EVALUATION OF EPA LEVEL I, II, AND III ASSESSMENTS AND THE EFFECTS OF LAND USE ON WETLAND COMMUNITIES

By

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EVALUATION OF EPA LEVEL I, II, AND III

ASSESSMENTS AND THE EFFECTS OF LAND USE

ON COMMUNITIES ON WETLAND COMMUNITIES

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Abstract: Effective tools are needed to monitor and assess wetland ecosystems. The U.S. Environmental Protection Agency (EPA) proposed a three level framework that includes landscape assessments (Level I), rapid assessments (Level II), and intensive surveys of wetland communities (Level III). The EPA conducted a national wetland condition assessment in 2011 using a new rapid assessment method (USA-RAM) that was not calibrated to specific regions. The objectives of this study were to compare the relationships between USA-RAM to the Level I and III assessments, analyze the influence of spatial scale on Level I analysis, and determine whether within-wetland or landscape features were more important in structuring macroinvertebrate communities. Plant communities from 22 wetlands of varying levels of landscape disturbance were surveyed in 2012 and 2013 and macroinvertebrate communities were surveyed twice in the 2013. Each wetland was assessed using USA-RAM. I analyzed land use in the buffer surrounding each wetland using the 2012 CropScape dataset at four spatial scales (100m, 300m, 500m, and 1000m). I found significant relationships between Level I assessments (e.g., the Landscape Development Intensity index) and the Level II assessment (USA-RAM) with the strongest relationships occurring within the 100m buffer around the wetlands. This is an important finding suggesting that computer assessments of the buffer can be used to predict stressors to wetlands. However, there were no significant relationships between the Level I and either the Level II (USA-RAM) or the Level III (plant and macroinvertebrate) assessments. Canonical Correlation Analysis (CCA) showed that land use within the 100m buffer explained the most variation among plant communities, while the 300m buffer explained the most variation among the aquatic insect genera. Land use explained more of the variation in aquatic insect genera than within-wetland variables, showing that even relatively less mobile and smaller taxa (as compared to waterfowl or muskrats) are affected by land use disturbances. Combined, my results suggest that land use can predict wetland disturbance as measured by the USA-RAM, but the USA-RAM does not correspond to wetland community condition; therefore the USA-RAM needs to be calibrated and potentially modified to accurately assess wetland community condition in the region.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

Figure Page

CHAPTER I

EVALUATION OF EPA LEVEL I, II, AND III ASSESSMENTS AND THE EFFECTS OF LAND USE ON WETLAND COMMUNITIES

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RAM) with the strongest relationships occurring within the 100m buffer around the wetlands. This is an important finding suggesting that computer assessments of the buffer can be used to predict stressors to wetlands. However, there were no significant relationships between the Level I and either the Level II (USA-RAM) or the Level III (plant and macroinvertebrate) assessments. Canonical Correlation Analysis (CCA) showed that land use within the 100m buffer explained the most variation among plant communities, while the 300m buffer explained the most variation among the aquatic insect genera. Land use explained more of the variation in aquatic insect genera than within-wetland variables, showing that even relatively less mobile and smaller taxa (as compared to waterfowl or muskrats) are affected by land use disturbances. Combined, my results suggest that land use can predict wetland disturbance as measured by the USA-RAM, but the USA-RAM does not correspond to wetland community condition; therefore the USA-RAM needs to be calibrated and potentially modified to accurately assess wetland community condition in the region.

Keywords USA-RAM · Wetlands · Land-use · Macroinvertebrates · Oklahoma

Introduction

Wetlands were not always thought of as valuable ecosystems and as a result they have been dramatically altered by human activities (Dahl 1990). Early settlers thought that wetlands were wastelands that prohibited further human settlement. This mindset led to the Swamp Lands Acts of the mid 1800s, which granted states the right to convert wetlands to agriculture. In the Midwestern United States, up to 90% of wetlands have been drained for agricultural land expansion, while across the United States, as a whole, over 50% of wetlands have been lost (Dahl 1990).

As wetlands are lost, the ecological services that they provide such as flood abatement, nutrient retention, aquifer recharge, and biodiversity enhancement are also lost (Smith *et al.* 2011). Wetlands contribute up to 40% of these services, while covering only 1.5% of the planet's surface (Zedler 2003). Consequently, there has been an increased interest in restoring and creating wetlands to help mitigate the effects of activities associated with agricultural land use (Dahl 2011). The Clean Water Act in 1972 was the first act that really started wetland protection. More recent legislation, such as the Farm Bill and wetland related programs including the Conservation Reserve Program, have helped to provide incentives to conserve and restore wetlands.

Assessing Wetlands

Effective monitoring and assessment programs are needed to manage and inventory the wetland resources that remain (Whigham 1999, Stevens and Jensen 2007, Stein *et al.* 2009). The U.S. Environmental Protection Agency (EPA) proposed a three tiered system of evaluating wetland quality (EPA 2006). The first tier is a Level I assessment, which uses a landscape analysis to assess wetland quality. This can be done remotely, usually using data obtained through geographic information systems (GIS). Level I assessments can provide a quick, coarse gauge of wetland quality in a region using few resources. The second tier is a Level II assessment, which uses a relatively quick on-the-ground assessment of potential stressors to a wetland with simple metrics, such as the U.S.A. Rapid Assessment Method, California Rapid Assessment, etc. The metrics are typically scored using rapid assessment protocols that provide an overall assessment score for individual wetlands. The last tier is a Level III assessment, which is an intensive field survey of wetlands by obtaining biological data and/or hydrogeomorphic functions as indicators of biologic integrity. Using a combination of these assessments has the potential to provide beneficial information about the quality of a wetland (EPA 2006, Reiss and Brown 2007, Stein *et al.* 2009). Many state and tribal agencies use a combination of these methods to monitor wetlands (EPA 2006).

In 2011, the Environmental Protection Agency (EPA) conducted a national wetland condition assessment program (NWCA) to survey the status of the nation's wetlands. They used all three wetland assessment levels to conduct their analysis and developed a new Level II assessment called the U.S.A. Rapid Assessment Method (USA-RAM). However, when the USA-RAM was created, it was not calibrated for wetlands within specific regions, and it was to be calibrated and tested during the NWCA (Scozzafava *et al.* 2011). Thirteen metrics, some of which have been useful in other rapid assessments protocols, were included in the USA-RAM (EPA 2011b). Therefore, it is important to determine if this Level II assessment method accurately describes disturbances to and the condition of wetlands in specific regions of the country.

Disturbance Surrounding Wetlands

Many wetlands, especially those in the Midwestern United States, are proximal to human disturbances in the landscape that come from agriculture (e.g., row crops, orchards, and pasture) as well as from roads and urbanization. These land use practices fragment and change the landscape, which affects the movement of species between habitat patches (Forman and Alexander 1998, Debinski and Holt 2000, Houlahan *et al.* 2006). Agricultural wetlands also receive high levels of sedimentation and nutrients through runoff from the surrounding landscape (Crumpton *et al.* 1993, Luo *et al.* 1997). High sedimentation rates change the hydrology of the wetland (Luo *et al.* 1997, Smith *et al.* 2011) altering the species composition within the wetland. Sedimentation can also bury estivating eggs and the seed bank (Gleason *et al.* 2003), and potentially clog gills of macroinvertebrates (Swenson and Matson 1976). Wetland vegetation helps retain much of the nutrients from runoff, preventing downstream impacts (Weisner *et al.* 1994). While this absorption and retention of nutrients is one of the key ecological roles of wetlands, it can also alter the species composition within wetlands to having more invasive or weedy species in higher nutrient systems (Bedford *et al.* 1999).

Urbanization and impervious surfaces within the landscape also have the potential to affect wetlands. These areas have high levels of runoff and increased pollutants from vehicles, construction, and industry (Hogan and Walbridge 2007, Lee *et al.* 2012), which may alter the hydrology of wetlands that are influenced by these land uses. Urbanization and impervious surfaces also lead to higher local land surface temperatures (Yuan and Bauer 2007), which leads to higher local evaporation rates. Impervious surfaces decrease the amount of water entering the ground (Arnold and Gibbons 1996), thereby increasing the need for wetlands to recharge groundwater and aquifers. All of these disturbances within the landscape likely have an adverse effect on wetland quality.

Level I and II assessments typically score these land use disturbances negatively. Some assessments score different land uses as having stronger or weaker influences on wetland condition than others (e.g., cropland, developed land, pasture; Brown and Vivas

2005, EPA 2011b, Dvorett *et al.* 2013). Knowledge of how these land uses and other disturbances affect wetland communities have helped to create these Level I and II assessments, but may not work well in analyzing disturbances across different regions.

Communities as Indicators of Disturbance

Biological communities are often used to assess land use disturbance. While a number of taxa have been used to study the effects of disturbance in a system (see Appendix 1), two of the most commonly used taxa are plants and macroinvertebrates. With respect to plant communities, a number of metrics including species richness, invasive species abundance, plant based indices of biotic integrity, and the floristic quality assessment index have been used to assess disturbances in wetlands (see Appendix 2). Many studies have found that communities will shift in response to different levels and types of human disturbance to having more invasive species or having lower species richness and diversity in more highly disturbed sites (Chipps *et al.* 2006, DeKeyser *et al.* 2009, Tsai *et al.* 2012, and Appendix 2). The types of disturbances that have been used to indicate these community changes include the amount of agriculture and impervious surfaces surrounding the wetlands (Mensing *et al.* 1998, Whited *et al.* 2000, Chipps *et al.* 2006, Rooney *et al.* 2012, Petersen and Westmark 2013, and Appendix 1). For example, Mensing *et al.* (1998) studied plant communities in 15 riparian wetlands and found that shrub carr vegetation diversity was strongly related to agriculture at the 1000m scale, where diversity decreased as agriculture increased. Rooney *et al.* (2012) found that a plant based index of biological integrity decreased as road cover increased in the landscape surrounding wetlands in Alberta. These and other studies highlighted in Appendix 2 show that agriculture and impervious surfaces are important forms of disturbance that affect plant communities.

Another group of organisms that have been used extensively to study the effects of land use and human disturbance are macroinvertebrates (see Appendix 3). Macroinvertebrates may respond to disturbance in a number of ways including having a greater proportion of tolerant species, lower species richness, and/or lower community evenness (Hall *et al.* 2004, Chipps *et al.* 2006, Meyer 2012, and Appendix 3). Macroinvertebrates have been used to assess disturbance including Indices of Biologic Integrity (IBI), the dominance of specific taxa (e.g., Ephemeropterans, Chironomids, Corixids, etc.), species richness, abundances, and weighted biomass (Appendix 3). However, when Mensing *et al.* (1998) studied 15 riparian wetlands in Minnesota, they found that macroinvertebrates were relatively unresponsive to land use disturbance, but were more responsive to some of the within-wetland characteristics, such as water quality. Another example from Meyer (2012), who studied 58 depressional wetlands in north-central Oklahoma, found that the local, within-wetland factors were better at modeling macroinvertebrates than landscape factors (Meyer 2012). These studies and others highlighted in Appendix 3 show that relative to plants, macroinvertebrates seem to be more responsive to water quality than landscape features. This response may be due to how the studies measured and recorded disturbance by either categorizing the majority of land use around wetlands (Tangen *et al.* 2003, Hall *et al.* 2004, Chipps *et al.* 2006, Campbell *et al.* 2009, Reece and McIntyre 1009) or by using large landscape analyses that may not show the impacts that are relevant to the wetland (Mensing *et al.* 1998, Angeler *et al.* 2008). One would expect that in more disturbed sites, the communities would shift to having an abundance of tolerant species such as Chironomids, while less disturbed sites would contain more species that are sensitive to disturbance (e.g., Trichopterans and Ephemoropterans; Merrit *et al.* 2008).

