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Daisuke Goto, William G. Wallace. Altered feeding habits and strategies of a benthic forage fish () in chronically polluted tidal salt marshes. *Marine Environmental Research*, 2011, 72 (1-2), pp.75. 10.1016/j.marenvres.2011.06.002 . hal-00720184

HAL Id: hal-00720184

<https://hal.science/hal-00720184>

Submitted on 24 Jul 2012

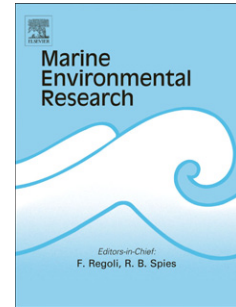
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Accepted Manuscript

Title: Altered feeding habits and strategies of a benthic forage fish (*Fundulus heteroclitus*) in chronically polluted tidal salt marshes

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PII: S0141-1136(11)00070-5

DOI: [10.1016/j.marenvres.2011.06.002](https://doi.org/10.1016/j.marenvres.2011.06.002)

Reference: MERE 3531

To appear in: *Marine Environmental Research*

Received Date: 14 August 2009

Revised Date: 18 March 2011

Accepted Date: 8 June 2011

Please cite this article as: Goto, D., Wallace, W.G. Altered feeding habits and strategies of a benthic forage fish (*Fundulus heteroclitus*) in chronically polluted tidal salt marshes, *Marine Environmental Research* (2011), doi: 10.1016/j.marenvres.2011.06.002

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1 **Altered feeding habits and strategies of a benthic forage fish (*Fundulus***
2 ***heteroclitus*) in chronically polluted tidal salt marshes**

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46 **Abstract.** Responses in feeding ecology of a benthic forage fish, mummichogs (*Fundulus*
47 *heteroclitus*), to altered prey resources were investigated in chronically polluted salt marshes (the
48 Arthur Kill–AK, New York, USA). The diet niche breadth of the AK populations of
49 mummichogs was significantly lower than that of the reference population, reflecting reduced
50 benthic macroinfaunal species diversity. Most of the AK populations also had two to three times
51 less food in their gut than the reference population. This disparity in gut fullness among the
52 populations appeared to be partly due to ingested prey size shifts; some of the AK populations
53 ingested fewer large prey than the reference population. Furthermore, benthic assemblages were
54 strongly associated with sediment-associated mercury; gut fullness of the AK populations also
55 significantly decreased with increasing mercury body burdens. These results indicate that chronic
56 pollution may have directly (chemical bioaccumulation) and indirectly (reduced prey
57 availability) altered the feeding ecology of mummichogs.

58
59 **Keywords** Trophic relationships; benthic environment; feeding strategy; pollution effects; diet
60 analysis; salt marshes; the Hudson River

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69 **1. Introduction**

70 Urban coastal habitats are often highly degraded due to a variety of human activities (e.g.,
71 accidental chemical discharges and dredging) (Able et al., 1998; Levin et al., 2001). Coastal
72 habitat degradation can alter food web structures and ecosystem functions including
73 biogeochemical cycling of materials from terrestrial, freshwater, and marine environments
74 (Levin et al., 2001). Among anthropogenic stressors, elevated levels of chemical pollutants can
75 have particularly persistent effects in aquatic organisms (Boesch et al., 2001). As the majority of
76 these anthropogenically introduced chemicals are bound to bottom sediments, benthic organisms
77 are especially vulnerable (Clements and Newman, 2002). Furthermore, since these benthic
78 organisms can also exhibit species-specific differential tolerance to a variety of chemical
79 pollutants, chemical pollution may ultimately alter the community structure and trophic
80 dynamics of benthic habitats (Clements and Newman, 2002).

81 In chronically degraded habitats, chemical pollutants can influence foraging behaviors (e.g.,
82 prey capture) of fish (Smith and Weis, 1997; Weis et al., 2001). Due to logistical difficulties,
83 however, it is often difficult to elucidate how altered foraging behaviors affect the feeding habits
84 of wild populations of fish (Clements and Newman, 2002; Weis et al., 2001). As both predator
85 and prey are often concurrently exposed to chemical pollutants in natural habitats, species-
86 specific differential sensitivities to these stressors can result in a variety of altered (often
87 unpredictable) predator-prey relationships (Livingston, 1984; Steimle et al., 1993).

88 Chemical pollutants can also indirectly restructure the biological assemblage, and these
89 indirect effects are often mediated through trophic relationships (Eby et al., 2005; Jeffree and
90 Williams, 1980). Deposit-feeding invertebrates, a common trophic group in coastal habitats, are
91 an important prey resource for fishes, shellfishes, and wading birds; anthropogenic impacts on

92 these invertebrates can thus have cascading effects on their predators (Pinnegar et al., 2000).
93 Direct impacts of habitat degradation on marine benthic macroinfaunal assemblages have been
94 extensively investigated (Boesch, 1982). However, there have been relatively few attempts to
95 examine indirect effects of pollution, including the feeding habits of benthivorous predators in
96 marine and estuarine ecosystems (e.g., Hinz et al., 2005; Powers et al., 2005). As a result, there is
97 little mechanistic understanding of pollution-induced alterations in trophic structure and
98 ecosystem function (Livingston, 1984; Rose, 2000).

99 In the present study, the impacts of habitat degradation on benthic trophic interactions were
100 investigated by examining the feeding habits and strategies of mummichogs, *Fundulus*
101 *heteroclitus* (Pisces: Cyprinodontidae), in chemically polluted tidal salt marshes in New York,
102 USA. Mummichogs are among the most abundant and productive fishes in tidal creeks and salt
103 marshes in the northwest Atlantic coast (Meredith and Lotrich, 1979). Mummichogs thus play an
104 essential role in salt marsh food webs as an intermediate predator linking benthic invertebrates,
105 and piscivorous fishes and wading birds (Kneib, 1986). Within salt marshes, mummichogs also
106 exhibit a distinct ontogenetic shift in habitat use from vegetated intertidal pools and creeks
107 during larval and juvenile stages to primarily subtidal creeks as an adult (Kneib, 1997; Teo and
108 Able, 2003). Subtidal food resources may, however, not be sufficient or easily accessible to adult
109 mummichogs (Weisberg and Lotrich, 1982; Weisberg et al., 1981). In foraging for sufficient
110 prey resources, adult mummichogs thus regularly migrate from subtidal marshes to intertidal
111 marsh surface following tidal cycles (Kneib, 1997; Teo and Able, 2003).

112 Because of their ontogenetic tidal migration and strong site fidelity within salt marshes (Able
113 et al., 2006; McMahon et al., 2005), mummichogs are a key species in transporting secondary
114 production of salt marsh ecosystems in the northwest Atlantic coast (Kneib, 1997). A subtidal

115 component of these salt marshes are frequently used as a feeding ground by juvenile and adult
116 migrant fish and shellfish species such as blue crab (*Callinectes sapidus*) and striped bass
117 (*Morone saxatilis*), which are potential predators of mummichogs (Nemerson and Able, 2003;
118 Peters and Schaaf, 1991). These migrant fishes and shellfishes consequently transport marsh
119 production incorporated by mummichogs to open coastal ecosystems (Kneib, 1997). Moreover,
120 through these trophic interrelationships, tidal migration of mummichogs from intertidal marsh
121 surface to subtidal creeks may also couple spatially semi-compartmentalized trophic components
122 (i.e., benthic invertebrates in intertidal marshes and migrant piscivores in subtidal creeks and
123 open bays) (Kneib, 1997; Weisberg et al., 1981). This cyprinodontid fish thus plays a critical role
124 in integrating and transporting a substantial portion of marsh productions to adjacent coastal
125 ecosystems (Kneib, 1997; Smith et al., 2000).

126 Relatively high tolerance to a variety of natural and anthropogenic stressors often allows
127 mummichogs to thrive even in highly polluted coastal habitats (Schulte, 2007; Weis, 2002). Due
128 to their opportunistic feeding habits, omnivorous predators such as mummichogs can also be an
129 integrative biological surrogate ‘sampler’ for the assessment of benthic habitat quality (e.g., Frid
130 and Hall, 1999; Link, 2004). Although mummichogs have frequently been used to assess
131 structural and functional recovery of restored salt marshes (e.g., Moy and Levin, 1991; Wozniak
132 et al., 2006), there are few studies assessing the trophic ecology of mummichogs in chemically
133 polluted habitats (e.g., Smith and Weis, 1997; Weis et al., 2001). Since mummichogs are an
134 important trophic link in urban coastal habitats, alterations in their relationships with prey (e.g.,
135 local elimination of benthic invertebrate species) may disrupt trophic coupling in urban tidal salt
136 marshes (Kneib, 2003). This decoupling of food chains within urban tidal salt marshes induced
137 by chemical pollution could disrupt secondary production and energy transport in coastal

138 habitats. Understanding the effects of habitat degradation on the trophic ecology of mummichogs
139 is, therefore, highly relevant to the structural and functional assessments of urbanized salt
140 marshes (Moy and Levin, 1991).

141

142 **2. Materials and methods**

143 *2.1. Study area*

144 The study was conducted in the Arthur Kill–AK (part of the New York/New Jersey–NY/NJ
145 Harbor Estuary Complex), which is a tidal waterway located between northern New Jersey and
146 Staten Island, New York, connecting Newark Bay and Kill van Kull to the north, and Raritan
147 Bay, New York Harbor, and the Atlantic Ocean to the south (Fig. 1). Due to the proximity to
148 industrial facilities, petroleum refining facilities, heavy shipping traffic, and sewage treatment
149 plants, AK is one of the most severely polluted areas in the NY/NJ Harbor Estuary system
150 (Steinberg et al., 2004). A variety of anthropogenic disturbances have resulted in a considerable
151 loss of biodiversity in this region over the years (Steinberg et al., 2004). Among the
152 anthropogenic stressors, chemical pollutants released with industrial and municipal discharges
153 have been major contributors to the deterioration of trophic structure and ecosystem function in
154 this region (Bopp et al., 2006). Since tides enter from both ends of AK, this region is very poorly
155 flushed, resulting in enhanced entrapment and accumulation of pollutants in sediments (Bopp et
156 al., 2006). Due to potential toxicity and transference up food chains (Goto and Wallace, 2009),
157 chemical pollution is a major concern in this coastal ecosystem (Bain et al., 2007; Steinberg et
158 al., 2004).

