

Impact of anthropogenic disturbance on wetland aquatic communities: changes in patterns of  
species abundance and richness as a function of distance from canals

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## SUMMARY

In aquatic systems, a major mechanism for anthropogenic impact has been the alteration of the natural hydrologic regime, including the disruption of natural flow, flood cycles and hydrologic connectivity. In temporary habitats, hydrological alternations can result in increases in water permanence and may convert ephemeral habitats into permanent. Since aquatic organisms segregate strongly along a temporary to permanent habitat gradient, added water permanence can result in important changes in the composition, structure and food-web dynamics of aquatic communities. In this study, we examined the impact of canals on the abundance and richness of Everglades aquatic communities. We sampled fish and invertebrate density and species composition in transects away from canals. At 5 different canal sites, we sampled two parallel 1-km transects, one bisected a vegetated marsh, and the other followed an airboat trail. We found evidence that canals had an effect on the distribution of large predatory fishes, small fishes and invertebrates. However, the effect was only detected in close proximity of canals. All densities increased within 5 m of canals relative to reference marsh sites, and no difference from references was detected at greater distances. Species richness adjusted for density was unaffected for large fish, but decreased as a function of distance from canals for the small fishes and invertebrates. Airboat trails only affected the distribution of large fishes. Large fish abundance was higher in airboat trails. A nutrient gradient may explain increases in abundances in proximity of canals. Future research will include assessment of whether a productivity gradient exist along the study transects.

## INTRODUCTION

Anthropogenic activities have altered freshwater ecosystems worldwide, severely threatening their ecological integrity (NCR 1992, Naiman and Turner 2000). These alterations have resulted in fragmentation and loss of habitat, decline in biodiversity, pollution, and invasion by non-indigenous species (Moyle and Leidy 1992, Allan and Flecker 1993, Warren and Burr 1994, Alford and Richards 1999, Brönmark and Hansson 2002). A major mechanism for impact has been the alteration of the natural hydrologic regimes of aquatic systems (Poff et al. 1997, Dudgeon 2000, Rosenberg et al. 2000). The addition of dams, impoundments, channels, canals, and levees has dramatically disrupted flow, natural flood cycles and the hydrologic connectivity of water bodies (Dynesius and Nilsson 1994, Power et al. 1995, Pringle et al. 2000). In the United States, efforts to satisfy human needs for water consumption, irrigation, power generation, flood control and land conversion, have resulted in the construction of over 2 million dams and 40,000 km of levees and dikes (Johnston 1989, Naiman and Turner 2000). Presently, only 2 % of American rivers flow naturally and close to 50 % of wetlands have been destroyed as a result of drainage and impoundment (Vitousek et al. 1997, Turner et al. 1998).

In temporary habitats, anthropogenic hydrologic alterations often result in changes in the amount, extent and timing of flooding (Gergel 2002). Water control structures may increase water permanence and convert ephemeral habitats into permanent habitats. Aquatic organisms have been shown to segregate strongly along a temporary to permanent habitat gradient (Wilbur 1980, Schneider and Frost 1996, Wellborn et al. 1996, Corti et al. 1997, Skelly et al. 1999). Temporary habitats typically lack predatory species, including fish, and provide a refuge for pond-breeding amphibians and invertebrates. Thus, added water permanence may result in

significant changes in species composition, community structure, and food web dynamics. These artificial, permanent habitats also appear particularly susceptible to invasion by non-indigenous species (e.g., Adams 2000).

In the Florida Everglades, drainage and impoundment have drastically changed natural hydrologic conditions and reduced the extent of shallow, seasonal wetlands by more than 40 % (Davis et al. 1994). Over 1000 miles of canals and 700 miles of levees presently compartmentalize the system and disrupt the historical north to south sheet flow of water over vegetated marshes (Light and Dineen 1994). Historically, the extent of inundation of marshes (and thus habitat for aquatic organisms) fluctuated seasonally solely in response to rainfall patterns (high in the summer and fall, low in the winter and spring). Disturbance from dry down periods in the late spring is known to cause significant fish mortality (Nelson and Loftus 1996), which may limit the abundance of fishes, particularly large predatory species (Loftus and Eklund 1994, Trexler et al. 2004). Today, inundation and flow are largely controlled by humans. In particular, the addition of canals has provided permanent deep-water refuges that historically did not exist (Gunderson and Loftus 1993). These artificial habitats may increase predatory fish densities (Loftus and Kushlan 1987, Howard et al. 1995) and result in important changes in the small fish and invertebrate communities of nearby marshes.

In this study, we examined the impact of canals on Everglades aquatic communities. More specifically, we asked: (1) How does the fish and invertebrate community differ with proximity to canals? (2) Over what distance is the influence of canals seen in these communities? (3) Does the effect differ between undisturbed and disturbed marshes (i.e., marshes impacted by airboat trails)? To address these questions, we sampled fish and invertebrate density and species composition in transects originating at canals and extending into un-channelized marshes. We

were particularly interested in learning how proximity to canals affected the small fish assemblage, which dominates fish standing crops and abundances (Loftus and Eklund 1994, Turner et al. 1999, Trexler et al. 2002) and constitutes an important prey source for wading birds (Frederick and Spalding 1994). If canals provide important habitat for predatory fish species, densities of large fishes may increase with proximity to canals. If predation is an important factor regulating small fish (and invertebrate) density (Kushlan 1976), we would expect their density to decrease in close proximity of canals. Off-road vehicles (including airboats) are known to negatively affect vegetation in Everglades habitats (NPS 2000 and references therein, Welch et al. 2002) and may also impact the distribution and abundance of fishes. Airboat trails may act as conduits for the movement of large predatory species from canals to inner marshes, especially in the dry season. Density of predators may be expected to be greater in these disturbed habitats that appear deeper and less densely vegetated.

