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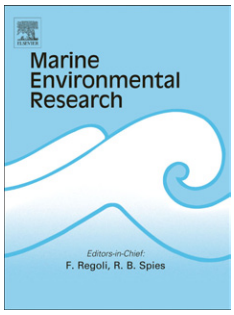


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# Altered feeding habits and strategies of a benthic forage fish (*Fundulus heteroclitus*) in chronically polluted tidal salt marshes

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

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

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## 1. Introduction

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

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

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

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

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

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

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

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

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

## 2. Materials and methods

### *2.1. Study area*

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

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

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

*2.2. Field sampling*

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

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

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

Benthic macroinvertebrates were also collected to assess prey availability for mummichogs. Macroinvertebrates were sampled by taking bulk surficial (top ~5 cm) sediments (one replicate per a randomly chosen 100 $m^2$ area) from the intertidal marsh at each study site, using a 15 x 15 cm-quadrat sampler ($n$ = 5 per site). Each sample was sieved through a 500 µm-mesh and fixed with 10% formalin stained with Rose Bengal in a glass jar in the field, which was then transported to the laboratory. Additional sediment samples were collected for analyses of trace metals, grain size, and total organic carbon (TOC) content. Surface water temperature, pH, salinity, and dissolved oxygen (DO) were also measured within the proximity of each replicate sample of benthic macroinvertebrates.

*2.3. Laboratory sample processing: Mummichogs*

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

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

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

Fish samples for Hg analysis (*n* = ~30 per site) were allowed to depurate gut contents for at least 24 hours in filtered 15‰ seawater and euthanized by an overdose of MS222. Then, fish samples were homogenized with a Polytron tissue homogenizer (Kinematica, Inc., Switzerland) in NANOpure® (reagent-grade) water. Homogenized samples were oven-dried (~60 °C) for 24 to 48 hours. Oven dried samples were processed and analyzed for mercury, as described below.

*2.4. Laboratory sample processing: Benthic macroinvertebrates*

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

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

*2.5. Laboratory sample processing: Sediments*

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

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

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

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

*2.6. Mummichog diet data analyses: Univariate measures*

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

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

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

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

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

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

*2.7. Mummichog diet data analyses: Multivariate analyses*

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

*2.8. Statistical analyses of relationships between environmental variables and benthic macroinfaunal prey assemblages*

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

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

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

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

## 3. Results

*3.1. Univariate comparisons of mummichog diet characteristics*

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

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reached an asymptote at all sites, indicating the adequacy of sample size for the diet analysis. 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 food (~9.00 mg·fish $g^{-1}$) than those from Richmond Creek (~5.00 mg·fish $g^{-1}$) and three times more than those from Main Creek and Neck Creek (~3.00 mg·fish $g^{-1}$) (Tukey HSD test, $p < 0.05$, Table 1). Although there were significant differences in prey species richness among populations (ANOVA, $F_{4, 249} = 2.6$, $p < 0.05$), the magnitude of the differences were small (Table 1). Fish from Main Creek and Neck Creek had significantly lower prey species richness than those from Lemon Creek (Tukey HSD test, $p < 0.05$, Table 1). A similar trend was also observed for diet niche breadth based on Shannon-Wiener index (ANOVA, $F_{4, 251} = 4.2$, $p < 0.01$), in which fish from Main Creek had a significantly lower niche breadth than those from Lemon Creek (Tukey HSD test, $p < 0.05$, Table 1). No significant difference was observed in prey species evenness among the populations (ANOVA, $F_{4, 251} = 1.5$, $p = 0.19$, Table 1).

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

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

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

*3.2. Multivariate comparisons of mummichog diet compositions*

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

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

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

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

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

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

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

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

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

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

Mean Hg whole body burdens in mummichogs were significantly different among the populations, ranging from 92 to 280 $ng \cdot g^{-1}$ dry weight (Tukey HSD test, $p < 0.05$**,** Fig. 6). Increasing Hg whole body burdens in mummichogs were significantly correlated with differences in gut fullness among the populations (least square regression, $p < 0.001$, Fig. 7).

## 4. Discussion

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

## 5. Conclusions

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

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

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

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

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**Figure captions**

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

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

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

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

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

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

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

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Table 1. Fish sample characteristics and mean values (± 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 |
| Diet indices | | | | | |
| Gut fullness | 9.0 ± 0.5a | 8.4 ± 0.5a | 5.0 ± 0.3b | 2.9 ± 0.3c | 3.3 ± 0.3c |
| Margalef's | 1.9 ± 0.10a | 1.7 ± 0.06ab | 1.8 ± 0.06ab | 1.6 ± 0.05b | 1.6 ± 0.06b |
| Shannon-Wiener | 1.2 ± 0.10a | 1.0 ± 0.05ab | 1.0 ± 0.06ab | 0.8 ± 0.05b | 0.8 ± 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).