159 Despite recent improvements in water quality of the NY/NJ Estuary Complex including AK
160 (Bopp et al., 2006; Steinberg et al., 2004), residual impacts of historic pollution can still be

161 observed at many areas in the NY/NJ Harbor, including altered benthic assemblages (Bain et al.,
162 2007). As the NY/NJ Harbor Estuary also provides essential habitats for resident and migrant
163 aquatic as well as terrestrial species (Able et al., 1998; Steinberg et al., 2004), predators ingesting
164 benthic organisms are still at risk of exposure to these pollutants (Steinberg et al., 2004).

165

166 2.2. *Field sampling*

167 Sampling sites chosen for this study were four tributaries of AK; Richmond Creek, Main
168 Creek, Neck Creek, and Mill Creek, and one regional reference site (near Raritan Bay) external
169 to AK; Lemon Creek (Fig. 1). Neck Creek, which is located on the northwestern Staten Island,
170 NY, is polluted with various chemicals due to its proximity to many oil refineries as well as
171 abandoned industrial facilities. Richmond Creek and Main Creek are tributaries of Fresh Kills
172 Complex, which is located on the western shore of Staten Island, and surrounded by the Fresh
173 Kills landfill. Mill Creek, on the southwestern shore of Staten Island, is located near the
174 confluence of AK and Raritan Bay and has been polluted from historic smelting activities.
175 Lemon Creek, which is located in the southeastern shore of Staten Island, is one of a few tidal
176 salt marshes that are relatively unaffected by human activities in the region.

177 Sampling of mummichogs was conducted in an intertidal area of each site during the summer
178 (July to September) of 2004 (two to three times per month). Mummichogs were collected using a
179 3.05 x 1.22 m-seine net with 6 mm-polyester mesh or unbaited cylindrical minnow trap with 6
180 mm-mesh. The sampling was continued until a representative size of fish population at each site
181 ($n = 100$ to 200) were captured. To maximize sampling efficiency and minimize the digestion of
182 gut contents, fish were collected within approximately 60 to 90 minutes after daytime high tides,
183 which is the peak of mummichogs' daily foraging activity (Weisberg et al., 1981). When using

184 minnow traps, fish were removed every 15 to 30 minutes to minimize the digestion of gut
185 contents (Beauchamp et al., 2007). Fish samples for diet analysis were immediately placed on ice
186 upon capture and then fixed with 10% formalin in the field (Bowen, 1996). Additional fish
187 samples for mercury (Hg) analyses were also collected and transported on ice to laboratory.

188 Benthic macroinvertebrates were also collected to assess prey availability for mummichogs.
189 Macroinvertebrates were sampled by taking bulk surficial (top ~5 cm) sediments (one replicate
190 per a randomly chosen 100 m² area) from the intertidal marsh at each study site, using a 15 x 15
191 cm-quadrat sampler ($n = 5$ per site). Each sample was sieved through a 500 μ m-mesh and fixed
192 with 10% formalin stained with Rose Bengal in a glass jar in the field, which was then
193 transported to the laboratory. Additional sediment samples were collected for analyses of trace
194 metals, grain size, and total organic carbon (TOC) content. Surface water temperature, pH,
195 salinity, and dissolved oxygen (DO) were also measured within the proximity of each replicate
196 sample of benthic macroinvertebrates.

197

198 2.3. Laboratory sample processing: *Mummichogs*

199 In the laboratory, fish samples for diet analysis remained in formalin for at least one week
200 (Bowen, 1996). Samples from the different sampling dates were pooled for each site. Each fish
201 was first sexed, measured (total length ± 0.01 mm), and weighed (wet weight ± 0.01 g). Then,
202 the entire intestinal tract was excised and preserved individually in 70% ethanol (Bowen, 1996).
203 Only the contents of the anterior 2/3 of intestines (equivalent to 'stomach') (Babkin and Bowie,
204 1928) were removed from the intestine, blotted dry, and placed on a pre-weighed petri dish to
205 weigh. All prey items were identified to the lowest practical taxonomic level under a dissecting

206 microscope using available species keys in literature (Fauchald, 1977; Weiss, 1995; Merritt and
207 Cummins, 1996; Pollock, 1998).

208 The diet composition was quantified by frequency of occurrence, count, and weight of each
209 diet item (Bowen, 1996). To estimate the percentage contribution by weight for small prey (e.g.,
210 meiofauna) and particulate matter (e.g., detritus) items, a modified version of the method by
211 Penczak (1985a) was used. Briefly, once identified, each diet item was separately squashed to a
212 uniform depth (assuming $1 \text{ mm}^3 = 1 \text{ mg}$ wet weight) between glass slides (Penczak, 1985a).
213 Then, to increase accuracy, the area of each diet item was captured with a digital camera through
214 a dissecting microscope (Bowen, 1996) and measured ($\pm 0.001 \text{ mm}^2$) using a digital imaging
215 software (Motic Images Plus 2.0 ML). The total reconstructed weight estimates of diet items
216 were then compared with the actual weight of gut contents to ensure the accuracy of the method
217 (Bowen, 1996).

218 Fish samples for Hg analysis ($n = \sim 30$ per site) were allowed to depurate gut contents for at
219 least 24 hours in filtered 15‰ seawater and euthanized by an overdose of MS222. Then, fish
220 samples were homogenized with a Polytron tissue homogenizer (Kinematica, Inc., Switzerland)
221 in NANOpure[®] (reagent-grade) water. Homogenized samples were oven-dried ($\sim 60 \text{ }^\circ\text{C}$) for 24 to
222 48 hours. Oven dried samples were processed and analyzed for mercury, as described below.

223

224 2.4. Laboratory sample processing: Benthic macroinvertebrates

225 Benthic samples remained in formalin for at least seven days and then were preserved in 70%
226 ethanol until species identification. Macroinvertebrates were identified to the lowest possible
227 taxonomic category (mostly to the species level) under a stereo microscope. Samples for each

228 species were counted to estimate its abundance and subsequently dried in an oven at 60 °C to a
229 constant weight for approximately two hours to estimate its biomass.

230

231 2.5. Laboratory sample processing: Sediments

232 Sediment samples for grain size, TOC, and trace metal analyses were sieved through a 500
233 µm-mesh to eliminate large objects (e.g., detritus) and macroinvertebrates. Subsamples of sieved
234 sediment samples (< 500 µm) were sieved further through a 73 µm-mesh and oven-dried (at 80
235 °C for ~48 to 72 hours) to estimate % fine sediment particles (< 73 µm). The remaining sieved
236 sediment subsamples (< 500 µm) were homogenized, oven-dried (at 60 °C for ~48-72 hours),
237 and then divided for TOC and trace metal analyses. TOC was determined by combustion of
238 dried, ground sediments with a benchtop muffle furnace (Type 1400, Barnstead International) at
239 550 °C for six hours.

240 Oven-dried (60 °C) subsamples of sediments were divided further into two subsamples [one
241 for silver (Ag), cadmium (Cd), copper (Cu), nickel (Ni), lead (Pb), and zinc (Zn); one for Hg].
242 For Ag, Cd, Cu, Ni, Pb, and Zn, subsamples were digested under reflux with concentrated Trace
243 Metal Grade nitric acid overnight at room temperature (~20 °C) and subsequently on a hotplate
244 (~80 °C) for ~48 to 72 hours. Once digested, samples were evaporated to dryness, re-suspended
245 in 2% nitric acid, and filtered. Metal concentrations in samples were analyzed with graphite
246 furnace atomic absorption spectrometer (for Ag, Cd, Cu, Ni, and Pb) or flame atomic absorption
247 spectrophotometer (for Zn) (3100AAS, Perkin Elmer, Inc.).

248 For Hg analysis of sediments as well as mummichogs, dried subsamples were weighed and
249 transferred to an acid-washed 60 ml-BOD bottle. Samples were digested with a mixture (1:4 v/v)
250 of concentrated Trace Metal Grade nitric and sulfuric acids in a water bath (~60 °C) for three

251 hours and cooled in a refrigerator (US EPA, 1992). Samples were then oxidized with 5%
252 potassium permanganate and 5% potassium persulfate at room temperature overnight (US EPA,
253 1992). The digested samples were transferred into a 100 ml-volumetric flask, and 12% sodium
254 chloride-hydroxylamine chloride was added to reduce excess potassium permanganate (US EPA,
255 1992). Samples were then brought to final volume with NANOpure[®] (reagent-grade) water,
256 filtered, and analyzed with cold vapor atomic absorption spectrometer (Flow Injection Mercury
257 System, FIMS 100, Perkin Elmer, Inc.) using stannous chloride as a reducing reagent. Finally,
258 measured concentrations of sediment-associated Ag, Cd, Cu, Ni, Pb, Zn, and Hg were compared
259 with the Effects Range Median (ERM) levels proposed by Long and Morgan (1990), which
260 suggest the threshold levels of trace metals for adverse effects in marine and estuarine
261 organisms.

