pm 2.7	14
Pigfish	Dry	Undisturbed	-12.4 \pm 0.8	7.4 \pm 0.1	5.6 \pm 3.3	2
	Wet	Disturbed	-13.5 \pm 1.0	7.5 \pm 1.1	2.1 \pm 1.0	4
	Wet	Undisturbed	-13.2	6.4	2.4	1
Pinfish	Dry	Disturbed	-14.3 \pm 0.5	7.4 \pm 0.3	6.8 \pm 1.5	5
	Dry	Undisturbed	-15.4 \pm 0.4	8.3 \pm 1.2	6.8 \pm 1.5	2
	Wet	Disturbed	-13.3 \pm 1.3	6.8 \pm 0.8	4.9 \pm 1.7	10
Pink shrimp	Wet	Undisturbed	-12.5 \pm 1.2	6.4 \pm 0.6	4.7 \pm 1.3	8
	Dry	Disturbed	-14.2 \pm 1.6	7.4 \pm 0.6	7.6 \pm 2.8	6
	Dry	Undisturbed	-15.8 \pm 2.3	7.1 \pm 1.0	8.0 \pm 2.5	4
Rainwater killifish	Wet	Disturbed	-12.2 \pm 0.8	5.2 \pm 1.3	3.4 \pm 1.5	5
	Dry	Disturbed	-14.2 \pm 0.9	6.1 \pm 0.7	8.2 \pm 2.1	7
	Dry	Undisturbed	-15	6.5	5.1	1
Silver perch	Wet	Disturbed	-12.8 \pm 0.5	7.4 \pm 0.4	7.2 \pm 1.8	7
	Wet	Undisturbed	-13.1 \pm 0.7	7.0 \pm 0.9	7.7 \pm 2.3	3
	Dry	Disturbed	-14.2 \pm 1.0	6.7 \pm 0.9	8.7 \pm 1.8	14
Silver perch	Dry	Undisturbed	-13.1 \pm 0.6	7.0 \pm 0.2	10.5 \pm 1.1	5
	Wet	Disturbed	-12.9 \pm 0.6	7.5 \pm 0.3	7.7 \pm 0.6	3
	Wet	Undisturbed	-13.5 \pm 0.9	8.1 \pm 0.7	10.1 \pm 2.7	2
Silver perch	Dry	Disturbed	-14.3 \pm 1.3	7.1 \pm 1.4	8.0 \pm 2.1	8
	Dry	Undisturbed	-14.9	7.5	5.8	1

198



199

200 **Figure 2.** Stable isotope biplots for carbon and nitrogen in the A) wet and B) dry season and
 201 carbon and sulfur in the C) wet and D) dry season collected in Florida Bay. Overlaid source
 202 values represent the trophic enrichment corrected mean \pm SD used in the Bayesian mixing
 203 models. POM source values were based on (Chasar *et al.*, 2005; Eggenberger *et al.*, 2019) but
 204 were not included as a potential source in the mixing models since they were indistinguishable
 205 from macroalgae.

206

207 **Results**

208 The four basal resource groups were isotopically distinct (Figure 2). Seagrass stable
209 isotope values had a mean $\delta^{13}\text{C} = -11.1 \pm 1.7$ (mean \pm SD), $\delta^{15}\text{N} = -0.3 \pm 3.1$, and $\delta^{34}\text{S} = 4.1 \pm$
210 4.2. Algae had a mean $\delta^{13}\text{C} = -16.2 \pm 3.4$, $\delta^{15}\text{N} = 1.7 \pm 3.0$, and $\delta^{34}\text{S} = 19.2 \pm 3.0$. Epiphytes had
211 carbon and sulfur isotope values that were between the seagrass and algal species, $\delta^{13}\text{C} = -13.1$
212 ± 2.4 and $\delta^{34}\text{S} = 10.3 \pm 2.4$, and the mean $\delta^{15}\text{N} = 0.5 \pm 0.5$. Mangrove stable isotope values were
213 the most negative for carbon and sulfur of all basal resources, $\delta^{13}\text{C} = -27.7 \pm 1.1$ and $\delta^{34}\text{S} = -$
214 13.5 ± 3.8 , and the mean $\delta^{15}\text{N} = 1.1 \pm 1.5$ (Table S2, Figure 2). Mean consumer isotope values
215 varied between -15.2 (bay anchovy) and -12.2 (pink shrimp) for $\delta^{13}\text{C}$ (Table 1, Figure 2). Pink
216 shrimp had the lowest $\delta^{15}\text{N}$ values, while bay anchovy collected in the wet season had the
217 highest $\delta^{15}\text{N}$ values. For $\delta^{34}\text{S}$, consumer species generally had lower values in the wet season,
218 except silver perch which had the highest $\delta^{34}\text{S}$ values in the wet season (Table 1, Figure 2).

219 We did not find any differences in resource use between consumers collected inside and
220 outside the die-off area (Table 2, Figure 3). In the wet season, mean basal resource use showed
221 low spatial variability across the bay, with all sites relying mainly on seagrass and low
222 intraspecies spatial variation in basal resource use (Table 2, S1, Figure 4). In the dry season,
223 there was more variation in both the mean and intraspecies resource use as compared to the wet
224 season (Table 2, S1, Figure 4).

225 Across all consumer species, seagrass was the most used resource in the wet season, but
226 this switched in the dry season, with epiphytes becoming the most used resource (Table 3). Mean
227 resource use of seagrass was 58% in the wet season and 30% in the dry season, while epiphyte
228 use was 17% in the wet season compared 44% in the dry season. Resource use was similar for

229 mangrove in both seasons (wet = 21%, dry = 23%). Algae use was low in both seasons (wet =
230 2%, dry = 5%; Table 2).

231 Each species displayed differential basal resource use in the amount of each resource, but
232 there were trends in seasonal variation of resource use (Table 3). Mojarra, pigfish, pinfish, and
233 rainwater killifish displayed similar seasonal patterns with seagrass being the most used resource
234 in the wet season, while epiphytes were the most used resource in the dry season (Table 3). Bay
235 anchovy displayed low seasonal variation and more equally distributed across epiphytes,
236 seagrass, and mangroves in both seasons (Table 3). Pink shrimp used seagrass most in both
237 seasons but was the only species to use > 10 % algae in any season (15% in dry season). The
238 most used resource for silver perch was epiphytes in both seasons, but unlike any other species,
239 epiphyte use was higher in the wet season than the dry season (Table 3).

240

241 **Table 2.** Mixing model results of source contribution (mean \pm SD) for each species and season
 242 collected inside (disturbed) and outside (undisturbed) the seagrass die-off area.

