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# Dynamics of Herbicide Concentrations in Mississippi Delta Oxbow Lakes and the Role of Planktonic Microorganisms in Herbicide Metabolism

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The small oxbow lakes central to the Mississippi Delta MSEA project provided a model system for evaluating the effects of watershed management and cropping practices on the dynamics of herbicide concentrations and planktonic populations. In 1996 and 1997, cotton was planted in about 50% of the area of three watersheds, and maximum fluometuron concentrations of 5.7, 5.0 and 12.4 μg $L^{-1}$ were observed in Beasley, Deep Hollow and Thighman lake water samples, respectively. The metabolite desmethyl fluometuron was present in all lakes (2.0 to 4.0 μg $L^{-1}$). In 1998, significant areas of Beasley and Thighman watersheds were planted in corn. Maximum concentrations of atrazine and metolachlor observed in Thighman lake water in 1998 were 15.0 and 7.2 μg $L^{-1}$, respectively, occurring in May. In Beasley Lake, maximum water concentrations of atrazine and metolachlor were 2.5 and 1.8 μg $L^{-1}$, respectively, occurring in late summer. Although differences in herbicide dissipation in the lakes may be partly explained by hydrology, cropping and management practices, the microbiological characteristics of the lake also need to be considered. Differences in planktonic populations and activity were observed among the lakes,

e.g., Thighman Lake had the highest enzymatic activity and bacterioplankton populations, and Beasley the lowest. Typically, the lowest suspended solids and the highest algal populations were found in Deep Hollow Lake during the rainy season, when the other lakes were sediment stressed. Certain algae have the potential for metabolism of atrazine and fluometuron via *N*-dealkylation. Laboratory studies indicated increased rates of fluometuron degradation when lake water samples were inoculated with the algae *Selenastrum capricornatum*, demonstrating the contribution of certain algae in herbicide metabolism.

Crop production in the Mississippi Delta region is highly dependent upon agrochemical use, thus non-point contamination of surface waters by herbicides is an environmental concern. In recent years, cropping systems and agrochemical use in this region have undergone dramatic changes. Understanding the relationship between crop management practices, pesticide use and maintenance of surface water quality is an important component of environmental stewardship, and a major objective of the Mississippi Delta Management System Evaluation Area (MDMSEA) project.

The occurrence and magnitude of herbicide concentrations observed in surface waters of the corn- and soybean-producing areas of the Midwestern U.S. have been well documented (*1*-*6*). The herbicides alachlor, atrazine, cyanazine, metolachlor and their respective metabolites are commonly observed in surface waters of this region and subsequently these contaminants drain into the Mississippi River system. By contrast, only limited studies have critically assessed the occurrence of herbicides in the Mississippi Delta region. A three-year survey of four Eastern Arkansas counties demonstrated that metolachlor, atrazine, norflurazon and cyanazine were observed in 7 to 13% of the samples (*7*). The occurrence of cotton and rice herbicides were studied in three Mississippi Delta streams (*8*); total concentrations of all pesticides exceeded 5 μg $L^{-1}$ with the order of occurrence molinate > fluometuron > cyanazine > metolachlor > norflurazon > atrazine > prometryn > propanil.

Oxbow lakes are common in the Mississippi Delta landscape. These water bodies are often used for recreational purposes (fishing and boating) and are important in wildlife habitat management. Despite the abundance of oxbow lakes in the Mississippi Delta landscape, little is known about the occurrence of pesticides in these water bodies and about their associated planktonic communities. The studies presented in this paper have been conducted with the following objectives:

- To characterize patterns of herbicide dynamics and planktonic populations in the MDMSEA oxbow lakes and evaluate the relationship to BMPs practiced in the watersheds.

- To ascertain the role of planktonic communities in herbicide transformations. Initial studies focused on the occurrence of the herbicide fluometuron that is traditionally applied at cotton planting. With changing cropping patterns in the MDMSEA watersheds the research scope was broadened to also study herbicides applied to corn, specifically, atrazine and metolachlor.