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**Table 2**

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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 | ERM[a] | Study sites | | | | |
|---|---|---|---|---|---|---|
| | | Lemon Creek | Mill Creek | Richmond Creek | Main Creek | Neck Creek |
| Temperature (°C) | | 25.2 | 28.3 | 29.6 | 26.0 | 31.6 |
| Dissolved oxygen ($mg \cdot l^{-1}$) | | 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 ($\mu g \cdot g^{-1}$, 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 |

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

Figure 1

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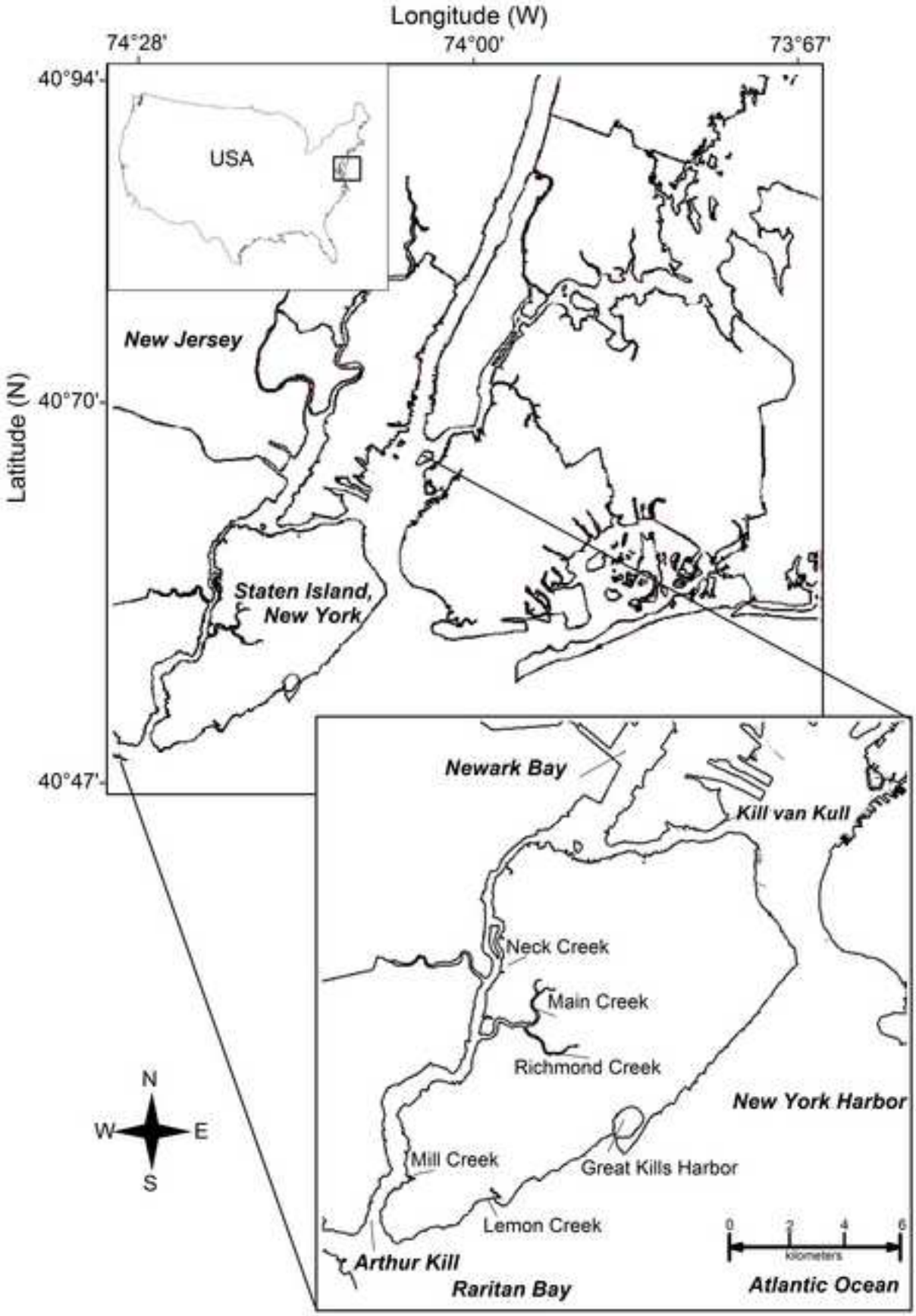

