Illinois Water Resources Center
Annual Technical Report
FY 2014

Introduction

In 2014, the Illinois Water Resources Center secured over $400,000 in base and leveraged funding to work on water resources issues in Illinois. We helped shape state nutrient policy, supported almost 4000 rural well owners, fund nine research projects, and provide opportunities for over 50 students.

Research Program Introduction

In 2014, IWRC funded nine research projects. Below are some of the notable results from of these projects.

Vegetation management impacts mosquito population

Relevance: Summer means mosquitoes. But in recent years, mosquitoes have gone from just annoying to potentially deadly. With mosquito-borne diseases such as West Nile Virus on the rise, managing mosquito populations is an increasingly important public health concern.

Response: Research conducted by Brian Allan at the University of Illinois looked the aquatic juvenile stage of the mosquito life cycle to see if vegetation management and composition in urban water bodies impact survival rates. Preventing juvenile mosquito development is considered one of the most effective strategies for reducing adult mosquito abundance and risk.

Results: Allan's research shows that mowing vegetation in stormwater detention areas during the growing season can substantially increases the mosquito population. Ditches and dry detention areas with unmanaged vegetation were associated with relatively low juvenile mosquito populations. Vegetation type seems less important than management strategy in population control, and Allan's research indicates the presence of cat tails might even be associated with lower mosquito populations.

IWRC funding seeds larger research into Woody Perennial Polyculture research

Relevance: Conversion of the Midwest from native vegetation to intensive row crop agriculture has severely altered the region's hydrologic cycle. Restoring the hydrologic cycle of the region, along with other valuable ecosystem services, has important implications for climate change adaptation and mitigation, water quality, and agricultural production. While a complete return to native flora is unrealistic, a transition to a woody, perennial polyculture that once dominated the parts of the region could provide both a profitable and ecologically sustainable hybrid.

Response: Graduate student Kevin Wolz, University of Illinois, used IWRC funding to begin compare the hydrologic impact of a woody, perennial polyculture to conventional agriculture.

Results: IWRC seed funding and the preliminary research results enabled the research group to secure funding for a new research site that is over 10 times the size and will be able to answer much deeper questions on hydrology, water quality, and other key environmental metrics of diverse agroforestry systems in the Midwest.

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Flame retardants in Illinois urban waterbodies are among the highest in the world

Relevance: Flame retardants may be a good thing in a house fire, but they are not such a good thing in aquatic environments. Some flame retardants are persistent, bioaccumulative, and potentially toxic. However, to date an assessment of levels of flame retardants in Illinois waters has not been conducted.

Response: In 2013-2014, IWRC funded research Da Chen, Southern Illinois University, to examine the sources and distribution of flame retardants in Illinois rivers and lakes. Fish were collected in 35 selected aquatic systems across the state.

Results: This research revealed polybrominated diphenyl ether (PBDEs) are the most abundant flame retardant pollutants in Illinois waters. The levels in fish collected for this research show that PBDE concentrations in impaired urban waters are higher than comparative waterbodies in the US, Belgium, Spain, and China. Although commercial PBDE products have been discontinued in 2004, their high environmental concentrations merit continuing monitoring. Researchers also recommend continued monitoring of emerging flame retardants as they replace PBDEs in many applications.

Research helps characterize microplastics and other litter in Illinois water bodies

Relevance: Our garbage often ends up in rivers and lakes in the form of microplastics and other anthropogenic litter. That littler has both environmental and economic impacts, it clogs sewers, and can hurt animals and possibly humans.

Response: In 2014, IWRC funded researcher Tim Hoellin, Loyola University of Chicago, to examine the dynamics of anthropogenic litter (AL) in rivers and the abundance, sources, and microbial communities on riverine microplastics.

Results: Hoellin found that the riparian zones typically had higher density of AL than benthic habitats, but that riparian AL assemblages typically had a higher prevalence of light-weight materials. Hoellin also demonstrated that riverine AL is highly mobile, and that hydrology influences the export of AL downstream. Wastewater treatment plants are a key source of microplastic discharge to rivers.

Restored wetlands may need time to perform as expected

Relevance: Wetlands are an important ecosystem for the biological removal of nitrate by denitrification. Wetland mitigation is a common practice used to satisfy the “no net loss” requirements of the Clean Water Act, but research indicates that restored wetlands often do not achieve similar rates of denitrification as compared to more pristine wetlands. Since this is a process performed by microbial communities, it is possible that there is an overlooked biological component.

Response: IWRC funded Angela Kent to (1) to show how denitrifying communities in restored wetlands can change over time, (2) to compare denitrifier composition and denitrification capacity across a chronosequence of restored wetlands, and (3) to identify environmental factors that may facilitate the restoration of denitrifying services by affecting the microbial organisms involved.

Results: The denitrifier communities in the restored sites were very different from the core reference community (Bray Curtis dissimilarity = 0.87 ± 0.02), and communities in older restoration projects were no more similar to the reference than those in younger restored wetlands. However, denitrification showed a non-significant increasing trend as the age of the restoration increased, which may indicate a rapid response to the development of wetland environmental factors, while the microbial community shows a lag in development.

Determining the Fate and Toxicity of Polycyclic Aromatic Hydrocarbons Associated with Coal-Tar and Other Carbonaceous Material Particles in Urban Lakes

Basic Information

Title:	Determining the Fate and Toxicity of Polycyclic Aromatic Hydrocarbons Associated with Coal-Tar and Other Carbonaceous Material Particles in Urban Lakes
Project Number:	2011IL239G
USGS Grant Number:	G11AP20211
Start Date:	8/1/2011
End Date:	7/31/2014
Funding Source:	104G
Congressional District:	15 IL
Research Category:	Water Quality
Focus Category:	Non Point Pollution, Sediments, Surface Water
Descriptors:	None
Principal Investigators:	Charles J. Werth, Michael Jacob Plewa

Publications

There are no publications.

IWRC ANNUAL REPORT 2015

USGS-NIWR Project Title:
Determining the Fate and Toxicity of Polycyclic Aromatic Hydrocarbons Associated with Coal-Tar and Other Carbonaceous Material Particles in Urban Lakes

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Problem and Research Objectives:
Particle associated contaminants (PACs) have resulted in the impairment of thousands of streams, lakes, and reservoirs; PACs were responsible for fish-consumption advisories for 39 percent of total river mileage and 43 percent of total lake acreage in the United States in 2008. Results from recent water quality surveys indicate that metal, polychlorinated biphenyl, and DDT concentrations in freshwater sediments have generally decreased since their peak in the mid 1970's, consistent with their use and regulatory histories. However, total concentrations of polycyclic aromatic hydrocarbons (SPAHs) have increased, and generally with increasing urbanization. PAHs are toxic to aquatic life, and many are probable or suspected carcinogens. This is of special concern because many urban surface waters are used for human recreation (e.g., fishing, swimming) and/or drinking water.

Sources of particle-associated PAHs in urban lake sediments are located both within and outside the watershed. They include point (e.g., industrial emissions) and nonpoint (e.g., automobiles) combustion sources, asphalt from roads and parking lots, vulcanized rubber products such as automobile tires, and coal-tar and asphalt based sealcoats on parking lots and driveway pavement and roofs. Results from a number of our recent studies indicate that coal-tar pavement sealcoat is fluvially transported into urban streams and lakes with runoff, and can be the dominant source of PAHs in urban streams and lakes.

The overall goal of this study is to determine the fate and toxicity of PAHs associated with coal-tar particles in urban lake sediments. The specific objectives of this study are listed below.

1) Determine the sorption equilibrium and desorption kinetics of PAHs in coal-tar and other carbonaceous material particles that comprise urban lake sediments. We hypothesize that sorption capacities are low and release rates are high for PAHs in coal-tar and other less condensed carbonaceous materials (CMs) compared to highly condensed CMs like black carbon char and soot.

2) Determine PAH losses and redistribution associated with coal-tar particles in urban lake sediments. We hypothesize that lower molecular weight PAHs are released from coal-tar particles soon after burial (weeks to months) and taken up by more strongly sorbing black carbon, and that higher molecular weight PAHs are only lost to black carbon over much longer time scales (i.e., years) as phenolic and heterocyclic compounds that comprise coal-tar degrade. As a result, we hypothesize that PAHs are largely conserved in lake sediments and are not significantly released to the water column, and that sediment pore-water concentrations of PAHs decrease with aging.

3) Determine the toxicity of PAHs associated with coal-tar and other carbonaceous material particles in urban lake sediments. We hypothesize that toxicity of pore water in sediments decreases with time as PAHs and other organic pollutants redistribute from less strongly sorbing CMs like coal-tar to more strongly sorbing black carbon, and as less recalcitrant pollutants are biologically degraded over time. Such information is important because these lakes are sources of recreation and/or drinking water for large populations, and understanding coal-tar contributions to toxicity is an important step in protecting the environment and public health.

Methodology:

The proposed work combines bench scale laboratory experiments, field experiments, and laboratory analysis of field samples. It is divided into four tasks that cover 1) lake core retrieval, analysis, treatment, and in situ placement, 2) sorption isotherm and desorption kinetic profile measurement, 3) PAH and CM analysis of in situ cores, and 4) toxicity analysis of in situ cores.

Task 1: Lake core retrieval, analysis, treatment, and in situ placement

Task 1 is complete and outlined in last year's report. Briefly, in situ cores were amended with carbonaceous materials (CMs) spiked with deuterated PAHs and placed into the top layer of sediment at Whitnall Park Pond in June 2013. These in situ cores have been retrieved over the past two years for analysis. The final set of samples is scheduled to be removed from Whitnall Park Pond in June 2015. Images of the in situ cores as well as the four types of CMs are shown below in Figures 1 and 2.

Figure 1: In situ core design and field deployment (credit: Victoria Boyd, UIUC and Peter Van Metre, USGS)

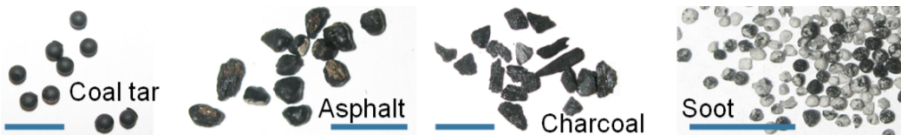

Figure 2: Carbonaceous materials, the scale bar is 1 cm (credit: Victoria Boyd, UIUC)

Each type of CM was spiked with three unique deuterated PAHs in a range of molecular weights. This allows for the transport of PAHs from each CM to be measured. Using a range of molecular weights in the spike makes is possible to determine the effects of PAH molecular weight on transport.

Task 2: Sorption isotherm and desorption kinetic profile measurement
Efforts on this task have just begun.

Task 3: PAH and CM analysis of in situ cores
Retrieval of in situ cores was completed with the help of divers as shown in Figure 3. Analysis of the in situ cores has started. During field deployment the samples were divided into different layers in order to determine if sediment depth had an impact on PAH transport. Mesh screens were added at that time to separate the different layers of sediment. This allowed for the samples to be divided back into the same depths after retrieval. Samples retrieved from the lake were frozen in order to preserve the layering of the sediment and cut open (Figure 4).

Figure 3: Divers retrieved in situ samples (credit: Victoria Boyd, UIUC)

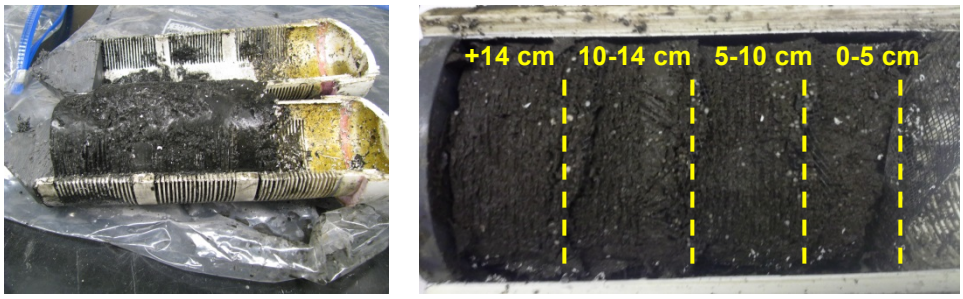

Figure 4: left) samples were frozen and cut open, right) mesh screens were used to separate sediment layers (credit: Victoria Boyd, UIUC)

The samples were wet sieved with nanopure water in order to separate the amended particles from the sediment. The sediment and water was collected and the particles were divided by hand (Figure 5).

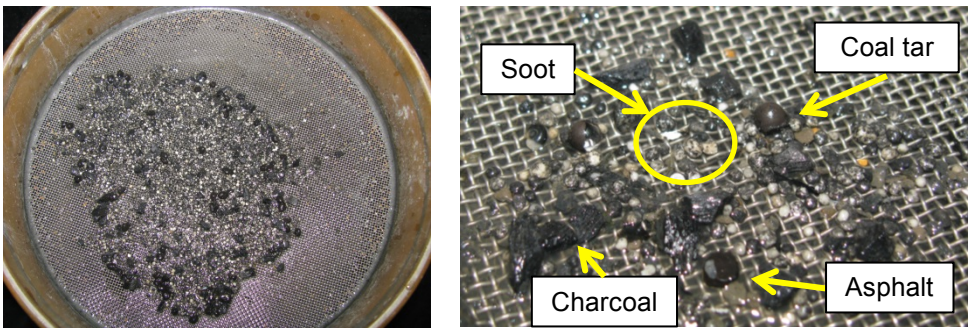

Figure 5: Particles were separated from the sediment by sieving (credit: Victoria Boyd, UIUC)

The sediment and particles are currently being analyzed for PAHs. Sediment samples are dried using a Labconco Freeze Dry System prior to analysis. Samples are extracted using EPA method 3534 for pressurized fluid extraction using a Dionex Accelerated Solvent Extractor. The resulting extract is cleaned following EPA method 3630c for silica gel clean up, and finally analyzed with GC/MS.

Task 4: Toxicity analysis of in situ cores
Mutagenicity and cytotoxicity (survivorship) experiments were performed using a single colony isolate of *S. typhimurium* TA100 that was grown overnight in 50 mL LB medium plus 100 µL ampicillin stock solution at 37°C with shaking (200 rpm). The following day, the overnight culture of TA100 was used to evaluate an extract of coal-tar for PAH-induced mutagenicity using a plate incorporation assay. For this assay, overlay tubes were prepared with 2 mL of histidine-biotin, supplemented over agar with 100 µL of overnight bacterial culture and ± 500 µL hepatic microsomal activation (S9) mix. The required volume of the test agent was added to the overlay tube, the tube was flamed sterilized and immediately poured onto a VB minimal plate. The plates were incubated at 37 °C for 36-72 h. Histidine revertant colonies were counted by hand or with a New Brunswick Biotran III automatic colony counter. To confirm the genotype of the TA100 cells, 100 µL of the bacterial cell suspension was added to an LB plate, and spread with a flamed glass rod. Flamed tweezers were used to place a crystal violet disk onto the center of the plate, and the disk was tapped lightly in place. Experiments are planned to evaluate the mutagenicity and cytotoxicity of coal tar extract surrogates containing only PAHs (not other components of the extract), and lake sediment extracts.

Principal Finding and Significance:
Background concentrations in sediment and the CMs were measured in order to have a baseline to compare the initial conditions of the sediment and CMs to the in situ field samples.

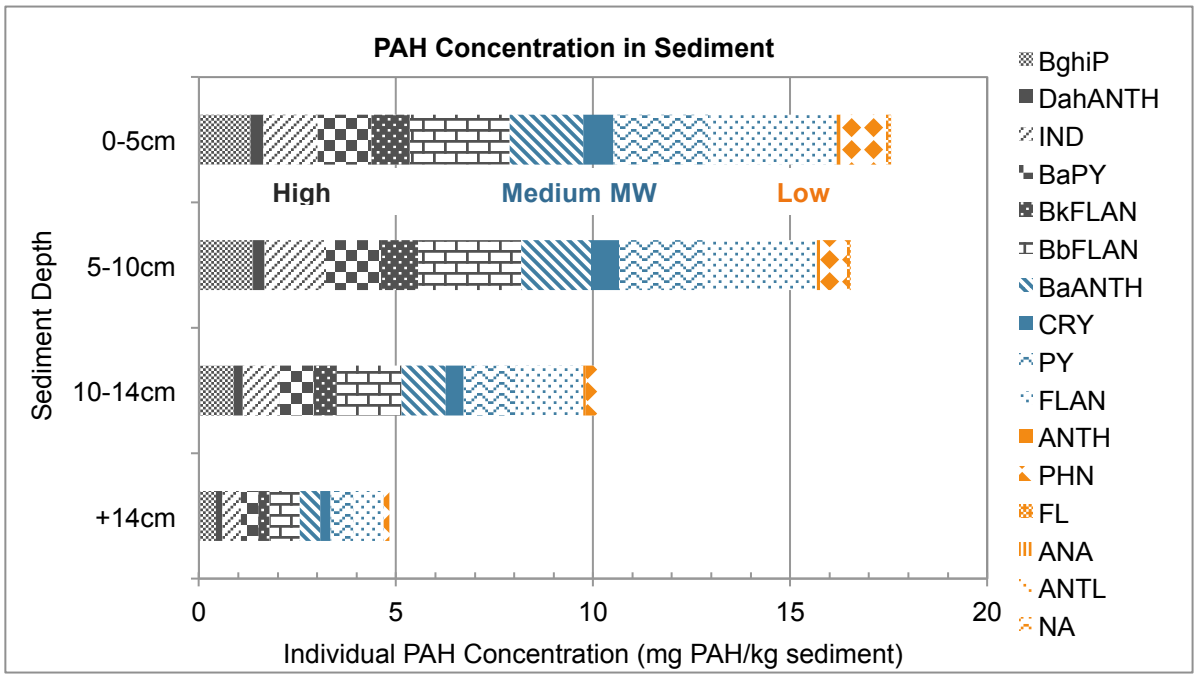

Figure 6: Background PAH concentrations in sediment (credit: Victoria Boyd, UIUC)

The concentration of PAHs in the background sediment decreased with increased sediment depth. This was expected since sediment age increases with depth; therefore, the PAHs in deeper sediment have had more time to degrade.

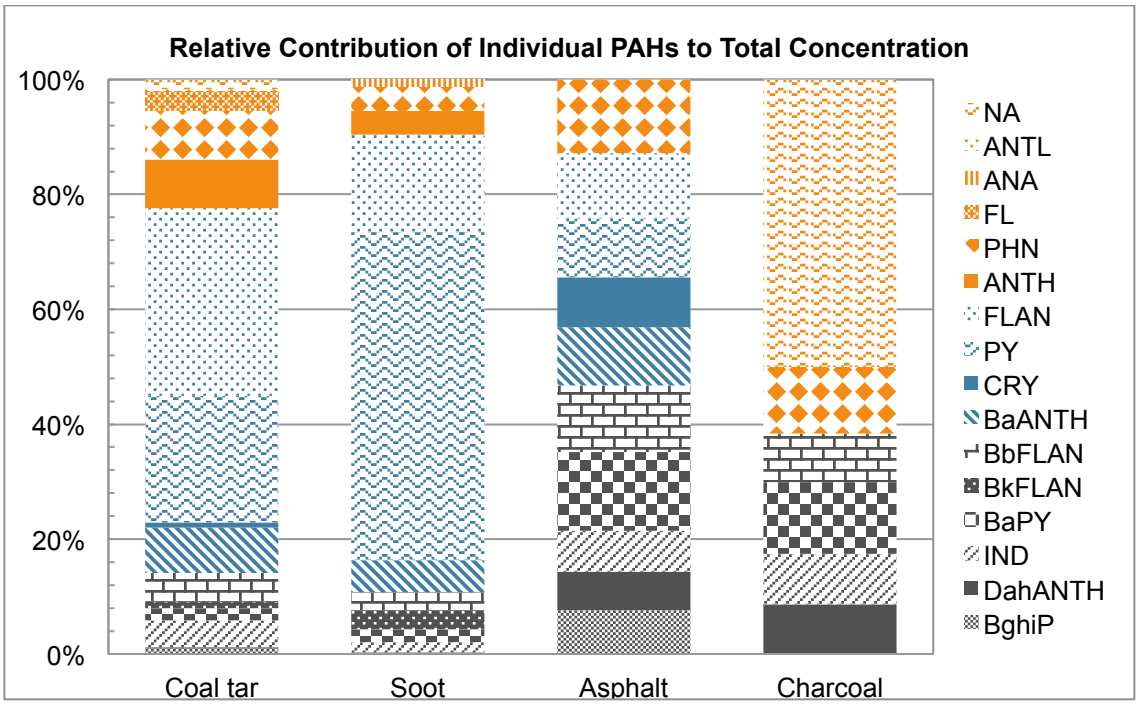

Figure 7: Relative abundance of individual PAHs in carbonaceous materials (credit: Victoria Boyd, UIUC)

Each CM has a unique PAH signature. Coal tar and soot are primarily made of medium molecular weight PAHs, specifically fluoranthene and pyrene. Conversely, charcoal has no detectable amount of medium molecular weight PAHs; it is mainly composed of naphthalene, the lightest PAH. Asphalt has the most equal distribution of PAHs. These trends are specific to the brands of CMs used in this study. For example, if a different brand of coal tar was selected the molecular weight distribution would be expected to change.

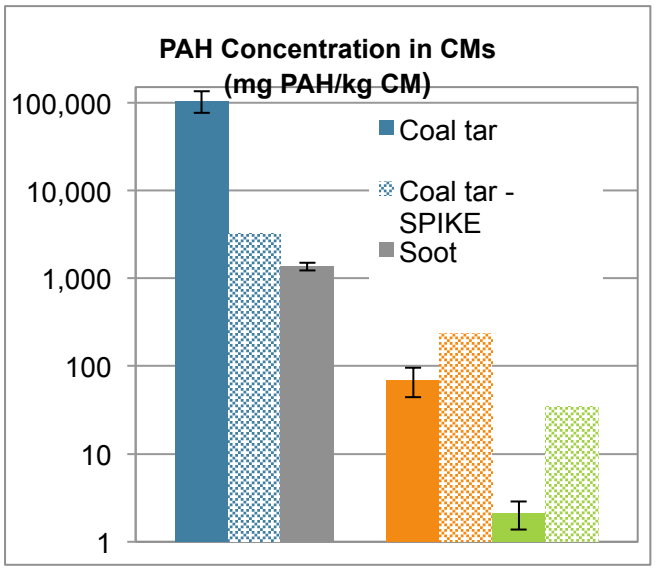

Figure 8: Background (solid bar) and spiked (patterned bar) PAH concentrations in carbonaceous materials (credit: Victoria Boyd, UIUC)

The background concentrations of PAHs in the CMs that were amended to the sediment are shown in the figure above in the solid bars. The PAH concentration in coal tar was two orders of magnitude greater than any of the other materials. The concentrations of the spiked PAHs are shown in patterned bars next to the background concentration. The spiked soot sample has not been analyzed yet, and replicates for the others are being run to confirm results.

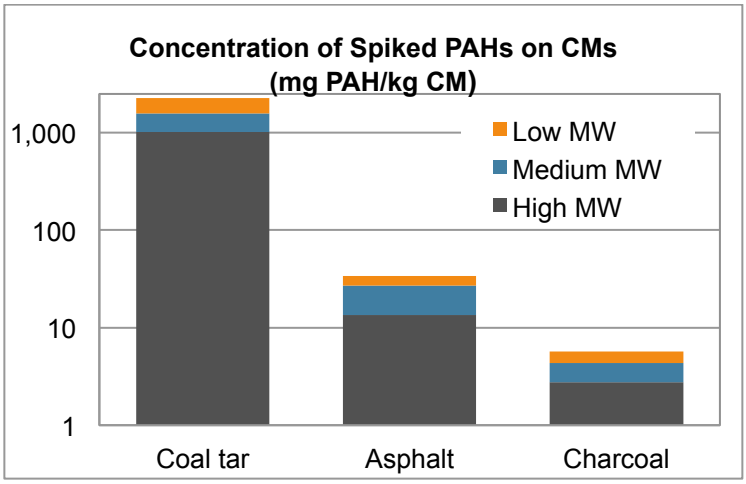

Figure 9: Individual concentrations of deuterated PAHs spiked onto carbonaceous materials (credit: Victoria Boyd, UIUC)

The figure above shows the concentration of the deuterated PAHs on each CM at the time of field deployment. Each CM was spiked with three unique deuterated PAHs of varying molecular weights. Approximately the same concentration of the three deuterated PAHs was introduced to a CM and allowed to equilibrate. For example, coal tar was spiked with similar concentrations of D10-acenaphthene, D12-benz[a]anthracene, and D12-benzo[b]fluoranthene. After a month of equilibration there was significant sorption of the high molecular weight spike (D12-benzo[b]fluoranthene) and lesser sorption of the other spikes. This trend was consistent across all CMs. This makes sense as the heavier PAHs are less soluble in water, leading to greater sorption onto the solid phase.

Analysis has started on in situ field samples. Preliminary results show the spiked PAHs can be detected in the sediment portion of the retrieved field samples (figure 10). Further analysis of sediment and CMs in all in situ field samples must be completed in order to draw any conclusions.

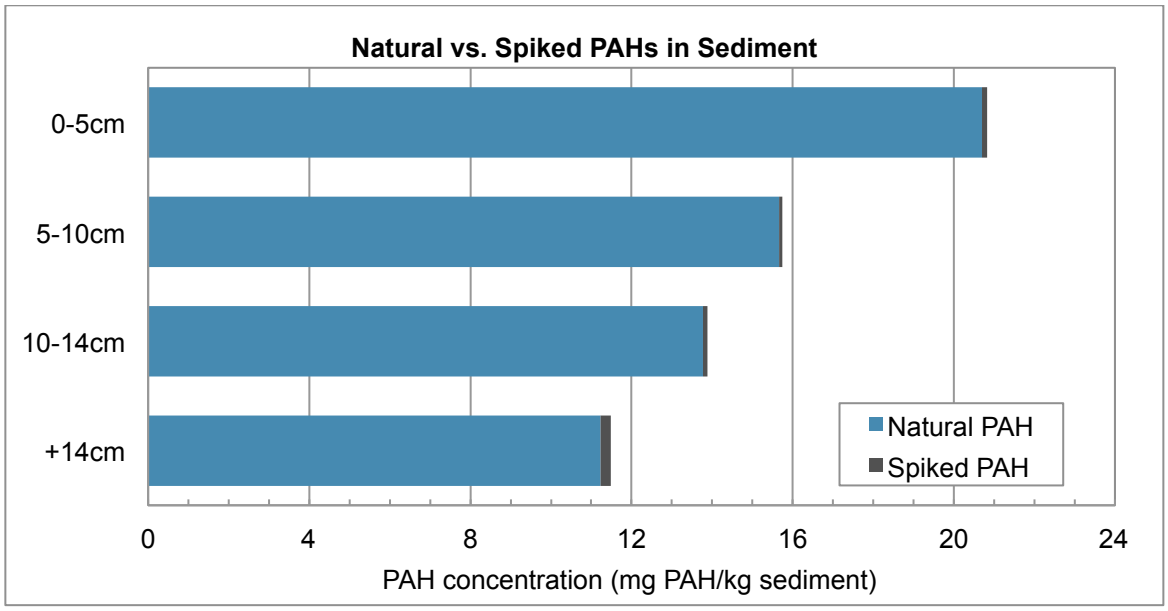

Figure 10: Natural and spiked PAH concentrations in in situ field sample (credit: Victoria Boyd, UIUC)

Concentration response curves are shown in Figure 11 for mutagenicity induced by benzo(α)pyrene and coal tar extract containing polyaromatic hydrocarbons in *S. typhimurium,* strain TA100, with and/or without mammalian hepatic microsomal activation. Benzo(a)pyrene with microsomal activation shows a linear increase in histidine revertants with increasing benzo(a)pyrene concentration. Coal tar extract with microsomal activation shows high numbers of histidine revertants as compared to coal tar extract without microsomal activation. Also, the response is nonlinear, showing a plateau at high loadings of extract.

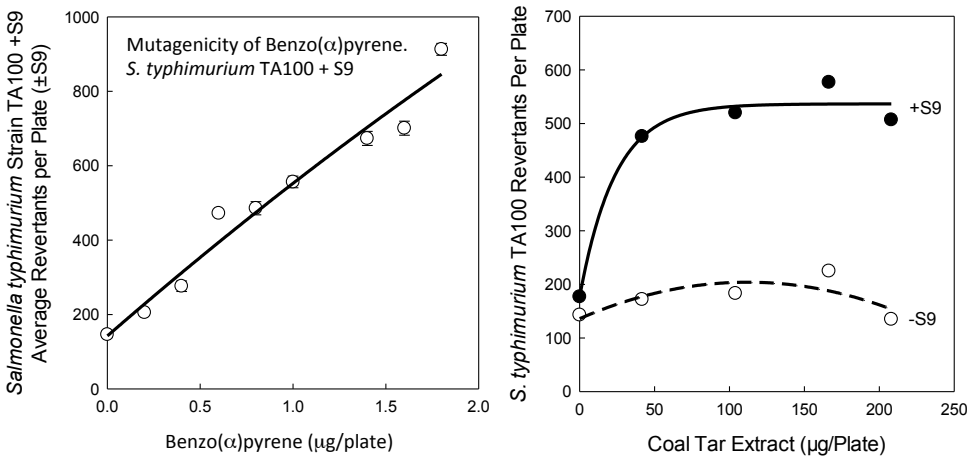

Figure 11. Mutagenic response of *S. typhimurium* to benzo(α)pyrene and a coal tar extract containing PAHs (credit: Azra Dad, UIUC).

Notable Achievements:

The analysis of background field samples and CMs has been completed. Two successful field sampling trips have resulted in the retrieval of the majority of in situ samples. The field sampling will conclude in June 2015 with the retrieval of the final two in situ samples. The successful detection of spiked PAHs in the in situ field samples and CMs was an important step in the project.

Students supported with funding:

In the summer of 2012 two graduate students were hired. Ms. Tory Boyd was hired to perform all work except toxicity testing. Ms. Boyd obtained her MS degree at Illinois, and the work in this proposal represents the bulk of her PhD thesis. The other graduate student is Ms. Azra Dad, who is performing the toxicity testing as part of her PhD thesis.

Publications and presentations:

This work was presented at the Environmental Engineering and Science Symposium at the University of Illinois on April 3, 2014 and at the Society of Environmental Toxicology and Chemistry in Vancouver, BC on November 10, 2014.

Photos and figures credited to Boyd, Dad, and Van Metre can be used in IWRC publications.

Hydro-meteorological response to tropical system precipitation in Illinois

Basic Information

Title:	Hydro-meteorological response to tropical system precipitation in Illinois
Project Number:	2014IL282B
Start Date:	5/1/2014
End Date:	12/31/2014
Funding Source:	104B
Congressional District:	16
Research Category:	Climate and Hydrologic Processes
Focus Category:	Climatological Processes, Floods, Drought
Descriptors:	None
Principal Investigators:	David Changnon

Publications

There are no publications.

U.S. Department of the Interior
United States Geological Survey Prime Award #G11AP20077
Illinois Water Resource Center, University of Illinois Subaward #2011-03502-03

Final Report

Hydro-meteorological Responses to Tropical System Precipitation in Illinois

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Hydro-meteorological Responses to Tropical System Precipitation in Illinois

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ABSTRACT

This study sought to characterize the impact of tropical system precipitation on streamflow values measured in nine Illinois watersheds and determine whether a predictive model based on antecedent and expected rainfall conditions could be developed. Unlike smaller-scale thunderstorm complexes which are difficult to forecast beyond 24 hours, there is much greater forecast lead time associated with tropical systems. The differences in spatial and temporal scales associated with two types of heavy rain producing storms suggests that hydro-meteorologists have a better opportunity to develop timely regional precipitation and streamflow forecasts as a tropical system approaches Illinois.

During a 100-year period (1913-2012) 26 tropical systems were found to have produced an average of an inch or more precipitation within a 24-hour window (i.e., event) in one or more of Illinois' nine climate divisions. Those climate divisions impacted by an event experienced a significant increase in monthly soil moisture levels as measured by the Palmer Drought Severity Index (PDSI). When pre- versus post-tropical system streamflow (ST) values were compared for the non-event watersheds an increase in ST of less than 50% occurred, while ST changes in watersheds that experienced an event increased by more than 500%. Factors that influenced the magnitude of increase included pre-storm ST conditions (i.e., was ST below, near or above average prior to the event), timing of event (i.e., summer or fall), and total storm precipitation.

The predictive ST model, where model output (e.g., post event ST estimates) could be integrated into decision support tools used by those impacted by flooding, was of limited use. Reasons for a lack of success were related to three primary issues, a small sample of events (i.e., 26) in the study, events occurring over a long period (100-years) when changes in land use and agricultural practices altered the surface hydrology (i.e., use of tiles, no-till agriculture, expanding crops in risky environments, etc.), and different surface characteristics (e.g., basin shape and size, soils, geomorphology, etc.) among the nine watersheds.

Despite the issues that add complexity to the rainfall to ST relationships over time, those who are tasked with forecasting tropical storm precipitation and related ST values have greater knowledge of how ST values increase and can provide more lead time to regional decision makers in affected watersheds.

Chapter 1: Introduction

Background

Heavy summer rainfall events in the Midwest are typically associated with mesoscale convective systems (MCS, Parker and Johnson 2000; e.g., Maddox et al. 1979; Fritsch et al. 1986; Junker et al. 1999; Anderson and Arrit 2001; Ashley et al. 2003; Tuttle and Davis 2006; Schumacher and Johnson 2005; Schumacher 2009). Various sub-types of MCS have been intensely studied, and their contribution to Midwest warm-season rainfall is well established (Maddox 1980; Rodgers et al. 1985; Fritsch et al. 1986; Parker and Johnson 2000; Ashley et al. 2003; Houze 2004; Ashley and Ashley 2008; Gallus et al. 2008; Kunkel et al. 2012). These types of systems can produce disastrous hydro-meteorological effects in Illinois that can lead to millions of dollars in flood-related damage (Angel et al. 1997) and even loss of life (Angel et al. 1997; Ashley and Ashley 2008). Although meteorological conditions dictate when, where and how long these heavy-rain producing systems will impact an area (Doswell et al. 1996), their impacts, either positive or negative, are largely determined by antecedent conditions (Changnon et al. 2003). For example, heavy MCS rainfall on areas experiencing drought could produce beneficial effects for crops, but the same rainfall on saturated soils and swollen streams could result in flooding (Changnon et al. 2003). One may wonder: *what other types of precipitation events could produce similar hydro-meteorological impacts in Illinois?*

Tropical systems have received little recognition as heavy rain producers in the Midwest, and studies have typically viewed these systems as interesting and rare case-studies (Changnon and Changnon 2004; Changnon and Changnon 2006; Gensini et al. 2011). However, Illinois has experienced heavy tropical system rainfall ($\geq$ 2.54 cm day^{-1}) once every 3.8 years from 1913-2012, and the period from 2005 to 2012 produced 7 such events (Haberlie et al. 2014). One such event, associated with the remnants of Hurricane Ike in 2008, produced once in 100 year rainfall and peak streamflow values in northern Illinois (Gensini et al. 2011). Tropical systems were found to produce heavy rainfall over a broader area (i.e., impacting on average three Illinois climate divisions) than MCS events. Furthermore, these tropical systems typically occur during neutral to wet conditions in Illinois. Tropical system events that occur under conditions where soils are wet and rivers are at or above average stream flow could further accentuate negative hydro-meteorological impacts in Illinois (Changnon et al. 2003). Moreover, recent studies have suggested that tropical systems in the Atlantic basin could become stronger and more numerous as anthropogenic warming continues (Emanuel 2005; Elsner et al. 2008; Bender et al. 2010), and some of those systems may increasingly impact the Midwest (Kunkel et al. 2012; Haberlie et al. 2014). Thus, there is reasonable motivation to understand the impacts of these events in Illinois, especially if their frequency increases. Building on the work of Haberlie et al. (2014), *this study will quantify the amount and impact of tropical system related precipitation in Illinois, with a focus on drought and stream flow response.*

Objectives of the Study

1) Produce information related to Illinois-impacting tropical systems that increases situational awareness for emergency managers, operational meteorologists, and agri-business interests

2) Develop a model that could assist interested parties in anticipating the impact of tropical systems in Illinois.

Chapter 2. Data and Approach

Tropical Systems impacting Illinois

Haberlie et al. (2014) identified heavy rainfall events (≥2.54 cm per day) associated with tropical systems that impacted Illinois from1913-2012 (Table 1). Climate divisions (CDs) that experienced a heavy tropical rainfall event, the dates the storm was in the region, the total number of climate divisions (CDs) impacted by each system, precipitation data, and monthly Palmer Drought Severity Index (PDSI) values before and after each event were used in this study. There were 26 tropical system events during the 100-year period that impacted one or more Illinois CDs (Table 1). On average a tropical system impacted 3.8 Illinois CDs or more than a third of the state.

Table 1. Tropical systems that impacted one or more Illinois climate divisions (CDs).

Year	Name	Regional Date(s)	IL CD(s)
1915	2	8/20-8/22	4-9
1916	14	10/19	2
1923	8	10/17	2, 4-9
1923	9	10/18-10/19	2, 5, 7, 9
1926	1	7/31-8/2	5
1932	3	9/2-9/4	9
1941	2	9/24-9/25	2-4, 6
1947	4	9/21	1
1948	5	9/5-9/6	8, 9
1949	10	10/6	1, 4, 5, 7
1950	Baker	9/1	7, 8
1955	5	8/29	3
1957	Audrey	6/28-6/29	5-7
1960	Ethel	9/17	5
1961	Carla	9/13-9/14	1-6
1965	Betsy	9/11-9/12	8, 9
1968	Candy	6/25-6/26	1-5, 7-9
1979	Bob	7/12-7/14	1
1979	Claudette	7/28-7/29	7, 8
2005	Arlene	6/12-6/14	9
2005	Dennis	7/11-7/17	8, 9
2005	Katrina	8/30-8/31	9
2005	Rita	9/25-9/26	7
2008	Gustav	9/4-0/5	2-6
2008	Ike	9/14-9/15	1-6
2012	Isaac	8/31-9/3	3-9

Long-term Illinois Watersheds selected for Study

In each of the Illinois' nine CDs a watershed with a long-term daily discharge record was identified and used in this study (Table 2, Figure 1). These watersheds varied in size and shape,

however, they were generally considered “virgin” watersheds with little or no impairments to the natural flow.

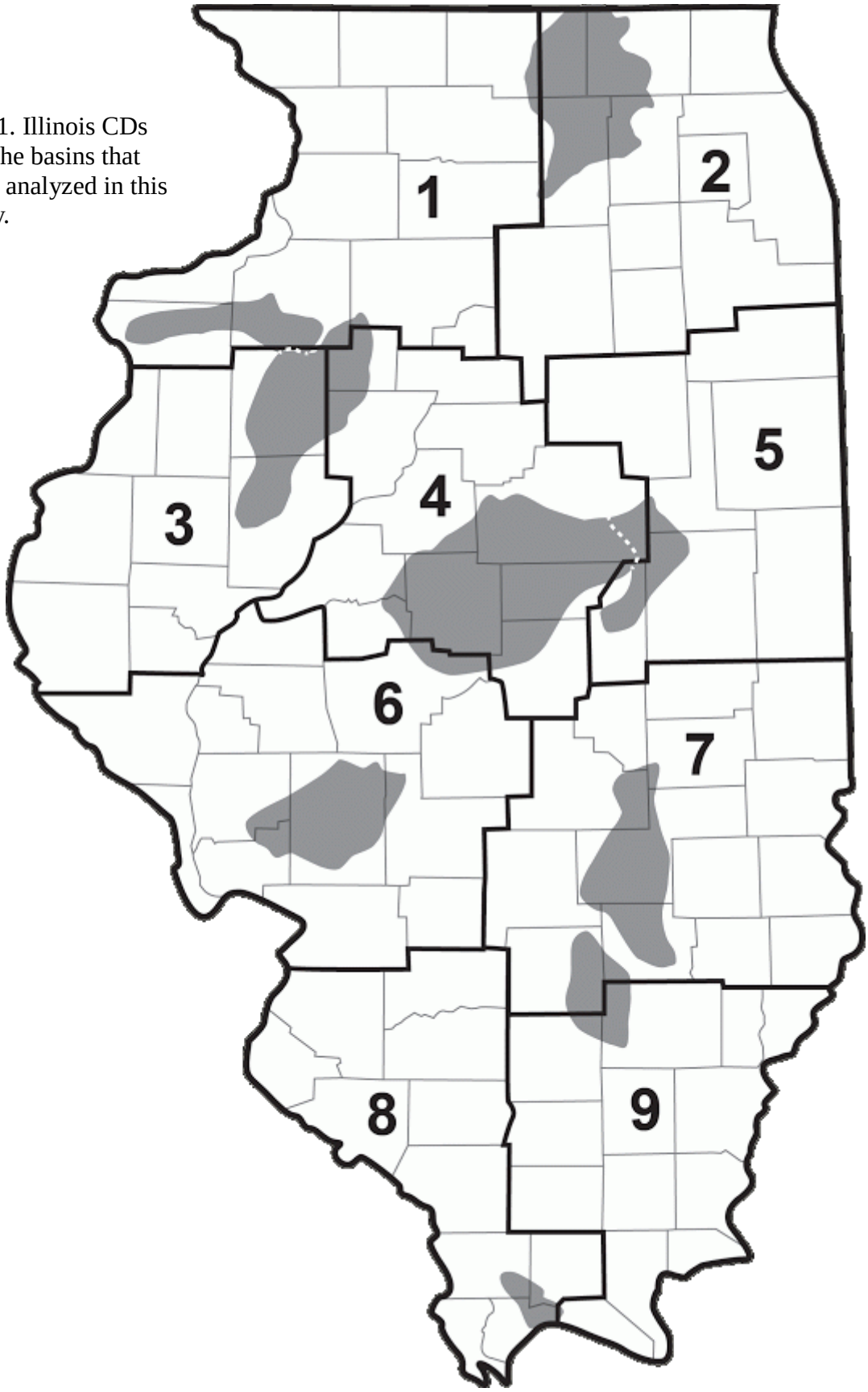

Fig. 1. Illinois CDs and the basins that were analyzed in this study.