## METHODS

Sampling of fish and invertebrates was conducted in September 2003 in transects along 5 canals: I-75, L-28, L-29 and L-67 in Water Conservation Area 3-A and C-111 in the Southern Everglades, Florida, USA (Fig. 1). Two transects were established at each canal site, one that followed an airboat trail and a parallel transect in the surrounding undisturbed marsh (approximately 350 m apart). Both transects originated at a canal and ran perpendicular for 1000 m. Sampling was conducted at 5, 100, 500, and 1000 m along each transect. Transects typically bisected *Eleocharis* spp.-dominated wet prairies, which constitute key habitats for small-bodied fishes (Gunderson and Loftus 1993, Loftus and Eklund 1994, Jordan et al. 1997).

Large fish (SL > 80 mm) were sampled using an airboat-mounted electrofishing unit (two-anode one-cathode apparatus with a Smith-Root GPP 9.0 control box). Electrofishing is an effective method for sampling large fish in marshes and electrofishing catch per unit effort (CPUE) provides a reliable index of fish abundance (Reynolds 1983, Burkhardt and Gutreuter 1995, Chick et al. 1999). Sampling was conducted in 2 5-minute bouts (pedal time) at each distance within each transect. For all bouts, electrofishing power was standardized at 1500 watts according to temperature and conductivity conditions (see Chick et al. 1999 for further details). All captured fish were placed in a holding tank, identified to species, measured (to nearest 1 mm SL), and released after full recovery.

Small-bodied fish (SL < 80 mm) and macroinvertebrates (length > 5 mm) were sampled with a 1-m<sup>2</sup> throw trap (Kushlan 1981, Jordan et al. 1997). Throw traps consisted of a square copper-pipe cage with sides covered by 2 mm mesh netting (no top or bottom). The trap was thrown into the water column, secured to the substrate by slightly pressing on it, and cleared with a combination of a bar seine (1 m length, 2 mm mesh) and two dipnets (1 mm and 5 mm mesh). We standardized sampling effort by seining until three consecutive seine hauls yielded no animals and then by sweeping until 10 consecutive empty sweeps were obtained. Three throw trap samples were taken at each distance within each transect. All animals were handpicked from the seine, preserved in 10 % formalin and brought to the laboratory for identification and enumeration.

### *Statistical analyses*

We examined variation in the abundance and richness of aquatic organisms as a function of distance from canals, transect type (marsh vs. trail), and sites (5 canals) with a repeated-measures ANOVA. Focal response variables included: mean CPUE of large fish (in number/5

min. electrofishing and averaged over the two bouts), mean densities of small fish, of the three most abundant small fish (least killifish *Heterandria formosa*, eastern mosquitofish *Gambusia holbrooki*, and bluefin killifish *Lucania goodei*) and of macroinvertebrates (all in numbers/m<sup>2</sup> and averaged over the three throw traps), and mean species richness measures. Species richness was determined separately for the large fish, small fish and macroinvertebrates. Richness measures were calculated as an index by dividing the total number of taxa obtained in each sample by the density or density estimate (CPUE) of the sample (Menhinick 1964). Taxonomic resolution was to the species level for all taxa except for 10 of the 32 macroinvertebrate groups found in the study.

We compared physical parameters (i.e., water depth, percent plant cover at the water surface, emerging stem density and periphyton volume) among distances, transects and canals with the same repeated measures ANOVA's model. In order to better satisfy assumptions of parametric tests, all variables were log-transformed ( $\ln$  of observed value + 1) prior to analyses. Post hoc multiple comparisons were performed using Tukey's tests. All analyses were performed using the mixed model procedure in SAS®.

## RESULTS

### *Physical parameters*

During the course of sampling (September 2003 wet season), water depths averaged 70 cm across study sites. Depth did not vary significantly with distance from canals or between the airboat trail and marsh transects (Fig. 2a) but varied among canals (Table 1). Depth was significantly lower at the C-111 canal transects (36 cm) in the Southern Everglades than at the

other four sites (66 cm) in WCA 3A (Tukey's pairwise comparisons, all  $p < 0.005$ ). Plant cover decreased with distance from canals, from 35 % in the immediate proximity of canals to less than 20 % at greater distances (5 m vs. 100 m,  $p = 0.039$ ; 5 m vs. 1000 m,  $p = 0.016$ ) (Fig. 2b). This is likely explained by the replacement of spike rush (*Eleocharis* spp.) and grasses (*Panicum*, *Paspalidium* spp.), which were the dominant vegetation in the inner marsh, with greater surface-area water lilies at the 5 m locations (i.e., *Nuphar*, *Nymphaea* and *Nolumbo* spp.). Stem density was unaffected by distance, but periphyton volume was significantly lower at 5 m relative to greater distances (5 m vs. 100 m,  $p = 0.066$ ; 5 m vs. 500,  $p = 0.016$ ; 5 m vs. 1000 m,  $p = 0.010$ ) (Table 1, Fig. 2c, d). None of the three vegetation measures was affected by transect type. Stem density varied among canals (Table 1); it was highest at the C-111 and I-75 canal transects and lowest at the L67 transects (C-111 vs. L-67,  $p = 0.003$ ; I-75 vs. L-67,  $p = 0.002$ ).

### *Large fish*

Large fish CPUE significantly varied with distance away from canals (Table 2). CPUE was greater at 5 m from canals than at greater distances (Fig. 3a). CPUE at 5 m averaged over 4 fish (SL > 8.0 cm) per 5 min electrofishing compared to less than 2 fish per 5 min at 100, 500, and 1000 m combined (5 m vs. 100 m,  $p = 0.0005$ ; 5 m vs. 500 m,  $p = 0.033$ ; 5 m vs. 1000 m,  $p = 0.0091$ ). CPUE was also greater in the airboat trail transect than in the nearby undisturbed marsh (Table 2). CPUE in the marsh transect averaged less than 1 fish/5 min. electrofishing compared to over 3 fish/5 min. electrofishing in the trail transect. Large fish species richness was not affected by either distance from canals or transect type (Table 2, Fig. 3b). Neither large fish density nor richness varied among sites (Table 2).