Figure 2

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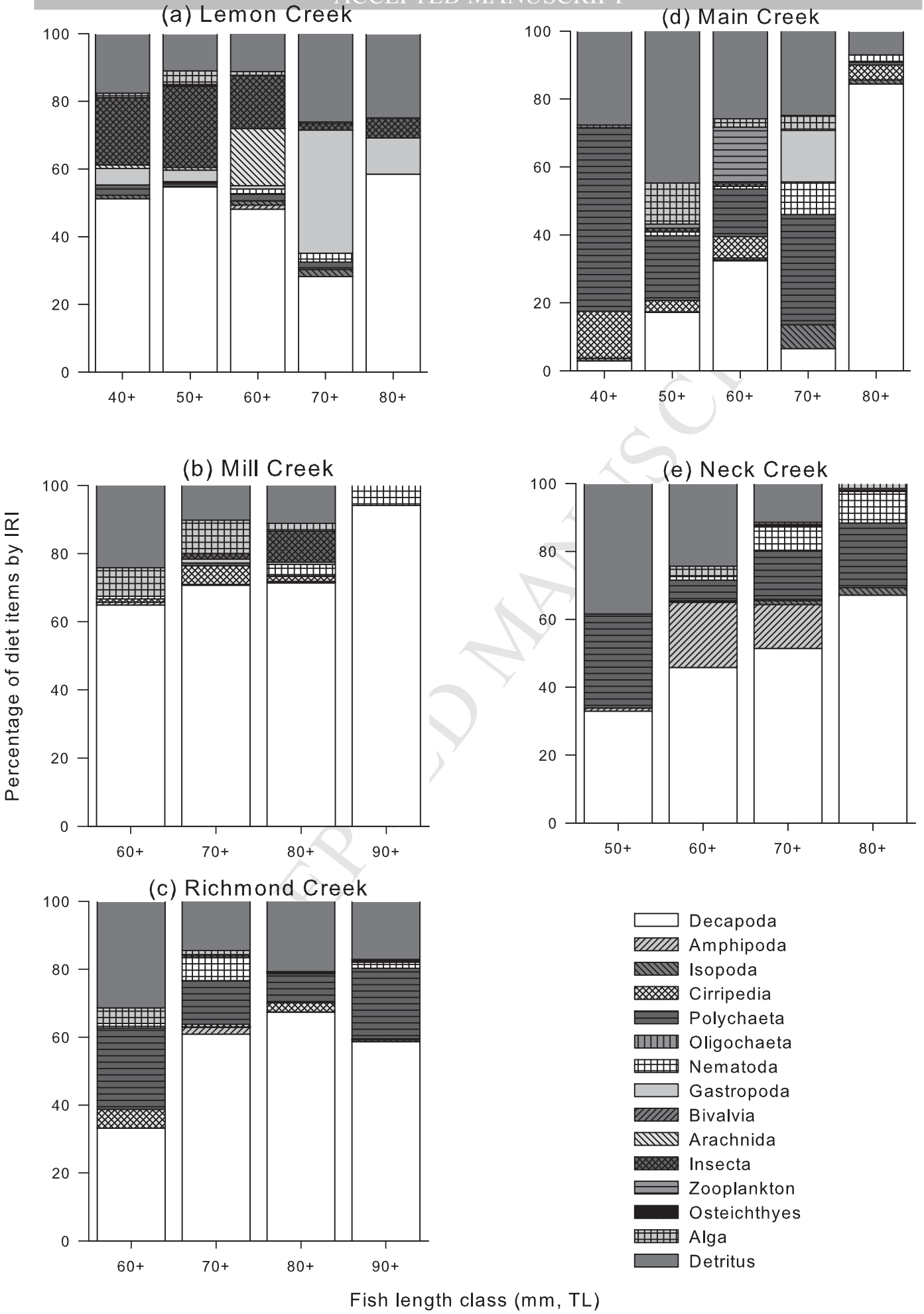

