Coastal residential canals harbor distinct water quality conditions and phytoplankton community composition

Jordana Cutajar, Blair Sterba-Boatwright, Michael S. Wetz

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- 1 Coastal residential canals harbor distinct water quality conditions and phytoplankton community
- 2 composition
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- 4 \*Jordana Cutajar<sup>1</sup>
- 5
- 6 Blair Sterba-Boatwright<sup>2</sup>7
- 8 Michael S. Wetz<sup>1</sup> 9
- 10 <sup>1</sup>Harte Research Institute for Gulf of Mexico Studies, Texas A&M University-Corpus Christi,
- 11 6300 Ocean Drive, Corpus Christi, Texas 78412
- <sup>12</sup> <sup>2</sup>Department of Mathematics and Statistics, Texas A&M University-Corpus Christi, 6300 Ocean
- 13 Drive, Corpus Christi, Texas 78412
- 14 \*Corresponding author
- 15 Email: jordanac@comcast.net
- 16 Phone: 954-253-7278
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### 28 ABSTRACT

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30 As urbanization increases, many regions around the globe are seeing an increase in the 31 number of residential canal systems along their coastlines. These canals possess unique attributes 32 that may facilitate water quality degradation, namely shallow depths, susceptibility to urban 33 runoff, and limited flushing. Despite this, there has been little research on the water quality and 34 phytoplankton dynamics of these systems. In this study, water quality and phytoplankton 35 biomass/composition were quantified once per month in the fall and winter, and twice per month 36 in the spring and summer, over a 1-year period at three sites along a mouth-interior gradient of a 37 canal system on North Padre Island (Corpus Christi, Texas, USA). It was hypothesized that 38 interior canal sites would exhibit symptoms of water quality degradation more regularly than a 39 site at the mouth, and interior canal sites would also have higher chlorophyll a and distinct 40 phytoplankton community composition compared to the mouth. Results showed that the mouth 41 site exhibited lower inorganic nutrient concentrations and less pronounced variability in salinity 42 and nutrients than the interior canal sites. Chlorophyll a was 2.5-3-fold higher in the canal sites 43 than at the mouth on average, and diatoms dominated at the mouth while a more diverse 44 assemblage dominated at the canal sites. Episodic to persistent hypoxia was observed at the canal 45 sites, but not at the mouth. These findings show that artificial coastal residential canal systems 46 have distinct and less desirous water quality conditions and phytoplankton communities, pointing 47 to a need for additional research on the attributes of canals that foster these conditions to support 48 effective ecosystem health management efforts.

49

50 Keywords: canal, water quality, nutrients, phytoplankton, oxygen, estuary

52

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62 INTRODUCTION

63 Estuaries and coastal wetlands have increasingly been modified by the creation of 64 residential waterfront properties with canal systems (Waltham and Connolly 2011). For example, Waltham and Connolly (2011) estimated that there was approximately 4000 km of canal length 65 66 and 270 km<sup>2</sup> covered by coastal residential canals globally. Approximately 77 percent of the 67 total global length of artificial coastal residential canals can be found in North America, but 68 locally dense concentrations of canals can also be found in Asia, Europe and Oceania (~7% of 69 total global length each) (Waltham and Connolly 2011). Canals are frequently used for boating 70 (Sagerman et al. 2020; Swett et al. 2009), swimming (Breen et al. 2018), fishing (Bisping & 71 Thompson 2017) and have been associated with increased property values depending on location 72 and proximity to the water (Nicholls and Crompton, 2017). Canals therefore represent a key 73 nexus between ecosystem and human/economic health. Despite the prevalence of canals 74 globally, as well as their potential importance to humans and wildlife, there have been few 75 studies on the ecology of these systems and specifically on water quality conditions in them, 76 especially from North America. This is despite inferences that can be drawn from a limited 77 number of prior studies, namely that there is a tendency towards poor water quality (e.g., low 78 oxygen levels in the water column, pesticide and metal build up in sediments and/or fish) in 79 residential canals that may be incompatible with good fisheries and/or human health (see e.g., 80 Lindall and Trent 1975; Maxted et al. 1997; Waltham et al. 2011; Waltham and Connolly 2011). 81 Canals in North America are unique in their design, favoring longer, narrower systems 82 and more dead ends than canals on other continents (Waltham and Connolly 2011). Historical 83 usage is an apparent reason for differences in design between countries, as canals in some 84 locations such as Europe were built to accommodate transportation and provide flood mitigation,

85 whereas in the U.S. canals are built primarily to provide recreational boating access and 86 waterfront views to residential communities (Nicholls and Compton 2017). Regardless of the 87 motivation behind canal construction, features of these artificial waterbodies can disrupt natural 88 systems and have unintended consequences on water quality. For example, it has been theorized 89 that the dead end design of residential canals as well as wind shielding caused by houses on the 90 sides of them can decrease flushing (Corliss and Trent 1971; Lindall and Trent 1975; Xing et al. 91 2018). Additionally, residential canals by definition are in developed areas where they are 92 susceptible to urban runoff and associated nutrient pollution (Ma et al. 2006; Ivey et al. 2020). In 93 brackish retention ponds, another type of artificial coastal waterbody that shares features with 94 canals such as close proximity to urban areas and restricted flow, previous research has shown 95 that they may experience persistent algal blooms and increased prevalence of harmful algal 96 blooms (HABs; Lewitus et al. 2003, 2008). Residential canals are also shallow and can exhibit low oxygen conditions (Lindall and Trent 1975), factors which can lead to enhanced exchange of 97 98 nutrients between the water column and sediments (Cloern 2001; Kemp et al. 2005). Some 99 canals may also contain acid sulphate sediments that favor dissimilatory nitrate reduction to 100 ammonium and further enhance exchange of ammonium from sediments to the water column 101 (Ferrada 2017). In general, limited flushing as well as exposure to potentially high external and 102 internal nutrient loadings has been shown to increase susceptibility of waterbodies to 103 eutrophication (e.g., Bricker et al. 2008).