### *Small fish and macroinvertebrates*

Small fish density and richness were affected by distance from canals but interestingly not in the same manner (Table 3). Density increased close to canals, whereas richness decreased. Density within 5 m of canals averaged 39 fish (SL < 8 cm) per m<sup>2</sup> compared to only about 12 fish per m<sup>2</sup> at 500 and 1000 m away from canals (5 m vs. 500 m,  $p = 0.025$ ; 5 m vs. 1000 m,  $p = 0.10$ ) (Fig. 3c). Richness decreased from 0.43 species per specimens caught at 500/1000 m away to 0.25 within 5 m of canals (5 m vs. 500 m,  $p = 0.025$ ; 5 m vs. 1000 m,  $p = 0.10$ ) (Fig. 3d). Species richness also varied with distance differently between the two transects (significant distance x transect interaction, Table 3). Richness differed between 5 m and the 500/1000 m distances only in the marsh transect, not in the trail transect (Marsh: 5 m vs. 500 m,  $p = 0.042$ ; 5 m vs. 1000 m,  $p = 0.016$ ; Trail: 5 m vs. 500 m and 5 m vs. 1000 m,  $p = 1.00$ ). Small fish density was unaffected by transect type (Table 3).

*H. formosa*, *G. holbrooki* and *L. goodei* accounted for 78 % of the total catch in throw traps, whereas 8 % consisted of juvenile centrarchids, including sunfishes (*Lepomis*, *Elassoma* and *Enneacanthus* spp.) and largemouth bass (*Micropterus salmoides*). Distance affected the densities of *H. formosa* and *L. goodei*, but not *G. holbrooki* or centrarchid densities (Fig. 4). Densities of *H. formosa* and *L. goodei* decreased from 13 and 6 fish per m<sup>2</sup> respectively at 5 m from canals to 6 and 2 fish per m<sup>2</sup> at 500 m from canals (*H. formosa*: 5 m vs. 500 m,  $p = 0.039$ ; 5 m vs. 1000 m,  $p = 0.040$ ; *L. goodei*: 5 m vs. 500 m,  $p = 0.029$ ) (Fig. 4a, b). No differences were detected at other distances. Densities of the three species and juvenile centrarchids were similar between trail and marsh habitats (Table 3). However, a transect by distance interaction was detected for the most abundant species, *H. formosa*. The density of *H. formosa* decreased significantly with distance only in the marsh transect but not in the trail transect (Marsh: 5 m vs.

500 m,  $p = 0.030$ ; 5 m vs. 1000 m,  $p = 0.021$ ; Trail: 5 m vs. 500 m and 5 m vs. 1000 m,  $p = 1.00$ ) (Fig. 4a).

Similar to the small fish results, the density of invertebrates increased whereas the number of taxa decreased as a function of distance from canals (Fig. 5). Invertebrate density decreased from 70 to 20 organisms/m<sup>2</sup> from the immediate proximity of canals to greater distances (5 m. vs. 100 m,  $p = 0.069$ ; 5 m vs. 500 m,  $p = 0.068$ ; 5 m vs. 1000 m,  $p = 0.032$ ). The grass shrimp *Palaemonetes paludosus* was the most abundant taxon (76 % of total catch), while odonate naiads, hemipterans, gastropods and decapods (in order of abundance) accounted for the remaining 24 % of the total catch. Richness was significantly lower at 5 m than at 100 and 1000 m from canals (5 m. vs. 100 m,  $p = 0.054$ ; 5 m vs. 1000 m,  $p = 0.040$ ) (Fig. 5b). Transect type had no effect on either invertebrate density or richness (Table 4).

We detected significant site effects on small fish and invertebrate densities, small fish species richness and *H. formosa* density (Tables 3, 4). Densities of small fish tended to be rather low at the L-29 canal transects, at least when compared to the I-75 and L-67 canal transects (L-29 vs. I-75,  $p = 0.060$ ; L-29 vs. L-67,  $p = 0.080$ ) (Fig. 6a). Similarly, densities of *H. formosa* were lower at the L-29 transects than at the I-75 and L-67 transects (L-29 vs. I-75,  $p = 0.010$ ; L-29 vs. L-67,  $p = 0.045$ ). For the invertebrates, densities were lower at the C-111 relative to the L-67 and I-75 sites (C-111 vs. L-67,  $p = 0.025$ ; C-111 vs. I-75,  $p = 0.028$ ) (Fig. 6b). Small fish species richness showed the opposite pattern. Richness was higher at the L-29 and C-111 canals than at the L-28, L-67 and I-75 sites ( $p < 0.03$  all comparisons).

## DISCUSSION

We found evidence that proximity to man-made canals had an effect on the distribution of aquatic organisms inhabiting Everglades marshes. As expected, the abundance of large, predatory fish increased near canals. Canals provide large areas of open, deep-water habitat for large fish species (Gunderson and Loftus 1993). These fishes are known to move great distances to take refuge in canals as water levels drop in the dry season (J. Trexler, unpublished radiotelemetry data), congregating at high densities until water levels increase in the wet season, when they may return to marshes. However, densities of predators may remain high during the wet season in canals (Trexler et al. 2002), making them function as sinks for predators with respect to marshes (Loftus and Kushlan 1987). Our results from sampling in the wet season agree with this notion. However, the effect of canals on predator abundance was seen over rather short distances. Predator abundances increased sharply only within 5 m of canals.