Figure 3

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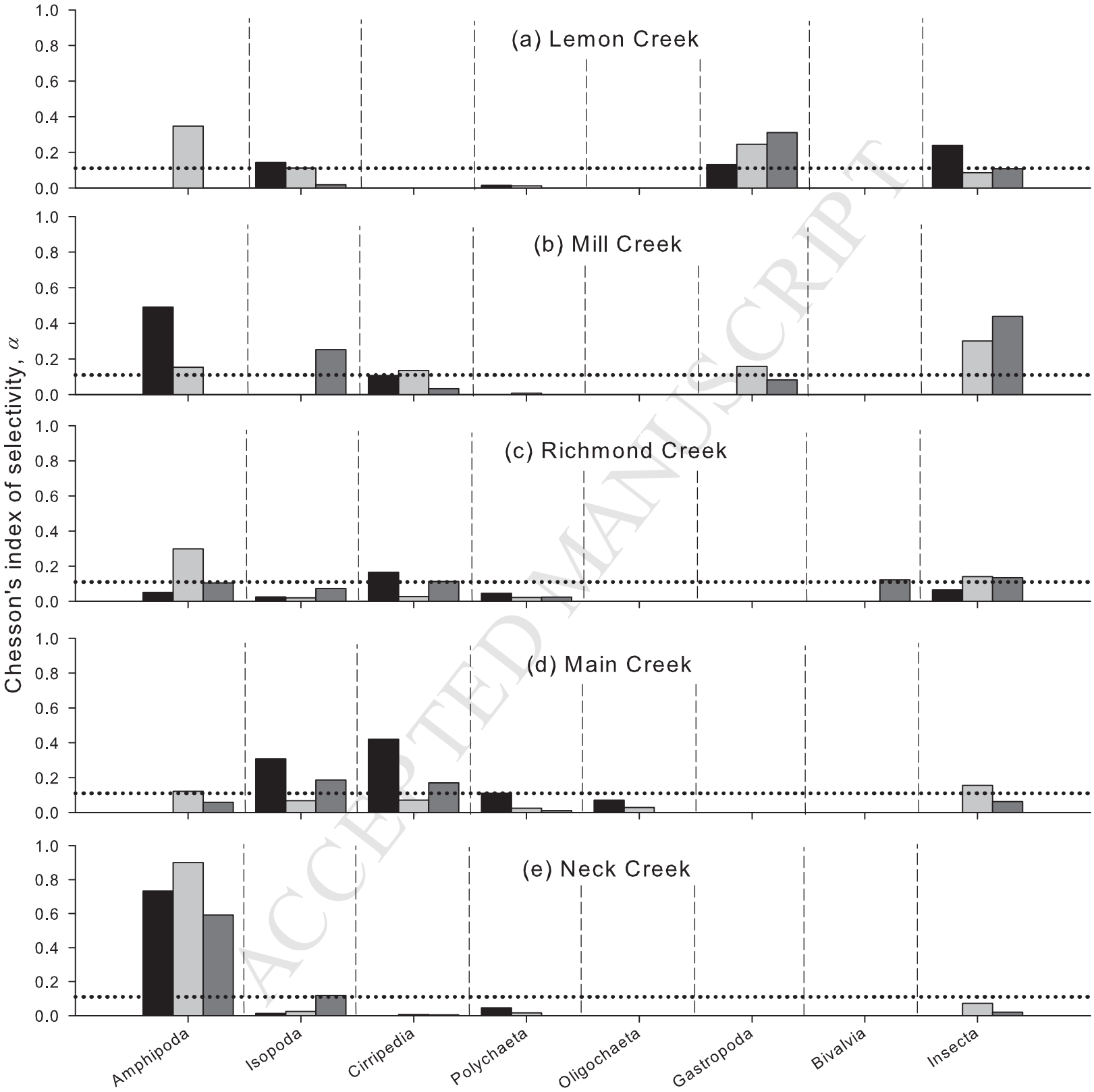