Species	Season	Habitat	Algae	Epiphytes	Seagrass	Mangrove	# of sites
Bay anchovy	Wet	Disturbed	0.02 \pm 0.01	0.30 \pm 0.01	0.35 \pm 0.01	0.33 \pm 0.01	12
	Wet	Undisturbed	0.02 \pm 0.01	0.30 \pm 0.01	0.35 \pm 0.01	0.33 \pm 0.01	6
	Dry	Disturbed	0.05 \pm 0.01	0.33 \pm 0.07	0.36 \pm 0.12	0.27 \pm 0.08	3
Mojarra	Wet	Disturbed	0.01 \pm 0.01	0.04 \pm 0.01	0.76 \pm 0.01	0.19 \pm 0.01	12
	Wet	Undisturbed	0.01 \pm 0.01	0.04 \pm 0.01	0.76 \pm 0.01	0.19 \pm 0.01	10
	Dry	Disturbed	0.01 \pm 0.01	0.58 \pm 0.09	0.23 \pm 0.07	0.18 \pm 0.05	14
	Dry	Undisturbed	0.01 \pm 0.01	0.59 \pm 0.07	0.22 \pm 0.01	0.17 \pm 0.07	2
Pigfish	Wet	Disturbed	0.01 \pm 0.01	0.08 \pm 0.01	0.66 \pm 0.01	0.25 \pm 0.01	4
	Wet	Undisturbed	0.01	0.08	0.66	0.25	1
	Dry	Disturbed	0.03 \pm 0.01	0.53 \pm 0.05	0.21 \pm 0.02	0.24 \pm 0.04	5
	Dry	Undisturbed	0.02 \pm 0.01	0.52 \pm 0.07	0.22 \pm 0.03	0.24 \pm 0.04	2
Pinfish	Wet	Disturbed	0.03 \pm 0.01	0.10 \pm 0.01	0.66 \pm 0.01	0.21 \pm 0.01	10
	Wet	Undisturbed	0.03 \pm 0.01	0.10 \pm 0.01	0.66 \pm 0.01	0.21 \pm 0.01	8
	Dry	Disturbed	0.02 \pm 0.01	0.60 \pm 0.11	0.14 \pm 0.07	0.23 \pm 0.05	6
	Dry	Undisturbed	0.02 \pm 0.01	0.55 \pm 0.10	0.11 \pm 0.01	0.32 \pm 0.09	4
Pink shrimp	Wet	Disturbed	0.01 \pm 0.01	0.05 \pm 0.01	0.80 \pm 0.01	0.15 \pm 0.01	5
	Dry	Disturbed	0.11 \pm 0.02	0.13 \pm 0.03	0.54 \pm 0.06	0.23 \pm 0.05	7
	Dry	Undisturbed	0.09	0.12	0.51	0.28	1
Rainwater killifish	Wet	Disturbed	0.02 \pm 0.01	0.33 \pm 0.01	0.46 \pm 0.01	0.19 \pm 0.01	7
	Wet	Undisturbed	0.02 \pm 0.01	0.33 \pm 0.01	0.46 \pm 0.01	0.19 \pm 0.01	3
	Dry	Disturbed	0.07 \pm 0.02	0.38 \pm 0.08	0.36 \pm 0.08	0.20 \pm 0.04	14
	Dry	Undisturbed	0.08 \pm 0.01	0.43 \pm 0.01	0.34 \pm 0.01	0.16 \pm 0.02	5
Silver perch	Wet	Disturbed	0.02 \pm 0.01	0.54 \pm 0.01	0.26 \pm 0.01	0.18 \pm 0.01	3
	Wet	Undisturbed	0.02 \pm 0.01	0.54 \pm 0.01	0.26 \pm 0.01	0.18 \pm 0.01	2
	Dry	Disturbed	0.04 \pm 0.01	0.40 \pm 0.10	0.34 \pm 0.11	0.23 \pm 0.06	8
	Dry	Undisturbed	0.04	0.46	0.26	0.24	1

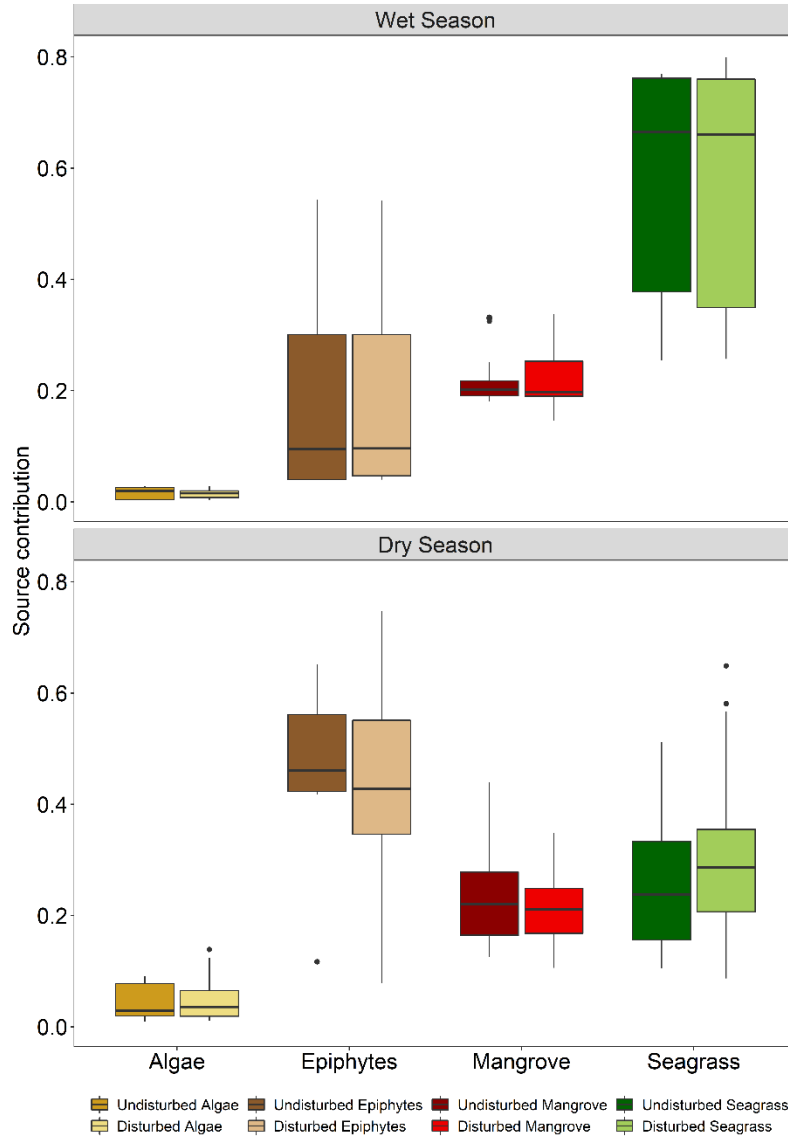
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244

245 **Table 3.** Mixing model results of source contribution (mean \pm SD) for all species collected in the
 246 wet and dry season in Florida Bay.