Table 2. Characteristics of Illinois basins used in study.

Climate Division	Streamflow Gage Number	Gage Site Location	Basin Size (mi^2)	Period of Record
CD1	05466500	Edwards near New Boston	445	1934-present
CD2	05440000	Kishwaukee near Perryville	1,099	1939-present
CD3	05570000	Spoon near Seville	1,636	1914-present
CD4	05582000	Salt Cr. near Greenview	1,804	1941-present
CD5	05572000	Sangamon near Monticello	550	1914-present
CD6	05587000	Macoupin near Kane	868	1921-33;1940-present
CD7	03379500	Little Wabash near Clay City	1,131	1914-present
CD8	03612000	Cache near Forman	244	1922-present
CD9	03380500	Skillet Fork near Wayne City	464	1914-21;1928-present

Examining the Impact of Tropical System Precipitation on Streamflow

Discharge data in feet per second from the USGS-NSIP (2014) were gathered for the day before and the last day of each tropical system passage in Illinois. These values were then compared to assess the impact of tropical systems on streamflow (ST) in Illinois. Average differences in post-tropical storm ST minus pre-tropical storm ST values were compared for two groups of watersheds, those that were impacted by a tropical system event (CD that experienced >2.54 cm per day rainfall from a tropical system) and those that were not.

Identifying Factors Contributing to Changes in Streamflow

Further multi-regression analysis was performed to assess the importance of antecedent soil moisture conditions, storm total rainfall, maximum 24-hour rainfall, basin size, time of year, days the tropical system was over Illinois, and the mean amount of rain per day produced by these systems. These variables were used to ascertain whether a predictive model could be developed to forecast potential tropical system-related ST for the state and the selected watersheds.

Chapter 3. Results

Descriptive Results: Impact of Tropical Precipitation on Selected Watersheds

Streamflow characteristics associated with the 26 tropical system events were evaluated individually for each watershed and averaged across the state. Each of these tropical system events evolved differently but all tracked from the Gulf of Mexico toward the eastern Midwest (Haberlie et al. 2014). The most recent tropical system was associated with Hurricane Isaac in 2012. Figures 2 and 3 show the rain bands associated with the remnants of Isaac as the tropical system began to impact parts of Illinois. This tropical system spent approximately two days over the Midwest. Figure 4 shows how precipitation associated with Isaac impacted the selected Illinois watersheds, especially those in CD 3-9. Although most watersheds in the southern three-quarters of Illinois exhibit a large response to the heavy rainfall associated with Isaac's remnants, the ST response appears to vary significantly from one watershed to another. The percent change in ST values were very large primarily due to the drought conditions experienced at the time of tropical system rainfall.

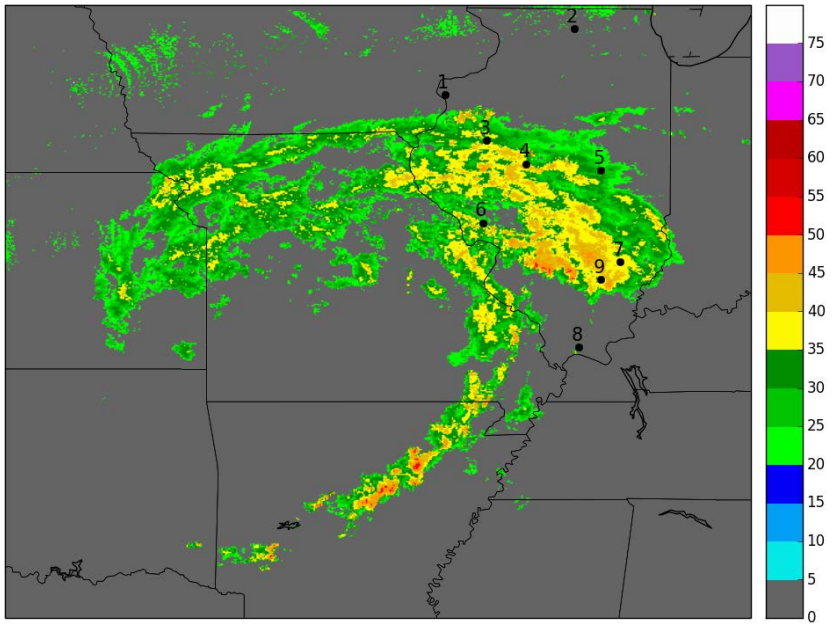

Figure 2. Radar image valid 1 Sep 2012 at 12:00 am CST.

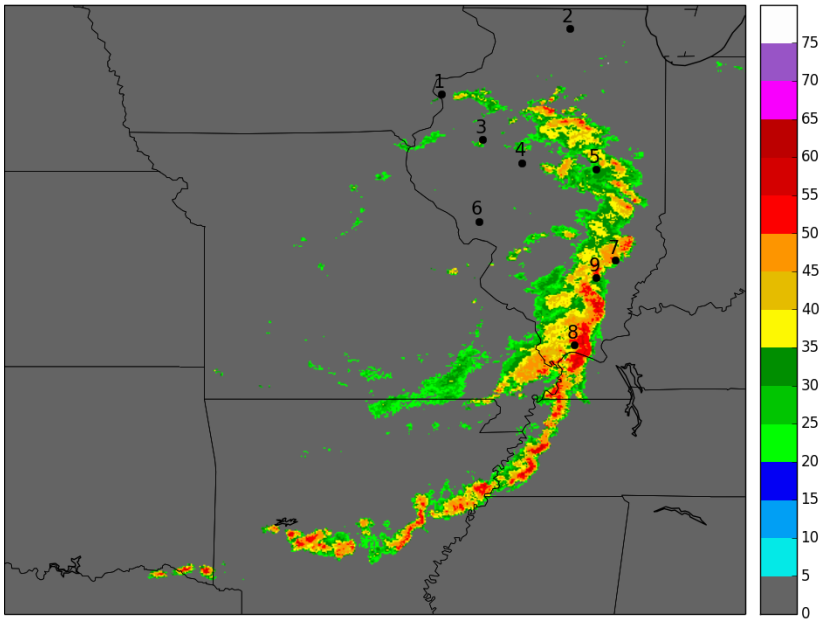

Figure 3. Radar image valid 1 Sep 2012 at 8:00 pm CST

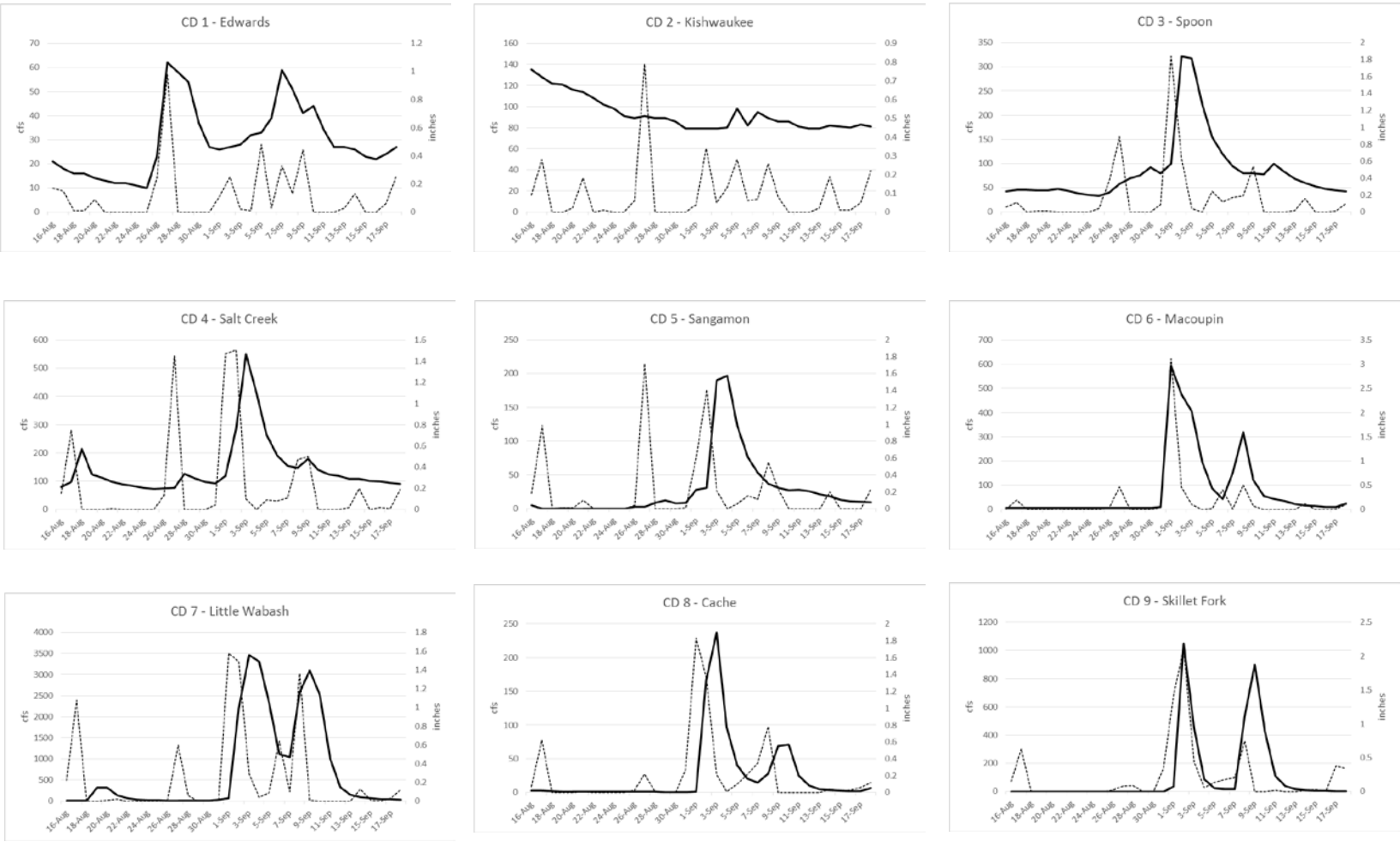

Figure 4. Streamflow (solid) and precipitation (dotted) for the two weeks before and after Hurricane Isaac traversed over Illinois in late August and early September of 2012. Second streamflow spike in CDs 6-9 was from synoptic frontal passage on 8 September 2012.

When examined at the watershed scale, the percent change in ST (post tropical system ST minus pre tropical system ST) depended greatly on whether the watershed was impacted by a tropical system event (>2.54 cm per day in that CD). For example, the Sangamon (CD5) experienced an average 38% increase in ST for non-events (i.e., CD did not experience >2.54 cm per day) with five cases when the ST actually decreased and nine cases with an increase (maximum increase in non-event was 164%), while during events the average ST increased 866% with increases ranging from 24% (1949 #10 event) to 2305% for Isaac.

The relationship of CD storm total rainfall to percent change in ST in each of the nine CDs for the 26 tropical storms was examined (Table 3). R-squared values ranged from 0.26 to 0.85 and were statistically significant (p-value <0.0051) for eight of nine watersheds (only the Macoupin River in CD 6 did not exhibit a statistically significant r-squared value). The relationships were higher in the northern half of the state.

Table 3. R-squared and p-values between CD storm total rainfall to percent change in streamflow for the nine Illinois CDs/Watersheds.

CD/Watershed	R-squared Value	p-value
1/Edwards	0.83	<0.0001
2/Kishwaukee	0.70	<0.0006
3/Spoon	0.70	<0.0001
4/Salt Cr.	0.85	<0.0001
5/Sangamon	0.72	<0.0001
6/Macoupin	0.26	0.244
7/Little Wabash	0.53	0.0051
8/Cache	0.64	0.001
9/Skillet Fork	0.63	0.0011

Hydro-climatic variables for each of the 26 tropical systems were examined (Table 4). Isaac's storm rainfall total (i.e., statewide average of 2.69 inches) was the highest and occurred during the most extreme drought conditions (i.e., statewide average PDSI value of -3.69), thus produced the most dramatic statewide change in streamflow (i.e., 50,483%) despite an actual increase of only 597 cfs (ninth highest value). Hurricane Carla produced the greatest estimated statewide rainfall rate of .13 in hr $^{-1}$ (Table 4). Since tropical systems typically spend at least a day over the Midwest, the statewide rainfall rates noted in Table 4 indicate that these systems produce large amounts of precipitation (>2.54 cm per day) over many hours. The average rainfall duration associated with tropical systems is longer than that for MCS events and contributes to very different hydrologic characteristics.

Table 4. State averages for each storm. *Hours in region based on 6 hour resolution of HURDAT database.

Storm	Preceding PDSI	Storm Total (in)	Hours in Corn Belt*	Inches Per Hour*	Percent Event CDs	CFS change	Percent CFS Change
Isaac	-3.69	2.69	24	0.11	78%	597	50483
1941-2	-2.86	1.05	24	0.04	38%	243	7661
1932-3	0.47	1.23	36	0.03	17%	530	6002
Betsy	1.14	0.82	30	0.03	22%	188	3167
1948-5	-0.41	0.66	30	0.02	22%	119	2635
Candy	0.31	2.41	24	0.10	89%	1531	2078
Carla	1.04	2.41	18	0.13	67%	1419	1944
1926-1	-0.20	1.01	42	0.02	20%	487	1844
Audrey	1.42	1.32	18	0.07	33%	2848	1794
1923-9	0.59	1.06	30	0.04	80%	1140	1342
1947-2	-0.82	0.75	6	0.12	11%	195	1096
1949-10	-0.38	0.98	18	0.05	44%	681	916
Gustav	2.02	1.95	30	0.07	56%	350	571
1915-2	2.34	1.71	54	0.03	75%	11899	523
Ike	2.02	2.43	24	0.10	67%	4235	496
Baker	2.14	0.45	12	0.04	22%	295	314
Dennis	-2.28	1.22	144	0.01	22%	18	291
Claudette	0.33	0.94	30	0.03	22%	1125	288
Arlene	-1.57	0.55	42	0.01	11%	93	266
1923-8	0.69	0.99	12	0.08	67%	106	171
Rita	-2.05	0.59	12	0.05	11%	18	116
1916-14	-0.94	0.59	12	0.05	0%	4.2	68
Ethel	0.35	0.24	12	0.02	11%	9	12
Bob	0.33	1.97	42	0.05	11%	-5	9
1955-5	-1.65	0.58	12	0.05	11%	2	-2.94
Katrina	-2.23	0.32	18	0.02	11%	-7	-5.8

Predictive Model

The relationships between hydro-climatic variables that were considered to influence ST (e.g., basin size, time of year of occurrence, soil moisture conditions as measured with PDSI, etc.) and the average pre-/post- change in ST for the 26 events were examined. The strongest relationship revealed by the statewide averages was between preceding PDSI and ST change, but only when preceding PDSI was positive (Figure 5; $r^2 = .31$, $p = .03$). This suggests that as statewide soil moisture increases, the average change in ST (associated with a tropical system) will linearly increase, but only in moist (positive PDSI) soil moisture conditions, regardless of rainfall amount.

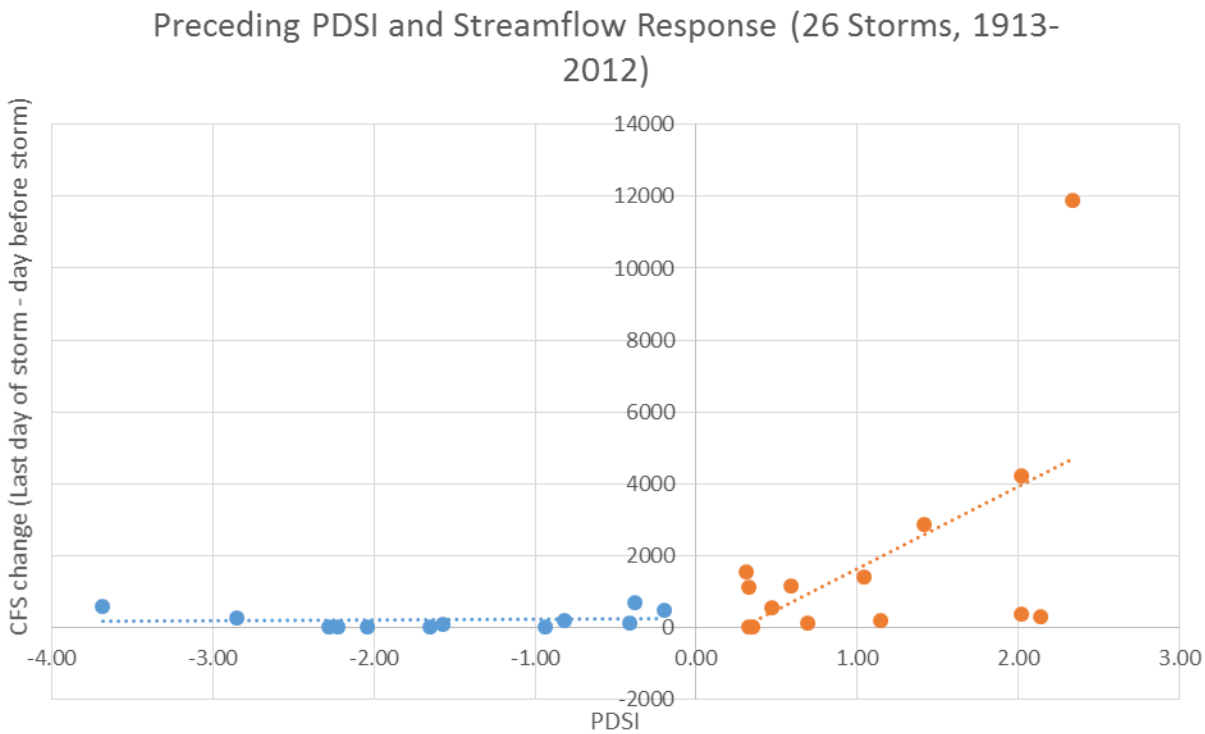

Figure 5. Preceding PDSI value to change in statewide change in streamflow for 26 tropical cyclones (1913-2012).

In developing a generalized model that could predict ST response to tropical system rainfall, we examined CD specific data for each storm. An unexpected result was the extreme responses to tropical system rainfall. Of specific interest was the data for CD 9 when Isaac traversed the state.

On August 31st, the daily average ST on the Skillet Fork at Wayne City (CD9) was at 0.12 cfs. Just two days later, rainfall from Isaac caused the Skillet Fork to swell to a daily average ST of 1050 cfs—a near-record daily ST value and a positive percentage change of 374,000% (USGS 2012). Isaac was associated with the largest statewide percent change in ST among the 26 tropical

systems (Table 4) and when examining the largest percent changes in daily average ST by CD four of the top 20 changes were associated with Isaac (Table 5). Upon further inspection of Table 5, other tropical systems are noted more than once suggesting that these extreme precipitation events impact broad areas of Illinois, not generally one CD. Furthermore, these results indicate that extreme, multi-day, heavy rainfall events associated with tropical systems can increase average daily ST significantly. This may be especially true in late summer when daily average ST in Illinois rivers can be at its lowest levels in the warm-season.

Table 5. Top 20 percent change magnitude increases in daily average streamflow associated with tropical systems.

Storm	CD	Basin Size	StormTotal	Max Day	Event	PriorPDSI	Day of Year	Days	Per Day	ChangeCFS	PercentChange
"Isaac"	9	464	4.44	2.11	1	-3.66	245	4	1.11	448.88	374066.67
"1941-2"	6	868	1.85	1.05	1	-2.77	267	2	0.93	1008.30	59311.76
"Isaac"	7	1131	3.41	1.58	1	-3.81	245	4	0.85	3451.20	39218.18
"Isaac"	8	244	3.72	1.83	1	-3.93	245	4	0.93	236.21	29900.00
"Betsy"	8	244	2.34	1.25	1	0.59	254	2	1.17	1474.00	24566.67
"1932-3"	9	464	2.95	1.52	1	0.55	247	3	0.98	288.70	22207.69
"1948-5"	8	244	1.76	1.14	1	0.70	249	2	0.88	572.50	16357.14
"1932-3"	7	1131	1.95	0.96	0	0.44	247	3	0.65	1767.00	13592.31
"Carla"	1	445	5.07	3.28	1	-0.39	256	2	2.54	2268.00	10309.09
"Candy"	8	244	1.06	1.05	1	0.67	177	2	0.53	438.60	9968.18
"Isaac"	6	868	3.73	3.11	1	-4.01	245	4	0.93	401.10	8185.71
"Audrey"	9	464	0.99	0.93	0	2.03	179	2	0.50	2024.00	7784.62
"1948-5"	9	464	2.15	1.30	1	0.78	249	2	1.08	564.40	7426.32
"1926-1"	7	1131	1.05	0.47	0	-0.58	213	3	0.35	373.00	6216.67
"Audrey"	6	868	2.66	2.65	1	1.96	179	2	1.33	16790.00	5416.13
"1947-2"	1	445	1.12	1.12	1	-0.89	264	1	1.12	949.00	4745.00
"Candy"	9	464	1.32	1.22	1	0.53	177	2	0.66	391.50	4121.05
"1947-2"	6	868	0.71	0.71	0	-1.77	264	1	0.71	277.20	4076.47
"1923-9"	7	1131	1.54	1.48	1	0.50	291	2	0.77	666.00	3700.00
"Betsy"	9	464	3.40	1.77	1	0.32	254	2	1.70	835.00	3630.43

Table 6 shows the correlation between values from every climate division. Per Day, Storm Total, and Max Day are all strongly correlated. Thus, the final regression variables, chosen for their non-collinearity, are Storm Total (total rainfall per event per CD), prior PDSI (PDSI monthly value before the event), Day of Year (Days since January 1st), and Days (Count of days tropical system was in the region).

Table 6. Correlation matrix comparing possible regression variables.

	PercentChange	StormTotal	Max Day	PriorPDSI	Day of Year	Days	Per Day
PercentChange	1.0000	0.2849	0.1685	-0.1398	0.0267	0.1139	0.0867
StormTotal	0.2849	1.0000	0.9368	0.1343	-0.0321	0.1873	0.8359
Max Day	0.1685	0.9368	1.0000	0.2124	-0.0265	0.0460	0.8844
PriorPDSI	-0.1398	0.1343	0.2124	1.0000	-0.0030	-0.2867	0.2979
Day of Year	0.0267	-0.0321	-0.0265	-0.0030	1.0000	-0.4628	0.1602
Days	0.1139	0.1873	0.0460	-0.2867	-0.4628	1.0000	-0.2173
Per Day	0.0867	0.8359	0.8844	0.2979	0.1602	-0.2173	1.0000

The generalized statewide regression model performed poorly ($r^2 = 0.11$) when predicting percent change in ST. This is likely caused by a heterogeneous response to tropical system rainfall in Illinois due to differences in soil types, geomorphology, site to site base flow, and year to year base flow and the fact that no one tropical system impacted all 9 CDs. We then addressed these issues by examining ST response in each basin separately (Table 7).

Table 7. Regression and coefficient results for each climate division.

	Storm Total		Prior PDSI		Day of Year		Days		Overall	
CD	p-value	parameter	p-value	parameter	p-value	parameter	p-value	parameter	p-value	r2
1	< 0.001	411.43	0.4	-29.1375	0.56	1.24	0.73	-19.4	< 0.001	0.79
2	0.002	494.4	0.11	129.45	0.56	2.81	0.56	72.49	0.007	0.58
3	< 0.001	1667.19	0.24	217.83	0.24	-13.25	0.51	-215.02	< 0.001	0.62
4	0.06	994.18	0.27	359.57	0.83	3.86	0.93	44.36	0.09	0.41
5	0.007	690.58	0.21	154.74	0.9	-0.72	0.78	-51.96	0.003	0.52
6	0.003	1969.85	0.19	417.76	0.11	-33.61	0.18	-806.03	0.007	0.53
7	0.07	3127.73	0.49	448.52	0.73	13.76	0.83	258.56	0.21	0.24
8	0.1	192.77	0.08	84.54	0.18	4.12	0.92	-10.59	0.1	0.34
9	0.62	348	0.12	532.6	0.85	4.25	0.64	327.53	0.57	0.14

Regression for individual CDs/watersheds explained more variability than the statewide model with the prediction model explaining 50 percent or more of the variance in five of the 9 CDs (Table 7). However, performance decreased with the southernmost CDs. Only Storm Total significantly explained percent change in ST, with values ranging from a 411% to 1970% increase in ST per inch or rain associated with the tropical systems.

Chapter 4. Conclusions, Implications, and Recommendations

Summary of Motivation and Exploratory Results

This research project examined the influence of tropical system precipitation on ST response in nine watersheds located within Illinois' nine climate divisions (CDs). A statistically significant percent change in streamflow was found for those watersheds experiencing a tropical system event (i.e., CD experiencing >2.54 cm per 24-hour period) versus those that don't. When the monthly PDSI value prior to the tropical system was average or above (i.e., generally wet soil moisture conditions), there was a linear relationship between pre-event PDSI and actual change in ST. In other words, the wetter the soil moisture conditions were prior to the event the larger the ST response. When soil moisture conditions were dry (negative PDSI values) the actual change in streamflow levels was small. This descriptive information may be useful to those faced with preparing for a relatively infrequent tropical system heavy precipitation event.

Summary or Implications for Forecasters and Decision Makers

Statewide average ST response to tropical systems is poorly predicted by surface-based observations alone. The best predictor of percent change in ST in each CD was storm total precipitation amount. This relationship had R-squared values that exceeded 0.52 in all but CD6 (Table 3). Furthermore, when all the variables were evaluated in a multi-regression model more than 50 percent of the variance was explained in CD 1-3 and 5 and 6. Although regression analysis

did not find a clear predictive tool for hydro-meteorologists tasked with forecasting post storm ST, four factors enhance the opportunity to improve short-term forecasts in Illinois including 1) greater forecast warning time (i.e., typically one to three days for tropical systems versus two to six hours for MCSs), 2) heavy precipitation and related high ST values will impact a larger region (i.e., tropical systems impact 3.8 CDs on average where MCSs impact less area), 3) under wet soil moisture conditions, the actual increase in ST is much greater, and 4) the more total precipitation expected from the tropical system, the larger the percent change in ST.

Limitations of Results

The lack of stronger predictive relationships is attributed to a number of factors. Importantly, these watersheds exist in parts of Illinois with very different surface soils and geomorphologies. Differences in basin characteristics such as basin size and shape impact the ST results. Finally, a small sample size (i.e., only 26 tropical storms in 100 years), changes in land cover (e.g., urbanization, farming in floodplains, etc.) and changing agricultural practices (e.g., types of row crops planted, post-harvest tilling of the surface, frequent updates to underground tiling systems, etc.) limit the ability to find commonality among these watersheds. Despite these limitations it appears that results from this study can provide additional information to those involved with predicting hydro-meteorological responses from tropical systems.

Recommendations for Future Studies

A more detailed study of these watersheds, their changing characteristics and man's influence on ST response to precipitation events such as tropical systems would potentially yield a better predictive model. A comparison of ST response to MCSs (i.e., shorter duration heavy precipitation events) versus tropical system events might shed further light on differences in hydrographs associated with the two types of events. This knowledge could then be integrated more fully into National Weather Service hydrologist forecasts, especially when tropical system events are expected one to three days in advance. Finally, we suggest that other watersheds in the eastern Midwest be assessed to expand the tropical system event sample size.

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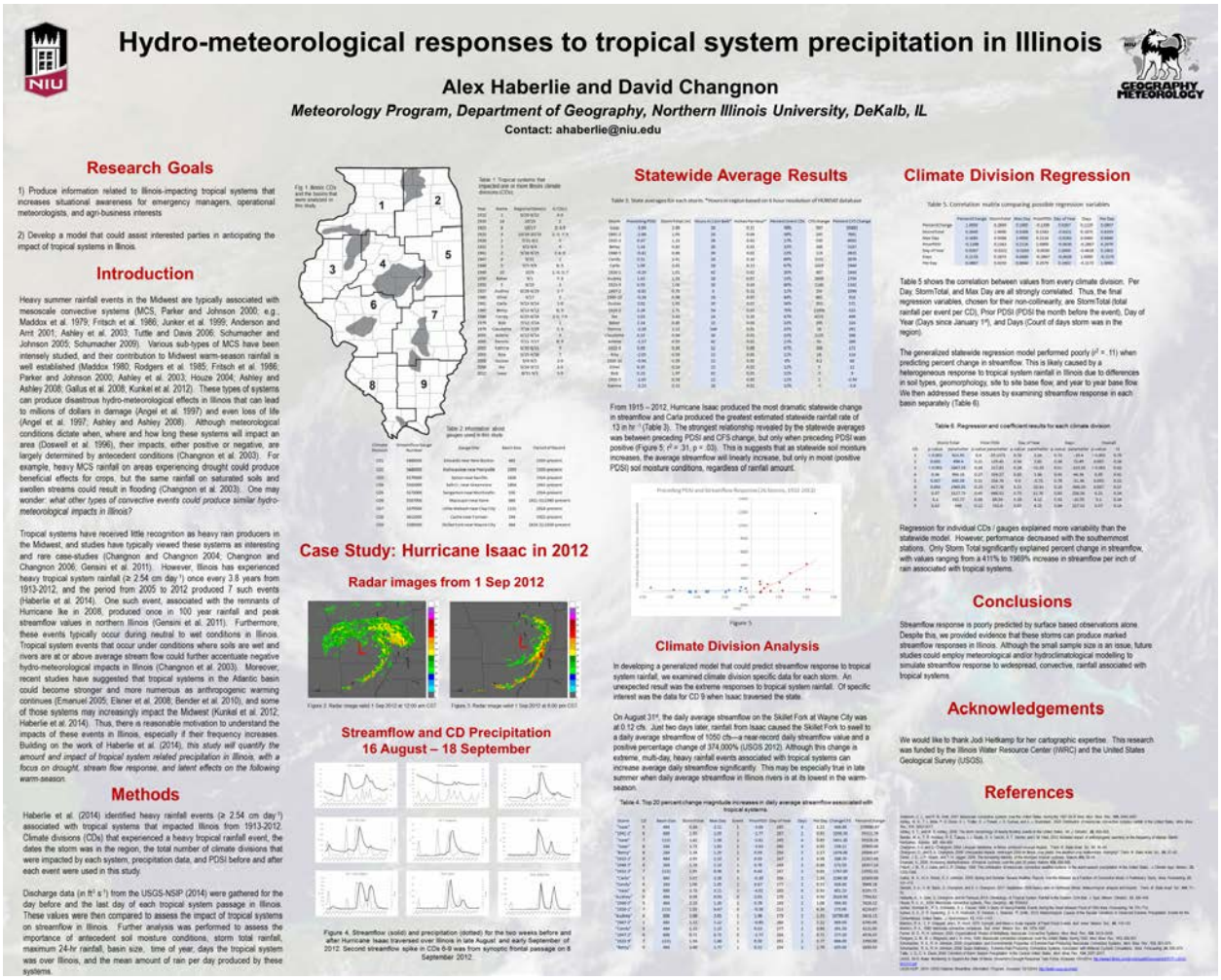
Appendices



Hydro-meteorological responses to tropical system precipitation in Illinois

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Research Goals

1) Produce information related to Illinois-impacting tropical systems that increases situational awareness for emergency managers, operational meteorologists, and agri-business interests

2) Develop a model that could assist interested parties in anticipating the impact of tropical systems in Illinois.

Introduction

Heavy summer rainfall events in the Midwest are typically associated with mesoscale convective systems (MCS, Parker and Johnson 2000; e.g., Maddox et al. 1979; Fritsch et al. 1986; Junker et al. 1999; Anderson and Arritt 2001; Ashley et al. 2003; Tuttle and Davis 2006; Schumacher and Johnson 2005; Schumacher 2009). Various sub-types of MCS have been intensely studied, and their contribution to Midwest warm-season rainfall is well established (Maddox 1980; Rodgers et al. 1985; Fritsch et al. 1986; Parker and Johnson 2000; Ashley et al. 2003; Houze 2004; Ashley and Ashley 2008; Gallus et al. 2008; Kunkel et al. 2012). These types of systems can produce disastrous hydro-meteorological effects in Illinois that can lead to millions of dollars in damage (Angel et al. 1997) and even loss of life (Angel et al. 1997; Ashley and Ashley 2008). Although meteorological conditions dictate when, where and how long these systems will impact an area (Doswell et al. 1996), their impacts, either positive or negative, are largely determined by antecedent conditions (Changnon et al. 2003). For example, heavy MCS rainfall on areas experiencing drought could produce beneficial effects for crops, but the same rainfall on saturated soils and swollen streams could result in flooding (Changnon et al. 2003). One may wonder: *what other types of convective events could produce similar hydro-meteorological impacts in Illinois?*

Tropical systems have received little recognition as heavy rain producers in the Midwest, and studies have typically viewed these systems as interesting and rare case-studies (Changnon and Changnon 2004; Changnon and Changnon 2006; Gensini et al. 2011). However, Illinois has experienced heavy tropical system rainfall (≥ 2.54 cm day^{-1}) once every 3.8 years from 1913-2012, and the period from 2005 to 2012 produced 7 such events (Haberlie et al. 2014). One such event, associated with the remnants of Hurricane Ike in 2008, produced once in 100 year rainfall and peak streamflow values in northern Illinois (Gensini et al. 2011). Furthermore, these events typically occur during neutral to wet conditions in Illinois. Tropical system events that occur under conditions where soils are wet and rivers are at or above average stream flow could further accentuate negative hydro-meteorological impacts in Illinois (Changnon et al. 2003). Moreover, recent studies have suggested that tropical systems in the Atlantic basin could become stronger and more numerous as anthropogenic warming continues (Emanuel 2005; Elsner et al. 2008; Bender et al. 2010), and some of those systems may increasingly impact the Midwest (Kunkel et al. 2012; Haberlie et al. 2014). Thus, there a reasonable motivation to understand the impacts of these events in Illinois, especially if their frequency increases. Building on the work of Haberlie et al. (2014), this *study will quantify the amount and impact of tropical system related precipitation in Illinois, with a focus on drought, stream flow response, and latent* effects on the following warm-season.

Methods

Haberlie et al. (2014) identified heavy rainfall events (≥ 2.54 cm day^{-1}) associated with tropical systems that impacted Illinois from 1913-2012. Climate divisions (CDs) that experienced a heavy tropical rainfall event, the dates the storm was in the region, the total number of climate divisions that were impacted by each system, precipitation data, and PDSI before and after each event were used in this study.

Discharge data (in ft^3 s^{-1}) from the USGS-NSIP (2014) were gathered for the day before and the last day of each tropical system passage in Illinois. These values were then compared to assess the impact of tropical systems on streamflow in Illinois. Further analysis was performed to assess the importance of antecedent soil moisture conditions, storm total rainfall, maximum 24-hr rainfall, basin size, time of year, days the tropical system was over Illinois, and the mean amount of rain per day produced by these systems.

Case Study: Hurricane Isaac in 2012

Radar images from 1 Sep 2012

Streamflow and CD Precipitation 16 August – 18 September

Statewide Average Results

Climate Division Analysis

Climate Division Regression

Conclusions

Acknowledgements

References

Year-round wetland microbial activity impacts on nitrogen cycling annual budgets: is restoration impacting greenhouse gas emissions in wetlands?

Basic Information

Title:	Year-round wetland microbial activity impacts on nitrogen cycling annual budgets: is restoration impacting greenhouse gas emissions in wetlands?
Project Number:	2014IL283B
Start Date:	5/1/2014
End Date:	12/31/2015
Funding Source:	104B
Congressional District:	IL-013
Research Category:	Water Quality
Focus Category:	Water Quality, Wetlands, Nutrients
Descriptors:	None
Principal Investigators:	Angela Kent

Publications

There are no publications.

The Effect of Agricultural Management Regimes on the Soil Microbial Denitrification Community Structure and its Impact on Nitrous Oxide Emissions

Summary:

Production agriculture to provide food for an ever-growing world population has resulted in the large scale use of inorganic fertilizers (Erisman, Sutton et al. 2008). The resulting increase of nitrogen in soil has far-reaching consequences (Canfield, Glazer et al. 2010). Nitrogen availability is regulated by microbial transformations in the nitrogen cycle, and agricultural inputs can disrupt the natural cycle (Galloway, Dentener et al. 2004). Important microbial transformations of the nitrogen cycle are nitrification and denitrification. Nitrification converts ammonium to nitrate, which presents pollution risks for groundwater and aquatic ecosystems. Denitrification can remove reactive N from terrestrial ecosystems through conversion of nitrate to molecular nitrogen via a stepwise process ($NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2$) but can also produce nitric oxide/nitrous oxide during incomplete denitrification (Zumft 1997). Nitrous oxide (N_2O) is an extremely potent greenhouse gas (Canfield, Glazer et al. 2010). A range of organisms are capable of denitrification including archaea, bacteria and fungi. The efficiency of denitrification (and production of GHG through incomplete denitrification) is influenced by the composition of the microbial community. Denitrification in soil also depends on oxygen availability, available carbon, redox conditions, pH, nitrate, and temperature, but most importantly on soil moisture, which plays a determining role in the other factors (Butterbach-Bahl, Baggs et al. 2013). The presence of more carbon and nitrogen substrates increases the rates of denitrification, which in turn increases the rate of incomplete denitrification and its byproduct of nitrous oxide. These factors can also influence the composition of soil microbial communities. These drivers of microbial communities and their denitrification processes are influenced by land use and management practices, thus investigating how microbial community structure and function differ among types of land use and also their response to specific ecological drivers can lead to better management of nitrous oxide emissions in intensively managed agricultural landscapes.

Methodology:

Soil samples will be obtained from fields that are managed with the following practices: chisel plow and ridge till, with and without cover cropping. We will also obtain soil samples from agricultural wetlands that treat agricultural runoff, and from denitrifying bioreactors that treat nitrate-laden tile drain effluent. The agricultural plots are located at the University of Illinois South Farm. The plots to be used are planted in a maize/soybean rotation and are fertilized with a surface liquid urea ammonium nitrate (UAN) broadcast application just prior to planting. The two denitrifying bioreactors planned for this study are also located at the University of Illinois South Farm. They consist of lined trenches filled with wood chips and receive effluent from a tile drain system from the study field. PVC sampling ports allow access for collecting woodchips from the depths of the bioreactor. The wetland sites are located on the Franklin Research and Demonstration Farm in McLean County, and these wetlands receive tile drain effluent for treatment. We will carry out sample collection over six sampling dates per management regime per year; the dates will be split between times expected to have relatively high N_2O emissions, such as following fertilizer application or incorporation of cover crop residues or during periods of high flow for the bioreactors, as well as during periods expected to have relatively low N_2O

emissions. A standard suite of soil chemical and physical parameters will be measured at the time of each sampling, including soil moisture, pH, SOM, total N and total C, and ammonium and nitrate concentrations.

Standard denitrification studies of soil or woodchip samples will be modified using several known methods to distinguish between contributions of bacteria and fungi. A modified acetylene inhibition method allows the measurement of the nitrous oxide produced by specific blockage of the last step of denitrification (Tiedje, Simkins et al. 1989, Royer, Tank et al. 2004). N_2O concentrations will be determined by use of gas chromatography using an electron capture detector.