Considering Scale in Wetland Assessments

Another important factor to consider in wetland assessment is how spatial scale impacts the relationships between the different types of assessments (e.g., Level I versus Level II). Different taxa interact with their habitat at different spatial extents depending on the size and dispersal ability of the organism (Levin 1992, Rooney *et al.* 2012). Therefore, the extent at which landscape disturbances will affect a community will depend on the organisms that are sampled and used to measure disturbance. A smaller, or less mobile organism will most likely be affected by disturbance at a close range (e.g., *Daphnia* species), while a larger, or more mobile organism can be affected by a disturbance happening at greater distances (e.g., waterfowl species; Levin 1992). For example, Rooney *et al.* (2012) found that plant based IBI's were best predicted by land use disturbances within 100m of wetlands, while bird-based IBI's were best predicted by land use disturbances within 500m of a wetland. This example and other studies highlighted in Appendix 4 document how different communities are affected by disturbance at different scales. In general, there is a tendency for stationary species, such as plants, to be affected by local disturbances, while more mobile species, such as birds, are affected by more regional and larger scales (Mensing *et al.* 1998, Whited *et al.* 2000, Rooney *et al.* 2012). These examples show that it is important to assess different scales during a Level I assessment to observe how disturbance affects different communities.

Predictive Models

A strong understanding of the relationships between land use and communities (as described above) may allow for the development of predictive models that relate Level I assessment variables to Level II and/or Level III assessment variables. Creating models to predict wetland condition using readily available landscape data (e.g., GIS from desktop computer) is valuable because these models can provide an initial assessment of wetland condition in the absence of field data. Land use models have the potential to identify areas of importance (e.g., high diversity) or areas of concern to help direct limited resources within different agencies.

One major consideration when creating land use models is how to calculate disturbance within the surrounding landscape. Not all forms of disturbance affect all of the taxa within a wetland equally (Appendix 1-4). For example, urban or industrial development would likely have a stronger influence on wetland condition than grazing land or a park (Rooney *et al.* 2012). One way to estimate disturbance is to calculate the percentage of the surrounding landscape that contains a specific type of disturbance (e.g., % agriculture, % urban). An alternative way to calculate landscape disturbance that may be more effective is the use of indices that assess the land use as a whole such as the Landscape Development Intensity index (LDI). The LDI was created to assess wetland disturbance and weights land use types more strongly or weakly (see Table 1) (Brown and Vivas 2005). The LDI provides a mechanism wherein disturbance based on a combined metric of the different land use types within a buffer surrounding a wetland can be used to compare it to community metrics.

Goals of this Study

When conducting the nation-wide wetland condition assessment, the EPA created a new rapid assessment method (USA-RAM). The USA-RAM was not calibrated for specific geographic regions, and therefore, may not accurately assess the condition and disturbances to wetlands in all regions of the country. I evaluated the relationships between Level I, II, and III assessments to assess the usefulness of using USA-RAM assessments in Oklahoma wetlands. I also assessed how different spatial scales of a Level I assessments influenced relationships with the Level II (USA-RAM) and Level III (plant and aquatic insect communities) assessments. Lastly, I assessed whether land use or within-wetland characteristics were more important in structuring wetland aquatic insect communities. This research is important because it evaluates how well the USA-RAM relates to wetland condition at a state scale. It also adds to the body of literature on how disturbances affect wetland communities and shows the importance of land use in structuring macroinvertebrate communities.

Methods

Wetland Site Selection

Wetlands that represented a gradient in land use characteristics were chosen out of a database of 50 wetlands that were previously sampled in Oklahoma in 2012 or 2013. The locations of all 50 wetlands were first entered into ArcGIS and a 500 meter buffer was placed around each wetland using ArcGIS 10. The national land cover dataset from the USGS (Price *et al.* 2006) was used to calculate the area of the major land use categories within each buffer. Land use disturbance was calculated using the Landscape

Development Index (LDI) which provides an estimate of disturbance in the buffer based on the proportion of land use types and predefined coefficients (see Table 1) for each land use type. The LDI is calculated according to:

$$\mathbf{LDI_{total} = \Sigma\, LU_{\mathit{i}} \cdot LDI_{\mathit{i}}},$$

where $\mathbf{LDI_{total}}$ = LDI value for the wetland, $\mathbf{LU}_{\boldsymbol{i}}$ = percent of the total area for land use type $\boldsymbol{i}$, and $\mathbf{LDI}_{\boldsymbol{i}}$ = landscape development intensity coefficient for land use $\boldsymbol{i}$. Lower LDI values indicate less disturbance in the surrounding landscape, while higher LDI values indicate more human disturbance to the surrounding land (Brown and Vivas 2005). LDI places weights on different land use types to determine disturbance and has been shown to work well for analyzing disturbance to wetlands and their communities (Cohen *et al.* 2004, Mack 2006, Chen and Lin 2011).

I selected 22 of these previously sampled wetlands that represented a range of wetland conditions based on the initial LDI values from low disturbance to high disturbance in their surrounding landscape.

<u>Study Area</u>

Twenty two wetlands were sampled in Oklahoma twice for invertebrates in the summer of 2013 (May and July), and once in either the summer (May through August) of 2012 or 2013 for vegetation. The latitude of the wetlands ranged from 33.800710°N to 36.982990°N and the longitude ranged from 94.594020°W to 98.075360°W. The wetland sample covered seven different ecoregions (Figure 1) and included hydrogeomorphic wetland classifications of both riverine and depressional wetlands (EPA 2011a). Wetlands ranged in size from 0.19 to 10.82 ha.

Level I Assessment

I first drew polygons along the edges of each wetland in ArcGIS using orthorectified aerial photographs. The polygons were then projected using the Albers North American Equal Area Conic projection so wetland area could be determined. I then drew buffers around each wetland polygon at different distances (100m, 300m, 500m and 1000m radii) to determine how disturbance in each buffer distance related to the wetland communities following the different radii used by Rooney *et al.* (2012). The 2012 CropScape land use dataset (USDA National Agricultural Statistics Service Cropland Data Layer 2012), which provides land use categories and crop type for each 30 m^2 pixel (e.g., Deciduous Forest, Developed Low Intensity, Corn, Soybean, etc.) was input and clipped to the different buffers using the extract by mask tool in ArcGIS. I then used the extract by mask tool to clip the land use data to each buffer size for each wetland. Then I used the tabulate area tool to calculate the areas of each land use type within each radius for CropScape data, which created a table for each radius of the amount of area that contained each land use and crop type. I then used the table created to calculate the LDI for each buffer radii. I also calculated land use scores (LUS), which is calculated the same as LDI, using coefficients developed for Oklahoma oxbow and riparian wetlands from Dvorett *et al.* (2013) (Table 1). The LUS is scored on a different scale from 0 to 1, where 1 represents no disturbance and 0 is highly disturbed. In addition to the LDI and LUS, I calculated the % total area agricultural lands for each row crops, pasture/hay, and row crops + pasture/hay. I also calculated % area developed, which was the sum of all developed land use types, and the % area of human disturbance, which was calculated as the sum of % area developed, the % area row crop, and the % area pasture/hay within each radius of the different wetland radii.

Level II Assessment

For the Level II assessment, I used the USA Rapid Assessment Method (USA-RAM; EPA 2011b). This assessment contains 12 different metrics that measure the condition and stress to a wetland. These stress metrics included stress to the buffer zone (Metric 3), alterations to the hydroperiods (Metric 8), stress to water quality (Metric 9), habitat/substrate alterations (Metric 10), percent cover of invasive plants (Metric 11), and vegetation disturbance (Metric 12). These metrics evaluated the amount of each stressor present to and around the wetland on a scale of 0 (the stressor is not present) to 3 (the stressor affects more than 2/3 of the wetland). Metric 11, however, was on a scale of 0 to 4 (0 = not present at all, 1 is <5%, 2 is 5-25%, 3 is 26-75%, and 4 is >75%). Each metric was summed up and each wetland was given a total stress rating (the sum of all Metrics, where lower values represent less stress) as well as scores for the individual metrics.

Level III Assessment

Plant communities were sampled in each wetland once during either the summer of 2012 or 2013 following the protocols outlined in the United States Environmental Protection Agency's 2011 National Wetland Condition Assessment Program (NWCA, EPA 2011a;) as shown in Figure 2. Briefly, each wetland contained five 10m X 10m plots that were arranged within the wetland where data could be collected that reflected vegetation composition across the wetland. All plants were identified to species, when possible, in collaboration with a botanist. Aerial percent covers were estimated in the field and recorded for each plant species within each plot for each plant strata size (e.g., submergent vegetation, floating or floating-leaved plants, short emergent plants < 0.5m, tall emergent plants > 0.5m, short woody plants < 0.5m, tall woody plants > 0.5m, and vines) where an overall percent cover could be larger than 100% since there could be multiple strata present. The percent covers were averaged across the plots to give the relative cover of each species for each wetland.

Composite sub-surface water samples were collected from each of the major habitat types that were present. These habitats included open water, submergent vegetation, and emergent vegetation, if available. A total of 4 samples were collected, where the dominant habitat was sampled twice across each wetland during both May and July 2013 sampling events. Water samples were analyzed for chlorophyll *a* using the fluorometric determination method as described by Clesceri *et al.* (2005). Water samples were also analyzed for orthophosphate and total phosphorus using the Standard Methods 4500 P-E method described by Clesceri *et al.* (2005) with a Hach DR 5000 UV-Vis spectrometer. Water temperature, dissolved oxygen, and turbidity were recorded and averaged from the same locations taken for water samples within each wetland using a Horiba U-50 multiparameter water quality meter. Dissolved oxygen was excluded from analysis due to the probe malfunctioning during one of the sampling periods.

Composite macroinvertebrate samples were collected with four 1m sweeps of a D-frame dip net (500 μm mesh) from the same locations as the water samples that were collected within each wetland in both May and July 2013. D-frame dip nets have been shown to collect a greater number of species, including fast swimming species, compared to the vegetation quadrat, water column, and benthic core method (Meyer *et al.* 2013). Macroinvertebrate samples were preserved in 70% ethanol, sorted, and identified under a microscope to the lowest taxonomic group possible in the lab following Merrit *et al.* (2008). In most cases, taxa were identified to genus, with the exception of Chironomidae and Simulidae, which were identified to family.

Data Analysis

I used regression analyses to compare Level I (LDI, LUS, and the % of each land use type) metrics at each buffer size and Level II assessments (each metric score individually and the sum of all metric scores). I compared the p-values and r^2 values of the LDI and LUS regressions of each buffer distance to the stressor scores to see if the coefficients developed for the LUS (from oxbow and riparian wetlands in Oklahoma; Dvorett *et al.* 2013) were a better fit for the wetlands than the ones developed for the LDI (from isolated depressional wetlands in Florida; Brown and Vivas 2005).

Macroinvertebrate data were totaled from the two sampling periods for each wetland for statistical analyses. Plant data was averaged from all plots within each wetland to give the relative abundance of each species present. Plant and macroinvertebrate data were used to determine several community metrics that include: richness (total number of taxa), Shannon-Wiener diversity (calculated as $\mathbf{H' = -\Sigma\, \boldsymbol{p_i} \ln \boldsymbol{p_i}}$, where $\boldsymbol{p_i}$ is the proportion of species $\boldsymbol{i}$), and evenness (calculated as $\mathbf{E = H'/log(N)}$, where n is the total number of species present). In addition, the floristic quality index (FQI) and the mean coefficient of conservatism (CoC) of the plant species found in each wetland were calculated from those developed by Dr. Bruce Hoagland for Oklahoma plants (unpublished data). For macroinvertebrate communities, % EPT (Ephemeroptera, Plecoptera, and Trichoptera), % Chironomids, and % Corixids were calculated, because these are common macroinvertebrate metrics that are used to assess wetland condition and are used in numerous indices of biological integrity (Tangen *et al.* 2003, Campbell *et al.* 2009, Bird *et al.* 2013). Each macroinvertebrate genera and the relative abundance of each family were compared to the land use metrics (percent cover of each land use type, LDI, LUS, etc.) using regression analysis in Minitab Version 15. These community metrics were compared to the land use metrics described to determine if there were significant relationships between land use disturbances and the richness and diversity, where each of the within-wetland variables listed in Table 2 were regressed against the land use variables for each spatial scale buffering wetlands shown in Table 2.