Although studies on water quality of coastal residential canals are limited, some potential ecological and economic impacts of poor water quality can be speculated upon. For example, poor water quality conditions should, in theory, lead to poor habitat quality for marine life. A synthesis of fish kills in coastal Texas found that a significant fraction of documented fish kills

108 occurred in coastal canals, primarily due to low dissolved oxygen (D.O.; Thronson and Quigg, 109 2008). Furthermore, a study of benthic invertebrate communities in coastal canals found that 110 dead end canals similar to those studied here tend to have benthic communities with lower 111 diversity and species richness compared to locations away from the interior of the canals, which 112 was attributed to both frequent low D.O. conditions and presence of fine-grained sediments 113 (Cosser 1989). In the aforementioned example of HABs in the analogous residential detention 114 ponds, the authors concluded that exchange with adjacent waterbodies may lead to "seeding" of 115 those waterbodies with HAB species from the detention ponds (Lewitus et al. 2003, 2008). In the 116 Florida Keys, it was shown that when nutrient-enriched water from manmade canals exchanged 117 with nearby coastal waters, water clarity decreased in the coastal waters, leading to a decrease in 118 coral productivity (Lapointe and Clark 1992). In terms of potential economic impacts, it has been 119 demonstrated that the value of coastal properties is dependent on water quality (Bonetti et al. 2016). For example, when a major HAB event affected southwest Florida, it was found that 120 121 property located within a mile of the coast sold for  $\sim 30\%$  less than properties in a neighboring 122 county that were not affected by the HAB (Bechard 2020).

123 As coastal populations increase, canal communities are likely to become more prevalent. 124 Because canals have attributes that in theory may facilitate water quality degradation that could 125 have concomitant impacts on the environment and economy, there is a need to quantify water 126 quality conditions in representative systems as a first step towards developing management 127 actions. The goal of this study was to assess the water quality and phytoplankton composition in 128 a residential canal system on the rapidly developing North Padre Island (Corpus Christi, Texas, 129 USA). This study tested the hypothesis that sites toward the interior of the canal system would 130 experience higher nutrient concentrations and phytoplankton biomass, more frequent hypoxia

131	and low pH water, and distinct phytoplankton community composition compared to the mouth of
132	the system. This study is one of the most comprehensive water quality/phytoplankton studies
133	from a coastal residential canal system to date, and results should be informative to scientists and
134	stakeholders looking to foster healthier canal systems in urbanized coastal areas.
135	
136	METHODS
137	Site characteristics
138	The study site was a residential canal system located on North Padre Island (Corpus
139	Christi, Texas, USA) (Fig. 1). Three sites were sampled: at the mouth of the canal system and
140	adjacent to the Upper Laguna Madre ("mouth"), a site mid-way between the mouth and the back
141	of the canal system ("middle"), and a site furthest interior in the canal system ("interior"). The
142	middle site is located approximately 3.2 km from the mouth of the canal system and is
143	surrounded by homes on either side. The interior site is located approximately 6.4 km from the
144	mouth of the canal system, with homes on one side of the canal.
145	Sample collection and processing
146	During the fall and winter, samples were collected once per month, while in the spring
147	and summer seasons, sampling switched to twice per month. There were 19 sampling trips total
148	from November 2020 until October 2021.
149	At each sampling location, acid-washed brown 1 L HDPE bottles were used to collect
150	water samples at 0.1 m below the surface. Water samples were immediately stored on ice for
151	transport to the lab. An additional water sample was taken at the surface for phytoplankton
152	analysis in acid washed 500 mL brown HDPE bottles. These samples were stored at ambient

153 temperature. Hydrographic data (dissolved oxygen, pH, conductivity, salinity, and temperature) 154 were collected from each sampling location at 0.5 m intervals in the water column using a YSI 155 multiparameter sonde. A stratification index was obtained by subtracting salinity recorded at the 156 bottom of the water column from salinity recorded at the top. In this case, the stratification index 157 refers to the strength of the halocline, as vertical temperature gradients were rare (data not 158 shown). Precipitation data was obtained from the National Oceanic and Atmospheric 159 Administration's National Weather Service and is represented as the cumulative rainfall for 7 160 days prior to a sampling date. 161 Once at the lab, water samples were filtered through precombusted 25 mm 0.7 µm pore 162 size GF/F filters. Filters were stored frozen (-20°C) in sealed Vacutainers until analysis. 163 Chlorophyll a was extracted from the filters by soaking for 18-24 hours in 90% HPLC-grade 164 acetone at -20°C, after which chlorophyll a was determined fluorometrically with a Turner 165 Trilogy fluorometer without acidification. Inorganic nutrients were determined using the filtrate 166 of water samples that were passed through a 25 mm GF/F filter and stored frozen (-20°C) until 167 analysis. After thawing to room temperature, samples were analyzed on a Seal QuAAtro 168 autoanalyzer. Standard curves with five different concentrations were run daily at the beginning 169 of each run. Fresh standards were made prior to each run by diluting a primary standard with low 170 nutrient surface seawater. Deionized water (DIW) was used as a blank, and DIW blanks were run 171 at the beginning and end of each run, as well as after every 8-10 samples to correct for baseline 172 shifts. Ambient water samples collected for phytoplankton enumeration were stored in amber 173 glass bottles and preserved using acid Lugol's solution (1 mL Lugol's: 60 mL sample water). An 174 inverted microscope and settling chambers were used for phytoplankton identification and 175 enumeration (Edler and Elbrächter 2010). 4-10 ml of sample was settled in chambers for 24

hours. Phytoplankton were counted at 20x magnification using an Olympus 1X-71 inverted
microscope. Samples for picoplankton were preserved using 80 µl of glutaraldehyde mixed with
4 mL of sample water and stored at -80°C until time of analysis. Samples were then thawed in
the dark and filtered through 20 µm Nytex mesh before being analyzed. Picoplankton abundance
and biomass were estimated using a BD Accuri C6 Plus Flow Cytometer. Each phytoplankton
taxa that was quantified was subsequently classified by shape using Hillebrand et al. (1999) and
Sun and Liu (2003) to calculate biovolume.