A variety of microbial functional groups are involved in nitrogen cycle activities including archaeal and bacterial and fungal denitrifiers, and nitrous oxide consuming bacteria which are not denitrifiers (Butterbach-Bahl, Baggs et al. 2013). We can assess the abundance of these groups using quantitative PCR (qPCR) of functional genes specific to them. We will use Illumina MiSeq amplicon sequencing of diagnostic functional genes to compare community composition of all abundant functional groups across management practices.

We will relate specific microbial populations or specific microbial assemblages to production of nitrous oxide by multivariate analysis of the sequencing and qPCR data. The analysis will also identify ecological drivers or management practices that influence microbial community structure and function.

Objectives and Expected Results

Objectives

The methods described above will allow us to analyze the soil and woodchip samples from a variety of agricultural management practices (and runoff mitigation strategies) for denitrification potential and nitrous oxide emissions. The molecular microbial ecology methods and ecological analyses will produce detailed information about size and composition of the microbial population present in those samples, and the ecological drivers that shape the communities and their functions. Our 4 major objectives for this work are listed below:

Objective 1: Measure N_2O emissions for different agricultural and nutrient management practices (ridge till and chisel plow with and without cover cropping; wetlands; and denitrifying bioreactors) for at least 6 sample dates over the course of the season.

Objective 2: Characterize the microbial community associated with production and consumption of N_2O in each management regime using functional gene markers for nitrifiers, denitrifiers, and N_2O-consuming microorganisms (including fungi). Characterize seasonal variations in the microbial community.

Objective 3: Determine the potential denitrification and incomplete denitrification rates for denitrifiers in soil or bioreactor woodchip samples from each management regime, and also determine the contributions of bacterial and fungal denitrifiers.

Objective 4: Determine the management factors and specific ecological drivers that have the

greatest influence on microbial community composition, denitrification, and N2O production in each management practice.

Expected Results

- We will be able to compare denitrifying activity in agricultural production to that of wetlands and bioreactors.
- Fungal denitrifiers will be important in wetlands and bioreactors since they are more abundant in less physically disturbed environments.
- The expectation for the nitrous oxide emissions is that soil in the agricultural fields will have a higher output than wetlands or bioreactors due to fertilization. Among agricultural management practices fertilized fields should have higher nitrous oxide emissions due to increased available nutrients.
- Seasonal patterns will be evident when soils are flooded and more denitrification can take place in the anoxic conditions. However, seasonal trends in agricultural fields will correspond more highly to nutrient inputs. It has also been noted that fungal populations increase over time following planting which may mean N_2O production will increase correspondingly.
- Understanding the abundance of and diversity of the microbes responsible for the production and consumption of nitrous oxide will be important in determining the best management regime.
- High nitrous oxide production will be correlated to low abundance of *nosZ*.

Conclusions

The relationship between the soil microbial community, agricultural nitrogen inputs, and management practices to the reactive nitrogen pollution produced from farming needs to be addressed. While some strategies have been developed to mitigate the runoff of nitrate into the aquatic environment (for example, denitrifying bioreactors and agricultural wetlands), their potential to generate nitrous oxide is a concern which needs to be more fully investigated. We propose that studying the microbial community composition in soil samples from a variety of agricultural management regimes along with analysis of denitrification potential and N_2O emissions will help us determine meaningful relationships of the community to GHG emissions. Combining these findings with results from other expert teams at the University of Illinois will allow an overarching model to be developed that can identify management practices for farmers which minimize soil, air, and water reactive nitrogen pollution while retaining good crop yields and revenue potential. The synergy from a multidisciplinary analysis of the problem by groups that have not traditionally worked together should advance our understanding of how to abate pollution caused by reactive nitrogen from agriculture.

Participants

The work will be carried out by graduate student Natalie Stevenson under the supervision of Professor Angela Kent. Two undergraduate students will participate in this project in Summer 2015.

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Changing food webs in Lake Michigan: Dreissena and the microbial loop

Basic Information

Title:	Changing food webs in Lake Michigan: Dreissena and the microbial loop
Project Number:	2014IL284B
Start Date:	5/1/2014
End Date:	12/31/2014
Funding Source:	104B
Congressional District:	IL-013
Research Category:	Biological Sciences
Focus Category:	Ecology, Invasive Species, Surface Water
Descriptors:	None
Principal Investigators:	Jerry H Kavouras, Jerry H Kavouras

Publications

There are no publications.

Changing Food Webs in Lake Michigan: *Dreissena* and the Microbial Loop

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SUMMARY OF PROBLEM AND RESEARCH OBJECTIVES

The zebra mussel (*Dreissena polymorpha*) and quagga mussel (*Dreissena bugensis*) are changing the Great Lakes ecosystem and surrounding fresh waters. They alter ecosystems through their voracious filter feeding ability that removes algae, bacteria, and protozoa from the water column which disrupts the food chain for native organisms and increases the clarity of the water, allowing photosynthesis to occur in deeper waters expanding the habitat of submergent aquatic macrophytes. The microbial loop is an important part of the aquatic food web that feeds organisms in the second trophic level. Dissolved organic matter (DOM), which is released into aquatic ecosystems by phytoplankton, zooplankton, and bacteria, is utilized in bacterial growth and reproduction. A reduction in DOM and/or secondary production would have serious consequences on aquatic food webs, which would be observed at all trophic levels. Previous studies have demonstrated that DOM can provide 10-25% of the zebra mussel's maintenance ration, which indicates the uptake of DOM may be metabolically significant to them. With the increasing number of *Dreissena* in our fresh waters, this is a serious threat to the microbial loop and secondary production. Potentially, *Dreissena* can change the species diversity of the microbial loop and/or the levels of secondary production. Changes in secondary production can result in a cascade effect observed in the other trophic levels, which can challenge the viability of macroscopic life in fresh waters.

The testable hypothesis is that filter-feeding activities of *Dreissena* modify the quantity and quality of dissolved organic matter in the water, which in turn influences the amount of bacterial secondary production and possibly microbial diversity in fresh waters.

- The *first objective* is to determine if filter feeding will change the quantity of carbon substrates in the environment.
- The *second objective* is to determine if filter feeding modifies the quality of carbon substrates in the environment.
- The *third objective* is to determine if carbon substrates modified by the mussels can be utilized by fresh water bacteria isolated from Illinois water ways.

METHODOLOGY

The utilization and/or modification of dissolved organic matter by *Dreissena* will be studied using a defined medium. Adult quagga and zebra mussels will be collected from field sites and washed to reduce the number of microbes attached to their shells. Individual mussels will be placed into a test tube with a defined medium that includes

one carbon substrate. Changes in substrate concentration of these proteins will be determined using the Bradford assay. Qualitative changes, e.g., proteins to peptides and amino acids, will be determined using polyacrylamide gel electrophoresis to visualize and compare carbon substrate profiles before and after processing by the mussels in the assay. The processed substrates will be filtered and purified, if necessary, before examination on polyacrylamide gels.

Pure cultures of heterotrophic bacteria will be isolated from Lake Michigan waters that will be studied and identified using the BIOLOG microplate assay according to manufacturer's instructions. A portion of the defined medium that was metabolized by the mussels in the test tube assays will be filtered to remove any microbes and then used to prepare minimal growth media. Growth media with unmodified and modified carbon substrates will be inoculated with the bacterial isolates. The growth rates in media with modified and unmodified substrates will be compared to determine if filter feeding changes the ability of bacterial species to reproduce in the environment.

PRINCIPAL FINDINGS AND OBJECTIVES

The project was not completed during the time period due to unforeseen circumstances. The fact that *Dreissena* were unavailable to collect during November through March due to the weather made it difficult to complete the work during these months. Data was collected for the first and third objectives.

Objective 1: Protein from the cyanobacterium *Spirulina* was utilized in this study. The microorganism's microbial products could be DOM utilized by bacteria in fresh waters and potential food for *Dreissena. Spirulina* powder sold as a natural food supplement was the source of protein. The data indicate that *D. polymorpha* and *D. bugensis* decreased the quantity of the protein over 24 hours, and hence the availability of the protein to bacteria (Fig. 1). Control tubes indicated that the mussels did not contribute proteins to the assay, so the proteins detected were from the *Spirulina* (Fig. 1). These results support the hypothesis that *Dreissena* filter feeding can change the quantity of carbon substrates in the environment available to bacteria.

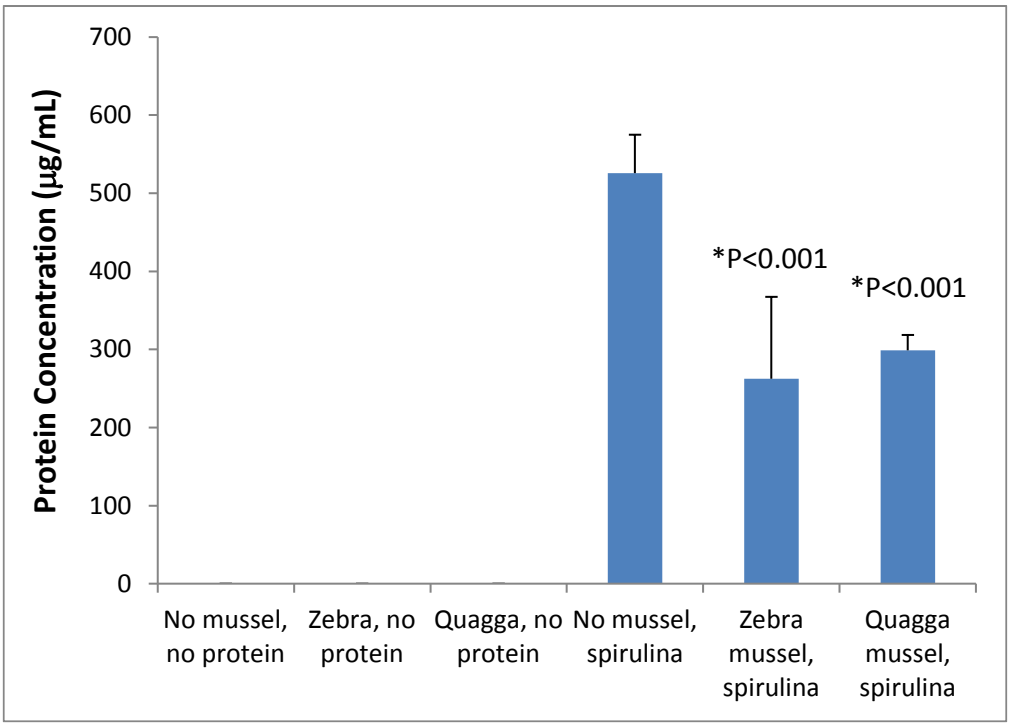

Figure 1. *Dreissena* decreases the quantity of *Spirulina* protein after 24 hours. The Bradford assay detected significantly lower levels of protein in test tubes with zebra and quagga mussels compared to tubes without the mussels. No proteins were detected in control tubes.

Objective 2: No work was completed on this objective.

Objective 3: The test tube assays confirmed that *Spirulina* is a viable protein source for use in this study to determine the effects of filter feeding on microbial diversity. The next step was to isolate bacteria that could metabolize the protein. A minimal culture medium was developed with the *Spirulina* protein as the sole carbon source. It was developed using the ingredients for minimal glucose medium, but substituting the glucose with *Spirulina* protein. Seventeen bacteria were isolated into pure cultures. The BIOLOG microplate identification system was utilized to determine species. Three bacteria were positively identified as *Aeromonas, Comamonas,* and *Exiguobacterium*. The remaining work for this objective was not completed.

STUDENT INFORMATION

Tami Roginski is an undergraduate Biology major who will complete a Bachelor of Science degree in December 2015. She plans to continue her studies at the graduate level at another institution.

PUBLICATION OR PENDING PUBLICATIONS

The results were presented by Ms. Roginski at the Lewis University STEM Undergraduate Research Experience on August 6, 2015. The work will continue during summer 2015 and should be completed by fall 2015. Findings will be presented at

regional and/or national meetings related to microbiology or ecology. A manuscript will be submitted to relevant journals with an appropriate readership. The support by the IWRC will be acknowledged.

Epikarstic groundwater ecosystems in Illinois: a sensitive by unstudied faunal element

Basic Information

Title:	Epikarstic groundwater ecosystems in Illinois: a sensitive by unstudied faunal element
Project Number:	2014IL285B
Start Date:	5/1/2014
End Date:	12/31/2014
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Congressional District:	IL-013
Research Category:	Biological Sciences
Focus Category:	Ecology, Groundwater, None
Descriptors:	None
Principal Investigators:	Steven J Taylor

Publications

There are no publications.

Progress report could not be obtained by the report deadline.

Anthropogenic litter and microplastic in urban streams: abundance, source, and fate

Basic Information

Title:	Anthropogenic litter and microplastic in urban streams: abundance, source, and fate
Project Number:	2014IL286B
Start Date:	5/1/2014
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Funding Source:	104B
Congressional District:	IL-013
Research Category:	Water Quality
Focus Category:	Water Quality, Surface Water, Ecology
Descriptors:	None
Principal Investigators:	Timothy Hoellein

Publications

There are no publications.

IWRC Progress Report, March 13, 2015

Title: Anthropogenic litter and microplastic in urban streams: abundance, source, and fate

Principal investigators:
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Introduction

The abundance of anthropogenic litter (i.e., garbage; AL) in marine ecosystems has received attention from researchers, the public, and the media. AL has many detrimental ecological effects such as ingestion and entanglement by animals and assistance in invasive species dispersal (Moore 2008). Accumulation of AL in the environment also presents an economic burden from costs related to clogged sewers and AL removal, and may represent potential hazard to human health (i.e., injury). Rivers transport materials such as organic matter, nutrients, and pollution between terrestrial and marine ecosystems. While many studies suggest rivers are an important source of AL to marine habitats (Stefatos 1999, Galgani et al. 2000, Hammer et al. 2012), the abundance, movement, and ecosystem effects of AL in rivers are unknown.

In addition to accumulations of large AL items, high concentrations of microplastic (i.e. <5 mm particles) have been measured in oceans worldwide (Browne et al. 2011). Sources of microplastic include industrial manufacturing pellets and fragmentation of larger plastic pieces. In addition, some personal care products and cleaning agents contain microplastic abrasives (Fendall and Sewell 2009), and washing machine effluent contains microplastic fibers from synthetic textiles (Browne et al. 2011). The latter two sources enter the domestic wastewater streams and wastewater treatment plants (WWTP). Microplastic particles and fibers are often unfiltered by WWTPs due to their small size (Fendall and Sewell 2009; Browne et al. 2011). Our preliminary studies were the first to show that showed that the North Shore Channel in Chicago, IL had surface water microplastic concentrations similar to oceanic gyres (McCormick et al. 2014) and that WWTP effluent was a point source. However, more work is needed to document this pattern across a larger geographic area.

In marine ecosystems, microplastic selects for unique microbial assemblages (Zettler et al. 2013), and ingestion by consumers can transport adsorbed contaminants, block digestion, and transfer from prey to predators (Cole et al. 2011). However, the ecological effects of riverine microplastic are unknown. Microplastic is buoyant and resistant to decomposition, so it presents a novel habitat which may select and disperse bacterial assemblages with unique metabolic capabilities in river networks. Our previous research showed bacterial communities on microplastic in an urban river were distinct from those in the water column and seston (McCormick et al. 2014). This was the first assessment of microplastic-attached microbial communities in rivers. Thus, more measurements on the interactions between microplastic and microbes are required to understand its potential ecological impacts.

Project Summary

This research represents a significant contribution to the study of AL in rivers. The project had 2 parts: 1) dynamics of AL in rivers, and 2) abundance, source, and microbial communities on riverine microplastic.

Research Objectives The proposed project addresses the following questions.

1a. What is the abundance, composition, and sources of AL?
b. How does AL move through rivers?

2a. What is the concentration of microplastic in urban rivers?
b. Does WWTP effluent serve as a point source for their entry into rivers and streams?
c. What types of microbes colonize microplastic in rivers and do they differ from those colonizing organic substrates?

Part 1. AL abundance, composition, and flux

Methods. *Study Sites.* We measured AL abundance and composition in 5 streams in the Chicago metropolitan region, which includes northeastern Illinois and northwestern Indiana. Study sites spanned an urban land-use gradient and had similar watershed sizes (Fitzpatrick et al. 2005; Table 1). AL was collected from the benthic and adjacent riparian zones in 3 reaches of each river (N=15). Reaches were located in publically accessible areas, including county parks and other recreational areas (Table 1). Permission and permits were obtained from county organizations before beginning the research.

Collection and categorization of benthic and riparian AL. We collected AL in June-October, 2014 (summer-autumn), except for 3 reaches sampled in 2013 (Table 1). Reach lengths were 50-100 m. AL was collected from the entire benthos of the reach, and from the riparian zone on one bank of the reach. We defined the riparian zone for this study as the area within 10 m of the water's edge. For consistency, the riparian bank chosen for AL collection was the one used to access the stream, (except for Hickory Creek at Hillcrest Road, which was inaccessible). To collect items, we slowly moved along the reach in teams of 2-3, picking up all AL. We have confidence in our estimates given the consistency with previous measurements (Hoellein et al. 2014), but note that some items may have been overlooked on the surface of the benthos. Also this method does not account for buried AL. However, any underestimates are equal across sites and dates, and establish our results as conservative. AL was transported to the lab in garbage bags labeled by collection site.

In the laboratory, AL was laid in a single layer on plastic sheets to air dry (~2-3 d) prior to counting, weighing, and categorizing each item. Dried dirt and debris were removed manually, and each AL item was weighed. We adapted a protocol from Cheshire et al. (2009) to categorize AL by material type, function, and most probable source. We classified AL into 11 material categories: ceramic, cigarettes, cloth, glass, metal, paper and cardboard, plastic, rubber, Styrofoam, wood, and 'other' (Appendix Table 1). We used a code to classify the item's function (e.g., cutlery, clothing, and cups; Appendix Table 1). Finally, we characterized each item according to most probable source using 6 categories: consumables, construction/industrial, recreation, domestic, fishing, and 'unknown.' Consumable were those materials associated with

smoking, eating, and drinking, and likely discarded by a person visiting the stream. Construction and industrial materials included pipes, manufactured wood, pallet wrap, and bricks. Recreation items were golf balls, tennis balls, and Frisbees. Items were classified as domestic if they originated from a home (e.g., kitchenware, appliances, and personal hygiene). We acknowledge the uncertainty of this source estimate. For instance, it is possible that an item classified as consumable, such as a beverage container, may have originated from a domestic source via wind or dumping of household trash. However, this approach has been used elsewhere to infer dominant AL sources (Hoellein et al. 2015, Ivar do Sul et al. 2011, Santos et al 2009).

We assessed anthropogenic activity in 4 ways: the presence and distance of a walking trail from the reach, the intensity of human activity, the number of parking spots, and the distance to a road (Table 2). We collected these data on the same date the reach was sampled at 11 of 15 sites (Table 1). Human activity data were collected at a later date than AL collection for Bunker Hill (Sep 16, 2014), Miami Woods (Sep 16, 2014), 26th Street Woods (Aug 4, 2014), and Pilcher Park (Sep 26, 2014). Trails were classified as near (<50 m from the stream), far (>50 m from the stream), or none (not present). We classified the intensity of human activity by the number of people observed at the reach or on a nearby reach trail during the sampling period (~3 h) as low (no people), medium (1-10 people), or high (>10 people). To quantify parking, we counted all parking spaces in the lot closest to the reach. Four reaches had no parking (3 in residential areas and 1 at a road intersection). We used the distance measuring tool on GoogleMaps to measure the distance from the sampled reach to the nearest road.

Riverine AL compared to marine benthos and beaches. We compared our results to AL density, mass, and composition from published studies conducted in rivers and oceans. Variation in methods, categories, and AL units complicates comparison across studies. For example, AL density is often reported as the number of items collected per unit area in benthic analyses (No. m^{-2}), but as number of items per length of transect (No. m^{-1}) in terrestrial and beach studies (Hoellein et al. 2014). Relative AL abundance is reported by material (e.g., glass, plastic, and metal) (Rech et al. 2014, Abu-Hilal 2009) or function (i.e., food-related, dumping activities, medical/personal) (Hoellein et al. 2015). To compare AL from this study to published values, we included studies which reported the number of items or mass per unit area and used similar material classifications. Studies included marine benthic habitats in European seas (Abu-Hilal and Al-Najjar 2009, Stefatos et al. 1999, Galgani et al. 2000), the open ocean (Pham et al. 2014, Pham et al. 2013), and near-shore habitats (Debrot et al. 2014, Donohue et al. 2001, Oigman-Pszczol and Creed 2007, Hess et al. 1999). Beach studies included ocean coastlines (Whiting 1998, Rosevelt et al. 2013, Madzena and Lasiak 1997, Smith and Markic 2013), estuaries (Rech et al. 2014), islands (Eriksson et al. 2013), and lakes (Hoellein et al. 2014).

AL flux: study sites. We examined movement of riparian zone AL at two spatial scales, seasonal (i.e., 3 times over 1 year), and biweekly (i.e., every 2 weeks during summer) in 2 riparian reaches of the North Branch of the Chicago River. The former was conducted at Bunker Hill Forest Preserve (Niles, IL) and the latter at Miami Woods (Morton Grove, IL). These 2 reaches were among the 15 sites used above, and are in the Cook County Forest Preserve network (Table 1).

AL flux: seasonal measurement. Our seasonal study measured the net accumulation of AL and export of marked AL items from the riparian zone over 1 year. In November 2013, all AL was cleared from a riparian quadrat (40 m length x 10 m width), directly adjacent to the water edge.

This set a 'blank slate' so that any AL collected on subsequent dates represented net accumulation. We measured net accumulation for 3 periods: winter/spring (Nov 26, 2013-Apr 25, 2014), summer (May 28-Sep 16, 2014), and fall (Sep 16-Dec 18, 2014). We did so by carefully searching the riparian quadrat and collecting all AL. The accumulated AL was taken o the laboratory for quantification and classification as described above.

At the same time we measured net accumulation, we measured export of marked AL items. To measure export, we selected 4 common AL categories: glass bottles, metal cans, plastic food containers/wrappers, and plastic bags. We marked 10 items from each category with spray paint and an identification number (N=40). On the start dates for the 3 seasonal sampling periods, the 40 marked AL items were haphazardly distributed throughout the riparian quadrat. The coordinates of each item's starting location within the quadrat were recorded. At the end of each period, we carefully searched the quadrat for the marked AL. In addition, we searched ~100 m downstream and 30 m inland from the quadrat. We recorded if the item remained in its starting location, moved within the quadrat, or was outside the quadrat (i.e., export). We established a new map for the locations of all marked AL items at the end of each sampling interval. Because a different color spray paint was used for each time period, some AL was tracked for 1 year. We removed all marked AL items in the quadrat after the final date (Dec 18, 2014). Finally, we note that all marked AL was originally collected from the study site or areas downstream, so this project represents no addition of new AL to the environment.

We calculated net accumulation and export rates from the collected data. We expressed net AL accumulation in units of No. d^{-1} and No. m^{-2} d^{-1}. We calculated AL export as the proportion of items lost per day (d^{-1}). We calculated the net accumulation and export rates for each season, and the mean annual export rate across the 4 AL categories, and calculated net flux of AL at our study site over the course of the year [Eq 1].

Eq. 1 $$Flux = Net\ accumulation - Export\ rate \times Initial\ AL\ standing\ stock$$

In this equation, we multiplied the mean annual export rate (d^{-1}) by the initial AL standing stock (No. m^{-2}). By subtracting this value from the net accumulation rate (No m^{-2} d^{-1}), we estimated net annual flux of AL (No m^{-2} d^{-1}). Finally, we calculated turnover time (d) for each AL type as the inverse of its mean output rate (d^{-1}). Turnover time represents the average time an item spends in the riparian habitat before being exported.

AL flux: biweekly measurement. We conducted an additional study to measure AL net accumulation and export in a riparian zone over shorter time intervals. This study was carried out over 18 weeks during summer 2014, starting on June 2. We visited the site every ~2 weeks (mean ± SE = 15.1 ± 1.3 d). We used the same quadrat dimensions, types of AL, and methods described above. The only difference in methods for this study was that we characterized 2 types of export. We noted if the item was out of the quadrat, but in the adjacent area (export: adjacent), or was not found (export: lost). To examine patterns between AL movement and stream hydrology, we obtained discharge data from the USGS for the North Branch of the Chicago River from Jun 2-Oct 2, 2014 (http://waterdata.usgs.gov/nwis/uv?site_no=05536000).

The effect of sampling interval on AL accumulation rates. In our seasonal and biweekly flux studies, we measured net AL accumulation rates in periods of 8-149 d. We combined our data with results from Smith and Markic (2013; *Figure 2* in that study).

Data Analysis. We used 2-way analysis of variance (ANOVA) to compare differences in AL density and mass among streams and between habitats (riparian and benthos). We conducted additional 2-way ANOVAs for each of the 11 material categories individually. Significant ANOVA results ($p<0.05$) were followed by Tukey's multiple comparison test. When data did not meet the assumptions of ANOVA, we applied a natural log transformation, or $\ln(x+0.5)$ when appropriate. However, several variables could not be transformed to meet the homoscedasticity and normality assumptions of ANOVA, which appears to be common in AL datasets (Hoellein et al. 2015). For these variables, we used a nonparametric statistical approach and performed two Kruskal Wallis tests. One tested for differences among streams and the other between habitats. This nonparametric approach limited our ability to test for an interaction effect, however, we found no significant interactions between stream and habitat for variables analyzed with ANOVA. All ANOVAs, Tukey's tests, and Kruskal Wallis tests were completed in SYSTAT 13.0 (Systat, Inc. Chicago, IL).

We used a nonmetric multidimensional scaling (nMDS) approach to analyze differences in AL composition among streams and between habitats (sensu Rech et al. 2014, Pham et al. 2014). We calculated Bray-Curtis similarity indices on $\log(x+1)$ transformed AL percent composition data for abundance and mass. The resulting distance matrix was visualized with nonmetric multidimensional scaling (nMDS) ordinations. We determined whether there were significant differences in AL composition among sites and between habitats using analysis of similarities (ANOSIM) analyses. We calculated all Bray-Curtis indices, nMDS coordinates, and ANOSIM analyses in Primer V.5 (Primer-E Ltd., Plymouth, United Kingdom). Finally, principal component analysis (PCA) was used to analyze relationships between variables associated with the anthropogenic activity at each reach and the density of each AL material type. We performed 2 PCA analyses for density of all 11 AL categories at our 2 habitats types (i.e., benthic and riparian). PCA was performed in PC-ORD V.6 (McCune and Mefford 2011) using a correlation matrix as our data included both environmental and AL density variables with varying units of measurement (Clarke and Warwick 2001).

Results & Discussion

AL abundance across streams and between habitats. Total AL density (No. m^{-2}) was significantly different among sites (2-way ANOVA, $p=0.006$; Figure 1A; Table 3), where the 3 most urbanized watersheds had the highest AL densities (Figure 1A), and AL in the two less urbanized watersheds was lower. There no difference in AL density between riparian and benthic zones (2-way ANOVA, $p=0.120$; Figure 1; Table 3), but there was a pattern of more AL in the riparian zone compared to the benthic zone (except Plum Creek; Figure 1A). Total AL mass (g m^{-2}) was highest at Turkey Creek and similar at other sites (2-way ANOVA, $p=0.005$; Figure 1B; Table 3). Benthic habitats had significantly greater AL mass than riparian zones (2-way ANOVA, $p<0.001$; Figure 1B; Table 3).

Density of AL by material type was variable among streams and between habitats. Plastic density was significantly greater in the riparian zone (2-way ANOVA, $p=0.002$; Table 3) and variable by site (2-way ANOVA $p=0.002$; Table 3). Styrofoam and paper were more abundant in the riparian zone, but there were no differences among sites (Table 3). Ceramic density was higher in the stream benthos, but did not differ among sites (Table 3). In contrast, rubber and cloth densities were similar between habitats, but variable among sites (Table 3). Finally, metal, glass, wood, cigarette, and 'other' AL did not differ between habitats or among sites (Table 3).

Patterns for the mass of each AL category were variable among streams and between habitats. In general, the heaviest AL types were highest in the benthos, including rubber (Kruskal Wallis p=0.039), metal (ANOVA p=0.003), and ceramic (ANOVA p=0.005; Table 3). In contrast, paper mass was greater in the riparian zone (Kruskal Wallis p=0.010; Table 3). Rubber and cloth mass were the only types that differed among sites (Table 3). Finally, the masses of plastic, Styrofoam, glass, wood, cigarettes, and 'other' did not differ between habitats or among sites (Table 3).

AL composition among streams and between habitats. While riparian zones typically had high density of AL (Figure 1A), a significant proportion of the AL assemblage consisted of light-weight materials such as plastic and Styrofoam (Table 4). For example, the relative abundance of plastic was higher in the riparian zone (48-65%) than in the river benthos (21-46%; Table 4). Benthic habitats had a lower AL density (Figure 1B), but heavier items such as metal, wood, and ceramic had greater relative abundance than riparian zones (Table 4). For example, metal and ceramic accounted for an average of 28% and 21% of the mass in benthic habitats, respectively, but 14% and 6% of the mass in riparian habitats (Table 4).

We calculated Bray-Curtis similarity indices for AL assemblages based on relative composition of AL density and mass. There was no significant dissimilarity in AL composition among streams (ANOSIM, R=0.084, p=0.140; Figure 2A) or between habitats (ANOSIM, R=0.133, p=0.139; Figure 2A). One riparian reach of Plum Creek strongly influenced the comparison (coordinates 2.03, -1.81; Figure 2A). This site had a very low AL density and half of the items were manufactured wood, a generally uncommon material in riparian sites elsewhere (Table 4). This site also lacked many of the AL types typical of other riparian zones such as glass, metal, paper, and Styrofoam (Table 4).

When comparing relative AL composition by mass, there was dissimilarity between habitats (ANOSIM R=0.267, p=0.027), although there were no significant differences among streams (ANOSIM R=0.036, p=0.321; Figure 2B). One riparian reach in Turkey Creek (coordinates -0.03, -2.39) and one in Plum Creek (coordinates -2.52, 0.71) are distinct on the nMDS ordination (Figure 2B). This Plum Creek reach is also distinct in the density nMDS ordination (Figure 2A). At the Turkey Creek reach, 3 tires accounted for over 96% of the site's mass. As a result, the relative contribution of rubber to the overall mass at this site was much higher than at other riparian sites (Table 4).

Comparing AL by most probable source showed distinctions between habitats. A higher proportion of AL in stream benthos came from construction and industrial sources than in riparian zones (Figure 3). This category included manufactured wood, metal, ceramic, and other building materials. In contrast, riparian habitats consisted of a high relative abundance of AL from consumable goods (Figure 3) associated with on-site littering. All recreation materials collected for this study were golf balls. Where present, golf balls were more abundant in the benthos than riparian zone (Figure 3). AL items associated with fishing were uncommon at all of the study sites (Figure 3).

Anthropogenic factors influencing AL density. We examined relationships between 4 variables related to anthropogenic activity and the density of our 11 AL categories using principle component analysis (PCA). We performed a separate PCA for benthic and riparian AL densities. The first 3 components of each PCA explained 60.5 and 63.4% of the data variation in the benthic and riparian zones, respectively (Table 5).

The first component of the PCA (PC1) explained 27.2 and 34.6% of the variation in the benthic and riparian habitats, respectively (Table 5). The second component (PC2) explained an additional 19.5 and 16.7% of the variation in the benthic and riparian habitats, respectively (Table 5). In benthic habitats, PC1 had a significant positive relationship with 3 measures of anthropogenic activity (number of parking spots, intensity of activity, and proximity of a trail) as well as all types of AL except ceramic, cigarettes, and 'other' (Table 6; Figure 4A). PC2 had a significant negative relationship with 3 measures of anthropogenic activity (the number of parking spots, distance to a road, and proximity of a trail). PC2 was negatively related to Styrofoam density, and positively related to densities of ceramic, glass, metal, rubber, and wood (Table 6; Figure 4A). PC3 showed no significant relationship with any human activity characteristics (Table 6). The heavy items, metal, rubber, and wood were clustered on the PCA diagram, and ceramic density was uncorrelated with any anthropogenic activities (Figure 4A).

In riparian habitats there was a significant negative relationship between PC1 and 2 measures of human activity (number of parking spots and intensity of activity) as well as the densities of all AL categories except for ceramic, cigarettes, and wood (Table 6; Figure 4B). In contrast, all 4 human activity characteristics showed a significant positive relationship with PC2. However, few AL categories were related to PC2 (ceramics and metal had a negative relationship and wood a positive relationship; Table 6; Figure 4B). Finally, PC3 had a significant negative relationship with the number of parking spots and the distance to a road and a significant positive relationship with paper and wood density in the riparian zone (Table 6). AL category vectors for plastic, rubber, glass, cloth, metal, and 'other' clustered in the PCA diagram (Figure 4B). Like benthic density, vectors for Styrofoam and paper densities were related to the number of parking lots and intensity of human activity, while ceramic had a negative relationship with all 4 site characteristic variables (Figure 4B).

AL density, mass, and composition across ecosystem types. The density of AL at our riparian sites was within the range reported in the literature for marine beaches, however, benthic AL density was higher than a majority of studies conducted in marine benthic environments (Table 7). Our mean (±SE) riparian AL density was 0.293 (±0.076) items m^{-2}, approximately the median of results assembled from other aquatic-terrestrial transitional habitats (Table 7). In contrast, mean (±SE) benthic AL density of 0.117 (±0.021) items m^{-2} was at least an order of magnitude above measurements in the marine benthos (Table 7). The only exception was marine density in the Gulf of Aqaba in the Red Sea which showed a mean (±range) of 2.8 (±0.9-5.9) items m^{-2} (Table 7; Abu-Hilal and Al. Najjar 2009). Far fewer studies report AL mass, yet our results for benthic and riparian habitats were consistent with the range reported in the literature from ocean sites (Table 8).

While AL density is variable, several trends emerge when comparing relative abundance of AL among studies by material type. For example, the relative abundance of metal was typically higher in benthic habitats than aquatic-terrestrial transition habitats (i.e., riparian zone and beaches). The abundance of metal in our benthic and riparian habitats (18%, and 9%, respectively) was comparable to the proportion of metal in marine benthic studies (range=3-27%) and higher than metal abundance in all but one beach (range=0-35%; Figure 5). The relative abundance of glass at our study sites was higher than all other studies except 2 beaches (Figure 5). While plastic was a major component of AL assemblages in rivers (range=30-55%) and marine benthic sites (range=19-64%), beaches were more likely to be dominated by plastic

(range=32-95%) (Figure 5). Styrofoam was uncommon in marine benthic sites (range=0-1%), relatively rare in rivers (range=0-15%), and most common on beaches (range=0-41%) (Figure 5). Finally, an important difference in AL composition between the marine benthos and other habitats was the prevalence of fishing items in ocean sites (Figure 5).

Seasonal flux: net accumulation and export rates

Patterns of net accumulation and export at the seasonal scale reveal that AL is highly mobile. Across the 3 seasonal intervals, mean (±SE) net accumulation of AL was 1.114 (±0.219) items d^{-1} or 0.0028 (±0.0005) items m^{-2} d^{-1}. Mean export rate for the AL types combined was 0.379 (±0.023) % d^{-1}, and was higher in spring and summer relative to fall (Table 9). Across AL types, there were no significant differences in export rates (1-way ANOVA, p=0.061). The mean (±SE) turnover time among the 4 AL types was 264 (±41) d, where aluminum cans had the shortest (197 d), and glass and plastic wrappers the longest (330 and 368 d, respectively) turnover times (Table 10). This suggests all 4 AL types are likely to leave the study reach within 1 year.

Using the original density of AL in the reach (0.9883 items m^{-2}), mean net accumulation rate, and mean total export rate, we calculated annual AL flux from this riparian zone site with the following calculation:

$$Flux = Net\ accumulation\ rate - Export\ rate$$
$$= 0.002785\ No.m^{-2}d^{-1} - (0.9883\ No.m^{-2})(0.003793\ d^{-1})$$
$$= -0.000964\ No.m^{-2}d^{-1}$$

The net flux of AL from the study quadrat over the course of the year was -0.000964 items m^{-2} d^{-1}. This is consistent with a mean turnover time of 264 d (i.e., < 1 y). Scaled to the quadrat dimensions (400 m^2) over the course of the year, the total export was 547 items y^{-1}, net accumulation was 407 items y^{-1} and the flux was a net loss of -141 items y^{-1}.

Biweekly flux: net accumulation and export rates. To complement our annual flux assessment, we measured net accumulation and export over shorter time scales. At a biweekly scale, net AL accumulation rates in the riparian zone were between 0.8-9 items d^{-1} (Mean (±SE)=3.435 (±1.050) items d^{-1} and 0.009 (±0.003) items m^{-2} d^{-1}). The biweekly accumulation rates were higher than those from the seasonal study (Table 9). Plastic and glass dominated AL input (Figure 6), and glass was typically broken bottles. There was no clear relationship between the river discharge and changes in input rates or relative AL composition (Figure 6).

Unlike net accumulation, export of marked AL items was related to AL material type, river discharge, and proximity of each item to the river edge. After 15 d, 100% of glass bottles, 60% of metal cans, 80% of plastic wrappers, and 70% of plastic bags remained in their original location (Figure 7). After 36 d, however, 30% of glass bottles, 20% of metal cans, 50% of plastic wrappers, and 50% of plastic bags were in their original locations (Figure 7). From that time onwards, the number of items remaining in their original locations was relatively constant (Figure 7). However, we note a later decline in the proportion of stationary plastic wrappers (Figure 7C). Overall, glass and metal were more frequently exported from the quadrat than plastic wrappers and bags (Figure 7). Exported plastic wrappers and bags that did move from their original location in the study quadrat were commonly exported near the vicinity of our quadrat (i.e., export: adjacent, Figure 7C, 7D), while glass and metal were lost. Many of the plastic items accumulated in a debris dam ~20 m inland from the study quadrat. The river

discharge peaked during the first third of our study, corresponding to the period of greatest AL movement (Figure 7).

Movement of AL relative to its original location also revealed the influence of flooding on AL redistribution. The third of the quadrat closest to the water's edge had the lowest proportion of AL items remaining in their original locations. By the second date, which was 36 days into the study, only 7% of items were in their original location in the section of the quadrat within 0-3.3 m of the water's edge, and by the end of the study, 0% of AL from this section remained (Figure 8A). In contrast, for the AL items in the middle (3.3-6.7 m from the water's edge) and inland (6.7-10 m from the water's edge) sections of the quadrat, 50% and 54% of items remained in their original locations during the second sampling date, respectively (Figure 8B, C). In addition, items in the middle and inland sections were more likely to remain in the vicinity of the quadrat when exported (export: adjacent), while items exported in the section near the water's edge were much less likely to be recovered in the vicinity of the quadrat (export: lost; Figure 8).

We used the same approach as in our seasonal study to calculate AL flux for the summer season at our biweekly study reach. Using the original standing stock density of AL in the reach (0.037 items m^{-2}), the mean net accumulation rate, and the final export rate, we calculated summer flux with the following calculation:

$$Flux = Net\ accumulation\ rate - Export\ rate$$
$$= 0.008588\ No.m^{-2}d^{-1} - (0.037\ No.m^{-2})(0.005328\ d^{-1})$$
$$= 0.008391\ No.m^{-2}d^{-1}$$

Thus, the net flux of AL from the study quadrat was 0.008391 items m^{-2} d^{-1}. Unlike the annual value, the net flux over summer was positive, suggesting net accumulation of AL during this time.

The effect of sampling interval on AL accumulation rates. Previous research suggests there is a relationship between accumulation rate (No. items m^{-2} d^{-1}) and sampling interval as a power function (Smith and Markic 2013). Net accumulation and sampling interval showed a significant relationship (R^2=0.559, p=0.005, Figure 9). In addition, we combined these data with similar measurements for a beach in Australia which showed the same relationship (Figure 2 from Smith and Markic 2013). When all data were combined, the power function maintained this significant relationship (R^2=0.872, p<0.001; Figure 9). These data, combined with the net positive flux of AL during our seasonal study, confirm that much of the AL found at a site is in motion, and may be missed altogether if sampling intervals are too far apart.

Estimation of total AL in rivers. We scaled up our density and mass measurements of AL to estimate the total abundance of AL in each river's benthos, riparian zone (10 m from the water's edge on each bank), and the two habitats combined. This estimation suggests that some of the rivers in this study may have over 700,000 AL items (Table 11) in that relatively small part of the watershed. The estimated total AL mass in the benthic + riparian zones in the study watersheds was 21-78 metric tons (Table 11). Using the annual export rate from our seasonal flux study, we calculated that up to 2,000 items d^{-1} are exported from the riparian zones at Salt Creek and N. Branch Chicago River, with less daily export at the others sites (Table 11).