Plant and macroinvertebrate community composition were compared to land use disturbance and within-wetland metrics (list of metrics in Table 2) using a Canonical Correlation Analysis (CCA) in CANOCO 5.03 software. A CCA was developed for each buffer distance (100m, 300m, 500m, and 1000m) for both plant and aquatic insect communities, first using only land use data to analyze the influence of spatial scale on the Level I assessment. Then, I used a CCA with both land use and within-wetland metrics for the buffer distance to determine if the importance of these all metrics changes with scale. For the plant communities, some of the within-wetland characteristics (phosphorus and chlorophyll *a*) were excluded from analysis since they were not collected at the same time as the plant communities. For this second analysis, I used forward selection CCA analysis to determine the top 5 variables that were most important in shaping the communities. For all CCA analyses, I ran a constricted CCA with 999 unrestricted permutations, with rare species (in <10% of sites) removed.

Results

<u>Wetland Condition</u>

The 22 wetlands exhibited a range of LDI scores from 1 to 4.86 and the LUS scores ranged from 1 to 0.40. A table showing the percent of each land use type and the LDI and LUS scores for the 100m buffer is inAppendix 5. There were no significant differences in LDI or LUS scores between the four buffer sizes (ANOVA, $P = 0.58$ and 0.51, respectively; Figure 3). LDI and LUS scores were highly related at all buffer distances ($P < 0.001$) and r^2 values increased as buffer distance increased from 0.59 for the 100m buffer to 0.79 for the 1000m buffer (data not shown). USA-RAM total stress rating ranged from 0 to 24, with an average total score in the wetlands of 9. Chlorophyll *a* levels increased from the first sampling period to the second sampling period (May = 17.6 ± 6.6, July = 81.0 ± 25.9, $P = 0.01$, Figure 4), while other metrics of water quality did not differ between sampling periods (all $P > 0.05$, data not shown).

One hundred twenty two different genera, encompassing 51 families and 9 orders of insects were found in the wetland samples (Appendix 6). Wetlands had between 17 and 45 total genera for both sample periods combined. The total abundance of aquatic insects per sample period ranged from 33 to 2277 and significantly greater abundances were observed in the second sample period (May = 163 ± 26.5 Standard Error, July = 655 ± 121, $P < 0.001$, Figure 5). The number of genera also increased from the first sampling period to the second sampling period (May = 17 ± 0.96, July = 24 ± 1.52, $P < 0.001$,

Figure 8). However, Shannon-Weiner diversity did not differ between sampling periods (May = 2.014 ± 0.09, July=1.858 ± 0.08, P = 0.11, data not shown). Evenness decreased from the first to the second sampling period (May=0.716 ± 0.026, July=0.588 ± 0.021, P = 0.002, data not shown). Wetlands had between <1% to 55% EPT and had between 0% to 49% Chironomids for both sampling periods combined.

One hundred seventy one species, encompassing 125 genera, 63 families, and 32 orders of plants were found in the wetland samples (Appendix 7). Wetlands had between 3 to 50 plant species present. A total of 11 non-native species of plants were found within the wetlands (Appendix 7). At most, 3 non-native species were present within an individual wetland. Plant Shannon-Weiner diversity values ranged from 0.19 to 2.60. Plant cover in the wetlands ranged from 6.6% to 98%. Average CoC values from the plants within the wetlands sampled ranged from 3.4 to 5.5, while the FQI scores ranged from 10.2 to 29.3.

Comparing Level I and Level II Assessments

LDI scores from the CropScape data were positively related to USA-RAM total wetland stressor scores for the 100 and 300 meter buffers (P < 0.001, r^2 = 0.47; Figure 6; P = 0.04, r^2 = 0.19; Figure 7, respectively); however, USA-RAM wetland stressor scores were not significantly related to LDI values at any of the larger buffers (all P > 0.05, data not shown). LDI scores from the 100m buffer were also positively related to the buffer metrics in USA-RAM (Metric 3; P = 0.003, r^2 = 0.37, data not shown) The 100m LUS was also significantly negatively related to the total USA-RAM wetland stressor scores, but did not explain as much variation as the LDI (P = 0.04, r^2 = 0.20; Figure 9); however, USA-RAM wetland stressor scores were not significantly related to the LUS scores for any of the larger buffers (all *P* > 0.05, data not shown).

Comparing Level I and Level III Assessments

When comparing community metrics to environmental and spatial variables with regression analyses, no landscape metrics were significantly related to the richness or diversity of aquatic insects (P > 0.05, data not shown). Neither LDI, LUS, nor any of the other disturbance metrics calculated at any of the buffers for the Level I assessment were significantly related to any of the community metrics of wetland aquatic insects (*P* > 0.05, data not shown). No landscape assessment at any buffer distance was related to the % EPT, % Chironomids, or % Corixids (*P* > 0.05, data not shown). Only the Level III assessment of water quality showed that turbidity was significantly negatively related to aquatic insect richness (*P* = 0.04, r^2 = 0.23, Figure 10).

When relating plant community metrics to environmental and spatial variables, the only land use variable that was significantly related to plant richness was the amount of water within 300m of the wetland, which was a negative relationship (*P* = 0.004, r^2 = 0.35, Figure 11). Neither LDI, LUS, nor any of the other disturbance metrics calculated at any buffers around the wetlands for the Level I assessment were significantly related to any of the community metrics of wetland plants (*P* > 0.05, data not shown). No landscape assessment at any buffer distance was related to the mean CoC value or the FQI score either (*P* > 0.05, data not shown).

Influences of Spatial Scale

When performing constrained CCA with only land use categories from the 2012 CropScape dataset, there was little differences in how much variation was explained by land use categories assessed at the different buffer distances for both plant and aquatic insect communities (Table 3). However, for plants, the most percent variation explained by the different land uses within the different buffer distances was the 100m buffer (48.9%). For the aquatic insect genera analysis, the 300m buffer explained the most variation among the communities (45.4%). For the aquatic insect family and order analysis, the 1000m buffers explained the most variation among the communities (57.1% and 49.1%, respectively).

Effects of Land Use and Within-Wetland Characteristics

When performing the forward selection CCA for aquatic insect genera, land use data from the 300m buffer was used since it explained the most variation between the communities. The top five land use and within-wetland characteristics that were selected and explained the most variation in the aquatic insect genera data were the relative percentage of mixed forest, chlorophyll *a* levels in July, the relative percentage of deciduous forest, chlorophyll *a* levels in May, and turbidity levels (Figure 12). The CCA had Eigenvalues of 0.2593 and 0.2174 for Axis 1 and 2, respectively. The variables above accounted for 38.1% of the variation in the communities. Of the explained variation, 31.5% was explained by within-wetland variables, 48.2% was explained by land use variables, and 20.3% was explained by combined effects (Figure 15).

When comparing the CCA analysis at different buffer distances for the aquatic insect genera, the CCA at the 300m buffer explained the most variation between communities (38.1%). Of the top 5 variables chosen during each analysis, the relative proportion of deciduous forest was an important variable in every analysis. Chlorophyll *a* levels in May were important in 3 of the 4 buffer distances (100m, 300m, and 1000m). Turbidity was important in the smaller buffers (100m and 300m), while the relative percent cover of grassland/herbaceous and pasture/hay were important in the larger buffers (500m and 1000m). Interestingly, the relative percent cropland was only important in the 100m buffer analysis. Other variables that were selected included phosphorus levels (100m), the relative percent mixed forest (300m and 1000m), chlorophyll *a* levels in July (300m), and the relative percent barren land (500m).

When performing the forward selection CCA for the aquatic insect family level analyses, land use from the 1000m buffer was used since it explained the most variation between communities. The top five land use and within-wetland characteristics that were selected and explained the most variation in the aquatic insect family data were for the 1000m buffer, which explained the most variation in the communities, were: the relative percentage of mixed forest, the relative percentage of developed land, average phosphorus levels, the relative percentage of barren land, and the relative percentage of pasture/hay (Figure 13). The CCA had Eigenvalues of 0.2773 and 0.2045 for Axis 1 and 2, respectively. The variables above accounted for 45.5% of the variation among the different communities. Of the explained variation, 3.3% was explained by within-wetland variables only, 44.0% was explained by land use variables only, and 52.7% was explained by combined effects (Figure 15).

When performing the forward selection CCA for the order level analysis, land use data from the 1000m buffer was used since it explained the most variation between communities. The top five land use and within-wetland characteristics that were selected and explained the most variation in the aquatic insect order data were for the 1000m buffer, which explained the most variation in the communities, were: chlorophyll *a* levels in July, the relative percentage of mixed forest, the relative percentage of pasture/hay, the relative percentage of grassland/herbaceous, the relative percentage of deciduous forest, and the average turbidity levels (Figure 14). The CCA had Eigenvalues of 0.1828 and 0.1144 for Axis 1 and 2, respectively. The variables above accounted for 48.7% of the variation among the different communities. Of the explained variation, 9.3% was explained by within-wetland variables only, 10.7% was explained by land use variables only, and 80.0% was explained by combined effects (Figure 15).

Discussion

A major objective of my research was to document and better understand the relationships between the three EPA wetland assessment levels in Oklahoma wetlands, and more specifically to determine the relationships between the USA-RAM and disturbance and wetland condition. I found that there were several significant relationships between the Level I and II assessments; however, there was only one significant relationship between Level I and III assessments, which was the relative amount of water within 300m of the wetland, and no significant relationships between the Level II and III assessments. The USA-RAM at the 100m buffer was relatively good at capturing land use disturbances (Level I assessments), which is in accordance with other studies that directly compared Level I and II wetland assessments (Reiss and Brown 2007, Stein *et al.* 2009, Margriter *et al.*2014). The lack of relationships at larger buffers is most likely due to the fact that the USA-RAM only measured stressors in distances 100 meters around the individual wetlands. As the buffer distance increased, the resolution of disturbances directly surrounding the wetland decreases. As such, it has been suggested that larger buffers do not accurately assess disturbances that are proximal to wetlands (Lammert and Allan 1999, Rooney *et al.* 2012). Similarly, Brown and Vivas (2005) concluded that a 100 meter buffer was adequate at capturing land use disturbance to wetlands when they developed the LDI. The relationships between USA-RAM Stressor scores and GIS land use data could be improved with increased resolution of the dataset (smaller pixel size) and additional land use categories as some of the stressors of the Level II assessment are not visible in the 30m^2 pixels of the CropScape dataset (e.g., inlets and outlets, herbicide application, etc.). Models could also be improved by calibrating for specific, individual ecoregions because communities can vary between different ecoregions (Nichols 1999, Sandin and Johnson 2000).

Surprisingly, the LUS (Dvorett *et al.* 2013) did not explain as much of the variation in USA-RAM scores as the LDI (Brown and Vivas 2005), which was created for isolated depressional wetland in Florida. This may be due to the fact that the LUS was created for oxbow and riparian wetlands, and that the current study included depressional wetlands in addition to oxbow and riparian wetlands. Therefore, it may be more beneficial to use the LDI and it may extend well to other areas of the country and for many wetland types and does not need to be calibrated specifically for each region or wetland type. For example, my findings are consistent with other studies showing that the

LDI is an important tool that can be widely used and that it works well at assessing wetland disturbances in many regions of the country including Florida (Cohen *et al.* 2004, Brown and Vivas 2005), Ohio (Mack 2006), Hawaii (Margriter *et al.* 2014), and even Taiwan (Chen and Lin 2011).

Other studies have reported contrasting relationships between land use (Level I) and wetland communities (Level III; Tables 1-3). I did not find strong relationships between any of the land use measurements and wetland community metrics. One factor to consider is that my analysis included several types of wetlands (e.g., oxbow and depressional), which may have affected my ability to develop significant models. For example, different wetland types can contain very different communities, which may have added variation to my data (Batzer and Wissinger 1996). Also, wetlands may have been in different stages along a wet-dry gradient, which can affect community composition (Casanova and Brock 2000). Another factor that may have contributed to variation in the data is that the wetlands were from seven different ecoregions in Oklahoma (Figure 1) and there can be strong ecoregional effects on community composition (Nichols 1999). All of these factors suggest that a larger dataset may be necessary to observe relationships and capture more of the potential variation that exists in wetland communities across large spatial and temporal scales.