## Materials and Methods

### Herbicide Assessments

The management practices, characteristics and maps of the watersheds of the three oxbow lakes studied, Beasley, Deep Hollow, and Thighman, are described in the overview chapter in this volume (*Chapter by Locke*). Surface water samples were collected monthly from triplicate permanent sampling rafts located in each of the three oxbow lakes as described elsewhere (*Chapter by Knight, this volume*). Samples were refrigerated immediately and stored at $5^{o}$ C until processing, typically within 24 h. Water samples (a total of 500 mL) were centrifuged in 250 mL polypropylene centrifuge bottles (10 min 6,000 g). The supernatant was decanted into glass bottles and acidified with 1 N HCl to a pH of 3.0, and the pellets dried at $60^{o}$ C for 24 h for determination of suspended solids. The acidified water samples were concentrated by passing through a 47 mm C-18 Empore® disc (3-M, distributed by Varian Instruments), thereafter herbicides and metabolites were eluted with 20 mL of ethyl acetate (*9*). Residual water was removed from the ethyl acetate by addition of 1.0 g anhydrous sodium sulfate and the volume reduced to 2.0 ml under $N_2$ gas. Fluometuron and its metabolites (desmethyl fluometuron [DMF], trifluoromethylphenylurea [TFMPU], and trifluoromethylaniline [TFMA] were determined by high pressure liquid chromatography (HPLC) with fluorescence detection as described elsewhere *(10).* Water samples collected in 1998 were analyzed for atrazine, cyanazine, de-ethyl atrazine [DEA], de-isopropyl atrazine [DIA], and metolachlor by gas chromatography/ion trap mass spectrophotometry. Gas chromatographic conditions, such as column type, temperature program, He flow rates, and injector type and settings were similar to that described by Thurman et al. (*11*). Ion trap mass spectroscopy was run in the selective storage mode to provide sensitivity in the low parts per trillion (ppt) range for all analytes. Efficiency of recovery of spiked herbicides and that of a surrogate (tetrabutylazine) from lake water samples from all three lakes was about 85% or greater using the Empore discs. As lower levels of fluometuron were observed in later years, fluometuron concentrations in 1998 and 1999 samples were confirmed by enzyme linked immunoabsorbant assays [ELISA] (*12*) using commercial kits (EnviroLogix Inc., Portland, ME) in addition to HPLC analysis.

**Microbiological Assessment of Lake Quality**

Bacterioplankton populations (total heterotrophic bacteria, gram-negative bacteria and fluorescent pseudomonads) were assayed by serial dilution and spiral plating as described elsewhere (*13*). Algal populations were estimated using a most-probable-number (MPN) technique using five replicate tubes and Bristol's mineral salts broth as media (*14*). Heterotrophic biological activity of water samples was assessed by determining fluorescein diacetate (FDA) hydrolytic activity (*13*).

**Algal Herbicide *N*-Dealkylation**

Laboratory incubation studies using $^{14}$C-labeled herbicides were conducted to ascertain the role of green algae in the metabolism of atrazine and fluometuron, using *Selenastrum capricorantum* as a model system, because of its relatively high fluometuron and atrazine dealkylating activity. Eight tubes containing *S. capricorantum* cell suspensions (two mL of $10^7$ cells mL$^{-1}$ in Bristols media) were treated with 15 μM of either $^{14}$C-atrazine or $^{14}$C-fluometuron. Four replicate tubes of cells treated with $^{14}$C-atrazine (1.07 kBq mL$^{-1}$) also received 15 μM of unlabeled fluometuron and four replicate tubes treated with $^{14}$C-fluometuron (0.62 kBq mL$^{-1}$) received 15 μM of unlabeled atrazine. Two uninoculated controls were included for each of the four herbicide treatments. Tubes were incubated on an illuminated shaking incubator (25 μEinsteins m$^{-1}$ s$^{-1}$, 24$^{o}$ C, 100 rpm). Aliquots (250 μL) were removed every 24 h over a 72 h incubation, extracted with 750 μL of acetone and analyzed for the corresponding herbicides and metabolites using radiological thin layer chromatography methods described elsewhere (*10*).

A second study evaluated degradation of $^{14}$C-fluometuron in lake water collected from Deep Hollow and Thighman Lakes and the effects of augmenting with *S. capricorantum*. Lake water (60 mL, collected February 1999, prior to herbicide application) was added to sterile 250-mL flasks, and $^{14}$C-fluometuron (73 Bq mL$^{-1}$) was added to attain a concentration of 180 nmole L$^{-1}$. No residual fluometuron (< 0.2 ppb) was detected via HPLC or ELISA in these water samples at initiation of the experiment. Three flasks from each lake were inoculated with *S. capricorantum* (2.5 mL log 7.5 cells mL$^{-1}$), or treated with 2.5 mL of Bristol's media and were incubated under static conditions (25 μEinsteins m$^{-1}$ s$^{-1}$, 24$^{o}$C). Aliquots (6 mL) were removed periodically (3 to 8 d), acidified to pH 3.0 with HCl, and phase partitioned twice with 6 mL of ethyl acetate. Radioactivity in aqueous and ethyl acetate fractions was determined, and the ethyl acetate was concentrated and analyzed by radiological thin layer chromatography. Algal populations in water samples were estimated at initiation and termination of the study using the MPN technique previously described.