Conclusion

We found that the riparian zones typically had higher density of AL than benthic habitats, but that riparian AL assemblages typically had a higher prevalence of light-weight materials. While benthic habitats typically had relatively low density to riparian habitats, AL mass at these locations was much higher than in the riparian zone. This is reflected in the higher relative abundance of heavy material types in the river benthos. It is likely that reach-specific characteristics (i.e., our 4 anthropogenic activity variables) have a greater influence on AL abundance and composition than watershed characteristics (i.e., urban land use, watershed size). Overall, we found that AL density and mass in urban rivers is in the range of data reported from other published studies, but the density of AL in river benthos was much higher than most marine benthic studies. Finally, we demonstrate that riverine AL is highly mobile, and that hydrology influences the export of AL downstream. Despite the limited spatial scale of our measurements (i.e., 10 m adjacent to the river), the abundance and mass of AL at the watershed scale is very high, and much of it is in motion. These data support anecdotal claims that rivers are source of AL to downstream habitats.

Part I: Tables and Figures

Table 1. Locations and land use characteristics for the 15 sampled stream reaches.

Stream	Urban Land Use (%)	Pop. Density (No. km^{-2})	Reach	Function	Date Sampled	City	County (State)	Latitude, Longitude
Salt Cr.	73*	1236*	26th St Woods	Forest Preserve	28-Oct-13	Berwyn	Cook (IL)	41.8426265, -87.8595240
			Sleepy Hollow Park	Residential	31-Jul-14	Elmhurst	DuPage (IL)	41.8809199, -87.9584902
			Bemis Woods	Forest Preserve	4-Aug-14	W. Springs	Cook (IL)	41.8266232, -87.9106167
Turkey Cr.	53^{+}	333^{+}	Hidden Lake	County Park	6-Jun-14	Merrillville	Lake (IN)	41.5035670, -87.3277269
			Broadway St	Commercial	6-Jun-14	Merrillville	Lake (IN)	41.5031540, -87.3367570
			Hidden Lake	County Park	7-Jun-14	Merrillville	Lake (IN)	41.5041679, -87.3305381
N. Br. Chi.R.	48*	572*	Bunker Hill	Forest Preserve	23-Sep-13	Niles	Cook (IL)	42.0004406, -87.7835676
			Miami Woods	Forest Preserve	2-Jun-14	Morton Grove	Cook (IL)	42.0274486, -87.7937181
			LaBagh Woods	Forest Preserve	30-Jul-14	Chicago	Cook (IL)	41.9780194, -87.7427133
Hickory Cr.	21*	352*	Pilcher Park	Nature Cent/Park	28-Oct-13	Joliet	Will (IL)	41.5262440, -88.0070321
			Hillcrest Rd	Residential	26-Sep-14	Joliet	Will (IL)	41.5251124, -88.0409162
			Schoolhouse Rd	Intersection	26-Sep-14	Joliet	Will (IL)	41.5169885, -87.9333094
Plum Cr.	8*	88*	Plum Cr For. Pres.	Forest Preserve	14-Aug-14	Beecher	Will (IL)	41.3931726, -87.6243616
			Goodenow Nat. Pres.	Forest Preserve	14-Aug-14	Beecher	Will (IL)	41.4036569, -87.6091785
			Ridgeland Ave	Residential	28-Sep-14	Chicago Heights	Cook (IL)	41.4827058, -87.5319383

* indicates data were obtained from Fitzpatrick et al. 2005. + indicates data were obtained from Northwestern Indiana Regional Planning Commission 2012.

Table 2. Site characteristics for the 15 reaches in the 5 study streams.

Reach	Distance to trail (m)	Parking spots	Distance to road (m)	Activity Observed
26th St Woods	43	40	93	Frequent walkers, cyclists
Sleepy Hollow	7	na	32	Moderate walkers
Bemis Woods	45	140	134	Little observed
Hidden Lake	na	105	27	Fishing, walking, vehicle traffic
Broadway St	na	130	140	Industrial employees
Hidden Lake	33	100	211	Recreational (sports fields)
Bunker Hill	121	250	229	Frequent walkers, cyclists
Miami Woods	20	180	230	Frequent walkers, cyclists
LaBagh Woods	30	200	154	Little observed
Pilcher Park	na	40	44	Little observed
Hillcrest Rd	na	na	20	Little observed
Schoolhouse Rd	na	na	5	Vehicle Traffic
Plum Cr Forest Pres	62	137	823	Little observed
Goodenow Nat. Pres	133	100	237	Little observed
Ridgeland Ave	na	na	20	Vehicle Traffic

na indicates that no trail or parking lot was present at the reach.

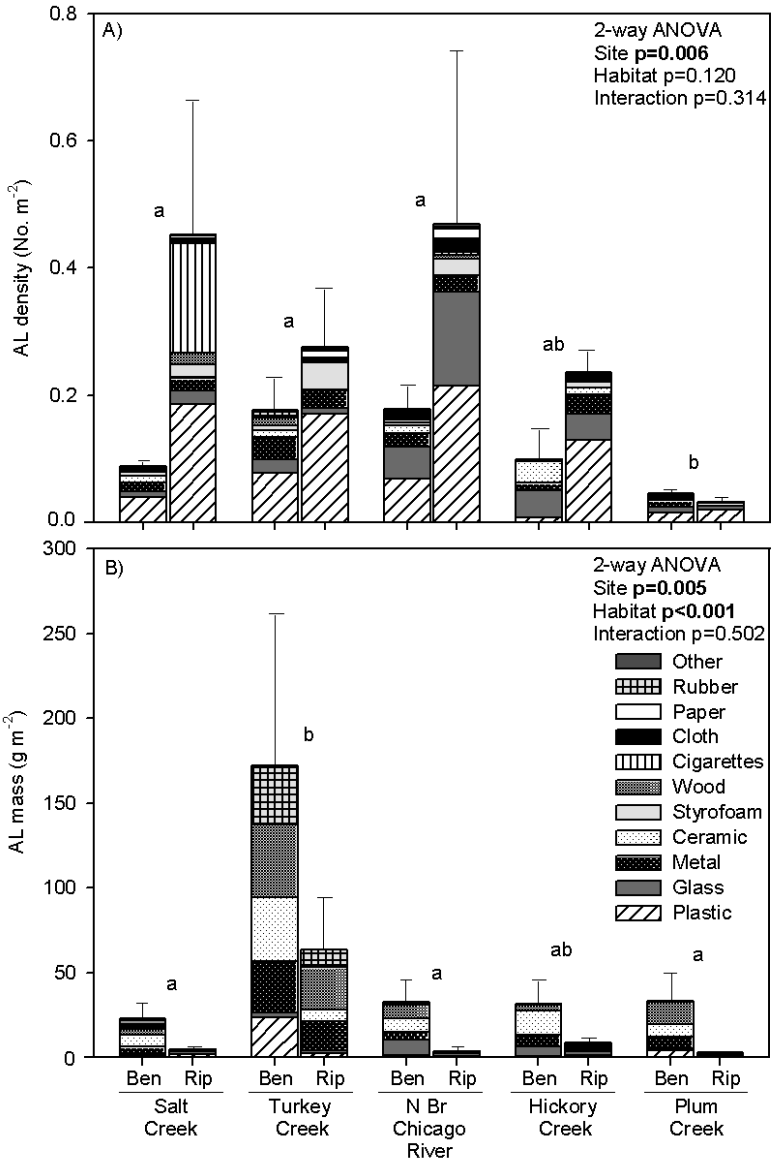

Figure 1. Density (A) and mass (B) of anthropogenic litter (AL) in 5 streams and 2 habitats (river benthos and riparian zone). Bars represent mean density and mass of combined AL categories with standard error bars. Each bar section represents the mean density or mass of that category in the reach. Letters indicate a difference between AL density or mass among sites using Tukey's test ($p \leq 0.05$).

Table 3. Summary of differences in AL density and mass between habitats and across sites using 2-way ANOVA and when necessary, the non-parametric Kruskal Wallis Test.

		Density		Mass				Density		Mass	
AL Type	Factor	Test Stat.	p-value	Test Stat.	p-value	AL Type	Factor	Test Stat.	p-value	Test Stat.	p-value
Total*	Habitat	2.64	0.120	17.43	**<0.001**	Paper	Habitat	6.85#	**0.009**	6.60#	**0.010**
	Site	4.94	**0.006**	5.19	**0.005**		Site	5.56#	0.235	5.88#	0.208
	Interaction	1.27	0.314	0.87	0.502		Interaction	-	-	-	-
Plastic*	Habitat	12.25	**0.002**	3.33	0.083	Cloth	Habitat	2.80#	0.095	0.74#	0.391
	Site	6.35	**0.002**	1.71	0.188		Site	13.81#	**0.008**	10.05#	**0.040**
	Interaction	2.50	0.075	1.54	0.228		Interaction	-	-		
Rubber	Habitat	1.71#	0.191	4.25#	**0.039**	Glass+	Habitat	0.01#	0.917	3.54	0.075
	Site	14.84#	**0.005**	12.03#	**0.017**		Site	8.38#	0.079	1.42	0.262
	Interaction	-	-	-	-		Interaction	-	-	0.76	0.564
Metal	Habitat	0.39	0.538	11.46+	**0.003**	Wood+	Habitat	0.19#	0.660	2.76	0.112
	Site	2.14	0.113	2.09+	0.120		Site	3.56#	0.468	0.22	0.925
	Interaction	0.83	0.524	0.121+	0.973		Interaction	-	-	0.23	0.920
Ceramic+	Habitat	6.54#	**0.011**	10.17	**0.005**	Cig.	Habitat	2.43#	0.119	2.28#	0.131
	Site	5.02#	0.285	0.89	0.487		Site	2.83#	0.587	2.45#	0.654
	Interaction	-	-	0.09	0.986		Interaction	-	-	-	-
Styro.	Habitat	6.97	**0.016**	2.44#	0.118	Other	Habitat	1.47#	0.226	3.71#	0.054
	Site	1.91	0.148	4.59#	0.332		Site	3.57#	0.467	2.12#	0.715
	Interaction	1.11	0.379	-	-		Interaction	-	-	-	-

Patterns in total AL density and mass and each category were compared between habitats (riparian and benthos) and across sites. Test statistics are ANOVA F-rations unless noted with #, which indicates Test Statistic from the non-parametric Kruskal Wallis Test. Symbol * indicates ln(x) transformation; + indicates ln(x+0.5) transformation.

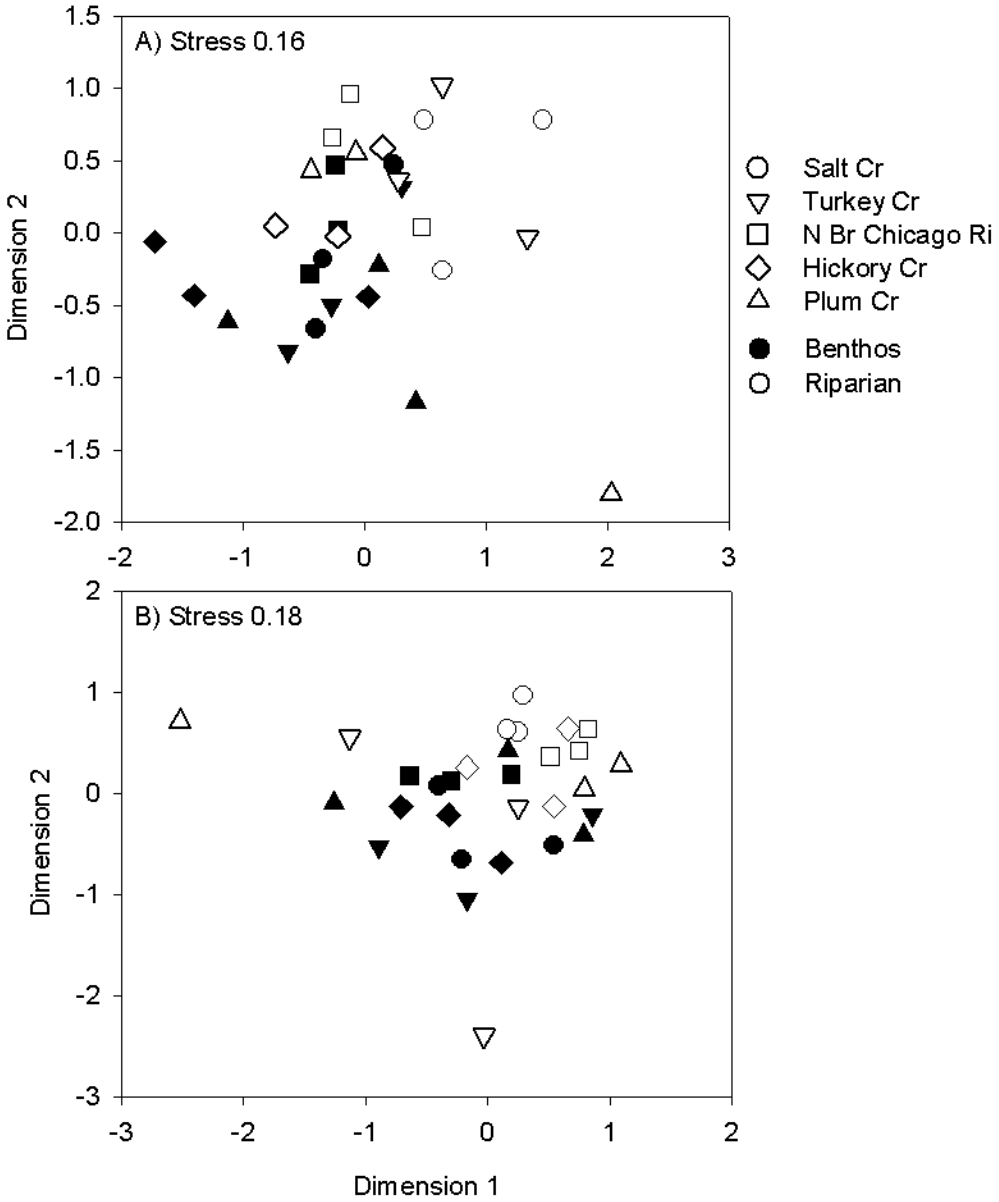

Figure 2. Non-metric multidimensional scaling (nMDS) ordination based on Bray-Curtis dissimilarity of anthropogenic litter (AL) composition in 5 streams and 2 habitats based on relative abundance (A) and relative mass (B) of the 11 AL categories. Percent composition data for AL abundance and mass was log(x+1) transformed.

Table 4. AL composition by abundance and mass for benthic and riparian habitats for the 5 streams (n=3 reaches per stream). Data values represent the mean percent contribution of each AL category for each site.

	Benthos					Riparian				
Density	Salt Cr	Turkey Cr	N Br Chi R	Hickory Cr	Plum Cr	Salt Cr	Turkey Cr	N Br Chi R	Hickory Cr	Plum Cr
Ceramic	12.5	8.3	5.1	28.3	7.5	0.1	0.4	0.3	5.2	0.7
Cigarette	0.0	0.0	0.2	0.5	2.2	19.9	0.3	0.7	0.0	0.7
Cloth	1.3	0.9	4.5	0.5	0.0	1.3	0.8	9.7	3.4	0.0
Glass	10.4	10.0	27.0	27.7	18.2	5.4	3.4	21.7	16.6	12.0
Metal	14.3	22.2	11.1	16.0	27.1	7.0	7.8	6.7	12.7	9.9
Other	0.9	2.4	0.4	1.4	4.5	0.2	0.4	0.6	1.3	0.0
Paper	0.0	0.3	1.6	0.0	0.0	1.5	3.7	3.9	1.3	0.0
Plastic	45.9	42.9	44.0	20.8	30.8	47.8	65.2	48.7	55.2	57.2
Rubber	3.2	3.7	1.0	0.5	0.0	0.2	2.3	0.2	0.0	0.0
Styrofoam	8.4	4.0	2.6	3.2	3.7	6.8	14.5	5.6	3.5	2.8
Wood	3.1	5.2	2.4	1.1	6.0	9.7	1.3	1.9	0.8	16.7
Mass										
Ceramic	19.5	19.7	20.0	32.0	12.1	0.0	18.4	0.3	10.8	0.3
Cigarettes	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0
Cloth	7.9	0.1	2.7	0.1	0.0	9.3	3.2	21.6	17.2	0.0
Glass	9.7	2.2	33.7	11.2	6.6	31.9	3.8	31.2	21.0	12.5
Metal	23.3	40.8	10.8	31.5	32.7	6.7	16.4	16.2	11.9	19.4
Other	4.2	1.0	0.2	2.0	6.0	0.7	0.1	1.4	10.5	0.0
Paper	0.0	0.0	0.1	0.0	0.0	1.3	0.1	6.9	0.2	0.0
Plastic	8.1	12.6	13.1	4.0	19.6	36.1	5.6	20.6	23.5	34.4
Rubber	11.4	10.3	0.6	0.5	0.0	0.2	32.2	0.3	0.0	0.0
Styrofoam	0.1	0.0	0.0	0.2	0.0	0.5	0.3	0.6	0.3	0.1
Wood	15.9	13.3	18.8	18.5	23.0	12.6	19.8	0.9	4.7	33.3

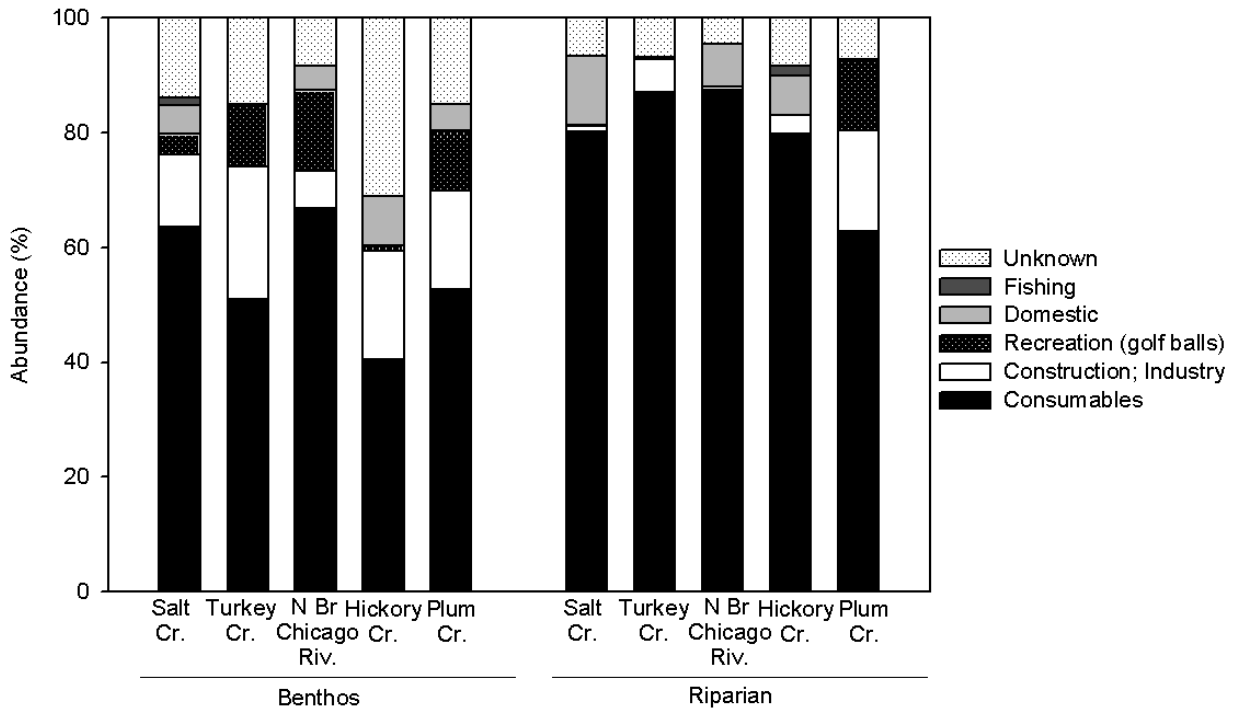

Figure 3. Mean relative proportion of sources contributing to anthropogenic litter (AL) collected from 5 streams.

Table 5. Contribution of first 3 PCA components in explaining variation in AL density and mass in stream habitats.

Density, benthos			*Density, riparian*		
Axis	Variation (%)	Cumulative variation (%)	Axis	Variation (%)	Cumulative variation (%)
1	27.22	27.22	1	34.62	34.62
2	19.45	46.66	2	16.65	51.27
3	13.81	60.48	3	12.13	63.40

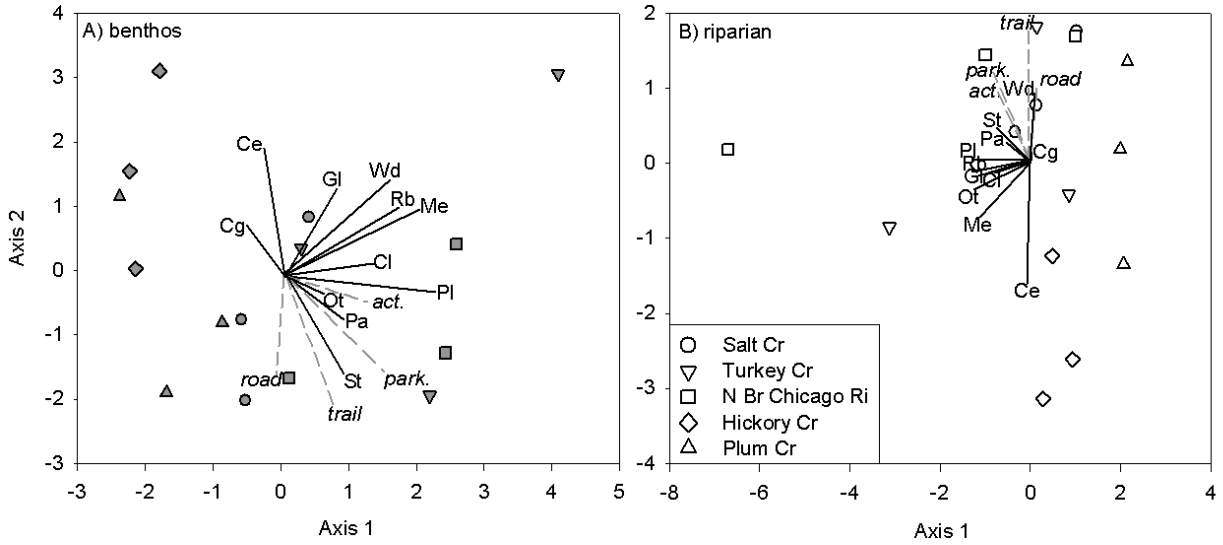

Figure 4. Principle component analysis (PCA) of site characteristics (presence and distance of a trail, number of parking spots, distance to a road, and level of human activity) (gray, dashed lines) and AL abundance at the 15 sampling sites. Abbreviations: park.=parking, act.=activity, Ce=ceramic, Cg=cigarettes, Cl=cloth, Gl=glass, Me=metal, Pa=paper and cardboard, Pl=plastic, Rb=rubber, St=Styrofoam, Wd=wood, Ot=other.

Table 6. Correlation coefficients for AL abundance and site characteristics for PCs 1, 2, and 3. Eigenvalues are considered significant at ≥0.3 and ≤-0.3, which are marked in bold.

	Benthos			Riparian		
	PC1	PC2	PC3	PC1	PC2	PC3
Site characteristics						
Parking	**0.597**	**-0.495**	0.183	**-0.554**	**0.563**	**-0.426**
Road	-0.048	**-0.569**	-0.088	0.093	**0.490**	**-0.602**
Activity	**0.519**	-0.153	-0.296	**-0.491**	**0.441**	0.118
Trail	**0.314**	**-0.727**	0.195	-0.018	**0.855**	0.298
AL abundance						
Ceramic	-0.125	**0.711**	**0.490**	-0.051	**-0.763**	0.193
Cigarettes	-0.234	0.280	0.068	-0.046	0.079	0.457
Cloth	**0.554**	0.065	**0.465**	**-0.896**	-0.065	-0.086
Glass	**0.335**	**0.485**	**0.712**	**-0.847**	-0.069	-0.192
Metal	**0.826**	**0.393**	-0.076	**-0.771**	**-0.352**	0.336
Other	0.247	-0.108	**-0.418**	**-0.856**	-0.168	-0.274
Paper	**0.375**	-0.251	**0.631**	**-0.368**	0.124	**0.497**
Plastic	**0.940**	-0.098	-0.075	**-0.891**	0.012	0.121
Rubber	**0.721**	**0.373**	**-0.441**	**-0.773**	-0.089	-0.095
Styrofoam	**0.376**	**-0.551**	0.240	**-0.506**	0.212	0.238
Wood	**0.663**	**0.540**	**-0.320**	0.059	**0.444**	**0.607**

Table 7. Published AL densities for worldwide benthic and aquatic-land transitional habitats.

Location	Ecosystem	Habitat	N	Measurement	AL Density (No. m^{-2})	Source
Benthic habitats						
N. Illinois/Indiana, USA	River	Benthos	15	Mean (±SE)	0.117 (0.021)	*This study*
N. Br. Chicago R., USA	River	Benthos	3	Mean (±SE)	0.076 (0.018)	Hoellein et al. 2014
Gulf of Aqaba, Red Sea	Marine	Benthos	6	Mean (Range)	2.8 (0.9-5.9)	Abu-Hilal and Al-Najjar 2009
Mediterranean Sea	Marine	Benthos	2	Mean	0.000165	Stefatos et al. 1999
Caribbean Islands	Marine	Benthos	24	Mean (Max)	0.0027 (0.0046)	Debrot et al. 2014
Condor Seamount, PT	Marine	Benthos	NR	Mean	0.00098	Pham et al. 2013
NW Hawaiian Islands	Marine	Benthos	2	Mean (Range)	0.000033	Donohue et al. 2001
European Seas	Marine	Benthos	18	Range	0-0.101	Galgani et al. 2000
Atlantic Ocean	Marine	Benthos	21	Range	0.0003-0.0032	Pham et al. 2014
Mediterranean Sea	Marine	Benthos	10	Range	0.0004-0.0032	
Arctic Ocean	Marine	Benthos	1	Mean	0.00136	
Armacao dos Buzios, BR	Marine	Subtidal	10	Mean (Range)	0.029 (0.003-0.065)	Oigman-Pszczol and Creed 2007
Aquatic-terrestrial transitional zones						
Combined study sites	River	Riparian	15	Mean (±SE)	0.293 (0.076)	*This study*
N. Br. Chicago R., US	River	Riparian	3	Mean(±SE)	0.095 (0.017)	Hoellein et al. 2014
Lake Michigan, US	Lake	Beach	3	Mean(±SE)	0.007(0.002)	
Lake Michigan, US	Lake	Beach	5	Mean(±SE)	0.009 (0.005)	Hoellein et al. 2015
Sea of Japan, Japan	Marine	Beach	18	Mean (Range)	3.41 (0.46-12.72)	Kusui and Noda 2003
Sea of Japan, Russia	Marine	Beach	8	Mean	0.21	
Gulf of Aqaba, Red Sea	Marine	Beach	3	Mean (Range)	4.51 (1.64-7.38)	Abu-Hilal and Al-Najjar 2004
Israel	Marine	Beach	6	Range	0.03-0.88	Bowman et al. 1998
Monterey Bay, USA	Marine	Beach	12	Mean (Range)	1 (0.03-17.1)	Rosevelt et al. 2013
Charlesworth Bay, AU	Marine	Beach	1	Standing stock	0.24	Smith & Markic 2013
Armacao dos Buzios, BR	Marine	Beach	10	Mean (Range)	0.138 (0.233-0.034)	Oigman-Pszczol and Creed 2007
Curacao, West Indies	Marine	Beach	5	Mean (±SD)	0.365 (0.410)	Nagelkerken et al. 2001

USA=United States, PT=Portugal, BR=Brazil, AU=Australia, NR = not reported.

Table 8. Published AL mass for worldwide benthic and aquatic-land transitional habitats.

Location	Ecosystem	Habitat	N	Measurement	AL Mass (g m^{-2})	Source
Benthic habitats						
Combined study sites	River	Benthos	15	Mean(±SE)	58.40 (16.74)	*This study*
N. Br. Chicago R., USA	River	Benthos	3	Mean(±SE)	13.43 (0.65)	Hoellein et al. 2014
Gulf of Aqaba, Red Sea	Marine	Benthos	6	Mean (Range)	310 (60-1060)	Abu-Hilal and Al-Najjar 2009
Aquatic-terrestrial transitional zones						
Combined study sites	River	Riparian	15	Mean(±SE)	16.74 (8.20)	This study
N. Br. Chicago R., USA	River	Riparian	3	Mean(±SE)	18.04 (5.10)	Hoellein et al. 2014
Lake Michigan, USA	Lake	Beach	3	Mean(±SE)	0.20 (0.12)	
Curacao, West Indies	Marine	Beach	5	Mean(±SD)	187 (532)	Nagelkerken et al. 2001
Sea of Japan, Japan	Marine	Beach	18	Mean (Range)	21.4 (1.4-73.3)	Kusui and Noda 2003
Sea of Japan, Russia	Marine	Beach	8	Mean (Max)	13.4 (46.9)	

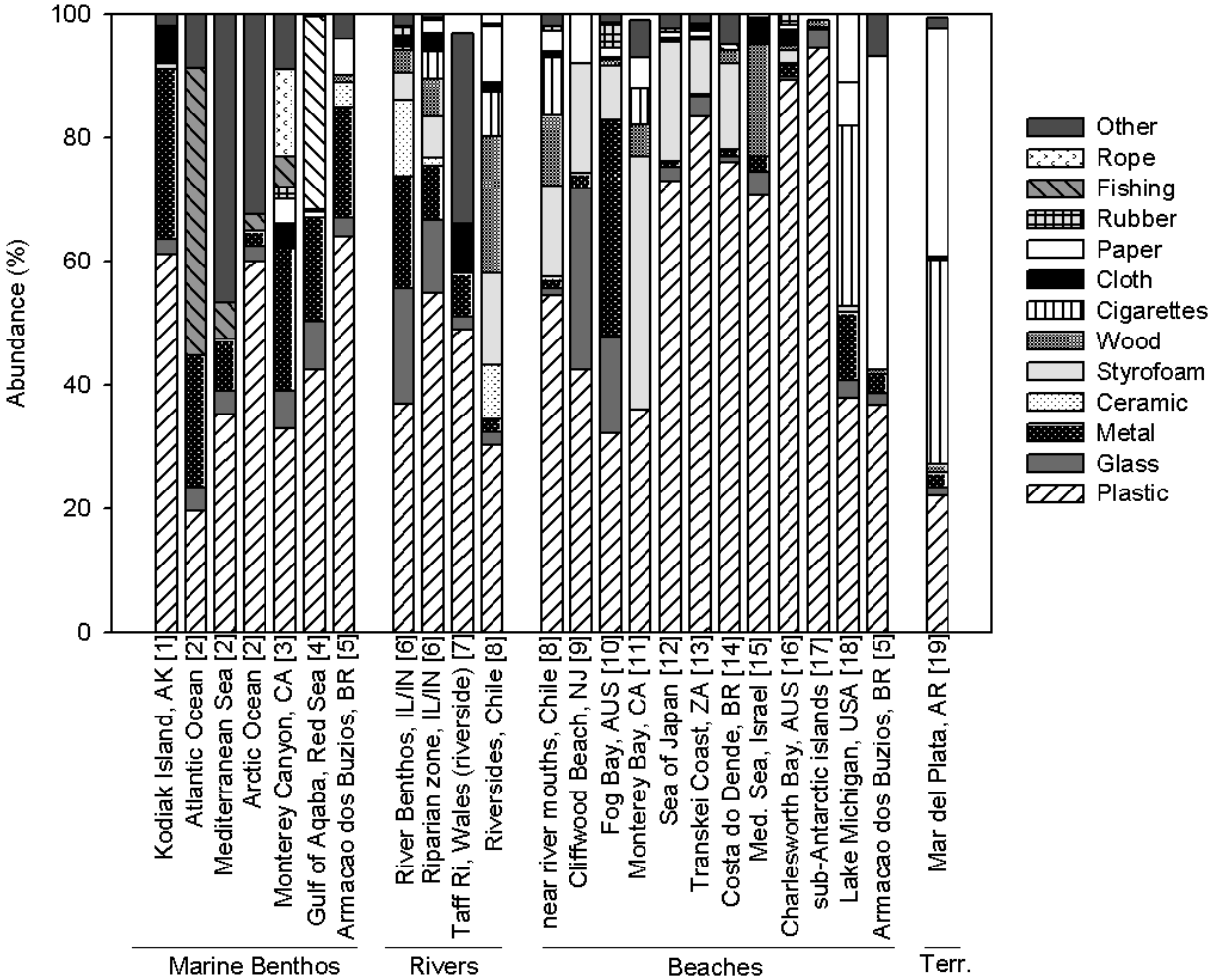

Figure 5. Relative abundance of anthropogenic litter (AL) categories in marine benthos, river, beach, and terrestrial habitats. Bars from this study represent the overall mean relative abundances for all riparian data combined and all benthic data combined**.** Numbers in brackets refer to the following sources: [1] Hess et al. 1999; [2] Pham et al. 2014; [3] Schlining et al. 2013; [4] Abu-Hilal and Al-Najjar 2009; [5] Oigman-Pszczol and Creed 2007; [6] this study; [7] Williams and Simmons 1999; [8] Rech et al. 2014; [9] Thornton and Jackson 1998; [10] Whiting 1998; [11] Rosevelt et al. 2013; [12] Kusui and Noda 2003; [13] Madzena and Lasiak 1997; [14] Santos et al. 2009; [15] Bowman et al. 1998; [16] Smith and Markic 2013; [17] Eriksson et al. 2013; [18] Hoellein et al. 2014; [19] Seco Pon and Becherucci 2012.

Table 9. Summary of net accumulation and export rates for the seasonal flux study.

Sampling Interval			Net Accumulation		Export rate (% d^{-1})					Export
Start	End	Days	No. d^{-1}	No. m^{-2} d^{-1}	Glass	Metal	Wrapper	Bag	Total	No. m^{-2} d^{-1}
26-Nov-13	25-Apr-14	149	0.8121	0.0020	0.2685	0.5369	0.3356	0.4698	0.4027	0.0039797
28-May-14	16-Sep-14	111	1.5405	0.0039	0.2815	0.4505	0.4204	0.4851	0.4022	0.0039748
16-Sep-14	18-Dec-14	93	0.9892	0.0025	0.3584	0.5376	0.0597	0.4032	0.3332	0.0032928

Table 10. Estimated turnover time for each type of item based on results from the seasonal flux study.

AL	Export (% d^{-1})	Turnover time (d)
Glass	0.3028	330
Metal	0.5083	197
Wrapper	0.2719	368
Bag	0.4527	221
Mean	0.3794	264

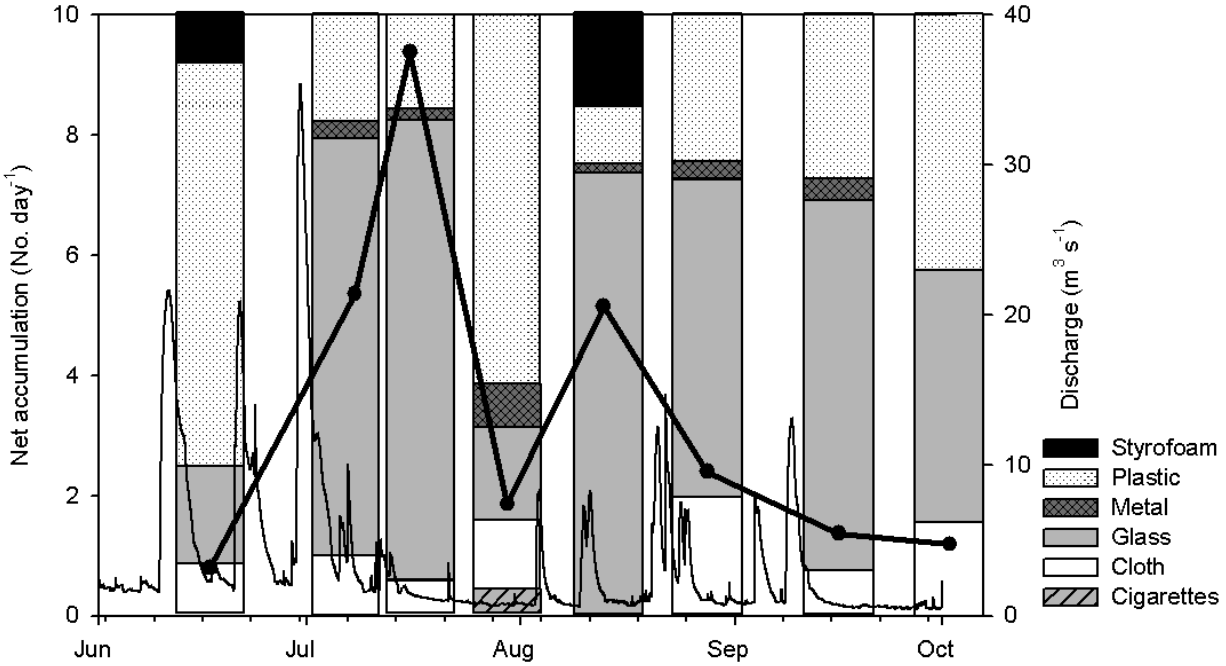

Figure 6. Composition of anthropogenic litter (AL) net accumulation and net accumulation rates for the biweekly flux study conducted in the riparian zone of the North Branch of the Chicago River in Miami Woods.

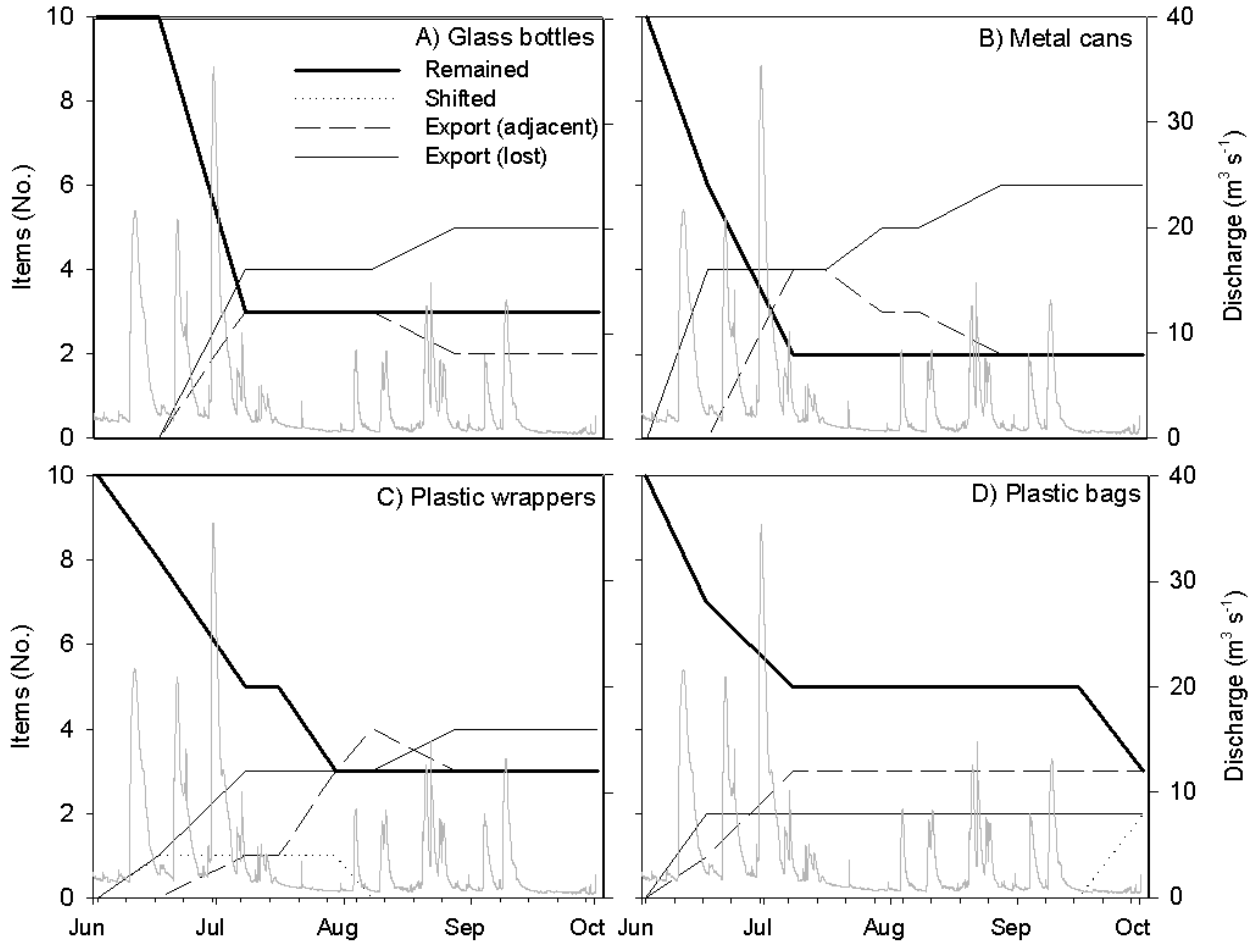

Figure 7. Movement patterns of marked glass bottles (A), metal cans (B), plastic wrappers (C), and plastic bags (D) during the biweekly flux study conducted in the riparian zone of the North Branch of the Chicago River in Miami Woods. 'Remained' indicates that the item remained in its original location between sampling periods. 'Shifted' indicates that the item moved between sampling periods but remained within the study quadrat. 'Exported (near)' indicates that the item was exported out of the quadrat but remained in the vicinity (within 100 m downstream and 40 m inland) of the study area. 'Exported (lost)' indicates the item was exported from the quadrat and was not in the vicinity of the study area.