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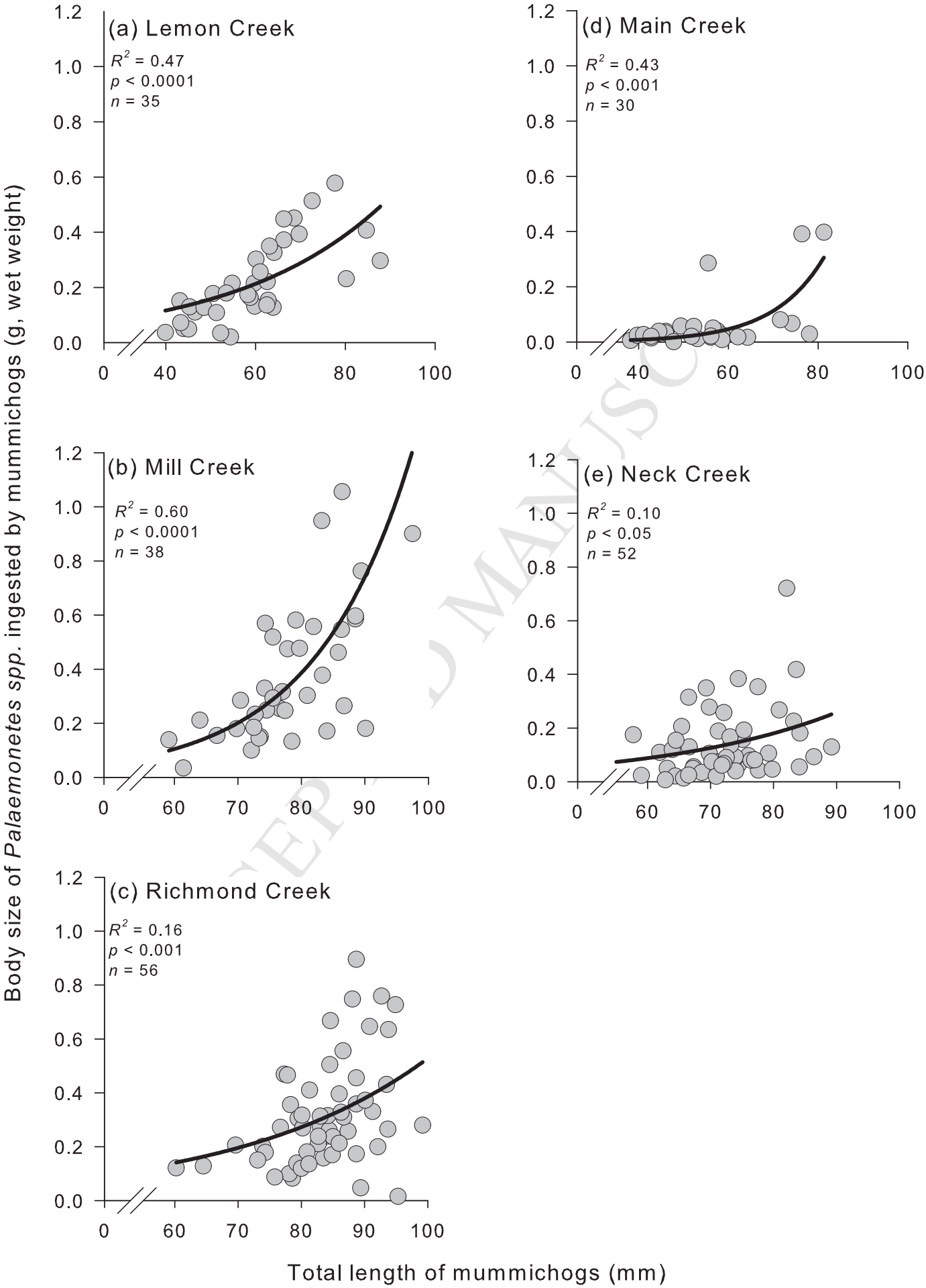

Figure 5a
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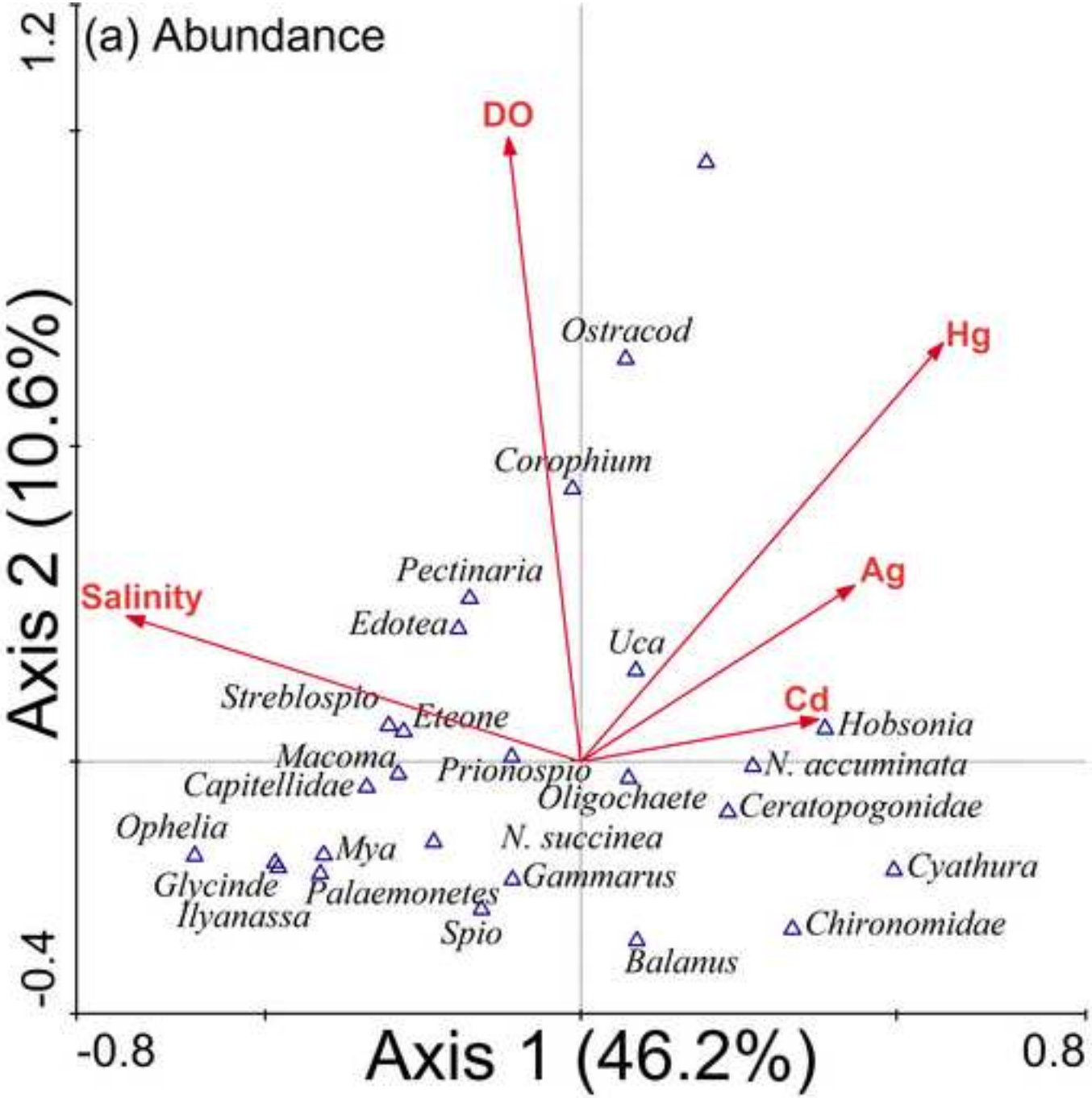