Species	Season	Algae	Epiphytes	Seagrass	Mangrove	# of sites
Bay anchovy	Wet	0.02 \pm 0.01	0.30 \pm 0.01	0.35 \pm 0.01	0.33 \pm 0.01	18
	Dry	0.05 \pm 0.01	0.33 \pm 0.07	0.36 \pm 0.12	0.27 \pm 0.08	3
Mojarra	Wet	0.01 \pm 0.01	0.04 \pm 0.01	0.76 \pm 0.01	0.19 \pm 0.01	22
	Dry	0.01 \pm 0.01	0.58 \pm 0.08	0.23 \pm 0.06	0.18 \pm 0.05	16
Pigfish	Wet	0.01 \pm 0.01	0.08 \pm 0.01	0.66 \pm 0.01	0.25 \pm 0.01	5
	Dry	0.02 \pm 0.01	0.52 \pm 0.05	0.21 \pm 0.02	0.24 \pm 0.04	7
Pinfish	Wet	0.03 \pm 0.01	0.10 \pm 0.01	0.66 \pm 0.01	0.21 \pm 0.01	18
	Dry	0.02 \pm 0.01	0.58 \pm 0.10	0.13 \pm 0.06	0.26 \pm 0.08	10
Pink shrimp	Wet	0.01 \pm 0.01	0.05 \pm 0.01	0.80 \pm 0.01	0.15 \pm 0.01	5
	Dry	0.11 \pm 0.02	0.12 \pm 0.03	0.54 \pm 0.06	0.23 \pm 0.05	8
Rainwater killifish	Wet	0.02 \pm 0.01	0.33 \pm 0.01	0.46 \pm 0.01	0.19 \pm 0.01	10
	Dry	0.07 \pm 0.01	0.39 \pm 0.07	0.35 \pm 0.07	0.19 \pm 0.04	19
Silver perch	Wet	0.02 \pm 0.01	0.54 \pm 0.01	0.26 \pm 0.01	0.18 \pm 0.01	5
	Dry	0.04 \pm 0.01	0.40 \pm 0.10	0.33 \pm 0.11	0.23 \pm 0.05	9

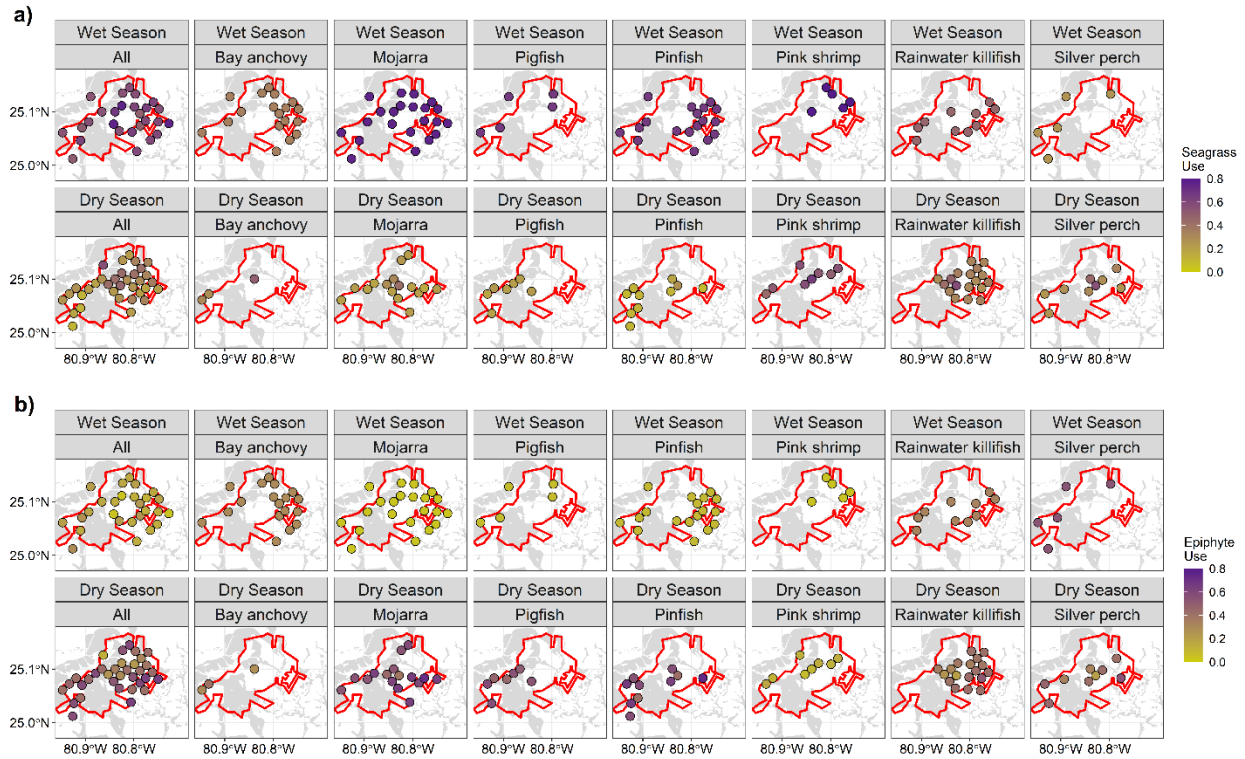
247



248

249 **Figure 3.** Source contribution across all seven consumer species each season inside (disturbed)

250 and outside (undisturbed) the seagrass die-off area.