The goal of any rapid Level II assessment is to evaluate the condition of a wetland using metrics that are easily and quickly measurable. The EPA developed the USA-RAM in 2011. However, it was not calibrated for wetlands within specific regions, including Oklahoma, but instead was developed to assess wetlands throughout the nation (Scozzafava *et al.*2011). There were strong relationships between the USA-RAM and land use disturbances in Oklahoma wetlands, but the USA-RAM was not related to any metric of wetland condition including FQI, % EPT, % Chironomids, species richness, or diversity. These results suggest that the USA-RAM needs further calibration within this region and potential modification to better represent wetland condition.

While landscape variables were not related to wetland communities, water quality variables were significant predictors of the diversity and richness of both aquatic insects and plants. Turbidity was a significant predictor of aquatic insect richness, where the number of genera decreased as turbidity increased. Turbidity directly affects species by clogging gills (Swenson and Matson 1976) and can indirectly affect species by warming the water, thereby decreasing the amount of dissolved oxygen available for invertebrates. Also, when there is clearer water, predators can be more efficient and abundant (Gardner 1981, Barrett *et al.* 1992), which may free up resources to support a higher diversity of lower consumer level taxa to co-exist (Menge and Sutherland 1979).

Land use models (Level I) have the potential to be fairly good at predicting stressors (USA-RAM) to the wetland. However, caution should be used when assessing disturbances at large buffer scales because it is possible that disturbances in a large buffer do not directly stress the wetland (Levin 1992, Lammert and Allan 1999). When creating models to predict stressors to wetlands, it may be most beneficial to look at the catchment area of the wetland (Silva and Williams 2001). Unfortunately, this is not always easy to do, especially in areas with low-relief topography. Furthermore, the location and placement of a stressor within a buffer could be important (Lammert and Allan 1999). For example, even if there is a stressor within a buffer, the wetland may be directly surrounded by a different land use type that protects the wetland from direct impacts of that land use disturbance. A better understanding of what factors are actually important in shaping wetland plant and animal communities is needed, since the Level III assessment was not strongly related to either the Level I or II assessment.

Another objective of this research was to determine how the spatial scale of the Level I assessment affected the relationships with aquatic insect and plant communities (Level III assessments). There was little difference in the amount of variation explained by land use variables measured at the different buffers. However, the 100m buffer explained the most variation in plant communities; while for aquatic insect genera, the 300m buffer explained the most variation between communities. When I analyzed a coarser scale of taxonomy for aquatic insects (family and order levels), the largest (1000m) buffers explained the most variation. Combined, these results show that aquatic insects and plants were affected by both disturbances measured in small buffers and by disturbances in larger buffers. These results suggest that the genera structure may be affected by local land use, but that the overall structure of the communities may be affected by regional land uses. This may be similar to other studies looking at taxonomic resolution of macroinvertebrates that found that family level analysis can detect coarse impacts of land use, but that genus level analysis can detect more subtle differences between communities (Waite *et al.* 2004, Chessman *et al.* 2007). This indicates that analyzing the correct spatial scale is important to understanding the relationship between communities and disturbances and that multiple scales may be necessary.

The third objective of this study was to evaluate whether within-wetland or land use characteristics were more important in structuring wetland communities. When looking at the amount of variation explained by either land use or the within-wetland characteristics (Table 3), aquatic insect genera analysis shows that both land use and within-wetland characteristics are relatively equal in explaining the variation among the communities. However, the within-wetland characteristics explained less for the family and order level analyses. When comparing land use and within-wetland variables combined in the forward selection CCA, land use variables explained more of the variation among the communities than within-wetland variables for both genera and family level analyses, while they were relatively equal for the order level analysis. The latter results suggest that land use may be more important in structuring what species are present, while the within-wetland metrics may be most important in shaping the abundance of each order. Studies looking at local and regional influences on wetland communities have had mixed results. Some have shown that regional influences are better at explaining variation in communities (Margriter *et al.* 2014), while others have shown that local influences are better at explaining variation (Whited *et al.* 2000, Meyer 2012, Johnston *et al.* 2013). Similar to the conclusions of Hall *et al.* (2004), my results suggest that both local (within-wetland characteristics) and regional influences (land use characteristics) are important in shaping the communities present within wetlands.

When comparing the variables selected for the CCA for the aquatic insect communities at different buffer scales the relative percent deciduous forest was consistently one of the top variables selected in the CCA. This result suggests that forests are important to aquatic insects in wetlands. An increase in forest cover may indirectly affect insects by providing cooler water temperatures through shading, by filtering runoff water going to wetlands, and by providing leaf litter for shredders (Cummins *et al.* 1989, Plenzler 2012). Another variable that was common among many different CCA's was the chlorophyll *a* concentration from the May sampling event. Since algae and plants are the base of the food chain within wetlands, it could be expected that they may be a limiting resource for aquatic insects. When it is the limiting resource, algae have been shown to be positively related to biomass and abundance of aquatic insects (Braccia *et al.* 2014). My results suggested that when there was a higher concentration of chlorophyll *a*, there was a greater abundance of aquatic insects (Figures 5 and 6). One more variable that was common among CCA's for aquatic insect communities was turbidity. This suggests that turbidity both limits how many and what species are present within wetlands.

The amount of variation explained increased from the genera to the order level analysis. However, there was a high proportion of unexplained variation between the communities. Differences in viable eggs and estivating larvae available may have made the variables measured unable to explain much of the data since some may persist in environments of unfavorable condition (Leibold 1995). Also, water chemistry metrics that were not measured in the current study such as dissolved oxygen and dissolved organic carbon may be better at explaining more of the variation between communities since macroinvertebrates are known to respond to these (Merrit *et al.* 2008, Plenzler 2012). Finally, the high amount of unexplained variation could come from differences in communities within each ecoregion and wetland type studied as mentioned above (Sandin and Johnson 2000).

Conclusion

My results suggest that land use disturbances (i.e., Level I) can be used to accurately predict USA-RAM (i.e., Level II assessment) score, but neither Level I or II accurately predicted wetland community condition (i.e., Level III assessment). This suggests that the USA-RAM needs to be calibrated to Oklahoma wetlands (and likely other regions of the country) before it is used to assess wetland condition. Therefore, it may be in the best interest of monitoring agencies to complete a Level I analysis to predict the stressors present to wetlands and then conduct more labor intensive Level III analysis of the communities to get a more accurate account of the wetland condition. Coarse, quick land use models (e.g., LDI) may be good to use to look at the stressors to the wetlands and may be widely adapted, but may not be good at predicting the communities within wetlands. Therefore, caution should be taken before any models are used, and should at least be first tested in a subset of sites in a study area.

Land use models have the potential to predict the stressors present to the wetland. However, these may not be particularly useful if they cannot predict community responses and wetland quality. More research is necessary in order to determine what stressors are actually important in structuring communities and the relationships between stressors and community responses. The proximity of a wetland to a stressor may be better and the land use separating the two may be more important than the stressor itself. Research looking at how stressors affect wetlands in different ecoregions is also necessary, since each ecoregional response may be different.

My results also suggest that spatial scales closer to wetlands (e.g., smaller buffer widths) are better predictors for genera analysis than larger buffer sizes when analyzing the landscape, which may be better for family and order level analyses. Similar to the suggestions of Brown and Vivas (2005), a 100 meter buffer around wetlands was identified as the best predictor of Level II stressors in the wetland. However, it may not be best at analyzing community responses. A catchment level analysis may be better, since a single size buffer may not adequately fit all wetlands.

Finally, land use variables may be more important than within-wetland variables for structuring aquatic insect genera. However, the within-wetland variables become more important as the analysis moves toward family and order-level analyses. This suggests that even relatively less mobile and small taxa (as compared to waterfowl) may be affected by land use disturbances.

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Tables

Table 1 Land use categories and their corresponding coefficients for calculating LDI and LUS. Coefficients were taken from Brown and Vivas (2005) and Dvorett *et al.* (2013) and site scores were determined using the 2012 CropScape dataset.

Land Use Classification	LDI Coefficient	LUS Coefficient
Open water, Emergent herbaceous wetlands, Woody wetlands	1.00	1.0
Deciduous forest, Mixed forest, Scrub/shrub	1.00	1.0
Evergreen Forest	1.58	1.0
Barren	1.00	0.5
Grassland/herbaceous	2.77	1.0
Pasture/hay	3.74	0.7
Cultivated Crops	4.54	0.3
Developed, Open Space	6.92	0.7
Developed, Low Intensity	7.47	0.2
Developed, Medium Intensity	8.66	0.0
Developed, High Intensity	10.00	0.0

Table 2 List of metrics and metric category used in regression analysis and Canonical Correlation Analysis to assess the relationships between Level I, II, and III assessments.

Metric	Metric Category
LDI and LUS	Land Use
Area	Within-wetland
Nutrients	Within-wetland
Distance to nearest water body	Within-wetland
Percent cropland/pasture	Land Use
Percent development	Land Use
Percent forested	Land Use
Percent water	Land Use
Chlorophyll *a*	Within-wetland
Level II Assessment (USA-RAM) Scores	Within-wetland

Table 3 Percent of the variation explained by land use calculations from 2012 CropScape data at different buffer distances, as well as the within-wetland metrics and the USA-RAM data in the Canonical Correlation Analyses.

	Plant	Insect Genera	Insect Family	Insect Order
100m	48.9	42.8	48.8	45.7
300m	45.9	45.4	50.9	48.2
500m	47.9	44.7	56.5	46.3
1000m	48.8	43.9	57.1	49.1
Within-wetland	16.2	45.4	43.4	39.2
USA-RAM	18.4	20.4	17.9	11.4

Figures

Figure 1 Wetland sample locations (n=22) and ecoregions in Oklahoma. Points represent wetlands sampled and polygons represent ecoregions within the state.

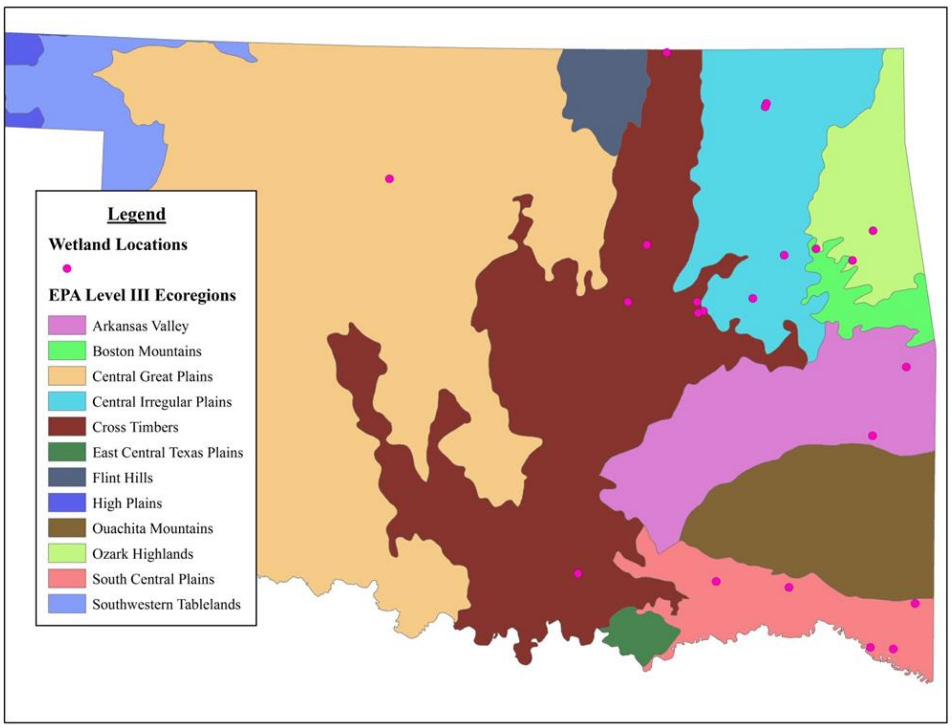


Figure 2 The standard setup used for sampling wetland vegetation. Dashed lines represent transects in the cardinal directions while the labeled squares represent the 10m x 10m vegetation sampling plots. This sampling technique was modified from the U.S. Environmental Protection Agency's National Wetland Condition Assessment: Field Operations Manual (2011a).