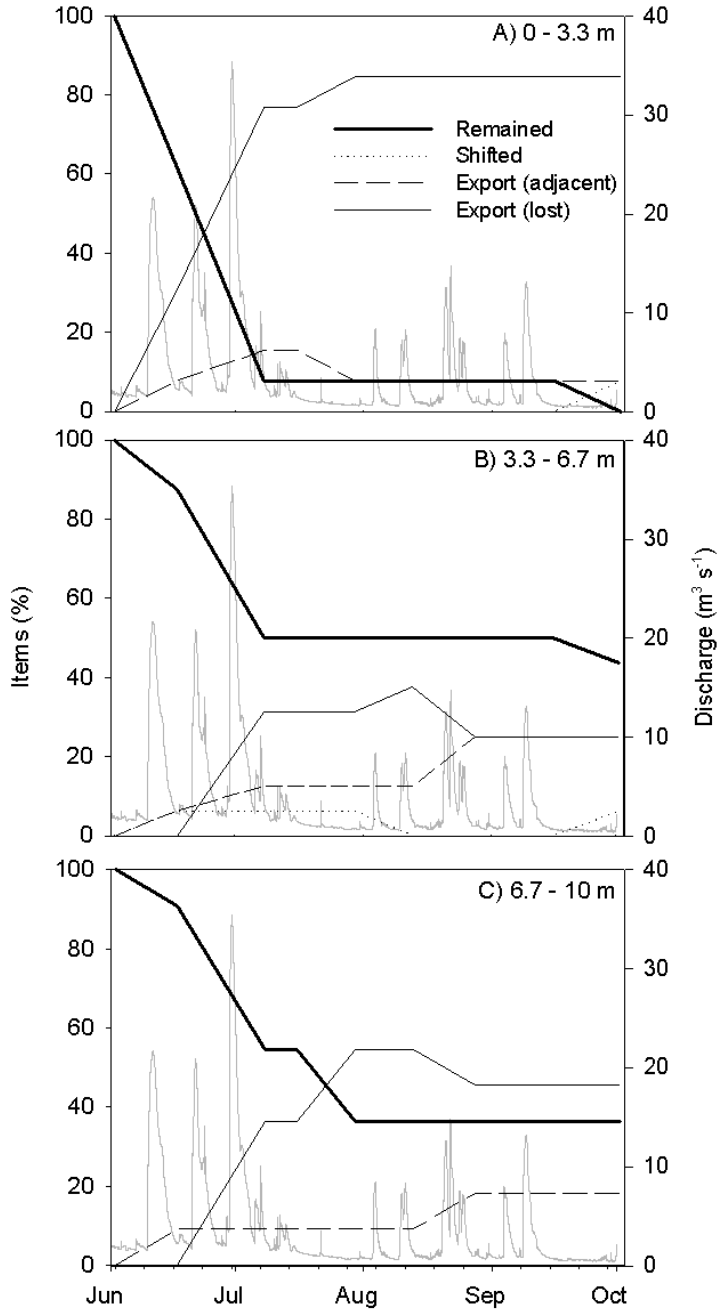

Figure 8. Movement patterns of marked anthropogenic litter (AL) items near the water's edge (0-3.3 m inland) (A), the middle (3.3-6.7 m inland) of the quadrat (B), and the most inland portion of the quadrat (6.7-10 m inland) (C) during the biweekly flux study conducted in the riparian zone of the North Branch of the Chicago River in Miami Woods. 'Remained' indicates that the item remained in its original location between sampling periods. 'Shifted' indicates that the item moved between sampling periods but remained within the study quadrat. 'Export (near)' indicates that the item was exported out of the quadrat but remained in the vicinity (within 100 m downstream and 40 m inland) of the study area. 'Export (lost)' indicates the item was exported from the quadrat and was not in the vicinity of the study area.

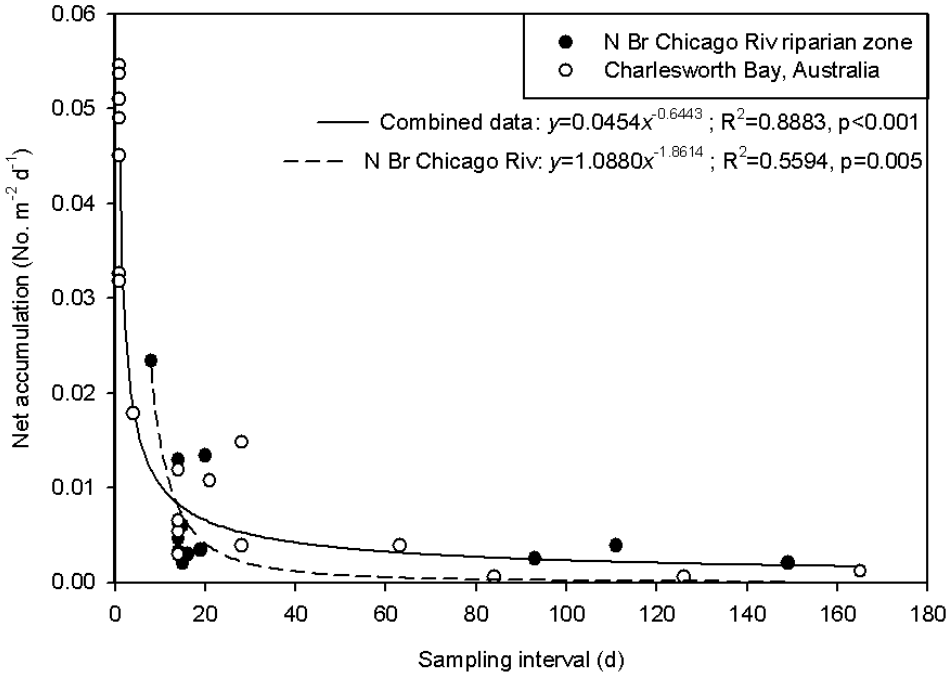

Figure 9. Plot of estimated daily accumulation rate of anthropogenic litter compared to time between sampling periods. The graph displays data from this study as well as data from Figure 2 of Smith and Markic 2013.

Table 11. Estimated total abundance of AL in the study streams. Riparian export rates are based on the mean export rate from our seasonal flux study.

	Salt Creek	Turkey Creek	N Br Chicago Riv	Hickory Creek	Plum Creek
Length (m)*	61355	19553	58874	39938	31182
Width (m)	18.6	8.9	15.1	14.9	5.5
Riparian Density (No. m^{-2})	0.452	0.275	0.470	0.236	0.032
Benthic Density (No. m^{-2})	0.088	0.176	0.178	0.099	0.045
Riparian River (No. items)	554,934	107,502	553,215	188,783	20,212
Benthos River (No. items)	100,596	30,545	158,090	58,609	7,795
Total River (No. items)	655,530	138,047	711,304	247,392	28,007
Riparian Mass (g m^{-2})	22.694	172.166	32.453	31.637	33.066
Benthos Mass (g m^{-2})	4.923	63.673	3.379	8.830	2.892
Riparian River (kg)	27,848	67,327	38,213	25,270	20,621
Benthos River (kg)	5,630	11,043	3,008	5,240	500
Total River (kg)	33,478	78,371	41,221	30,510	21,121
Total River (metric ton)	33	78	41	31	21
Riparian Export (No. d^{-1})	2,105	408	2,099	716	77
Riparian Export (No. y^{-1})	768,478	148,870	766,097	261,429	27,989

*Lengths were obtained from U.S. Geological Survey. National Hydrography Dataset high-resolution flowline data. The National Map, accessed March 10, 2015.

Part II. Microplastic concentration and bacterial colonization

Methods *Study sites.* During Jul-Oct 2014, we sampled 8 streams in Chicago metropolitan area and 2 in central Illinois (IL) (N=10 total) that receive treated WWTP effluent. Streams varied in their discharge and relative influence of effluent. The WWTPs were variable in the size of municipalities which they service, volume of treated effluent released, and treatment methods used for filtration and disinfection.

Microplastic collection and quantification. Microplastic was collected from streams with neuston nets (0.52 × 0.36 m) of 333 µm mesh. In the North Shore Channel, we deployed nets behind a stationary boat. All of the other streams were wadeable. At these streams, we stood behind the nets, took care not to disturb the net tail, and held the nets at the water's surface. Deployment time (typically 15-20 minutes) and depth of net submergence were recorded. Water velocity was measured at the center of each net during each deployment (Marsh-McBirney Flo-Mate Model 2000 Portable Flowmeter, Loveland, CO). After 15-20 min, all collected material was rinsed from the net into 1 L plastic containers (N=4 downstream and 4 upstream) with unfiltered site water, and then placed into a cooler on ice for transport to the laboratory where they were stored at 4ºC until measurement of microplastic concentrations.

To collect samples for bacterial measurements, additional net samples were collected downstream. Material from the nets was rinsed onto a sterile white tray. Individual microplastic particles were picked using sterilized forceps and placed in a 160 mL sterile specimen container with ~20 mL of site water. Organic material from the sample was removed in the same fashion. To measure water column bacteria, 2 L of unfiltered site water from the water column at the upstream and downstream sites were collected. The specimen containers and 2 L water column samples were transported on ice to the laboratory where they were stored at 4ºC until processing. Samples for DNA extraction were processed within 72 h, and samples for microplastic counts were processed over 6-8 weeks. Also collected were triplicate, 20 mL filtered water samples (glass microfiber filter; GF/F; Sigma-Aldrich Co., St. Louis, MO) to measure dissolved nutrients at the upstream and downstream sites. Filtered water samples were frozen at -20ºC until solute analyses.

A protocol designed for the quantification of marine samples to measure microplastic concentrations was adapted for this study (Baker et al. 2011, Eriksen et al. 2013). Samples from the net collections were first run through 4.75 mm and 330 µm stacked sieves. The remaining 0.330-4.75 mm fractions were stored in glass beakers in a drying oven at 75 ºC. Organic material was degraded through a wet peroxide oxidation (0.05 M Fe(II) and 30% hydrogen peroxide) at approximately 75ºC. Plastic is resistant to wet peroxide oxidation (Baker et al. 2011, Eriksen et al. 2013). Samples then went through a salinity-based density separation using sodium chloride, where microplastic floated and heavier inorganic material was drained from the sample (Baker et al. 2011). Microplastic was filtered and counted under a dissecting microscope. The microplastic type (i.e., fragment, pellet, foam, film, or fiber) was recorded for each particle in each field of view. All particles of fragments, pellets, foam, and film were counted individually. Due to the abundance of fibers and their tendency to stick to the filter, microplastic fiber particles were counted using a sub-sample approach. For each sample, 3 random subsamples of each quadrat of the filter were counted. Each subsample was 3% of the filter area. The mean value from 12 subsamples was scaled up in proportion to the whole filter to determine microplastic fiber abundance for the sample. Concentration was calculated by dividing the number of particles by

water volume (No. items m^{-3}), or surface area (No. items km^{-2}). All reagents were checked for microplastic contamination, and none was found. Control samples were processed identically to environmental samples to measure procedural contamination (N=5) No microplastic contamination of fragments, pellets, or foam was found. Mean procedural contamination by microplastic fibers was 4.67 per sample, which was subtracted from each environmental sample.

DNA extraction and sequencing. DNA was extracted from microplastic, suspended organic matter, downstream water column, and upstream water column samples using MoBio Powersoil DNA extraction kits (MoBio Laboratories, Carlsbad, CA). For the microplastic and organic matter samples, material collected manually from the net samples was placed into 2 mL microcentrifuge tubes for DNA extraction. For the water column samples, 500 mL of 2 L water samples was filtered using Millipore Sterivex 0.22 µm filter cartridges (N= 4 downstream and 4 upstream). The filters were removed from cartridges, cut with a sterilized razorblade, and placed into 2 mL microcentrifuge tubes for DNA extraction (Crump et al. 2003).

Bacterial assemblages were profiled via next-generation amplicon sequencing of 16S rRNA genes. PCR amplification was performed using primers CS1_515F and CS2_806R, which amplify the V4 hypervariable region of bacterial and archaeal 16S rRNA genes (Caporaso et al. 2011). For all samples, successful DNA amplification was confirmed by agarose gel electrophoresis. Amplicons were sequenced in a paired end format using the Illumina MiSeq platform (Caporaso et al. 2012) by the DNA Services Facility, University of Illinois at Chicago. Sequences were processed by using MOTHUR v.1.33.0 as described (Schloss et al. 2011). Briefly, paired reads were assembled and demultiplexed, and any sequences with ambiguities or homopolymers longer than 8 bases were removed from the data set. Sequences were aligned using the SILVA-compatible alignment database available within MOTHUR. Sequences were trimmed to a uniform length of 293 base pairs and chimeric sequences were removed using Uchime (Edgar et al. 2011). Sequences were classified using the MOTHUR-formatted version of the RDP training set (v.9) and any unknown (i.e., not identified as bacterial), chloroplast, mitochondrial, archaeal and eukaryotic sequences were removed. Sequences were clustered into operational taxonomic units (OTUs) based on 97% sequence identity. In order to avoid biases associated with uneven numbers of sequences across samples, the entire dataset was randomly subsampled to 14,541 sequences per sample.

Data analysis. We used 2-way analysis of variance (ANOVA) to compare differences in total microplastic concentration among streams and between sampling locations (upstream and downstream, or whether WWTP effluent was present). We applied a natural log transformation to ensure concentration data met the assumptions of ANOVA. After applying a Bonferroni correction (α=0.05/9=0.006), we then performed multiple comparison of microplastic concentration at each stream, since there was a significant interaction between stream and sampling location in our 2-way ANOVA. Since the proportion of effluent entering our study streams was variable, we also calculated the ratio of microplastic downstream to upstream. Upstream and downstream samples were independent of one another, so one replicate each from downstream and upstream were randomly paired to calculate the ratios. We performed a 1-way ANOVA on the natural log of this ratio to detect differences among streams. We followed this 1-way ANOVA with Tukey's multiple comparison test. All ANOVAs and Tukey's tests were completed in SYSTAT 13.0 (Systat, Inc. Chicago, IL).

The bacterial assemblages on microplastic, organic matter, upstream water column, and downstream water column samples were compared by calculating the Bray-Curtis similarity index for each pair of samples and visualizing the resulting distance matrix using non-metric multidimensional scaling (nMDS) run within MOTHUR. The statistical significance of differences in assemblages between sample types based on the Bray-Curtis index was assessed by the analysis of molecular variance (AMOVA) run within MOTHUR. Microbial diversity based on observed numbers of OTUs, Chao1 richness, and the inverse Simpson and Shannon-Weiner (H') indices were calculated for each sample using MOTHUR.

Principle findings/results. *Microplastic concentration.* There was a significant interaction between stream and presence of effluent (i.e., upstream versus downstream) (2-way ANOVA, $p<0.001$; Figure 10A). Microplastic concentration was significantly different among streams (2-way ANOVA $p<0.001$; Figure 10A). At all sites microplastic concentration downstream of the WWTP effluent was higher than upstream (except Little Kickapoo and Goose Creeks; Figure 10A). The only stream that had significant differences in downstream and upstream microplastic concentration with a Bonferroni Correction applied was Higgen's Creek (Figure 10A). There were significant differences in the ratios of downstream to upstream microplastic concentration (Figure 10B). Sampling methodologies may explain the pattern observed at Goose Creek. The upstream sampling location in this creek had a lower discharge compared than all other upstream sites, except Little Kickapoo and Springbrook Creeks. The water upstream at Goose Creek was also very shallow, so that only one-third of the net was submerged. This resulted in a low volume of water being collected, and thus a low number of microplastic particles generated a potentially artificially-high concentration.

Bacterial communities on microplastic. There were distinct differences in the bacterial assemblages among sample types (Figure 11). Bray-Curtix index scores were significantly differen when comparing all sample types (AMOVA, p value <0.001) and when comparing any one category to another (Table 12). To complete this part of our analysis, we will use one-way ANOVA to assess the effects of sample type on microbial diversity, which we will follow Tukey's multiple comparison test. <u>During March 15-31, 2015, we will be identifying bacterial taxa genera which make the largest contributions to the dissimilarities between sample types</u> (based on the Bray-Curtis index) with a SIMPER analysis run in Primer 6 (Primer-E Ltd., Plymouth, United Kingdom). Multiple analyses are ongoing, and take long processing time due to the high number of microbial sequences in each sample. We will compare microplastic to organic material, microplastic to non-plastic downstream substrates, and microplastic to the upstream water column. We will compare our results to previous work which identified *Pseudomonas* as a common bacteria genus on plastic (McCormick et al. 2014). Initial results suggest that variation in community composition within microplastic samples may be greater than the within variation of other substrates. We will also analyze whether the dominant bacteria genera on microplastic differ among sites.

Conclusion. Overall, our results suggest that WWTP effluent is a point source of microplastic to rivers. Furthermore, this research and previous studies (McCormick et al. 2014, Zettler et al. 2013, Harrison et al. 2014) demonstrate that microplastic supports unique bacterial communities in comparison to natural substrates. <u>The plastic may provide a novel habitat for bacteria, or if taxa colonizing microplastic have plastic-degrading metabolic capabilities, microplastic may</u>

provide a novel carbon source (McCormick et al. 2014). Research on the ecological impacts of microplastic in freshwater environments is lacking in comparison to marine ecosystems. This research provides a foundation for future studies analyzing biofilm activity on microplastic and its effect on higher trophic levels.

Part II: Tables and Figures

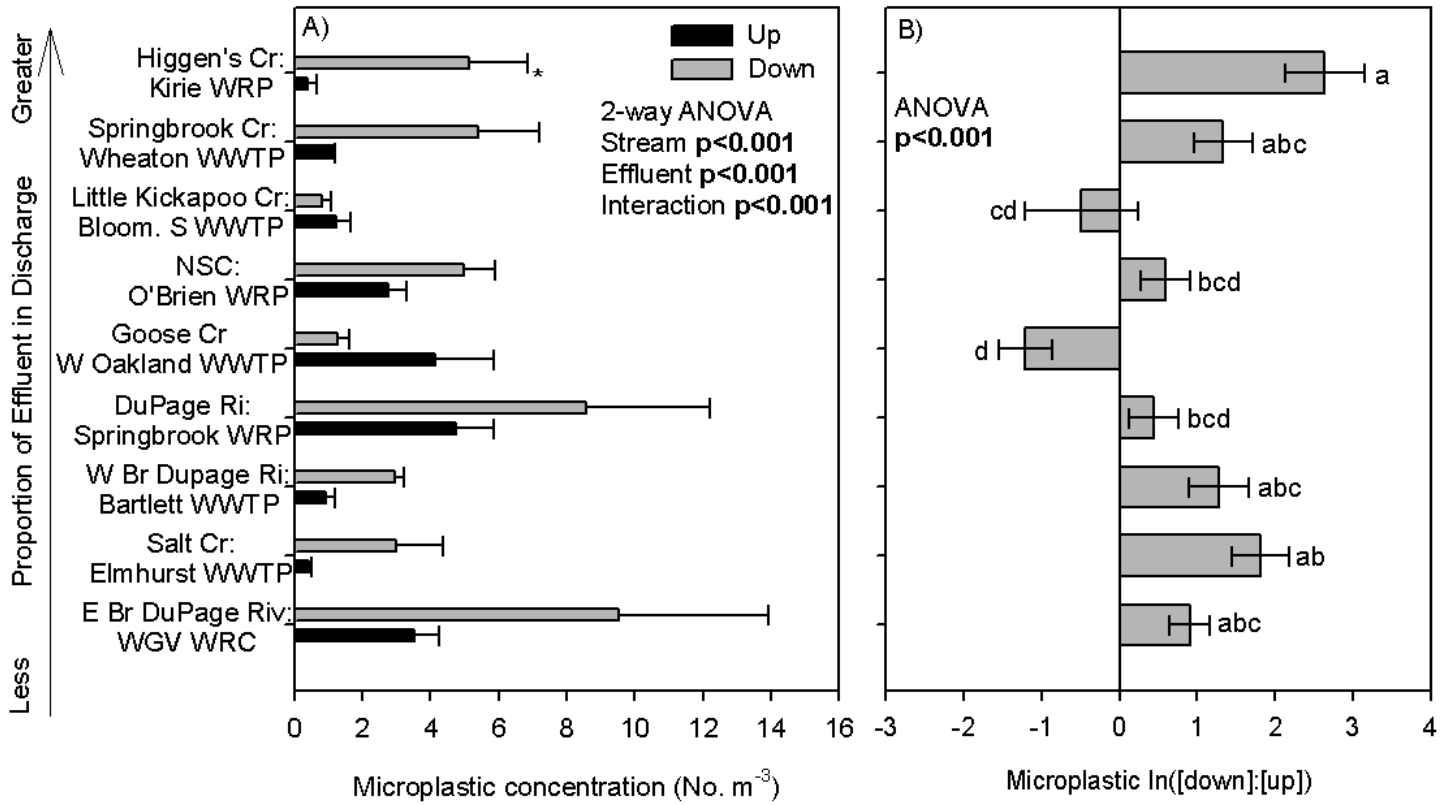

Figure 10. A) Microplastic concentration upstream and downstream B) ratio of microplastic concentrations downstream and upstream at each of our study streams. Bars represent mean (SE). * indicates significant difference in downstream and upstream concentrations with a Bonferroni Correction. Letter's represent Tukey's test results.

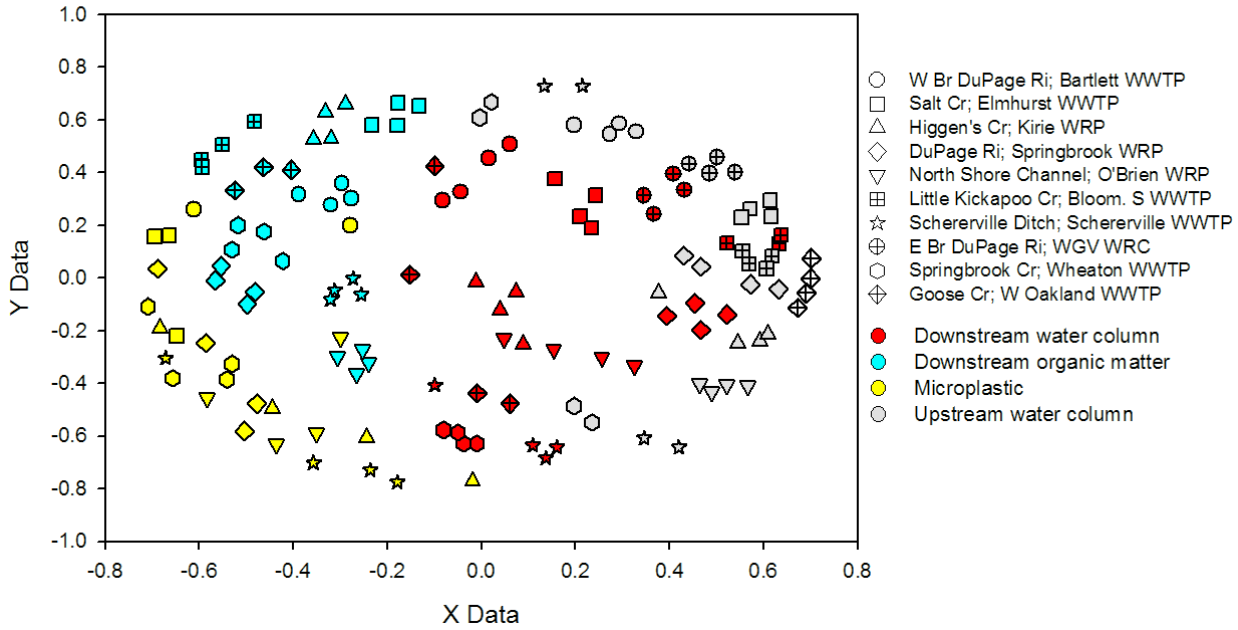

Figure 11. nMDS ordination of 16S sequencing data (Bray-Curtis dissimilarity) comparing community structures of bacteria collected in our 10 study sites.

Table 12. Differences in bacterial community composition based on a comparison of the Bray-Curtis dissimilarity index for 4 sample types (U=upstream water column; D=downstream water column; O=organic material; P = microplastic). P-values were calculated using the AMOVA run within MOTHUR

Comparison	p-value
All sample types	<0.001
D-O	<0.001
D-P	<0.001
D-U	<0.001
O-P	<0.001
O-U	<0.001
P-U	<0.001

Student information

1. Amanda McCormick, *MS student, Loyola University Chicago*
Ms. McCormick worked on this project as her MS thesis (2013-2015) at Loyola. She is currently writing together the thesis which will be published in two manuscripts. Her anticipated date of graduate with the MS is August 2015.

2. Joshua Hittie, *undergraduate student, Loyola University Chicago*
Mr. Hittie assisted in collecting AL and microplastic samples during the summer of 2014. He also assisted in categorizing AL and processing microplastic samples in the lab. He is a senior and will graduate in May 2015.

3. Melaney Dunne *undergraduate student, Loyola University Chicago*
Ms. Dunne assisted in processing microplastic samples. She is a senior and will graduate in May 2015.

Communication of results

A. Publication: We plan to publish this research in 2 peer-reviewed publications with submission in summer 2015.

1. The first paper will consist of the research described in Part 1 above (AL abundance, composition, and flux

2. The second paper will include the data in Part 2: microplastic concentration and microbial colonization research.

B. Presentations:

Completed

1. McCormick, A. and T.J. Hoellein. Oct 15, 2014. Anthropogenic litter and microplastic in urban streams: abundance, source, and fate. Illinois Water Conference. Champaign, IL

Upcoming

2. The microplastic research will also be presented in an oral presentation by Amanda McCormick at the annual Society for Freshwater Science meeting in Milwaukee, WI in May 2015.

3. The AL density and flux data will be presented in an oral presentation by Timothy Hoellein at the annual Society for Freshwater Science meeting in Milwaukee, WI in May 2015.

4. Data on microplastic abundance and microbial community composition will be presented at Loyola University Chicago, Weekend of Excellence (dedicated to undergraduate research) by Joshua Hittie.

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Appendix Table 1. Classification of AL by material and item type.

Material	Code	Litter form (examples)
Ceramic	CE01	Construction material (brick, cement, pipes)
Ceramic	CE02	Bottles & Jars
Ceramic	CE03	Ceramic fragments
Ceramic	CE04	Other (specify)
Cigarettes	CG01	Cigarettes, butts & filters
Cloth	CL01	Clothing, shoes, hats & towels
Cloth	CL02	Backpacks & bags
Cloth	CL03	Canvas, sailcloth & sacking
Cloth	CL04	Rope & string
Cloth	CL05	Carpet & furnishing
Cloth	CL06	Other cloth (including rags)
Glass	GL01	Bottles & jars
Glass	GL02	Tableware (plates & cups)
Glass	GL03	Light bulbs
Glass	GL04	Fluorescent light tubes
Glass	GL05	Glass buoys
Glass	GL06	Glass fragments
Glass	GL07	Other
Metal	ME01	Tableware (plates, cups & cutlery)
Metal	ME02	Bottle caps, lids & pull tabs
Metal	ME03	Aluminum drink cans
Metal	ME04	Other cans (< 4 L)
Metal	ME05	Gas bottles, drums & buckets (> 4 L)
Metal	ME06	Foil wrappers
Metal	ME07	Fishing related (sinkers, lures, hooks, traps & pots)
Metal	ME08	Fragments
Metal	ME09	Wire, wire mesh & barbed wire
Metal	ME10	Other, including appliances
Paper & Cardboard	PC01	Paper (including newspapers & magazines)
Paper & Cardboard	PC02	Cardboard boxes & fragments
Paper & Cardboard	PC03	Cups, food trays, food wrappers, cigarette packs
Paper & Cardboard	PC04	Tubes for fireworks
Paper & Cardboard	PC05	Other
Plastic	PL01	Bottle caps & lids
Plastic	PL02	Bottles < 2 L
Plastic	PL03	Bottles, drums, jerrycans & buckets > 2 L
Plastic	PL04	Knives, forks, spoons, straws, stirrers, (cutlery)
Plastic	PL05	Drink package rings, six-pack rings, ring carriers

Plastic	PL06	Food containers and wrappers
Plastic	PL07	Plastic bags (opaque & clear)
Plastic	PL08	Toys
Plastic	PL09	Gloves
Plastic	PL10	Cigarette lighters
Plastic	PL12	Syringes
Plastic	PL13	Baskets, crates & trays
Plastic	PL14	Plastic buoys
Plastic	PL15	Mesh bags (vegetable, nets, bags)
Plastic	PL16	Sheeting (tarp or woven plastic bags, palette wrap)
Plastic	PL17	Fishing gear (lures)
Plastic	PL18	Monofilament line
Plastic	PL19	Rope
Plastic	PL20	Fishing net
Plastic	PL21	Strapping
Plastic	PL22	Fibreglass fragments
Plastic	PL23	Resin pellets
Plastic	PL24	Other
Rubber	RB01	Balloons, balls & toys
Rubber	RB02	Footwear (flip-flops)
Rubber	RB03	Gloves
Rubber	RB04	Tires
Rubber	RB05	Inner-tubes and rubber sheet
Rubber	RB06	Rubber bands
Rubber	RB07	Condoms
Rubber	RB08	Other
Styrofoam	FP01	Foam sponge
Styrofoam	FP02	Cups & food packs
Styrofoam	FP03	Foam buoys
Styrofoam	FP04	Insulation & packaging
Styrofoam	FP05	Other
Wood	WD01	Corks
Wood	WD02	Fishing traps and pots
Wood	WD03	Ice-cream sticks, chopsticks & toothpicks
Wood	WD04	Processed timber and pallet crates
Wood	WD05	Matches & fireworks
Wood	WD06	Other
Other	OT01	Paraffin or wax
Other	OT02	Sanitary (diapers, cotton buds, feminine hygiene)
Other	OT03	Appliances & Electronics
Other	OT04	Batteries

Other	OT05	Other

Statewide surveillance of emerging flame retardant contamination in Illinois waters via fish monitoring

Basic Information

Title:	Statewide surveillance of emerging flame retardant contamination in Illinois waters via fish monitoring
Project Number:	2014IL287B
Start Date:	3/1/2014
End Date:	2/28/2015
Funding Source:	104B
Congressional District:	IL-013
Research Category:	Water Quality
Focus Category:	Water Quality, Toxic Substances, None
Descriptors:	None
Principal Investigators:	Da Chen

Publications

There are no publications.

Project Title

Statewide Surveillance of Emerging Flame Retardant Contamination in Illinois Waters via Fish Monitoring

Principle Investigator

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Problem and Research Objectives

Human activities generate large amounts of a wide range of pollutants. These are subsequently released into water systems via point or non-point discharges. Some may impair water quality, aquatic ecosystem and human health. Current efforts of monitoring anthropogenic pollution in Illinois have mainly focused on legacy pollutants such as heavy metals, pesticides, and polychlorinated biphenyls (PCBs). There is a lack of information on the distribution of emerging contaminants, such as flame retardants (FRs), in Illinois aquatic environments. Flame retardants are of high concern as some of them are identified as High Production Volume (HPV) substances and have demonstrated environmental persistence, bioaccumulative and toxic potentials. They may impair aquatic ecosystems via bioaccumulation and biomagnification to reach high levels in aquatic species (e.g. fish and birds). The bioaccumulation in edible fish also represents potential risks to human health. However, to date nothing has been done to assess the contamination of FRs in Illinois waters and organisms on a state scale.

This project aimed to investigate the spatial distribution of BFRs in rivers and lakes across the State of Illinois via fish monitoring. The findings would reveal the status of FR contamination and associated risks to ecosystems in Illinois waters. Specific research objectives of this project were to: (1) determine priority FR contaminants and their levels in Illinois fish; (2) assess distribution patterns of FRs in Illinois waters; and, (3) identify impaired aquatic systems.

Methodology

Samples and study sites. Fish samples utilized by this project were retrieved from the Illinois Fish Contaminant Monitoring Program (FCMP). Common carp (*Cyprinus carpio*) and channel catfish (*Ictalurus punctatus*) were selected as the bio-monitoring species for this project. They were collected from a total of 35 selective aquatic systems across the state in 2011-2013 (Figure 1). These sites cover a number of ecologically or economically important aquatic systems and represent different water use types, including potential hotspots, unimpaired/impaired urban streams and lakes, and unimpaired /impaired rural/agricultural steams and lakes. At each site, a minimum of

three (and preferably five) fish, with the smallest fish being at least 75% of the length of the largest fish, were collected. Fillet tissues were removed from individual fish and then combined to produce a composite fillet sample by species. A total of 42 carp composite and 31 catfish composite samples were analyzed for flame retardants.

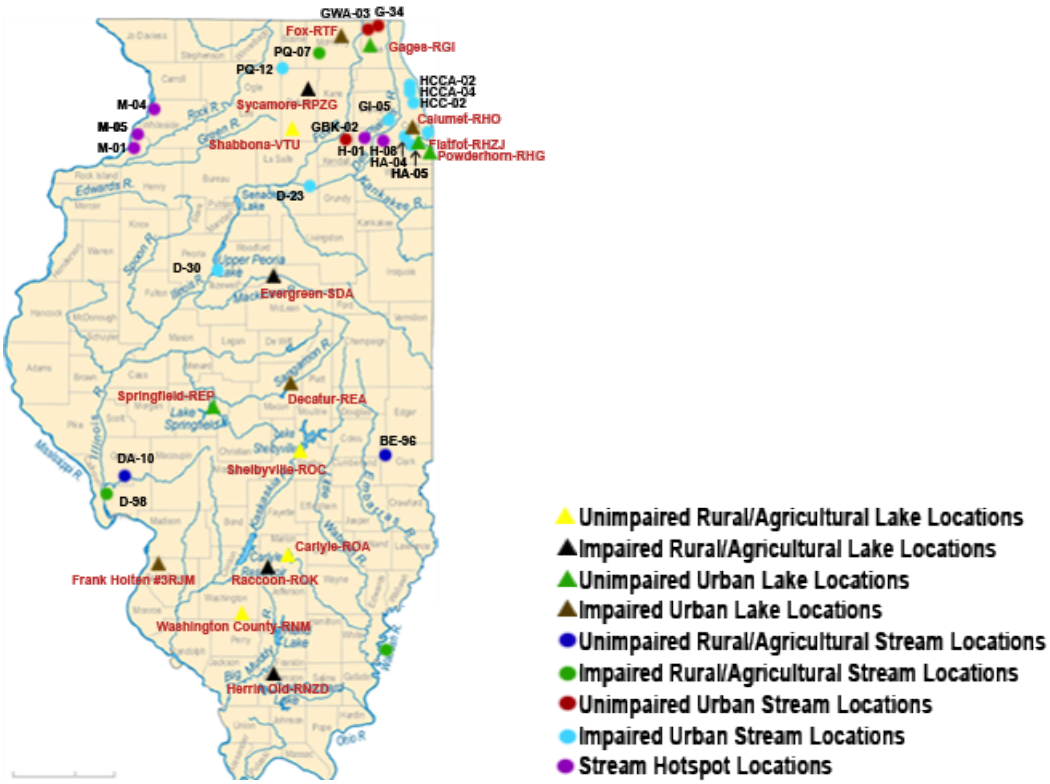

Figure 1. Station sites where fish samples were collected and analyzed in this project.

Residue analysis. Chemical residue analysis followed the protocols shown in Figure 2 (Chen et al., 2012). In brief, composite fish (2-3 gram) was ground with diatomaceous earth, spiked with internal standards, and then subject to accelerated solvent extraction. The extract was purified by gel permeation chromatography, followed by solid phase extraction cleanup. Final extract was analyzed on gas chromatography – mass spectrometry (GC-MS). A total of 20 polybrominated diphenyl ether (PBDE) congeners, 27 non-PBDE BFRs, and 19 DP-related compounds were determined in this project (Table 1).

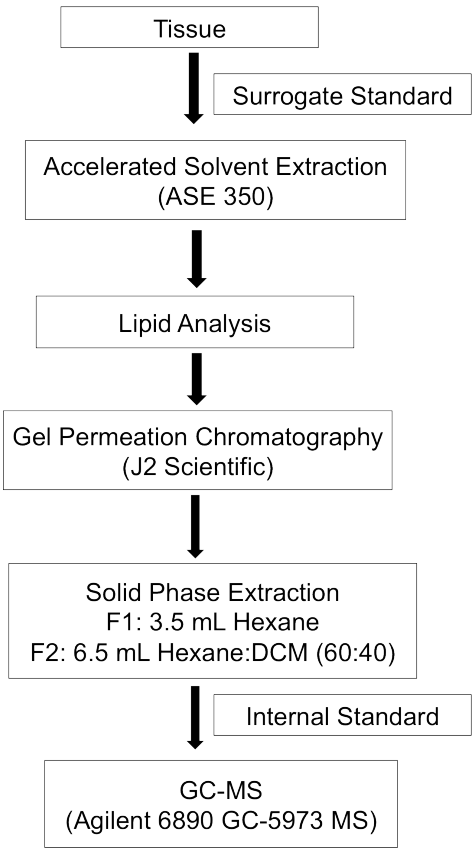

Figure 2. Flow chart of analytical methodology

Principle Findings and Significance

1. PBDEs are the main flame retardants in Illinois waters.

The data from this project revealed that PBDE compounds were the most abundant flame retardant pollutants in Illinois water. PBDE levels in fish were up to 8200 ng/g lipid weight (lw), comprising more than 98% of total FR concentrations in fish from all sites. Hexabromocyclododecane (HBCD) and dechlorane plus (DP) were also frequently detected, but at much lower concentrations (i.e. generally less than 100 ng/g lw). Other flame retardant analytes were generally below detection limits or detected at very low frequencies (i.e. < 4%). Although commercial PBDE products have been discontinued in 2004, the study revealed that PBDEs are still the main flame retardants present in Illinois aquatic environments. Their high concentrations in fish from some sites merit continuing monitoring.

2. Wide distribution of PBDEs in Illinois waters.

Although the levels differed between sites, PBDEs were detected in both species from all study sites, revealing a statewide contamination. PBDEs have been used for decades in a variety of consumer products. They can be released into the environment via multiple

pathways and persist in the environment for a long time. They are bioavailable to organisms, resulting in wide contamination in fishes. This is consistent with other studies in the US or the rest of the world, demonstrating a universal contamination of PBDEs in aquatic organisms (Law et al., 2014).