251
 252 **Figure 4.** Spatial distribution of a) seagrass and b) epiphyte resource use in the wet (top row) and
 253 dry (bottom row) seasons. Each dot represents the resource use for seagrass or epiphytes as
 254 estimated from the Bayesian mixing models for that species (or if all the average of all species
 255 collected at that site) in a given season at that sampling location. The red outline represents the
 256 seagrass die-off footprint. All = the average of all species collected at a given sampling location.
 257 Mojarra = Silver Jenny mojarra

258 **Discussion**

259 The 2015 seagrass die-off in Florida Bay led to the loss of key structural habitat for many
 260 consumer species (Hall *et al.*, 2016). The seagrass loss in Florida Bay not only reduced critical
 261 habitat for consumer species but also the main energy source, either directly or indirectly (i.e.
 262 epiphyte pathway) for seagrass-associated consumers. Contrary to our hypothesis, we were
 263 unable to detect a significant, sustained shift in basal resource use between sampling locations
 264 that were affected and unaffected by the seagrass die-off three years post disturbance (Table 2,

265 Figure 3). Chasar *et al.* (2005) also found no shift in stable isotope values for consumers before
266 and after the 1987-1991 seagrass die-off in Florida Bay. Therefore, there may not be lasting or
267 legacy effects of seagrass die-off on the basal resource use of mesotrophic seagrass consumers in
268 Florida Bay. Our results confirm that the food web in Florida Bay is driven by benthic
269 production, as has been shown for seagrass food webs generally (Kitting *et al.*, 1984; Moncreiff
270 and Sullivan, 2001; Chasar *et al.*, 2005). We found a consistent pattern of seasonal variation in
271 basal resource use, with seagrass being the most important resource in the wet season and
272 epiphytes being the most important resource in the dry season. Mangroves were a consistent
273 secondary source with similar use in both seasons (~20%), but algal resource use, whether
274 benthic or pelagic, was low for all consumer species in both seasons. Overall, there was a lack of
275 spatial structuring and minimal spatial variation in basal resource use across the bay, especially
276 in the wet season.

277 Many estuarine consumers are opportunistic feeders shifting resource use based on
278 availability (Deegan and Garritt, 1997; Moncreiff and Sullivan, 2001; Olin *et al.*, 2012). Both
279 widespread seagrass die-offs in Florida Bay have likely led to a decline in epiphyte and seagrass
280 production, but consumers do not appear to shift resource use away from these basal resources
281 (Figure 3). One interpretation of this would be that disturbance does not affect the basal resource
282 use of seagrass food webs in Florida Bay. Instead, consumers might concentrate foraging effort
283 into the remnant patches of seagrass, thus increasing competition, with possible negative effects
284 on secondary production (Macreadie *et al.*, 2010). Consumers might also increase movement and
285 home range size or increase variation in resource use in response to lower amounts of basal
286 resources following the die-off in order to meet energetic demands (Schradin *et al.*, 2010; Lesser
287 *et al.*, 2020; Santos *et al.*, 2022). Alternatively, consumers might display shifts in basal resource

288 use immediately following disturbance but display rapid recovery in the timeframe before
289 sampling occurred. Further investigation into these potential hypotheses will provide vital
290 information into the mechanisms leading to a lack in shift of basal resource use following the
291 seagrass die-off found in this study.

292 Our results reveal that there is a strong seasonal variation in energy flow in the seagrass
293 ecosystems of Florida Bay. Seagrass and mangrove production typically enters the food web
294 through the detrital pool (Cebrian, 1999), and our results reveal that there are seasonal
295 differences in the reliance on brown vs green energy pathways. Basal resources that enter the
296 food web through the green pathway, in our case epiphyte and algal sources, are generally higher
297 quality with lower C:N and C:P ratios and more readily enter the food web compared to sources
298 that enter the food web through the detrital pool (Cebrian, 1999; Evans-White and Halvorson,
299 2017). However, in Florida Bay, sources that make up the majority of the detrital pool (seagrass
300 and mangrove) have higher production than basal resources that contribute to the green pathway,
301 with seagrass the highest production of the basal resources (Smith *et al.*, 2020). Seagrass
302 productivity is seasonal, peaking during summer months (Fourqurean *et al.*, 2001), with a lag
303 before it enters the detrital pool and becomes available to consumers (Cebrian, 1999). Although
304 not always equivalent to production, standing stock biomass of epiphytes is highest during the
305 dry season (Frankovich and Zieman, 2005). It is likely that resource use in Florida Bay follows
306 trends in basal resource production availability regardless of the type of pathway, leading to the
307 seasonal patterns observed in this study. Other coastal systems have displayed seasonal shifts in
308 resource use that are likely related to changes in productivity (Olin *et al.*, 2012; Rezek *et al.*,
309 2017; Jankowska *et al.*, 2019).

310 In the wet season, there was little spatial variation in intraspecific resource use across the
311 bay, despite variability in both the amount and configuration of seagrass habitat as well as
312 several other environmental variables (Fourqurean *et al.*, 1992; Boyer *et al.*, 1997; Kelble *et al.*,
313 2007; Herbert *et al.*, 2011). Spatial variation in production, habitat amount, habitat configuration,
314 and environmental factors (e.g. depth, salinity) have all been shown to affect resource use
315 (Livingston, 1984; Deegan and Garritt, 1997; Olin *et al.*, 2012; Nelson *et al.*, 2015; Giraldo *et*
316 *al.*, 2017; Jankowska *et al.*, 2018; Santos *et al.*, 2022). Our results do not point to these factors
317 influencing basal resource use of seagrass, epiphytes, mangroves, or algae in the wet season as
318 there is little intraspecific spatial variation among the sites sampled, despite broad spatial
319 sampling (Table 3, S1, Figure 4). However, in the dry season, there was more intraspecific
320 spatial variation in basal resource use across all species (Table 3, S1, Figure 4). This variation
321 could be related to seasonal differences in the spatial variation of environmental factors that are
322 known to differ seasonally in our sampling area (Kelble *et al.*, 2007) or related to seasonal
323 differences in productivity as seagrass consumers have been shown to increase spatial variation
324 in resource use under varying levels of primary productivity (Lesser *et al.*, 2020).

325 Though we found minimal spatial variation in basal resource use, we did find spatial
326 variation in species occurrence, and there was variation in basal resource use among species
327 (Table 3). The seagrass die-off could have altered food web structure, especially at higher trophic
328 levels, through a shift in the amount or configuration of seagrass habitat which has been shown
329 to influence species occurrence through alterations in habitat quality, predation efficiency, and
330 competition (Fahrig, 2003; Boström *et al.*, 2011; Santos *et al.*, 2018). Patterns in species
331 occurrence could also be due to the spatial variability in environmental conditions across the bay
332 (Sheridan *et al.*, 1997; Kelble *et al.*, 2013). Environmental conditions vary naturally across the

333 bay, but the seagrass die-off led to local alterations of environmental conditions and could have
334 influenced species occurrence (Fredley *et al.*, 2019; Rodemann *et al.*, 2021). Therefore, further
335 research is needed into the relationship between species occurrence, environmental variables,
336 and seagrass seascape characteristics to better understand the effect of the seagrass die-off on
337 food web structure through the alteration of species composition across the seascape.