## Results and Discussion

### Occurrence of Fluometuron In Lake Water

Fluometuron was detected in all three lakes during the 1996 and 1997 growing seasons (Table I), with the highest concentrations observed in Thighman Lake. The highest fluometuron concentrations were typically observed in June, one to two months after cotton planting. The fluometuron metabolite, DMF, also was found in all three lakes when fluometuron was detected (data only shown for 1997, Figures 1a-c). In 1998 and 1999, corn was planted on most of Thighman watershed previously planted to cotton and little fluometuron was applied to this watershed. Thus, limited detections of fluometuron or DMF (<0.1 μg $L^{-1}$) were observed in Thighman Lake water during these years. In Beasley watershed, corn replaced cotton on about 30% of the crop land in 1998, and maximum concentrations of fluometuron were about half that observed in 1996 and 1997. During 1998 and 1999, rainfall in the Delta during June and July was about 40% that of 1996 and 1997, which resulted in substantially reduced runoff. However, during 1998 and 1999, lower maximum fluometuron concentrations and less frequent detections were observed in water samples from Deep Hollow Lake compared to Beasley Lake, even though a similar percentage of the total watershed was planted in cotton in each watershed. The reduced fluometuron levels in Deep Hollow Lake may be attributed to the intensive best management practices e.g., reduced tillage and winter wheat cover crop that were implemented at the Deep Hollow watershed.

**Table I. Maximum Fluometuron Concentrations Observed in the Three MDMSEA Oxbow Lakes, 1996 to 1999.**

| *Year* | *Beasley* | *Deep Hollow* | *Thighman* |
|---|---|---|---|
| | *μg $L^{-1}$* | *μg $L^{-1}$* | *μg $L^{-1}$* |
| 1996 | 2.6 ± 2.2[a] | 1.4 ± 0.8 | 12.4 ± 7.6 |
| 1997 | 5.7 ± 3.4 | 5.0 ± 0.6 | 11.2 ± 8.9 |
| 1998 | 1.6 ± 1.2 | 0.4 ± 0.1 | < 0.1 |
| 1999 | 2.8 ± 0.6 | 1.2 ± 0.8 | < 0.1 |

NOTE: Measurements are mean and standard deviation of three replicates

The seasonal dynamics of fluometuron and DMF concentrations for the three MDMSEA lakes in 1997 are presented in Figures 1a-c. In Thighman Lake, relatively high levels of fluometuron (11.2 μg $L^{-1)}$ were observed in the June sample (about two months after planting) coinciding with the maximum level of DMF detected. By July, concentrations of fluometuron and DMF were about 80% lower than that found in June. In Beasley and Deep Hollow Lakes, lower concentrations of fluometuron were observed, with both fluometuron and DMF being relatively more persistent in those lakes compared

to Thighman Lake. The accumulation and persistence of DMF in Beasley and Deep Hollow Lakes suggests that *in situ* fluometuron degradation may have occurred in lake water. No other fluometuron metabolite, e.g., TFMPU or trifluoromethylaniline (TFMA) was observed in any water sample during this study. Studies by others have shown that TFMA but not TFMPU occurring in water samples from the Mississippi Delta (*8*) and Texas Playa lakes (*15*) where fluometuron was used in the watersheds. Patterns of fluometuron and DMF occurrence and dissipation in Beasley and Deep Hollow Lakes were similar to that described in the Mississippi Big Sunflower River (*8*).

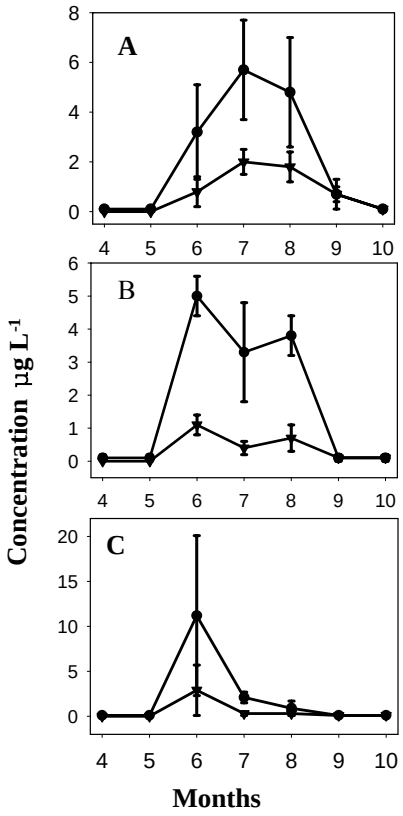


*Figure 1. Fluometuron* (●) *and desmethyl fluometuron* (▼) *concentrations observed in Beasley Lake (A), Deep Hollow Lake (B), and Thighman Lake (C) in 1997, mean and standard deviation of three replicate samples per occasion.*

**Occurrence of Other Herbicides in Lake Water**

As limited detections of fluometuron were observed in lake water samples in 1998, water samples were analyzed via GC/MS for other herbicides. The herbicides atrazine and metolachlor were applied to corn planted in Beasley and Thighman watersheds, and different patterns of accumulation and persistence of these herbicides were observed in Thighman compared to Beasley lake samples. Within a month following corn planting, peak concentrations were observed in Thighman Lake in May (Figure 2b). In Beasley Lake, peak concentrations of atrazine were observed in September (Figure 2a). The dealkylated metabolite, de-ethyl atrazine (DEA), accumulated in higher proportions to atrazine in Beasley Lake compared to Thighman Lake, suggesting that *in situ* degradation was perhaps occurring in the former. The second dealkylated metabolite, de-isopropyl atrazine (DIA), was also found in both lakes coincident with peak atrazine/DEA concentrations. Concentrations of DIA however were about 20 to 25% that of DEA (less than 0.4 $\mu g\ L^{-1}$). Atrazine concentration in Thighman Lake exceeded the Maximum