183 Statistical analyses

184 Data were analyzed using R-Studio. Since some data were non-normal, transformed 185 values were analyzed using Kruskal-Wallis one way Analysis of Variance, with Wilcoxon post 186 hoc testing to determine if there were significant differences in environmental and phytoplankton 187 parameters based on site. Non-transformed values were used to run a Principal Component 188 Analysis (PCA) to identify relationships among the environmental variables. A general linear 189 model (GLM) was created for each phytoplankton group using all available water quality and 190 nutrient variables, as well as the categorical variables site and season, to identify potentially 191 significant explanatory variables. Phytoplankton and nutrient values were log transformed, as 192 necessary. The dredge function was used to examine a variety of different models. Top models 193 for each phytoplankton group were then selected and graphed to check for linearity, 194 homoscedasticity, independence and normality.

195

196 RESULTS

197 Environmental variables

198	There were no statistically significant differences in mean temperature among the three
199	sites (Table 1; p=0.81). The lowest temperatures were recorded in February 2021 following a
200	freeze event (15.7-17.7°C), while the highest temperatures were in August (30.1-31.6°C; Fig.
201	2a). Notable wet periods (rain > 25 mm) in the 7 d prior to sampling were observed on $5/2/21$ ,
202	5/17/21, 6/7/21, 7/9/21, 9/20/21, 10/4/21 and 10/20/21 (Suppl. Fig. 1). Mean salinity ranged from
203	26.5-30.4 but was not statistically different between sites (Table 1; p=0.40). Higher salinity was
204	observed in the winter and spring, while the lowest salinity generally occurred during the
205	summer (Fig. 2b). Following a rain event in early July 2021, the middle and interior canal sites
206	had their lowest recorded salinities (14.9 and 5.1, respectively) (Fig. 2b), whereas the mouth's
207	lowest recorded salinity (20.3) occurred later that month. The only instances of stratification
208	documented throughout the study occurred following heavy rainfall events, particularly in the
209	beginning of May and July 2021 (Suppl. Fig. 2). The intensity of stratification varied based on
210	site. Generally, the mouth experienced little to no stratification, while rain events created
211	occasional stratification at the middle and interior sites.
212	Mean surface pH ranged from 7.99-8.15 but was not statistically different between the
213	three sites (Table 1; p=0.07). The mouth and interior sites experienced reduced pH's (7.04, 7.02
214	respectively) in early May 2021 (Fig. 3a). Mean bottom pH was significantly lower at the
215	interior site (7.94 $\pm$ 0.23) compared to the mouth (8.19 $\pm$ 0.21) and middle site (8.12 $\pm$ 0.19)
216	(Table 1). The interior site notably low pH (7.19) in early July 2021 (Fig. 3b). Mean surface
217	D.O. was lowest at the interior site (4.02 $\pm$ 1.57 mg/L), while the mouth and middle sites had a
218	higher mean surface D.O. (5.38 $\pm$ 1.26 mg/L and 5.03 $\pm$ 2.23 mg/L respectively) and were not
219	statistically different from one another (Table 1). The highest surface D.O. was recorded in
220	February 2021 following the freeze event, reaching 7.99 mg/L at the mouth, 8.08 mg/L at the

221	interior site, and 11.02 mg/L at the middle site (Fig. 3c). The lowest recorded surface D.O. levels
222	were observed in mid-August to early September 2021, ranging from 1.88-2.80 mg/L (Fig. 3c).
223	Bottom D.O. was generally lower than surface D.O. (Table 1; Fig. 3d). Mean bottom D.O. was
224	similar between canal sites, but significantly lower at both the interior (1.98 $\pm$ 2.67 mg/L) and
225	middle site (2.67 $\pm$ 3.77 mg/L) compared to the mouth (4.70 $\pm$ 1.34 mg/L) (Table 1). As with
226	surface D.O., highest levels at the bottom were found in February 2021 following the freeze.
227	Instances of hypoxia (D.O. < 2.0 mg/L) were common at the interior site, occurring in March,
228	May, and July-October 2021 (Fig. 3d). The middle site also had instances of hypoxia in April
229	and mid-August 2021 (Fig. 3d).
230	N+N concentrations were not different on average between the middle (2.64 $\pm$ 2.87 $\mu M$ )
231	and interior (3.50 $\pm$ 3.65 $\mu$ M) sites but were significantly higher at those two sites compared to
232	the mouth (0.36 $\pm$ 0.20 $\mu M,$ p <0.01) (Table 1, Figure 4a). Several instances of relatively high
233	( $\geq$ 5 µM) N+N concentrations were observed at the canal sites, including November-December
234	2020, February-March 2021, early May 2021 and mid July 2021. The former two events did not
235	correspond with rainfall, but the February-March 2021 event did follow a freeze that resulted in a
236	large fish kill in the canals. The May and July high N+N concentrations co-occurred with rain
237	and a concomitant decrease in salinity (Fig. 4a). Mean ammonium concentrations were highest at
238	the interior site (7.52 $\pm$ 6.51 $\mu M$ ), followed by the middle (5.64 $\pm$ 4.13 $\mu M$ ) and mouth (3.79 $\pm$
239	1.86 $\mu$ M), although there was no statistically significant difference between sites (p= 0.21)
240	(Table 1, Fig. 4b). Ammonium was relatively low at all three sites in December 2020 and
241	January 2021 (Fig. 4b). Following the freeze in February 2021, the interior site experienced the
242	highest recorded ammonium level (25.1 $\mu$ M) while the middle site experienced the second
243	highest ammonium level (17.8 µM) (Fig. 4b). Several additional instances of relatively high (≥5