Table 1. Summary of target analytes

PBDEs	Non-PBDE BFRs	DP-related compounds
BDE-28	2,4,6-tribromophenyl allyl ether	Di-dechlorinated DP
BDE-47	2,2',4,5,5'-pentabromobipheny	Mono-dechlorinated DP
BDE-49	2,2',4,4'5,5'-hexabromobiphenyl	Chlordene Plus
BDE-66	Bis(2-ethyl-1-hexyl)tetrabromophthalate	Dibromoaldrin
BDE-85	1,2-bis-(2,4,6-tribromophenoxy)ethane	Dechlorane Plus Mono Adduct
BDE-99	decabromodiphenyl ethane	DEC-601
BDE-100	2,3,-dibromopropyl tribromophenyl ether	DEC-602
BDE-138	ethylenebistetrabromophthalimide	DEC-603
BDE-153	2-ethylhexyl-2,3,4,5-tetrabromobenzoate	DEC-604
BDE-154	hexabromobenzene	DEC-604CB
BDE-183	hexabromocyclododecane	Br-DEC604
BDE-196	Hexachlorocyclo-pentadiene	Br_2-DEC604
BDE-197	Hexachlorocyclopentadienyl-dibromocyclooctane	Cl_4-DEC604
BDE-201	Pentabromobenzyl acrylate	Br_2Cl_2-DEC604
BDE-202	pentabromobenzyl bromide	Hexachloro(phenyl)norbornene
BDE-203	Pentabromobenzene	VCH-dechlorane plus
BDE-206	pentabromoethylbenzene	syn-dechlorane plus
BDE-207	Pentabromochlorocyclohexane	anti-dechlorane plus
BDE-208	Pentabromophenyl allyl ether	
BDE-209	2,3-dibromopropyl pentabromophenyl ether	
	Pentabromotoluene	
	TetraBromo-p-Xylene	
	1,3,5-Tribromobenzene	
	1,2,5,6-tetrabromocyclooctane (alpha-TBCO)	
	1,2-dibromo-4-(1,2-dibromoethyl)cyclohexane	
	tetrabromo-o-chlorotoluene	
	2,3,5,6-tetrabromo-p-xylene	

3. PBDE levels differed between water use types.

In this project, our study sites include multiple water use types, i.e. potential hotspots, unimpaired/impaired urban streams and lakes, and unimpaired /impaired rural/agricultural steams and lakes. The data indicated statistically significant differences in PBDE levels in fish from different water use types (Analysis of Variance or ANOVA, $p < 0.001$) (Figure 3). Fish from stream hotspot sites (N = 5) and impaired urban lakes/streams (N = 12) contained much greater PBDE concentrations than those from unimpaired rural/agricultural/urban lakes (N = 14) and from impaired rural/agricultural lakes (N = 4). For example, common carp and channel catfish from stream hotspots

contained median ΣPBDE concentrations of 3,210 and 2,950 ng/g lw, respectively. These two species contained median ΣPBDE concentrations of 2,174 and 2,910 ng/g lw, respectively, at impaired urban lakes and streams. Fish from impaired rural/agricultural lakes or unimpaired waters contained median concentrations of generally lower than 70 ng/g lw.

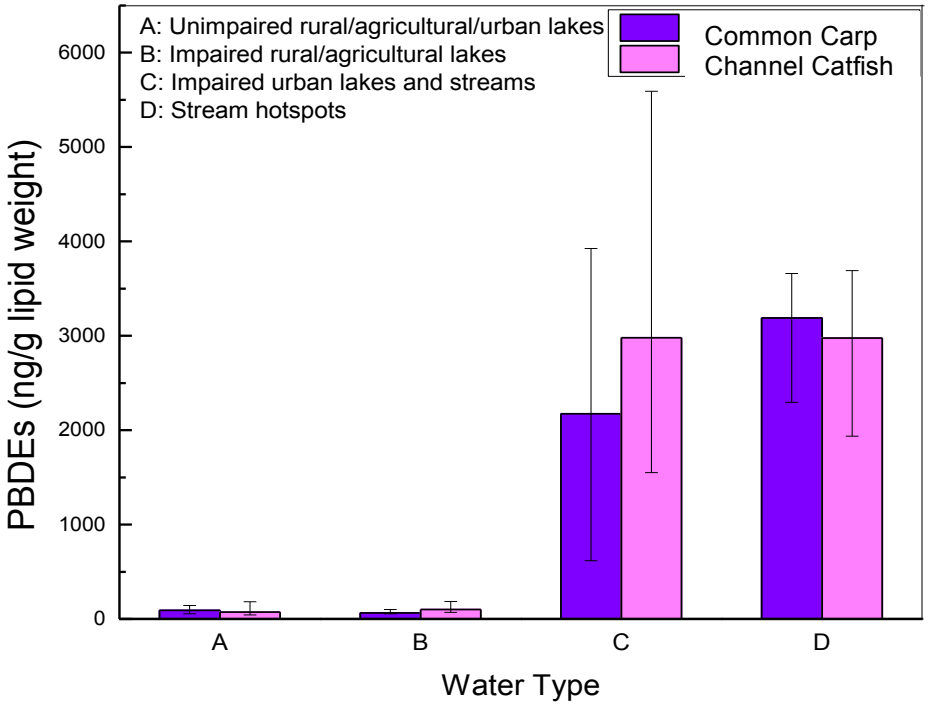

Figure 3. Median PBDE concentrations (ng/g lw) in fish from different water use types. Error bars represent 75 and 25 percentiles, respectively.

The elevated PBDE contamination in urban waters was likely due to greater human population densities in urban versus agricultural/rural areas. A greater human population density may be associated with a more abundance of consumer products (e.g. thermoplastics, textiles, and electronics) that contain PBDEs, which consequently resulted in an elevated release of PBDEs to the environment. Typical sources of PBDEs in urban waters may include wastewater effluents, surface runoff, or industrial releases.

High contamination in fish from some impaired urban rivers, such as the sites at the Cal-Sag Channel at Rt. 83 and the Little Calumet River, may indicate potential sources nearby. The future work is needed to identify sources and transport pathways in order to prevent or reduce the contamination. High levels in fish may also represent potential human health risks via the consumption of edible fishes collected from these rivers. Currently there is a lack of fish consumption advisory for PBDEs in Illinois.

4. Comparison with international studies

A comparison with international studies revealed that PBDE contamination in fish from impaired Illinois urban waters (including hotspots and impaired urban lakes/streams) was at the higher end of the range of concentrations (Table 2).

Table 2. Reported PBDE concentrations (ng/g lw) in carp from this study and studies from other regions.

Watershed	Country	PBDE Concentration Range and Median (in parentheses)	Reference
Impaired IL urban waters	US	642-8200 (2350)	This study
Hyco River, NC	US	1500-12900 (9140)	Chen et al. 2011
Dan River, NC	US	310-1900 (1120)	Chen et al. 2011
Roanoke River, NC	US	160-1300 (440)	Chen et a. 2011
Hardley Lake	US	760-2500 (1600)	Dodder et al. 2002
Detroit River	US	29-50.2 (40.7)	Rice et al. 2002
Gila River, AZ	US	0.2-108 (24.9)	Echols et al. 2013
Lake Erie	US	1.5-101.8 (11)	Perez-Fuentetaja et al. 2010
Canal Lake	Belgium	Non-detection-420 (140)	Covaci et al. 2005
Anoia and Cardener River	Spain	29-744 (43)	Labanderia et al. 2007
Tai Lake	China	1.1-97.6 (23.1)	Su et al. 2014
Dongjiang River	China	42-263 (115)	He et al. 2012
Yangtze River	China	17-1100 (140)	Xian et al. 2008

5. PBDE congener profiles were consistent across sites.

Although PBDE concentrations differed between water use types, the BDE congener compositions (i.e. percentage of each congener to the total PBDEs in concentration) were consistent across sites. The congener compositions were dominated by BDE-47, followed by BDE-100, -99, -154, and -153 (Figure 4). This represents a typical pattern resulting from the PentaBDE contamination (Law et al., 2014). The U.S. has consumed the majority of the world's PentaBDE production. Major congeners in the commercial PentaBDE products, such as BDE-47 and -99, are highly bioaccumulable compared to other congeners. Therefore, BDE-47 dominates the PBDE contamination patterns in most aquatic organisms.

5. Further studies are needed to elucidate non-PBDE flame retardant contamination in Illinois waters.

Our data revealed that PBDEs were still the major flame retardants in Illinois fishes and that non-PBDE flame retardants were not detectable or detected at very low frequencies. However, given that commercial PBDE mixtures have been phased out from the North American market, many alternative flame retardant chemicals are used to replace PBDEs in a variety of consumer products (Covaci et al., 2010). Some of them have already demonstrated environmental persistence and bioaccumulation (Covaci et al., 2010).

Future studies are needed to monitor the contamination and distribution of these alternative flame retardants in Illinois aquatic ecosystems.

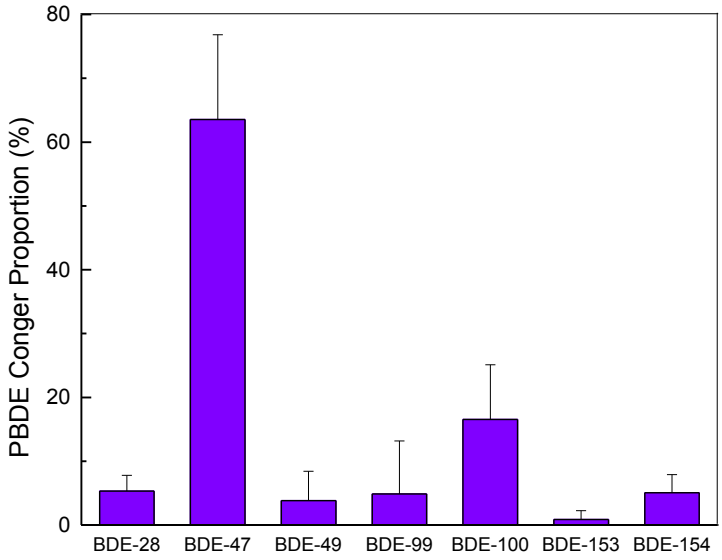

Figure 4. PBDE conger compositions in Illinois fish. Error bars represent standard deviations.

Notable Achievements

(1) The project revealed PBDEs are the most abundant flame retardant pollutants in Illinois waters. Although commercial PBDE products have been discontinued in 2004, their high environmental concentrations merit continuing monitoring. The project also raised the need of monitoring HBCD and DP, as well as many other non-PBDE flame retardants in Illinois fishes. These emerging flame retardants are subject to increased usage in order to replace PBDEs in many applications, which may result in increasing concentrations over time.

(2) The project mapped the spatial contamination pattern of flame retardants in Illinois waters. It developed a baseline for understanding flame retardant contamination in Illinois aquatic systems and identified contamination hotspots. The levels in fish collected from impaired urban waters are in the higher end of the range of concentrations reported in international studies. Potential human exposure and heath risks via edible fishes from impaired urban waters merit further investigations. The findings would essentially contribute to the database of water pollution developed by the Illinois EPA Fish Contaminant Monitoring Program and other agencies.

Students Supported With Funding

Malgorzata Widelka, Department of Zoology, Southern Illinois University, Bachelor of Science (awarded in May 2014).

Hillary Marler, Department of Zoology, Southern Illinois University, doctoral student (expected to be awarded in December 2017).

Publications and Presentations

Widelka, M., H. Marler, M. Lydy, T. Hornshaw, D. Chen. 2014. Statewide Surveillance of emerging flame retardant contamination in Illinois waters via fish monitoring. Illinois Water Conference, Urbana, Illinois, October 14-15.

Widelka, M., H. Marler, M. Lydy, T. Hornshaw, D. Chen, 2014, Statewide Surveillance of emerging flame retardant contamination in Illinois waters via fish monitoring, Southern Illinois University Carbondale Annual Undergraduate Research and Creative Activities Forum, Carbondale, IL, April 7.

Widelka, M., H. Marler, M. Lydy, T. Hornshaw, D. Chen, 2013, Statewide Surveillance of emerging flame retardant contamination in Illinois waters via fish monitoring, The 34th Society of Environmental Toxicology and Chemistry (SETAC) North American Annual Meeting, Nashville, TN, November 17-21.

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Restoration trajectories of restored floodplain wetlands and their ecosystem services

Basic Information

Title:	Restoration trajectories of restored floodplain wetlands and their ecosystem services
Project Number:	2014IL288B
Start Date:	3/1/2014
End Date:	2/28/2015
Funding Source:	104B
Congressional District:	IL-013
Research Category:	Water Quality
Focus Category:	Wetlands, Water Quality, None
Descriptors:	None
Principal Investigators:	Angela Kent

Publications

There are no publications.

Problem and research objectives

Wetlands are an important ecosystem for the biological removal of nitrate by denitrification. Wetland mitigation is a common practice used to satisfy the "no net loss" requirements of the Clean Water Act, but monitoring is rarely carried out to determine whether function is adequately replaced by restored wetlands [1]. Restored wetlands often do not achieve similar rates of denitrification as compared to more pristine wetlands [2, 3]. Fluctuating anaerobic hydrologic conditions are known to favor denitrification activity because it is an anaerobic process, but simply restoring the hydrology does not always restore denitrification capacity as expected [4]. Since this is a process performed by microbial communities, it is possible that there is an overlooked biological component. It is essential that we investigate how microbial communities differ between restored and natural wetlands and whether this influences function in order to better understand the ability of restored wetlands to perform desired functions. The objectives of this project aim (1) to show how denitrifying communities in restored wetlands can change over time, (2) to compare denitrifier composition and denitrification capacity across a chronosequence of restored wetlands, and (3) to identify environmental factors that may facilitate the restoration of denitrifying services by affecting the microbial organisms involved.

Methods

Soil was collected from 45 wetlands located across the state of Illinois, including 30 restored wetlands, 15 of which were paired with adjacent reference wetlands. Samples were collected both during a drought in 2012 and again in 2013, which was a much wetter year. Four paired restored and reference wetlands were also analyzed during 2007, so a historical comparison could be made for a subset of the sites, and three of these paired sites were revisited in 2014 to continue the time series. Potential denitrification rates were determined in the lab by an acetylene reduction assay. Changes in denitrifier composition were determined by performing terminal restriction fragment length polymorphism (T-RFLP) of the nitrous oxide reductase (*nosZ*) gene in bacteria [5]. Redundancy analysis (RDA) was used to identify a core set of denitrifier populations that represent natural reference wetlands. A suite of environmental data was collected simultaneously, including total soil N and C, available N and P, pH, and moisture content, which was combined with information about plant community characteristics determined by our collaborators.

Principle findings

In order to determine whether denitrifying microbial communities are becoming more similar to a reference wetland community, denitrifer communities from four different restored wetlands were compared to a core reference community that was identified using redundancy analysis (RDA). These four sites were selected because they were sampled in 2007, 2012, and 2013, and three of them were sampled again in 2014. In general, there is a non-significant trend showing decreasing dissimilarity of denitrifier assemblages in restored wetlands to this core reference community (as determined by the Bray Curtis dissimilarity index) as the restored wetlands age, which indicates that the communities are becoming more similar to the core reference community over time (Fig. 1). Two of the sites showed a significant decrease in dissimilarity: Site 028 in Stephenson County ($p < 0.05$) and Site 029 in Sangamon County ($p<0.01$).

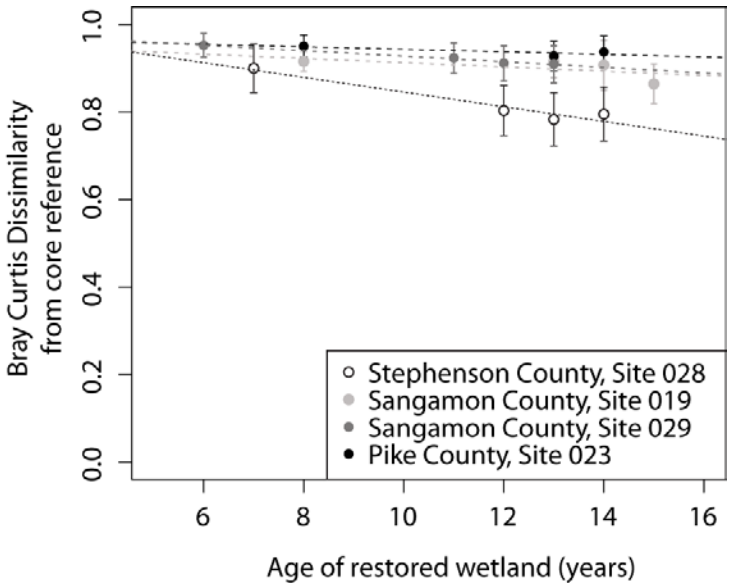

Figure 1. Denitrifier community dissimilarity between restored sites and a core reference community determined by RDA. Significant trends (regression significance at $p < 0.05$; indicating increasing similarity) were seen for Site 028 in Stephenson County (white) and for Site 029 in Sangamon County (dark grey), but not for the other two sites. Error bars represent the 95% confidence interval.

Contrary to what has been found previously, there was no significant difference between the denitrification capacities of restored and reference wetlands in either 2012 or 2013 (Fig. 2), though the drought in 2012 significantly limited denitrification capacity for both. Thus, chronosequence analysis was performed only for the wetlands sampled in 2013 to avoid confounding factors related to the drought. The denitrifier communities in the restored sites were very different from the core reference community (Bray Curtis dissimilarity = 0.87 ± 0.02), and communities in older restoration projects were no more similar to the reference than those in younger restored wetlands (Fig. 3a). However, denitrification showed a non-significant increasing trend as the age of the restoration increased (Fig. 3b), which may indicate a rapid response to the development of wetland environmental factors, while the microbial community shows a lag in development.

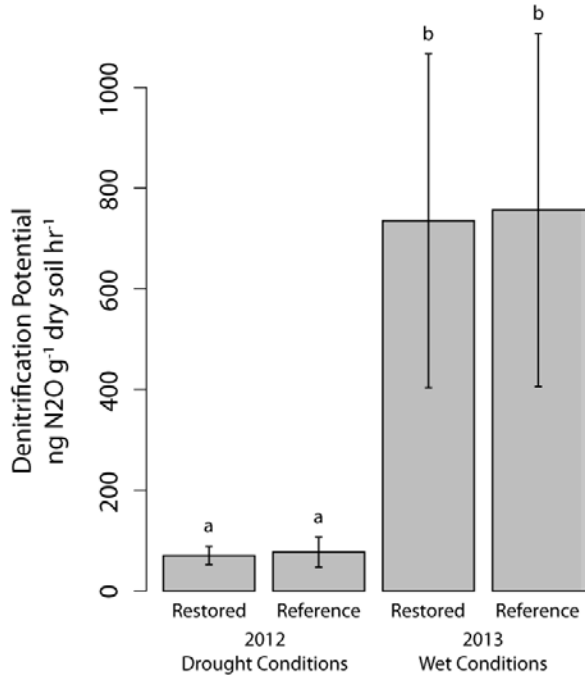

Figure 2. Average denitrification capacity for restored and reference wetlands under drought conditions and under more normal conditions. Error bars represent the 95% confidence interval.

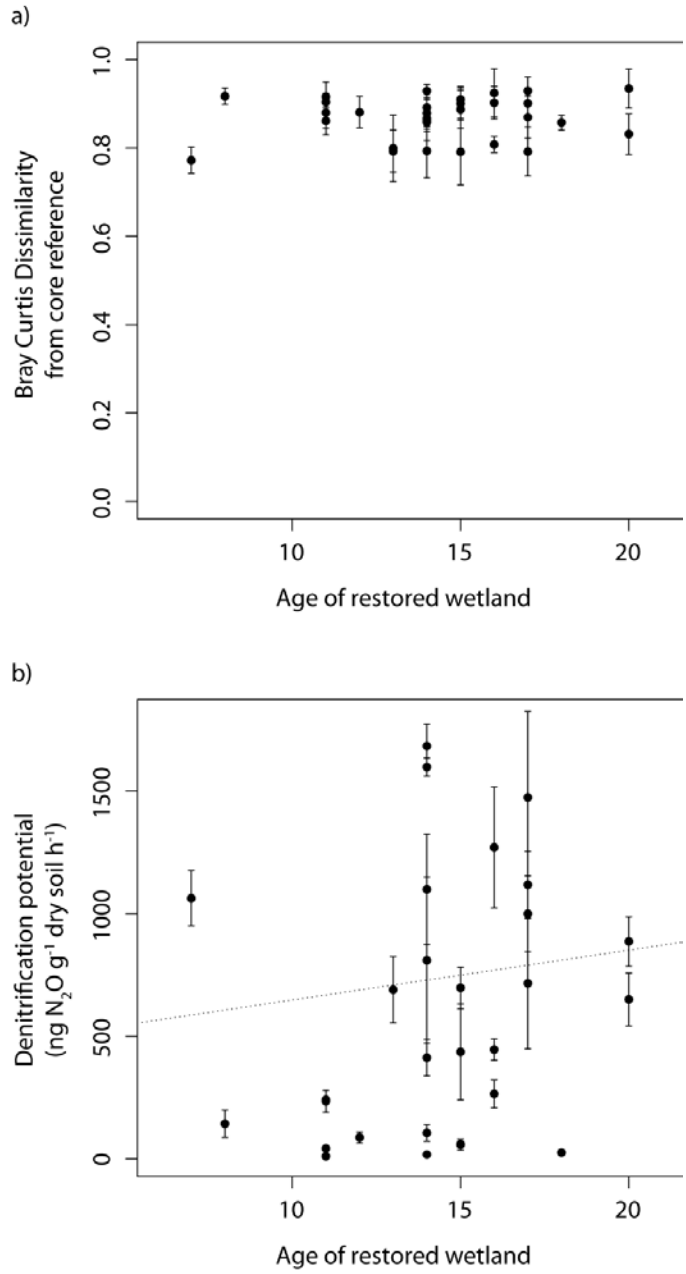

Figure 3. (a) Dissimilarity between denitrifier communities in a chronosequence of restored wetlands compared to a reference core, and (b) denitrification potential of each restored site plotted by the age of the restoration in 2013 only. There was no trend in denitrifier community over time, and the dotted line in panel b shows a non-significant upward trend in denitrification rates as the restored sites get older.

In order to determine which factors influence denitrifier composition, permutational analysis of variance (PERMANOVA) models were built using soil factors and plant factors separately, which was visualized by correspondence analysis (CA) plots (Fig. 4).

The factors influencing communities in restored and reference wetlands were different, indicating that the denitrifiers are responding to different drivers in these systems. The soil factors with the most influence on denitrifier communities in the restored sites were moisture ($R^2 = 0.09$, $p < 0.001$), pH ($R^2 = 0.07$, $p < 0.001$), and total N ($R^2 = 0.02$, $p < 0.01$) and the plant factors with the most influence were mean coefficient of conservatisms (mean C; $R^2 = 0.06$, $p < 0.001$), floristic quality index (FQI; $R^2 = 0.03$, $p < 0.001$), and the percent of perennial species present ($R^2 = 0.03$, $p < 0.01$). Soil factors with the greatest influence on the communities in the reference sites were pH ($R^2 = 0.12$, $p < 0.001$), nitrate ($R^2 = 0.03$, $p < 0.001$), and moisture ($R^2 = 0.03$, $p < 0.001$), while the plant factors with the greatest influence were the percent native species present ($R^2 = 0.10$, $p < 0.001$), mean C ($R^2 = 0.03$, $p < 0.01$), and FQI ($R^2 = 0.03$, $p < 0.01$).

The factors that show up as important most often seem to be moisture and pH, as well as FQI and mean C, most of which were correlated to denitrification (Fig. 5). However, moisture was not a significant explanatory variable for potential denitrification, presumably because the laboratory assays are provided with adequate water regardless of the moisture content of the soil, while available phosphate was the most important factor, even though it did not appear to affect the denitrifier composition at all. The mismatch between factors that influence denitrification rates and those that influence community

composition further support the idea that denitrifier communities experience a lag in development following restoration. Put together, our results show that denitrifying services might not always attain restoration goals if factors influencing the development of microbial communities, like soil pH and moisture, are not addressed.

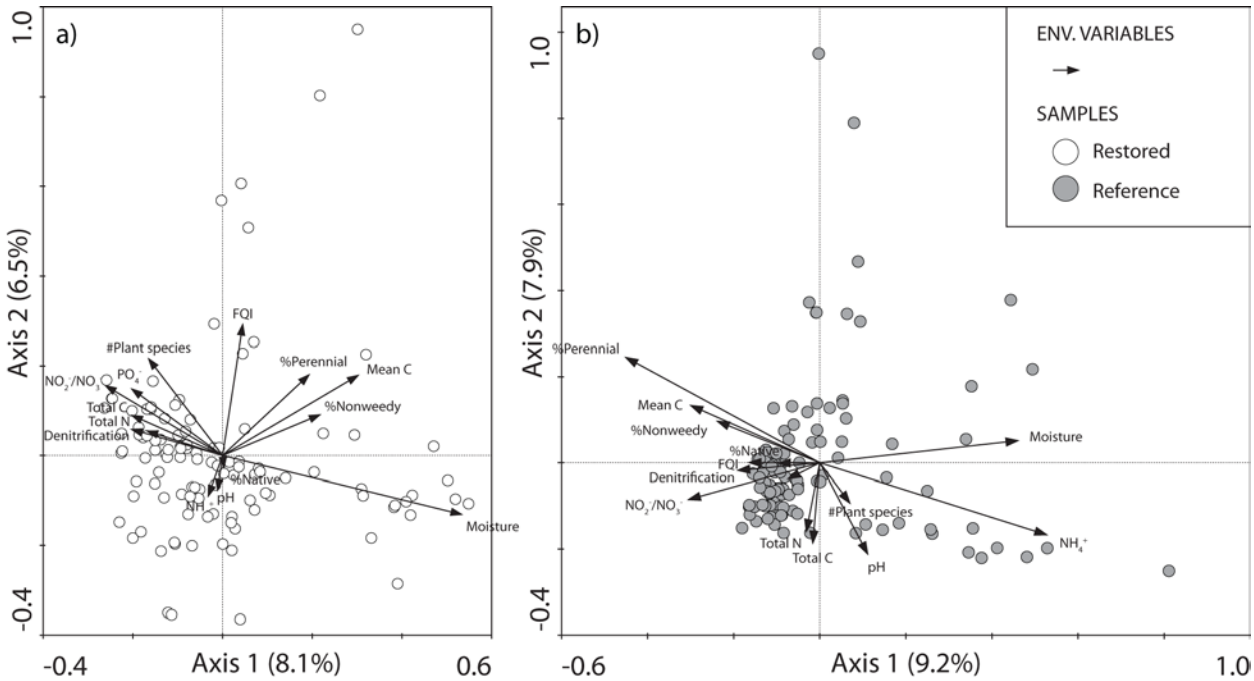

Figure 4. Correspondence analysis (CA) plots showing factors influencing denitrifier communities in restored and reference wetlands separately. Separate permutational analysis of variance (PERMANOVA) models showed that the three most influential factors for the restored sites were moisture, pH, and total N, as well as mean C, FQI, and the percent perennial plant species (% perennial). The most influential factors for the reference sites were pH, nitrate, and moisture, as well as the percent native species present (% native), mean C, and FQI.

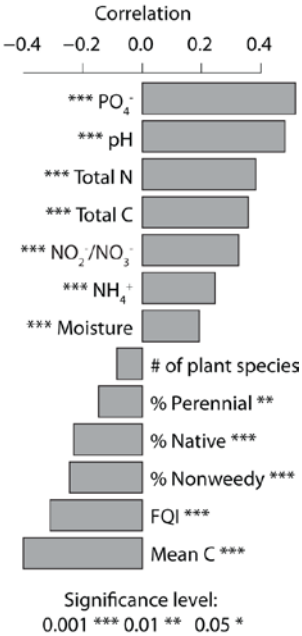

Figure 5. Correlation of soil and plant factors with potential denitrification rates in 2013. In general soil factors were positively correlated, while plant factors were negatively correlated with denitrification. Asterisks (*) represent the level of significance for each correlated term as determined by a simple linear regression model

Students involved in this project

There are two graduate students who contributed work on this project, including field collection and laboratory analyses. Dora Cohen is a third year Ph.D. student in the program for Ecology, Evolution, and Conservation Biology, while Natalie Stevenson is a second year masters student in Natural Resources and Environmental Science. Two undergraduate students also assisted with this project. Jonathan Bressler majored in Natural Resources and Environmental Sciences and was an intern in the summer internship program offered by the National Great Rivers Research and Education Center (NGRREC) during the summer of 2012. Emily Mackley majored in Integrative Biology and was an intern in the NGRREC summer internship during 2013.

Publications or pending publications

Parts of this work have been presented in the form of posters at conferences. Dora Cohen presented the poster "Microbial community composition has ramifications for denitrification capacity in restored wetlands" at the Joint Aquatic Sciences Meeting in Portland, OR on May 22, 2014. Natalie Stevenson presented a poster at the Illinois Water Conference in Urbana, IL on Oct. 14, 2014. Dora Cohen is currently using the data to write a manuscript entitled "Drought limitation and resiliency of denitrifiers in restored Illinois wetlands".

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Hydrologic Impacts of an alternative agricultural land use: a woody perennial polyculture

Basic Information

Title:	Hydrologic Impacts of an alternative agricultural land use: a woody perennial polyculture
Project Number:	2014IL289B
Start Date:	3/1/2014
End Date:	2/28/2015
Funding Source:	104B
Congressional District:	IL-013
Research Category:	Biological Sciences
Focus Category:	Management and Planning, Conservation, Ecology
Descriptors:	None
Principal Investigators:	Evan DeLucia, Kevin Wolz

Publications

There are no publications.

Illinois Water Resources Center Final Report

March 10, 2015

Project Title: Influence of Water Quality and Stormwater Management on the Ecology of Mosquito-Borne Disease

Principal Investigator:

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Problem and Research Objective:

Urban ecosystems are notoriously prone to outbreaks of mosquito-borne diseases due to the extensive adaptations by mosquito vectors to urban environments (Bradley and Altizer 2007). Mosquitoes undergo a 'complex' life cycle involving both aquatic juvenile stages (egg, larva and pupa) and an adult terrestrial stage. Mosquitoes that have adapted to reproduce in urban environments are those that are best able to utilize the many nutrient-rich aquatic habitats that urban development often provisions. Preventing juvenile mosquito development in these habitats is considered one of the most effective strategies for reducing adult mosquito abundance and public health risk due to mosquito-borne disease, and is thus a primary focus of urban mosquito control programs (Floore 2006). This is largely accomplished through the application of insecticides targeting the juvenile life stages (e.g. larvicides), and by modifying conditions or human behaviors that create or maintain productive environments for mosquito development (e.g. source reduction, public education, etc.). For these prevention programs to be effective requires identifying risk factors for vector mosquito production in urban aquatic habitats.

Infrastructure for managing urban stormwater runoff provides abundant aquatic habitats for *Culex pipiens* and *Cx. restuans*, the primary mosquito vectors of West Nile virus (WNV) in Illinois. These structures may be particularly prolific habitats for vector mosquitoes when receiving nutrient-enriched runoff or septic effluent. High-nutrient loading in aquatic habitats encourages oviposition by these species and can alleviate negative effects of larval competition on the rate of juvenile development and survivorship (Reiskind et al. 2004, Alto et al. 2012, Kraus and Vonesh 2012). Additionally, adult mosquitoes emerging from nutrient-rich environments are typically larger and possess greater nutrient reserves, which can enhance longevity, fecundity and dispersal range (Akoh et al. 1992, Drummond et al. 2006, Alto et al. 2012), and may ultimately alter their potential to acquire and transmit a pathogen.

In addition to altering conditions for developing mosquitoes, the retention of nutrient-rich runoff in stormwater management structures facilitates invasion by cattails (*Typha* spp.) and other emergent, aquatic macrophyte species tolerant of saturated, eutrophic conditions (McCormick et al. 2004, Kettenring et al. 2011). These invasive plants can competitively displace desired vegetation (e.g. turfgrass, native flood-tolerant or aquatic flora, etc.), resulting in dense, monospecific stands reaching several meters in height (Kominkova et al. 2000, Vaccoro et al. 2009), with an above-ground biomass often exceeding 1 kg / m^2 (Mason and Bryant 1975). To control this undesirable vegetation, managers of stormwater infrastructure may employ annual or semi-annual mowing, herbicide applications, or prescribed burning.

Emergent, aquatic plant growth and its management can have variable effects on the abundance and species composition of juvenile mosquitoes, depending on the plant species, type of aquatic habitat, and temporal concordance between vegetation management activities and hydroperiod (Walton 2012). In semi-permanent or permanent habitats (e.g., constructed wetlands, retention ponds), dense aquatic vegetation can provide juvenile mosquitoes refuge from predation (De Szalay and Resh 2000), and is therefore often characterized as a significant risk factor for the production of vector mosquitoes. Reducing the density of emergent vegetation has been shown to be an effective environmental management tool for diminishing juvenile mosquito abundance in these settings (De Szalay et al. 1995, Thullen et al. 2002, Lawler et al. 2007). However, vegetation management practices can also promote eutrophic conditions that support high juvenile mosquito densities when the resulting plant litter is later inundated and decomposes, particularly in ephemeral or seasonally flooded aquatic environments (Jiannino and Walton 2004, Walton and Jiannino 2005). Thick accumulations of plant litter in these habitats can also alter drainage, creating fragmented, nutrient-rich habitats that obstruct the movement or functional response of predators that regulate juvenile mosquito populations (Berkelhamer and Bradley 1989).

The overall objectives of this study were to examine the potential consequences to vector mosquito production resulting from colonization of two common stormwater management tools (ditches and dry detention basins) by one of the most widespread and invasive, aquatic macrophytes groups in urban landscapes (cattails), and from actions taken to control these invasive macrophytes (mowing), and to evaluate whether these effects of invasive vegetation or vegetation management on the larval ecology of vector mosquitoes are mediated by water quality. The specific goals of this study were:

Goal 1. Conduct a field survey of stormwater ditches and dry detention basins to evaluate relationships among plant composition (grasses vs. cattails), plant management (mowing), water quality and the abundance of juvenile *Culex* spp. mosquitoes.

Goal 2. Determine whether the type of plant detritus in aquatic habitats (cattails vs. turfgrass), or exogenous orthophosphate enrichment, influence the oviposition response of *Culex* spp. mosquitoes, and whether differences in the responses of *Cx. pipiens* and *Cx. restuans* to these factors are consistent with differences in the distribution of their juveniles in storm water BMPs observed in Goal 1.

Goal 3. Compare the effects of plant detritus type (cattails vs. turfgrass) and concentration in aquatic habitats on the duration of juvenile development, adult body size and adult longevity of *Cx. pipiens* and *Cx. restuans*. Determine whether these parameters in *Cx. pipiens* are influenced by exogenous orthophosphate enrichment of cattail infusions.

Methodology:

Goal 1.

A total of 36 stormwater management structures (16 dry detention basins and 20 drainage ditches) were selected in the communities of Urbana, Champaign and Savoy, IL, based on dominant plant species composition and the presence of standing water in the early spring (≈1 month prior to sampling juvenile mosquitoes). Grasses (primarily turfgrass) were the dominant vegetation at 11 sites, and the remaining 25 sites were colonized by cattails. Vegetation within 17 of the study sites was mowed at least once during the study period as part of existing municipal or private maintenance programs.

Juvenile mosquito abundance and water quality at each study site were assessed at ≈2 week intervals, from 3 June to 16 August, 2013 (total 5-6 inspection dates per site). When standing water was present, a sample of mosquito larval and pupae was collected (20 dips per sample), and specimens were counted and identified to species (larvae) or genus (pupae). Hand-held meters were used to collect field measurements of dissolved oxygen, pH, conductivity, salinity and total dissolved solids. A water sample was collected from each mosquito sampling location, transported to the lab on ice and analyzed immediately for total reactive phosphorus and nitrates by colorimetry (Hach 2009). The relative influences of various site characteristics (e.g. vegetation type, mowing history, type of structure) on larval abundance were evaluated as fixed effects in a General Linear Mixed Model (GLMM). Water quality measures and lagged intervals of cumulative rainfall and mean daily air temperature in the two weeks preceding sampling, were explored as covariates. A spearman correlation analysis was also used to compare relationships between juvenile mosquito abundance and each water quality parameter.

Goal 2.

Two field assays was performed to compare how the type and quantity of nutrient enrichment in aquatic habitats affects the oviposition response of *Cx. pipiens* and *Cx. restuans*. Each assay was performed at 5 replicate sites in Urbana, IL. At each site, ovitraps representing each of seven treatments were randomly assigned to sheltered locations at least 10 m apart along the margins of woody vegetation. Ovitraps were prepared by submerging a quantity of plant substrate in 8 liters of tap water in a 19 L pail. Egg rafts were collected daily from the water surface of ovitraps and the position of each ovitrap was rotated among locations within each study site. At the end of each week, ovitrap positions were re-randomized. First instar larvae hatching from field-collected egg rafts were identified to species. For both *Cx. pipiens* and *Cx. restuans*, the daily numbers of egg rafts in each ovitrap, as a proportion of the daily total collected at each site, were fit to a GLMM with a binomial distribution and logit link function.

Trap location nested within study site was included as a random effect. Post-hoc, pairwise comparisons between the average proportion of egg rafts collected in each treatment were performed using a sequential Bonferoni adjustment.

In the 1st experiment, we compared oviposition by *Culex* spp. mosquitoes in infusions prepared from cattail or turfgrass clippings (4.5 g / L) and receiving one of three levels of orthophosphate enrichment: a) no enrichment, b) weekly addition of 0.27 mg PO_4-P / L / week, and c) a single dose of 0.81 mg PO_4-P / L added at the start of the assay. A seventh treatment consisted of weekly enrichment with 0.27 mg PO_4-P / L / week in the absence of a plant substrate. The weekly enrichment rate used in this assay (0.27 mg PO_4-P / L) is comparable to the median concentration of reactive phosphorus measured in storm water BMPs contemporaneously with a high abundance of *Cx.* spp. larvae in Goal 1 (>3 per dip; Fig. 1). Collections were made 04 September to 01 October, 2013.

In the 2nd experiment, we varied the amount of plant substrate used in each trap, comparing 0.75, 1.5 and 4.5 g / L of turfgrass clippings, and 0.75, 1.5, 4.5 and 9 g / L of cattail clippings. Collections of *Culex* spp. egg rafts were made 25 June to 21 July, 2014.

Goal 3.

Three laboratory experiments were performed to examine how nutrient enrichment of the aquatic environment from plant detritus and exogenous orthophosphate inputs affect the duration of juvenile development, and the size and longevity of adult mosquitoes. Experiments were performed by adding newly-hatched larvae to 340 ml of a plant infusion (with or without the addition of orthophosphate) that was allowed to decompose for 3 days prior to the experiment. After completing larval development, individual pupae were transferred from the plant infusion to screen-covered, 50 ml plastic centrifuge tubes provisioned with water-soaked cotton balls and observed every 12 hours to record the time of adult eclosion and time of death. After death, the length of one wing was measured as a proxy for adult body size (van den Huevel 1963).

The first laboratory experiment examined the effect of high orthophosphate enrichment (0.81 mg PO_4-P / L) on the juvenile development rate, adult size and adult longevity of *Cx. pipiens* reared in an infusion of decomposing cattails (4.5 g / L). Experiments were performed at 2 larval densities (10 and 20 per replicate). Each treatment (+/- orthophosphate enrichment x 2 larval densities) was replicated four times in this experiment.

The remaining laboratory experiments evaluated the influence of plant substrate type (cattails vs. turfgrass) and concentration on the juvenile development rate adult size and adult longevity of *Cx. pipiens* (Exp. 2) and *Cx. restuans* (Exp. 3). In both experiments, infusions were prepared using three concentrations of cattails (1.5, 4.5, and 9.0 g / L), and two concentrations of turfgrass (0.75 and 1.5 g / L). Additional concentrations of turfgrass (0.375 g / L) and cattails (0.75 g / L) were added in experiment 3 (*Cx. restuans*). In both experiments, 10 newly hatched larvae were added to each replicate, and each treatment was replicated 5 times.

Principle Findings and Significance.

Goal 1.

1) A substantial increase in the abundance of juvenile mosquitoes (*Cx.* sp. pupae and larvae of *Cx. pipiens* and *Cx. restuans*) occurred immediately following mowing of vegetation in stormwater structures (Fig. 2). Abundance of *Cx.* spp. juveniles was low in habitats with unmanaged vegetation or vegetation mowed >2 weeks previously. The class of structure (dry detention basin or ditch) was not a significant influence on the abundance of mosquito juveniles.
2) Dominant plant type mediated the response of *Cx. pipiens* to mowing. The abundance of *Cx. pipiens* larvae and *Cx.* sp. pupae were much greater in recently mowed turfgrass habitats than in recently mowed habitats colonized by cattails (Fig. 2a, 2c). The abundance of *Cx. restuans* larvae was not significantly influenced by plant type (Fig. 2b).
3) Total reactive phosphorus concentration varied greatly among samples (Fig. 1), and was positively correlated with the abundance of *Cx. restuans* larvae (P=0.004) and *Culex* sp. pupae (P=0.038). The abundance of *Cx. pipiens* larvae was positively correlated with reactive phosphorus concentration in dry detention basins (P=0.036), but not in ditches (P>0.05). Relationships between juvenile mosquito abundance and nitrate concentration were not significant.

Goal 2.