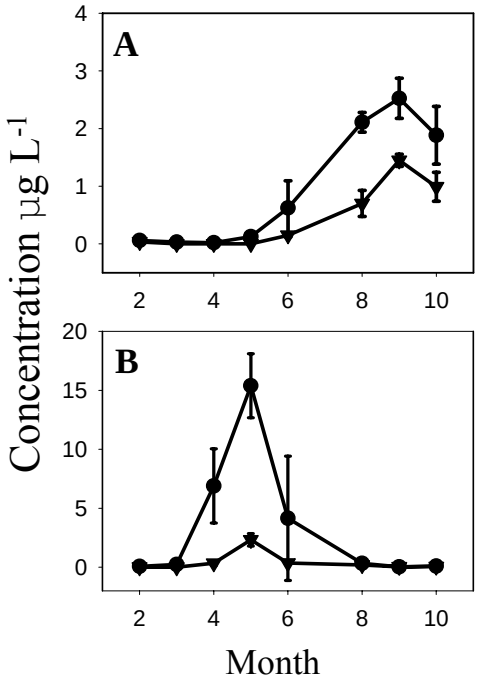


*Figure 2. Atrazine* (●) *and de-ethyl atrazine* (▼) *concentrations observed in Beasley Lake (A) and Thighman Lake (B) in 1998, mean and standard deviation of three replicate samples per occasion.*

Concentration Load (MCL) of 3 μg $L^{-1}$ for drinking water set by the EPA (*16*), and Canadian water quality standards of 2 μg $L^{-1}$ (*17*) for three months during the growing season. Levels of atrazine observed in Thighman Lake were greater than those reported in several Mississippi Delta streams (*8*), however concentrations are similar to those reported in the Midwest (*1-6*).

Metolachlor was applied at the same time as atrazine in the watersheds, and the dynamics of metolachlor appearance and dissipation followed the same dynamics as atrazine in Beasley and Thighman Lakes (Figure 3). Neither atrazine nor metolachlor were applied in Deep Hollow watershed, and neither was detected in water samples from Deep Hollow Lake. In Beasley Lake watershed, atrazine and metolachlor were incorporated in the soil, while they were surface applied in Thighman Lake watershed. This may have been one factor for delayed peak appearance in Beasley water samples. Studies on a small Pennsylvania watershed indicated that when atrazine was incorporated, its transport was reduced by about 50% (*18*). The presence of vegetative filter strips in Beasley watershed may have also temporarily impeded movement of both herbicides. The nature of the water flow through Thighman Lake suggests that hydrological characteristics were responsible for rapid flushing of atrazine, fluometuron and metolachlor following peak periods of run off, while Beasley Lake was a closed system, retaining herbicides that entered this system.

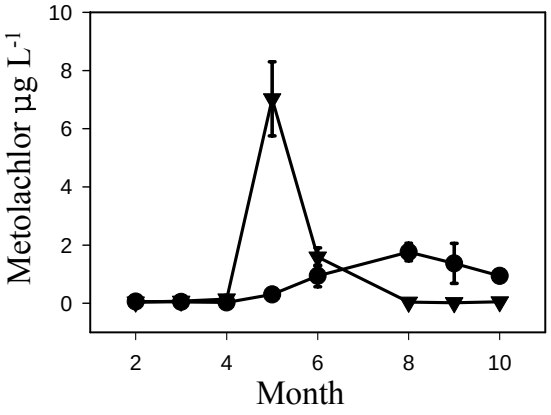


*Figure 3. Metolachlor occurrence in Beasley* (●) *and Thighman* (▲) *Lakes during 1998, mean and standard deviation of three replicate samples per occasion.*

Cyanazine was applied at planting to corn and postemergence directed to cotton later in the growing season, so the cotton can avoid injury. In 1998, maximum concentrations of cyanazine were observed in August and September in water samples collected from Beasley and Deep Hollow Lakes, while maximum but much lower concentrations were observed in Thighman Lake in April corresponding with application to corn (Figure 4). Cyanazine concentrations detected in Beasley Lake were five-fold higher than in Deep Hollow Lake, although applied to a similar percentage of the watershed, indicating that filter strips and other edge of field practices alone were not effective in reducing cyanazine movement into the lake water. Lower cyanazine concentrations found in Deep Hollow Lake compared to Beasley Lake suggest that the conservation management practices (reduced tillage and cover crop) were successful in minimizing runoff of this herbicide.