244 uM) ammonium concentrations were observed, especially at the interior site and frequently 245 coinciding with rainfall (Fig. 4b). Average phosphate concentrations were highest at the interior 246 site  $(2.83 \pm 4.70 \ \mu\text{M})$ , followed by the middle  $(0.88 \pm 0.87 \ \mu\text{M})$  and mouth  $(0.20 \pm 0.14 \ \mu\text{M})$ 247 (Table 1, Fig. 4c). While phosphate concentrations were not statistically different between canal 248 sites, they were both higher than at the mouth site on average (p<0.01; Table 1). The interior site 249 experienced the highest orthophosphate values in May (15.1  $\mu$ M) and July (16.4  $\mu$ M), 250 corresponding with rain events (Fig. 4c). The middle site also saw relatively high concentrations 251 during those timeframes, reaching 2.28  $\mu$ M and 3.09  $\mu$ M in May and July respectively. Mean 252 dissolved inorganic nitrogen to orthophosphate ratios were similar between the mouth (42.1  $\pm$ 253 47.1) and middle sites (28.4  $\pm$  44.8), but significantly lower at the interior site (8.1  $\pm$  7.2; p<0.01; 254 Table 1). Relatively high ratios (>16) were occasionally observed, most frequently during the 255 summer at the mouth and middle sites, while relatively low ratios (<16) were more common in 256 the winter and at the interior site (Suppl. Fig. 3). Mean silicate concentrations were lowest at the 257 mouth (53.4  $\pm$  31.8  $\mu$ M), followed by the middle (73.9  $\pm$  34.0  $\mu$ M) and interior sites (130  $\pm$  61.9 258  $\mu$ M) (Table 1, Fig. 4d). All three sites experienced relatively low silicate in the fall and winter of 259 2020 and higher concentrations from mid-April 2021 onward, especially coincident with rain 260 events in May and July (Fig. 4d). Mean dissolved inorganic nitrogen to silicate (DIN:Si) ratios 261 were low for all three sites (<0.18). A brief increase was observed in March 2021, with the 262 middle site experiencing the highest recorded ratio of 1.10 and the interior site reaching 0.64 263 (Suppl. Fig. 3).

Collectively, PC's 1 and 2 explained 72.9% of the variance (Fig. 5). PC1 explained
47.4% of the variability and had salinity that was inverse to precipitation, silicate, stratification
and phosphate. Thus, PC1 likely represents a freshwater inflow index. PC2 explained 25.5% of

267	the variability and had temperature that was inverse to DIN:Si and D.O. Thus, PC2 likely
268	represents an ecosystem metabolism index. While chlorophyll $a$ was not as well represented in
269	PC's 1 and 2, we can still infer a negative correlation between it and Secchi, DIN:PO4, and pH,
270	and a positive correlation with N+N and ammonium.
271	Phytoplankton dynamics
272	Chlorophyll <i>a</i> concentrations were not different on average between the middle (16.1 $\pm$
273	10.6 $\mu$ g L <sup>-1</sup> ) and interior canal sites (19.5 ± 9.02 $\mu$ g L <sup>-1</sup> ), but were significantly higher at those
274	two sites compared to the mouth (6.40 $\pm$ 3.31 µg L <sup>-1</sup> ) (p <0.01) (Table 1, Figure 6). At the
275	mouth, chlorophyll <i>a</i> was consistently $<15 \ \mu g \ L^{-1}$ throughout the study. In contrast, chlorophyll <i>a</i>
276	concentrations >15 $\mu$ g L <sup>-1</sup> were frequently observed at the canal sites (Fig. 6). The interior site
277	experienced its highest recorded chlorophyll <i>a</i> concentration of 43 $\mu$ g L <sup>-1</sup> in January 2021, while
278	the middle site experienced a peak of 45 $\mu$ g L <sup>-1</sup> in February 2021 (Fig. 6). The most pronounced
279	bloom periods either occurred during dry periods (January-February 2021) or with a lag
280	following a rainy period (June 2021) (Figs. 2b,6).
281	On average, diatoms were the largest contributor to community biovolume at the mouth
282	(63.7 $\pm$ 22.0%), followed by dinoflagellates (17.3 $\pm$ 16.3%). All other groups contributed <10%
283	(Table 1; Fig. 7). Dinoflagellates became a slightly more important contributor during the late
284	summer-fall months, while the contribution from diatoms lessened (Fig. 7). The phytoplankton
285	community was consistently more diverse at the canal sites. For example, dinoflagellates (26.3 $\pm$

286 26.9%), picoeukaryotes ( $25.8 \pm 24.1\%$ ), diatoms ( $23.5 \pm 27.7\%$ ) and picocyanobacteria ( $12.0 \pm$ 

287 11.7%) were all important contributors to community biovolume on average at the middle site

288 (Table 1; Fig. 7). Phytoplankton community composition appeared to be dynamic, as consistent

289 seasonal patterns were not observed except for picoeukaryote dominance during the April-early

290 June 2021 timeframe (Fig. 7). During instances of notably high (>20  $\mu$ g L<sup>-1</sup>) chlorophyll a 291 observed in fall 2020 through spring 2021, diatoms (primarily Chaetoceros sp.) and/or mixed 292 communities dominated, while euglenoids dominated a bloom in June (Fig. 7). At the interior 293 site, picoeukaryotes  $(37.9 \pm 26.3\%)$ , diatoms  $(26.2 \pm 28.5\%)$ , dinoflagellates  $(12.6 \pm 18.8\%)$  and 294 picocyanobacteria ( $10.8 \pm 8.8\%$ ) were important contributors to community biovolume (Table 1; 295 Fig. 7). Phytoplankton community composition appeared to be dynamic at the interior site as well and no consistent seasonal patterns were observed with the exception of picoeukaryote 296 297 dominance during spring-early summer (Fig. 7). During instances of notably high (>20  $\mu$ g L<sup>-1</sup>) 298 chlorophyll a observed in early August 2021, diatoms dominated, whereas dinoflagellates 299 dominated during a bloom in January 2021, and the community was mixed during an early 300 October 2021 bloom (Fig. 7).