Figure 5b
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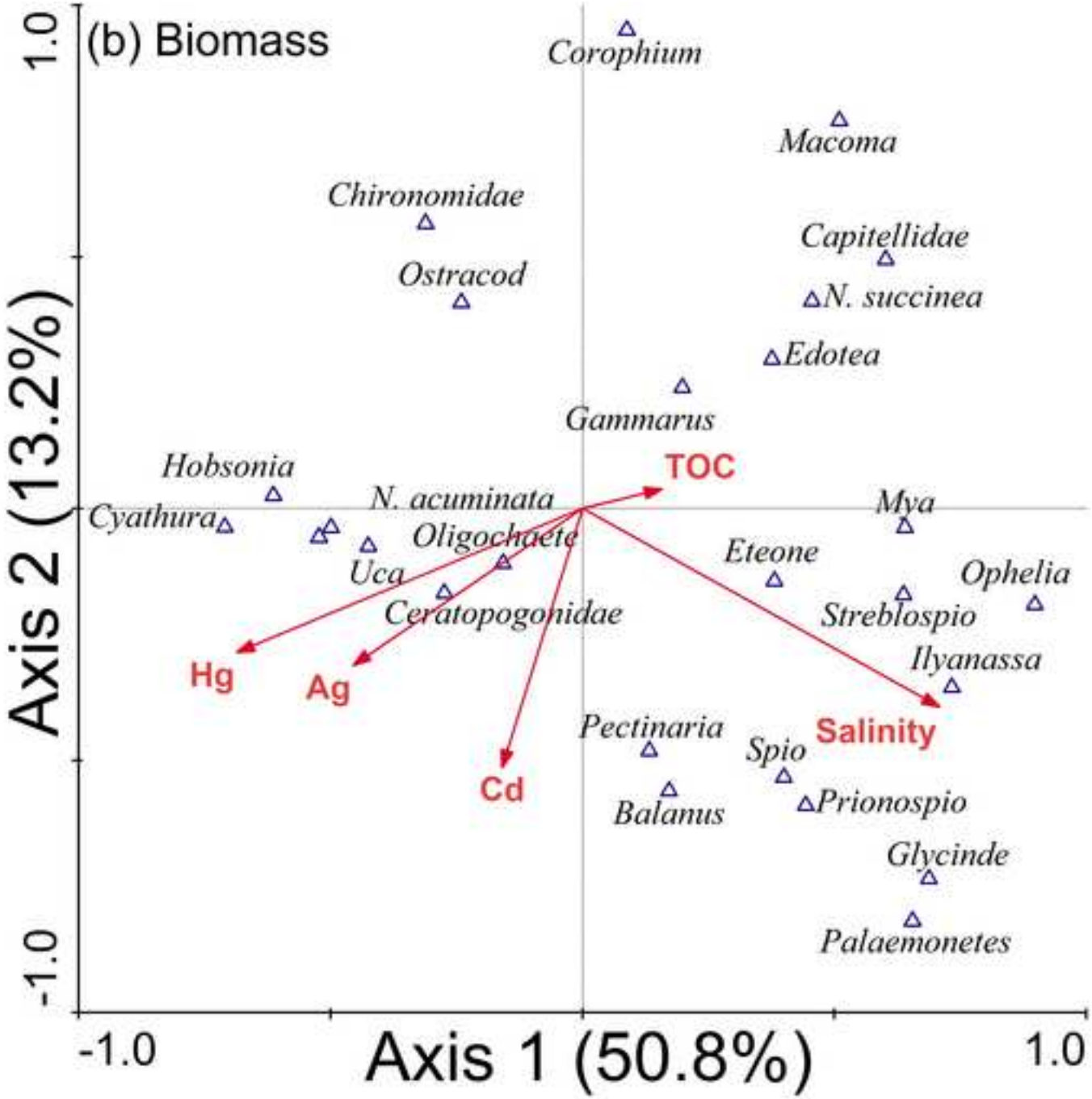