1) In the first field assay where a single concentration of plant substrate was evaluated (4.5 g / L), the type of substrate had asymmetrical effects on the oviposition behavior of *Cx. pipiens* and *Cx. restuans* in the absence of orthophosphate enrichment (open bars, Fig. 3). Consistent with our observations in Goal 1, *Cx. pipiens* deposited a greater proportion of egg rafts in turfgrass infusion and *Cx. restuans* preferred to oviposit in cattail infusions. Similar results were observed when oviposition preferences were compared over a range of turfgrass and cattail concentrations (Fig. 4). The greatest proportion of *Cx. pipiens* egg rafts were collected in the highest concentration of turfgrass substrate (4.5 g / L), whereas *Cx. restuans* deposited a significantly greater proportion of egg rafts in the highest concentration of cattails (9.0 g / L).
2) Weekly enrichment of plant infusions with PO_4-P (0.27 mg / L / wk) did not affect the oviposition behavior of either species (light grey bars, Fig. 3). However, both species responded positively to cattail infusions enriched with a single, large orthophosphate dose (0.81 mg PO_4-P / L; dark grey bars, Fig. 3).

Goal 3.

1) Enrichment of cattail infusions with 0.81 mg PO_4-P / L did not significantly affect the juvenile development rate of *Cx. pipiens* males or females (Table 1).
2) Orthophosphate enrichment of cattail infusions had a significant negative effect on the longevity of adult *Cx. pipiens* males (P=0.006), but only a marginally negative effect on their wing length (P=0.059). The mean wing length of adult *Cx. pipiens* females was not

significantly affected by orthophosphate enrichment, and enrichment only had a marginally positive effect on adult female longevity (P=0.051).

3) At a comparable concentration of plant substrate (1.5 g / L), the rate of juvenile development (Fig. 5), and the size and longevity of adult mosquitoes (Fig. 6) were greater when *Cx. pipiens* and *Cx. restuans* were reared in turfgrass infusions, compared with cattail infusions. The lowest concentration of cattail substrate (0.75 g / L) did not provide sufficient resources for juvenile development of *Cx. restuans*; average survivorship of larvae to pupation was <10 %.

These finding demonstrate that mowing of vegetation in stormwater management structures during the growing season substantially increases the immediate risk of these habitats supporting vector mosquito production. Conversely, ditches and dry detention basins with unmanaged vegetation were associated with a relatively low abundance of *Culex* spp. juveniles. These results indicate that caution should be used when conducting maintenance activities that result in the deposition of plant debris in locations likely to collect standing water. Vegetation management practices applied during the growing season should be coordinated with public health agencies so that larval surveillance, and if necessary insecticide treatments, can be implemented to prevent production of vector mosquitoes. When possible, vegetation management should be performed in the early spring or late fall, when enrichment of aquatic habitats is unlikely to coincide with the peak seasonal activity of vector mosquitoes and WNV transmission risk.

Results from our study indicate that colonization by cattails does not appear to increase public health risks associated with mowing of vegetation in dry detention basins and ditches. In recently mowed structures, the presence of cattails was associated with a lower relative abundance of larvae of *Cx. pipiens*, the primary epidemic vector of WNV. Addition of cattail detritus resulted in aquatic habitats that were less attractive to gravid *Cx. pipiens* than adding turfgrass detritus, although this difference was less apparent when cattail infusions were enriched with a high concentration of orthophosphate, or when a high concentration of cattail detritus was present (9 g / L). Aquatic habitats containing cattail detritus generally provided a poorer quality environment for the juvenile development, and produced adult *Cx. pipiens* and *Cx. restuans* mosquitoes with a lower resistance to starvation (i.e. reduced longevity when deprived of food) – decreasing the likelihood of mosquitoes produced by these habitats surviving long enough to acquire and transmit a pathogen.

Similar to previous studies (Mercer et al. 2005, Gingrich 2006, Young et al. 2014), we observed a positive association between reactive phosphorus and juvenile mosquito abundance. Average mosquito abundance and reactive phosphorus concentration were both highest in recently mowed habitats. Although we did not attempt to identify the source of phosphorus loading, it is likely that these relatively high reactive phosphorus concentrations were a result of both direct inputs from runoff and endogenous release from decomposing plant detritus deposited by mowing, and represents an overall increase in nutrient availability in mowed structures. However, we regard the study design in Goal 2 and Goal 3 (experiment 1) to represent a realistic scenario for possible interactions between cattail detritus and exogenous inputs of

biologically-active phosphorus from stormwater runoff. The P concentrations used in the lab and field experiments (0.27 and 0.81 mg PO_4-P / L) are within the moderate to high range of soluble reactive P reported from urban stormwater runoff (Waschbusch et al. 1999), and comparable or greater P loading may occur in stormwater BMPs intercepting highly-enriched runoff from sources such as fertilized lawns or compost amendments used for erosion control (Waschbusch et al. 1999, Faucette et al. 2005).

Notable Achievements.

This study provides the first evidence of invasive aquatic plants and their management mediating the abundance and species composition of mosquitoes of public health significance in two common urban stormwater management tools; ditches and dry detention basins. Our study identifies a potential mechanism for the observed effects of plant species composition effects on juvenile vector communities in these habitats (i.e. oviposition preference) and demonstrates that that plant species composition can also influence important traits of adult mosquitoes produced in stormwater management structures. Our results also suggest an interaction between plant type and high exogenous phosphorus enrichment in stormwater infrastructure that could alter the species composition or abundance of juvenile vectors.

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Publications and Presentations. One manuscript describing the results in this report is currently in review in the journal Ecological Applications, and a second if being prepared and will be submitted to another peer-reviewed journal for publication. Research results also have been resented at the Annual Meeting of the Illinois Mosquito and Vector Control Association.

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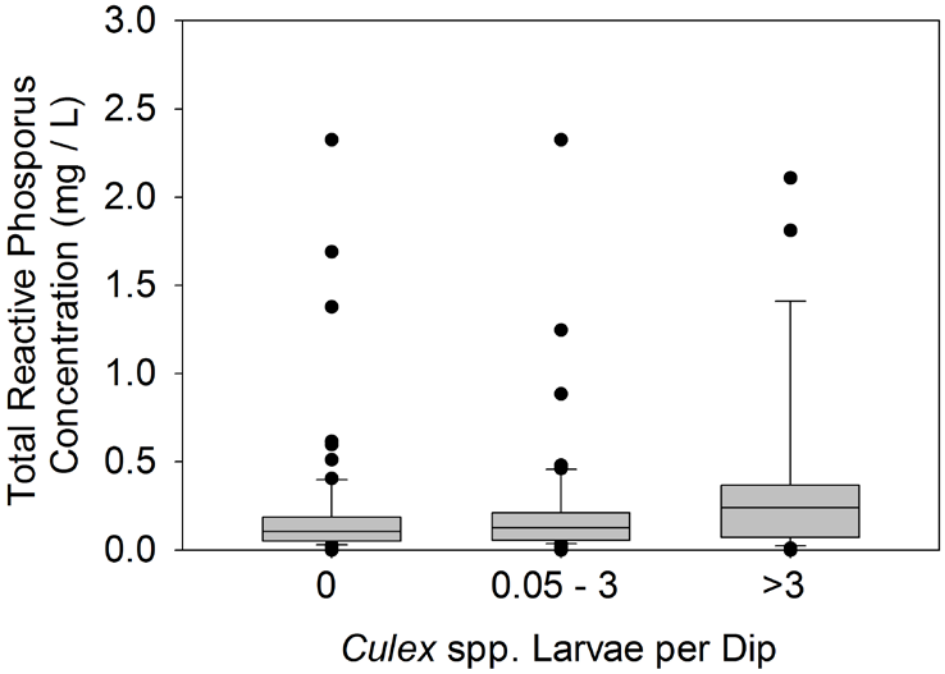

Figure 1. Box plot comparison of total reactive phosphorus concentration by larval mosquito abundance class.

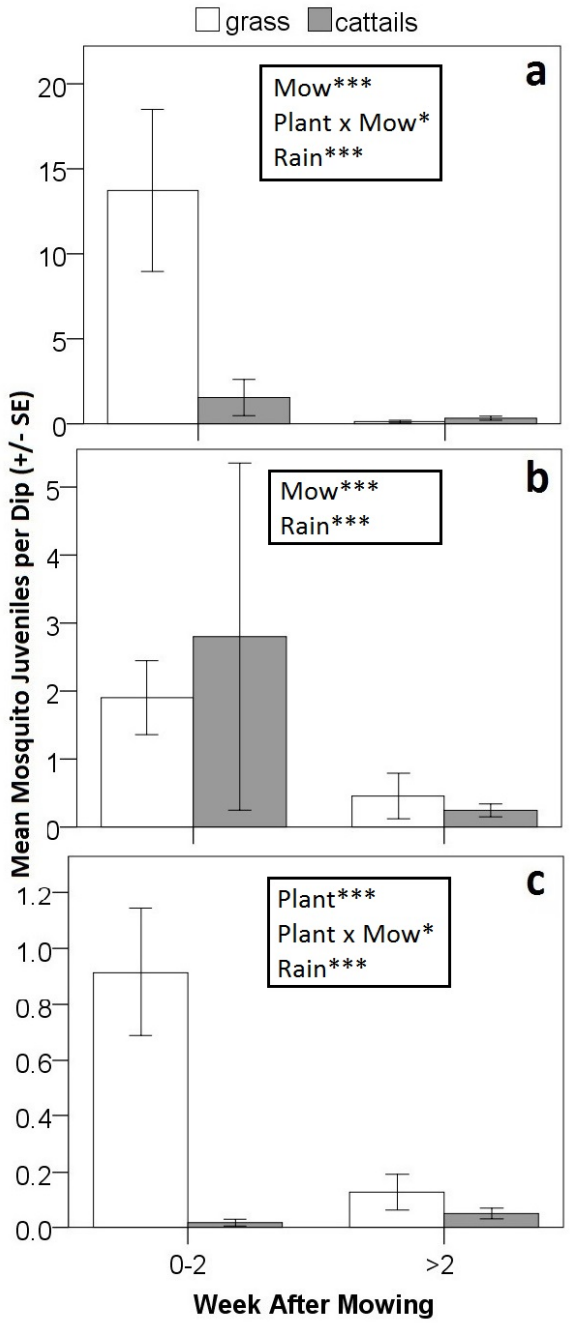

Figure 2. Effect of plant type and recent mowing on the average numbers of *Cx. pipiens* larvae (a), *Cx. restuans* larvae (b), and *Culex* sp. pupae (c) collected from stormwater ditches and detention basins, 03 June to 16 August, 2013 (observations from dry sites excluded from calculation of average values). Significant effects and interactions in GLMMs are shown for each group [**Plant** = dominant plant type (grass or cattails), **Mow** = vegetation mowed ~≤ 2 weeks prior to sampling, **Rain** = cumulative rainfall 0 to 8 days prior to sampling; * p-value < 0.05, ** p-value < 0.01, *** p-value < 0.001].

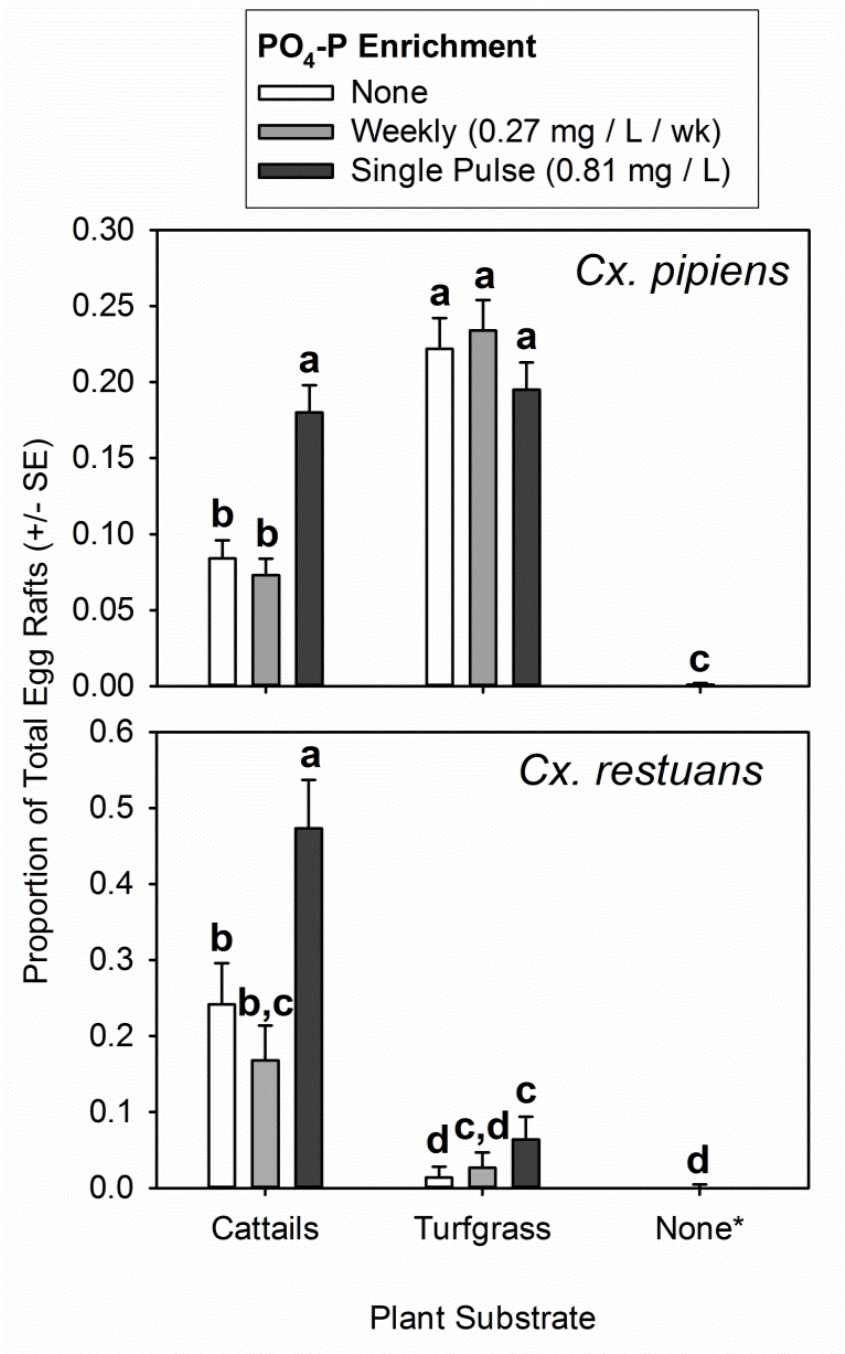

Figure 3. Influence of orthophosphate enrichment on the proportion of eggs rafts of Cx. *pipiens* and *Cx. restuans* collected ovitraps baited with either cattails or turfgrass clippings (4.5 g / l). For each species, columns with the same letter are not significantly different (P>0.05; post-hoc test in GLMM with Bonferroni correction).

* Traps without plant substrate were enriched with 0.27 mg PO_4-P / L / week.

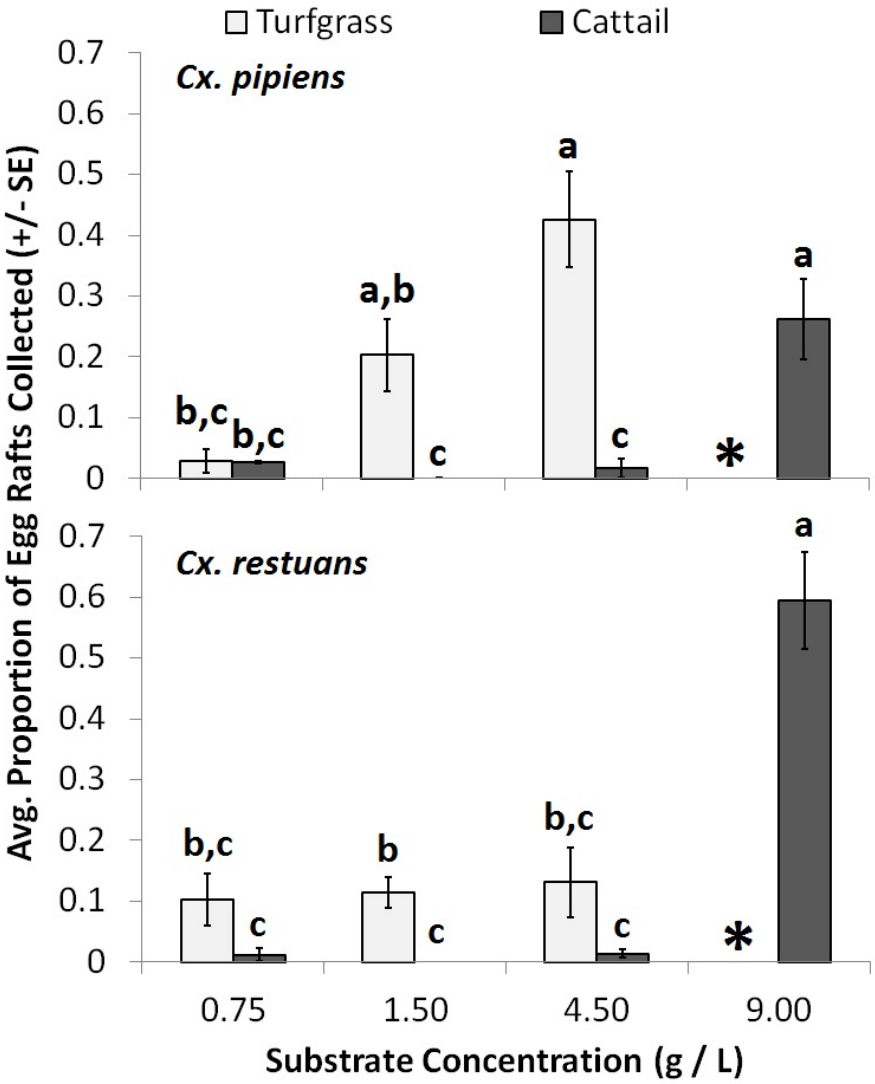

Figure 4. Proportion of *Cx. pipiens* and *Cx. restuans* egg rafts collected from infusions containing varying concentrations of turfgrass or cattail substrate. Turfgrass was not evaluated at the 9.0 g / L concentration (indicated by the asterisk in figure). For each species, columns with the same letter are not significantly different (P>0.05; post-hoc test in GLMM with Bonferroni correction).

Table 1. General linear models evaluating the effects of larval rearing density and orthophosphate enrichment (0.81 mg PO_4-P / L) of cattail infusions (4.5 g / l) on the duration of juvenile development, adult longevity and wing length of male and female *Cx. pipiens*.

Response Variable	Parameter[1]	Males			Females		
		Coeff. (± 95% CI)	t	p-value	Coeff. (± 95% CI)	t	p-value
1 / duration of juvenile development	Intercept	0.098 (0.002)	52.8	<0.001	0.084 (0.003)	32.5	<0.001
	Orthophosphate	0.000 (0.002)	<0.1	0.946	0.003 (0.003)	0.9	0.377
	Larval Density	0.004 (0.002)	1.8	0.089	**0.011 (0.003)**	**3.7**	**0.003**
1 / adult longevity	Intercept	0.581 (0.017)	34.6	<0.001	0.397 (0.026)	15.2	<0.001
	Orthophosphate	**-0.064 (0.019)**	**-3.3**	**0.006**	0.065 (0.030)	2.2	0.051
	Larval Density	**-0.186 (0.019)**	**-9.6**	**<0.001**	-0.007 (0.030)	-0.2	0.817
wing length	Intercept	0.361 (0.004)	100.4	<0.001	0.283 (0.004)	75.2	<0.001
	Orthophosphate	-0.009 (0.004)	-2.1	0.059	0.003 (0.004)	0.7	0.511
	Larval Density	**-0.039 (0.004)**	**-9.4**	**<0.001**	**-0.014 (0.004)**	**-3.2**	**0.007**

[1] The interaction term (orthophosphate x larval density) was excluded from the final models as it was not a significant effect for any of the response variables for males or females.

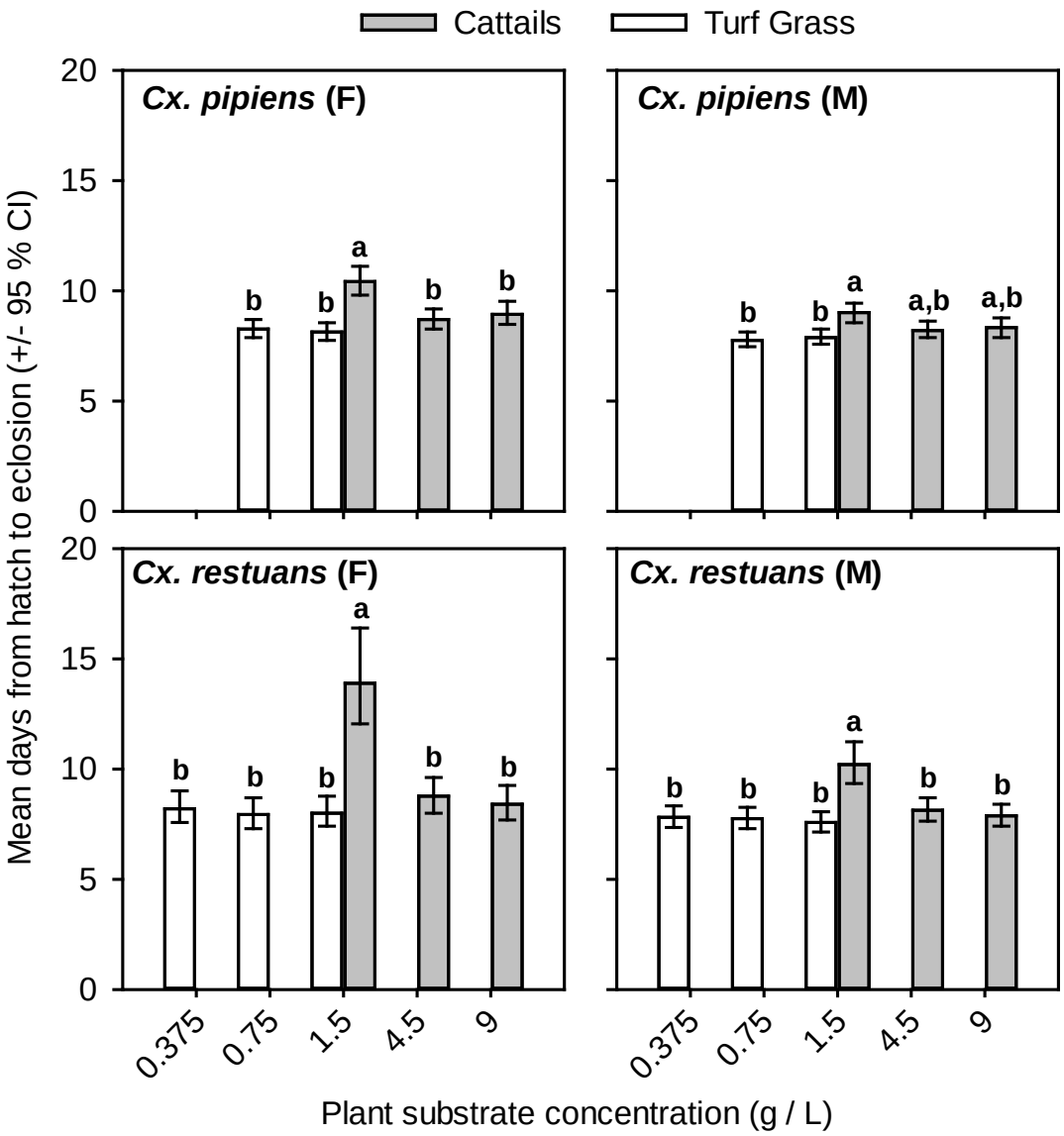

Figure 5. Juvenile development time of male (M) and female (F) *Cx. pipiens* and *Cx. restuans* reared in infusions containing varying concentrations of turfgrass or cattail substrates. Columns within each group that share the same letter were not significantly different (P>0.05; post-hoc test with sequential Bonferroni correction in GLM using 1 / Days to Eclosion as the response variable).

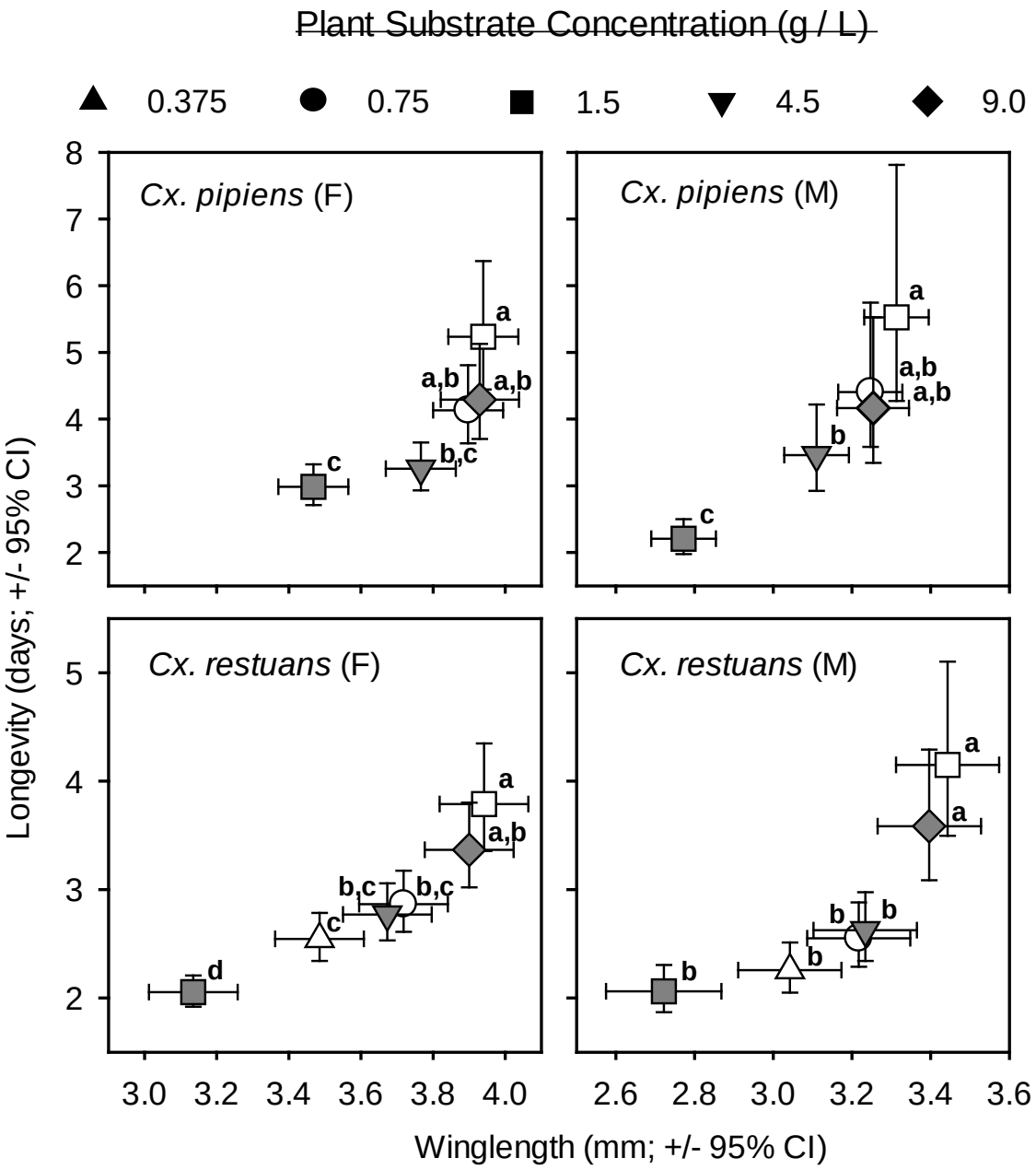

Figure 6. Average wing length and longevity of adult male (M) and female (F) *Cx. pipiens* and *Cx. restuans* reared in infusions containing varying concentrations of turfgrass (open symbols) and cattails (closed symbols). Average longevity within treatments for each group that share the same letter were not significantly different (P>0.05; post-hoc test with sequential Bonferroni correction in GLM using 1 / days surviving as the response variable). A significant linear relationship was observed between adult winglength and transformed adult longevity (1 / days surviving) for male (R^2 = 0.546) and female (R^2 = 0.512) *Cx. pipiens*, and male (R^2 = 0.436) and female (R^2 = 0.509) *Cx. restuans*.

Influence of water quality and stormwater management on the ecology of mosquito-borne disease

Basic Information

Title:	Influence of water quality and stormwater management on the ecology of mosquito-borne disease
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End Date:	2/28/2015
Funding Source:	104B
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Research Category:	Biological Sciences
Focus Category:	Ecology, Management and Planning, Surface Water
Descriptors:	None
Principal Investigators:	Brian Allan, Andrew Mackay

Publications

There are no publications.

Illinois Water Resources Center Final Report

March 10, 2015

Project Title: Influence of Water Quality and Stormwater Management on the Ecology of Mosquito-Borne Disease

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Problem and Research Objective:

Urban ecosystems are notoriously prone to outbreaks of mosquito-borne diseases due to the extensive adaptations by mosquito vectors to urban environments (Bradley and Altizer 2007). Mosquitoes undergo a 'complex' life cycle involving both aquatic juvenile stages (egg, larva and pupa) and an adult terrestrial stage. Mosquitoes that have adapted to reproduce in urban environments are those that are best able to utilize the many nutrient-rich aquatic habitats that urban development often provisions. Preventing juvenile mosquito development in these habitats is considered one of the most effective strategies for reducing adult mosquito abundance and public health risk due to mosquito-borne disease, and is thus a primary focus of urban mosquito control programs (Floore 2006). This is largely accomplished through the application of insecticides targeting the juvenile life stages (e.g. larvicides), and by modifying conditions or human behaviors that create or maintain productive environments for mosquito development (e.g. source reduction, public education, etc.). For these prevention programs to be effective requires identifying risk factors for vector mosquito production in urban aquatic habitats.

Infrastructure for managing urban stormwater runoff provides abundant aquatic habitats for *Culex pipiens* and *Cx. restuans*, the primary mosquito vectors of West Nile virus (WNV) in Illinois. These structures may be particularly prolific habitats for vector mosquitoes when receiving nutrient-enriched runoff or septic effluent. High-nutrient loading in aquatic habitats encourages oviposition by these species and can alleviate negative effects of larval competition on the rate of juvenile development and survivorship (Reiskind et al. 2004, Alto et al. 2012, Kraus and Vonesh 2012). Additionally, adult mosquitoes emerging from nutrient-rich environments are typically larger and possess greater nutrient reserves, which can enhance longevity, fecundity and dispersal range (Akoh et al. 1992, Drummond et al. 2006, Alto et al. 2012), and may ultimately alter their potential to acquire and transmit a pathogen.

In addition to altering conditions for developing mosquitoes, the retention of nutrient-rich runoff in stormwater management structures facilitates invasion by cattails (*Typha* spp.) and other emergent, aquatic macrophyte species tolerant of saturated, eutrophic conditions (McCormick et al. 2004, Kettenring et al. 2011). These invasive plants can competitively displace desired vegetation (e.g. turfgrass, native flood-tolerant or aquatic flora, etc.), resulting in dense, monospecific stands reaching several meters in height (Kominkova et al. 2000, Vaccoro et al. 2009), with an above-ground biomass often exceeding 1 kg / m^2 (Mason and Bryant 1975). To control this undesirable vegetation, managers of stormwater infrastructure may employ annual or semi-annual mowing, herbicide applications, or prescribed burning.

Emergent, aquatic plant growth and its management can have variable effects on the abundance and species composition of juvenile mosquitoes, depending on the plant species, type of aquatic habitat, and temporal concordance between vegetation management activities and hydroperiod (Walton 2012). In semi-permanent or permanent habitats (e.g., constructed wetlands, retention ponds), dense aquatic vegetation can provide juvenile mosquitoes refuge from predation (De Szalay and Resh 2000), and is therefore often characterized as a significant risk factor for the production of vector mosquitoes. Reducing the density of emergent vegetation has been shown to be an effective environmental management tool for diminishing juvenile mosquito abundance in these settings (De Szalay et al. 1995, Thullen et al. 2002, Lawler et al. 2007). However, vegetation management practices can also promote eutrophic conditions that support high juvenile mosquito densities when the resulting plant litter is later inundated and decomposes, particularly in ephemeral or seasonally flooded aquatic environments (Jiannino and Walton 2004, Walton and Jiannino 2005). Thick accumulations of plant litter in these habitats can also alter drainage, creating fragmented, nutrient-rich habitats that obstruct the movement or functional response of predators that regulate juvenile mosquito populations (Berkelhamer and Bradley 1989).

The overall objectives of this study were to examine the potential consequences to vector mosquito production resulting from colonization of two common stormwater management tools (ditches and dry detention basins) by one of the most widespread and invasive, aquatic macrophytes groups in urban landscapes (cattails), and from actions taken to control these invasive macrophytes (mowing), and to evaluate whether these effects of invasive vegetation or vegetation management on the larval ecology of vector mosquitoes are mediated by water quality. The specific goals of this study were:

Goal 1. Conduct a field survey of stormwater ditches and dry detention basins to evaluate relationships among plant composition (grasses vs. cattails), plant management (mowing), water quality and the abundance of juvenile *Culex* spp. mosquitoes.

Goal 2. Determine whether the type of plant detritus in aquatic habitats (cattails vs. turfgrass), or exogenous orthophosphate enrichment, influence the oviposition response of *Culex* spp. mosquitoes, and whether differences in the responses of *Cx. pipiens* and *Cx. restuans* to these factors are consistent with differences in the distribution of their juveniles in storm water BMPs observed in Goal 1.

Goal 3. Compare the effects of plant detritus type (cattails vs. turfgrass) and concentration in aquatic habitats on the duration of juvenile development, adult body size and adult longevity of *Cx. pipiens* and *Cx. restuans*. Determine whether these parameters in *Cx. pipiens* are influenced by exogenous orthophosphate enrichment of cattail infusions.

Methodology:

Goal 1.

A total of 36 stormwater management structures (16 dry detention basins and 20 drainage ditches) were selected in the communities of Urbana, Champaign and Savoy, IL, based on dominant plant species composition and the presence of standing water in the early spring (≈1 month prior to sampling juvenile mosquitoes). Grasses (primarily turfgrass) were the dominant vegetation at 11 sites, and the remaining 25 sites were colonized by cattails. Vegetation within 17 of the study sites was mowed at least once during the study period as part of existing municipal or private maintenance programs.

Juvenile mosquito abundance and water quality at each study site were assessed at ≈2 week intervals, from 3 June to 16 August, 2013 (total 5-6 inspection dates per site). When standing water was present, a sample of mosquito larval and pupae was collected (20 dips per sample), and specimens were counted and identified to species (larvae) or genus (pupae). Hand-held meters were used to collect field measurements of dissolved oxygen, pH, conductivity, salinity and total dissolved solids. A water sample was collected from each mosquito sampling location, transported to the lab on ice and analyzed immediately for total reactive phosphorus and nitrates by colorimetry (Hach 2009). The relative influences of various site characteristics (e.g. vegetation type, mowing history, type of structure) on larval abundance were evaluated as fixed effects in a General Linear Mixed Model (GLMM). Water quality measures and lagged intervals of cumulative rainfall and mean daily air temperature in the two weeks preceding sampling, were explored as covariates. A spearman correlation analysis was also used to compare relationships between juvenile mosquito abundance and each water quality parameter.

Goal 2.

Two field assays was performed to compare how the type and quantity of nutrient enrichment in aquatic habitats affects the oviposition response of *Cx. pipiens* and *Cx. restuans*. Each assay was performed at 5 replicate sites in Urbana, IL. At each site, ovitraps representing each of seven treatments were randomly assigned to sheltered locations at least 10 m apart along the margins of woody vegetation. Ovitraps were prepared by submerging a quantity of plant substrate in 8 liters of tap water in a 19 L pail. Egg rafts were collected daily from the water surface of ovitraps and the position of each ovitrap was rotated among locations within each study site. At the end of each week, ovitrap positions were re-randomized. First instar larvae hatching from field-collected egg rafts were identified to species. For both *Cx. pipiens* and *Cx. restuans*, the daily numbers of egg rafts in each ovitrap, as a proportion of the daily total collected at each site, were fit to a GLMM with a binomial distribution and logit link function.

Trap location nested within study site was included as a random effect. Post-hoc, pairwise comparisons between the average proportion of egg rafts collected in each treatment were performed using a sequential Bonferoni adjustment.

In the 1st experiment, we compared oviposition by *Culex* spp. mosquitoes in infusions prepared from cattail or turfgrass clippings (4.5 g / L) and receiving one of three levels of orthophosphate enrichment: a) no enrichment, b) weekly addition of 0.27 mg PO_4-P / L / week, and c) a single dose of 0.81 mg PO_4-P / L added at the start of the assay. A seventh treatment consisted of weekly enrichment with 0.27 mg PO_4-P / L / week in the absence of a plant substrate. The weekly enrichment rate used in this assay (0.27 mg PO_4-P / L) is comparable to the median concentration of reactive phosphorus measured in storm water BMPs contemporaneously with a high abundance of *Cx.* spp. larvae in Goal 1 (>3 per dip; Fig. 1). Collections were made 04 September to 01 October, 2013.

In the 2nd experiment, we varied the amount of plant substrate used in each trap, comparing 0.75, 1.5 and 4.5 g / L of turfgrass clippings, and 0.75, 1.5, 4.5 and 9 g / L of cattail clippings. Collections of *Culex* spp. egg rafts were made 25 June to 21 July, 2014.

Goal 3.

Three laboratory experiments were performed to examine how nutrient enrichment of the aquatic environment from plant detritus and exogenous orthophosphate inputs affect the duration of juvenile development, and the size and longevity of adult mosquitoes. Experiments were performed by adding newly-hatched larvae to 340 ml of a plant infusion (with or without the addition of orthophosphate) that was allowed to decompose for 3 days prior to the experiment. After completing larval development, individual pupae were transferred from the plant infusion to screen-covered, 50 ml plastic centrifuge tubes provisioned with water-soaked cotton balls and observed every 12 hours to record the time of adult eclosion and time of death. After death, the length of one wing was measured as a proxy for adult body size (van den Huevel 1963).

The first laboratory experiment examined the effect of high orthophosphate enrichment (0.81 mg PO_4-P / L) on the juvenile development rate, adult size and adult longevity of *Cx. pipiens* reared in an infusion of decomposing cattails (4.5 g / L). Experiments were performed at 2 larval densities (10 and 20 per replicate). Each treatment (+/- orthophosphate enrichment x 2 larval densities) was replicated four times in this experiment.

The remaining laboratory experiments evaluated the influence of plant substrate type (cattails vs. turfgrass) and concentration on the juvenile development rate adult size and adult longevity of *Cx. pipiens* (Exp. 2) and *Cx. restuans* (Exp. 3). In both experiments, infusions were prepared using three concentrations of cattails (1.5, 4.5, and 9.0 g / L), and two concentrations of turfgrass (0.75 and 1.5 g / L). Additional concentrations of turfgrass (0.375 g / L) and cattails (0.75 g / L) were added in experiment 3 (*Cx. restuans*). In both experiments, 10 newly hatched larvae were added to each replicate, and each treatment was replicated 5 times.

Principle Findings and Significance.

Goal 1.

1) A substantial increase in the abundance of juvenile mosquitoes (*Cx.* sp. pupae and larvae of *Cx. pipiens* and *Cx. restuans*) occurred immediately following mowing of vegetation in stormwater structures (Fig. 2). Abundance of *Cx.* spp. juveniles was low in habitats with unmanaged vegetation or vegetation mowed >2 weeks previously. The class of structure (dry detention basin or ditch) was not a significant influence on the abundance of mosquito juveniles.
2) Dominant plant type mediated the response of *Cx. pipiens* to mowing. The abundance of *Cx. pipiens* larvae and *Cx.* sp. pupae were much greater in recently mowed turfgrass habitats than in recently mowed habitats colonized by cattails (Fig. 2a, 2c). The abundance of *Cx. restuans* larvae was not significantly influenced by plant type (Fig. 2b).
3) Total reactive phosphorus concentration varied greatly among samples (Fig. 1), and was positively correlated with the abundance of *Cx. restuans* larvae (P=0.004) and *Culex* sp. pupae (P=0.038). The abundance of *Cx. pipiens* larvae was positively correlated with reactive phosphorus concentration in dry detention basins (P=0.036), but not in ditches (P>0.05). Relationships between juvenile mosquito abundance and nitrate concentration were not significant.