301 The GLM for dinoflagellates explained 54% of their variability and was represented by 302 the following explanatory variables: chlorophyll, D.O, log(N+N), log(orthophosphate), site and 303 Secchi. The GLM for diatoms explained 34% of their variability and was represented by the 304 following explanatory variables: depth, log(precipitation), and silicate. The GLM for 305 picoyanobacteria explained 65% of their variability and was represented by the following 306 explanatory variables:  $\log(DIN:Si)$ ,  $\log(N+N)$ , salinity and season. Finally, the GLM for 307 picoeukaryotes explained 71% of their variability and was represented by the following 308 explanatory variables: site, log(DIN:SI), log(N+N), and log(orthophosphate). Full statistical 309 output from each model can be found in Supplemental Table 1.

310

311 DISCUSSION

312 Residential canals are expanding worldwide, but few studies have documented the water 313 quality of canals or their susceptibility to water quality degradation. Canals tend to be shallow, 314 often experience restricted flushing, and receive urban stormwater runoff, all of which set the 315 stage for them to exhibit symptoms of water quality degradation. In this study, water quality 316 conditions were assessed over the course of one year at three sites from the mouth to interior of a 317 coastal residential canal system. During the sampling period, the region experienced a typical 318 seasonal temperature cycle, wet and dry conditions, and extreme weather phenomena including a 319 freeze and heavy rainfall events. It was hypothesized that sites toward the interior of the canal 320 system would experience higher nutrient concentrations and phytoplankton biomass, more 321 frequent hypoxia and low pH water, and distinct phytoplankton community composition 322 compared to the mouth of the system.

323 Results showed that the canal sites generally had higher nutrient (orthophosphate, N+N, 324 silicate) concentrations than the mouth of the canal. Other studies have reported similar patterns 325 of increased nutrients in or near canal structures (Corliss and Trent 1971), especially in the wake 326 of storm events that would presumably deliver nutrients from surrounding urban areas 327 (Wachnicka et al. 2020). Indeed, both canal sites experienced increased nutrient levels following 328 heavy rains that occurred in May 2021 and late June/early July 2021. Interestingly, the canal 329 sites also saw increased ammonium (both sites) and nitrate (interior site only) levels following a 330 freeze in February 2021. From February 14-20, 2021, Corpus Christi experienced record low air 331 temperatures ranging from -8.3 to -1.6  $^{\circ}$ C. The freeze led to fish kills across the Texas coast, 332 with estimates of about 3.8 million fish impacted (TPWD 2021). Dead fish were observed at all 333 three sites in February following the freeze, however more dead fish were observed in the canals 334 (Cutajar, pers. obs.). This may have been due to the canals constricted nature, which caused dead

335	fish to concentrate in them. Previous studies showed that ammonium and phosphate can be
336	remineralized from dead fish in the environment (Vargo et al. 2008; Walsh et al. 2009; Killberg-
337	Thoreson et al. 2014), a phenomenon that has been previously documented in the adjacent
338	Laguna Madre of Texas (Buskey et al. 1997). Aside from these episodic nutrient pulses, several
339	studies from canals of various designs have determined that they often contain organic rich
340	sediments that promote efflux of nutrients into the water column (e.g., Azzoni et al. 2015; Ma et
341	al. 2023). The episodic and/or persistent presence of hypoxic or anoxic conditions in bottom
342	waters of the canal sites is another factor that would enhance ammonium and phosphate release
343	to the water column (Cowan and Boynton 1996; Ma et al. 2006). Furthermore, Macklin et al.
344	(2014) found that coastal residential canals may receive more pronounced discharge of
345	groundwater compared to undisturbed locations. This is relevant because studies in numerous
346	coastal systems (Paerl 1997; Statham 2012; Webb et al. 2018), including in Texas (Douglas et al.
347	2020), have shown that groundwater can be an important source of nutrients to coastal
348	waterbodies. Despite the observations here of episodically high inorganic nutrient
349	concentrations, the overall average inorganic nutrient concentrations were not necessarily
350	indicative of a eutrophied ecosystem. However, data from a limited number of samples showed
351	consistently high dissolved organic nitrogen (DON) concentrations at all three sites here, but
352	especially in the canals (mouth: 38.9 $\pm$ 10.5 $\mu M,$ n=17; middle: 48.9 $\pm$ 12.6 $\mu M,$ n=17; interior:
353	59.1 $\pm$ 19.1 $\mu$ M, n=15). Previous studies have shown that some fraction of marine DON pools is
354	capable of supporting algal growth and overall eutrophication (Bronk et al. 2007; Seitzinger and
355	Sanders 1997), and the concentrations observed in the canals are relatively high compared to
356	many other locations worldwide (Berman and Bronk 2003; Mulholland et al. 2009; Wetz et al.
357	2017).

358	Canal sites also had higher chlorophyll a concentrations than at the mouth, a finding that
359	has been noted elsewhere as well (Maxted et al. 1997). Additionally, the relatively high average
360	chlorophyll <i>a</i> concentration at the canal sites (16.1-19.5 $\mu$ g L <sup>-1</sup> ) as well as the frequent
361	observation of concentrations > 20 $\mu$ g L <sup>-1</sup> offers further support to the notion that the canal sites
362	were affected by nutrient pressures. Results from the PCA showed that chlorophyll a was
363	positively correlated with N+N and ammonium, and negatively correlated with Secchi depth, pH
364	and the ratio of dissolved inorganic nitrogen to phosphate. The negative correlation with Secchi
365	is unsurprising if we consider that algal biomass is likely the main contributor to light
366	attenuation. Likewise, positive correlation with ammonium and N+N points to a role for nutrient
367	availability as an important factor in phytoplankton growth here. Less clear is the negative
368	relationship with the dissolved inorganic nitrogen to phosphate rate. On first glance we would
369	interpret this as implying that phytoplankton biomass is generally higher under low inorganic
370	nitrogen to phosphate ratios, suggestive of selective drawdown of nitrogen and thus consistent
371	with longstanding dogma that nitrogen limits phytoplankton growth in marine waters (Ryther
372	and Dunstan 1971; Paerl 2018). However, additional research would be needed in the study
373	system to determine which nutrient(s) actually limit phytoplankton growth. Regardless,
374	following the February 2021 freeze, a spike in chlorophyll a concentrations was observed at the
375	mouth and middle canal site, suggestive of a role for the nutrients that were released from
376	decaying fish. A previous fish killing freeze event that occurred in 1989 in the Upper Laguna
377	Madre was followed by high ammonium levels and a brown tide algal bloom (Buskey et al.
378	1997). Likewise, chlorophyll a generally increased at the canal sites coincident with the May and
379	late June/early July rain events and associated nutrient spikes. One important factor that we did
380	not measure is residence time, and thus we cannot discount its role. It is well documented that