Figure 6

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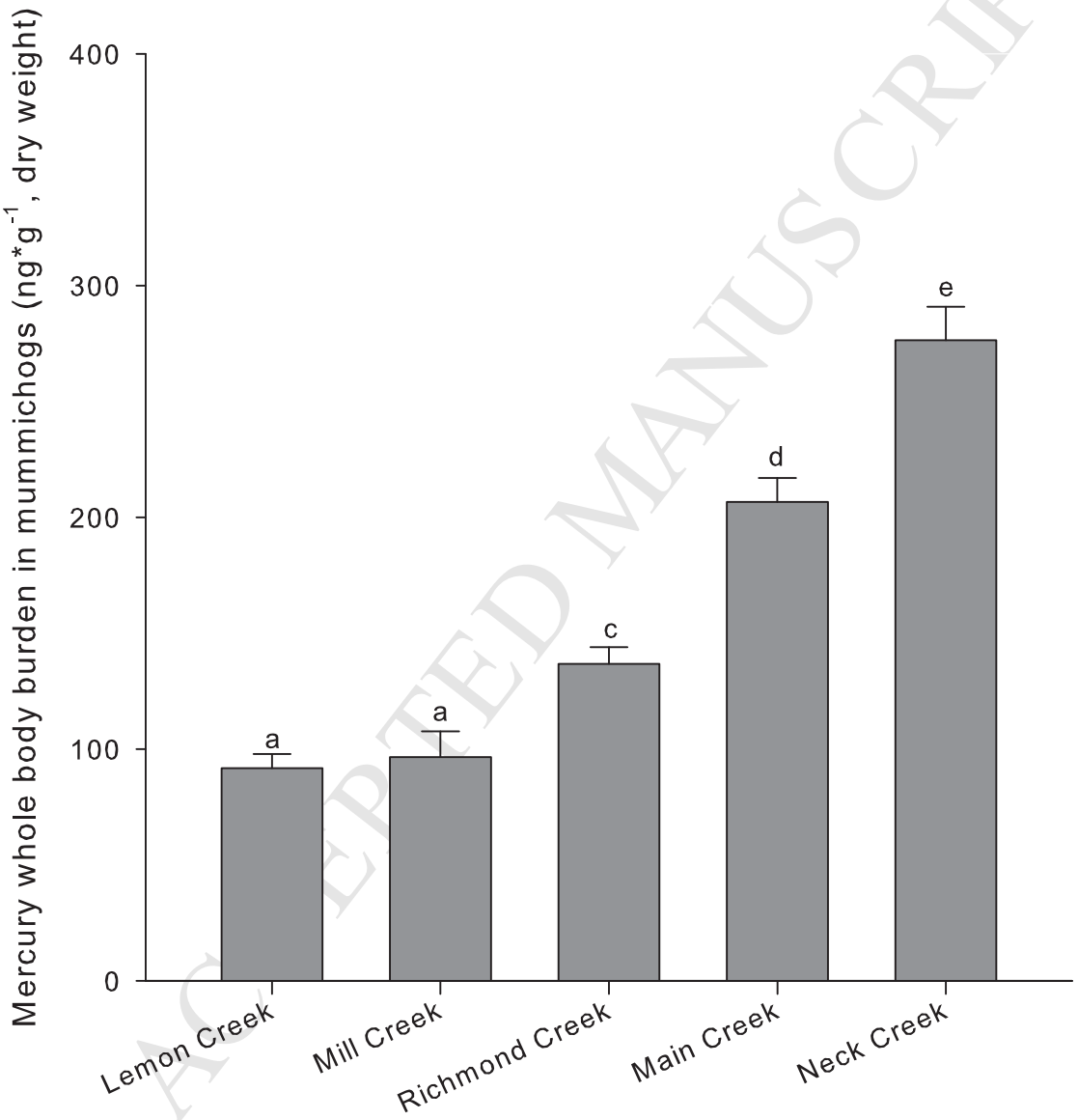