Goal 2.

1) In the first field assay where a single concentration of plant substrate was evaluated (4.5 g / L), the type of substrate had asymmetrical effects on the oviposition behavior of *Cx. pipiens* and *Cx. restuans* in the absence of orthophosphate enrichment (open bars, Fig. 3). Consistent with our observations in Goal 1, *Cx. pipiens* deposited a greater proportion of egg rafts in turfgrass infusion and *Cx. restuans* preferred to oviposit in cattail infusions. Similar results were observed when oviposition preferences were compared over a range of turfgrass and cattail concentrations (Fig. 4). The greatest proportion of *Cx. pipiens* egg rafts were collected in the highest concentration of turfgrass substrate (4.5 g / L), whereas *Cx. restuans* deposited a significantly greater proportion of egg rafts in the highest concentration of cattails (9.0 g / L).
2) Weekly enrichment of plant infusions with PO_4-P (0.27 mg / L / wk) did not affect the oviposition behavior of either species (light grey bars, Fig. 3). However, both species responded positively to cattail infusions enriched with a single, large orthophosphate dose (0.81 mg PO_4-P / L; dark grey bars, Fig. 3).

Goal 3.

1) Enrichment of cattail infusions with 0.81 mg PO_4-P / L did not significantly affect the juvenile development rate of *Cx. pipiens* males or females (Table 1).
2) Orthophosphate enrichment of cattail infusions had a significant negative effect on the longevity of adult *Cx. pipiens* males (P=0.006), but only a marginally negative effect on their wing length (P=0.059). The mean wing length of adult *Cx. pipiens* females was not

significantly affected by orthophosphate enrichment, and enrichment only had a marginally positive effect on adult female longevity (P=0.051).

3) At a comparable concentration of plant substrate (1.5 g / L), the rate of juvenile development (Fig. 5), and the size and longevity of adult mosquitoes (Fig. 6) were greater when *Cx. pipiens* and *Cx. restuans* were reared in turfgrass infusions, compared with cattail infusions. The lowest concentration of cattail substrate (0.75 g / L) did not provide sufficient resources for juvenile development of *Cx. restuans*; average survivorship of larvae to pupation was <10 %.

These finding demonstrate that mowing of vegetation in stormwater management structures during the growing season substantially increases the immediate risk of these habitats supporting vector mosquito production. Conversely, ditches and dry detention basins with unmanaged vegetation were associated with a relatively low abundance of *Culex* spp. juveniles. These results indicate that caution should be used when conducting maintenance activities that result in the deposition of plant debris in locations likely to collect standing water. Vegetation management practices applied during the growing season should be coordinated with public health agencies so that larval surveillance, and if necessary insecticide treatments, can be implemented to prevent production of vector mosquitoes. When possible, vegetation management should be performed in the early spring or late fall, when enrichment of aquatic habitats is unlikely to coincide with the peak seasonal activity of vector mosquitoes and WNV transmission risk.

Results from our study indicate that colonization by cattails does not appear to increase public health risks associated with mowing of vegetation in dry detention basins and ditches. In recently mowed structures, the presence of cattails was associated with a lower relative abundance of larvae of *Cx. pipiens*, the primary epidemic vector of WNV. Addition of cattail detritus resulted in aquatic habitats that were less attractive to gravid *Cx. pipiens* than adding turfgrass detritus, although this difference was less apparent when cattail infusions were enriched with a high concentration of orthophosphate, or when a high concentration of cattail detritus was present (9 g / L). Aquatic habitats containing cattail detritus generally provided a poorer quality environment for the juvenile development, and produced adult *Cx. pipiens* and *Cx. restuans* mosquitoes with a lower resistance to starvation (i.e. reduced longevity when deprived of food) – decreasing the likelihood of mosquitoes produced by these habitats surviving long enough to acquire and transmit a pathogen.

Similar to previous studies (Mercer et al. 2005, Gingrich 2006, Young et al. 2014), we observed a positive association between reactive phosphorus and juvenile mosquito abundance. Average mosquito abundance and reactive phosphorus concentration were both highest in recently mowed habitats. Although we did not attempt to identify the source of phosphorus loading, it is likely that these relatively high reactive phosphorus concentrations were a result of both direct inputs from runoff and endogenous release from decomposing plant detritus deposited by mowing, and represents an overall increase in nutrient availability in mowed structures. However, we regard the study design in Goal 2 and Goal 3 (experiment 1) to represent a realistic scenario for possible interactions between cattail detritus and exogenous inputs of

biologically-active phosphorus from stormwater runoff. The P concentrations used in the lab and field experiments (0.27 and 0.81 mg PO_4-P / L) are within the moderate to high range of soluble reactive P reported from urban stormwater runoff (Waschbusch et al. 1999), and comparable or greater P loading may occur in stormwater BMPs intercepting highly-enriched runoff from sources such as fertilized lawns or compost amendments used for erosion control (Waschbusch et al. 1999, Faucette et al. 2005).

Notable Achievements.

This study provides the first evidence of invasive aquatic plants and their management mediating the abundance and species composition of mosquitoes of public health significance in two common urban stormwater management tools; ditches and dry detention basins. Our study identifies a potential mechanism for the observed effects of plant species composition effects on juvenile vector communities in these habitats (i.e. oviposition preference) and demonstrates that that plant species composition can also influence important traits of adult mosquitoes produced in stormwater management structures. Our results also suggest an interaction between plant type and high exogenous phosphorus enrichment in stormwater infrastructure that could alter the species composition or abundance of juvenile vectors.

Students Supported With Funding. These funds helped to support an indenependent study by an undergraduate student (Ellie Moen) who assisted with the collection of field data described under Goal 1. Funds were also used to provide materials required for field and laboratory assays (Goals 2 and 3) that were performed as independent research projects by one undergraduate student, Ellie Moen, and one first-year veterinary medicine student, Matt Holland.

Publications and Presentations. One manuscript describing the results in this report is currently in review in the journal Ecological Applications, and a second if being prepared and will be submitted to another peer-reviewed journal for publication. Research results also have been resented at the Annual Meeting of the Illinois Mosquito and Vector Control Association.

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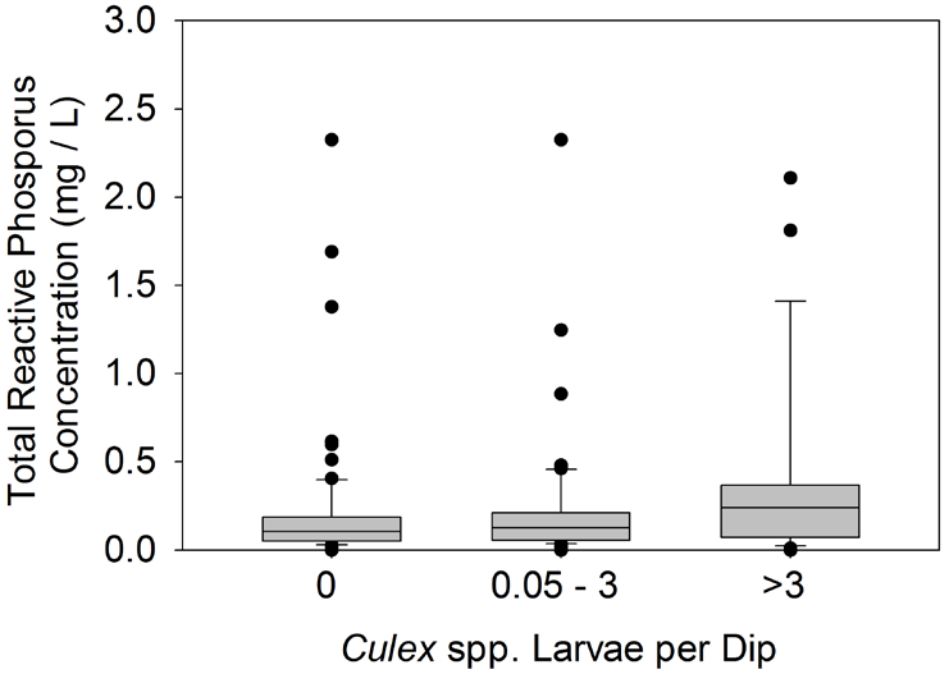

Figure 1. Box plot comparison of total reactive phosphorus concentration by larval mosquito abundance class.

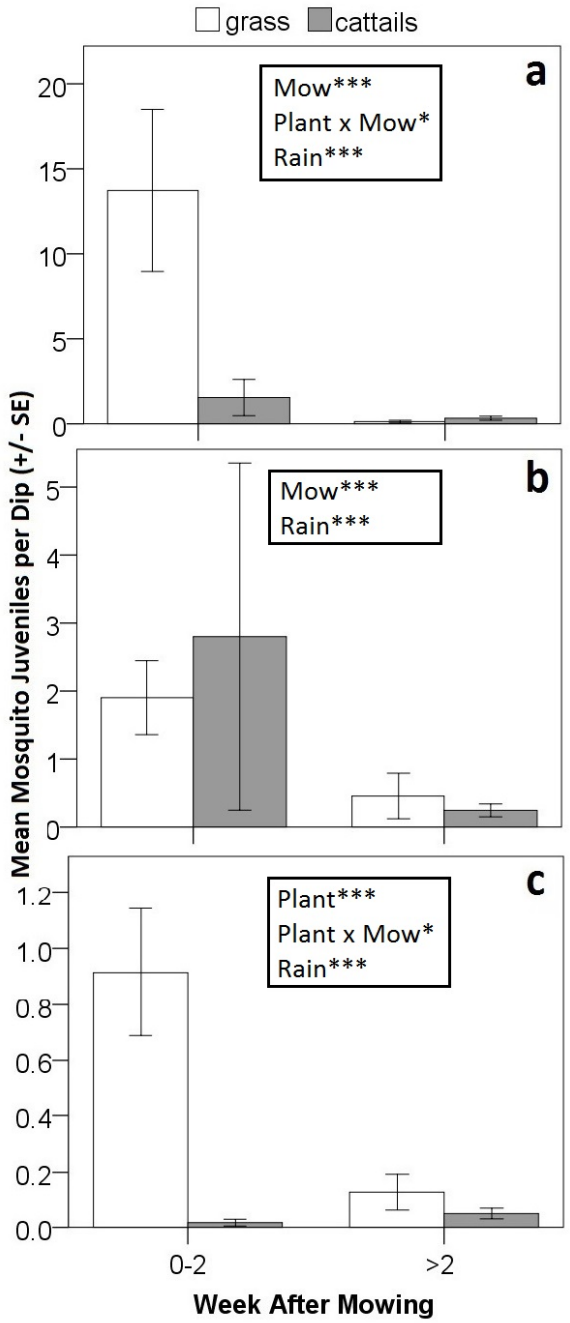

Figure 2. Effect of plant type and recent mowing on the average numbers of *Cx. pipiens* larvae (a), *Cx. restuans* larvae (b), and *Culex* sp. pupae (c) collected from stormwater ditches and detention basins, 03 June to 16 August, 2013 (observations from dry sites excluded from calculation of average values). Significant effects and interactions in GLMMs are shown for each group [**Plant** = dominant plant type (grass or cattails), **Mow** = vegetation mowed ~≤ 2 weeks prior to sampling, **Rain** = cumulative rainfall 0 to 8 days prior to sampling; * p-value < 0.05, ** p-value < 0.01, *** p-value < 0.001].

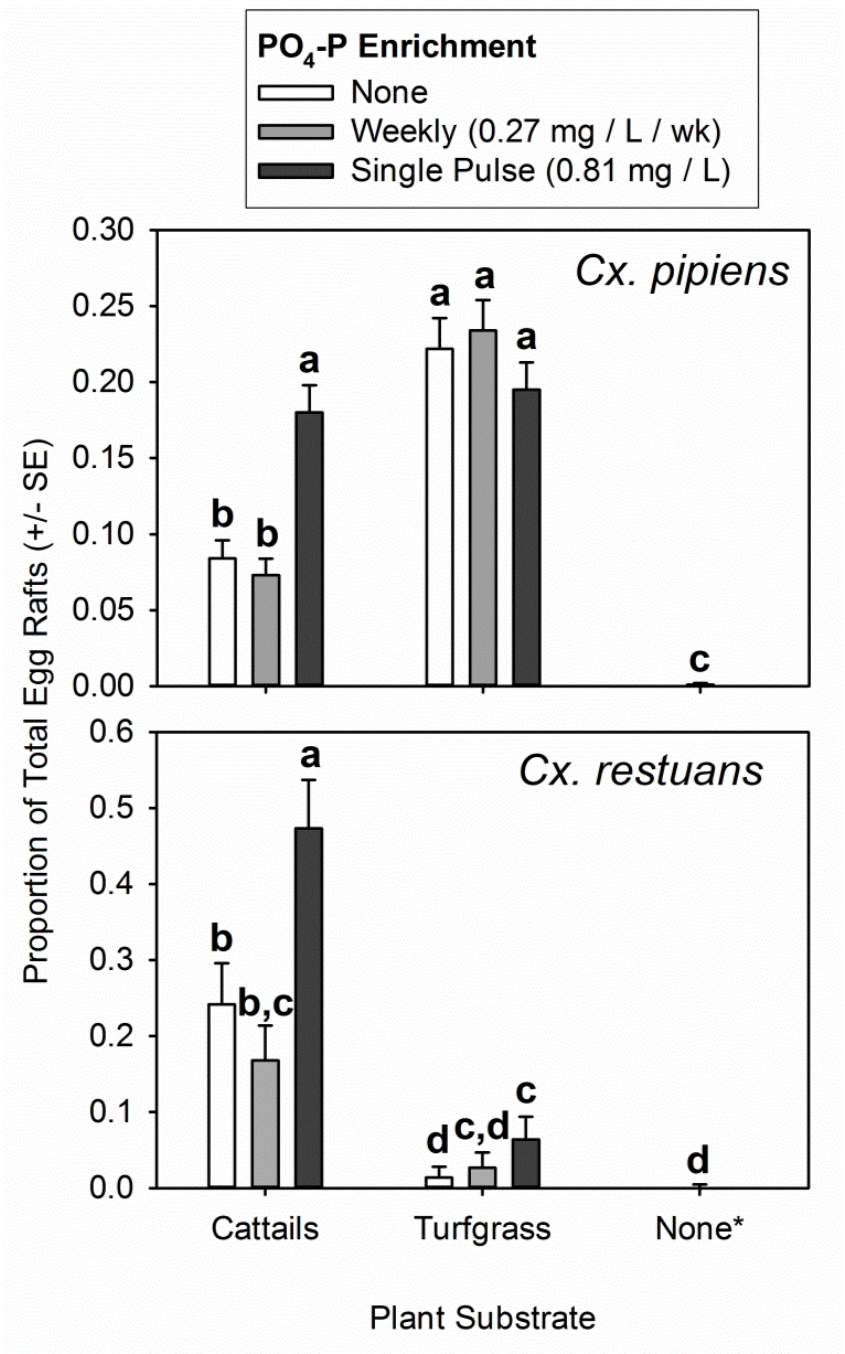

Figure 3. Influence of orthophosphate enrichment on the proportion of eggs rafts of Cx. *pipiens* and *Cx. restuans* collected ovitraps baited with either cattails or turfgrass clippings (4.5 g / l). For each species, columns with the same letter are not significantly different (P>0.05; post-hoc test in GLMM with Bonferroni correction).

* Traps without plant substrate were enriched with 0.27 mg PO_4-P / L / week.

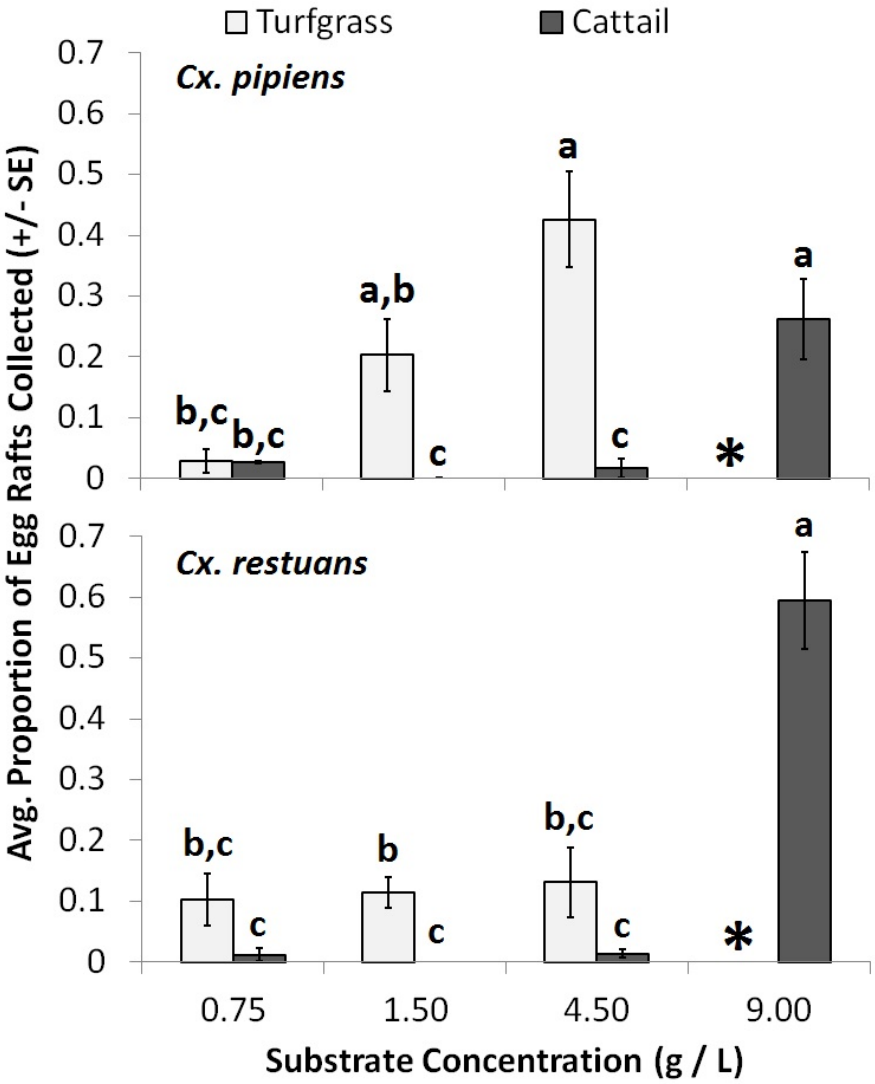

Figure 4. Proportion of *Cx. pipiens* and *Cx. restuans* egg rafts collected from infusions containing varying concentrations of turfgrass or cattail substrate. Turfgrass was not evaluated at the 9.0 g / L concentration (indicated by the asterisk in figure). For each species, columns with the same letter are not significantly different (P>0.05; post-hoc test in GLMM with Bonferroni correction).

Table 1. General linear models evaluating the effects of larval rearing density and orthophosphate enrichment (0.81 mg PO_4-P / L) of cattail infusions (4.5 g / l) on the duration of juvenile development, adult longevity and wing length of male and female *Cx. pipiens*.

Response Variable	Parameter[1]	Males			Females		
		Coeff. (± 95% CI)	t	p-value	Coeff. (± 95% CI)	t	p-value
1 / duration of juvenile development	Intercept	0.098 (0.002)	52.8	<0.001	0.084 (0.003)	32.5	<0.001
	Orthophosphate	0.000 (0.002)	<0.1	0.946	0.003 (0.003)	0.9	0.377
	Larval Density	0.004 (0.002)	1.8	0.089	**0.011 (0.003)**	**3.7**	**0.003**
1 / adult longevity	Intercept	0.581 (0.017)	34.6	<0.001	0.397 (0.026)	15.2	<0.001
	Orthophosphate	**-0.064 (0.019)**	**-3.3**	**0.006**	0.065 (0.030)	2.2	0.051
	Larval Density	**-0.186 (0.019)**	**-9.6**	**<0.001**	-0.007 (0.030)	-0.2	0.817
wing length	Intercept	0.361 (0.004)	100.4	<0.001	0.283 (0.004)	75.2	<0.001
	Orthophosphate	-0.009 (0.004)	-2.1	0.059	0.003 (0.004)	0.7	0.511
	Larval Density	**-0.039 (0.004)**	**-9.4**	**<0.001**	**-0.014 (0.004)**	**-3.2**	**0.007**

[1] The interaction term (orthophosphate x larval density) was excluded from the final models as it was not a significant effect for any of the response variables for males or females.

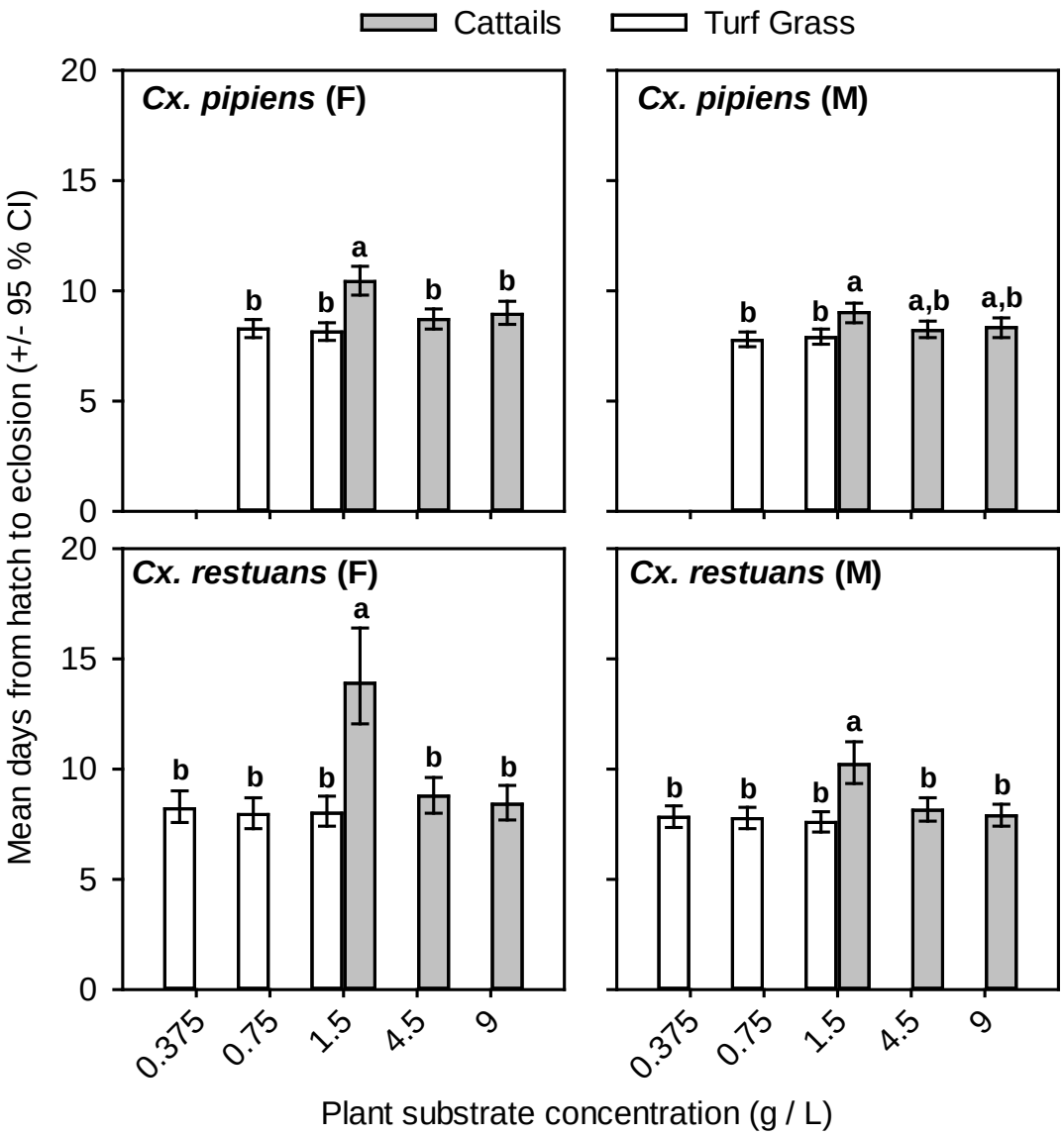

Figure 5. Juvenile development time of male (M) and female (F) *Cx. pipiens* and *Cx. restuans* reared in infusions containing varying concentrations of turfgrass or cattail substrates. Columns within each group that share the same letter were not significantly different (P>0.05; post-hoc test with sequential Bonferroni correction in GLM using 1 / Days to Eclosion as the response variable).

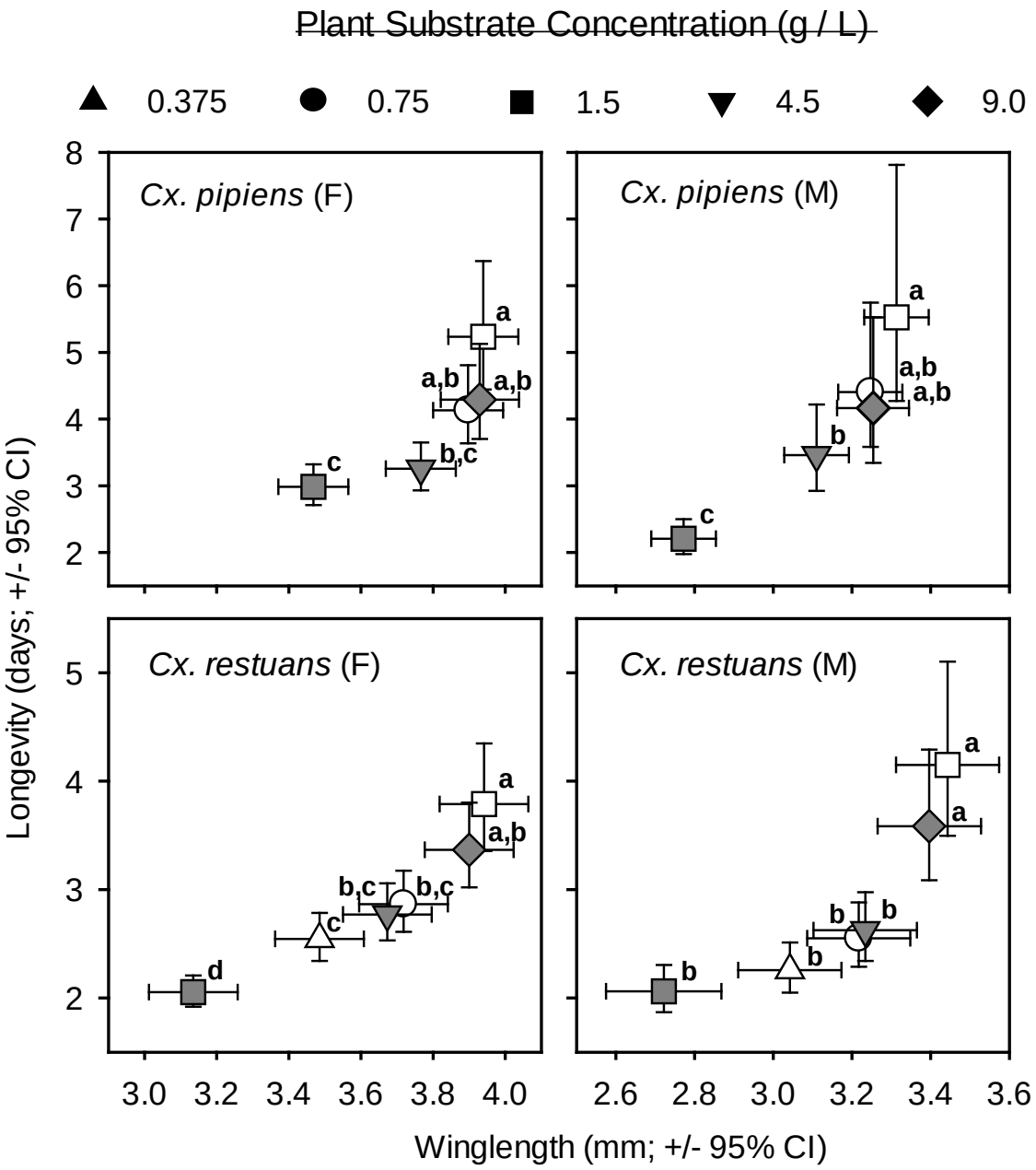

Figure 6. Average wing length and longevity of adult male (M) and female (F) *Cx. pipiens* and *Cx. restuans* reared in infusions containing varying concentrations of turfgrass (open symbols) and cattails (closed symbols). Average longevity within treatments for each group that share the same letter were not significantly different (P>0.05; post-hoc test with sequential Bonferroni correction in GLM using 1 / days surviving as the response variable). A significant linear relationship was observed between adult winglength and transformed adult longevity (1 / days surviving) for male (R^2 = 0.546) and female (R^2 = 0.512) *Cx. pipiens*, and male (R^2 = 0.436) and female (R^2 = 0.509) *Cx. restuans*.

3-D Modeling of the Chicago Sanitary and Ship Canal from the Cal-Sag Junction to the Lockport Powerhouse, Illinois

Basic Information

Title:	3-D Modeling of the Chicago Sanitary and Ship Canal from the Cal-Sag Junction to the Lockport Powerhouse, Illinois
Project Number:	2014IL291S
USGS Grant Number:	G14AP00005
Sponsoring Agency:	COE_UIL
Start Date:	12/1/2013
End Date:	10/31/2014
Funding Source:	104S
Congressional District:	IL-015
Research Category:	Climate and Hydrologic Processes
Focus Category:	Models, Hydrogeochemistry, Hydrology
Descriptors:	None
Principal Investigators:	Marcelo Horacio Garcia

Publications

There are no publications.

Progress report could not be obtained by the report deadline.

Information Transfer Program Introduction

None.

Transferring Information to the People of Illinois

Basic Information

Title:	Transferring Information to the People of Illinois
Project Number:	2014IL277B
Start Date:	3/1/2014
End Date:	2/28/2015
Funding Source:	104B
Congressional District:	IL-013
Research Category:	Not Applicable
Focus Category:	Education, Management and Planning, None
Descriptors:	None
Principal Investigators:	Lisa Merrifield

Publications

There are no publications.

Illinois Water Resources Center
Technology Transfer Report

In 2014, IWRC engaged in a number of technology transfer and outreach strategies. Below are details on our significant impacts and accomplishments.

Illinois launches nutrient loss reduction strategy

Relevance: Gulf hypoxia starts in the Midwest. U.S. EPA has charged all Midwest states with developing plans to help reduce nutrient run off into streams and rivers. In 2014, Illinois EPA began to develop such a strategy that would use existing resources to improve local and Gulf water quality.

Response: Illinois EPA and the Illinois Department of Agriculture asked IWRC to facilitate the strategy development. Through a series of meetings with a policy working group, IWRC and the state agencies worked through issues of agricultural runoff, urban point sources discharge, and urban non-point runoff. The policy workgroup looked at both existing resources and infrastructure and at funding, technological, and societal gaps that need to be addressed over the year long process.

Results: In November of 2014, Illinois EPA released the Illinois Nutrient Loss Reduction strategy for public comment. The comment period ended in January and those comments are now being incorporated. In 2015 IWRC will help Illinois EPA and the Illinois Department of Agriculture begin to implement the strategy.

IWRC provides technical assistants to rural communities

Relevance: Small communities and rural areas face real challenges obtaining and maintaining safe water supplies. Where larger metropolitan areas have more resources and more funding opportunities, small communities often do not have the funding to train personnel and maintain drinking water and waste water infrastructure. Technical assistance can be one inexpensive and convenient way to support small community water quality challenges.

Response: IWRC partners with the Illinois State Water Survey on two websites funded by the US EPA, in partnership with the Rural Community Assistance Partnership (RCAP), which address education and outreach barriers to safe water. Both websites share a key mission, to distill best-available information into user-friendly content and lessons that serve small communities and rural areas across the United States.

Results:

PrivateWellClass.org centers on a 10 week email course (The Private Well Class) that teaches homeowners how to properly care for and maintain their water well. This includes introductory information on geology, well contamination and water testing. The site is designed to serve the 45 million Americans who rely on a private well for their drinking water and includes a pre- and post-test

quiz to test knowledge improvement. Understanding how to prevent groundwater contamination, both on the property and via cross-connection control, is addressed in the lessons as well as during a series of three live webinars. To date, more than 3,860 individuals have signed up for The Private Well Class. During 2015-2016, the program will be expanded to include a nationwide training program for public health professionals on private well risk assessment and public outreach.

SmallWaterSupply.org is a one-of-a-kind resource website on many drinking water, wastewater, and utility management topics. It aggregates information from across the web so that users can save significant time finding the documents, training events, and news they need. The site is designed to serve water operators in small communities and tribes across the United States. The website team has indexed more than 80,000 events and 15,000 documents to date. A biweekly newsletter reaches nearly 1,300 water industry professionals with the most timely and relevant information available. During 2015-2017, SmallWaterSupply.org will collaborate with two water research centers to improve adoption of innovative treatment technologies at small water systems.

Over 200 attend Illinois Water 2014

Relevance: Located in the heart of the University of Illinois campus, IWRC interacts with water resource professionals at most major universities in the state, the USGS Water Science Center, and state agencies. IWRC is strategically places to bring together those professionals to share the latest science and provide networking opportunities for key partners and students.

Response: In 1998, IWRC started the Illinois Water Conference series. Every other year, IWRC invites researchers and water resource professionals and students around the state to present papers and poster on the state of their scientific pursuits.

Results: In 2015 over 200 professionals attended the Illinois Water Conference. Thirty students presented papers or posters on their work. IWRC used the water conference to kick of our 50th anniversary celebration. Sessions dealt with water supply planning, water quality monitoring, nutrient loss, and other timely issues.

IWRC co-chairs the 2015 Illinois River Conference

Relevance: The Illinois River bi-sects the state, draining a large section of central Illinois from the Chicago metro area to intensely managed rural landscapes. A principal tributary of the Mississippi River, the Illinois River is a key transportation route as well as an avenue for tourism. The health and productivity of the river is important to a wide variety of stakeholders in the region including industry, municipal, and environmental users.

Response: The Illinois River Conference has been a biennial mainstay in Illinois. IWRC has co-sponsored the Illinois River conference since the mid-90s, chairing sessions, designing programs, and organizing workshops at each conference. In 2014, IWRC became a co-chair for the conference and is not integral in the planning and organization of the event.

Results: In 2015, the Illinois River Conference will draw people from all sectors to talk about policy and management of the river. The conference has a loyal following, but with increased IWRC engagement, will hopefully draw an even broader audience.

USGS Summer Intern Program

None.

Student Support					
Category	**Section 104 Base Grant**	**Section 104 NCGP Award**	**NIWR-USGS Internship**	**Supplemental Awards**	**Total**
Undergraduate	8	0	0	0	8
Masters	16	2	0	0	18
Ph.D.	21	0	0	0	21
Post-Doc.	1	0	0	0	1
Total	46	2	0	0	48

Notable Awards and Achievements

In 2014, IWRC leveraged base funding to achieve an operating budget of over $400,000.

We funded or supported over 50 students through small grants and the Illinois Water Conference.

We funded 9 research projects, helping to identify: best management practices in detention ponds to minimize mosquito populations, flame retardant levels in Illinois water, and sources and distribution of microplastics in water. Another project helped to better understand ecosystem services in constructed wetlands. (See research introduction for additional projects and introductions)

IWRC facilitated the state of Illinois Nutrient Loss Reduction Strategy. (See Information Transfer for details)

IWRC reached 1300 rural water industry professionals with continuing education and technical assistance resources.(See Information Transfer for details)

Almost 4000 private well owners have participated in the IWRC designed Private Well Class to learn about how to maintain a safe drinking water supply.

Publications from Prior Years

1. 2012IL250B ("The Impacts of Cellulosic Biofuel Feedstock Production on Midwest US Hydrologic Cycle") - Articles in Refereed Scientific Journals - VanLocke, A., T.E. Twine, M. Zeri, and C. J. Bernacchi. 2012. A regional comparison of water use efficiency for miscanthus, switchgrass and maize. Agricultural and Forest Meteorology, 164(2012), 82-95.
2. 2011IL219B ("Influence of Algal/ Bacterial Interactions on Denitrification in Stream Biofilms") - Articles in Refereed Scientific Journals - Kalscheur, K.N., M. Rojas, C.G. Peterson, J.J. Kelly, and K.A. Gray. 2012. Algal exudates and stream organic matter influence the structure and function of denitrifying bacterial communities. Microbial Ecology. 64: 881-892.
3. 2010IL207B ("Linking microbial community structure to water quality function: investigating nitrogen cycling during early floodplain development") - Articles in Refereed Scientific Journals - Peralta, A.L., J.W. Matthews, D. N. Flanagan, and A.D. Kent. 2012. Environmental factors at dissimilar spatial scales influence plant and microbial communities in restored wetlands.
4. 2010IL207B ("Linking microbial community structure to water quality function: investigating nitrogen cycling during early floodplain development") - Articles in Refereed Scientific Journals - Peralta, A.L., S. Ludmer, and A.D. Kent. 2013. Hydrologic history constrains microbial community composition and nitrogen transformations under experimental drying/wetting treatments. Soil Biology & Biochemistry 66: 29-37.
5. 2010IL207B ("Linking microbial community structure to water quality function: investigating nitrogen cycling during early floodplain development") - Articles in Refereed Scientific Journals - Peralta, A.L., J.W. Matthews, and A.D. Kent. 2014. Habitat specialization along a wetlands moisture gradient differs between ammonia-oxidizing and denitrifying microorganisms. Microbial Ecology 68:339-350 doi: 10.1007/s00248-014-0407-4
6. 2010IL207B ("Linking microbial community structure to water quality function: investigating nitrogen cycling during early floodplain development") - Articles in Refereed Scientific Journals - Peralta, A.L., S. Ludmer, J.W. Matthews, and A.D. Kent. 2014. Bacterial community response to changes in soil redox potential along a moisture gradient in restored wetlands. Ecological Engineering 73: 246–253.
7. 2010IL202B ("An Agent-Based Model of Nitrogen and Carbon Trading at the Watershed Scale") - Articles in Refereed Scientific Journals - Ng, T. L., J. W Eheart, X. Cai, J. B. Braden, and G. F. Czapar. 2012. Agronomic and stream nitrate load responses to incentives for bioenergy crop cultivation, reductions of carbon emissions and fertilizer use. Journal of Water Resources Planning and Management, ASCE, 140(1), 112–120.
8. 2010IL202B ("An Agent-Based Model of Nitrogen and Carbon Trading at the Watershed Scale") - Articles in Refereed Scientific Journals - Ng, T. L., J. W Eheart, X. Cai, and J. B. Braden. 2011. An agent-based model of farmer decision-making and water quality impacts in a watershed under markets for carbon allowances and second-generation bioenergy crops. Water Resources Research, 47, W09519, doi:10.1029/2011WR010399.
9. 2009IL184G ("Attachment and Transport Mechanisms of Cryptosporidium parvum oocysts in karst aquifers: Role of Natural Organic Matter and Iron Oxide") - Articles in Refereed Scientific Journals - Liu, Y., M. S. Kuhlenschmidt, T. B. Kuhlenschmidt, and T. H. Nguyen. 2010. Composition and conformation of Cryptosporidium parvum oocyst wall surface macromolecules and their effect on adhesion kinetics of oocysts on quartz surface. Biomacromolecules, 11(8), 2109–2115, doi: 10.1021/bm100477j.
10. 2009IL184G ("Attachment and Transport Mechanisms of Cryptosporidium parvum oocysts in karst aquifers: Role of Natural Organic Matter and Iron Oxide") - Articles in Refereed Scientific Journals - Liu, Y., C. Zhang, M. Hilpert, M. S. Kuhlenschmidt, T. B. Kuhlenschmidt, and T. H. Nguyen. 2012. Transport of Cryptosporidium parvum oocysts in a silicon micromodel. Environmental Science and Technology, 6(3), 1471–1479, doi: 10.1021/es202567t.

11. 2009IL171B ("Nitrate Remeidation by Iron Redox Reactions in Soils") - Articles in Refereed Scientific Journals - Su, K., A.Radian, Y. Mishael, L. Yang, and J.W. Stucki. 2012. Nitrate reduction by redox- activated, polydiallyldimethylammonium-exchanged ferruginous smectite. Clays and Clay Minerals, 59, 464-472.
12. 2009IL171B ("Nitrate Remeidation by Iron Redox Reactions in Soils") - Articles in Refereed Scientific Journals - Pentrak, M., L. Pentráková, A. Radian, Y.G. Mishael, and J.W. Stucki. 2014. Nitrate reduction by redox-modified smectites exchanged with chitosan. Clays and Clay Minerals, 62, 403-414.