262

263 *2.6. Mummichog diet data analyses: Univariate measures*

264 Population-specific characteristics of the diet composition of mummichogs were first examined
265 using the following univariate measures; gut fullness index (= 100 x weight of gut
266 contents/weight of fish) (Weisberg et al., 1981), prey species richness [Margalef's index, $d = (S -$
267 $1)/\log_e N$; S = the total number of prey species, N = the total number of individuals] (Clifford and
268 Stephenson, 1975), diet niche breadth [Shannon-Wiener diversity index, $H' = -\sum p_i \cdot \log_e(p_i)$; p_i
269 = the proportion of the abundance of the i th prey species (Shannon and Weaver, 1949), as
270 suggested by Marshall and Elliott (1997)], and prey species evenness [Heip's evenness = $(e^{H'} -$
271 $1)/(S - 1)$] (Heip, 1974). The representativeness of diet composition derived from the fish
272 samples examined was evaluated using a species-accumulation curve, which was based on a
273 cumulative number of species in the samples taken in a random order with 9999 permutations

274 (Clarke and Gorley, 2006). The sample size was considered adequate if the species-accumulation
275 curves reached an asymptote.

276 The univariate measures were transformed using a logarithmic transformation, where
277 necessary. The normality of data was tested using Shapiro-Wilk's W test. The homogeneity of
278 variance was tested using Levene's test. The statistical significance of differences in the
279 univariate indices among populations was tested using one-way analysis of variance (ANOVA),
280 which, if significant, was followed by Tukey's honestly significant difference (HSD) test. A
281 relationship between the body size of grass shrimp (a dominant prey item) ingested and fish total
282 length was tested using least squares regression, which was fitted with the Levenberg-Marquardt
283 algorithm. All univariate statistical analyses were performed using Statistica 7.1 (Statsoft, Inc,
284 USA).

285 Population-specific relative importance of each diet item (i) was evaluated by the index of
286 relative importance (IRI_i) (Bowen, 1996). In this study, the diet data were pooled into 14 diet
287 categories, according to their taxonomic relationships (mostly at the class level) for IRI . IRI_i is
288 based on percentage contributions by frequency of occurrence ($\%FO_i$), count ($\%N_i$), and weight
289 ($\%W_i$) [$IRI_i = \%FO_i \times (\%N_i + \%W_i)$] (Cortés, 1997), which was adjusted to percentage for each
290 prey item ($\%IRI_i = 100 \times IRI_i / \sum IRI_i$), as suggested by Cortés (1997).

291 Population-specific prey selectivity was estimated using Chesson's index, α , which is based on
292 the relative occurrence of the i th prey item in the environment (p_i) and in the gut content (r_i) [α_i
293 $= (r_i/p_i) / \sum (r_i/p_i)$; $i = 1, \dots, m$; $0 \leq \alpha_i \leq 1$] (Chesson, 1978). Inference of prey selectivity is
294 based on a critical value for no or neutral selection that is defined as $\alpha_{\text{neutral}} = 1/m$, where m = the
295 number of diet categories; if $\alpha_i > 1/m$, prey i is actively selected by a predator, while if $\alpha_i < 1/m$,
296 prey i is avoided by a predator (Chesson, 1978). In this study, α_i was calculated for eight ($m = 8$,

297 i.e., $\alpha_{\text{neutral}} = 0.125$) benthic macroinvertebrate taxonomic classes; Amphipoda, Isopoda,
298 Cirrledia, Polychaeta, Oligochaeta, Gastropoda, Bivalvia, and Insecta (aquatic insects only). In
299 calculating α_i , the diet data were pooled for length classes, 60+ mm, 70+ mm, and 80+ mm at
300 each site, as only these classes were reasonably represented at all sites.

301

302 2.7. *Mummichog* diet data analyses: Multivariate analyses

303 The statistical significance of differences in count and weight of diet composition among sites
304 and length classes was tested using two-way crossed analysis of similarity (ANOSIM) with
305 randomization permutation ($n = 9999$) at the significance level of 5%, which, if significant, was
306 followed by pairwise comparisons (Clarke and Gorley, 2006). Two-way crossed analysis of
307 similarity percentages (SIMPER) was then used to identify dominant diet categories contributing
308 to dissimilarity among sites and length classes (Platell et al., 1998). In ANOSIM and SIMPER,
309 the diet data for length classes, 60+ mm, 70+ mm, and 80+ mm were used. All analyses of
310 ANOSIM and SIMPER were performed using PRIMER v6.1.13 (PRIMER-E Ltd, Plymouth,
311 UK).

312

313 2.8. *Statistical analyses of relationships between environmental variables and benthic*

314 *macroinfaunal prey assemblages*

315 To examine relationships between benthic macroinfaunal prey assemblages and
316 physicochemical properties of the environment, canonical correspondence analysis (CCA) with
317 Hill's scaling on inter-species distances was used (ter Braak and Šmilauer, 2002). Prior to CCA,
318 all assemblage data were transformed using log-transformation [$\log_e(x + 1)$ for the abundance
319 data and $\log_e(x + 0.001)$ for the biomass data] (McCune and Grace, 2002), while all the

320 environmental variables were normalized (i.e., mean = 0, standard deviation =1) (ter Braak and
321 Šmilauer, 2002). The statistical significance of all canonical axes was tested using a Monte Carlo
322 permutation test (999 unrestricted permutations under reduced model at $p < 0.05$) (ter Braak and
323 Šmilauer, 2002). Redundant environmental variables with Variance Inflation Factor (VIF) > 20
324 (an indication of multicollinearity) were removed from further analyses (ter Braak and Šmilauer,
325 2002). The importance of environmental variables was assessed using intraset correlation with
326 the CCA axes (ter Braak, 1986). The relative importance of environmental variables after
327 removal of redundant variables was ranked using forward selection with Monte Carlo
328 permutation tests (999 unrestricted permutations under full model at $p < 0.05$) (ter Braak and
329 Šmilauer, 2002). All CCAs were performed using CANOCO 4.5 (Microcomputer Power, New
330 York, USA).

331

332 *2.9. Statistical analyses of potential effects of Hg on mummichogs*

333 To assess potential effects of Hg on mummichogs, first, the statistical significance of
334 differences in Hg whole body burdens among populations was tested using one-way ANOVA,
335 which, if significant, was followed by Tukey's HSD test. Direct effects of Hg bioaccumulation
336 on the feeding habits of mummichogs were then examined by testing a relationship between Hg
337 whole body burdens and gut fullness index using least squares regression.

338

339 **3. Results**

340 *3.1. Univariate comparisons of mummichog diet characteristics*

341 Only a small fraction of the fish samples examined in this study had an empty gut (0% to 6%)
342 (Table 1). Prey species accumulation curves showed that the number of species found in guts had

343 reached an asymptote at all sites, indicating the adequacy of sample size for the diet analysis.
344 There were significant differences in average gut fullness among the populations (ANOVA, $F_{4,282} = 37$, $p < 0.001$), in which fish from Lemon Creek and Mill Creek had nearly two times more
345 food ($\sim 9.00 \text{ mg}\cdot\text{fish g}^{-1}$) than those from Richmond Creek ($\sim 5.00 \text{ mg}\cdot\text{fish g}^{-1}$) and three times
346 more than those from Main Creek and Neck Creek ($\sim 3.00 \text{ mg}\cdot\text{fish g}^{-1}$) (Tukey HSD test, $p <$
347 0.05 , Table 1). Although there were significant differences in prey species richness among
348 populations (ANOVA, $F_{4,249} = 2.6$, $p < 0.05$), the magnitude of the differences were small
349 (Table 1). Fish from Main Creek and Neck Creek had significantly lower prey species richness
350 than those from Lemon Creek (Tukey HSD test, $p < 0.05$, Table 1). A similar trend was also
351 observed for diet niche breadth based on Shannon-Wiener index (ANOVA, $F_{4,251} = 4.2$, $p <$
352 0.01), in which fish from Main Creek had a significantly lower niche breadth than those from
353 Lemon Creek (Tukey HSD test, $p < 0.05$, Table 1). No significant difference was observed in
354 prey species evenness among the populations (ANOVA, $F_{4,251} = 1.5$, $p = 0.19$, Table 1).

356 In general, decapods (mainly *Palaemonetes spp.*) were the dominant (%*FO* and %*W*) and most
357 important (%*IRI*) diet category observed in all populations, though there was some ontogenetic
358 variation, especially in the AK populations (Fig. 2a through e). Decapods generally became
359 increasingly important (%*IRI* = 2.9% to 94%) with the total length of fish from all sites (Fig. 2a
360 through e). A moderate amount (%*W* = 11% to 35%) of detritus was also frequently (%*FO* =
361 30% to 100%) found in the gut of mummichogs from all sites. At the AK sites (except for Mill
362 Creek), another important diet category was polychaetes (mostly *Nereis spp.*) (%*IRI* = 17% to
363 24%, Fig. 2c through e), which were frequently observed (%*FO* = 30% to 82%) and contributed
364 up to $\sim 40\%$ by weight to the total gut contents. Gastropods (mostly *Melampus bidentatus*),
365 arachnids (mostly trombididae and ixodides), and insects (mostly diptera larvae) were frequently

366 observed in fish from Lemon Creek (especially larger fish, >70+ mm TL) (%*FO* = 18% to 67%
367 for gastropods, 17% to 62% for arachnids, and 33% to 61% for insects). Nematodes, though their
368 contribution to total gut content weight was negligible (< 1.0%), were also frequently observed
369 in the guts of fish from all sites (%*FO* = 6% to 50%). Cirripedes (*Balanus spp.*) were frequently
370 observed in the guts of fish from Mill Creek, Richmond Creek, and Main Creek (%*FO* = 9% to
371 50%). Oligochaetes, bivalves, and teleosts were rarely found in the guts of mummichogs.

372 Prey selection by mummichogs based on the Chesson's index showed that only the Lemon
373 Creek and Mill Creek populations actively selected gastropods ($\alpha = 0.13$ to 0.31) and insects ($\alpha =$
374 0.23 to 0.43) (Fig. 3a and b). All populations of mummichogs consistently avoided bivalves and
375 oligochaetes (Fig. 3a through e). Although the selectivity of polychaetes by the AK populations
376 were higher than that by the Lemon Creek population, polychaetes were not actively selected (α
377 < ~0.11) (Fig. 3a through e). The Neck Creek population actively selected only amphipods ($\alpha =$
378 0.59 to 0.73) (Fig. 3e).