residential canal systems often have reduced flushing/long residence time due to lack of water

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382 exchange (Goodwin 1991) and that poorly flushed, long residence time conditions can amplify 383 phytoplankton growth (Anderson et al. 2002). 384 Along with having higher chlorophyll a, the canal sites also had distinct phytoplankton 385 community composition compared to the mouth. The mouth was characterized by diatom 386 dominance for much of the year, as is common in estuaries in Texas and elsewhere (Carstensen 387 et al. 2015; Cloern 2018; Chin et al. 2022). The Texas coast, including the Upper Laguna Madre, 388 is exposed to strong winds throughout the year (Carlin et al. 2016; Reisinger et al. 2017; see also: 389 https://windexchange.energy.gov/maps-data/325). Consequently, the water column of Texas 390 estuaries routinely experiences vigorous mixing that favors diatoms through resuspension, 391 nutrient exchange and potentially other ecophysiological factors (Jäger et al. 2008; Depauw et al. 392 2012; Dorado et al. 2015). In contrast, vigorous mixing can negatively affect dinoflagellate 393 populations through such factors as cell damage and by placing them at a competitive 394 disadvantage for nutrients (Smayda 1997; Hall and Paerl 2011). GLM results show that depth 395 and silicate had a negative influence on diatoms, while precipitation was positive. The negative 396 influence of depth seemingly points to resuspension and/or access to sediment-derived nutrients 397 as important factors in the prevalence of diatoms at the study location, as shallower depths led to 398 higher biovolume and vice versa. Along those lines, it must be noted that on average, the mouth 399 was shallower than the canal sites (Cutajar, unpubl. data). The negative influence of silicate can 400 perhaps be interpreted in the context of nutrient uptake preferences, pointing to increased diatom 401 biovolume as silicate is drawn down, whereas the positive influence of precipitation likely 402 indicates that the increased nutrient availability caused by rainfall events may stimulate diatom

403 growth. However, the low explanatory power of the GLM for diatoms indicate that additional

404 factors not quantified here are likely influential as well. In contrast to the mouth, the canal sites 405 exhibited a more diverse phytoplankton community. This diversity may be a result of the 406 pronounced environmental variability that occurs in the canals (Ma et al. 2006; Zhang et al. 407 2017; Kolarova and Napiórkowski 2022), primarily in the form of variable nutrient availability 408 and ratios, episodic rainfall events that seemed to influence the water column in the canals but 409 not at the mouth, and episodic stratification. Dinoflagellates, picoeukarvotes and 410 picocyanobacteria were particularly important contributors to phytoplankton composition at the 411 canal sites. An earlier study conducted in this canal system (Tominack et al. 2022) as well as a 412 study conducted in a dead end canal systems in Delaware (USA; Ma et al. 2006) both 413 documented the prevalence of dinoflagellate blooms. GLM results for dinoflagellates indicate 414 that Secchi depth, orthophosphate and chlorophyll a were positive influences on dinoflagellate 415 biovolume, while D.O. and N+N were negative influences. The positive influences of Secchi 416 depth, orthophosphate, and chlorophyll a seemingly indicate that conditions favoring higher light 417 penetration and increased nutrient availability/overall phytoplankton biomass are conducive to 418 dinoflagellates in the study area. The negative influence of N+N is possibly reflective of the 419 competitive advantage that other, non-dinoflagellate functional groups may have in the presence 420 of oxidized nitrogen forms (Li et al. 2009; Dagenais-Bellefeuille and Morse 2013). The negative 421 influence of D.O. in unlikely to suggest a negative physiological impact on dinoflagellates, as 422 suboxic conditions were not pronounced in the surface waters where phytoplankton samples 423 were collected. One possibility is that it may simply point to a relationship derived from the 424 relatively higher respiratory potential of dinoflagellates that leads to lower D.O. levels. 425 Picoeukaryotes have been shown to respond to pulsed inputs of oxidized N (Bec et al. 2005; Ivey 426 et al. 2020), which may explain their prevalence at the canal sites during the rainy period in

427 summer 2021. GLM results are consistent with a role for nutrients in picoeukaryote growth, showing a positive influence of N+N concentration as well as site (with higher concentrations in 428 429 the interior and middle sites), and negative influence of orthophosphate and DIN:Si. Incidentally, 430 the picoeukaryotes tended to be more prevalent when dinoflagellates were less prevalent, lending 431 support to the hypothesis that nitrogen form played a role in shaping community dynamics in the canal. Aside from nutrient factors, the influence of site may highlight unique characteristics of 432 433 the canal sites, such as long residence time, that may be influential on the picoeukaryotes. 434 Picoplankton in general are known to be favored by reduced flushing (Philps et al. 1999; 435 Badylak and Phlips, 2004). Picocyanobacteria are typically associated with warm temperatures 436 and can be associated with a range of nutrient conditions (e.g., Kuosa 1991; Gaulke et al. 2010; 437 Wang et al. 2011). GLM results showed a negative influence of the winter season, but no clear 438 temperature effect otherwise. Winter season and the overall period of relatively cool temperatures (<20°C) are short in the study region, suggesting that any temperature effect is 439 440 limited to this narrow time window compared to more temperate systems when cooler waters 441 may persist from fall through spring. A slight positive influence of salinity suggests a preference 442 of picocyanobacteria for higher salinity and/or lower inflow conditions (Phlips et al. 1999). The 443 positive influence of N+N and negative influence of DIN:Si points to nutrient availability and the 444 relative availability of nitrogen vs. phosphorus as affecting picocyanobacteria in the study area. 445 Euglenoids were briefly a dominant functional group at the middle canal site in late June when 446 salinities were still relatively high at the site. Euglenoids are often associated with high nutrient 447 concentrations (Nunes et al. 2018), but may also be susceptible to hydrologic displacement 448 (Roelke et al. 2013). We speculate that the rain in May could have created favorable nutrient 449 conditions for the euglenoids, yet a lag phase was required for them to achieve dominance of the