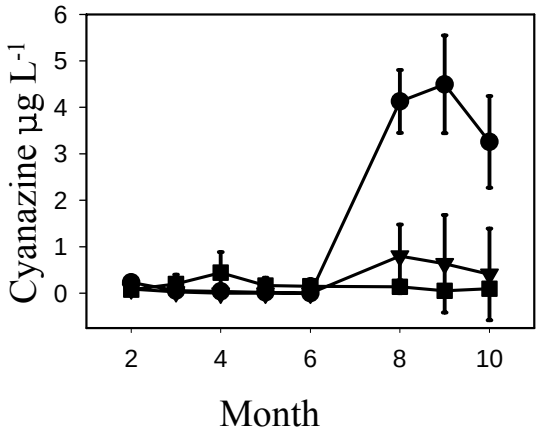


*Figure 4. Occurrence of cyanazine in Beasley* (●)*, Deep Hollow* (▲) *and Thighman* (■) *Lakes, 1998, mean and standard deviation of three replicate samples per occasion.*

**Microbiological Characteristics of Lake Quality**

Seasonal dynamics of suspended solids and FDA hydrolytic activity within the three lakes from January 1997 to July 1999 are summarized in Figures 5a and b. Maximum suspended solids were observed from January to June, corresponding with maximum precipitation and runoff. Relative levels of suspended solids observed in Deep Hollow during the rainy seasons ranged from 6 to 45% of suspended solids found in Beasley and Thighman Lakes. This dramatic reduction in suspended solids indicates that the combination of agronomic (reduced tillage and winter cover crop) and edge of field best management practices (filter strips and structural BMPs) used in Deep Hollow Lake

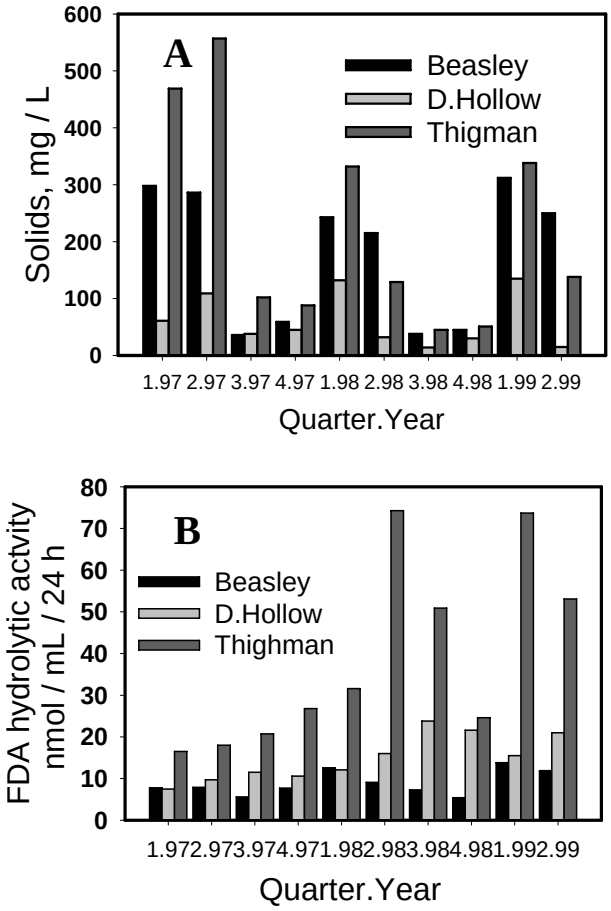


*Figure 5. Quarterly mean suspended solids (A) and fluorescein diacetate hydrolytic activity (B) of the three MDMSEA oxbow lakes, 1997 to 1999.*

watershed were effective in reducing runoff of sediment. Even though Beasley Lake watershed implemented edge of field BMPs, these alone were not effective in reducing sediment loss to the lake. The observations confirm other reports (*Chapter by Knight, this volume*) that both Beasley and Thighman Lakes were still sediment stressed.

Over the study period, levels of FDA activity were greatest in Thighman Lake, intermediate in Deep Hollow and lowest in Beasley Lake (Figure 5b). A 1996 assessment of these lakes did not show these differences in FDA activity (*13*). Seasonal populations