379

380 3.2. Multivariate comparisons of mummichog diet compositions

381 The diet compositions (both by weight and count) of mummichogs were significantly different
382 among the populations. Although the diet compositions were significantly different both among
383 sites (ANOSIM, $p < 0.001$ by weight and count) and length classes (ANOSIM, $p < 0.05$ by
384 weight and count), the global *R* values for the among-site comparisons ($R = 0.27$ by weight and
385 $R = 0.29$ by count) were considerably higher than those for the among-length class comparisons
386 ($R = 0.044$ by weight and $R = 0.043$ by count). Thus, pairwise comparisons were done among
387 sites only.

388 The *post-hoc* pairwise comparisons by weight among sites indicated that the diet composition
389 of the Lemon Creek population was significantly different from those of the AK populations (R
390 = 0.49 to 0.71, $p < 0.05$), with an exception of the Mill Creek population ($R = 0.24$, $p > 0.05$).
391 The differences in diet composition among these populations were mainly due to fish from
392 Lemon Creek ingesting substantially more gastropods and less polychaetes (SIMPER, relative %
393 contributions to dissimilarity = 22% to 25% by gastropods and 11% to 19% by polychaetes) than
394 those from Richmond Creek, Main Creek, and Neck Creek, as well as more decapods than those
395 from Main Creek (SIMPER, relative % contributions to dissimilarity = 19%).

396 The pairwise comparisons by count showed that the diet composition of fish from Lemon
397 Creek was significantly different from all the AK populations ($R = 0.30$ to 0.78 , $p < 0.05$). A
398 significant difference was also found between Mill Creek and Richmond Creek ($R = 0.38$, $p <$
399 0.05). The differences by count between Lemon Creek and the AK populations were due to the
400 ingestion of more insects and gastropods by fish from Lemon Creek than those from Mill Creek,
401 Main Creek, and Neck Creek (SIMPER, relative % contributions to dissimilarity = 15% to 17%
402 for insects and 12% to 17% for gastropods), whereas fish from Richmond Creek ingested fewer
403 gastropods and more polychaetes than those from Lemon Creek (SIMPER, relative %
404 contributions to dissimilarity = 16% for gastropods and 14% for polychaetes).

405

406 3.3. Relationships between body sizes of mummichogs and *Palaemonetes* spp.

407 The body size of an individual *Palaemonetes* spp. (a dominant diet item by weight) ingested by
408 mummichogs was significantly correlated with the total length of mummichogs from all sites
409 (least squares regression, $p < 0.05$, Fig. 4a through e). However, there was a substantial
410 difference in the length-specific maximum size of *Palaemonetes* spp. ingested by mummichogs

411 among the populations. The gut contents of mummichogs from Main Creek and Neck Creek
412 contained mostly small *Palaemonetes* spp. ($< \sim 0.20$ g) and only a few individuals larger than
413 0.40 g (Fig. 4d and 4e).

414

415 *3.4. Relationships between environmental variables and benthic macroinfaunal prey* 416 *assemblages*

417 Among trace metals measured, only mercury content in the sediment at the Arthur Kill (AK)
418 sites (varying from 0.80 to 2.46 $\mu\text{g}\cdot\text{g}^{-1}$ dry weight) was consistently above the ERM level,
419 whereas none of trace metals at Lemon Creek exceeded the ERM levels (Table 2). Canonical
420 correspondence analysis (CCA) showed significant relationships between environmental
421 variables and benthic macroinfaunal communities in both abundance ($F = 11, p < 0.001$) and
422 biomass ($F = 14, p < 0.001$). More than 70% of the total variances (71% for abundance and 75%
423 for biomass) in macroinfaunal communities were explained by the environmental variables
424 measured. The majority of the variances was explained by the first two statistically significant
425 canonical axes (Monte Carlo permutation test, $p < 0.001$) for both abundance (46% by the 1st
426 axis + 11% by the 2nd axis = 57% of the total variance) and biomass data (51% by the 1st axis +
427 13% by the 2nd axis = 64% of the total variance) (Fig. 5a and b). Stepwise forward selection
428 identified and ranked the environmental variables that significantly contributed to the total
429 explained variances in macroinfaunal compositions. For the abundance data, salinity (eigenvalue,
430 $\lambda = 0.22$; % variance explained = 27%), mercury ($\lambda = 0.14$; 17%), surface dissolved oxygen ($\lambda =$
431 0.09 ; 11%), silver ($\lambda = 0.09$; 11%), and cadmium ($\lambda = 0.05$; 6.0%) significantly contributed to
432 the total variance (Fig. 5a). For the biomass data, salinity ($\lambda = 0.36$; 29%), mercury ($\lambda = 0.23$;

433 19%), cadmium ($\lambda = 0.15$; 12%), total organic carbon ($\lambda = 0.12$; 9.7%), and silver ($\lambda = 0.06$;
434 4.9%) significantly contributed to the total variance (Fig. 5b).

435

436 3.5. Relationships between mercury whole body burdens and gut fullness of mummichogs

437 Mean Hg whole body burdens in mummichogs were significantly different among the
438 populations, ranging from 92 to 280 ng·g⁻¹ dry weight (Tukey HSD test, $p < 0.05$, Fig. 6).

439 Increasing Hg whole body burdens in mummichogs were significantly correlated with
440 differences in gut fullness among the populations (least square regression, $p < 0.001$, Fig. 7).

441

442 4. Discussion

443 Chronic habitat degradation has substantially altered biological assemblages of benthic habitats
444 in the Arthur Kill (AK) over the years (Crawford et al., 1994; Steinberg et al., 2004). In the
445 current study, benthic macroinfaunal assemblages in most of the AK salt marshes still exhibited
446 characteristics frequently observed in chronically polluted coastal benthic habitats, including
447 numerical dominance by a few small-bodied polychaete and oligochaete species (e.g.,
448 *Streblospio benedicti*).

449 Alterations in benthic macroinfaunal assemblages clearly resulted in markedly differences in
450 the feeding habits of mummichogs between the AK and reference (near Raritan Bay)
451 populations. The average relative gut fullness of mummichogs at the peak of their feeding
452 activity (i.e., daytime ebbing tides) was substantially lower at most of the AK sites (especially,
453 Main Creek and Neck Creek) than at the reference site (Lemon Creek). This disparity in gut
454 fullness among the populations appeared to be partly because of shifts in size of ingested prey;
455 the amount and frequency of consumption of large-bodied prey such as gastropods and decapods

456 were much lower in most of the AK populations than in the reference population. Furthermore,
457 diet niche breadth and prey species richness of mummichogs were also lower at these northern
458 AK sites (Main Creek and Neck Creek), where the benthic macroinfaunal communities were also
459 especially impoverished (e.g., lower species diversity), than at the other AK sites. Only the
460 reference (Lemon Creek) population frequently ingested insects and gastropods, reflecting the
461 higher occurrence of these macroinvertebrates in the environment. The reduction or
462 disappearance of sensitive relatively large invertebrate groups such as many gastropods and
463 bivalves from benthic macroinfaunal communities is commonly observed in degraded salt
464 marshes (Warwick and Clarke, 1994; Weston, 1990). Although a large number of gastropods
465 were found at one of the AK sites, Mill Creek, a majority of them were *Illynassa ventriretta*,
466 which is far larger than the gape size of mummichogs (Vince et al., 1976; Werme, 1981). These
467 gastropods were thus not ingested by mummichogs. Meiofauna such as nematodes were fairly
468 frequently (6% to 50%) ingested by different sizes of fish from all sites. Because of their low
469 contributions by weight (< 1%), however, their functional importance to adult mummichogs may
470 be negligible (Penczak, 1985b).

471 Alterations in interspecific interactions in degraded habitats may benefit some pollution-
472 tolerant species in certain cases (Boesch, 1982; Steimle et al., 1993). For example, due to lack of
473 or reduced competition and predation, pollution-tolerant benthic invertebrates (e.g., opportunistic
474 deposit-feeding annelids) often dramatically increase in abundance at the initial stage of
475 recolonization following anthropogenic disturbances, which may in turn provide an increased
476 prey resource to their predators (Jeffree and Williams, 1980; Powers et al., 2005). The benefit of
477 an enhanced prey resource may, however, depend on the “preferred” prey type of predators
478 (Steimle, 1994; Steimle et al., 1993). In the present study, although the diet composition of the

479 reference (Lemon Creek) population of mummichogs was significantly different from those of
480 the AK populations, there were only minor differences among the AK populations. Since many
481 dominant polychaete and oligochaete species at the AK sites were subsurface deposit-feeders,
482 which may not be easily accessible to mummichogs, differences in abundances of these deposit-
483 feeders among the sites may not have had measurable impacts on the diet compositions of
484 mummichogs in AK.

485 In general, despite their high abundance in urban tidal salt marshes, oligochaetes are rarely
486 found in the gut of mummichogs (Moy and Levin, 1991; Smith et al., 2000). Although it is
487 possible that rapid digestion of soft-bodied invertebrates (e.g., polychaetes and oligochaetes)
488 may influence their percentage contributions in the diet of mummichogs (Smith et al., 2000), the
489 present study indicated that enhanced abundances of oligochaetes in AK appear to have little or
490 no effect on the feeding habit of mummichogs. Werme (1981) has experimentally demonstrated
491 that mummichogs are much less efficient in foraging on sediment-dwelling prey than another
492 fundulid species, striped killifish (*Fundulus majalis*). Striped killifish often co-occurs with
493 mummichogs in tidal salt marshes in the northwest Atlantic coast, competing for prey resources
494 (Weisberg, 1986; Werme, 1981). Unlike striped killifish, however, mummichogs appear to
495 preferentially capture epibenthic prey, while striped killifish forage for potential prey items by
496 aggressively disturbing sediments (Sardá et al., 1998; Werme, 1981). As a result, these fundulid
497 species, despite a high degree of diet overlap, tend to have different 'preferences' in prey type
498 (Sardá et al., 1998; Weisberg, 1986); mummichogs generally 'prefer' relatively mobile prey such
499 as decapods and amphipods in intertidal marsh surface (Kneib and Stiven, 1978), while striped
500 killifish 'prefer' sedentary sediment-associated prey such as gastropods and bivalves (Brousseau

501 et al., 2008). Further studies would, however, still require determining the importance of
502 subsurface deposit-feeders to the diet of mummichogs.