450	community. Indeed, euglenoid biomass decreased following increased rainfall in early July,
451	suggestive of a "wash out" effect (Tan et al. 2019). Unfortunately, frequent "non-detects" for
452	euglenoids prevented establishment of linear models to quantitatively assess influential
453	environmental factors.
454	One of the two physico-chemical indicators of ecosystem health, pH, displayed little
455	difference between sites whereas the other, D.O., showed obvious differences. pH was generally
456	within a range expected for a waterbody with salinities at or near seawater on average.
457	Ephemeral low pH (<7.20) was observed on May 2 <sup>nd</sup> , 2021, at both the mouth and interior site,
458	but only at the surface. An algal bloom was in place at the interior site but not at the mouth. This
459	event also coincided with rainfall that led to a sharp decrease in salinity at the interior site, while
460	salinity at the mouth site was minimally affected. Thus, while the cause of the low pH at the
461	interior canal site may be attributed to an influx of low pH runoff as well as respiration during
462	the algal bloom, it is not clear what caused the low pH at the mouth. Another instance of low pH
463	(7.19) was observed at the bottom of the interior site on July 9 <sup>th</sup> , 2021, which coincided with
464	both stratification and low D.O. (1.18 mg/L), suggestive of the influence of respiratory
465	processes. Regardless of cause, this combination of low pH and low D.O. is indicative of a
466	"multi-stressor" situation that has potential to be harmful to shell-forming organisms in particular
467	(Clark and Gobler 2016; Tomasetti et al. 2018). In contrast to pH, D.O. was generally lower at
468	the canal sites compared to the mouth. The PCA results show a negative correlation of D.O. with
469	temperature, consistent with known solubility effects of temperature on D.O. A slightly lower PC
470	loading score was obtained for D.O. on the inflow axis, where D.O. had a positive correlation
471	with salinity. This could be interpreted as showing negative effects of low salinity events on
472	D.O., perhaps due to organic matter loadings and increased respiration that would result from

473	those events. Only one instance of hypoxia was detected at the surface, which occurred on
474	September 3 <sup>rd</sup> , 2021, at the interior site. On that day, salinity was relatively high, stratification
475	was not evident, and chlorophyll $a$ was low. Thus, the proximate cause cannot be established. In
476	contrast to the surface, hypoxia occurred regularly at the bottom of the canal sites, especially at
477	the interior site. Causes of the March and April 2021 hypoxic events are unknown, as salinity
478	was 31.3-32.6 and thus solubility effects that are common with hypersalinity would not be
479	expected, stratification was not evident, and chlorophyll <i>a</i> was low. One possibility is that these
480	events may represent a lagged response to earlier high chlorophyll a events at those sites,
481	resulting in low D.O. that developed as water temperature and microbial respiration of the algal
482	biomass increased (e.g., Rabalais et al. 2001). In early-mid May at the interior canal site,
483	stratification was present and chlorophyll a was high early in the period, pointing to degradation
484	of the algal bloom and respiratory processes in general in the bottom water as a driver of the low
485	D.O. For the period of early July through mid-October when hypoxia was remarkably persistent
486	at the interior site, a combination of factors likely were at play including temperatures that
487	consistently exceeded 25°C, chlorophyll <i>a</i> that was frequently >20 $\mu$ g/L, and ephemeral
488	stratification that occurred coincident with rain events.

489

### 490 Management recommendations

491 Results from this study demonstrate that the canal system of North Padre Island, Texas, 492 does indeed contain unique water quality and phytoplankton community characteristics relative 493 to surrounding waterbodies, with various water quality indicators pointing to a susceptibility to 494 poor water quality conditions (high nutrient levels, high phytoplankton biomass, low dissolved 495 oxygen) in the canals. To ameliorate the poor water quality conditions now or in the future,

496 several options are available to local communities. For existing canals, mitigation strategies for 497 hypoxia/anoxia have been proposed/employed, primarily in the form of fountains and other 498 aeration devices that help oxygenate the system and reduce the stratification observed during 499 rainfall events (Maxted et al. 1997; Koweek et al. 2020; Ma et al. 2023). Nutrient management is 500 also paramount, but to be effective it requires knowledge of nutrient sources. In canal systems 501 that are being noticeably impacted by nutrient pollution, source tracking studies with isotopic 502 tracers are recommended (Bannon et al. 2008). Once nutrient sources are determined, best 503 management practices can be applied. For example, if a canal is impacted by sewage, action 504 should be taken to address failing septic tanks and/or sewer lines. In canals affected by runoff of 505 fertilizer, a fertilizer ordinance could be applied and has helped to reduce total nitrogen and 506 phosphorus levels in some South Florida coastal waterbodies following enactment of the 507 ordinance (Motsch 2018). Other best management practices, such as planting vegetative buffers, have been effective in reducing nitrogen inputs to river-fed agricultural canals (Castaldelli et al. 508 509 2015), but a similar logic could be applied to estuarine-fed residential canals. Alternative canal 510 designs should also be considered. For example, a previous study that evaluated dead end canals 511 in Maryland and Delaware recommended that canal systems be designed with features such as 512 rounded corners and ample connection with nearby waterbodies to enhance flushing rates 513 (Maxted et al., 1997). Other studies recommend that canals be situated so that the canal axis is 514 parallel to wind patterns to allow for improved flow and water quality (Lindall and Trent, 1975; 515 Marvin et al., 1990). Ultimately, coastal residential canals are growing worldwide in their areal 516 fingerprint, requiring additional attention from a monitoring, resource management and 517 environmental engineering standpoint if we are to ensure that these waterbodies can support

- 518 healthy aquatic communities and meet the needs of surrounding communities from a human
- 519 health, aesthetic and recreational use standpoint.