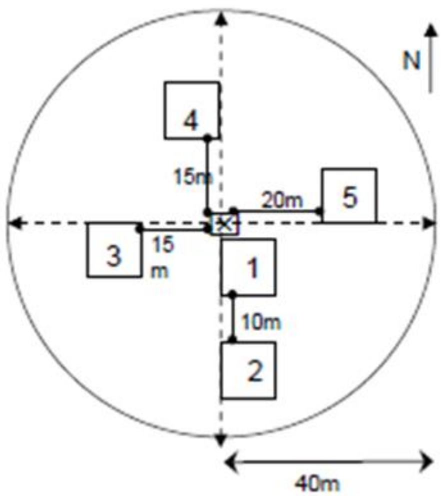


Figure 3 Average LDI and LUS scores from 2012 CropScape data for 22 wetlands sampled in Oklahoma at four different buffer distances (100m, 300m, 500m, and 1000m). LDI and LUS values did not differ between the different buffers (ANOVA, P = 0.58 and 0.51, respectively).

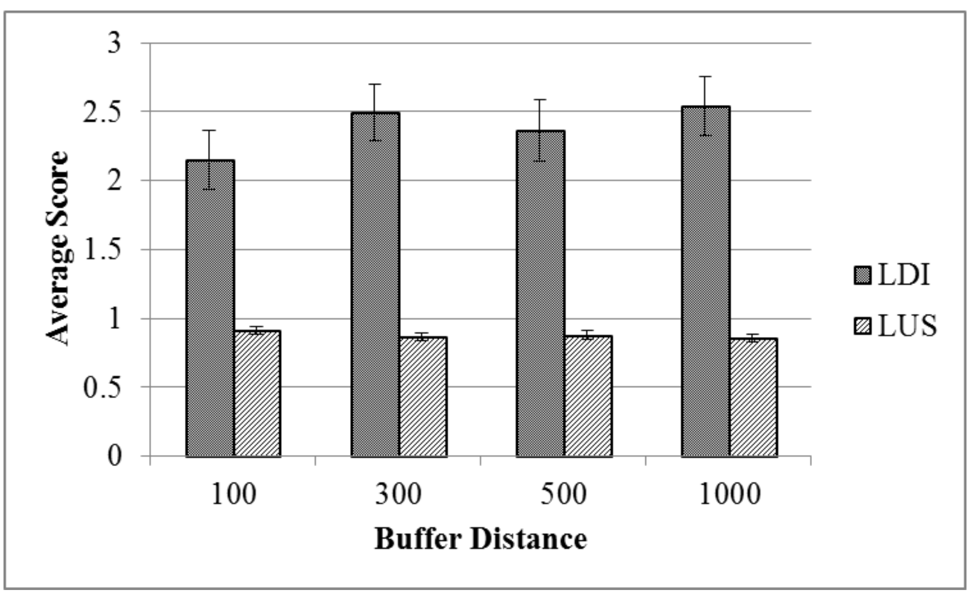


Figure 4 The average chlorophyll *a* concentrations in the 22 wetland samples in Oklahoma for May and July 2013 (Paired t-test, $P = 0.01$).

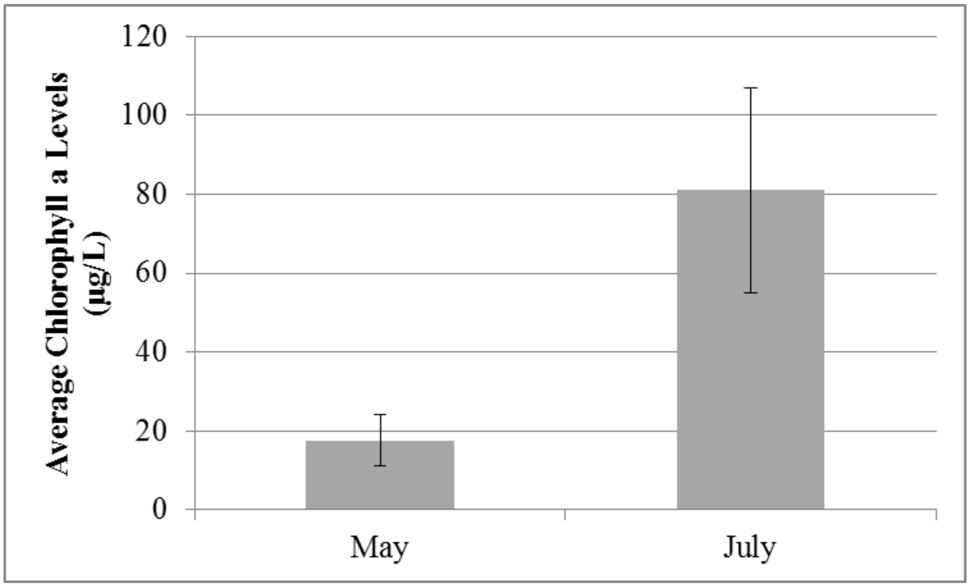


Figure 5 The average number of aquatic insects in the 22 wetland samples in Oklahoma for May and July 2013 (Paired t-test, $P < 0.001$).

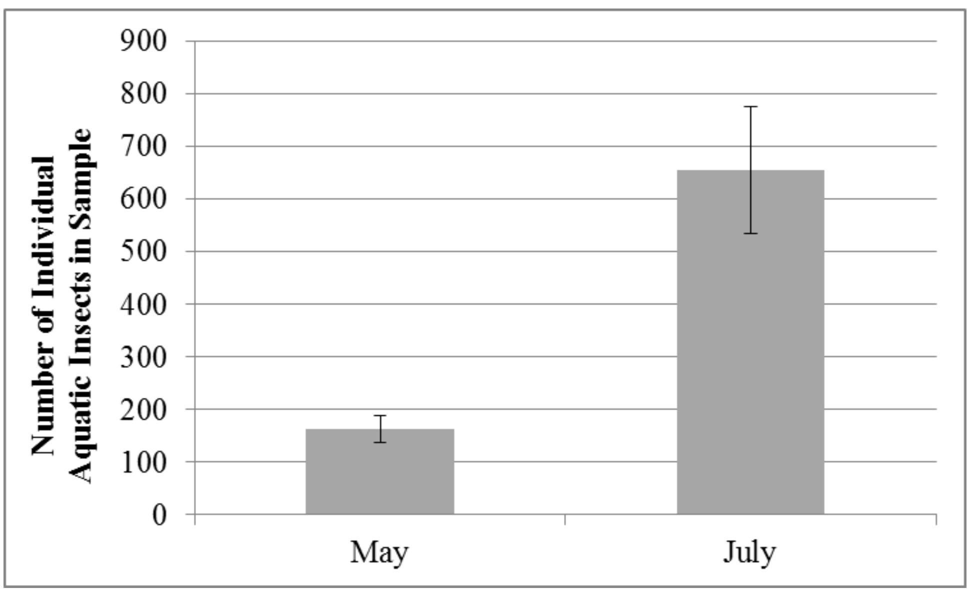


Figure 6 The relationship between 100m LDI values and USA-RAM stressor scores. Higher LDI and USA-RAM stressor scores represent wetlands with more disturbances to the surrounding landscape (Regression, P = 0.002, r^2 = 0.40).

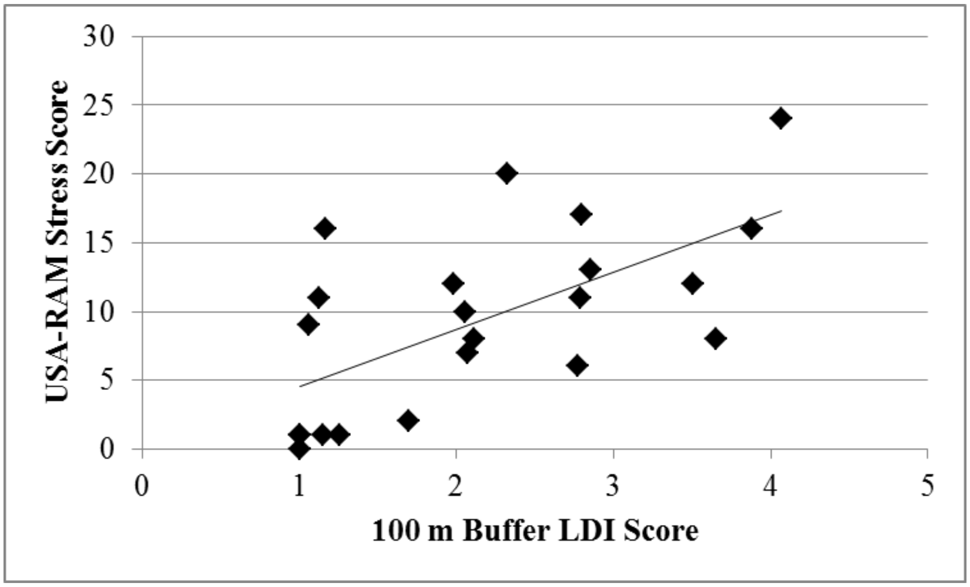


Figure 7 The relationship between 300m LDI scores and USA-RAM stressor scores. Higher LDI scores represent wetlands with more land use disturbances (Regression, P = 0.04, r^2 = 0.19).

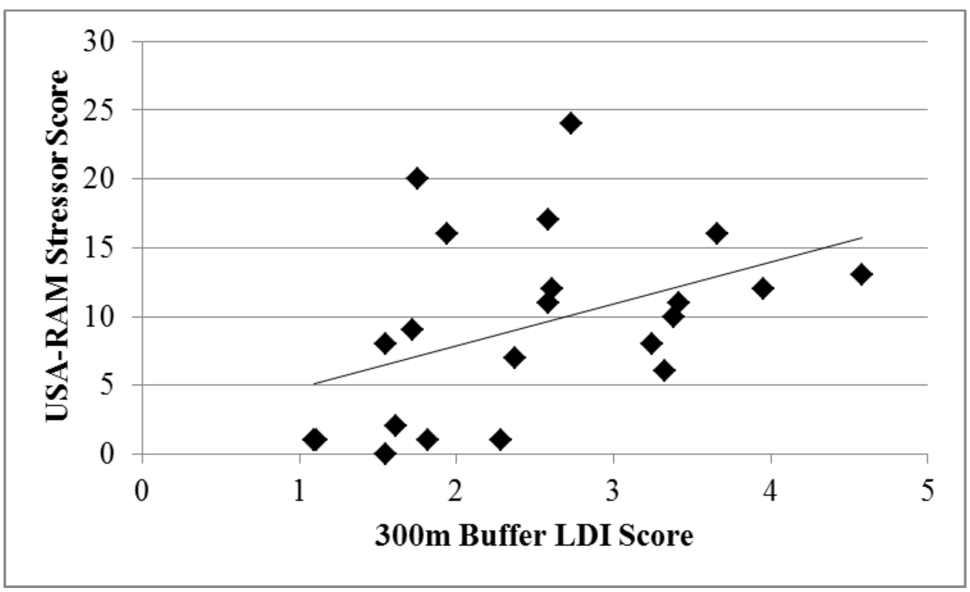


Figure 8 The relationship between 100m LDI values and the USA-RAM Metric 3 score (Stressors to the Buffer). Higher LDI and USA-RAM Metric 3 values represent wetlands with more land use disturbance (Regression, P = 0.003, r^2 = 0.37).