503 High abundances of some epibenthic decapods such as grass shrimp (*Palaemonetes spp.*) may
504 also have masked some indirect effects of altered prey resources on the feeding habits of
505 mummichogs in AK. Grass shrimp was relatively larger than other potential prey items available
506 and one of the most frequently found prey items at both the AK and reference sites. The body
507 size of individual grass shrimp ingested by mummichogs was, however, considerably smaller at
508 two of the northern AK sites, Main Creek and Neck Creek, than at the other sites. These site-
509 specific relationships between mummichogs and grass shrimp observed in the current study may
510 have been due to altered feeding behaviors induced by chemical pollutants such as mercury
511 (Smith and Weis, 1997; Weis et al., 2001).

512 Canonical correspondence analysis in the current study indicated that a relatively large
513 proportion of among-site variance of benthic macroinfaunal communities (in both abundance and
514 biomass) was strongly associated with increasing concentrations of sediment-associated mercury
515 in the AK sites. Furthermore, a decrease in gut fullness of the AK (particularly, Main Creek and
516 Neck Creek) populations was also significantly correlated with increasing Hg whole body
517 burdens in mummichogs. Similarly, Weis et al. (2001) have also shown that mummichogs from
518 severely chemically polluted coastal habitats in New York and New Jersey (including AK)
519 ingested considerably fewer grass shrimp than those from the reference sites. Experimental
520 studies have demonstrated that anthropogenically introduced toxic chemicals such as mercury
521 can impair mummichogs' ability to capture epibenthic decapods (Smith and Weis, 1997; Weis et
522 al., 2001). These altered feeding behaviors ultimately reduce the ingestion of decapods by
523 mummichogs (Smith and Weis, 1997; Weis et al., 2001). In addition to prey availability,

524 chemical pollutants may thus also directly influence the feeding habits of fish by altering their
525 feeding behaviors in urban coastal habitats.

526 Like many other fish species, mummichogs are a visual (Weisberg et al., 1981) and size-
527 selective predator (Kneib, 1986), influencing prey abundance and distribution in tidal salt
528 marshes (Kneib and Stiven., 1982; Vince et al., 1976). Smith and Weis (1997) have
529 demonstrated that chronic exposure to chemical pollutants such as mercury can also alter size-
530 selective feeding of mummichogs. In addition to direct effects of anthropogenic stressors, altered
531 size-selective predation pressure may also lead to a shift in size structure of benthic
532 macroinvertebrate assemblages (Kneib and Stiven., 1982; Vince et al., 1976). Santiago-Bass et
533 al. (2001) have shown that the body size of the AK (Piles Creek, New Jersey) population of grass
534 shrimp is considerably larger than that of a reference (Tuckerton, New Jersey) population,
535 suggesting that grass shrimp may not only benefit from the reduced predator pressure from
536 mummichogs, but also be more tolerant to chemical pollution than mummichogs (Weis et al.,
537 2000). These species-specific differential responses to chemical pollutants suggest that altered
538 feeding behaviors of predators can also have top-down (reverse) cascading effects on their prey
539 (i.e., reduced predation pressure) in severely polluted benthic habitats (Santiago-Bass et al.,
540 2001; Weis et al., 2000).

541 Both altered feeding behaviors (e.g., reduced prey capture efficacy) and prey availability (e.g.,
542 disappearance of large-bodied prey) may also influence ontogenetic diet shifts of mummichogs
543 in urban benthic habitats. In general, large adult mummichogs ($> \sim 70$ mm TL) feed on decapods
544 such as fiddler crab (*Uca spp.*) and grass shrimp (*Palaemonetes spp.*) (Kneib and Stiven, 1978),
545 whereas small adults and juveniles ($< \sim 40$ mm TL) feed on small crustaceans such as copepods
546 (Able et al., 2008; Smith et al., 2000). In the current study, the amount of large prey such as

547 decapods such as grass shrimp generally increased with the body size of mummichogs, whereas
548 the amount of other prey items (especially polychaetes) decreased at most sites. Mummichogs
549 from the northern AK sites (Main Creek and Neck Creek) that ingested only small shrimp,
550 however, mainly substituted with an increased ingestion of other prey including amphipods and
551 polychaetes until they reached the largest size group (> 80 mm TL). In fact, the Neck Creek
552 population continued to actively select amphipods even when they reached the largest size group.

553 Increased abundance and biomass of polychaetes are commonly observed in degraded marine
554 benthic habitats (Warwick and Clarke, 1993; Weston, 1990), including the AK tidal salt marshes.
555 In the current study, higher relative proportion (by weight) of polychaetes was more frequently
556 observed in the diets of fish from the northern AK sites (Main Creek, Richmond Creek, and
557 Neck Creek) than those from the southern AK (Mill Creek) and the reference (Lemon Creek)
558 sites. The most frequently observed polychaete species in the present study was *Nereis spp.*,
559 which was also one of the most abundant polychaete species (particularly, *N. acuminata*) found
560 in the AK benthic macroinfaunal communities. Moreover, *Nereis spp.* were relatively larger than
561 other polychaete species found in AK; they may thus have been more conspicuous to
562 mummichogs than other abundant, but smaller polychaete species such as *Hobsonia florida* and
563 *Streblospio benedicti*.

564 Previous studies have suggested that an increased ingestion of detritus may indicate pollution
565 impacts on the feeding habits of mummichogs in urban salt marshes with impoverished prey
566 resources (Smith and Weis, 1997; Weis et al., 2001). In the current study, however, there was no
567 consistent difference in the amount of detritus ingested by mummichogs among the populations.
568 A frequent occurrence (~31% to 100%) of detritus with an average of ~20% (by weight) of total
569 gut contents was observed in all populations of mummichogs. This frequent ingestion of detritus

570 is not uncommon for many benthivorous predators such as mummichogs (Gerking, 1994; Moy
571 and Levin, 1991). Mummichogs are also unlikely to use detritus as a primary food source for
572 their growth or metabolism (Prinslow et al., 1974; White et al., 1986). Furthermore, in the
573 current study, a large amount of detritus was often found in the guts of mummichogs with
574 sediment-dwelling invertebrates such as polychaetes, but not with highly mobile epibenthic prey
575 such as grass shrimp. The occurrence of detritus may thus have been due to an accidental
576 ingestion (Kneib and Stiven, 1978; Penczak, 1985b). In habitats with limited prey resource such
577 as urbanized tidal salt marshes, however, mummichogs may continue to unintentionally ingest
578 detritus in search for sediment-dwelling prey as a short-term strategy (Weisberg and Lotrich,
579 1986). Altered prey availability in the environment may thus indirectly influence the amount of
580 detritus ingested by mummichogs.

581 Macroalgae (e.g., *Ulva spp.*) were also frequently found in the gut contents of mummichogs in
582 the present study. The frequency of occurrence was slightly higher in the reference (Lemon
583 Creek) and Mill Creek populations than the other populations, whereas the relative proportion
584 (by weight) was much larger in the AK populations than the reference population. Unlike
585 detritus, the ingestion of macroalgae by mummichogs may not be accidental (Fell et al., 1998),
586 though mummichogs often feed on small crustaceans and insects that aggregate around
587 macroalgae (Werme, 1981). The amount of plant material in the gut of some benthivorous fishes
588 in coastal habitats including mummichogs is known to increase with their body size (Able et al.,
589 2008; Smith et al., 2000). In general, larvae and juveniles of these benthivorous fishes in coastal
590 habitats are often primarily carnivorous, feeding mostly on meiofauna and macroinfaunal larvae,
591 while adults become more omnivorous, ingesting a progressively increasing amount of plant
592 material as they grow (Kneib and Stiven, 1978; Smith et al., 2000). Although the morphology of

593 their digestive tract suggests that mummichogs are not well suited for herbivorous feeding habits
594 (Weisberg, 1986; Werme, 1981), Moerland (1985) has experimentally demonstrated that there is
595 a temperature-dependent cellulase activity in the digestive tract of mummichogs. It is thus
596 possible that mummichogs may partially substitute an insufficient invertebrate prey supply with
597 macroalgae, when necessary.

598 Non-random perturbations such as chemical pollution to trophic structures may provide a
599 unique opportunity to elucidate intricate trophic interactions in aquatic ecosystems (Fleeger et
600 al., 2003; Raffaelli, 2005). The results based on the Chesson's selectivity index in the current
601 study revealed the highly flexible nature of the overall feeding strategy of mummichogs, mostly
602 corresponding to general patterns in benthic macroinfaunal communities, as suggested by
603 previous studies (e.g., Moy and Levin, 1991). A large amount of decapods (mostly,
604 *Palaemonetes spp.*) was frequently observed in the gut of adult mummichogs, due most likely to
605 their high abundance (personal observation) and relatively large body size. However,
606 mummichogs did show some preference to amphipods, isopods, and aquatic insects, as well as
607 avoidance of oligochaetes and bivalves, regardless of changes in their availabilities in the
608 environment. For example, although the Main Creek population ingested more polychaetes than
609 the reference population, this was not resulted from active selection, but from an increased
610 abundance of polychaetes and possibly a reduced ingestion (most likely reduced prey-capture
611 efficiency) of preferred prey (e.g., decapods). These results suggest that even at degraded
612 habitats with impoverished prey resources, mummichogs still exhibit some prey preference as
613 well as their trophic adaptability.

