**BIOASSESSMENT OF WETLANDS USING INVERTEBRATES: STUDIES FROM THE SOUTHEASTERN U.S.**

by

BAGIE MARIAM GEORGE

(Under the direction of Darold Batzer)

ABSTRACT

In order to assess the efficacy of using invertebrates to assess impacts on wetlands, three wetland habitats were studied. In the Okefenokee Swamp, invertebrates were used to determine concentrations of the heavy metal mercury. Levels of mercury were higher in amphipods than either odonates or crayfish. Invertebrates were useful for monitoring temporal changes in mercury bioavailability. In a forested floodplain system, the impacts of logging on invertebrates were determined. Numbers of several invertebrate taxa decreased in harvested treatments as compared to control treatments, while others increased. In order to determine the effects of peripheral tree harvest on invertebrate numbers, isolated depressional wetlands were studied. Invertebrate numbers did not change in depressional wetlands after peripheral logging.

INDEX WORDS: Wetlands, Invertebrates, Mercury, Logging

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by

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## DEDICATION

*To my parents, C.A. and Mariam George, and my brother and sister, Agie and Cigie George for your endless support, guidance and love.*

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# CHAPTER 1

## INTRODUCTION AND LITERATURE REIVEW

Because of their abundance and diversity in freshwater habitats, aquatic invertebrates have been useful organisms for biologically monitoring streams, rivers, and lakes. Biomonitoring is defined as using biological responses to determine changes in the environment (Hare 1992; Rosenberg and Resh 1993). Measuring fluctuations in diversity and abundance of invertebrates is one method to determine impacts. Others include changes in genetic composition and bioaccumulation of toxicants. Advantages of using invertebrates for biomonitoring include: 1) invertebrates are ubiquitous and occur in large numbers throughout many habitats, 2) they are easy to sample, and 3) invertebrates are easy to manipulate in studies (Rosenberg and Resh 1993).

Although the use of invertebrates as biomonitors is now common in rivers, streams, and lakes, aquatic invertebrate biomonitoring is just beginning to be used in wetland habitats (Rader et al. 2001). This study looks at three different wetland systems and assesses the impacts of anthropogenic manipulations on the invertebrates. The first study focuses on spatial and temporal variations of mercury levels in Okefenokee invertebrates, the second on the direct effects on invertebrates of logging in bottomland hardwood forest wetlands, and the last on the indirect effects on invertebrates of peripheral logging around (but not in) depressional wetlands.

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# CHAPTER 2

# INVERTEBRATES AS BIOINDICATORS OF MERCURY LEVELS IN THE OKEFENOKEE SWAMP OF SOUTHEAST GEORGIA

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## INTRODUCTION

Wetlands are often sinks for heavy metals (Moore et al 1995, Rood 1996, St. Louis et al. 1996, Heyes et al. 1998, Niamo et al. 2000). Elemental mercury and methyl mercury are heavy metals of particular concern to humans. Elemental mercury poses little risk but it becomes an issue when it is methylated to form methyl mercury (Morel 1998). This form of mercury is a neurotoxin that poses serious problems to ecosystems. Methyl mercury is frequently produced at high rates in wetlands (St. Louis et al. 1994).

The bioavailability of mercury in wetland aquatic environments appears to be dependent on water temperature, dissolved oxygen levels, and hydrology. High temperature reduces dissolved oxygen level, which in turn enhances methylation because mercury bound to sediment is released into the water column when oxygen levels are low (Henry et al. 1995). If dissolved oxygen levels remain high the release of methyl mercury from sediment is reduced (Henry et al. 1995). Intermittent or shallow flooding can increase methyl mercury bioavailability to organisms (Morel 1998). Mercury levels were higher in fish collected from South Carolina wetlands that had frequent fluctuations in water levels than those that were deep and permanent (Snodgrass et al. 2000).

Although mercury is often found in wetlands, the source of contamination is not easily determined. Both anthropogenic and natural processes are possible sources of bioavailable mercury. Some of the greatest contaminant loads of mercury detected from wetlands have been from the Florida Everglades (Facemire 1995, Rood 1996), where peat and natural mineral deposits are possible sources of mercury contamination (Rood 1996, Vaithiyanathan et al. 1996). In the Okefenokee Swamp, decomposition of peat is considered to be an important natural source of methyl mercury (Porter et al. 1999).

Possible external anthropogenic inputs of mercury into wetlands include burning of fossil fuels, medical waste incineration, agriculture, and mining (Rood et al. 1995).

There is increasing evidence that anthropogenic emissions significantly increase mercury levels in precipitation (Rolfhus & Fitzgerald 1995, Keeler et al. 1995). In precipitation, however, methyl mercury constitutes a small part of total mercury, typically < 1% of total concentrations (Bloom & Watras 1989, Downs et al. 1998). The mercury in precipitation becomes a problem if it accumulates where conditions suitable for methylation develop. Besides input in precipitation, some mercury may enter wetland habitats via runoff, and a positive correlation has been found between sediment mercury concentrations and watershed area (Wiener et al. 1990).

In wetland and aquatic habitats, mercury can be closely bound to sediments or particulate matter in the water column. Sediment is thought to be the major source of methyl mercury available to both aquatic and terrestrial vertebrates (Larsorsa & Allen-Gill 1995, Tremblay et al. 1996). Various factors can affect sediment mercury concentrations. In lakes, for example, a decrease in pH enhances release of mercury found in the sediment (French et al. 1999). Variations in temperature, dissolved oxygen, depth, and pH all affect the uptake of mercury by various organisms from sediment (Downs et al. 1998, Rood 1996).

Mercury in diets probably has more influence on bioaccumulation of mercury in vertebrates than direct exposure to mercury in the water column (Bhattacharya & Sarkar 1991, Downs et al. 1998). Top aquatic predators depend either directly or indirectly on plants and invertebrates. Several wetland plants sequester mercury and various levels of accumulation have been recorded from plant tissues. The uptake of mercury from

sediments by plants may contribute to increased mercury bioavailability (Rood et al. 1995). Submersed species of aquatic plants can sequester high levels of mercury (Thompson-Roberts et al. 1999), and bryophytes, both feather and *Sphagnum* mosses, sequester the highest levels of mercury of any plants recorded. However, many common wetland species such as the yellow water lily sequester minimal amounts of mercury in their tissues (Thompson-Roberts et al. 1999), and the leaves of most trees and shrubs have very low levels of mercury (Moore et al. 1995). Decaying aquatic vegetation may play a role in increasing mercury levels in wetlands. As plant tissues decompose under anoxic conditions, methyl mercury is released from the methylation of inorganic mercury (Heyes et al. 1998).

Invertebrates are used frequently as bioindicators of heavy metal contamination. For example, mollusks are used as bioindicators of contamination from an assortment of heavy metals (Eiseman et al. 1997). However, relatively little research has been conducted on bioaccumulation of mercury in invertebrates (Hall et al. 1998, Odin et al. 1995, Visman et al. 1995). Under laboratory conditions amphipods efficiently bioaccumulated mercury from both algae and sediment (Lawrence & Mason 2000). Bioaccumulation of mercury by invertebrates allows mercury to become available to organisms higher in the food chain.

Wetland fish can bioaccumulate particularly high levels of methyl mercury (Bloom 1992, Mason et al. 1994, Kannan et al. 1997, Wong et al. 1997, Hall et al. 1998). As a result, piscivorous birds become susceptible to bioavailable mercury (Gariboldi et al. 1997). Bioaccumulation of mercury in large predators is of considerable concern. Much of the furor about mercury in the Florida Everglades resulted because high levels

were detected in the liver of an endangered Florida panther (Roelke et al. 1991). Since alligators are long-lived wetland predators, there are also concerns about the potential to bioaccumulate mercury in their tissues (Kahan & Tuansel 1999).

The Okefenokee Swamp is one of the largest freshwater wetlands in North America. It is approximately 3781 $km^2$ and provides habitat for a variety of organisms (Porter et al. 1999). The Okefenokee Swamp has many characteristics that could lead to mercury accumulation and bioavailability problems including high water temperature, frequent anoxic conditions, low pH (<4.0), intermittent hydrology, peat deposits, and abundant *Sphagnum* mosses. Recently the Georgia Department of Natural Resources placed restrictions on the consumption of two species of fish (bowfin, *Amia calva* and flier, *Centrarchus* sp.) from the Okefenokee due to high levels of mercury. As a result, questions have developed about the role invertebrates might play in the biomagnification of mercury through the food web of the Okefenokee Swamp. The objective of this project was to describe spatial and temporal variation of mercury levels in Okefenokee invertebrates.

**MATERIALS & METHODS**

Thirty-two sites were chosen in the Okefenokee that were distributed across the range of hydrological units and vegetative communities present in the swamp. They include, sites centered around Grand Prairie, Double Lakes, Durden Prairie, Chase Prairie, Floyd's Prairie, and Billy's Lake (Fig 2.1). At each site, sampling was stratified to include shrub, prairie (lily and/or sedge marsh), and cypress habitats. In addition, samples were collected in managed boat trails and any deepwater habitats that were present such as lakes, rivers, or canals. Sampling was conducted in December, 1998,

May, 1999, August, 1999, December, 1999, May, 2000, and August, 2000. At each sampling location we collected amphipods as available for thirty minutes using sweep nets. Beginning in May, 1999, we began collecting Odonata nymphs (primarily Anisoptera), and beginning in December, 1999, we added crayfish to the list. Invertebrates were placed in plastic vials and transported on ice back to the laboratory, and then frozen. We did not wait for organisms to clear their guts because analyses of sediments indicated mercury levels were low (<0.002 ppm).

Amphipod, odonate, and crayfish samples were sent to Clemson Institute of Environmental Toxicology at Clemson University, SC, for analysis. Total mercury levels were determined using Atomic Absorption Spectrophotometery (AAS) as described by Waldrop (1999). Validation trials were conducted in conjunction with each sampling run and lower detection limits were 0.25ppb.

Statistical analysis was conducted using SAS version 8. Variations in mercury levels among locations, subhabitats, sample dates, and organisms were assessed concurrently using 4-way ANOVA. However, because each location-date-subhabitat-organism combination was not replicated, we could not test for statistical interactions among factors in that analysis. Thus, a series of separate 3-way ANOVA's were used to address variation within each of the six locations, within each of the six dates, within each of the five subhabitats, and for each of the three organisms. When ANOVA's were significant, post-hoc Tukey tests were used to separate means. When variances were not equal, mercury levels were log (x+1) transformed prior to analysis.

To determine if hydrological patterns influenced mercury levels, each of the thirty-two sites was ranked according to the number of times that the site was flooded.

For example, if a site flooded once out of six sampling dates it received a rank of 1, if a site flooded twice it received a rank of 2, and so on (with the highest rank being a 6). The sites were then grouped into three categories. Sites that were classified 1, 2, or 3 were grouped together as ephemerally flooded, sites that were classified as 4 or 5 were grouped together as intermittently flooded, and sites that were classified as 6 were grouped as permanently flooded. Mercury levels in amphipods from sites that were flooded ephemerally, intermittently, or permanently were compared using ANOVA.

**RESULTS**

Highly significant differences (Table 2.1) in mercury levels were found among organisms (Fig 2.2, $F_{2, 184}$=93.07, P<0.0001) and sample dates (Fig 2.3, $F_{5, 184}$=16.16, P<0.0001). Mercury concentrations in amphipods were dramatically higher than in odonates or crayfish. Mercury concentrations in amphipod tissue averaged 3.8 ppm and levels often exceeded 20 ppm in some sample pools. Invertebrates collected in the May 2000, sampling period had the highest levels of mercury. At that time, levels averaged 5.7 ppm, and levels as high as 86 ppm were encountered. However, overall levels of mercury were similar among all six locations (Fig 2.4, $F_{5, 184}$=1.67, P=0.1448) and all five subhabitats ($F_{4, 184}$=1.67, P=0.3527, Fig 2.5) (Table 2.1).

Patterns among individual organisms (amphipods, odonates, and crayfish):

For amphipods, mercury concentrations peaked during the May 2000, sampling date ($F_{5, 65}$=18.95, P<0.0001, Fig 2.6). In contrast, odonates had low levels of mercury during May 1999 and May 2000 (Fig 2.7). For crayfish, levels remained similar among all dates sampled. For both amphipods and crayfish, concentrations of mercury were similar among all six locations (both P> 0.05). However, for odonates, mercury levels

were marginally higher at Floyd's Prairie ($F_{5, 78}$=1.67, P=0.0311). For all three organisms, mercury levels were similar in all five subhabitats (all P>0.05)

Patterns among dates (12/98, 5/99, 8/99, 12/99, 5/00, and 8/00):

Higher levels of mercury in amphipods than odonates or crayfish were apparent for every sampling date except August 2000, when levels of mercury did not differ among organisms (P=0.7332). Mercury levels in invertebrates were similar among all five subhabitats for every date except May 2000, when significantly higher levels of mercury were present in invertebrates from the boat trails ($F_{4, 33}$=3.31, P=0.0219, Fig 2.8). For every sampling date, mercury levels in invertebrates were similar among all six locations.

Patterns among individual locations (Grand Prairie, Chase Prairie, Durden Lake, Double Lakes, Floyd's Prairie, Billy's Lake):

Consistent with the overall analysis, mercury levels in amphipods were higher than in either odonates or crayfish at each of the six locations. A peak in mercury concentration occurred during May 2000 at the Grand Prairie, Chase Prairie, Durden Lake, and Billy's Lake locations (all P<0.05). However at the Double Lakes site, the peak in May 2000 was not evident, and instead levels in December 1999 and August 1999 somewhat exceeded May 2000 levels. The Floyd's Prairie area was dry in May 2000 and samples could not be collected; otherwise mercury levels were similar across sampling dates ($F_{4, 9}$= 0.69, P=0.6184). At every sampling location, mercury concentrations in invertebrates were similar among the five subhabitat classes (all P>0.05).

Patterns among subhabitats (prairie, shrub, cypress, trails, and deepwater habitats):

The higher levels in amphipods that are evident in the overall analysis also existed at each of the five subhabitats (all $P<0.05$). A peak in mercury levels in invertebrates occurred during the May 2000 sampling period for cypress, shrub, and boat trail habitats (all $P<0.05$). For each of the subhabitats, mercury levels in invertebrates among the six locations were similar (all $P>0.05$).

Hydrology patterns

December 1998 was the wettest month of the study when all 32 sites were flooded, while May 2000 was the driest month with only 17 sites flooded (Table 2.2). Mercury levels in amphipods were highest in sites that were flooded for all six sampling dates (permanently flooded sites) ($P=0.0117$, Fig 2.9) as compared to sites that were flooded for < 6 sampling dates (either intermittently or ephemerally flooded sites). We were concerned that this pattern simply developed because permanent water sites were some of the only locations flooded during the May 2000 peak in mercury. Thus a second analysis was conducted using just the December 1998 sampling date when all sites were flooded. At that time mercury levels in amphipods were similar among ephemerally, intermittently, and permanently flooded sites ($P=0.2928$).

**DISCUSSION**

Levels of mercury detected in Okefenokee invertebrates were unusually high, even for wetlands. We frequently encountered mercury levels in excess of 20 ppm, and levels averaged 1.6 ppm. In comparison, mercury levels in invertebrates of the Florida Everglades averaged 0.3 ppm (Scheidt 2000), and levels averaged 0.1 ppm in small

depressional wetlands in South Carolina (Snodgrass et al. 2000). While mercury levels in the Okefenokee Swamp invertebrates might seem high, the elevated levels encountered in this study may have resulted because we sampled amphipods, which are not typically sampled in other wetlands, and because we sampled six times over a two-year period, increasing our chances of sampling during temporal peaks in mercury bioavailability.

Variation in mercury levels among organisms

Most of the explained variation in mercury levels in Okefenokee invertebrates (Table 2.1) was related to differences among amphipods, odonates, and crayfish. Mercury levels in amphipods collected from the Okefenokee Swamp averaged 4.0 ppm and levels >20 ppm were detected in several sample pools. In contrast, concentrations of mercury in odonates and crayfish averaged only 0.18 ppm and 0.23 ppm, respectively. Possible reasons for higher levels detected in amphipods may include close association of amphipods with sediment or with mercury sequestering plants.

Sediment is a likely source for mercury transfer to aquatic invertebrates, and as detritivores, amphipods probably frequent the sediment boundary to feed (Pennak 1989). Mercury is often bound to sediments and is released via methylation (Raldua & Pedrocchi 1996) or by drying and reflooding of habitats (Snodgrass et al. 2000; Warren et al. 2001). Studies have indicated that amphipods exposed to sediments spiked with methyl mercury showed an increased uptake of mercury as compared to those exposed to spiked pore water (Lawrence & Mason 2001). The relationship between sediment and mercury is complex and there are many interacting factors, but amphipods that are associated with or feeding on sediment undoubtedly increased their degree of exposure to mercury.

Biomagnification of mercury from plants or periphyton might also explain higher levels of mercury in amphipods. Mercury concentrations in periphyton and plant material are 3-10 times lower than those in invertebrates that feed on this material (Mason et al. 2000). Some of the highest levels of mercury have been documented from *Sphagnum* mosses, with low levels occurring in water lily (Cymerman & Kempers 1995, Moore et al. 1995, Thompson-Roberts et al. 1999). We collected and analyzed plant material from the Durden Lake area; mercury levels generally were low, but the highest levels (0.43 ppm) were collected from *Sphagnum* mosses. In the Okefenokee, we observed that amphipods were abundant in and around *Sphagnum* mosses, and this relationship might explain the higher levels of mercury in the amphipods. Algae and plant detritus are two important foods of amphipods that might be the source of mercury for those organisms. Previously accumulated mercury becomes methylated in decaying plant material (Heyes et al. 1998), which can result in increased levels of mercury that are bioavailable to invertebrates. In addition, it has been shown that amphipods sequester mercury much more efficiently from mercury-spiked algae in comparison to mercury-spiked pore water (Lawrence & Mason 2000).