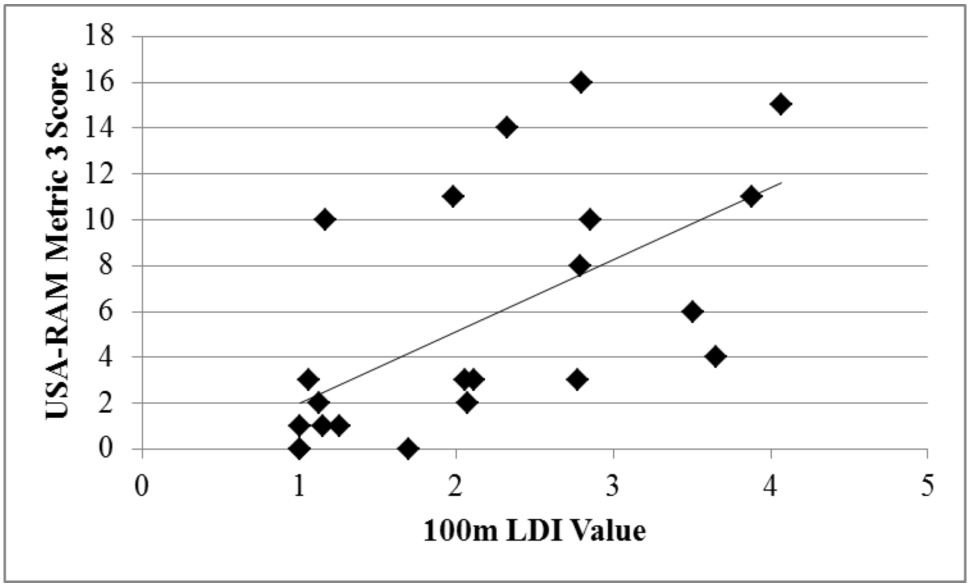


Figure 9 The relationship between 100m LUS score the USA-RAM stressor scores. Lower LUS values represent wetlands with more disturbances to the surrounding landscape, while higher USA-RAM stressor scores represent wetlands with more disturbance (Regression, P = 0.038, r^2 = 0.1976).

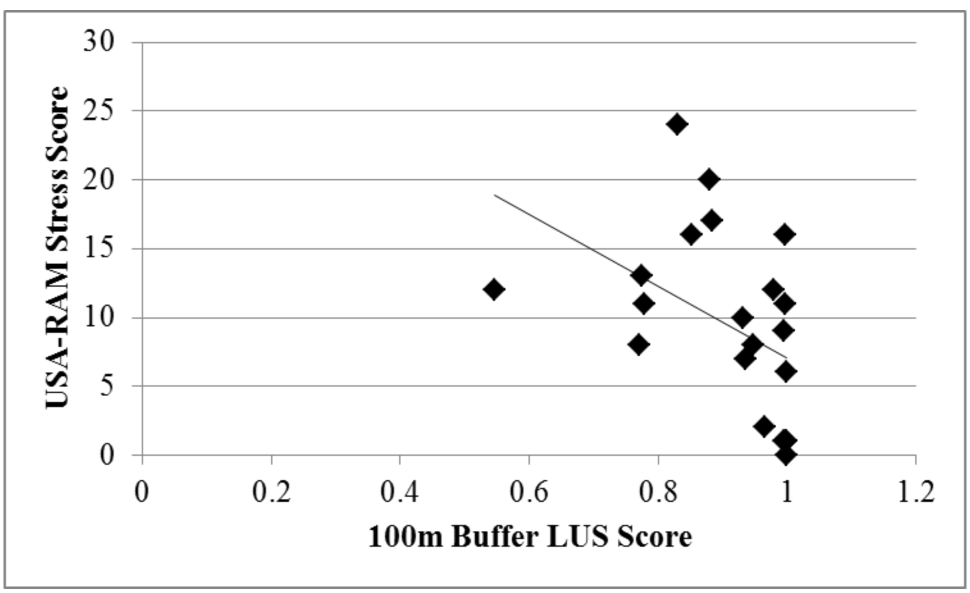


Figure 10 The relationship between turbidity and the number of aquatic insect genera found for both sampling periods combined in 22 Oklahoma wetlands (Regression, P = 0.04, $r^2 = 0.23$).

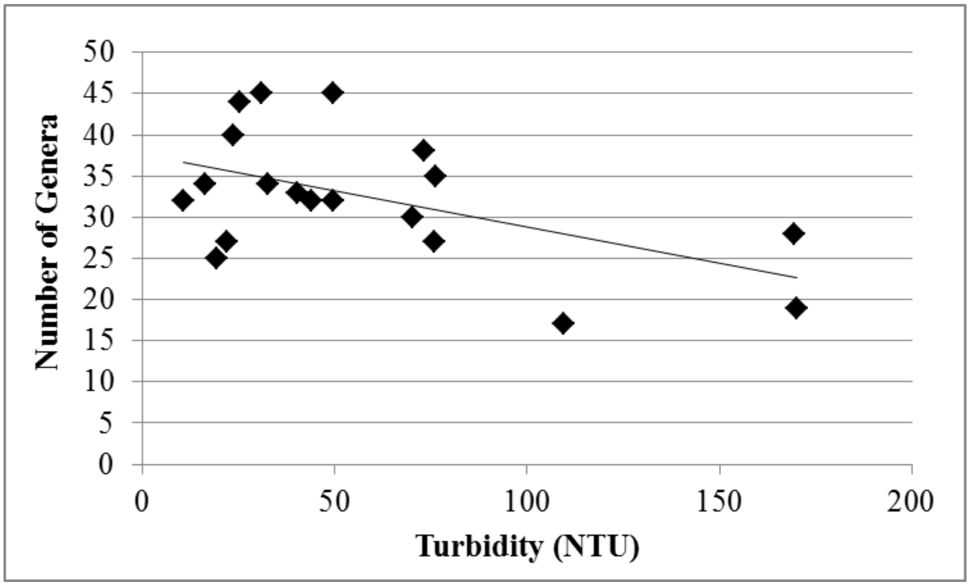


Figure 11 The relationship between the proportion of water within 300m of the wetland and plant species richness within the 22 wetlands sampled in Oklahoma (Regression, P = 0.004, $r^2 = 0.35$).

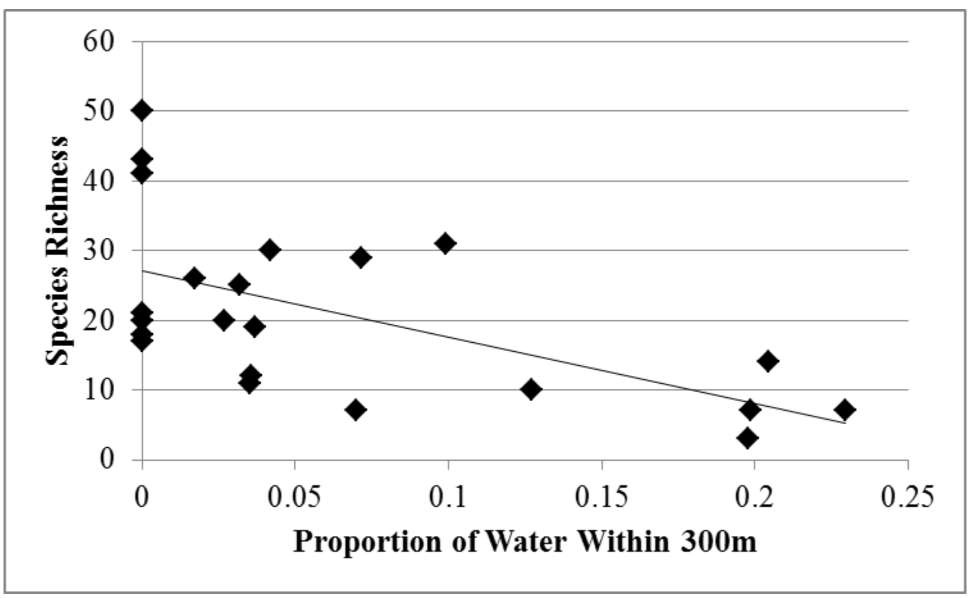


Figure 12 CCA of wetland aquatic insect genera at the 300m buffer distance with the top five contributing land use and within-wetland variables separating genera and communities. Vectors (arrows) point in the direction of increasing values for the respective variables, with longer vectors indicating stronger correlations between vectors and axes shown. Invertebrate genera can be found in Appendix 1. Turbidity = average turbidity levels between May and July, CHLAMay = Chlorophyll *a* concentrations during the May 2013 sampling event, Deciduous = the relative percentage of deciduous forest within 300m of the wetland, CHLAJULY = chlorophyll *a* concentrations during the July 2013 sampling event, Mixed Forest = the relative percentage of mixed forest within 300m of the wetland.

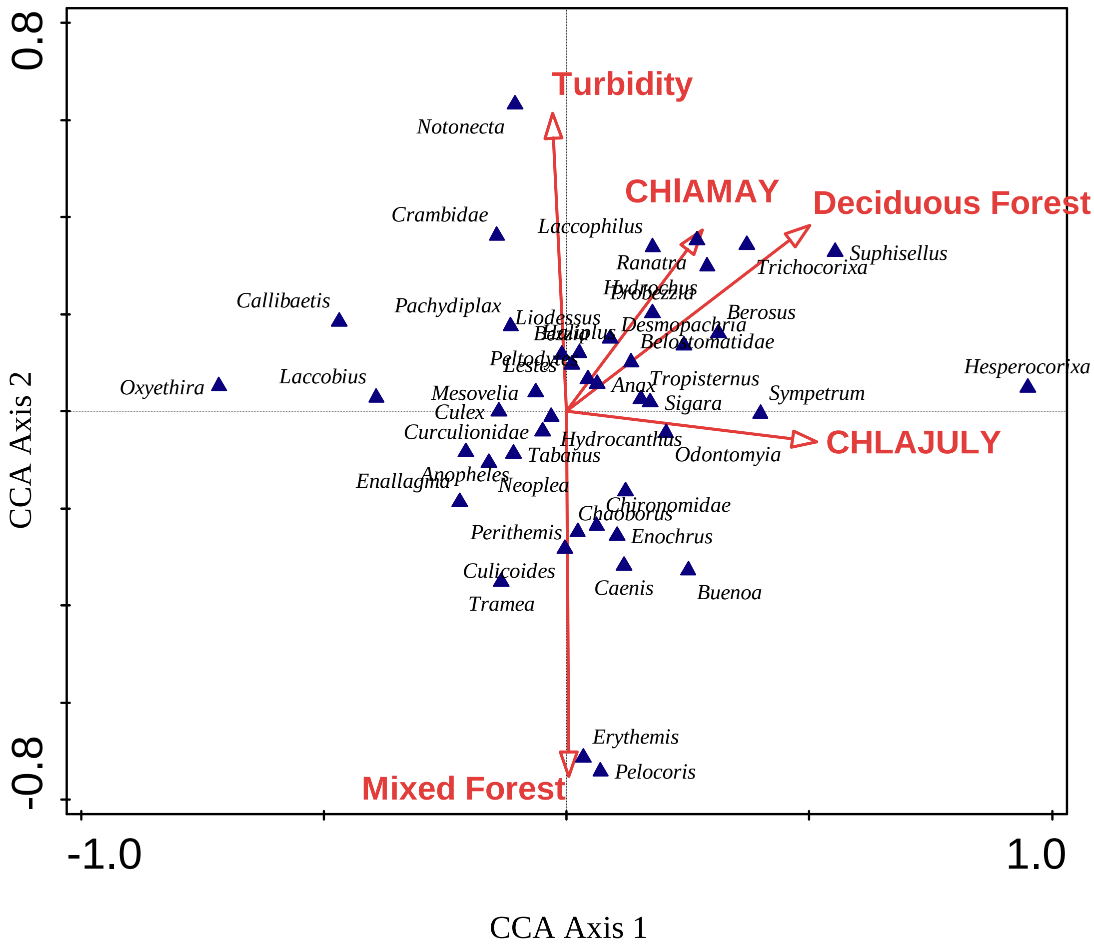


Figure 13 CCA of wetland aquatic insect family at the 1000m buffer distance with the top five contributing land use and within-wetland variables separating families and communities. Vectors (arrows) point in the direction of increasing values for the respective variables, with longer vectors indicating stronger correlations between vectors and axes shown. Invertebrate families can be found in Appendix 1. Mixed Forest = the relative percentage of mixed forest within 1000m of the wetland, Pasture/Hay = the relative percentage of pasture and hayed land within 1000m of the wetland, Developed = the relative percentage of developed land within 1000m of the wetland, p = the average phosphorus levels between May and July 2013, and Barren = the relative percentage of barren land within 1000m of the wetland.