Surprisingly, the abundances of small-bodied fishes and invertebrates also increased in the close proximity of canals. There are several plausible explanations for this result. The hydrologic pattern of recurrent dry down is a strong limiting factor on the abundance of aquatic organisms (Kushlan 1976, Loftus and Eklund 1994, Trexler et al. 2004). Thus, access to dry season refugia is a key factor affecting the survival of fishes and invertebrates and their abundances (Turner et al. 1999). Added water permanence provided by canals, while increasing the density of predators, may also provide dry down refugia for the smaller fish and invertebrates and lessen the regulatory effect of hydrology on their populations. Furthermore, species that appear to be strongly regulated by hydrological patterns seemed to be the ones becoming more abundant in the vicinity of canals. Abundances of *H. formosa* and *L. goodei* increased in close

proximity of canals, whereas no effect was detected for *G. holbrooki*. Trexler et al. (2001) found a similar result in a previous transects study. Ruetz et al. (2004) have shown that dry down events have strong effects on densities of *L. goodei* and *H. formosa*, whereas the effect is weak on densities of *G. holbrooki*.

The fact that consumer densities still increase in spite of higher densities of predators may provide additional support to the idea that Everglades consumer populations may not be limited by piscivorous fishes (Eklund and Loftus 1994). Kushlan (1976) suggested that fish predators have an effect on prey densities only when the frequency of disturbance (dry down events) is low as is predicted by theory (Menge and Sutherland 1987). Canals may be able to artificially nullify or at least lessen the limiting effect of hydrology and allow for a top-down effect on small fish and invertebrates. However, our study did not yield any evidence of such an effect, providing further support to the idea that predator regulation is not an important force structuring the Everglades aquatic community.

Increases in abundances of organisms may result in response to changes in basal productivity. Although both theoretical and empirical evidence show that increases in productivity may cause a variety of changes in community structure (Power 1992, Abrams 1993), increases in consumer abundance have been reported in response to enrichment (Rader and Richardson 1992, Turner et al. 1999, Trexler et al. 2001) and as a function of natural variation in nutrient availability in Everglades communities (Trexler et al. 2002). The Everglades are a naturally oligotrophic system, characterized by low phosphorus availability (Davis 1994). Phosphorus addition, associated with agricultural runoff from canals, has resulted in gradients in enrichment as a function of distance from these sources (Davis 1994, Doren et al. 1997, Childers et al. 2003). However, these gradients have been documented at greater spatial scales (e.g., 1-10

km) that those examined by our study. It is not clear whether a gradient exists in the 5 m to 1000 m distance range. To elucidate this issue, future work includes the collection of soil nutrient data, specifically total phosphorus from the flocculent detrital layer, along all sampling sites in our transects.

Higher productivity may not only affect the abundance of organisms but also their diversity. Generally, productivity and diversity can relate by a unimodal relationship (Leibold et al. 1997). Diversity may increase with increases in productivity, but it decreases at the highest levels of productivity. In our study, small fish and invertebrate diversities both decreased in close proximity of canals, whereas no effect was detected on the diversity of predatory species. If productivity is very high within 5 m of canals, it may explain this decrease in small fish and invertebrate diversity. However, we might expect a similar effect on the richness of predatory fish. Another explanation might be that selective predation by large fishes is negatively affecting diversity. Only prey species that can coexist with high densities of predators are found in close proximity of canals. Further research is needed to better understand the mechanism behind these differences in species richness as a function of distance from canals.

ORV use in the Everglades has resulted in soil displacement, changes in water flow and plant community composition (NPS 2000). In our study, disturbance from airboat trails had no effect on the small fish and invertebrate assemblages, but affected the abundances of predatory fish. Greater numbers of predatory fish were found along airboat trails, suggesting that trails may be preferred habitats. Habitat selection in fishes is known to be affected by resource availability and profitability (Werner and Hall 1976); however, prey densities were not greater in trail transects. Nevertheless, it is plausible that prey profitability was in fact higher in trails due to a greater ability to capture prey in trails relative to undisturbed marshes. Variation in habitat

structural complexity can strongly influence the ability of predators to catch prey (Crowder and Cooper 1982, Savino and Stein 1989). Although we did not detect large differences between transects in the vegetation characteristics described here (e.g., surface plant cover and stem density), we suspect differences in plant composition may be affecting predatory fish distribution (J.S. Rehage, personal observation). Future work will include more detailed analyses of compositional vegetation differences between transect types in order to clarify this issue.

Man-made structures are a permanent feature of the Everglades ecosystem. Ongoing restoration efforts call for the removal of only 240 miles of levees and canals (CERP 1999). Thus, the study of the impact of canals on fish communities and on other components of the ecosystem is a critical component of our understanding of Everglades ecology and has important implications for ongoing restoration efforts. Gaining a clear understanding of ecological patterns in pulse systems like the Everglades also requires research to address how ecological processes respond to seasonal hydrological patterns. Ongoing work includes sampling the study transects again in the dry season. We expect the effect of canals to increase in magnitude as predator densities increase in canals in response to dry-down conditions. Perhaps the distance over which we note changes in fish community composition will also increase. In addition, we plan to compare patterns of abundance and diversity from transects to interior sites that are part of our long-term monitoring study.

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Table 1. Results from repeated measures ANOVA's testing effects of distance, transect and site on physical parameters: water depth, percent plant cover, stem density (number/m<sup>2</sup>) and periphyton volume.

<i>Source of Variation</i>	Water depth			% Plant cover		Stem density		Periphyton volume	
	<i>df</i>	<i>F</i>	<i>p</i>	<i>F</i>	<i>p</i>	<i>F</i>	<i>p</i>	<i>F</i>	<i>p</i>
Distance	3, 12	0.01	0.998	6.0	0.011	1.5	0.271	6.3	0.012
Transect	1, 4	3.6	0.130	0.3	0.646	0.01	0.936	1.7	0.275
Site	4, 4	60.2	0.0007	0.4	0.800	18.0	0.008	2.7	0.202
Distance x Transect	3, 12	1.2	0.434	2.5	0.118	3.4	0.053	0.6	0.623
Distance x Site	12, 12	8.4	0.034	2.8	0.053	3.1	0.033	1.4	0.311

Table 2. Results from repeated measures ANOVA's testing effects of distance, transect and site on large fish CPUE (number/5 min. electrofishing) and large fish richness.