If biomagnification of mercury is occurring within invertebrates in the Okefenokee, one might expect that predators such as odonates and large, long-lived organisms such as crayfish would have higher levels of mercury than amphipods. The concentrations of mercury in predatory insects has been reported to increase by a factor of 2-5 over that found in prey (Mason et al. 2000). However, predatory odonates collected from the Okefenokee had mercury levels ranging from 0.01-3.53 ppm, levels that were dramatically lower than those in amphipods (Fig. 2.2). Studies from lakes

reported that levels of mercury in odonates and amphipods were similar, with levels averaging 0.13 ppm in amphipods and 0.12 ppm in odonates (Wong et al. 1997). In the Okefenokee, biomagnification of mercury by odonates may be similar to other habitats, but the relationship between mercury and amphipods is apparently unique. Further, it seems unlikely that odonates in the Okefenokee accumulate mercury from amphipods.

Crayfish consume many of the same foods as amphipods, have similar habitats, and are larger and longer-lived than amphipods, yet mercury levels in these organisms was lower than in amphipods. Studies elsewhere have shown that mercury levels in crayfish are usually lower than other corresponding predatory insects but higher than other insect groups (Mason et al. 2000). Mercury levels in crayfish in the Okefenokee were similar to levels in the Everglades (Scheidt et al. 1997).

Temporal variation in mercury levels

Onset of drought and extensive fires are two factors that might explain the observed temporal change in mercury levels in invertebrates. Drought conditions might increase bioavailable mercury because the increase in temperature causes a decrease in dissolved oxygen, which in turn promotes the formation of methyl mercury (Morel et al. 1998). When mercury levels peaked in May 2000, precipitation levels were very low (Fig 2.9) and only 17 of the 32 sample sites were flooded (Table 2.2). Both Snodgrass et al. (2000) and Hall et al. (1998) reported that levels of mercury in invertebrate increased after reflooding of a dried habitat. However, when many sites in the Okefenokee reflooded in August 2000, mercury levels in invertebrates declined rather than increased (Fig 3.9). The relationship between hydroperiod and mercury appears to be different in the Okefenokee than in other habitats. In the Okefenokee, permanently flooded sites

(lakes, canals, boat trails) had somewhat higher mercury levels in amphipods than those habitats that were intermittently or ephemerally flooded (Fig 2.11). This suggests that mercury levels in Okefenokee invertebrates are less affected by drying and reflooding cycles than elsewhere.

Forest fires occurred occasionally in the wetland portion of the Okefenokee Swamp, and it is believed that burning of plant material and peat might release mercury into the air (Lamontagne et al. 2000). In the Okefenokee there was a gradual increase in mercury levels in invertebrates after a large fire in the summer of 1999, but levels of mercury did not peak until a full year later in May 2000 (Fig 2.11). In Canada, extensive wildfires did not increase levels of mercury in zooplankton or fish (Garcia & Carignan 1999, 2000). The peak for mercury in Okefenokee invertebrates during May 2000 was probably most influenced by drought, although fire might also have been involved.

Spatial variation in mercury levels

While the Okefenokee Swamp clearly had organisms (amphipods) and time periods (May 2000) that were "hot" for mercury, we did not detect any "hot" spots for mercury. A 1994 study found that Chase Prairie had significantly higher levels of mercury in crayfish when compared to the Chesser Prairie location (Arnold 2000), but in this study, levels in Chase Prairie were generally low. We detected some minor variations in mercury levels among sub-habitats. Levels of mercury in invertebrates peaked during the May 2000, sampling period in all habitats except the deepwater lakes and canals. However, odonates comprised most of the samples collected in the deepwater habitats, and odonates do not appear to accumulate large amounts of mercury in the Okefenokee. We observed that mercury levels in the invertebrates collected from

trail habitats were particularly high in May 2000 and speculated that sediments in trails might contain high levels of mercury. However, mercury levels were similar between sediment samples from a trail that contained invertebrates with high levels of mercury and sediment samples collected from adjacent natural wetland (unpublished data).

Implications

Amphipods appear to play an important role in the biomagnification of mercury in the Okefenokee Swamp. Although crayfish are commonly used as bioindicators of heavy metal contamination, amphipods clearly accumulated greater levels of mercury in Okefenokee food webs. In fact, if we had relied solely on crayfish to monitor mercury, we would have developed a very different picture of mercury distributions. Amphipods might be especially useful for detecting high levels because, as Sferra et al. (1999) reported, mercury toxicity in amphipods exceeded 4.1 ppm. Many organisms might die before accumulating such high levels. It follows that since amphipods are often a major source of food for fish they might be contributing to the high levels of mercury in Okefenokee Swamp fish.

This study indicates that mercury monitoring programs must address temporal variation. Our study clearly indicates that there were dramatic variations in invertebrate mercury levels among dates. The combination of drought and fire might have played a prominent role in the May 2000 of mercury peak. Perhaps, sampling should be conducted more frequently around these events to better ensure that mercury levels in the environmental are detected.

Since mercury levels were similar among all locations and habitats, it suggests that the source of mercury for the Okefenokee Swamp is probably atmospheric

deposition. Over the past 150 years, atmospheric inputs of mercury have tripled in the U.S. (Mason et al. 1994). Even the most remote sites receive significant inputs of mercury by atmospheric transport (Fitzgerald et al. 1998). Although the Okefenokee Swamp is one of the largest wilderness areas in the eastern United States, mercury contamination is a problem that needs to be closely monitored.

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Table 2.1. ANOVA table of mercury concentrations among locations, habitats, dates, and organisms.

Table 2.1.

| Source | DF | Type III SS | Mean Square | F Value | Pr > F |
|---|---|---|---|---|---|
| Location | 5 | 2.4819 | 0.4963 | 1.67 | 0.1448 |
| SubHabitat | 4 | 1.3239 | 0.3309 | 1.11 | 0.3527 |
| Date | 5 | 24.0676 | 4.8135 | 16.16 | <0.0001 |
| Organism | 2 | 55.4481 | 27.7240 | 93.07 | <0.0001 |
| Error | 184 | 54.8100 | | | |
| Total | 200 | 128.0000 | | | |

Table 2.2 Total numbers of sites flooded (N= 32) during each of six sampling dates.

Table 2.2.

| | Dec 1998 | May 1999 | Aug 1999 | Dec 1999 | May 2000 | Aug 2000 |
|---|---|---|---|---|---|---|
| Number of sites flooded during each sampling date. | 32 | 22 | 21 | 23 | 17 | 23 |

Figure 2.1 Map of the Okefenokee Swamp showing the locations (stars) of the six locations where invertebrates were collected.

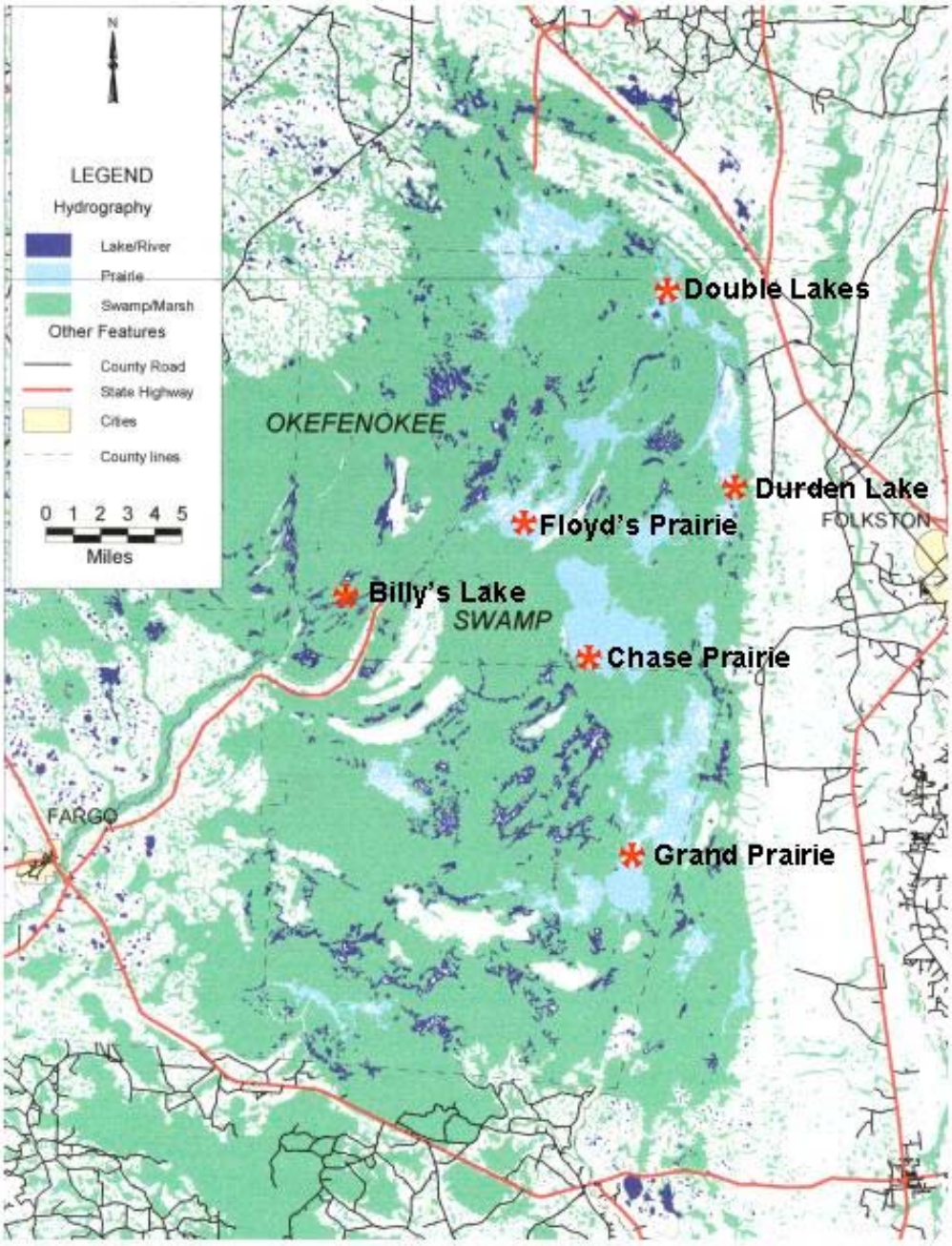

N
LEGEND
Hydrography
Lake/River
Prairie
Swamp/Marsh
Other Features
County Road
State Highway
Cities
County lines
0 1 2 3 4 5
Miles
Double Lakes
OKEFENOKEE
Durden Lake
FOLKSTON
Floyd's Prairie
Billy's Lake
SWAMP
Chase Prairie
FARGO
Grand Prairie

Figure 2.2 Variation in total mercury concentrations among amphipods, odonates and crayfish collected from the Okefenokee Swamp ($P<0.001$). Bars indicated by the same letter are not significantly different ( $P>0.05$).

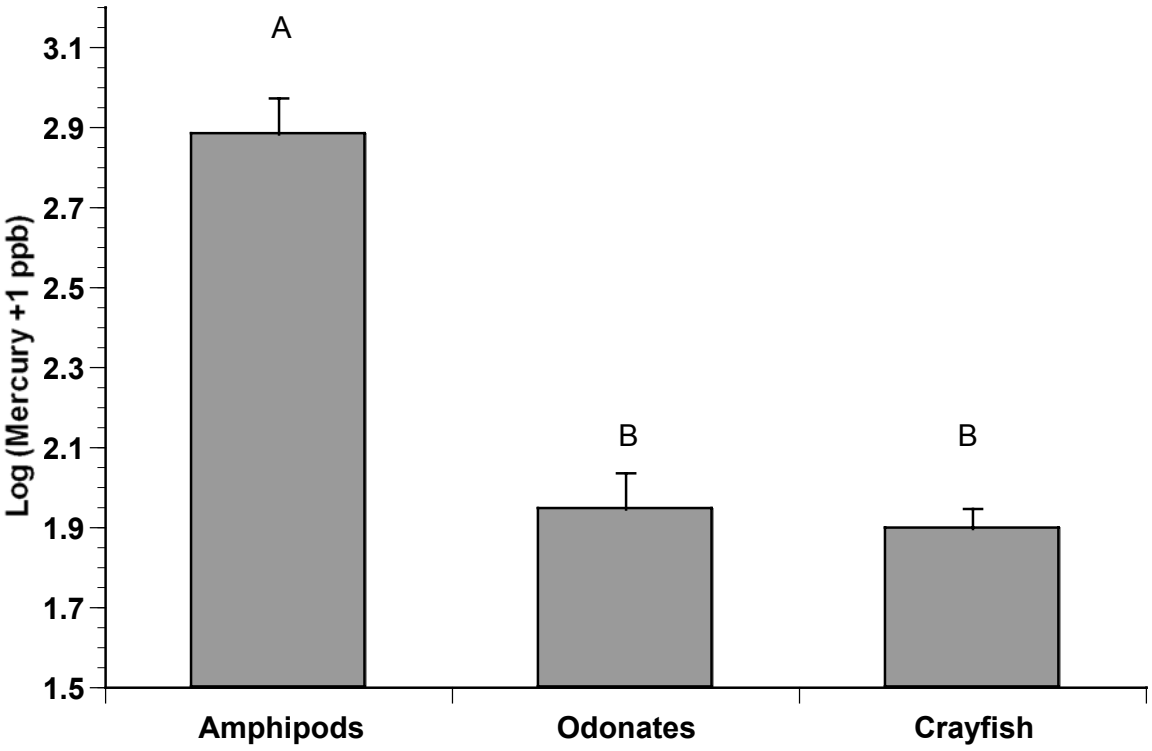

A
B
B
3.1
2.9
2.7
2.5
2.3
2.1
1.9
1.7
1.5
Log (Mercury +1 ppb)
Amphipods
Odonates
Crayfish

Figure 2.3 Variation in total mercury concentrations in invertebrates among six sampling dates at the Okefenokee Swamp ($P<0.001$). Bars indicated by the same letter are not significantly different ( $P>0.05$).

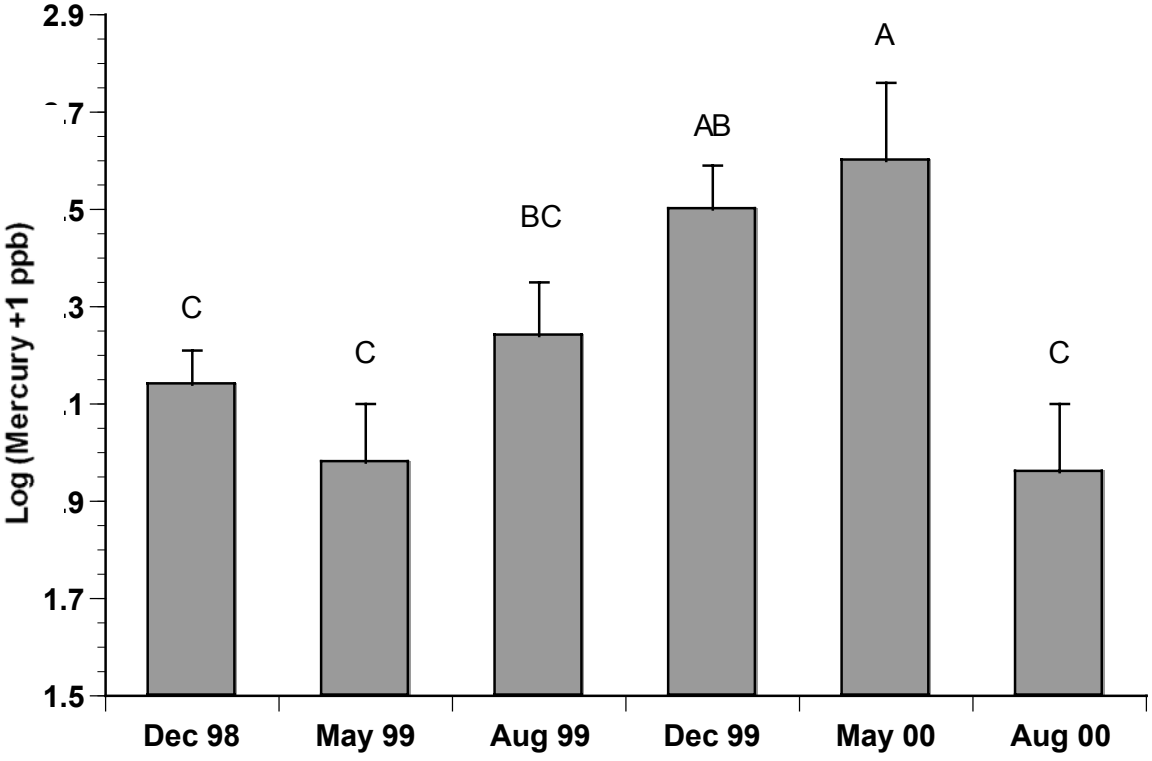

2.9
1.7
1.5
Log (Mercury +1 ppb)
C
C
BC
AB
A
C
Dec 98
May 99
Aug 99
Dec 99
May 00
Aug 00

Figure 2.4 Variation in total mercury concentrations in invertebrates among six locations at the Okefenokee Swamp (P=0.15).