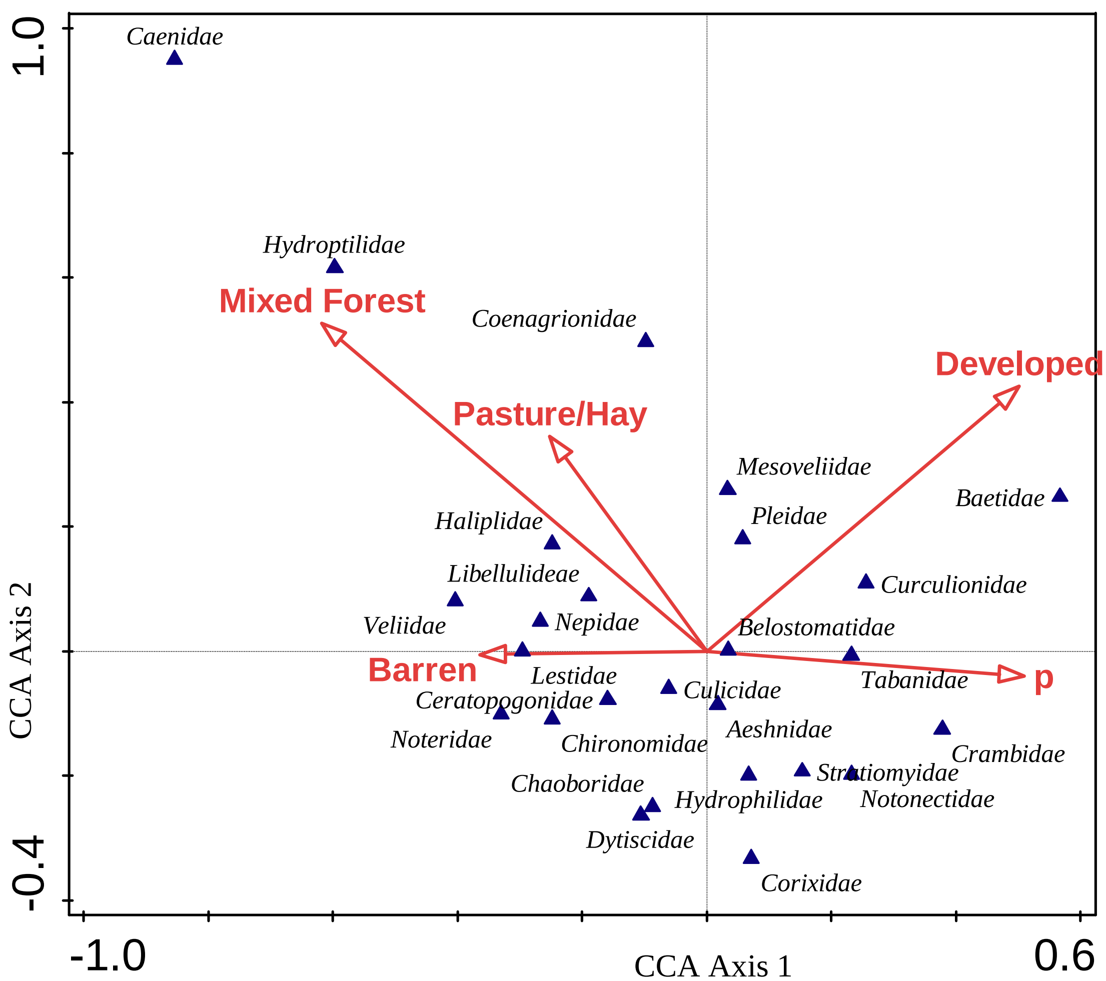


Figure 14 CCA of wetland aquatic insect orders at the 1000m buffer distance with the top five contributing land use and within-wetland variables separating orders and communities. Vectors (arrows) point in the direction of increasing values for the respective variables, with longer vectors indicating stronger correlations between vectors and axes shown. Invertebrate orders can be found in Appendix 1. Grassland/Herbaceous = the relative percent of grassland and herbaceous land within 1000m of the wetland, CHLAJULY = chlorophyll *a* concentrations during the July 2013 sampling event, Deciduous Forest= the relative percentage of deciduous forest within 1000m of the wetland, Turbidity = average turbidity levels between May and July, and Pasture/Hay = the relative percentage of pasture and hayed land within 1000m of the wetland.

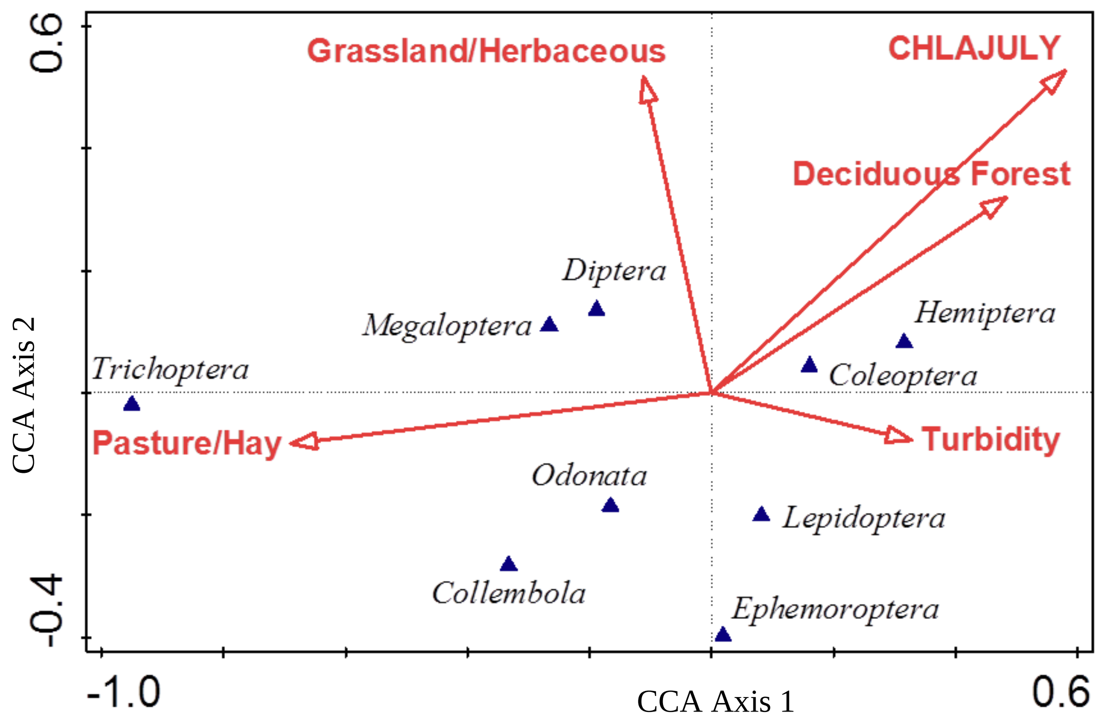
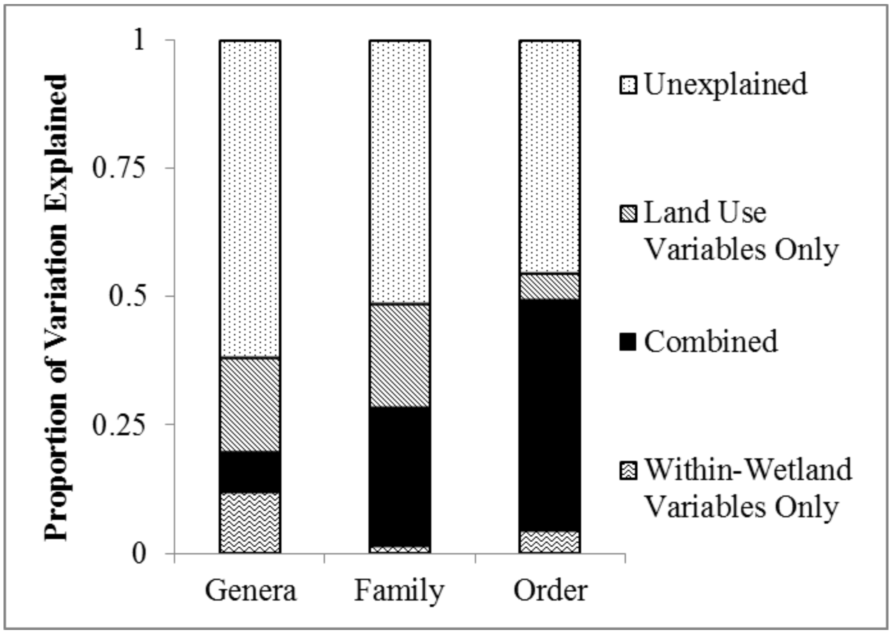


Figure 15 Comparison of the amount of variation explained by within-wetland and land use variables for the aquatic insect communities comparing a genera-level, family-level, and order-level analysis in CCA from wetlands sampled in Oklahoma.



APPENDICES

Appendix 1. Studies showing the relationship between many different taxa (e.g., diatoms, fish, amphibians, reptiles, and birds) and disturbance.

Study	System	Sample Size	Dependent Variable	Independent Variable	Results
Barret and Guyer 2008	Streams in western Georgia	16 samples of 12 second or third order streams	Herpetofaunal species richness	Primary land use within watershed (reference, pasture, urban, developing)	Amphibian richness was lowest in urban watersheds, Differences in species composition among different land use practices within watersheds.
Brazner *et al.* 2007	Coastal wetlands in the Great Lakes	450 locations along the Great Lakes shoreline	Different taxa (fish, birds, diatoms, and amphibians)	Human disturbance index	Taxon indicators were better than functional indicators. Wetland fish and bird indicators were the most responsive to human disturbance.
Chipps *et al.* 2006	Wetlands in Upper Missouri River basin in North Dakota	10 wetlands	Wetland water quality and algae	Low or High agriculturally impacted wetlands (Low is <5% of area within 150 m, High is >33% of area within 10m)	Higher impacted wetlands had higher phosphorus and alkalinity. Higher impacted wetlands had less sensitive diatoms.
Lammert and Allan 1999	Streams in southeastern Michigan	Six 100m stream reaches	Fish assemblages	Land use at different scales (within 50m, 125m, and entire subcatchment of stream section), instream habitat characteristics	Fish showed a stronger relationship to flow variability and immediate land use.
Lane and Brown 2007	Isolated wetlands in Florida	70 small wetlands	Diatom assemblages	Landscape Development Intensity Index (LDI) within 100m	They identified indicator species to create a Diatom Index of Wetland Condition.

Appendix 1. Continued

Study	System	Sample Size	Dependent Variable	Independent Variable	Results
Mensing *et al.* 1998	Riparian wetlands in Minnesota	15 riparian wetlands associated with low-order streams	Abundance, species richness, and Shannon diversity for birds, fish, and amphibians	Land use within catchment at 4 scales (500m, 1000m, 2500m, 5000m)	Bird and fish richness and diversity had a negative relationship with agriculture. Amphibians and birds responded more to local scales (500m and 100m) of human disturbance, while fish responded to more to the more regional scales (2500m and 5000m) of human disturbance
Petersen and Westmark 2013	Suburban wetlands in Minnesota	6 wetlands	Bird use of wetlands	Land cover within 500m	Bird use of wetlands was negatively associated with more urban cover of the landscape.
Rooney *et al.* 2012	Wetlands in Alberta's Beaverhills watershed	45 wetlands	Bird-based index of biotic integrity (IBI)	Land use within different buffering areas (100m, 300m, 500m, 1000m, 1500m, 2000m, 3000m.	IBI scores were significantly predicted by every spatial scale. Bird-based IBI scores were best predicted from land use within 500m. Road cover and proportion of disturbed land were consistent with the predictors of the IBI scores.
Whited *et al.* 2000	Wetlands in Minnesota	40 wetlands	Wetland bird assemblages	Landscape variables for 3 spatial scales (500m, 1000m, and 2500m)	Roads had the highest impact on bird assemblages at the 500m scale. Species richness was lowest in the urbanizing ecoregion, but the community patterns were not correlated to any landscape variables.