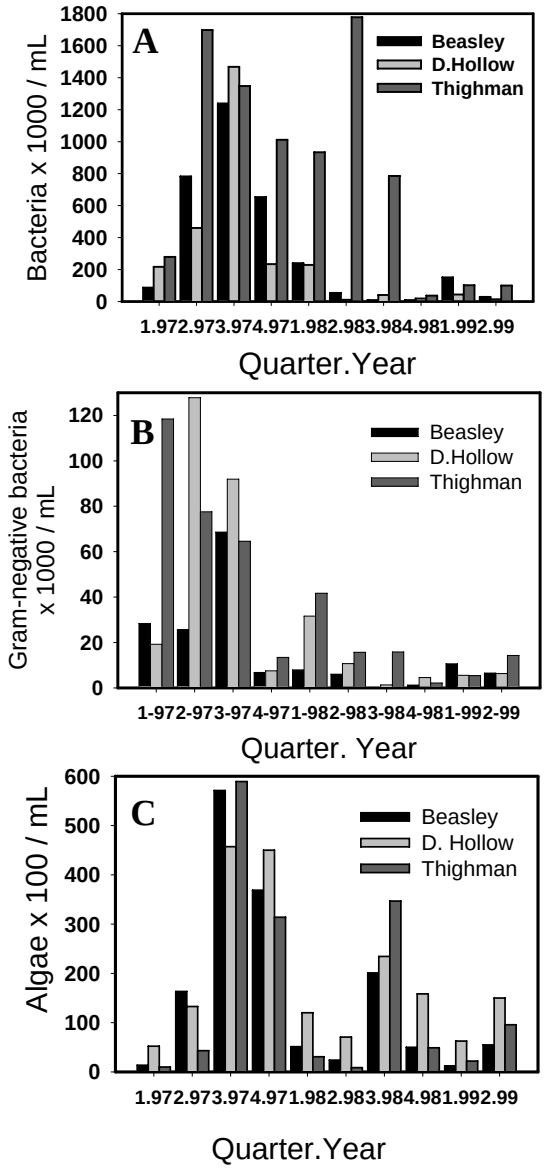


*Figure 6. Quarterly total bacterioplankton (A), Gram-negative bacterioplankton (B) and algal (C) population dynamics in three MDMSEA oxbow lakes 1997 to 1999.*

of total heterotrophic bacteria, gram-negative bacteria and algae are presented in Figures 6a to 6c. No consistent seasonal pattern of total heterotrophic bacteria or gram-negative bacteria was obvious during this study. Overall, Thighman Lake typically maintained the highest bacterioplankton populations compared to the other lakes, corresponding with higher rates of FDA activity. Thighman watershed was the only MDMSEA watershed with catfish ponds. The catfish ponds may have contributed carbon, nutrients and bacterial loading into this lake that might result in the higher populations of heterotrophic bacteria and subsequent FDA activity observed. In 1996, gram-negative bacteria, especially fluorescent pseudomonads, were a more abundant component of the bacterioplanktonic population in all three lakes (*13*). The highest density of algal populations occurred in late summer early fall corresponding with the warmest water temperatures. During periods of highest sediment load (winter and spring), Deep Hollow Lake typically maintained the highest algal populations, coinciding with the lowest levels of suspended solids. Thighman Lake has been shown to support the best establishment of introduced sports fish (*Chapter by Knight, this volume*). The productivity of Thighman Lake corresponds with certain biological indicators of lake productivity, e.g., highest levels of FDA activity and bacterial populations, although not evident in green algal populations. Herbicide biotransformations, especially co-metabolic transformations, are mediated by the magnitude, composition and activity of the indigenous microbial community. Based upon enzymatic FDA activity as an indicator of co-metabolic activity, Thighman Lake should support the highest level of herbicide degradation, Deep Hollow Lake intermediate and Beasley the least conducive, although not evident from these field assessments.

**Planktonic Herbicide Transformations**

There are many gaps in scientific assumptions that consider the toxicity of herbicides and other xenobiotics to aquatic organisms. The sensitivity of the planktonic community to various herbicides such as atrazine (*17*, *19*) and fluometuron (*20*, *21*) has been well established. The role of various terrestrial bacteria and fungi in the metabolism of herbicides in soil has been well characterized (*22*, *23*). Although algae are dominant components of the aquatic microflora, the contribution of these planktonic organisms in aquatic herbicide metabolism is poorly understood.

Previous investigations assessed the potential of fifteen green algae and two cyanobacteria cultures to metabolize fluometuron (*10*). As summarized in Table II, seven of fifteen green algae strains studied transformed fluometuron to the *N*-demethylated metabolite desmethyl fluometuron. The highest activity was found in two genera, *Ankistrodesmus* and *Selenastrum*, with lower levels of activity observed in strains of *Chlorella* and *Pediastrum*. One of these fluometuron-metabolizing algal strains (*Ankistrodesmus nannoselene*) was isolated from Deep Hollow Lake. The dominant genera of green algae observed in the MDMSEA lakes is *Chlorella* (unpublished data),

typically associated with disturbed, sediment stressed lakes. Of the *Chlorella* isolates tested either low of no fluometuron de-methylating activity was observed. These studies also demonstrated that algal strains that demethylated fluometuron were also capable of dealkylating atrazine with the ethyl group preferentially removed (*10*).

**Table II. Fluometuron *N*-demethylation Activity in Cell Suspensions of Various Genera of Green Algae and Cyanobacteria.**

| *Genus* | *Strains tested* | *Activity nmol desmethyl fluometuron formed 24 $h^{-1}$* |
|---|---|---|
| *Ankistrodesmus* | 2 | 2.2 –2.4 |
| *Anabaena* | 1 | <0.1 |
| *Chlorella* | 4 | <0.1 – 0.2 |
| *Chlamydomonas* | 1 | < 0.1 |
| *Oscillatoria* | 1 | <0.1 |
| *Pediastrum* | 1 | 0.4 |
| *Scenedesmus* | 2 | < 0.1 |
| *Selenastrum* | 3 | 1.6 – 1.8 |
| *Spirogyra* | 1 | <0.1 |

NOTE: Data summarized from reference 12.