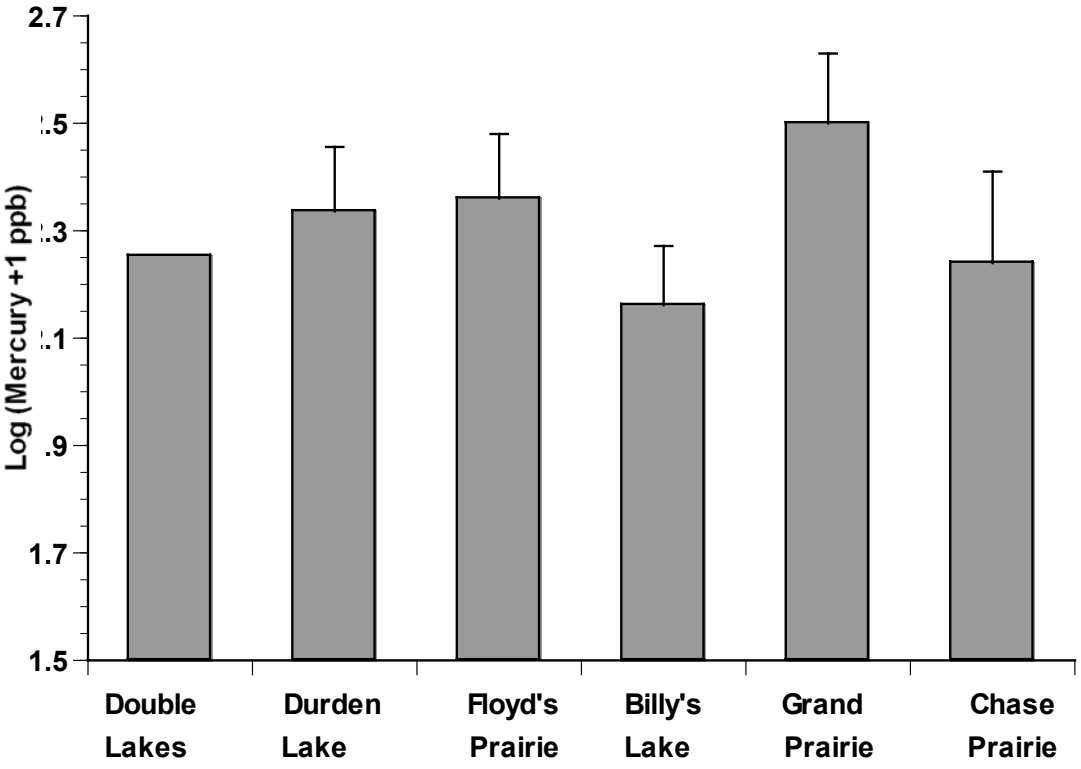

2.7
.5
.3
.1
.9
1.7
1.5
Log (Mercury +1 ppb)
Double Lakes
Durden Lake
Floyd's Prairie
Billy's Lake
Grand Prairie
Chase Prairie

Figure 2.5 Variation in total mercury concentrations in invertebrates among subhabitats at the Okefenokee Swamp (P=0.35).

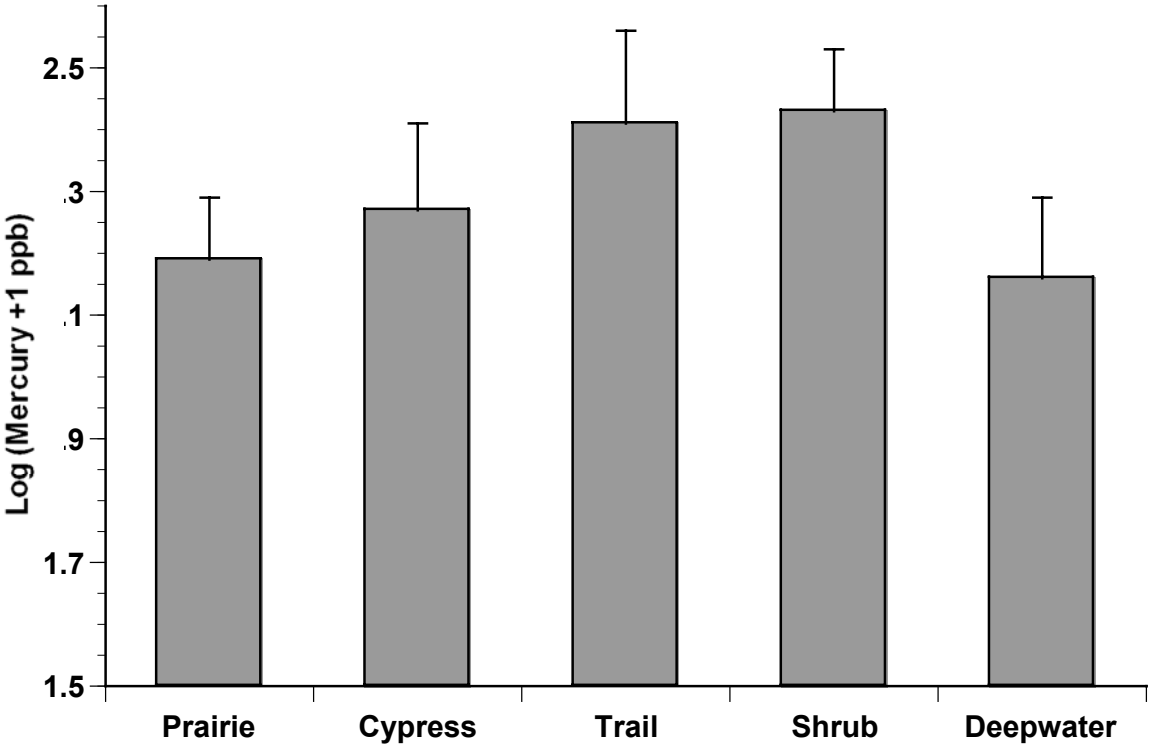

Log (Mercury +1 ppb)
2.5
.3
.1
.9
1.7
1.5
Prairie
Cypress
Trail
Shrub
Deepwater

Figure 2.6 Variation in total mercury concentrations in amphipods over six sampling dates ($P < 0.0001$). Bars indicated by the same letter are not significantly different ($P > 0.05$).

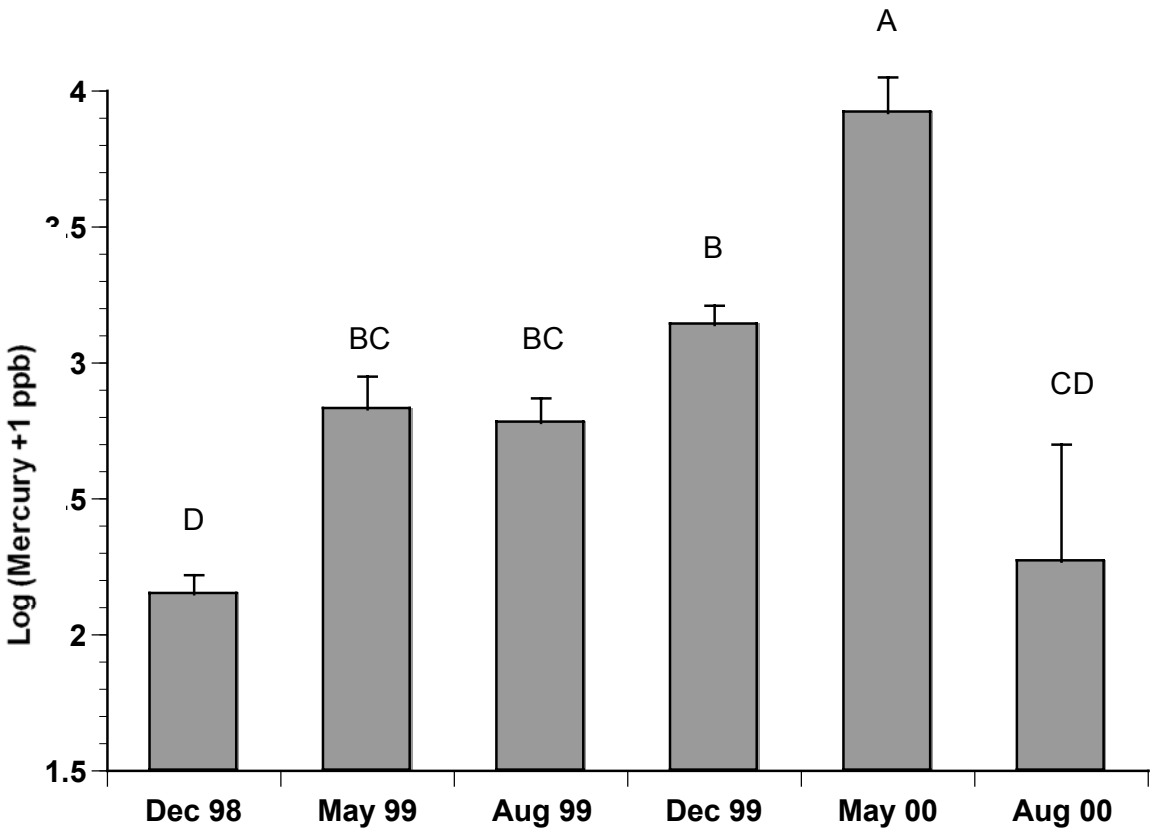

Log (Mercury +1 ppb)
4
3.5
3
2.5
2
1.5
D
BC
BC
B
A
CD
Dec 98
May 99
Aug 99
Dec 99
May 00
Aug 00

Figure 2.7 Variation in total mercury concentrations in odonates over six sampling dates ($P = 0.0311$). Bars indicated by the same letter are not significantly different ($P > 0.05$).

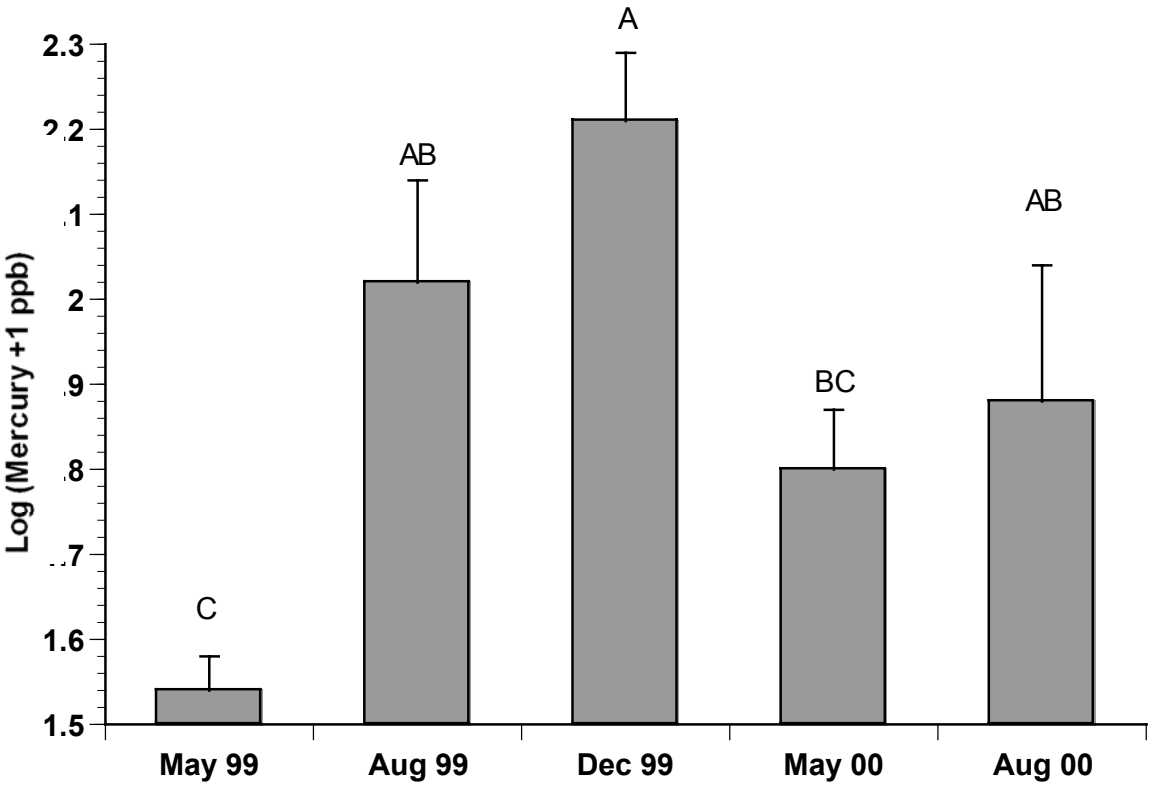

A
AB
AB
BC
C
2.3
2.2
2.1
2
1.9
1.8
1.7
1.6
1.5
Log (Mercury +1 ppb)
May 99
Aug 99
Dec 99
May 00
Aug 00

Figure 2.8 Variation in total mercury concentrations in invertebrates from trail habitats during the six sampling dates ($P = 0.0219$). Bars indicated by the same letter are not significantly different ( $P > 0.05$).

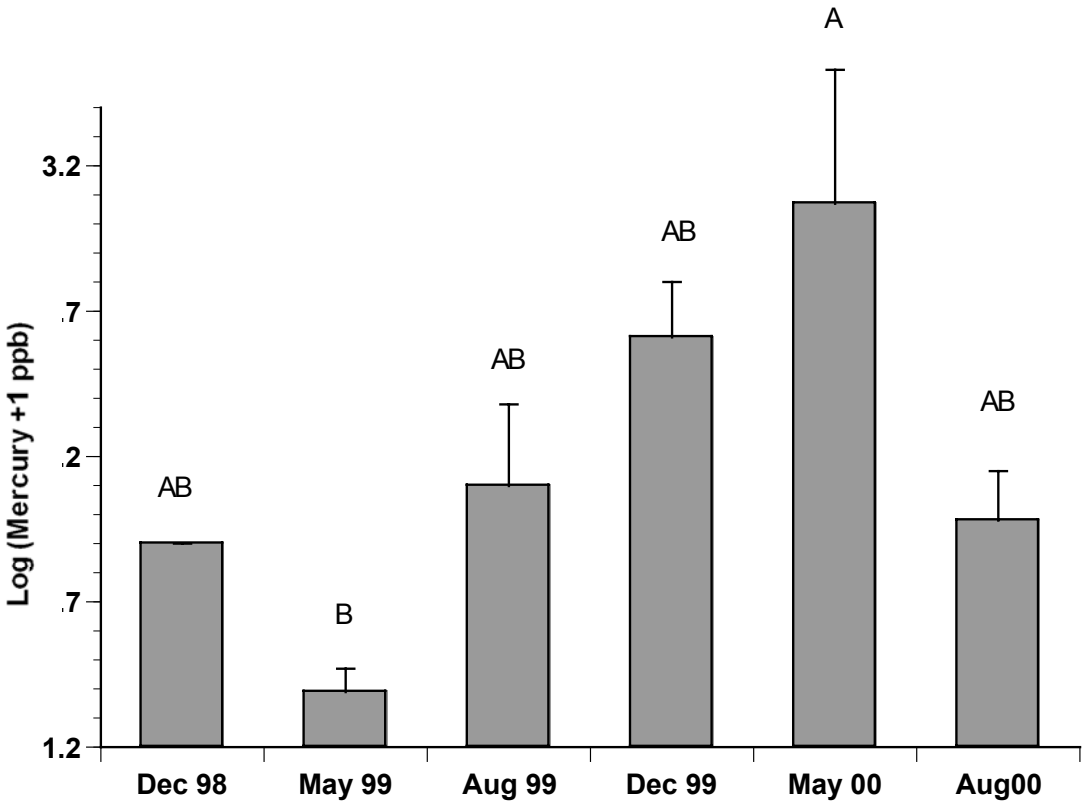

A
AB
AB
AB
AB
B
3.2
.7
.2
.7
1.2
Log (Mercury +1 ppb)
Dec 98
May 99
Aug 99
Dec 99
May 00
Aug00

Figure 2.9 Average mercury concentrations over two years in amphipods from ephemerally, intermittently, and permanently flooded sites of the Okefenokee Swamp ($P=0.0117$). Bars indicated by the same letter are not significantly different ($P>0.05$).