<i>Source of Variation</i>	CPUE			Species richness	
	<i>df</i>	<i>F</i>	<i>p</i>	<i>F</i>	<i>p</i>
Distance	3, 12	11.2	0.0009	0.4	0.729
Transect	1, 4	13.9	0.020	1.2	0.328
Site	4, 4	3.5	0.129	0.6	0.677
Distance x Transect	3, 12	1.0	0.436	0.7	0.572
Distance x Site	12, 12	1.9	0.149	0.6	0.824

Table 3. Results from repeated measures ANOVA's testing effects of distance, transect and site on small fish density, small fish species richness, and *H. formosa*, *G. holbrooki*, *L. goodei* and juvenile Centrarchid (sunfishes and largemouth bass) density.

Source of Variation	df	Small fish density		Small fish richness		<i>H. formosa</i>		<i>G. holbrooki</i>		<i>L. goodei</i>		Centrarchids	
		F	p	F	p	F	p	F	p	F	p	F	p
Distance	3, 12	4.0	0.038	4.1	0.032	4.0	0.037	1.8	0.204	4.0	0.040	1.0	0.439
Transect	1, 4	0.5	0.500	2.8	0.167	0.01	0.917	0.2	0.701	1.0	0.389	0.03	0.865
Site	4, 4	5.9	0.040	10.1	0.023	11.7	0.008	3.8	0.107	0.7	0.652	5.5	0.065
Distance x Transect	3, 12	2.6	0.11	3.6	0.048	3.5	0.051	1.5	0.264	2.8	0.096	1.2	0.351
Distance x Site	12, 12	1.6	0.22	0.9	0.549	2.9	0.046	0.9	0.554	1.3	0.343	0.5	0.851

Table 4. Results from repeated measures ANOVA's testing effects of distance, transect and site on invertebrate density (number/m<sup>2</sup>) and invertebrate species richness.

<i>Source of Variation</i>	Invertebrate density			Species richness	
	<i>df</i>	<i>F</i>	<i>p</i>	<i>F</i>	<i>p</i>
Distance	3, 12	4.4	0.027	4.1	0.032
Transect	1, 4	4.5	0.102	1.9	0.240
Site	4, 4	11.0	0.020	5.2	0.070
Distance x Transect	3, 12	1.2	0.350	1.2	0.366
Distance x Site	12, 12	1.1	0.437	1.2	0.385

## FIGURES

Fig. 1. Map of the 5 sampling sites (I-75, L-28, L-29, L-67 in WCA-3A and C-111 in the Southern Everglades) and diagram of the two study transects (undisturbed marsh and airboat trail) sampled at each site (approximately 350 m apart).

Fig. 2. Physical parameters measured in 1-m<sup>2</sup> quadrats along marsh and trail transects: (a) water depth (cm), (b) percent plant cover, (c) stem density (number/m<sup>2</sup>), and (d) periphyton volume (mL). Data are means  $\pm$  1 SE.

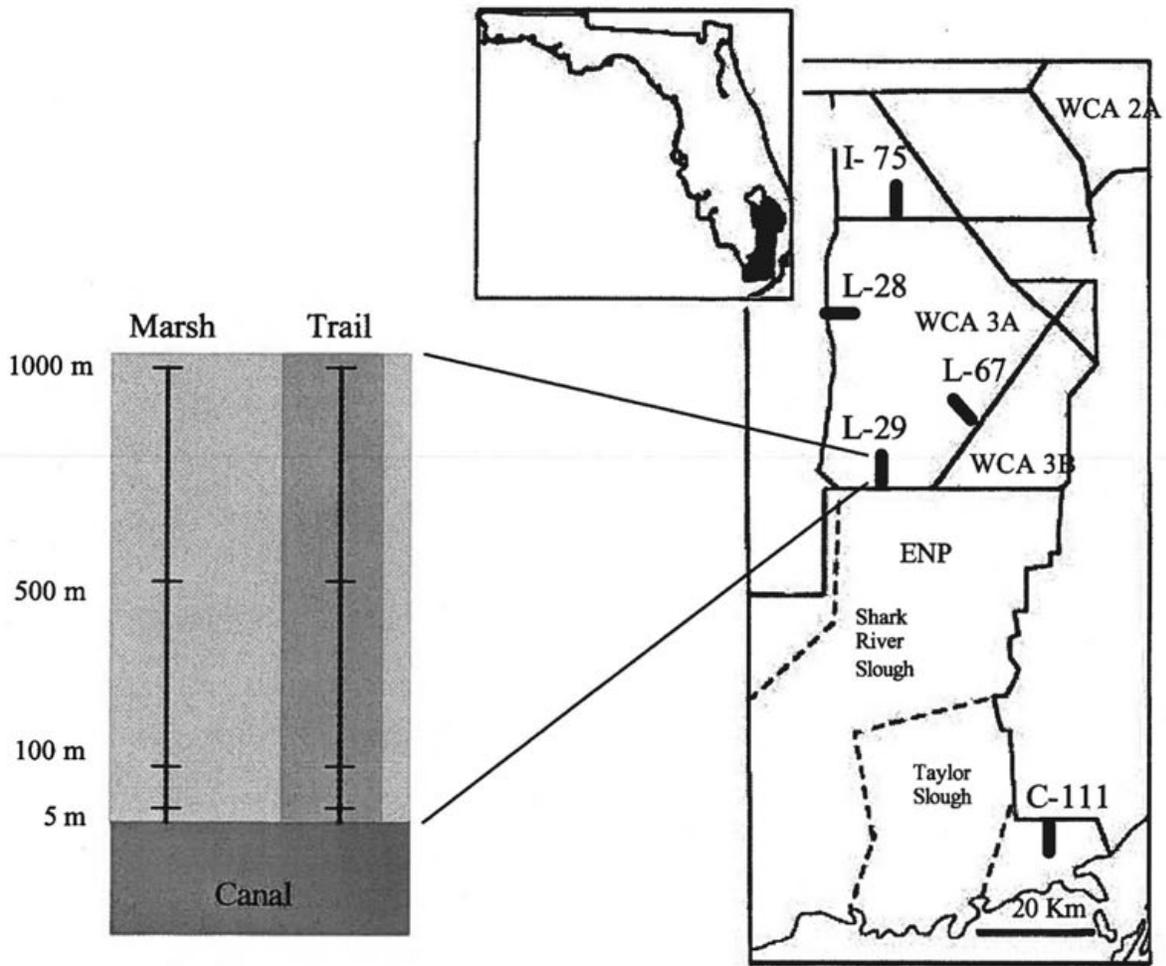
Fig. 3. (a) Large fish (SL > 8 cm) CPUE (number/5 min. electrofishing), (b) large fish species richness, (c) small fish (SL < 8 cm) density (number/m<sup>2</sup>), and (d) small fish species richness along marsh and trail transects. Species richness was calculated as the number of species per sample/number of specimens per sample. Data are means  $\pm$  1 SE.