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### 750 FIGURE LEGENDS

- Figure 1: Map of study sites. Green dot in inset indicates location of Corpus Christi, Texas
  (USA).
- 754 Figure 2: Surface water temperature (°C) (A) and surface salinity (B) over time.
- Figure 3: Surface pH (A), bottom pH (B), surface D.O. (mg/L) (C) and bottom D.O. (mg/L) (D) over time.
- Figure 4: N+N (A), ammonium (B), orthophosphate (C), and silicate (D) over time. All
  concentrations are μM.
- 762 Figure 5: Principal Components Analysis of environmental variables collected during this study.
- Figure 6: Chlorophyll a (µg/L) over time.
- Figure 7: Percentage contribution of functional groups to phytoplankton community biovolume.
- 767 (Top) Mouth, (Center) middle, and (Bottom) interior sites.

Table 1: Mean ± Standard Deviation of environmental variables or percent contribution for each

roup of phytoplankton to total biovolume for each site. Different letters (a.b.c) indicate

significant differences between sites, as determined by Wilcox post-hoc tests. Variables with the
same letter are not significantly different. \*Sample size too small, or all observations in the same
site.

Variable	Mouth		Middle		Interior	
Surface Temp. (°C)	$25.0\pm4.9$	a	$25.7\pm5.1$	a	$25.4\pm5.0$	a
Surface Salinity	$30.4 \pm 4.5$	a	$29.1\pm6.0$	а	$26.5\pm8.2$	a
Surface pH	$8.12\pm0.33$	а	$8.15\pm0.19$	а	$7.99 \pm 0.29$	а
Bottom pH	$8.19\pm0.21$	a	$8.12\pm0.19$	а	$7.94 \pm 0.23$	b
Surface DO (mg/L)	$5.38 \pm 1.26$	a	$5.03 \pm 2.23$	a	$4.02 \pm 1.57$	b
Bottom DO (mg/L)	$4.70 \pm 1.34$	a	$2.67\pm3.77$	b	$1.98\pm2.67$	b
Secchi depth (m)	$0.65\pm0.19$	a	$1.04\pm0.27$	b	$0.68\pm0.19$	a
N+N (µM)	$0.36\pm0.20$	a	$2.64 \pm 2.87$	b	$3.50\pm3.65$	b
Ammonium (µM)	$3.79 \pm 1.86$	а	$5.64 \pm 4.13$	a	$7.52\pm6.51$	a
Orthophosphate (µM)	$0.20\pm0.14$	а	$0.88\pm0.87$	b	$2.83 \pm 4.70$	b
Silicate (µM)	$53.4\pm31.8$	a	$73.9 \pm 34.0$	b	$130\pm61.9$	с
DIN:PO <sub>4</sub>	$42.1\pm47.1$	a	$28.4\pm44.8$	а	$8.1 \pm 7.2$	b
DIN:Si	$0.1 \pm 0.1$	a	$0.2\pm0.2$	а	$0.1 \pm 0.2$	a
Chlorophyll (µg L <sup>-1</sup> )	$6.4 \pm 3.3$	a	$16.1 \pm 10.6$	b	$19.5\pm9.0$	b
Diatoms (%)	$63.7\pm22.0$	a	$23.5 \pm 27.7$	b	$26.2\pm28.5$	b
Dinoflagellates (%)	$17.3\pm16.3$	a	$26.3\pm26.9$	а	$12.6\pm18.8$	a
Cryptophytes (%)	$2.5 \pm 2.9$	a	$5.2\pm9.9$	а	$6.2 \pm 11.3$	a
Euglenoids (%)	$0.4 \pm 0.4$	a	$6.8\pm17.4$	b	$2.2 \pm 3.7$	a
Picoeukaryotes (%)	$9.0\pm6.4$	а	$25.8\pm24.1$	b	$37.9\pm26.3$	с
Picocyanobacteria (%)	$5.3 \pm 5.3$	а	$12.0\pm11.7$	a	$10.8\pm8.8$	a
Chlorophyte (%)	$0.0\pm0.0$	а	$0.0\pm0.2$	a	$3.5\pm11.0$	a
Cyanobacteria (%)	$1.6 \pm 6.8$	a	$\overline{0.0\pm0.0}$	a	$0.0 \pm 0.0$	a
Raphidophyte (%)	$0.1 \pm 0.3$	a	$0.2 \pm 0.6$	a	$0.6 \pm 2.3$	a
Silica Flagellate (%)	$0.1 \pm 0.2$	a	$0.1 \pm 0.3$	a	$0.0 \pm 0.2$	a



Figure 1.









Figure 4.











Figure 7.

Highlights

- Residential canals are an increasingly common feature of coasts worldwide.
- Canals possess attributes that may lead to water quality degradation.
- Hypoxia, high nutrients and chlorophyll were frequently observed in a Texas canal.
- Conditions in the canal favored a unique assemblage of phytoplankton.

### Author CRediT statement – Cutajar et al.

Jordana Cutajar: Conceptualization, Methodology, Software, Validation, Formal analysis, Investigation, Data curation, Writing – Original draft, Visualization, Project administration Blair Sterba-Boatwright: Formal analysis, Writing – Review and editing Michael Wetz: Conceptualization, Methodology, Validation, Writing – Review and editing, Resources, Supervision, Funding acquisition

### **Declaration of interests**

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

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

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