614

615 **5. Conclusions**

616 The present study showed that chronic chemical pollution appeared to have directly (through
617 chemical bioaccumulation) and indirectly (through reduced benthic prey availability) altered
618 feeding habits and strategies of mummichogs in highly urbanized tidal salt marshes of the Arthur
619 Kill (AK). Since benthic macroinvertebrates are highly responsive to altered environmental
620 conditions in urban coastal habitats, the direct trophic relationship with these invertebrates may
621 have integrated overall anthropogenic impacts on benthic habitats. Furthermore, as chemical
622 pollutants such as mercury can be transferred through food chains and accumulate in predators
623 (Goto and Wallace 2009), the diet habits of the AK populations of mummichogs may also have
624 been influenced by chemically induced alterations in feeding behaviors.

625 The predominant prey of adult mummichogs in the present study, grass shrimp, are also among
626 the most productive macroinvertebrate species in tidal salt marshes in the northwest Atlantic
627 coast and known to feed not only on plant material, but also on a variety of benthic meiofauna
628 and macroinfauna (Gregg and Fleeger, 1998). Due to their intimate association with the cycling
629 of materials and energy in coastal ecosystems, partial trophic decoupling between mummichogs
630 and grass shrimp observed in some of the AK sites could thus disrupt not only energy transfer to
631 mummichogs, but also ecosystem functioning in these highly urbanized salt marshes.

632 Altered prey resources often induce functional responses in predators such as food
633 consumption and growth rates, including mummichogs (Goto and Wallace, 2010), which could
634 also influence trophically mediated ecosystem processes (e.g., energy flow and nutrient cycling)
635 (Eby et al., 2005; Peterson et al., 2000). The importance of integrity of trophic structure to
636 ecosystem functioning is widely recognized (Clements and Newman, 2002). Empirical and
637 experimental studies have demonstrated a close association between biodiversity and ecosystem
638 functioning; low-biodiversity ecosystems (e.g., urban tidal salt marshes) are likely to have low

639 buffering capacity toward further anthropogenic disturbances (Woodward et al., 2005).
640 Urbanized coastal ecosystems with simplified trophic structure are likely to be at a vulnerable
641 state, potentially facing food chain collapse from a loss of a member of the food web (Clements
642 and Newman, 2002). Furthermore, tidal salt marshes are among the most productive
643 environments (Levin et al., 2001), supporting adjacent coastal ecosystems (Deegan et al., 2000).
644 The spatial movement of marsh production toward open coastal habitats is intimately associated
645 with the trophic interactions between resident and migrant nekton (“trophic relay”) in salt
646 marshes (Kneib, 1997). As a key resident species in urban salt marshes, monitoring the feeding
647 ecology of mummichogs (e.g., trophic relationships with grass shrimp) is, therefore, critical in
648 understanding the structural and functional responses to restorations of chronically degraded
649 coastal habitats.

650

651 **Acknowledgements**

652 This project was supported by New York Sea Grant (project number, R/CTP-39) funded under
653 award # NA16RG1645 from the National Sea Grant College Program of the U.S. Department of
654 Commerce’s National Oceanic and Atmospheric Administration, to the Research Foundation of
655 State University of New York on behalf of New York Sea Grant. The statements, findings,
656 conclusions, views, and recommendations are those of the author(s) and do not necessarily
657 reflect the views of any of those organizations. Additional funding was also provided by the
658 Graduate Center, the City University of New York under the Doctoral Student Research Grant
659 Program. We greatly appreciate field and laboratory assistance provided by M. Perez and D.
660 Seebaugh and useful comments provided by R. R. Veit, J. W. Rachlin, J. R. Waldman, P. Weis,
661 and an anonymous reviewer.

662

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894 **Figure captions**

895 **Figure captions**
 896 Fig. 1. Map of study sites in the Arthur Kill (Mill Creek, Richmond Creek, Main Creek, and
 897 Neck Creek) and Raritan Bay (Lemon Creek), New York, USA.

898
 899 Fig. 2. Contributions by index of relative importance (*IRI*) (%) of diet categories (Decapoda,
 900 Amphipoda, Isopoda, Cirrledia, Polychaeta, Oligochaeta, Nematoda, Gastropoda, Bivalvia,
 901 Arachnida, Insecta, zooplankton, Osteichthyes, Alga, and detritus) to the length class (mm, TL)-
 902 specific diet compositions of mummichogs from (a) Lemon Creek, (b) Mill Creek, (c) Richmond
 903 Creek, (d) Main Creek, and (e) Neck Creek.

904
 905 Fig. 3. Chesson's index prey selectivity by 60+ (black bars), 70+ (gray bars), and 80+ (dark gray
 906 bars) mm-size classes of mummichogs from (a) Lemon Creek, (b) Mill Creek, (c) Richmond
 907 Creek, (d) Main Creek, and (e) Neck Creek. Dotted horizontal lines indicate a critical value for
 908 no or neutral selection ($\alpha_{\text{neutral}} = 0.125$).

909
 910 Fig. 4. Relationships between the total length (mm) of mummichogs and body size (g, wet
 911 weight) of *Palaemonetes spp.* in the gut contents of mummichogs from (a) Lemon Creek, (b)
 912 Mill Creek, (c) Richmond Creek, (d) Main Creek, and (e) Neck Creek. Solid lines indicate a
 913 best-fit regression.

914
 915 Fig. 5. Biplot representation (Axes 1 and 2) of canonical correspondence analysis (CCA) on (a)
 916 abundance and (b) biomass of benthic macroinvertebrates in relation to environmental variables
 917 at study sites. Environmental variables are represented by arrows; species are represented by
 918 open triangles. The length of arrows indicates the relative importance of environmental variables.
 919 Arrows pointing in the same direction indicate positive correlation, while arrows pointing in the
 920 opposite direction indicate negative correlation. DO = dissolved oxygen. TOC = total organic
 921 carbon. *Eteone* = *Eteone lactea*, *Hobsonia* = *Hobsonia florida*, *Streblospio* = *Streblospio*
 922 *benedicti*, *Edotea* = *Edotea triloba*, *Macoma* = *Macoma balthica*, *Mya* = *Mya arenaria*,
 923 *Cyathura* = *Cyathura polita*, *Gammarus* = *Gammarus mucronatus*, *Ilyanassa* = *Ilyanassa*
 924 *trivittata*, *Glycinde* = *Glycinde salitaria*, *Balanus* = *Balanus improvises*, *Prionospio* =
 925 *Prionospio sp.*, *Pectinaria* = *Pectinaria gouldii*, *Corophium* = *Corophium volutator*, *Uca* = *Uca*
 926 *pugnax*, *Ophelia* = *Ophelia denticulate*, *Palaemonetes* = *Palaemonetes larvae*, and *Spio* = *Spio*
 927 *setosa*.

928
 929 Fig. 6. Mean mercury whole body burdens ($\text{ng}\cdot\text{g}^{-1}$, dry weight) in mummichogs from the study
 930 sites. Different letters indicate statistically significant difference among the populations (one-
 931 way ANOVA followed by Tukey's HSD test).

932

933 Fig. 7. A relationship between mean mercury whole body burdens ($\text{ng}\cdot\text{g}^{-1}$, dry weight) and gut
934 fullness ($\text{mg}\cdot\text{g}^{-1}$, wet weight) of mummichogs from the study sites. A solid line indicates a best-
935 fit regression.

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Table 1. Fish sample characteristics and mean values (\pm SE) of indices (gut fullness, prey species richness, prey species diversity, and prey species evenness) for the gut contents of mummichogs from the study sites.

	Study sites				
	Lemon Creek	Mill Creek	Richmond Creek	Main Creek	Neck Creek
Sample size	$n = 44$	$n = 44$	$n = 56$	$n = 73$	$n = 68$
Size range (TL, mm)	43–88	59–100	60–99	41–81	45–89
(g, wet weight)	1.1–10.0	3.1–16.0	2.7–15.0	41.0–81.0	0.80–7.6
Empty stomach (%)	0.0	0.0	1.6	5.6	5.9
<u>Diet indices</u>					
Gut fullness	$9.0 \pm 0.5a$	$8.4 \pm 0.5a$	$5.0 \pm 0.3b$	$2.9 \pm 0.3c$	$3.3 \pm 0.3c$
Margalef's	$1.9 \pm 0.10a$	$1.7 \pm 0.06ab$	$1.8 \pm 0.06ab$	$1.6 \pm 0.05b$	$1.6 \pm 0.06b$
Shannon-Wiener	$1.2 \pm 0.10a$	$1.0 \pm 0.05ab$	$1.0 \pm 0.06ab$	$0.8 \pm 0.05b$	$0.8 \pm 0.06b$
Heip's	0.84 ± 0.03	0.92 ± 0.03	0.93 ± 0.02	0.92 ± 0.03	0.88 ± 0.03