338 Consumer species were haphazardly collected from trawl samples and were not measured
339 before processing for stable isotope analysis. Because of this sampling, it is possible that
340 differences in resource use could be related to ontogenetic diet shifts that are related to body size
341 and are observed in some of the species sampled (Scharf *et al.*, 2009; Barbosa and Taylor, 2020).
342 For example, pinfish display ontogenetic diet shifts throughout switching from more carnivorous
343 to omnivorous diets throughout their lifecycle (Stoner, 1980; Barbosa and Taylor, 2020). The
344 spatial variation in resource use of pinfish observed in the dry season could be related to spatial
345 differences in size classes of life history stages across Florida Bay.

346 Contrary to expectations, algal resources were not a major contributor to the food web in
347 Florida Bay (Table 3, Figure 3). Our results are in contrast to food webs from other seagrass
348 ecosystems where algal sources have been shown to be a major energy source for consumers
349 (Deudero *et al.*, 2014; Choi *et al.*, 2017; Lesser *et al.*, 2020). Because POM stable isotope values
350 from other studies throughout Florida Bay overlapped with stable isotope values with the
351 macroalgae (Figure 2; Chasar *et al.*, 2005; Eggenberger *et al.*, 2019), POM falls within the
352 multispecies algal group. We can conclude that this basal resource is not a major contributor to
353 the Florida Bay food web. The low amount of water column production entering the food web is
354 not surprising as Florida Bay is normally a phosphorus limited oligotrophic estuary characterized
355 by having low water column production (Fourqurean *et al.*, 1993; Philips and Badylak, 1996).

356 Because all species sampled, including pelagic consumers, are not heavily relying on water
357 column production, there appears to be strong benthic-pelagic coupling in this system (Lawrence
358 *et al.*, 2004; Chasar *et al.*, 2005). For example, bay anchovy are a widely known planktivorous
359 feeder (Sheridan, 1978), but even bay anchovy derived only a maximum of 6% of their energy
360 from algae (Table 3). Our results do not suggest that bay anchovy in Florida Bay are not
361 planktivorous or feeding on zooplankton, but bay anchovy production is not mainly derived from
362 water column production. Bay anchovy have been found to use benthic production as the
363 primary basal resource in other seagrass systems (Wilson *et al.*, 2010), and high reliance on
364 benthic production for pelagic consumers is not unique to our study and has been shown in other
365 studies in Florida Bay (Chasar *et al.*, 2005; Malinowski *et al.*, 2019). Benthic-pelagic coupling of
366 energy can happen in multiple ways, but two common transfers of organic material from benthic
367 into pelagic habitats are through resuspension/bioturbation and movement of organisms
368 (Lawrence *et al.*, 2004; Griffiths *et al.*, 2017). One possible mechanism of benthic-pelagic
369 coupling is through copepod movement. Copepods that live mainly in seagrass beds have been
370 shown to migrate between seagrass beds and pelagic environments, potentially providing a link
371 between benthic productivity to pelagic predators (Walters and Bell, 1994). In Florida Bay, there
372 was a significant number of benthic harpacticoids found in pelagic zooplankton net tows,
373 suggesting this type of migration might be prevalent (Kelble *et al.*, 2010). Determining the
374 mechanism of benthic-pelagic coupling in Florida Bay will be crucial for predicting how food
375 webs will respond to future disturbances.

376 Our results reveal that there is no shift in basal resource use at the time of sampling across
377 the affected areas, but that does not mean that there was no effect from the die-off. Other studies
378 have shown mixed results regarding changes in trophic contribution in response to changes in

379 seagrass cover; in some cases consumers have shifted trophic pathways (Calizza *et al.*, 2013) or
380 showed no response (Chasar *et al.*, 2005). Samples in this study were taken three years after the
381 start of the 2015 die-off (Hall *et al.*, 2016), and it is possible that the food web could have
382 recovered by the time of sampling. Trawl surveys at pre-existing monitoring stations were used
383 to collect consumers (Kelble *et al.*, 2013), which led to the lower sampling stations at
384 undisturbed sites and low sample numbers of some species (Table 2). Possibly, these lower
385 sampling numbers could limit the ability to detect changes in basal resource use. Additionally,
386 areal coverage of *T. testudinum* decreased markedly in the die-off area, but remnant patches
387 remained and in some locations recruitment of *H. wrightii* was rapid, resulting in comparably
388 dense meadows by 2017 (Furman *et al.*, 2021). Thus, enough seagrass and epiphyte production
389 may have been present to prevent basal resource shifts by the consumers sampled in this study,
390 especially those in the eastern portion of the die-off area (Rodemann *et al.*, 2021). Future studies
391 should include the areal coverage of seagrass around the sampling locations to test for the effects
392 of seagrass cover and patterning on food web structure.

393 Stable isotope analysis combined with Bayesian mixing models are a useful tool for
394 estimating the resource use of consumer species, but like all methods are not without limitations.
395 One such limitation is choosing the trophic enrichment factors (TEFs, also referred to as diet
396 discrimination factors) for each stable isotope in the model, as these can vary based on
397 environmental conditions, physiological factors, and diet quality (Vander Zanden and
398 Rasmussen, 2001; McCutchan *et al.*, 2003; Caut *et al.*, 2009), and these estimates can influence
399 mixing model results (Bond and Diamond, 2011). This variation is particularly true for $\Delta^{15}\text{N}$
400 values, which have the largest fractionation and most variation with each trophic compared to
401 carbon and sulfur (Post, 2002; McCutchan *et al.*, 2003). Like other studies our estimates are

402 likely influenced by our estimation of TEFs. However, unlike most stable isotope studies our
403 models include $\delta^{34}\text{S}$, which is valuable for delineating basal resources in coastal systems
404 (Connolly *et al.*, 2004). The variation of $\delta^{13}\text{C}$ and $\delta^{34}\text{S}$ of basal resources in our system provide a
405 wide mixing space and combined with the inherent low fractionation values of both carbon and
406 sulfur stable isotopes with each trophic step, our models are likely less sensitive to changes in
407 TEF compared to studies that do not include $\delta^{34}\text{S}$. Because $\Delta^{15}\text{N}$ are so variable, and basal
408 resources can vary little in $\delta^{15}\text{N}$ values, future research should investigate situations where $\delta^{15}\text{N}$
409 values are not useful for estimating resource use of consumers.