As both atrazine and fluometuron were applied in the MDMSEA watersheds, laboratory studies were conducted to assess the ability of *S. capricornatum* to transform atrazine and fluometuron when both herbicides were present (Table III). Similar levels of fluometuron demethylation were observed in the presence or absence of atrazine. Atrazine de-ethylation activity in this study was about 30% that of fluometuron demethylation activity, and a 70% reduction of atrazine dealkylation activity was observed in the presence of fluometuron. These studies indicate that when mixtures of the two herbicides were present fluometuron was preferentially metabolized by *S. capricorantum* versus atrazine. Although atrazine is considered algistatitic to *S. capricorantum* (*17*, *19*), the potential of this species and others to detoxify this and other herbicides needs to be considered in ecotoxicity studies.

In the laboratory incubation study, minimal degradation of fluometuron occurred in non-inoculated lake water from either Deep Hollow or Thighman Lakes during a 28-d laboratory incubation (Figures 6 a and b), with only 3% of the radioactivity recovered as DMF. In this radiological laboratory study the concentration used (180 nM) is equivalent to 41 $\mu$g $L^{-1}$, several fold greater than maximum lake concentrations of fluometuron observed in the field. Augmentation of water with log 6.2 cells of *S. capricornatum* resulted in decreased ethyl acetate-extractable $^{14}$C-fluometuron, with a corresponding increase in DMF. Ethyl acetate recovered greater than 98% of the initial radioactivity

**Table III. Atrazine and Fluometuron *N*-dealkylation by *Selenastrum capricornatum* When Applied Alone or in Combination During a 72 h Incubation.**

| *Treatment* | *$^{14}$C-De-ethylatrazine* | *$^{14}$C-Desmethyl fluometuron* |
|---|---|---|
| | *nmol formed* | |
| $^{14}$C-atrazine | 1.4 ± 0.3 | |
| $^{14}$C-atrazine + fluometuron | 0.4 ± 0.1 | |
| $^{14}$C- fluometuron | | 4.8 ± 1.8 |
| $^{14}$C- fluometuron + atrazine | | 3.9 ± 2.0 |

Mean and standard deviation of four replicates

added from non-inoculated lake water throughout the study, while at the termination of the study about 90% was recovered from water inoculated with *S. capricornatum* (data not shown). The second demethylated metabolite, TFMPU, was only observed in augmented water from Thighman Lake (5% extractable radioactivity).

During the initial two weeks of incubation, there was a loss of chlorophyll from lake water augmented with *S. capricornatum*. Fluometuron herbicidal mode of action in plants is inhibition of photosynthesis II electron transport, which results in loss of chlorophyll and other pigments (*24*). Fluometuron has been shown to cause a loss of chlorophyll in the algae *Chlorella vulgaris* and *Chlorococcum humicola* (*20*, *21*), however, concentrations that inhibit chlorophyll accumulation were much higher than that used in the present study. Algal populations were initially log 6.4 cells $mL^{-1}$ in augmented lake water samples and declined to about log 5.2 cells $mL^{-1}$ at the termination of the study. Algal populations in non-inoculated lake water did not change during the study (about log 4.6 cells $mL^{-1}$). Predation by protozoa and other zooplankton may have been responsible for the decline in algal population of the introduced *S. capricornatum* in inoculated water samples. Relative rates of fluometuron degradation observed in these studies using natural water samples, were similar to pure culture studies when low inoculum densities of *S. capricornatum* (*10*). These studies of atrazine and fluometuron metabolism demonstrate the ability of certain green algae to transform herbicides commonly used in the Mississippi Delta.

Studies presented here describe the dissipation of herbicides in laboratory or microcosm systems, and attempts to demonstrate the role of specific planktonic organisms using augmentation with pure cultures. Microbial pesticide transformations typically occur via activity of a consortium of microorganisms, thus more research needs to be conducted towards assessing the role of the total planktonic community. Molecular biology tools, e.g., reverse transcriptase polymerase chain reaction, offer the potential for studying *in situ* gene expression of pesticide metabolizing enzymes. Future assessments using these techniques may aid in defining the role of planktonic communities in metabolizing xenobiotics under natural conditions.