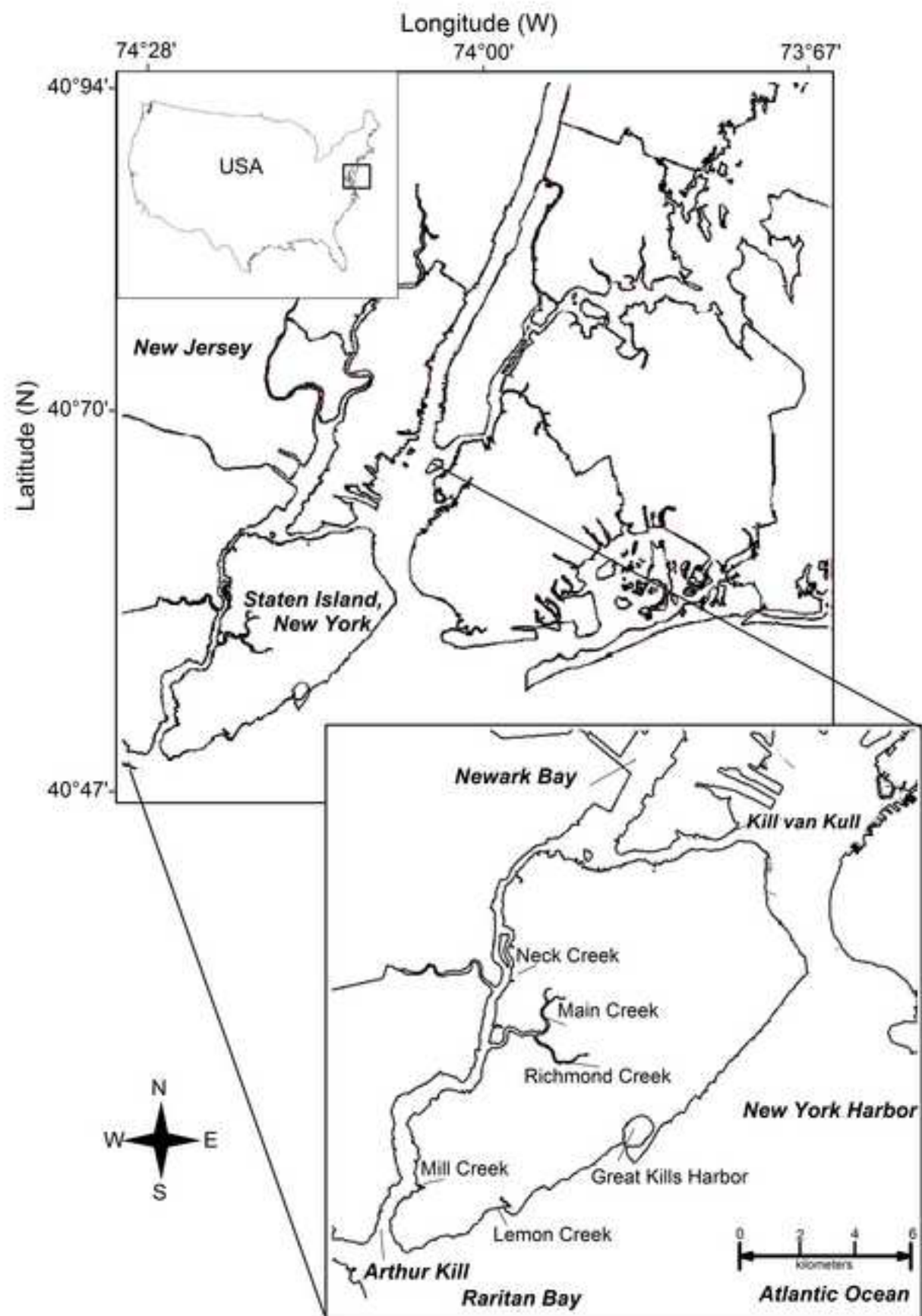
Different letters indicate statistically significant difference in indices among sites (one-way ANOVA followed by Tukey's HSD test).

Table 2. Mean values of physicochemical environmental variables of the study sites during the summer of 2004. Metal concentrations above the ERM levels are indicated in **bold**

Environmental variables	Study sites					
	ERM ^a	Lemon Creek	Mill Creek	Richmond Creek	Main Creek	Neck Creek
Temperature (°C)		25.2	28.3	29.6	26.0	31.6
Dissolved oxygen (mg·l ⁻¹)		6.13	6.53	3.07	2.80	16.1
Salinity		6.3	14.7	5.0	6.0	14.0
pH		7.68	7.26	7.88	7.93	8.70
Fine sediment particles (%)		33.5	74.7	63.8	17.5	59.5
Total organic carbon (%)		5.95	4.92	2.38	5.43	12.5
Trace metals (µg·g ⁻¹ , dry weight)						
Ag	3.7	0.83	1.38	0.78	1.59	1.37
Cd	9.6	1.20	2.88	1.58	2.90	1.08
Cu	270	125	912	99.7	216	153
Hg	0.71	0.28	0.98	0.80	1.86	2.46
Ni	51.6	27.9	36.5	34.0	70.3	49.0
Pb	218	83.9	656	91.9	171	215
Zn	410	220	1190	289	411	339

^aERM = Effects Range Median = the threshold levels of trace metals for potential adverse effects in organisms (Long and Morgan, 1990).

Figure 1



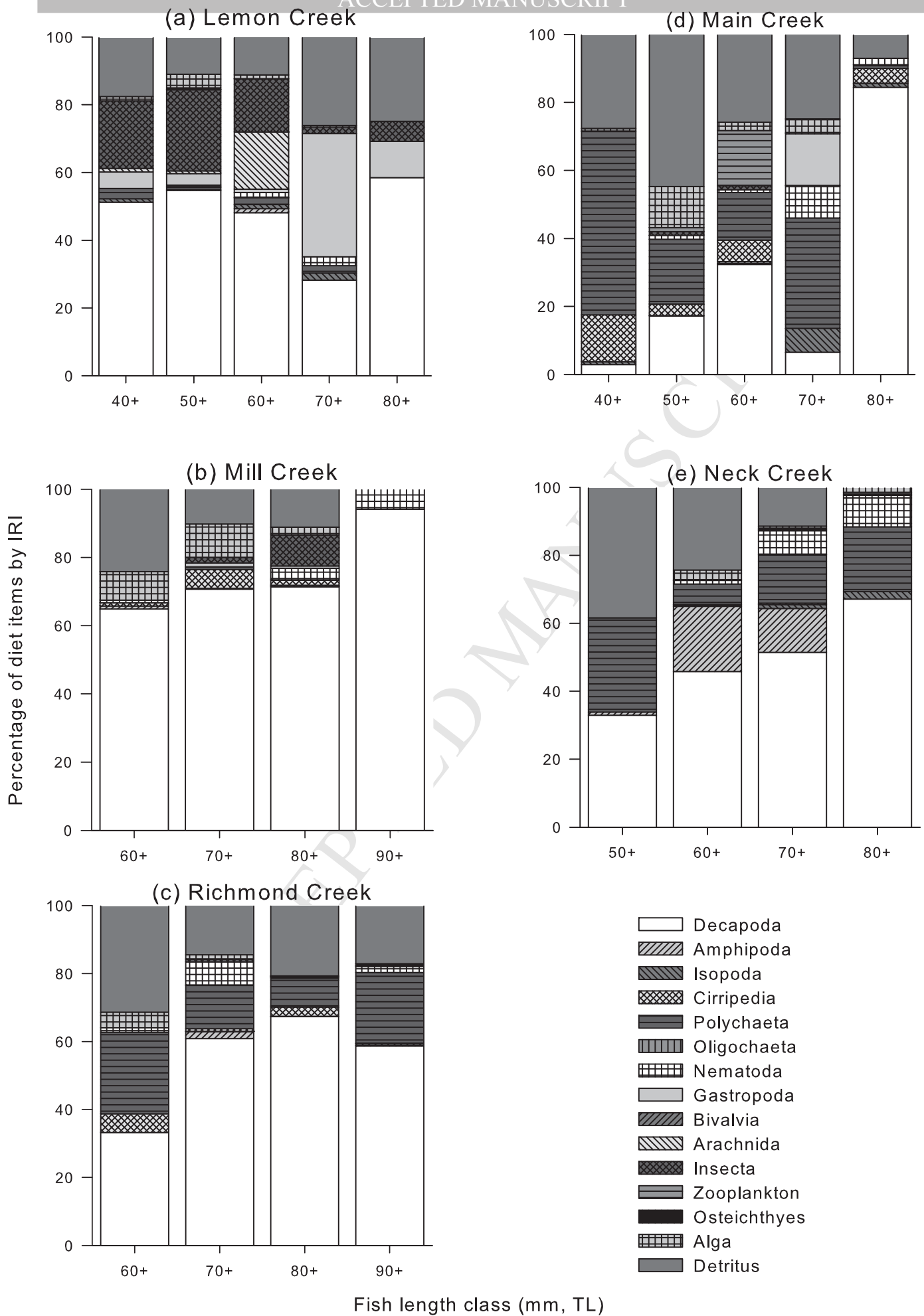
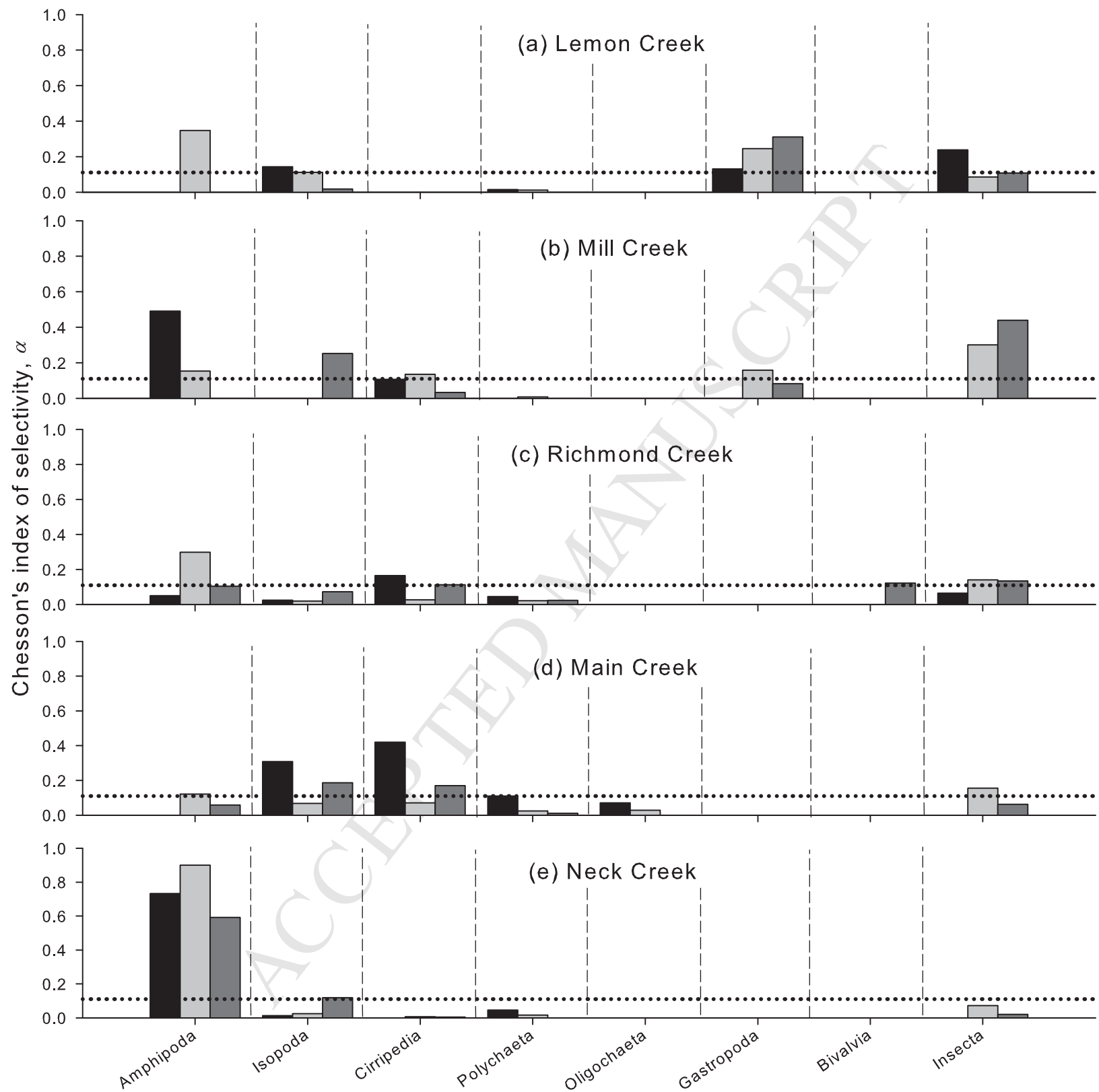