410 We found no differences in resource use of the consumers sampled in Florida Bay inside or
411 outside the die-off area. As seagrass is lost, and epiphytes no longer have a substrate to grow on,
412 seagrass consumers do not appear to change energy use to compensate for this loss in production.
413 These die-off events have a two-fold effect on seagrass consumers as they decrease the critical
414 structural habitat and they decrease the primary production sources that fuel the food web. Based
415 on our results we would expect there to be a decrease in biomass of seagrass nekton that
416 corresponds to the decrease in the production of seagrass and epiphytes. Following the first
417 widespread seagrass die-off in Florida Bay, the densities of seagrass-associated prey species
418 were significantly lower in algae and mud bottom habitats compared to dense seagrass beds
419 (Sheridan *et al.*, 1997). Many species (e.g. pinfish) use seagrass beds as nursery habitat and
420 transfer seagrass production to offshore predators when migrating to adult habitats, so a decrease
421 in biomass would not only affect food webs within the seagrass beds (Beck *et al.*, 2001; Nelson
422 *et al.*, 2012). If consumers respond by decreasing production, then the loss in seagrass could
423 have cascading effects that reach outside of the initial die-off areas (Lesser *et al.*, 2021).

424 This study investigated the seasonal resource use of common seagrass consumers inside
425 and outside the 2015 seagrass die-off area in Florida Bay three years after the disturbance. Our
426 results showed that consumer resource use did not differ inside or outside the die-off area, but
427 there were seasonal differences in resource use, as seagrass was the most important resource in
428 the dry season, while epiphytes were the most important resource in the dry season. Intraspecific
429 variation in resource use was low in general, but there was more variation in the dry season
430 compared to the wet season. This study highlights that food webs in Florida Bay display similar
431 spatial characteristics in resource use, and that there appears to be no legacy effect on resource
432 use after a major disturbance event.

433

434 **Acknowledgements**

435 We would like to thank Natasha Viadero, Ian Smith, Kelly Montenero, Joe Contillo, Tori
436 Furka, Laura McDonald, and Holly Mayeux for help with field collection and data processing.
437 Benjamin Harlow was responsible for running the stable isotope analysis. This work was
438 supported by Louisiana Sea Grant, The National Academies of Science, Engineering, and
439 Medicine Gulf Research Program, NSF (DEB-1832229), and the Critical Ecosystems Study
440 Initiative (CESI) cooperative agreement P16AC00032. This is contribution XXX from the
441 Coastlines and Oceans Division in the Institute of Environment at Florida International
442 University.

443 **Author contributions**

444 WRJ, ROS, ZWF, BTF, MOH, CRK, JSR, and JAN designed the study. WRJ and JRR collected
445 and processed samples. WRJ analyzed the data and wrote the first draft. All authors contributed
446 substantially to revising the manuscript.

447

448 **Data availability**

449 Data used in this study are available on Github (doi XXX). Doi will be generated when accepted
450 for publication.

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Supplementary Information

Table S1. Mean resource use and species caught at each site sampled in Florida Bay for the wet and dry season.

Site	Lat	Lon	Season	Algae	Epiphytes	Seagrass	Mangrove	Species
123	25.012	-80.926	Wet	0.01 ± 0.01	0.29 ± 0.36	0.51 ± 0.36	0.19 ± 0.01	Mojarra, Silver perch
229	25.026	-80.794	Wet	0.02 ± 0.01	0.15 ± 0.14	0.60 ± 0.22	0.24 ± 0.08	Bay anchovy, Mojarra, Pinfish
147	25.046	-80.91	Wet	0.02 ± 0.01	0.16 ± 0.16	0.63 ± 0.15	0.20 ± 0.01	Mojarra, Pinfish, Rainwater killifish
258	25.047	-80.766	Wet	0.02 ± 0.01	0.15 ± 0.14	0.59 ± 0.22	0.24 ± 0.08	Bay anchovy, Mojarra, Pinfish
271	25.058	-80.752	Wet	0.02 ± 0.01	0.15 ± 0.14	0.59 ± 0.22	0.24 ± 0.08	Bay anchovy, Mojarra, Pinfish
133	25.061	-80.947	Wet	0.02 ± 0.01	0.21 ± 0.21	0.54 ± 0.22	0.23 ± 0.06	Bay anchovy, Mojarra, Pigfish, Pinfish, Silver perch
242	25.062	-80.804	Wet	0.02 ± 0.01	0.15 ± 0.15	0.63 ± 0.15	0.20 ± 0.01	Mojarra, Pinfish, Rainwater killifish
226	25.064	-80.83	Wet	0.02 ± 0.01	0.22 ± 0.17	0.56 ± 0.14	0.20 ± 0.01	Pinfish, Rainwater killifish
159	25.07	-80.907	Wet	0.02 ± 0.01	0.26 ± 0.22	0.51 ± 0.19	0.21 ± 0.03	Pigfish, Pinfish, Rainwater killifish, Silver perch
256	25.074	-80.787	Wet	0.02 ± 0.01	0.24 ± 0.13	0.49 ± 0.16	0.25 ± 0.08	Bay anchovy, Pinfish, Rainwater killifish
225	25.077	-80.842	Wet	0.03	0.04	0.76	0.19	Mojarra
294	25.078	-80.728	Wet	0.03	0.04	0.77	0.19	Mojarra
169	25.081	-80.893	Wet	0.02 ± 0.01	0.19 ± 0.15	0.56 ± 0.19	0.23 ± 0.07	Bay anchovy, Mojarra, Pinfish, Rainwater killifish
283	25.081	-80.75	Wet	0.02 ± 0.01	0.14 ± 0.14	0.59 ± 0.22	0.24 ± 0.08	Bay anchovy, Mojarra, Pinfish
269	25.083	-80.776	Wet	0.02 ± 0.01	0.14 ± 0.14	0.59 ± 0.22	0.25 ± 0.08	Bay anchovy, Mojarra, Pinfish
268	25.0965	-80.7877	Wet	0.02 ± 0.01	0.24 ± 0.13	0.49 ± 0.16	0.25 ± 0.08	Bay anchovy, Pinfish, Rainwater killifish
176	25.1	-80.866	Wet	0.01 ± 0.01	0.17 ± 0.19	0.55 ± 0.30	0.26 ± 0.10	Bay anchovy, Mojarra
239	25.1	-80.839	Wet	0.01 ± 0.01	0.14 ± 0.17	0.67 ± 0.18	0.18 ± 0.03	Mojarra, Rainwater killifish, Pink shrimp
292	25.105	-80.748	Wet	0.02 ± 0.01	0.19 ± 0.15	0.56 ± 0.19	0.23 ± 0.07	Bay anchovy, Mojarra, Pinfish, Rainwater killifish
281	25.107	-80.774	Wet	0.02 ± 0.01	0.12 ± 0.12	0.64 ± 0.20	0.22 ± 0.08	Bay anchovy, Mojarra, Pinfish, Pink shrimp
267	25.109	-80.799	Wet	0.01 ± 0.01	0.13 ± 0.12	0.61 ± 0.18	0.25 ± 0.06	Bay anchovy, Mojarra, Pigfish, Pinfish
253	25.111	-80.825	Wet	0.03	0.04	0.76	0.2	Mojarra
291	25.118	-80.76	Wet	0.02 ± 0.01	0.16 ± 0.14	0.61 ± 0.20	0.21 ± 0.07	Bay anchovy, Mojarra, Pinfish, Rainwater killifish, Pink shrimp
174	25.128	-80.889	Wet	0.02 ± 0.01	0.21 ± 0.21	0.54 ± 0.22	0.23 ± 0.06	Bay anchovy, Mojarra, Pigfish, Pinfish, Silver perch
279	25.133	-80.797	Wet	0.01 ± 0.01	0.20 ± 0.22	0.56 ± 0.25	0.22 ± 0.07	Bay anchovy, Mojarra, Pigfish, Pink shrimp, Silver perch
265	25.135	-80.823	Wet	0.01 ± 0.01	0.17 ± 0.18	0.55 ± 0.29	0.27 ± 0.10	Bay anchovy, Mojarra
278	25.145	-80.809	Wet	0.01 ± 0.01	0.17 ± 0.18	0.57 ± 0.32	0.24 ± 0.13	Bay anchovy, Pink shrimp
123	25.012	-80.926	Dry	0.03	0.57	0.11	0.29	Pinfish
135	25.036	-80.924	Dry	0.03 ± 0.01	0.56 ± 0.10	0.19 ± 0.08	0.22 ± 0.01	Pigfish, Pinfish, Silver perch
228	25.038	-80.806	Dry	0.03	0.64	0.22	0.13	Mojarra
147	25.046	-80.91	Dry	0.03	0.42	0.12	0.44	Pinfish
257	25.06	-80.778	Dry	0.03	0.42	0.34	0.16	Rainwater killifish
133	25.061	-80.947	Dry	0.04 ± 0.03	0.42 ± 0.17	0.25 ± 0.07	0.30 ± 0.06	Bay anchovy, Mojarra