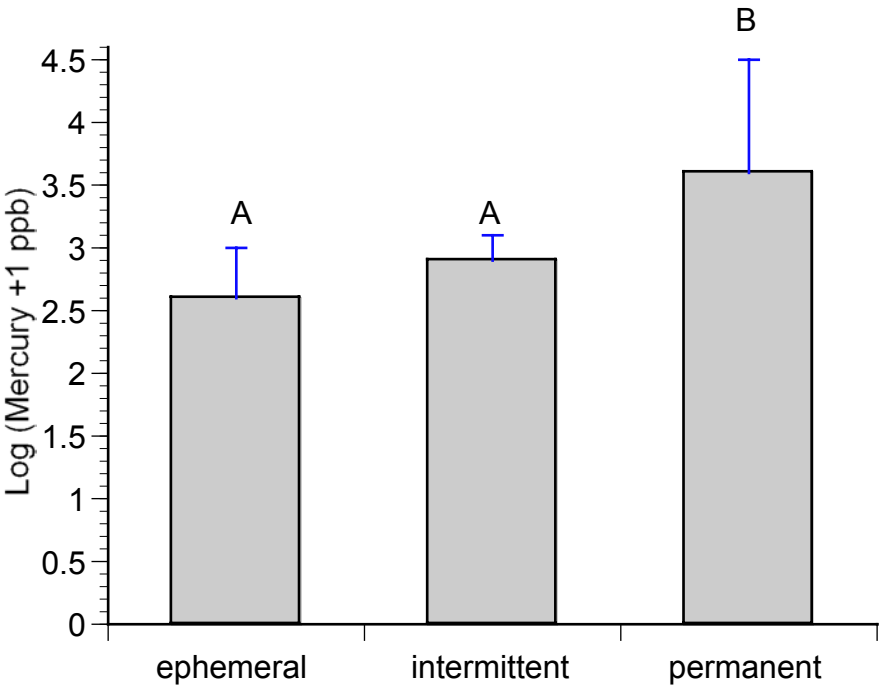

B
A
A
4.5
4
3.5
3
2.5
2
1.5
1
0.5
0
Log (Mercury +1 ppb)
ephemeral
intermittent
permanent

Figure 2.10  Monthly rainfall levels (bars) at the Okefenokee Swamp National Wildlife Refuge Headquarters from July 1998 – August 2000, and mercury levels in invertebrates (line) from December 1998 to August 2000.

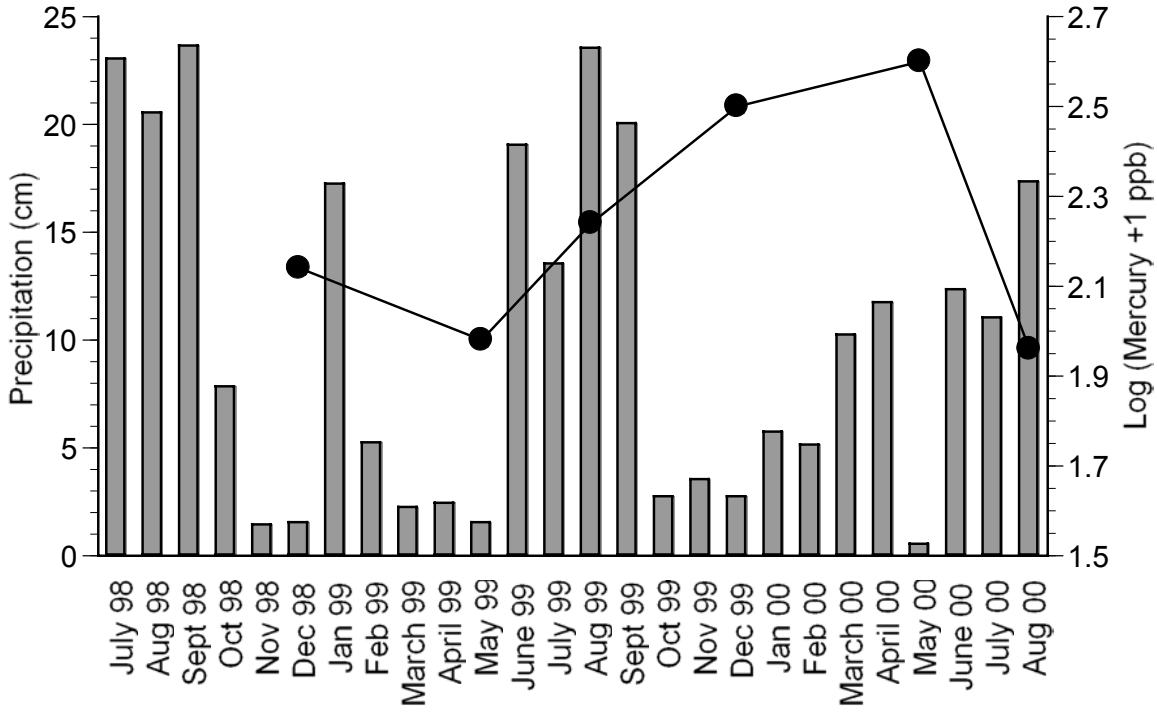
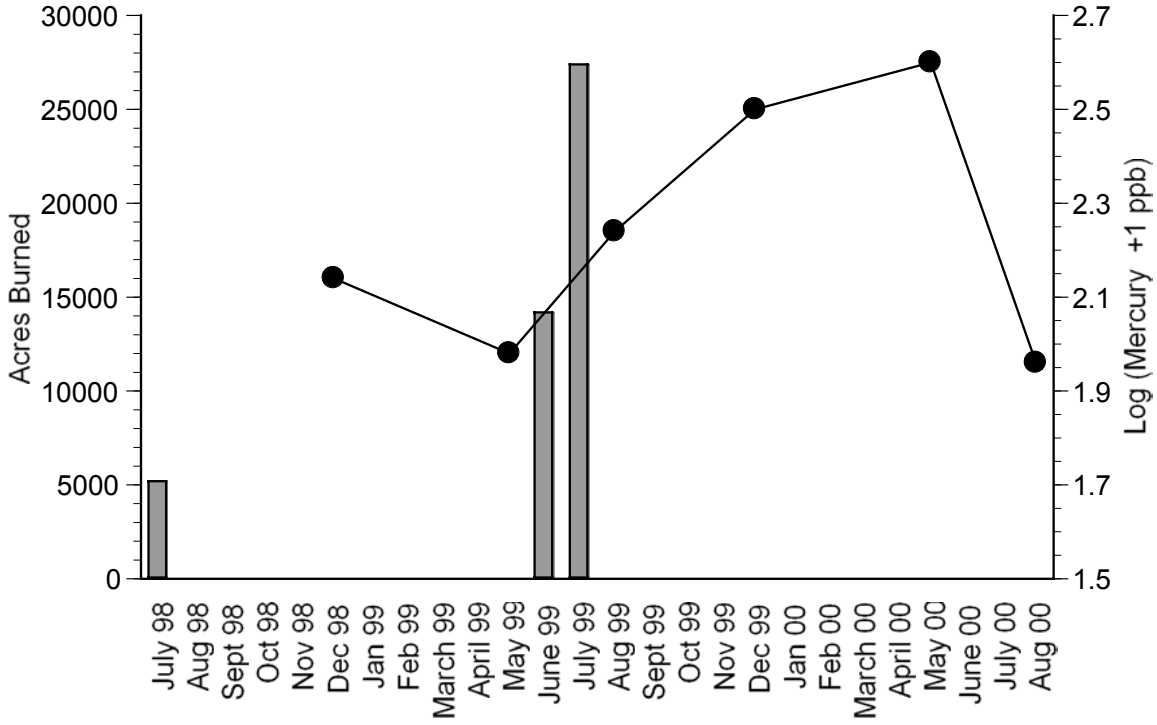

Precipitation (cm)
Log (Mercury +1 ppb)
25
20
15
10
5
0
2.7
2.5
2.3
2.1
1.9
1.7
1.5
July 98
Aug 98
Sept 98
Oct 98
Nov 98
Dec 98
Jan 99
Feb 99
March 99
April 99
May 99
June 99
July 99
Aug 99
Sept 99
Oct 99
Nov 99
Dec 99
Jan 00
Feb 00
March 00
April 00
May 00
June 00
July 00
Aug 00

Figure 2.11 Total wetland acres burned (bars) in the Okefenokee Swamp from July 1998 to August 2000 (data provided by Okefenokee National Wildlife Refuge personnel), and mercury levels in invertebrates (line) from December 1998 to August 2000.


30000
25000
20000
15000
10000
5000
0
Acres Burned
2.7
2.5
2.3
2.1
1.9
1.7
1.5
Log (Mercury +1 ppb)
July 98
Aug 98
Sept 98
Oct 98
Nov 98
Dec 98
Jan 99
Feb 99
March 99
April 99
May 99
June 99
July 99
Aug 99
Sept 99
Oct 99
Nov 99
Dec 99
Jan 00
Feb 00
March 00
April 00
May 00
June 00
July 00
Aug 00

# CHAPTER 3

# IMPACTS OF TREE HARVEST PRACTICES ON AQUATIC INVERTEBRATES IN FLOODPLAIN FORESTS AND DEPRESSIONAL WETLANDS

George, B.M. & Batzer, D.P. To be submitted to Southern Journal of Applied Forestry

## INTRODUCTION

Bottomland hardwood forests (Kellison & Young 1997, Lockaby et al. 1997, Perison et al. 1997) and small depressional ponds (Batzer et al. 2000, Prenger & Crisman 2001) are wetlands of the southeastern United States that are frequently associated with timber harvest. Various physical features of wetlands can be affected by logging. For example, the removal of trees can cause a wetland forest to become wetter (Dube 1995; Sun et al. 2000) because evapotranspiration rates decrease and runoff rates increase (Richarson & McCarthy 1994; Crownover et al. 1995). The loss of canopy cover can increase levels of sunlight that in turn can increase water temperature. Post-harvest site preparations can also affect wetlands. Fertilization of new trees causes increased levels of nutrients (N, $PO_4$, ortho-$PO_4$ and urea) (Shepard 1994). Transformation of inorganic forms of nitrogen and phosphorus to organic forms can occur within wetlands and this process can be affected by logging induced variations in hydrology (Walbridge & Lockaby 1994).

Changes in biota also can occur when timber in and around wetlands is harvested. Excessively wet conditions that can occur after logging may impair germination of some seeds (McKee 1985). On the other hand, certain hydrophytes (e.g. sedges) may flourish from logging induced wetting (Batzer et al. 2000). These changes in hydrology and vegetation may in turn influence invertebrate compositions (Ward 1992). Finally, harvest methods can affect salamander and reptile densities (Lockaby et al. 1997; Perison et al. 1997). However, most biotic impacts of logging in and around wetlands remain poorly understood (Prenger & Crisman 2001).

Invertebrates are particularly important to the ecologies of bottomland hardwood floodplains and depressional wetlands (Batzer & Wissinger 1996). These invertebrates are important as predators as well as prey items for higher organisms. In addition to their role in the trophic structure of communities, invertebrates can function as useful bioindicators of environmental impacts in wetlands (Radar et al. 2001) because they are ubiquitous, occur in large numbers, and are easy to sample and manipulate in studies (Rosenberg & Resh 1993).

This study will focus on two wetland systems, bottomland hardwood floodplains and isolated depressional wetlands. Bottomland hardwood forests contain valuable timber and resources that are increasingly being exploited (Kellison & Young 1997). Leaf litter inputs, which are likely the food base for bottomland hardwood floodplains food webs, will be reduced after mature trees are removed. Removing trees will also cause light levels to increase dramatically, thus increasing the possibility of algal blooms.

While depressional wetland systems can also be logged directly (Prenger & Crisman 2001), in many tree plantations it is no longer cost-effective to harvest them. However, the timber that surrounds depressional wetlands may be intensively managed, and this study will address some indirect impacts of logging (i.e., peripheral shading, fertilization, and buffer strips). As isolated wetlands rely hydrologically on precipitation and run off, fluctuations in evapo-transpiration from the surrounding trees might impact the hydrology (Sun et al. 2000) and the invertebrate community of these depressional wetlands.

## MATERIALS & METHODS

*Effects of Logging on Invertebrates in a Bottomland Hardwood Wetland*

Study Site

Bottomland hardwood forests are found on the floodplains of streams and rivers and are especially important wetland habitats for the Southeastern United States (Mitsch and Gosselink, 1993). A seasonal cycle of drying and flooding creates a nutrient rich environment that supports a diverse and productive plant and animal community (Smock 1999). Rivers associated with these floodplains mainly drive the hydrology of these systems. The bottomland hardwood floodplain used for this study was located along the lower Coosawhatchie River in Jasper County, South Carolina (Burke & Eisenbies 2000). The soils there are classified into the Brookman series, which consists of dark clayey subsoil and a thick black loamy surface layer. Annual precipitation ranges from 127 cm to 152 cm per year. This habitat typically floods in winter and stays flooded into late spring.

A 150 ha portion of mature bottomland hardwood forest in the Coosawhatchie floodplain was divided into three sections. The section farthest downstream was clear-cut, the section in the middle was partially cut, and the upstream section remained intact and served as a control. In the partially cut section several "islands" of trees were retained. Logging was conducted in the fall of 1999 when the site was dry and impacts such as puddling, rutting, and compaction were minor. In both the clear-cut and partial cut sections, machine-mounted rotary saws were used to cut all stems and tops of trees. All stems were cut, even some that were not removed from the site. Where heavy equipment was used, machines were operated over beds of cut limbs to reduce impacts on soils. The clearcut and partially cut sections were allowed to regenerate naturally.

Invertebrate Sampling and Experiments

To assess invertebrate response to logging, we first collected two years of pre-harvest baseline data in each of the three treatment areas (December 1997–April 1999). Each year, samples were collected in December or January soon after the site floods and a second set of samples in April as water levels were receding. Aquatic invertebrate samples were collected using D-framed sweeps nets. In each of the three sections on each date ten 1-m sweep were collected at random locations along a representative transect. Samples were preserved in 95% ethanol, and transported to the laboratory for processing. Invertebrates were removed manually using dissecting scopes, identified to genus or family using keys in Pennak (1989), Thorp & Covich (1991), and Merritt & Cummins (1996), and enumerated.

Two factors associated with logging that might significantly influence post-harvest invertebrate communities on this floodplain were reduced leaf litter inputs and increased light levels. To test the influence of leaf litter reductions, twelve 1m x 1m plots were selected in the unmanipulated control area of the Coosawhatchie floodplain. That area was selected for this experiment because we wanted to manipulate leaf litter input in isolation while all of the other variables that might be associated with logging remained unchanged. Six plots were randomly designated as controls and six plots were designated for leaf litter exclusion. Each exclusion plot was cleared of leaf litter in November 1999 while the area was still dry. Then, square (1m x 1m) sheets of hardware cloth wire mesh (12mm) were laid over all 12 plots (exclusion and control). Leaves that accumulated on screening were removed from exclusion plots to keep them litter free. Leaf litter on mesh over reference plots was removed then placed on the soil surface, so the only difference

between exclusion and control plots was the amount of litter. Once per month when the sites were flooded, invertebrate samples were taken in all 12 plots using a D-frame sweep net, with a single one sweep taken across the center of each plot. Samples were processed as described above.

To determine the influence of increased insolation on invertebrates, twelve 3m x 3m plots were chosen in the clear-cut section of the Coosawhatchie floodplain. Six plots were selected randomly as controls and six plots were selected for artificial shading. Shade cloth panels (3m x 3m) supported by wooden and metal frames were suspended over each plot selected for shading. Control plots received ambient inputs of light. Each month that the site was flooded (February 2000, April 2000, March 2001) invertebrate samples were collected from each plot using a D-framed net. One 1-m sweep was taken in the center of each plot. Samples were preserved in 95% ethanol and processed in the laboratory. In the summer of 2001, biomass of herbaceous plants was measured in the center of each plot by clipping a 50 x 50 cm quadrat. Clipped material was oven dried for 48 hours at 60°C and dry mass determined.

Analyses

Differences in invertebrate abundance among the control, partially cut, and clearcut treatments were determined using ANOVA in SAS version 8. Analyses used total invertebrate abundance and the abundances of common families (those that comprised >1% the total). Differences in invertebrates between litter exclusion and control plots and between open and shaded plots were determined using t-tests. Data were log (x+1) transformed prior to analyses to equalize variance.

*Effects of Logging Peripheral Upland Pines on Embedded Depressional Wetlands and Efficacy of Riparian Buffer Strips on Reducing Impacts*

Study Sites

Small isolated depressional wetlands are scattered across the Coastal Plain of the Southeastern United States (Crownover et al. 1995; Leslie et al. 1997). Because of their small size, it is often no longer profitable to harvest trees in most depressional wetlands. However, many are embedded in pine stands that are routinely harvested and concerns have developed that adjacent logging activity might affect pond ecologies (Batzer et al. 2000).

In cooperation with International Paper Company, two sets of four wetlands were selected in 1997 (another two sets were included in the original study, but these wetlands never flooded because of drought and so they were deleted from consideration). One of the four sets of wetlands was located near Rincon, Georgia, and the second was located near Jones, Georgia. At each location, three of the four wetlands were located in a loblolly pine (*Pinus taeda L.*) plantation that was scheduled for logging the next year. The fourth wetland was located in an adjacent stand of similar aged pine that was not scheduled for logging. In each logged tract, buffers of plantation pine were maintained around two of the three wetlands (selected randomly). A 10-m wide buffer of pine was maintained around one wetland and a 30-m buffer was maintained around another wetland. Each buffer width was measured from the wetland's edge, determined by the presence of hydrophytic vegetation.

Invertebrate Sampling and Experiments

One year prior to harvest (1998) and for three years after harvest (1999-2001) aquatic invertebrate communities were sampled in all ponds. Using a D-frame sweep net, three 1-m sweeps were collected at randomly selected locations in each wetland. As nets sieved the water column, scraped the bottom, and swept the subsurface and above surface portions of emergent plants, a fairly complete sample of the invertebrate community was collected. Samples were collected in winter soon after the sites had flooded and again in early spring when water levels were receding. Samples were preserved with 95% ethanol and transported back to the laboratory for processing.