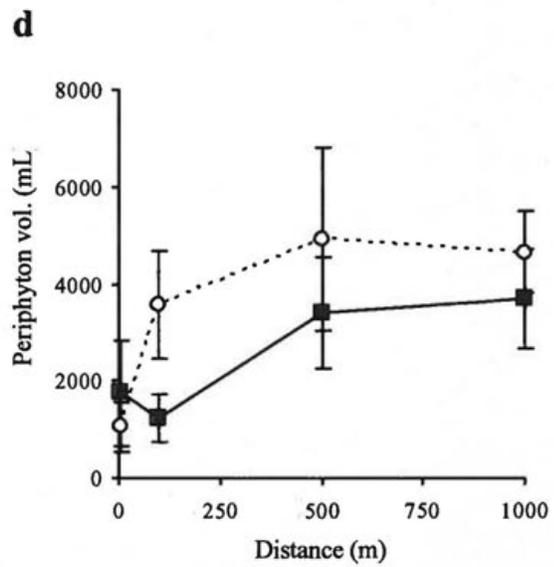
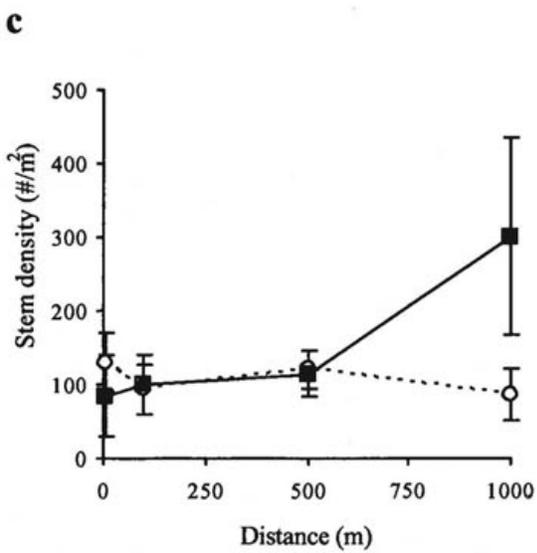
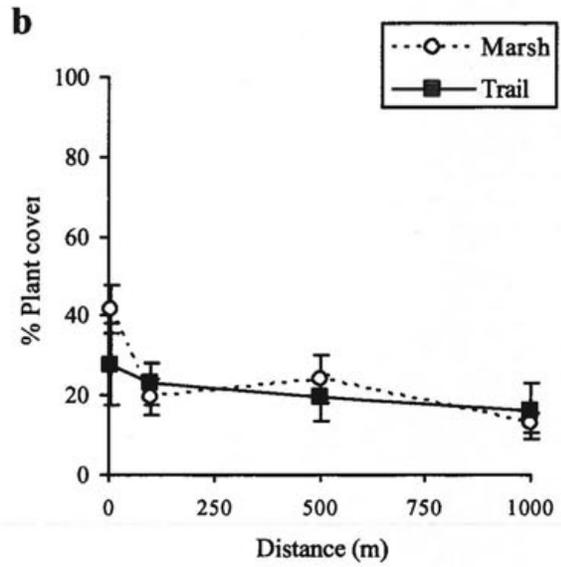
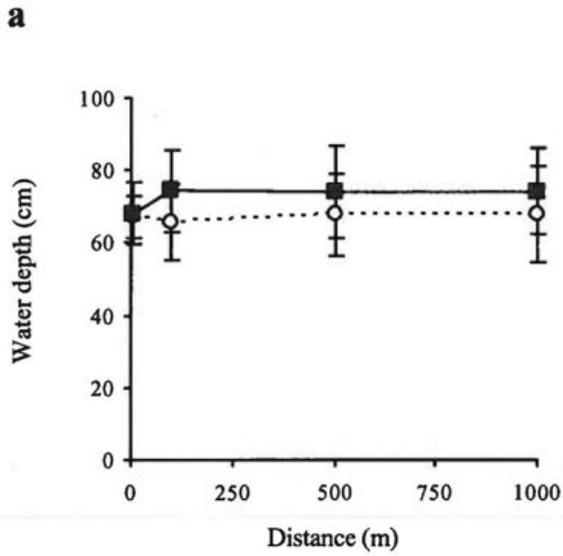
Fig. 4. (a) *H. formosa*, (b) *G. holbrooki*, (c) *L. goodei*, and (d) Centrarchid (sunfishes and largemouth bass) densities (number/m<sup>2</sup>) collected in throwtraps along marsh and trail transects. Data are means  $\pm$  1 SE.