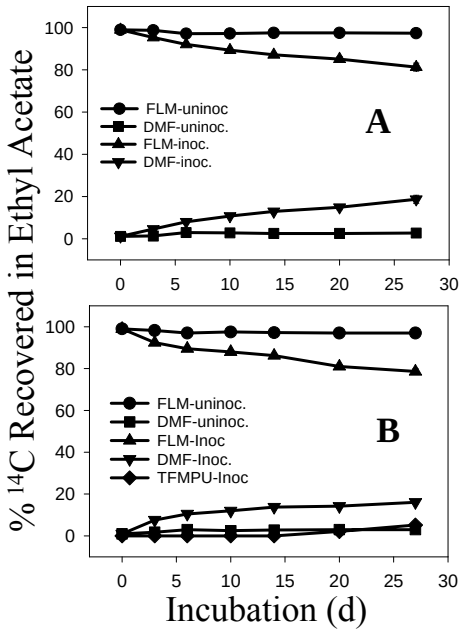


*Figure 7. Degradation of $^{14}$C-ring labeled fluometuron in water from Deep Hollow (A) and Thighman (B) Lakes as affected by inoculation with log 6.2 cells of S. capricornatum; mean of three replicates.*

**Acknowledgments**

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## References

1. Thurman, E.M.; Goolsby, D.A.; Meyer, M.T.; Koplin, D.W. *Environ. Sci. Technol.* **1991,** *25,* 1794-1796.
2. Goolsby, D.A.; Battaglin, W.A. In Selected papers on agricultural chemicals in water resources of the Midwestern United States; Goolsby, D.A.; Boyer, I.L.; Mallard G.E. Eds.; U.S. Geological Survey, Open File Report 93-418, **1991,** Washington D.C. pp. 1-25.
3. Thurman, E.M.; Goolsby, D.A.; Meyer, M.T.; Koplin, D.W. *Environ. Sci. Technol.* **1991,** *25,* 1794-1796.

4. Thurman, E.M.; Goolsby, D.A.; Meyer, M.T.; Koplin, D.W. *Environ. Sci. Technol.* **1992,** *26,* 2440-2447.
5. Jaynes, D.B.; Hatfield, J.L.; Meek, D.W. *J. Environ. Qual.* **1998,** *28,* 45-59.
6. Blanchard, P.E.; Lerch, R.N. *Environ. Sci. Technol.* **2000,** *34,* 3315-3322.
7. Senseman, S.A.; Lavy, T.L.; Mattice, J.D.; Gbur, E.E.; Skulman, B.W. *Environ. Sci. Technol.* **1997,** *31,*395-401.
8. Coupe, R.H.; Thurman, E.M.; Zimmermann, L.R. *Environ. Sci. Technol.* **1998,** *32,* 3673-3680.
9. Senseman, S.A.; Lavy, T.L.; Mattice, J.D.;Myers, B.D.; Skullman, B.W. *Environ. Sci. Technol.* **1993,** 27, 516-519.
10. Zablotowicz, R.M., Schrader, K.K.; Locke, M.A. J. Environ. Sci. Health. 1998, B33, 511-528.
11. Thurman, E.M.; Meyer, M.; Pomes, M.; Perry, C.A.; and Schwab, A.P. *Anal. Chem.* **1990,** *62,* 2043-2048.
12. Bastian, K.C.; Thurman, E.M.; Reibich, R.A. In *Proc. 28th Mississippi Water Resources Conference,* Daniel, B.J. Ed., **1998,** Mississippi State Water Resources Institute, p45-55.
13. Zablotowicz, R.M.; Locke, M.A.; Hoagland, R.E.; Knight, S.S. Cash, B. *Environ. Toxicol.* **2001,** *16,* 9-19.
14. Starr, R.C. *Amer. J. Botany.* **1964,** *51,* 1013-1044.
15. Thurman, E.M.; Bastain, K.C.; Mollhagen, T. *The Sci. Total Environ.* **2000,** *248,* 189-200.
16. U.S. Environmental Protection Agency. *The national water quality inventory. The 1992 report to Congress.* USEPA, Washington, D.C.
17. Trotter, D.M.; Baril, A.; Wong, M.P.; Kent, R.A. Canadian water quality Guidelines for Atrazine. Inland Waters Directorate, Water Quality Branch, *Sci. Ser.* 168, **1990,** Ottawa, Ontario.
18. Hall, J.K.; Hartwig, N.L.; Hoffman, L.D. *J. Environ. Qual.* **1983,** *12,* 336-340.
19. Caux, P.-Y.; Mènard, L.; Kent, R.A.; *Environ. Polut.* **1996,** *92,* 219-225.
20. Younnis, M.E.; Osman, M.E.H.; Soliman, A.I. *Quatar Univ. Sci.* **1988,** *8,* 85-101.
21. Younnis, M.E.; Osman, M.E.H.; Soliman, A.I. *Microbios,* **1991,** *67,*75-86.
22. MacRae, I.C. *Revs. Environ. Conatam. Toxicol.* **1989,** *109,* 1-88.
23. Zablotowicz, R.M.; Hoagland, R.E.; Locke, M.A. In *Pesticide Remediation in Soils and Water,* P.C. Kearney; T. Roberts (Eds.) Wiley and Sons, Chichester, U.K. **1997,** p217-250.
24. WSSA. In *WSSA Herbicide Handbook,* 7th ed.; Weed Science Society of America, Champaign, IL, **1994,** pp.135-137.