Surface waters depths were measured using meters as described by Richter (1997). These meters measure both maximum and minimum water depths over a given sampling period. Temperature, pH, electrical conductivity, and dissolved oxygen of the water were measured using portable field meters during each sampling visit. In summer the biomass of herbaceous plants was measured by clipping two 50 x 50 cm quadrats near each wetland's center (locations randomly selected). Clipped plant material was oven dried for 48 hours at 60°C and dry mass determined.

In this system, likely impacts of logging to invertebrates included the loss of peripheral shading and fertilization. In one pond without a buffer we installed shaded plots similar to those described above in the floodplain system. However, that pond did not flood due to drought conditions so the effects of retaining shade could not be evaluated. To assess the post harvest management practice of aerial fertilization, two depressional wetlands set in mature pine stands were selected. One wetland was designated as a control and one was fertilized. In December 1999, an operational rate of

65 kg of diammonium phosphate was applied per hectare to the latter wetland. Each month that sites were flooded, invertebrate samples were collected. Ten 1-m sweeps were taken using a D-frame sweep net. Samples were preserved with ethanol and transported back to the laboratory for processing.

Analyses

Variation in plant biomass among the 0-m buffer, 10-m buffer, 30-m buffer and reference wetlands were determined using ANOVA. Differences in invertebrate densities among 0-m buffer, 10-m buffer, 30-m buffer, wetlands were determined using ANCOVA. We could not use ANOVA for this analysis because the reference wetlands did not flood and thus we relied on temporal changes to assess differences among buffer widths. The differences in invertebrate densities and plant biomass between the fertilized and reference ponds were determined using t-tests. All data were log (x+1) transformed prior to analysis to equalize variance.

**RESULTS**

*Effects of Logging on Invertebrates in a Bottomland Hardwood Wetlan*ds

Crustaceans, mollusks, dipterans, and oligochaetes dominated the pre-harvest invertebrate community (Table 3.1) and at that time abundances of each of these taxa as well as total invertebrate abundance were similar (P=0.56) among the three study areas (Figs. 3.1, 3.2, 3.3, 3.4, 3.5). After logging, total invertebrate abundance remained similar (P=0.45) among the three study areas (Fig 3.1). However, significant responses were detected for several individual taxonomic groups.

Of the common crustaceans in the study sites, numbers of Asellidae (Fig 3.2A, P=0.0035) and Crangonyctidae (Fig 3.2B, P=0.0231) were lower in the clearcut and

partially cut areas than the uncut control. A significant overall response was also detected for Cambaridae (P=0.0453). However, while the pattern of response for crayfish appeared to parallel responses of the other crustaceans, post hoc tests could not discriminate specific differences among treatments (Fig. 3.2C).

Of the common mollusks in the study sites, planorbid snail numbers were significantly lower (P=0.0006) in the clearcut treatment site than in the control (Fig. 3.3A). Numbers of physid snails were not affected by logging (Fig. 3.3B, P=0.2604). Abundance of Sphaeriidae fingernail clams in the partially cut area (P=0.001) were greater than either the control or clearcut areas (Fig. 3.3C).

Chironomidae and Culicidae were the only insects that were abundant (comprised > 1% of total) in these study sites. Preliminary analysis suggested that midges in the genus *Chironomus* were responding differently than other chironomids and thus *Chironomus* and non-*Chironomus* midges (mostly *Polypedilum*) were analyzed separately. *Chironomus* midges were uncommon in all portions of the floodplain during the pre harvest years, but these midges became very abundant in the clear-cut area after the harvest (Fig. 3.4A). However, because distributions of *Chironomus* midges were highly variable even in the clearcut, differences in abundance among treatments were not statistically significant (P=0.0766). In contrast, non-*Chironomus* midges were abundant in all areas prior to harvest, but after harvest numbers were significantly lower (P=0.0053) in the clearcut treatment area than the control area (Fig. 3.4B). Numbers of Culicidae did not differ among treatments (P=0.1434, Fig. 3.4C). The last common group of invertebrates at the site was Oligochaeta, and their numbers remained similar among the three treatment areas (P=0.8426, Fig. 3.5).

In the small plot experiment where leaf litter was manipulated, we were only able to duplicate patterns associated with logging for a single taxon. Numbers of Crangonyctidae were significantly lower (P=0.0043) in plots where leaf litter was removed than in reference plots (Fig 3.6A). Asellidae (P=0.0049) and non-*Chironomus* (P=0.0045) midges, which both declined after logging, were both significantly higher rather than lower in leaf litter exclusion plots (Figs 3.6B&C). Impacts of clearcut harvest on crustaceans, mollusks, and insects were not reversed in plots that were shaded. This lack of invertebrate response occurred despite the fact that shading reduced the extensive bloom of herbaceous growth that occurred throughout the rest of the clear-cut area (P= 0.0210, Fig 3.7).

*Effects of Logging Peripheral Upland Pines on Embedded Depressional Wetlands and Efficacy of Riparian Buffer Strips on Reducing Impacts*

The most common invertebrates collected from the depressional wetlands were Crangonyctidae, Asellidae, Chironomidae, and Culicidae (Table 3.1). Because the sites located in Rincon flooded only once after harvest it was difficult to assess the impacts harvest had on invertebrates, although on that one date communities were similar in all four study ponds. At Rincon, maximum water depths (Fig 3.8), herbaceous plant biomass (Fig 3.9) and water chemistry were similar among 0-m buffer, 10-m buffer, 30-m buffer and reference sites.

Despite the drought the 0-m, 10-m, and 30-m buffered wetlands near Jones flooded yearly after peripheral harvest. However, the reference site never flooded over the years 1999-2001 (Fig. 3.10). Since the reference did not flood it was not possible to statistically compare invertebrates or water quality between buffered sites and the

reference. However among buffer sites, as at Rincon, densities of Crangonyctidae, Asellidae, Chironomidae and Culicidae were similar (all P> 0.05, Fig. 3.11). In the 0-m, 10-m, and 30-m buffer ponds, pH, temperature, and dissolved oxygen levels remained similar. After harvest, herbaceous plant growth increased in the three ponds located in the clearcut regardless of buffer width, while it remained limited in the reference site (P=0.018, Fig 3.12).

Numbers of Asellidae (P=0.2505, Fig 3.13B) and Chironomidae (P=0.1456,Fig 3.13C) were similar in the experimentally fertilized and reference depressional wetlands. However, the fertilized wetland had higher densities of Crangonyctidae than the reference wetland (P=0.0007, Fig 3.13A). Prior to fertilization (1998 collecting season) Crangonyctidae were rarely collected from either of these study ponds. With the increase of nutrients there was also an increase in herbaceous plant growth in the fertilized pond (P=0.0116, Fig 3.14). However, the majority of the herbaceous growth consisted of fern species instead of the sedge bloom that has been previously documented to follow a harvest (Fig 3.14; Batzer et al. 2000); sedges were the plants that responded at the Jones sites.

Although we were unable to evaluate effects of the loss of shading directly (because our experimental pond with shade plots remained dry), the buffers provided some evidence that variation in shade did not significantly affect plants or invertebrates in these ponds. At both Rincon and Jones, herbaceous plant and invertebrate communities remained similar among 0-m, 10-m, and 30-m buffer ponds.

## DISCUSSION

The invertebrate responses to timber harvesting were different in the floodplain and depressional wetland studies. In the floodplain study, several species of Crustacea, Mollusca, and Diptera were negatively impacted by harvest, while *Chironomus* midges benefited. In the depressional wetland where only peripheral trees were harvested, invertebrate populations did not change.

Variations in hydrological patterns induced by harvest might affect invertebrates. Because hydrologies of depressional wetlands are driven by precipitation and ground water (Faires et al. 1996) and harvesting of trees decreases evapotranspiration rates, these systems often get wetter after a harvest (Sun et al. 2000). In the Jones sites, aquatic invertebrates probably were benefited from timber harvest because the ponds in the clearcut were some of the few ponds in the region to flood during the 1999-2001 drought. However, in the depressional wetlands at Rincon, peripheral harvest of trees did not affect water levels and invertebrate densities were not affected. At the Coosawhatchie floodplain it seemed unlikely that logging-induced changes in hydrology would cause the observed changes in invertebrates. Here, the associated river largely drives the hydrology of the floodplain system and any effect of logging on hydrology would be overwhelmed by the flood pattern of the river. Furthermore there was a decrease in most invertebrates after the floodplain was logged. Increases in surface water from logging should not negatively impact most aquatic invertebrates, especially during a drought. It seems unlikely that hydrological changes induced by harvest caused declines in aquatic invertebrates on the Cooswhatchie floodplain. Batzer et al. (2000) hypothesized that invertebrates in wetlands typically deal with tremendous natural variation in hydrology,

and thus minor logging induced changes in hydrology may not significantly impact invertebrates.

Nutrient regimes were probably altered in both the floodplain and depressional wetlands, either from disturbance of soils or direct fertilization. However, impacts of nutrients on wetland invertebrates were unclear. In our fertilization experiment the only change associated with fertilizer was an increase in numbers of Crangonyctidae. However, at the Coosawhatchie floodplain, densities of Crangonyctidae and most other invertebrates decreased rather than increased in the clearcut area where nutrient availability should be the highest.

Harvest-induced increases in herbaceous plant growth were pronounced in both the floodplain and one of the two depressional systems. However, we found little evidence that these plant responses impacted invertebrate densities. Most invertebrate densities in the Coosawhatchie clearcut declined as herbaceous plant densities increased. In the shade plots where herbaceous plant biomass was kept, low invertebrate densities were not affected.

We suspect that the decrease of leaf litter input associated with logging might be an important influence on some invertebrates. Leaf litter can be an important food source for invertebrates, and decreases in leaf litter in streams have affected invertebrate densities (Wallace et al. 1997). In our leaf litter experiment, densities of Crangonyctidae declined in plots where litter was removed as compared to reference plots. The reverse pattern for Asellidae and Chironomidae (primarily *Polypedilum*) was perplexing. Apparently leaf litter was not particularly important to those organisms. In the depressional systems where only peripheral trees were harvested, leaf litter inputs by

wetland trees were minimally affected, and this may explain in part why invertebrate densities were not negatively impacted by that harvest.

Conclusions

Until now studies have reported that timber harvest had positive effects on invertebrate abundances (Batzer et al. 2000, Prenger & Crisman 2001). However, we found that impacts on invertebrates were largely dependent on whether harvesting of trees was direct or peripheral. Impacts to invertebrates were negligible when timber harvest was peripheral (in the depressional wetland study) but, when harvest was direct (in the floodplain study) invertebrate abundances dramatically decreased. If depressional wetlands were themselves logged the impacts to the invertebrates might mimic those observed in the floodplain study.

This study suggests that the impacts of direct logging might have significant effects on wetland ecosystems. Not only did invertebrate numbers decline, but there was a change in the community structure. The invertebrate community in the floodplain prior to harvest was largely comprised of Crangonyctidae, Asellidae, and non-*Chironomus* midges, but after harvest the dominant taxon collected was *Chironomus*. Declines in invertebrate abundances persisted for two years after harvest, and in combination with changes in community structure, these changes might have serious consequences for higher organisms that are dependent upon invertebrate communities.

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Table 3.1 Invertebrates collected from the Coosawhatchie floodplain and depressional wetland near Jones, GA.

| | Floodplain Forests % of Total | Depressional Wetlands % of Total |
|---|---|---|
| NON-ARTHROPODA | | |
| Oligochaeta | 2 | * |
| Mollusca | | |
| Planorbidae | 12 | -- |
| Physidae | 9 | -- |
| Sphaeriidae | 3 | -- |
| ARTHROPODA | | |
| Copepoda | * | * |
| Amphipoda | | |
| Crangonyctidae | 5 | 8 |
| Isopoda | | |
| Asellidae | 20 | 66 |
| Anostraca | | |
| Chirocephalidae | -- | * |
| Decapoda | | |
| Cambaridae | 1 | * |
| Cladocerans | | |
| Daphniidae | 3 | * |
| Ephemeroptera | | |
| Baetidae | * | -- |
| Odonata | * | * |
| Gomphidae | * | -- |
| Lestidae | -- | * |
| Libellulidae | -- | * |
| Hemiptera | | |
| Corixidae | * | * |
| Gerridae | * | * |
| Gyrinidae | * | -- |
| Hydrometridae | * | -- |
| Notonectidae | * | * |
| Coleoptera | | |
| Curculionidae | * | * |
| Dytiscidae | * | * |
| Hydrophilidae | * | * |
| Lampyridae | * | * |
| Noteridae | * | -- |
| Staphylinidae | * | -- |

| | | |
|---|---|---|
| Diptera | | |
| Chaoboridae | * | * |
| Chironomidae | 34 | 20 |
| Culicidae | 4 | 42 |
| Stratiomyidae | * | -- |
| Syrphidae | * | -- |
| Tabanidae | * | -- |
| Tipulidae | * | -- |
| Trichoptera | | |
| Hydroptilidae | * | -- |
| Phryganeidae | * | -- |
| Polycentropodidae | * | -- |

* asterisks indicate total abundances of < 1%

-- dashes indicate taxon was not collected from the site

Figure 3.1. Total invertebrate abundance among clearcut, partially cut and control treatments in both pre harvest and post harvest years at the Coosawhatchie floodplain. Densities were similar among the three treatments in both pre harvest (P=0.5602) and the post harvest periods (P=0.4578).

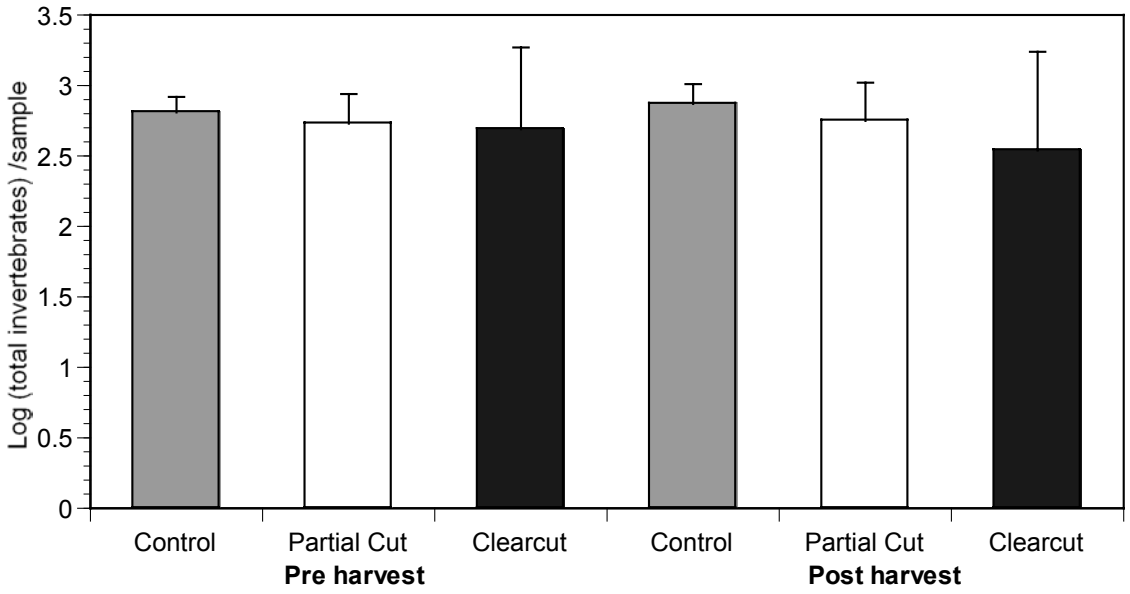
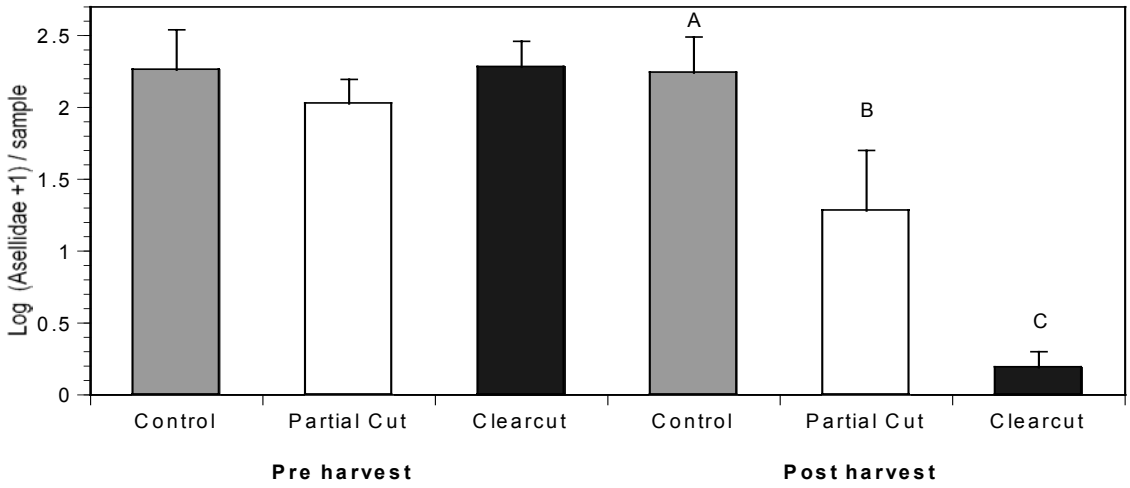
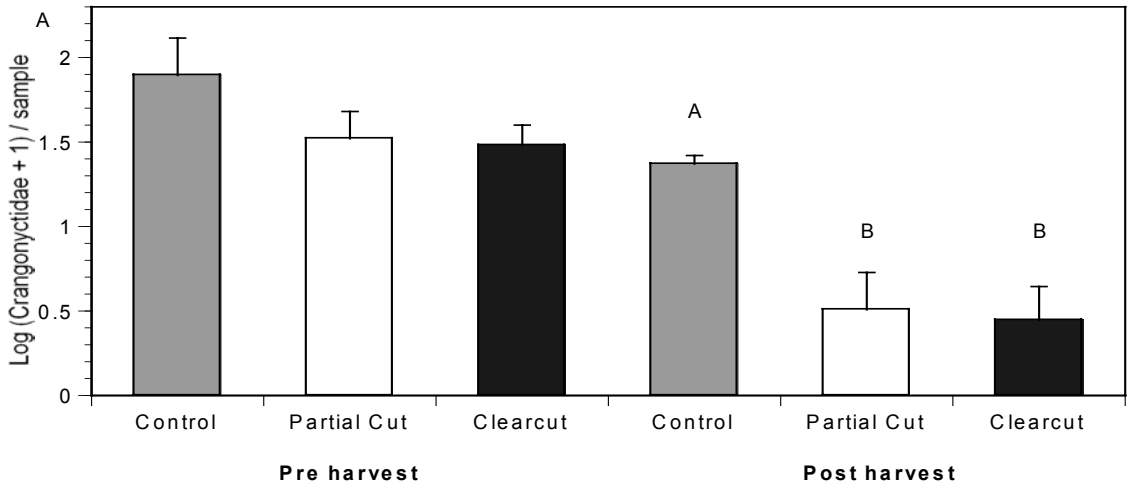

3.5
3
2.5
2
1.5
1
0.5
0
Log (total invertebrates) /sample
Control
Partial Cut
Clearcut
Control
Partial Cut
Clearcut
Pre harvest
Post harvest

Figure 3.2. Numbers of crustaceans among clearcut, partially cut, and control treatments at the Coosawhatchie floodplain during pre harvest (all $P>0.05$) and post harvest years: A) Asellidae ($P=0.0035$); B) Crangonyctidae ($P=0.0231$); C) Cambaridae ($P=0.0453$). Bars indicated by the same letter are not significantly different ($P>0.05$).