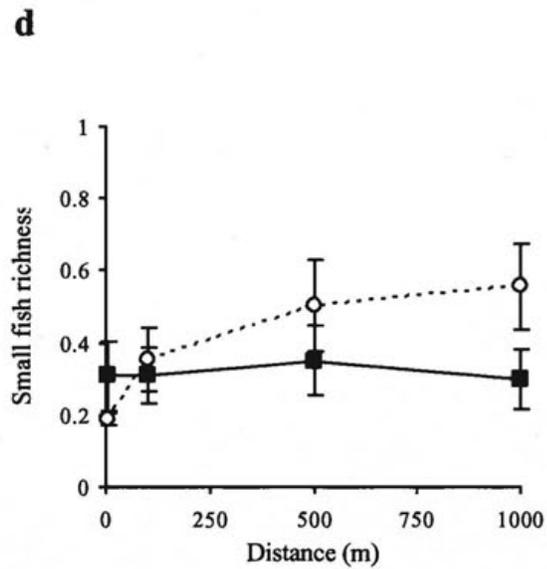
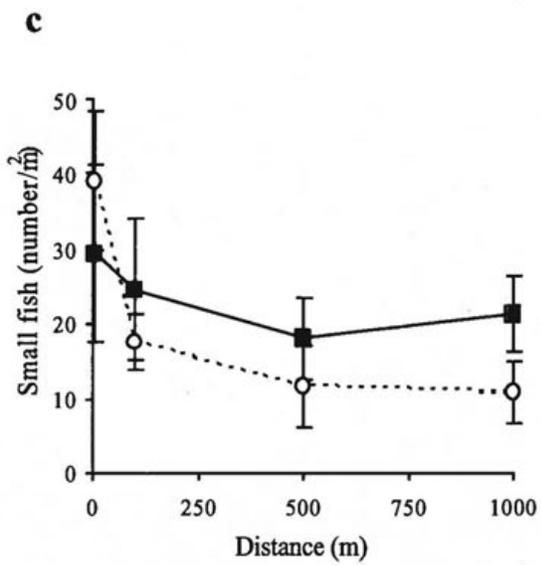
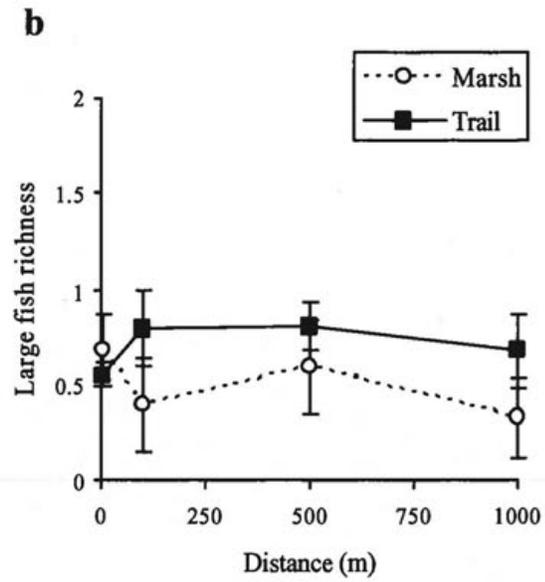
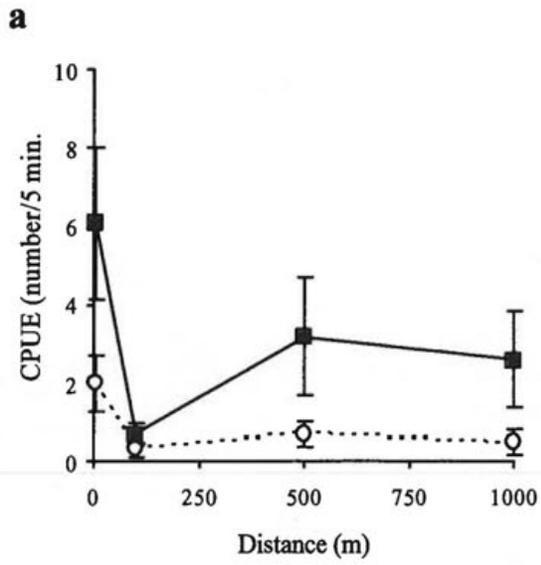
Fig. 5. (a) Macroinvertebrate density (number/m<sup>2</sup>), and (d) richness along marsh and trail transects. Richness was calculated as the number of taxa per sample/number of specimens per sample. Data are means  $\pm$  1 SE.