Figure 3



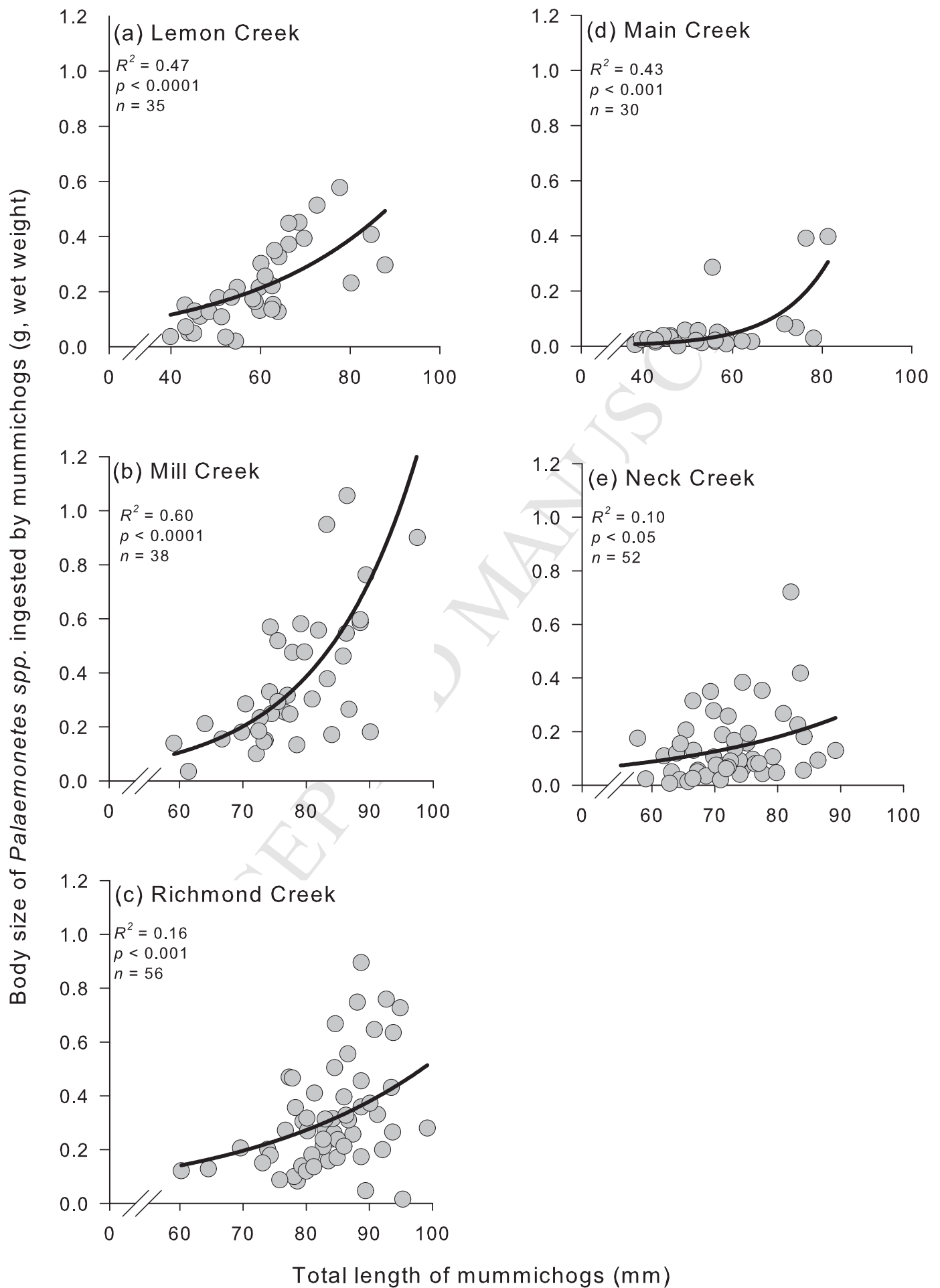


Figure 5a

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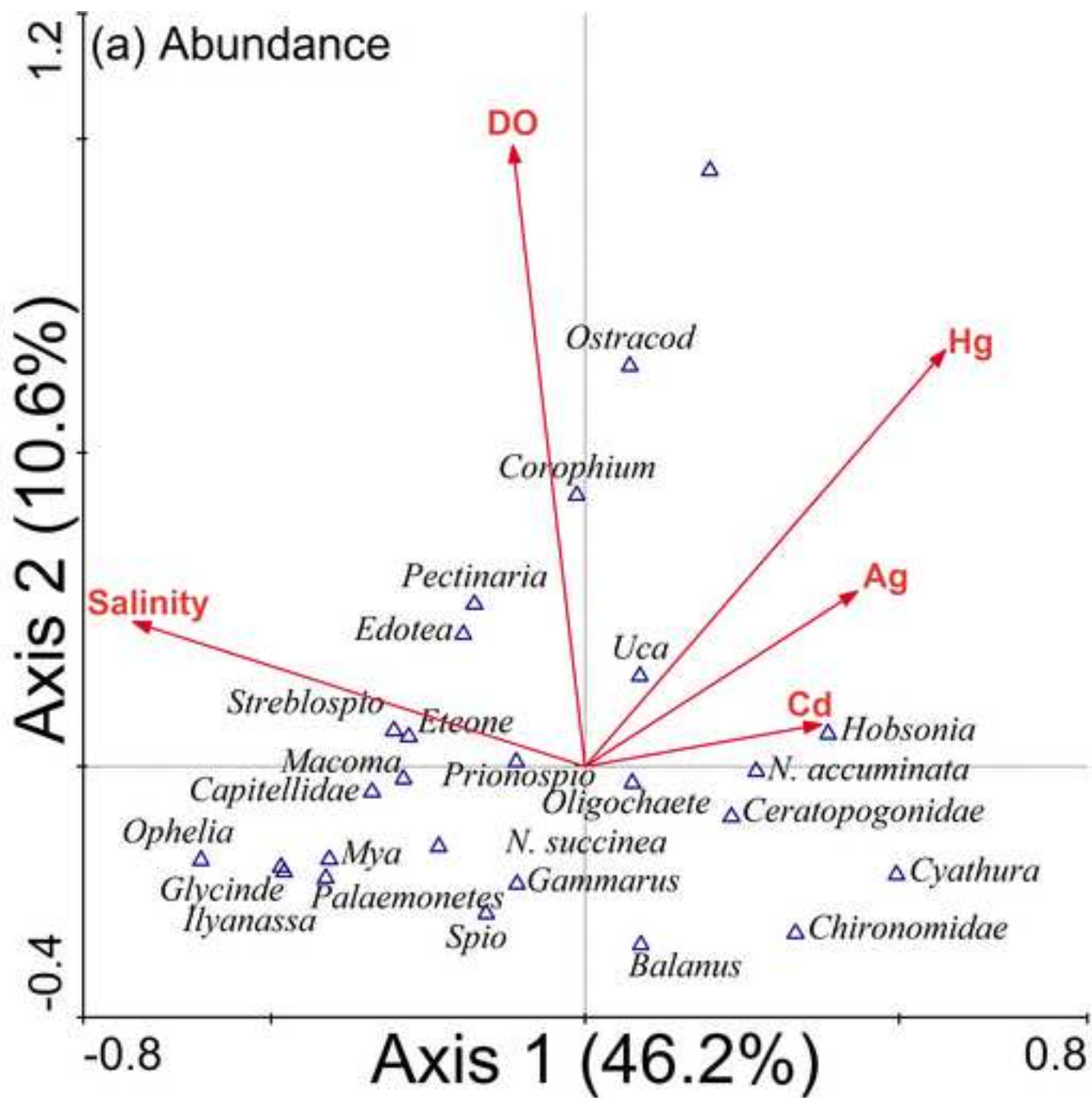


Figure 5b

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