A. Asellidae




B. Crangonyctidae




C. Cambaridae

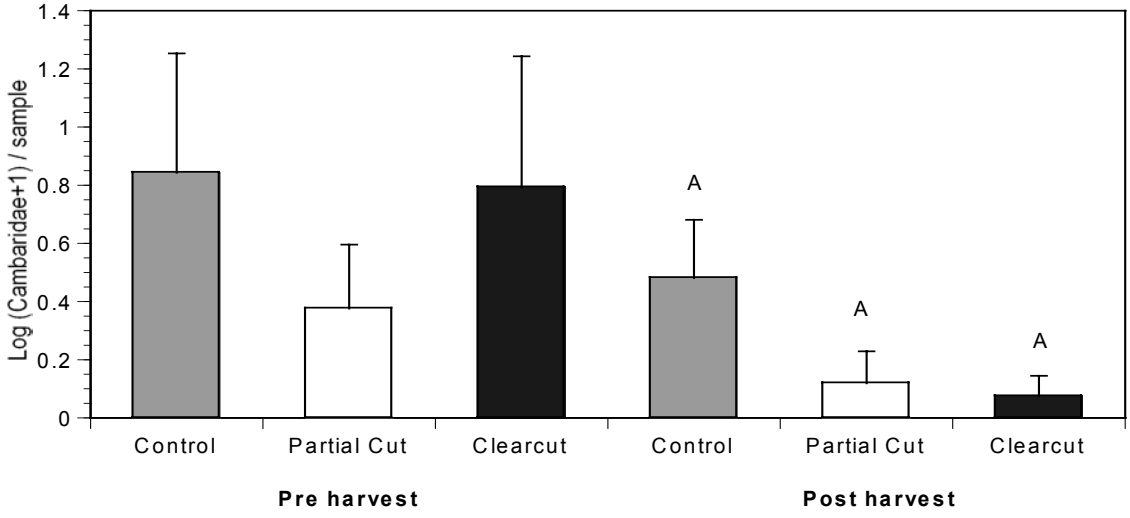
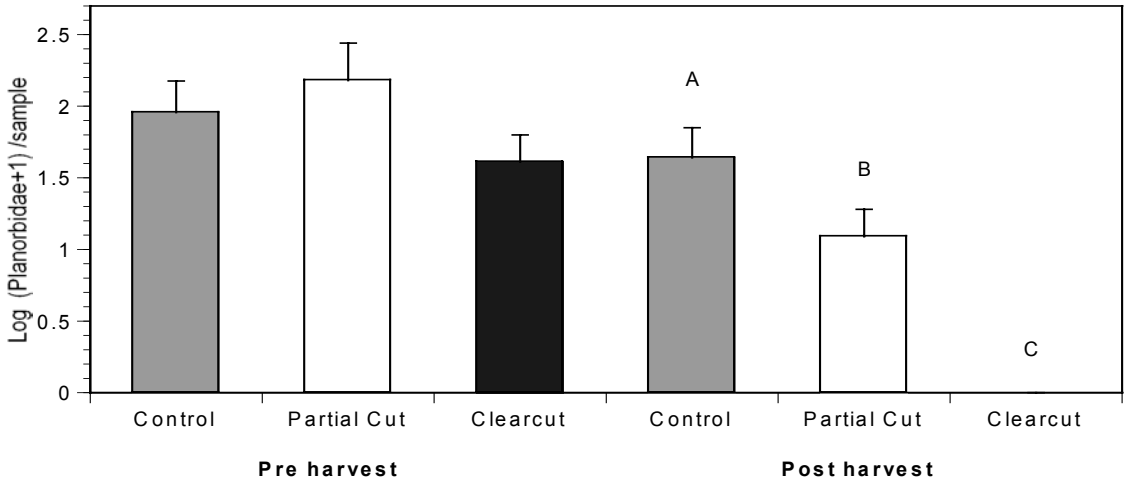
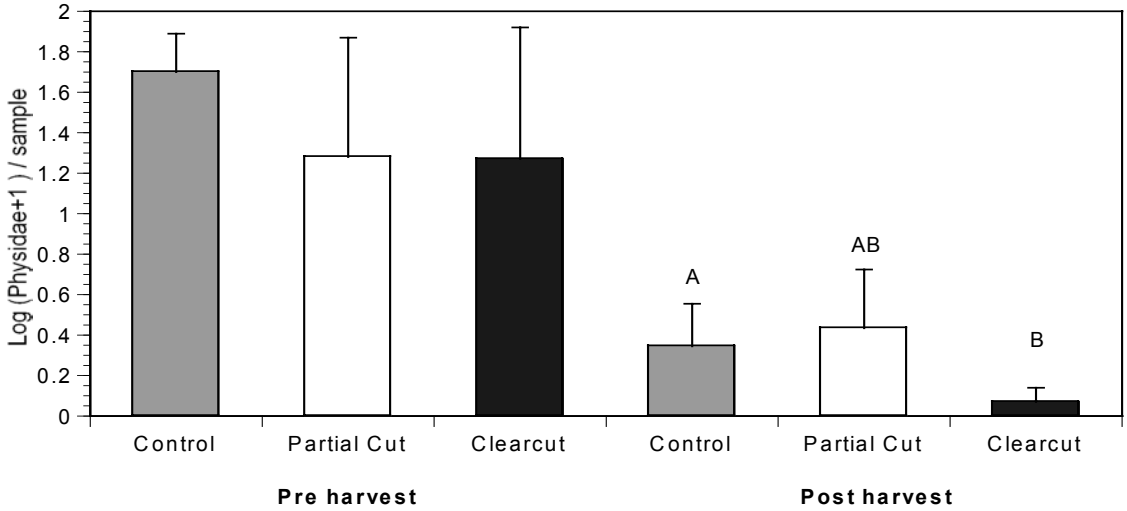


Figure 3.3 Numbers of mollusks among clearcut, partially cut, and control treatments in the Coosawhatchie floodplain during pre harvest (all $P>0.05$) and post harvest years: A) Planorbidae ($P=0.0006$); B) Physidae ($P=0.2604$); C) Sphaeriidae ($P=0.001$). Bars indicated by the same letter are not significantly different ($P>0.05$).

A. Planorbidae




B. Physidae




C. Sphaeriidae

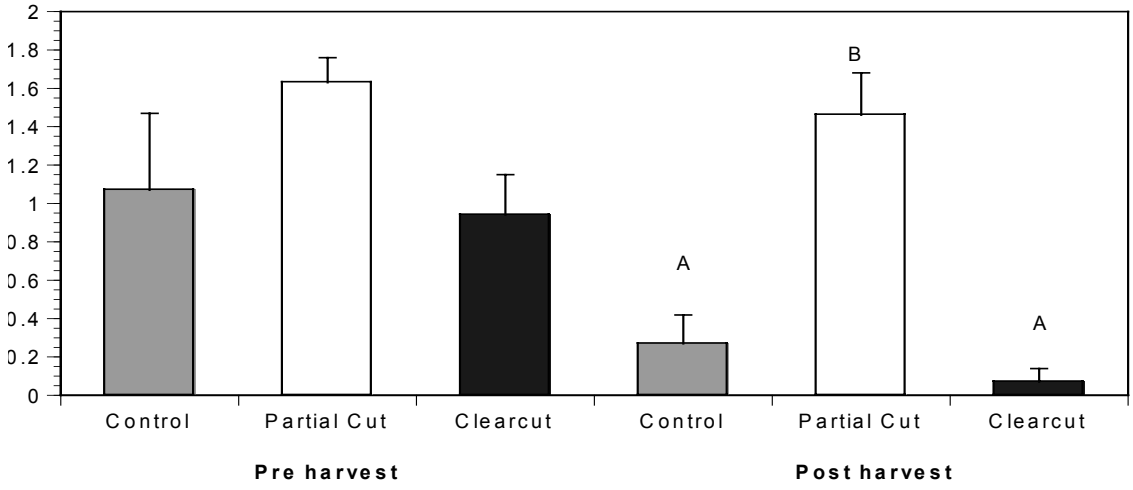
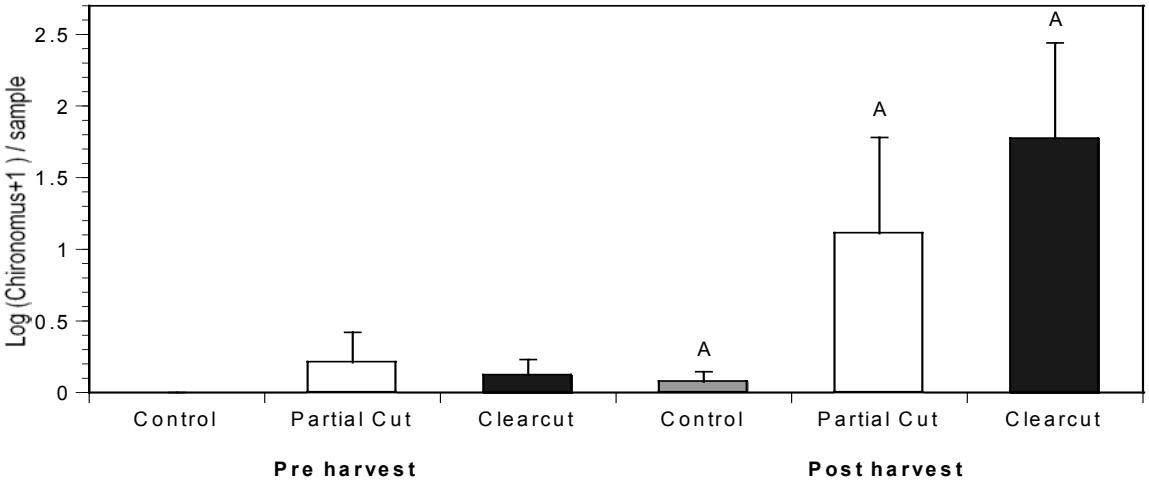
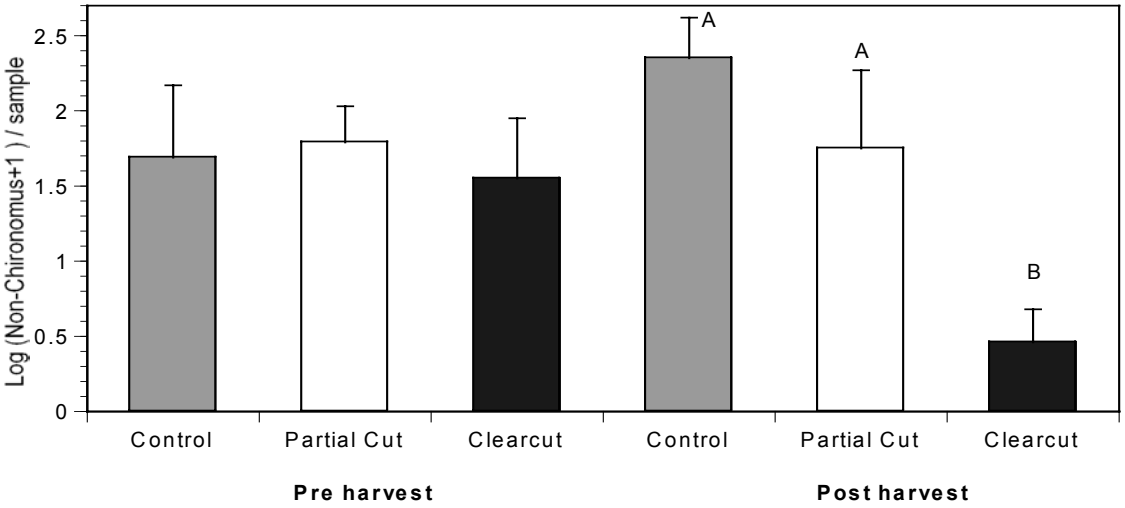


Figure 3.4. Numbers of Diptera among clearcut, partially cut, and control treatments in the Coosawhatchie floodplain during pre harvest (all P>0.05) and post harvest years: A) *Chironomus* (P=0.0766); B) Non-*Chironomus* midges (P=0.0053); C) Culicidae (P=0.1434). Bars indicated by the same letter are not significantly different (P>0.05).

A. *Chironomus*




B. Non-*Chironomus* midges




C. Culicidae

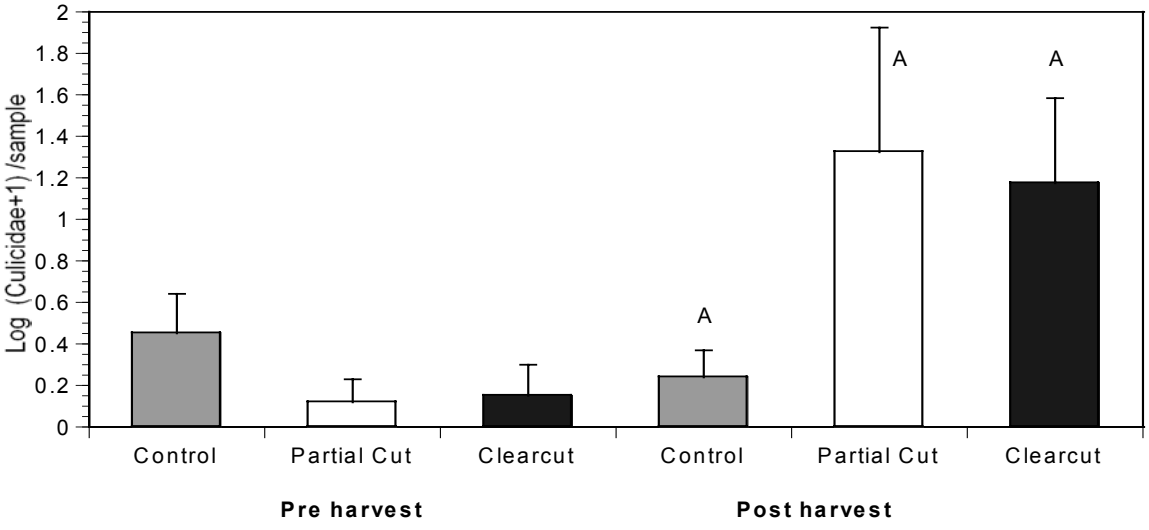


Figure 3.5 Numbers of Oligochaeta among clearcut, partially cut, and control treatments in the Coosawhatchie floodplain during pre harvest ($P>0.05$) and post harvest years ($P=0.8426$).