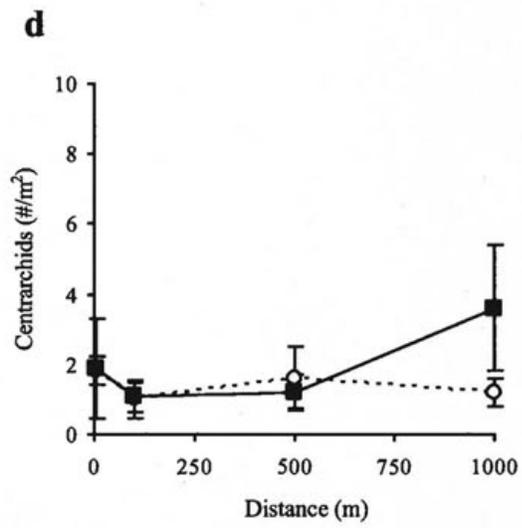
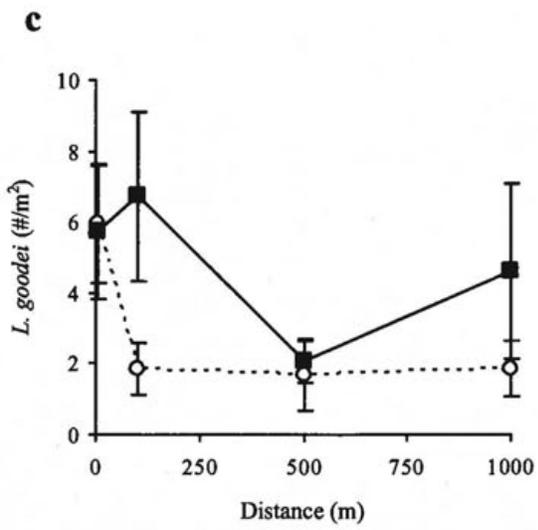
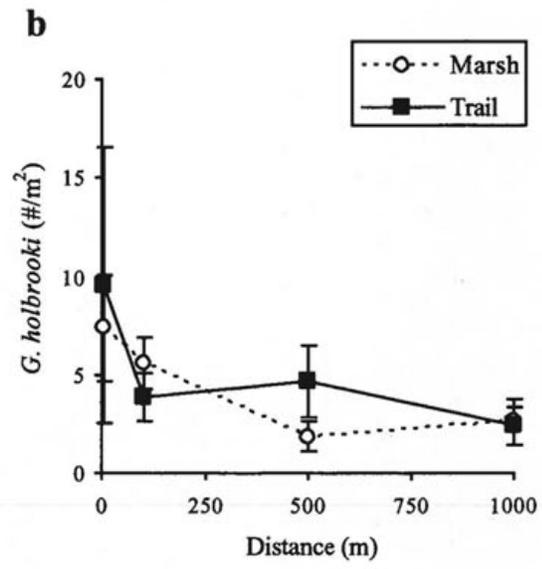
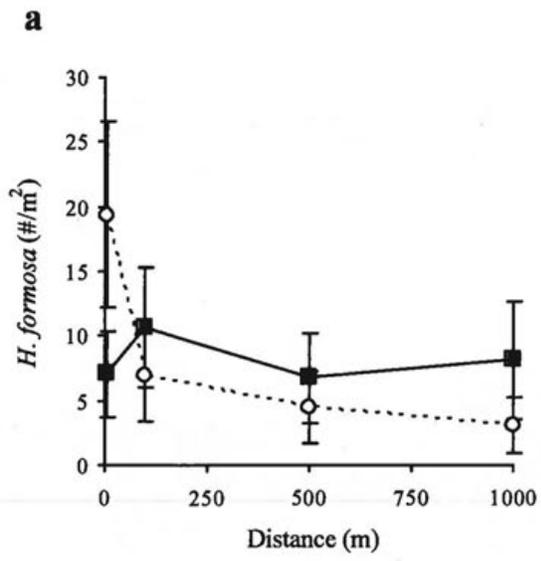
Fig. 6. (a) Small fish and (b) invertebrate density (numbers per m<sup>2</sup>) at each canal site. Data are means ( $\pm 1$  SE) of all distances at both transects.

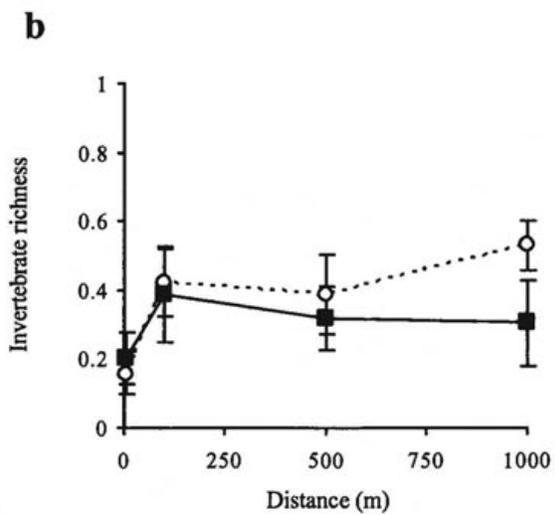
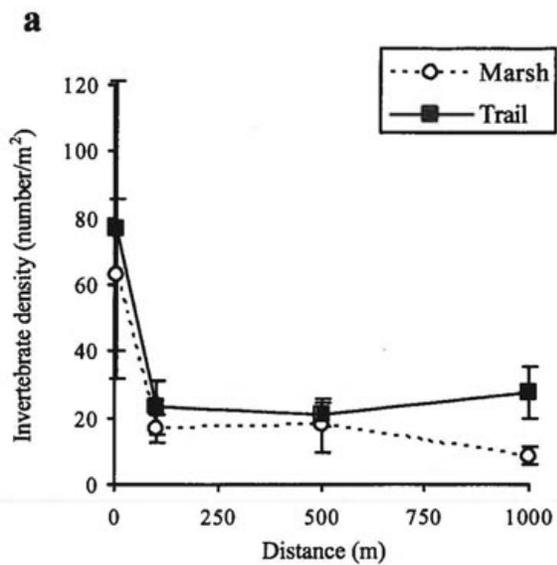
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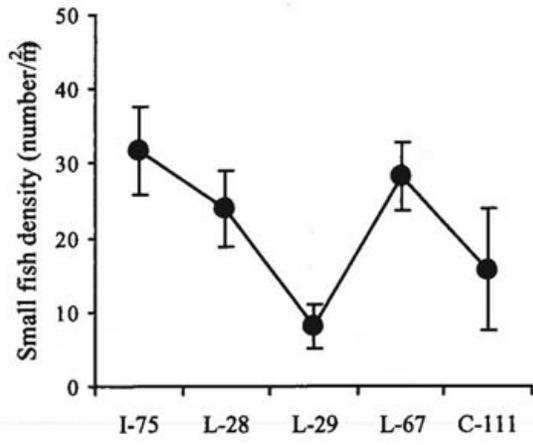








**a**



**b**