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Figure 7

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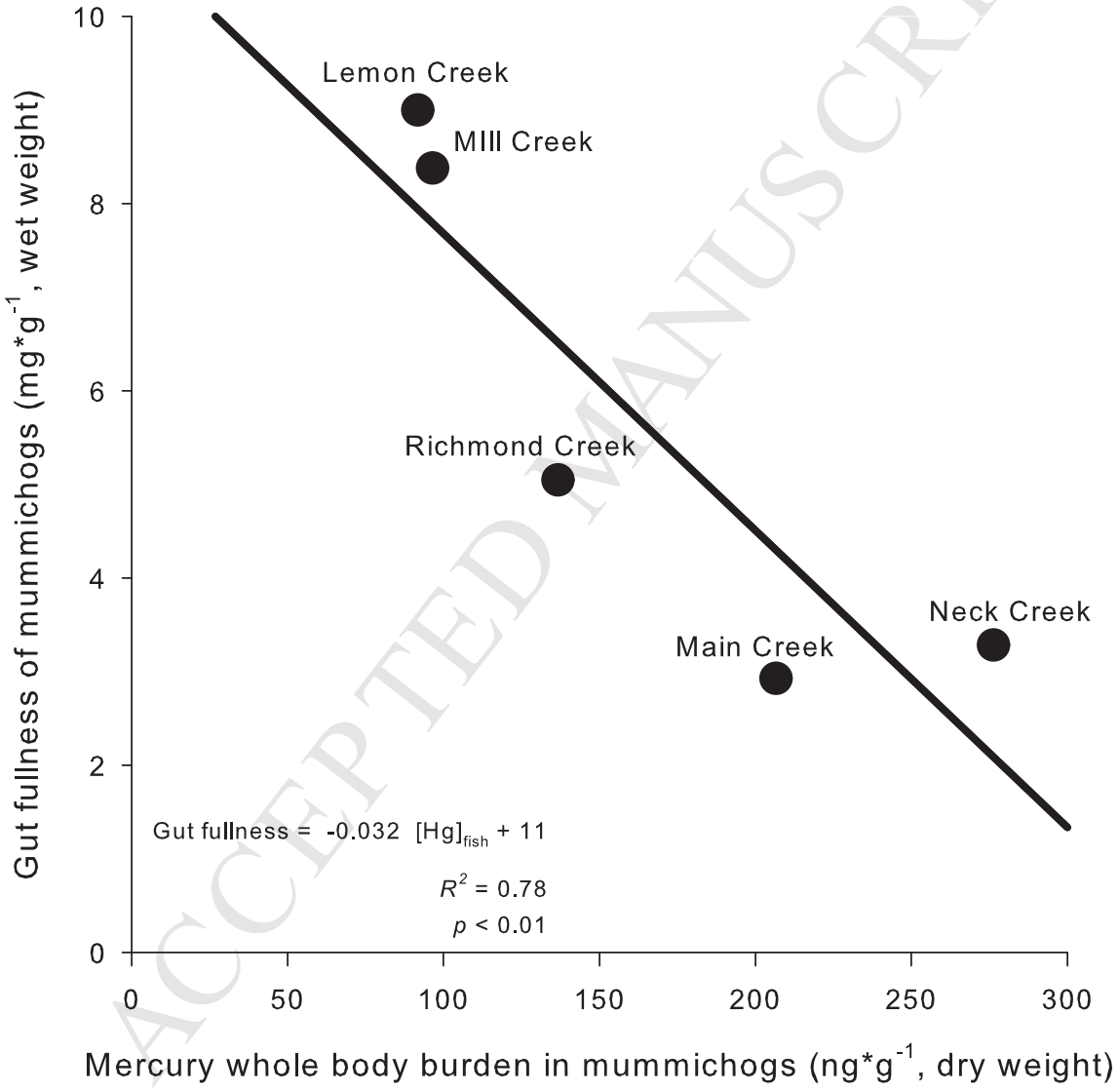


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