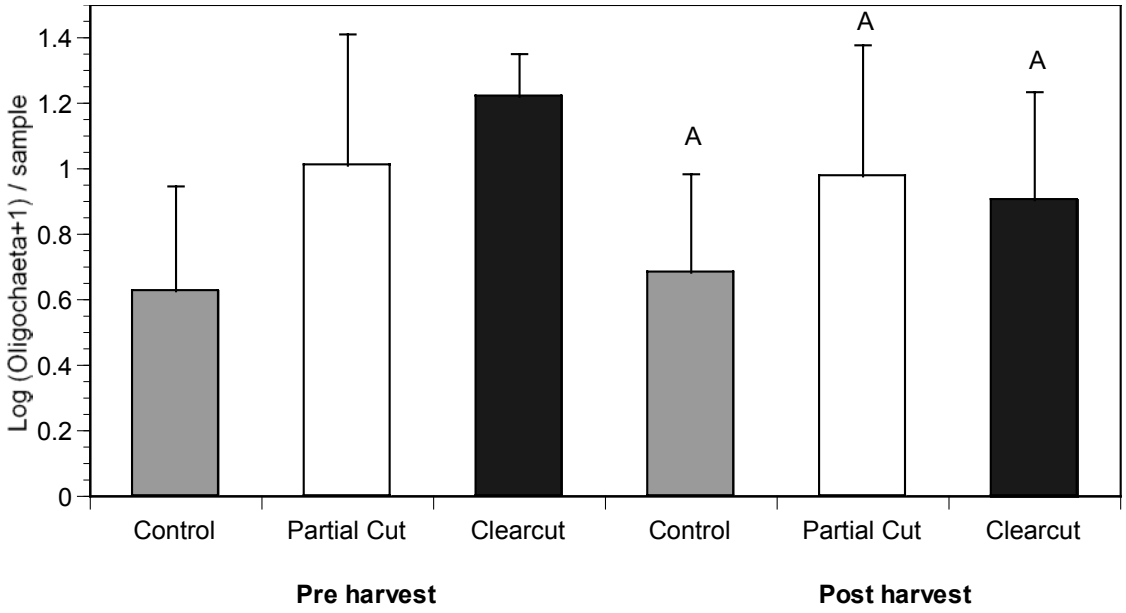
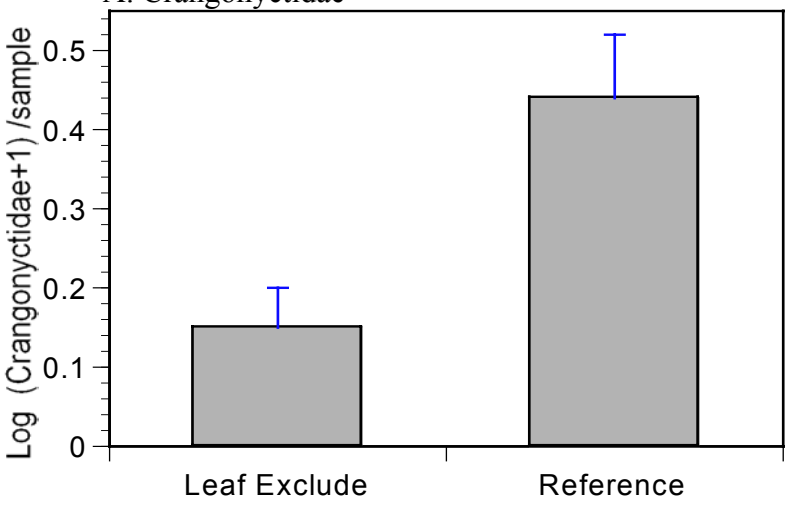
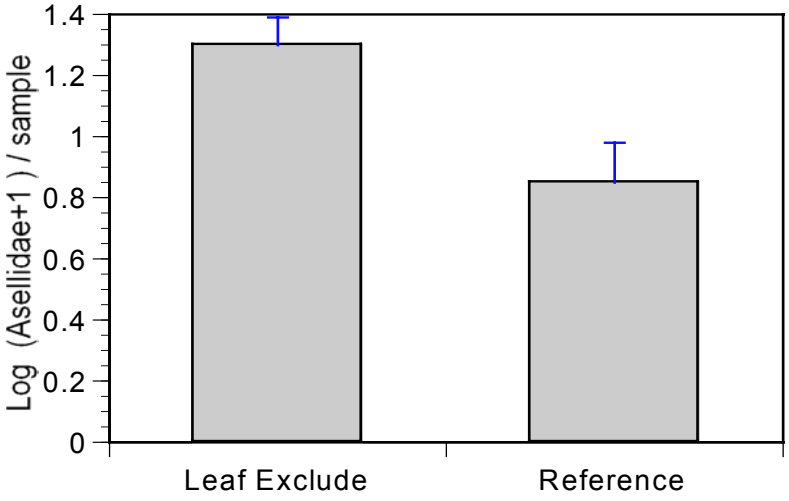

Log (Oligochaeta+1) / sample
0
0.2
0.4
0.6
0.8
1
1.2
1.4
A
A
Control
Partial Cut
Clearcut
Control
Partial Cut
Clearcut
Pre harvest
Post harvest

Figure 3.6. Numbers of common invertebrates from the Coosawhatchie floodplain collected from plots with leaf litter excluded vs. reference plots with ambient levels of litter: A) Crangonyctidae (P=0.0043); B) Asellidae (P=0.0049); C) Chironomidae (P=0.0045).

A. Crangonyctidae




B. Asellidae




C. Chironomidae,

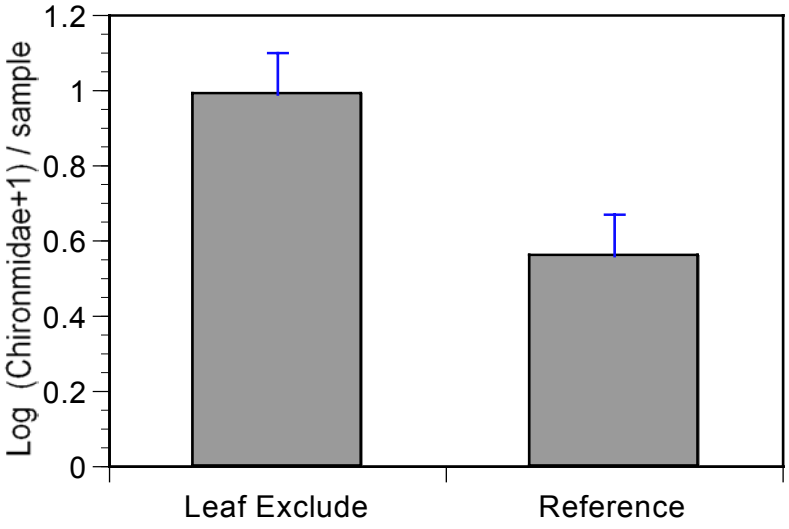


Figure 3.7. Herbaceous plant dry mass in shade manipulation plots at the Coosawhatchie floodplain. Dry mass was significantly lower in shaded plots (P=0.0210).

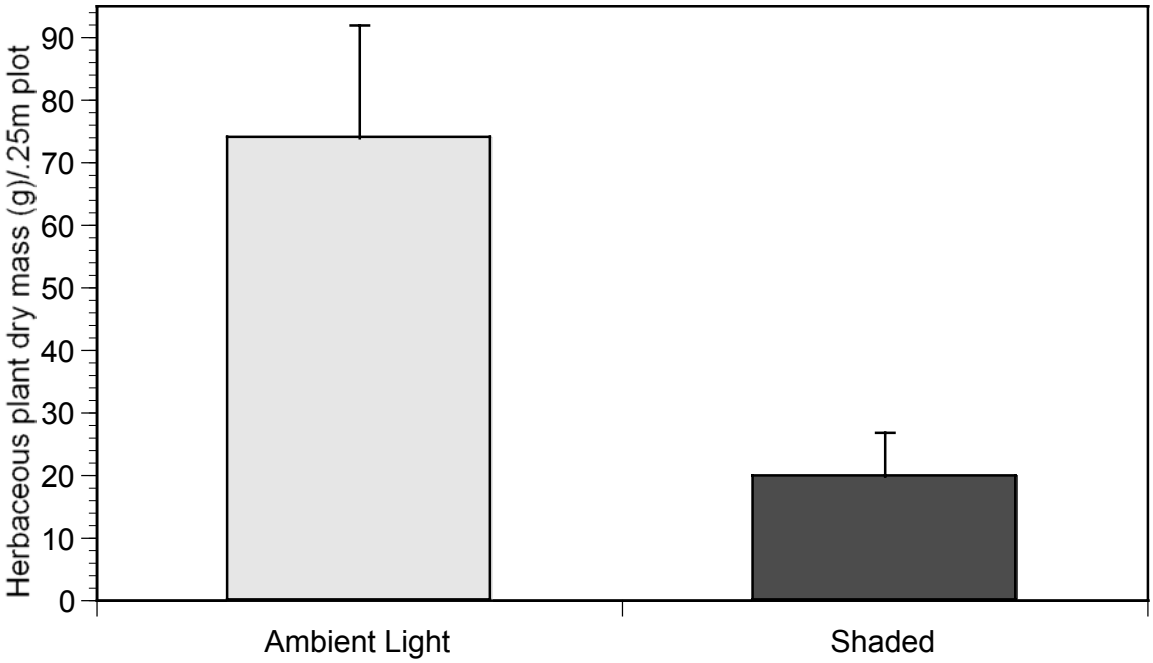

Herbaceous plant dry mass (g)/.25m plot
90
80
70
60
50
40
30
20
10
0
Ambient Light
Shaded

Figure 3.8 Maximum water depths from 1998 to 2001 in 0-m buffer, 10-m buffer, 30-m buffer and reference depressional wetlands located at Rincon, GA.

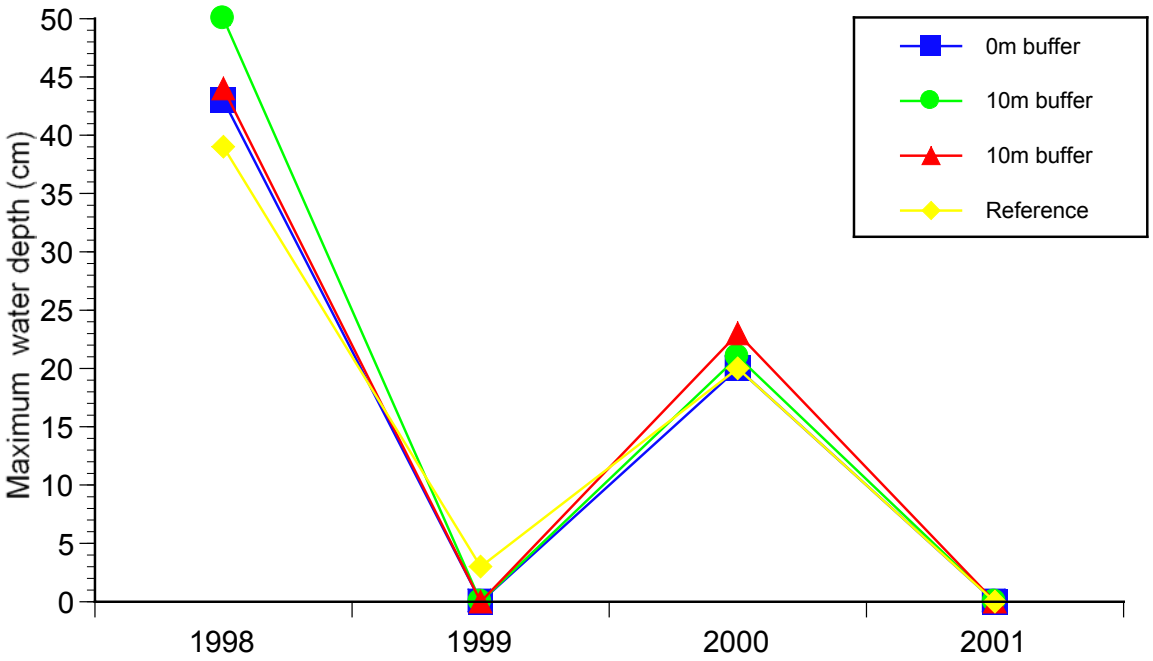

Maximum water depth (cm)
50
45
40
35
30
25
20
15
10
5
0
1998
1999
2000
2001
0m buffer
10m buffer
10m buffer
Reference

Figure 3.9 Herbaceous plant dry mass in 0-m buffer, 10-m buffer, 30-m buffer and reference depressional wetlands located at Rincon, GA.

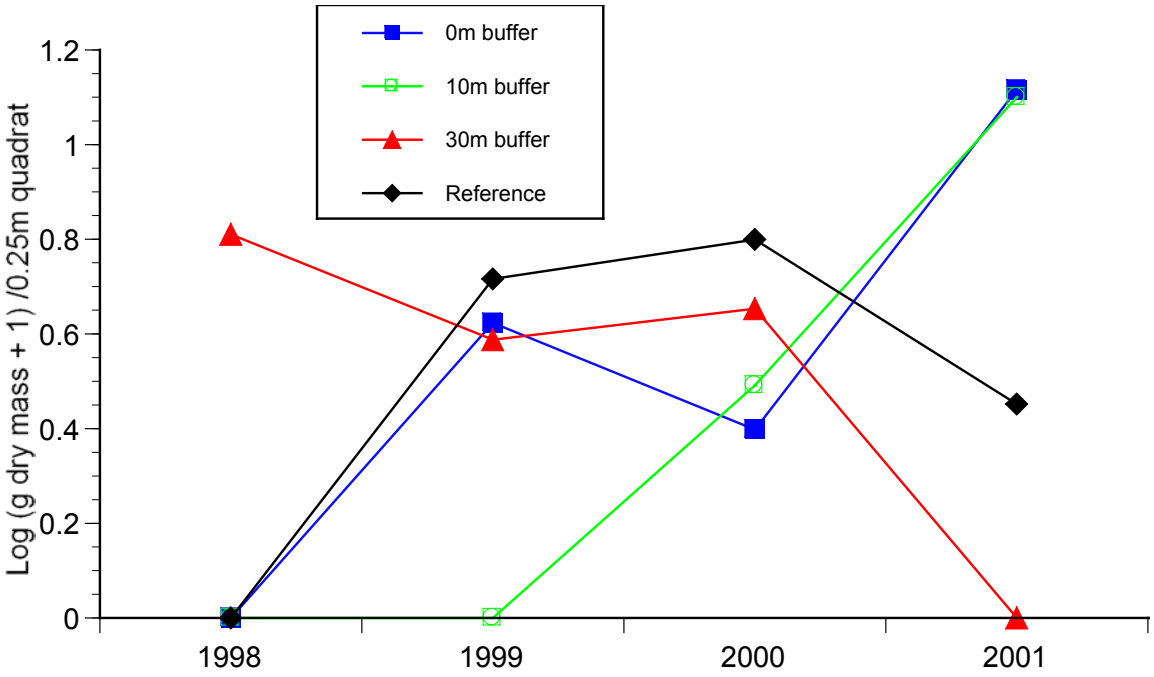

0m buffer
10m buffer
30m buffer
Reference
Log (g dry mass + 1) /0.25m quadrat
1.2
1
0.8
0.6
0.4
0.2
0
1998
1999
2000
2001

Figure 3.10. Maximum water depths from 1998 to 2001 in 0-m buffer, 10-m buffer, 30-m buffer, and reference depressional wetlands located at Jones, GA.