242	25.062	-80.804	Dry	0.03	0.42	0.33	0.17	Rainwater killifish
226	25.064	-80.83	Dry	0.05 ± 0.05	0.54 ± 0.15	0.26 ± 0.08	0.15 ± 0.02	Mojarra, Rainwater killifish
159	25.07	-80.907	Dry	0.03	0.58	0.12	0.28	Pinfish
145	25.072	-80.933	Dry	0.05 ± 0.04	0.46 ± 0.19	0.25 ± 0.13	0.25 ± 0.02	Bay anchovy, Pigfish, Pinfish, Pink shrimp, Silver perch
256	25.074	-80.787	Dry	0.03	0.6	0.2	0.18	Mojarra
225	25.077	-80.842	Dry	0.03 ± 0.02	0.53 ± 0.11	0.24 ± 0.09	0.21 ± 0.03	Mojarra, Pigfish, Pinfish, Rainwater killifish, Silver perch
169	25.081	-80.893	Dry	0.02 ± 0.01	0.45 ± 0.09	0.22 ± 0.04	0.31 ± 0.05	Mojarra, Pigfish, Silver perch
283	25.081	-80.75	Dry	0.03	0.64	0.22	0.13	Mojarra
158	25.083	-80.919	Dry	0.03 ± 0.04	0.42 ± 0.21	0.27 ± 0.17	0.27 ± 0.04	Mojarra, Pigfish, Pinfish, Pink shrimp
269	25.083	-80.776	Dry	0.04 ± 0.03	0.64 ± 0.12	0.19 ± 0.08	0.13 ± 0.02	Mojarra, Pinfish, Rainwater killifish, Silver perch
255	25.085	-80.802	Dry	0.05 ± 0.05	0.54 ± 0.15	0.26 ± 0.08	0.15 ± 0.02	Mojarra, Rainwater killifish
240	25.088	-80.828	Dry	0.03 ± 0.02	0.30 ± 0.12	0.46 ± 0.14	0.21 ± 0.05	Mojarra, Pinfish, Rainwater killifish, Silver perch
224	25.09	-80.853	Dry	0.06 ± 0.04	0.26 ± 0.18	0.43 ± 0.14	0.26 ± 0.01	Mojarra, Rainwater killifish, Pink shrimp
173	25.092	-80.879	Dry	0.02 ± 0.01	0.59 ± 0.05	0.20 ± 0.02	0.19 ± 0.03	Mojarra, Pigfish
282	25.094	-80.762	Dry	0.08 ± 0.01	0.43 ± 0.01	0.35 ± 0.01	0.14 ± 0.01	Rainwater killifish
268	25.0965	-80.7877	Dry	0.03	0.42	0.3	0.2	Rainwater killifish
254	25.098	-80.814	Dry	0.03	0.43	0.28	0.25	Silver perch
176	25.1	-80.866	Dry	0.07 ± 0.05	0.44 ± 0.10	0.25 ± 0.06	0.24 ± 0.01	Pigfish, Rainwater killifish
239	25.1	-80.839	Dry	0.04 ± 0.03	0.35 ± 0.17	0.43 ± 0.16	0.18 ± 0.03	Bay anchovy, Mojarra, Pinfish, Rainwater killifish, Pink shrimp, Silver perch
281	25.107	-80.774	Dry	0.03	0.38	0.32	0.23	Rainwater killifish
267	25.109	-80.799	Dry	0.08 ± 0.03	0.24 ± 0.16	0.41 ± 0.11	0.27 ± 0.02	Rainwater killifish, Pink shrimp
253	25.111	-80.825	Dry	0.08 ± 0.03	0.26 ± 0.17	0.44 ± 0.13	0.22 ± 0.02	Rainwater killifish, Pink shrimp
280	25.12	-80.785	Dry	0.06 ± 0.03	0.32 ± 0.15	0.41 ± 0.12	0.22 ± 0.02	Rainwater killifish, Pink shrimp, Silver perch
237	25.126	-80.863	Dry	0.03	0.13	0.56	0.17	Pink shrimp
290	25.131	-80.771	Dry	0.03	0.42	0.32	0.18	Rainwater killifish
279	25.133	-80.797	Dry	0.03	0.41	0.33	0.18	Rainwater killifish
265	25.135	-80.823	Dry	0.03	0.54	0.24	0.2	Mojarra
278	25.145	-80.809	Dry	0.03	0.59	0.22	0.17	Mojarra