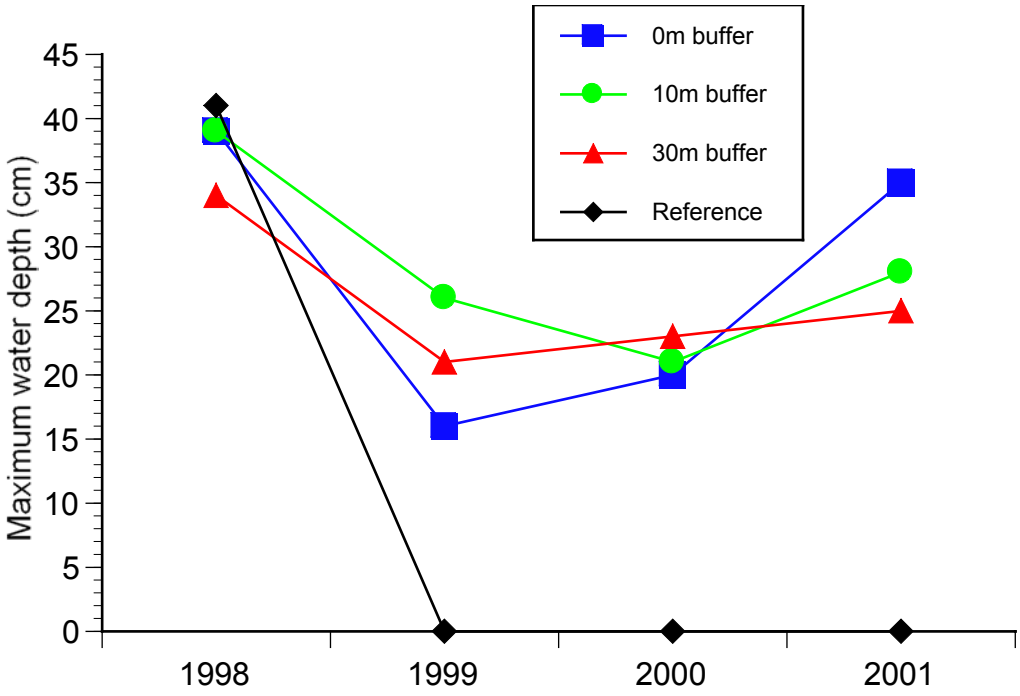
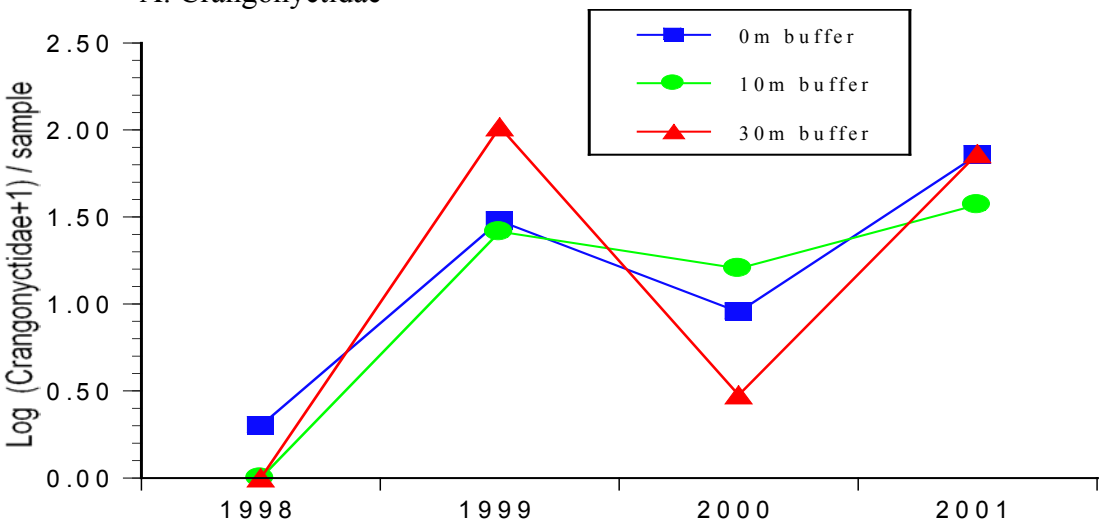
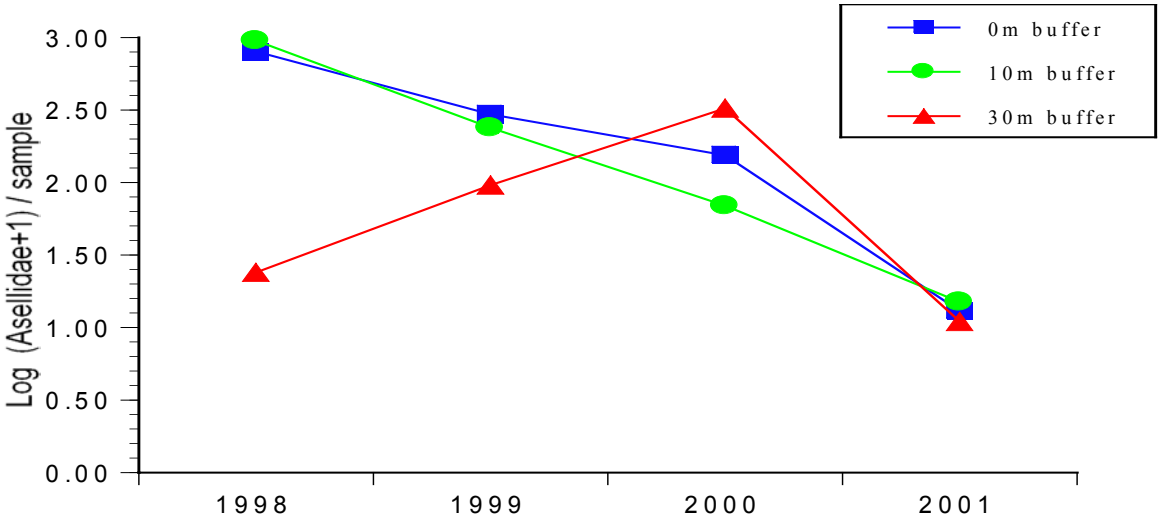

0m buffer
10m buffer
30m buffer
Reference
Maximum water depth (cm)
45
40
35
30
25
20
15
10
5
0
1998
1999
2000
2001

Figure 3.11. Numbers of common invertebrates in depression wetlands at Jones, GA with 0-m, 10-m, and 30-m buffer widths: A) Crangonyctidae; B) Asellidae; C) Chironomidae. Data for the reference wetland are not presented because it only held water in 1998. A sample consisted of three 1-m sweeps with a D-framed net, and multiple samples in each year were averaged.

A. Crangonyctidae




B. Asellidae




C. Chironomidae

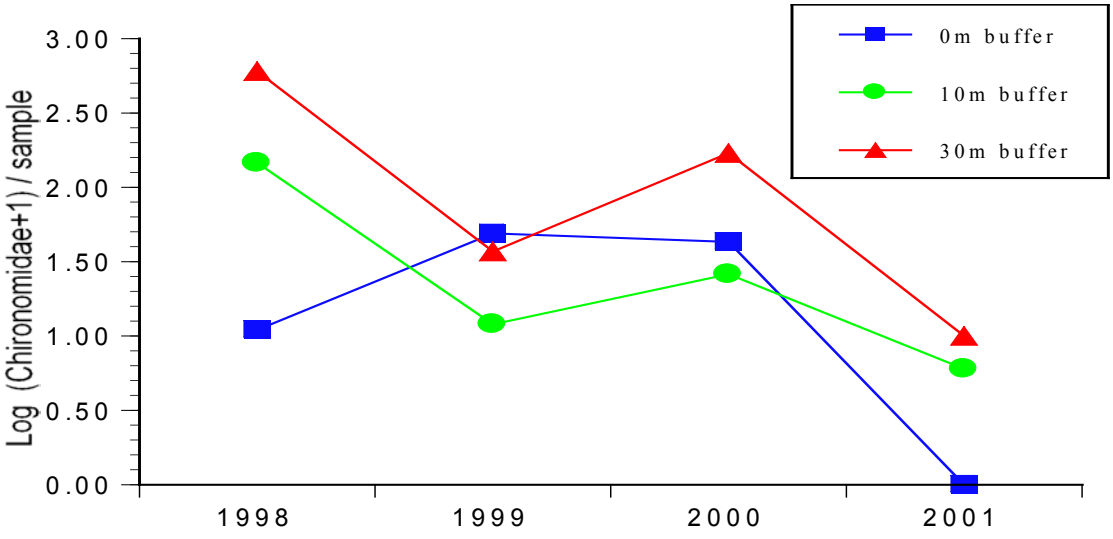


Figure 3.12. Herbaceous plant dry mass in 0-m buffer, 10-m buffer, 30-m buffer and reference depressional wetlands located at Jones, GA.

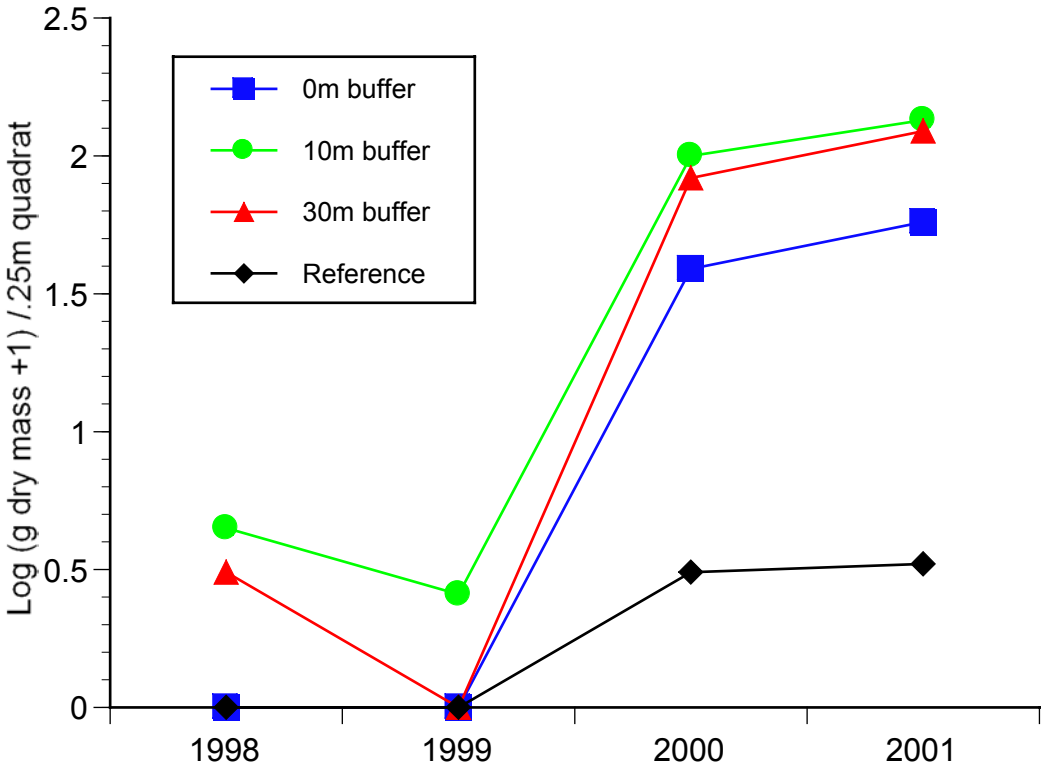
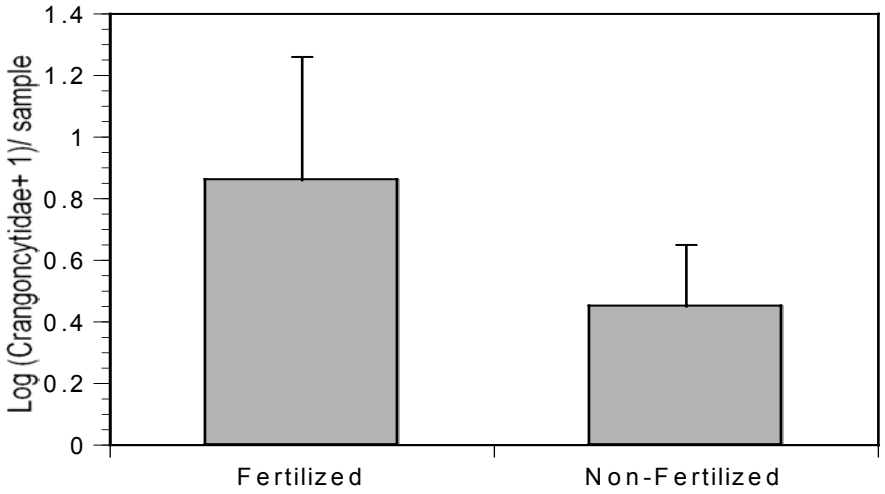
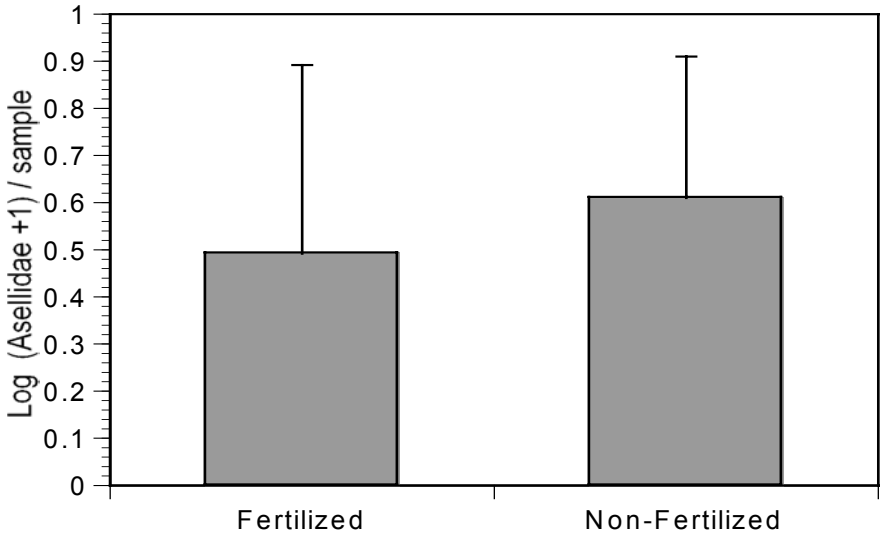

2.5
2
1.5
1
0.5
0
Log (g dry mass +1) /.25m quadrat
0m buffer
10m buffer
30m buffer
Reference
1998
1999
2000
2001

Figure 3.13. Abundances of common invertebrates in a fertilized wetland and a non fertilized wetland: A) Crangonyctidae (P=0.0007); B) Asellidae; C) Chironomidae.

A. Crangonyctidae




B. Asellidae




C. Chironomidae

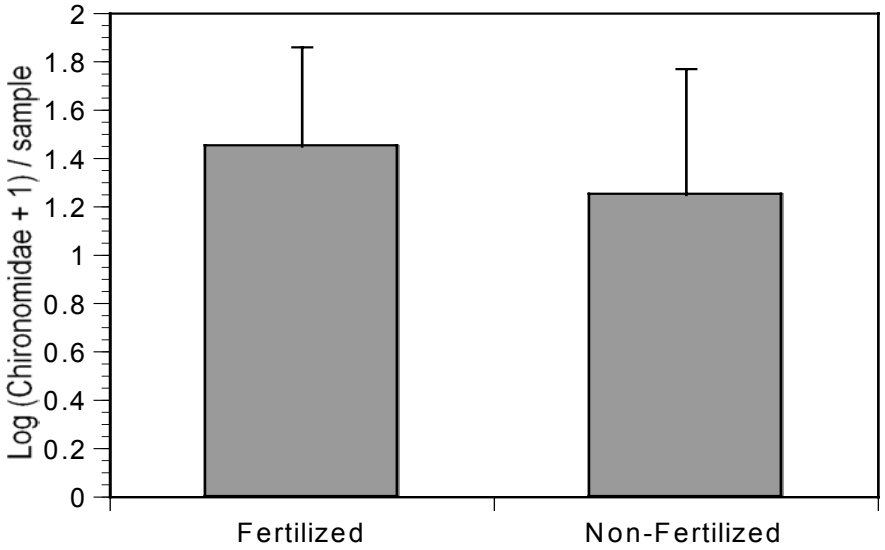
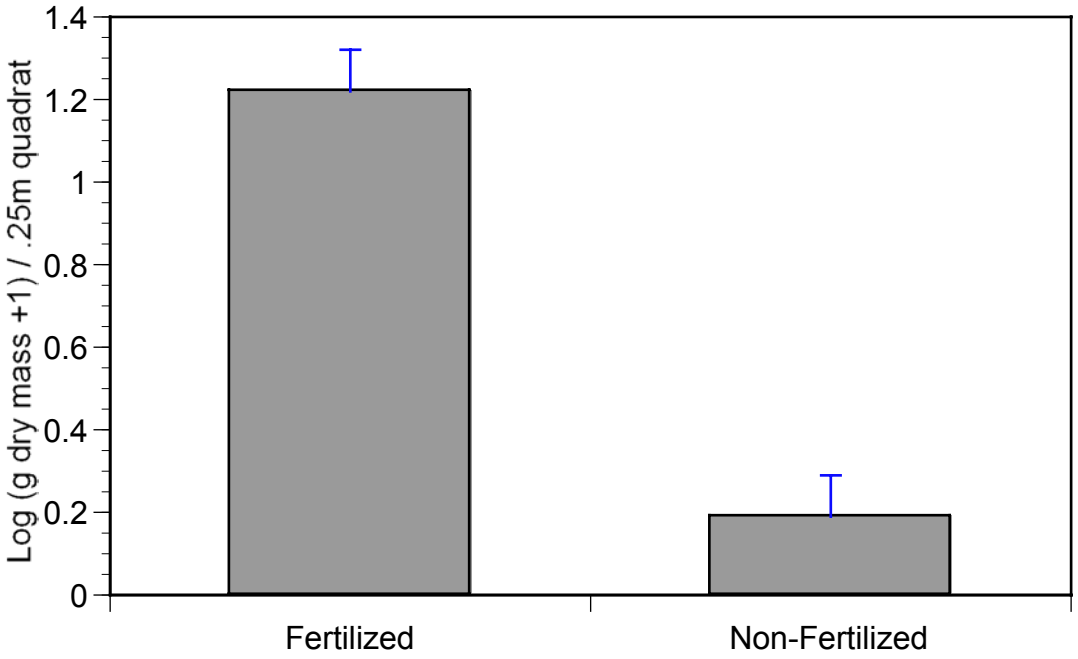


Figure 3.14 Herbaceous plant dry mass of a fertilized wetland and a non-fertilized wetland in Rincon, GA (P=0.0116).


1.4
1.2
1
0.8
0.6
0.4
0.2
0
Log (g dry mass +1) / .25m quadrat
Fertilized
Non-Fertilized

# CHAPTER 4

# CONCLUSIONS

Based on the results of these studies in the Okefenokee Swamp, Coosawhatchie floodplain, and Georgia Coastal Plain depressional wetlands, invertebrate bioassessment had the following advantages:

- Invertebrates readily sequestered heavy metals and these organisms could be used to detect the presence of heavy metals, such as mercury, in wetland habitats.
- Invertebrates could be used to detect spatial and temporal patterns of mercury within wetland habitats.
- Invertebrate numbers changed dramatically in response to direct harvesting of trees within a floodplain system and could be used to assess these impacts.

Although invertebrate bioassessment of wetlands clearly had virtues, problems with the technique include:

- Mercury concentrations were different among invertebrates, and each presented a different picture of mercury levels and variation.
- Mechanisms that caused variations in the temporal and spatial patterns of mercury were difficult to establish
- Mechanisms that caused changes in invertebrate numbers (in the floodplain system) after tree harvest were difficult to isolate.

- The responses of invertebrates to tree harvests differed depending upon which taxa of invertebrates were analyzed.
- Herbaceous plants responded to peripheral harvesting of trees while invertebrates did not. This suggests that herbaceous plants may be a more sensitive indicator of harvesting impacts than invertebrates.
- Yearly variations in weather affected invertebrate numbers in depressional wetlands and in some years invertebrates could not be used for assessment of tree harvesting.