Table S2. Isotope values (mean \pm SD) and elemental concentration of basal resources. POM source values are based on (Chasar *et al.*, 2005; Eggenberger *et al.*, 2019) but were not included as a potential source in the mixing models

Source	$\delta^{13}\text{C}$	%C	$\delta^{15}\text{N}$	%N	$\delta^{34}\text{S}$	%S
Epiphytes	-13.6 ± 2.2	48	0.4 ± 0.5	8	10.1 ± 2.6	2
Seagrass	-11.3 ± 1.6	37.8	-0.5 ± 3.1	2.1	4.0 ± 3.9	0.5
Mangrove	-27.7 ± 1.1	44	1.1 ± 1.4	1.5	-13.5 ± 3.8	0.5
Algae	-18.6 ± 3.8	30	2.6 ± 2.5	2.3	19.6 ± 3.2	2.7
POM	-19.4 ± 4.1		3.3 ± 3		18.0 ± 2.0	

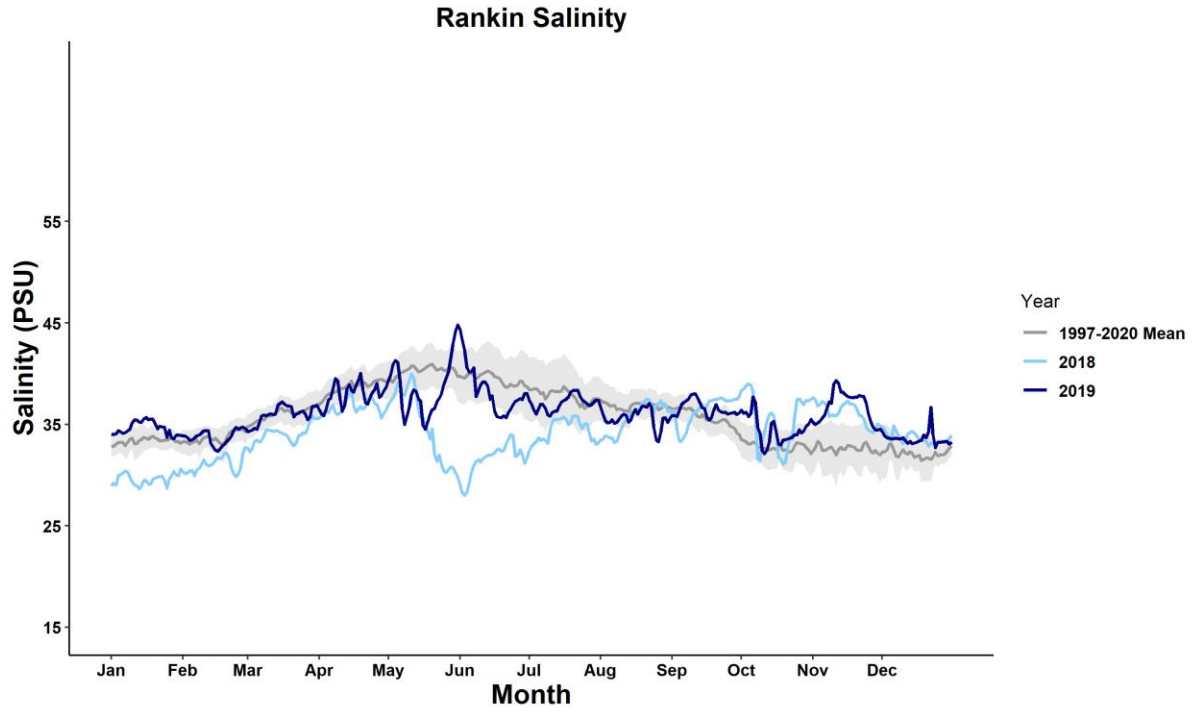


Figure S1. Average salinity in Rankin Basin in 2018 and 2019 compared to mean salinity between 1993-2020.

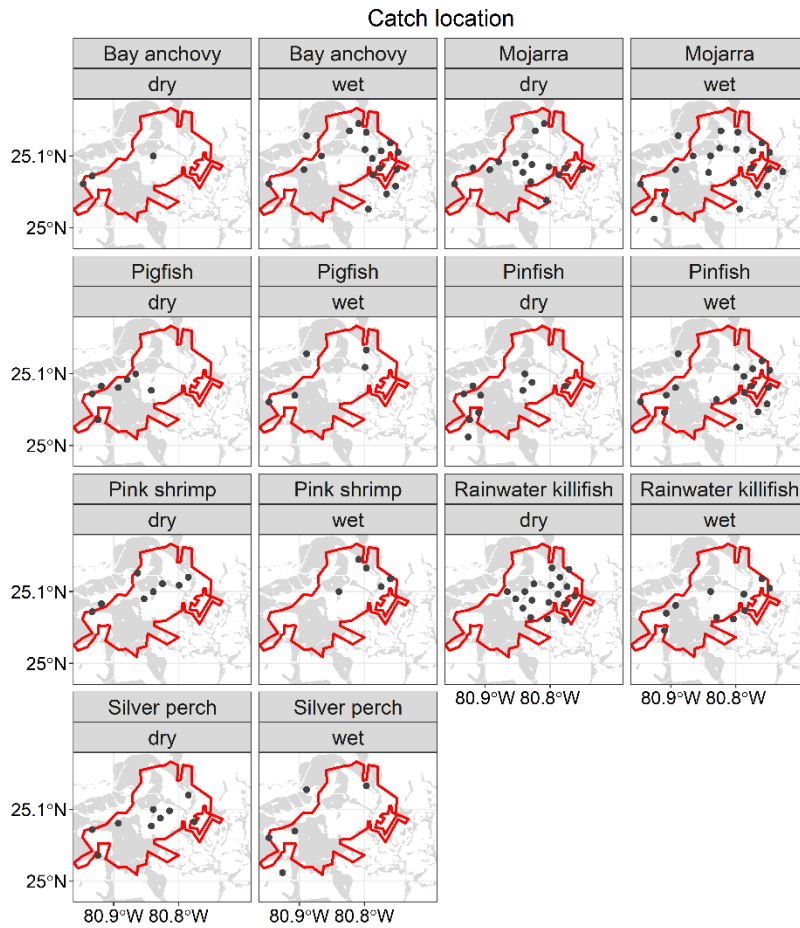


Figure S2. Site locations where species were collected in Florida Bay for the wet and dry seasons. Mojarra = Silver Jenny mojarra