Appendix 2. Studies showing relationships between wetland vegetation and disturbance.

Study	System	Sample Size	Dependent Variable	Independent Variable	Results
Brazner *et al.* 2007	Coastal wetlands in the Great Lakes	90 locations along the Great Lakes shoreline	Vegetation (species richness, proportion of native taxa, proportion of invasive taxa, and proportion of obligate wetland taxa)	Human disturbance index (HDI)	Vegetation was strongly responsive to human disturbance.
Chipps *et al.* 2006	Wetlands in Upper Missouri River basin in North Dakota	10 wetlands	Vegetation (species richness, proportion of invasive taxa)	Low or High agriculturally impacted wetlands (Low is <5% of area within 150 m, High is >33% of area within 10 m)	Higher impacted wetlands had higher proportion of invasive plant species, while having lower overall plant richness.
Chu and Molano-Flores 2013	Wetlands in Northeastern Illinois	14 wetlands	Floristic Quality Assessment scores	Impervious surfaces, development, buffer area, wetland size	Positive relationship between wetland size and FQA score, positive relationship between impervious surfaces and percent native species, an increase in species richness post development, also larger wetlands had a higher percentage of native species.
De Cauwer and Reheul 2009	Wet meadows in Belgium	99 parcels of wet meadows	Plant species	Grassland management technique (pastures used at high or low stocking rate, hayfields used at high or low mowing frequency, abandoned hayfields and hay pastures)	Species richness was negatively related to intensity of grassland management.
DeKeyser *et al.* 2009	Prairie pothole wetlands in Montana, North Dakota, and South Dakota	193 wetlands	Plant communities	Disturbances and land uses that were represented included: rangeland grazing (light, moderate, heavy), pasture grazing (light, moderate, heavy), hayland, Conservation Reserve Program (CRP) grasslands, cultivation, urbanization, restored native prairie, idle lands (native and pasture), fire, drought, and pluvial conditions.	Higher disturbance related to increases in invasive species, plant species were related to disturbance levels, many parameters related to decreasing plant community composition.

Appendix 2. Continued

Study	System	Sample Size	Dependent Variable	Independent Variable	Results
Hargiss *et al.* 2008	Prairie pothole wetlands in Montana, North Dakota, and South Dakota	215 wetlands	Index of plant community integrity	Visual inspection of disturbance within the wetland and surrounding landscape	All vegetation metrics tested were significant in indicating disturbance level.
Houlahan *et al.* 2006	Wetlands in Ontario, Canada	74 wetlands	Plant species richness and community composition	Land use characteristics (forest cover, water cover, road density, and agriculture cover) at different landscape scales(0-100m,0-250m, 0-300m, 0-400m, 0-500m, 0-750m, 0-1000m, 0-1250m, 0-1500m, 0-1750m, 0-2000m, 0-2250m, 0-2500m, 0-3000m, 0-4000m)	Positive relationship between wetland size and plant species richness, Landscape properties were significant predictors of plant species richness, the most significant scales were between 250m and 300m that affected wetland plant diversity.
Johnston and Brown 2013	Wetlands along the Great Lakes	48 freshwater coastal wetlands	Plant community composition	Water chemistry and aerial fraction of land uses	Land use was a better predictor of plant communities than water chemistry.
Lopez and Fennessy 2002	Wetlands in Ohio	20 depressional wetlands	FQAI	Relative disturbance within 100m of wetland edge	As relative disturbance increased, FQAI score decreased.
Mensing *et al.* 1998	Riparian wetlands in Minnesota	15 riparian wetlands associated with low-order streams	Abundance, species richness, and Shannon diversity of plants	Land use within catchment at 4 scales (500m, 1000m, 2500m, 5000m)	Vegetation richness and diversity had a negative relationship with agriculture.
Miller and Wardrop 2006	Central Pennsylvania headwater wetlands	40 headwater wetlands	Floristic Quality Assessment Index scores	Level 2 rapid assessment of buffering area to calculate disturbance	The floristic quality assessment index scores were highly correlated to disturbance.
Rooney *et al.* 2012	Wetlands in Alberta's Beaverhills watershed	45 wetlands	Plant-based index of biotic integrity (IBI)	Land use within different buffering areas (100m, 300m, 500m, 1000m, 1500m, 2000m, 3000m.	IBI scores were significantly predicted by every spatial scale. Road cover and proportion of disturbed land were consistent with the predictors of the IBI scores.

Appendix 2. Continued

Study	System	Sample Size	Dependent Variable	Independent Variable	Results
Tsai *et al.* 2012	Playa wetlands in Southern High Plains of Texas	80 playa wetlands	Plant community metrics	Landscape variables within 3 km (# of playas, percentage of urban area, percent area in CRP program), Local factors (water depth, sediment depth, playa area)	Water depth had a negative relationship with all plant community metrics. Wetlands with more cropland had more exotic species. However, wetland size had a very weak relationship with species richness.

Appendix 3. Studies showing relationships between macroinvertebrates and disturbance.

Study	System	Sample Size	Dependent Variable	Independent Variable	Results
Angeler *et al.* 2008	Wetlands in Spain	Resting eggs from 12 dry wetlands	Density of Branchiopods	Land use at different scales (100 m, 1 km, 5 km, 10 km), water quality	Local scales (100 m) influenced water quality, while only the 10 km land use scale influence densities (densities negatively related to cropland).
Azrina *et al.* 2006	Langat River, Malaysia	4 upper reaches and 4 lower reaches	Macroinvertebrate richness, diversity, and abundance	Water Quality, anthropogenic impacts	Total suspended solids and conductivity were negatively related to richness. Urban runoff negatively affected richness.
Bird *et al.* 2013	Wetlands in South Africa	90 isolated depressional wetlands	Macroinvertebrate variables (richness, IBI scores, etc.)	Human disturbance at 3 scales (within wetland, 100 m, and 500 m) grouped into 6 categories (0 being no human disturbance to 6 being highly disturbed	No clear relationship between macroinvertebrates and human disturbance.
Brazner *et al.* 2007	Coastal wetlands in the Great Lakes	75 coastal wetlands	Macroinvertebrates communities (richness and function)	Human disturbance index	Macroinvertebrates were relatively unresponsive to human disturbance.
Campbell *et al.* 2009	Wetlands and farm ponds in Minnesota	40 wetlands and farm ponds	Chironomid richness	Majority of land use within a 500 meter buffer (natural wetlands, ponds in a non-grazed grassland, ponds in a grazed grassland, pond with a lot of row crops)	Chironomid richness decreased as agricultural use of the surrounding lands increased, increased turbidity and Total nitrogen as agricultural use of the surrounding lands increased
Chipps *et al.* 2006	Wetlands in Upper Missouri River basin in North Dakota	10 wetlands	Wetland water quality and macroinvertebrate communities	Low or High agriculturally impacted wetlands (Low is <5% of area within 150 m, High is >33% of area within 10m)	Higher impacted wetlands had higher phosphorus and alkalinity. Higher impacted wetlands had higher Culicidae biomass, while having lower macroinvertebrate diversity and chironomidae abundance.
Hall et al. 2004	Playa wetlands in Southern High Plains of Texas	38 wetlands	Macroinvertebrate diversity	Predominant land use within 100m of wetland (agriculture, range, or CRP, insular characteristics	Land use had an effect on species richness only in first sampling period, some insular characteristics had an effect on species richness. They concluded that both insular and landscape characteristics influenced macroinvertebrate diversity.

Appendix 3. Continued

Study	System	Sample Size	Dependent Variable	Independent Variable	Results
Lammert and Allan 1999	Streams in southeastern Michigan	Six 100m stream reaches	Macroinvertebrate assemblages	Land use at different scales (within 50m, 125m, and entire subcatchment of stream section), in stream habitat characteristics	Land use immediate to tributaries predicted the macroinvertebrate community condition better than larger scales. However, in stream habitats explained more of the variance.
Mensing *et al.* 1998	Riparian wetlands in Minnesota	15 riparian wetlands associated with low-order streams	Abundance, species richness, and Shannon diversity for macroinvertebrates	Land use within catchment at 4 scales (500m, 1000m, 2500m, 5000m)	Macroinvertebrates were relatively unresponsive to human disturbance. They were more responsive to within wetland characteristics.
Meyer 2012	Wetlands in north-central Oklahoma	58 depressional wetlands	Invertebrate communities metrics	Local (predominant land use surrounding wetlands, soil, slope, plant cover) and landscape (land use of either cropland, range, or pasture within 1km and 2km)	Local factors (within wetland) explained more of macroinvertebrate communities than landscape factors. However, sampling date explained most of the variation.
Miserendino and Masi 2010	Patagonian low order streams	18 sites	Benthic invertebrates	Different land uses (native forest, pine plantation, pasture, harvest forest, urban, and reference urban)	Macroinvertebrate assemblage structure was altered by land use practices. Shredder richness was clearly higher at native and harvest forest than exotic pine plantations and total density was significantly higher at urban and harvest forest
Reece and McIntyre 2009	Wetlands in northern Texas	73 playa wetlands	Adult Odonate diversity	Predominant land use covering >75% area within 0.5km (either cropland or grassland)	Traditional community metrics showed no significant difference in diversity of Odonate assemblages between the two land use types.
Schäfer et al. 2006	Wetlands in Sweden	9 wetlands	Culicidae and Dytiscidae	Landscape variables a 5 different spatial scales (from 100m to 3000m)	Culicidae abundance was higher with a higher proportion of forest and standing water bodies. Dytiscidae abundance was only related to the amount of water bodies in the landscape.

Appendix 3. Continued

Study	System	Sample Size	Dependent Variable	Independent Variable	Results
Tangen *et al.* 2003	Wetlands in central North Dakota	24 prairie pothole wetlands	Macroinvertebrate IBI scores	Low impact (wetland basin primarily composed of grassland), Severe impact (wetland basin has >50% cropland), Moderate impact (in between low impact and severe impact)	No strong relationship between macroinvertebrate IBI scores and land use disturbance

Appendix 4. Studies showing the relationship of disturbance at different scales influencing communities.

Study	System	Sample Size	Dependent Variable	Independent Variable	Results
Angeler *et al.* 2008	Wetlands in Spain	Resting eggs from 12 dry wetlands	Density of Branchiopods	Land use at different scales (100 m, 1 km, 5 km, 10 km), water quality	Local scales (100 m) influenced water quality, while only the 10 km land use scale influence densities (densities negatively related to cropland).
Bird *et al.* 2013	Wetlands in South Africa	90 isolated depressional wetlands	Macroinvertebrate variables (richness, IBI scores, etc.)	Human disturbance at 3 scales (within wetland, 100 m, and 500 m) grouped into 6 categories (0 being no human disturbance to 6 being highly disturbed	No clear relationship between macroinvertebrates and human disturbance at any scale.
Houlahan *et al.* 2006	Wetlands in Ontario, Canada	74 wetlands	Plant species richness and community composition	Land use characteristics (forest cover, water cover, road density, and agriculture cover) at different landscape scales(0-100m,0-250m, 0-300m, 0-400m, 0-500m, 0-750m, 0-1000m, 0-1250m, 0-1500m, 0-1750m, 0-2000m, 0-2250m, 0-2500m, 0-3000m, 0-4000m)	Positive relationship between wetland size and plant species richness, Landscape properties were significant predictors of plant species richness, the most significant scales were between 250m and 300m that affected wetland plant diversity
Lammert and Allan 1999	Streams in southeastern Michigan	Six 100m stream reaches	Macroinvertebrate and fish assemblages	Land use at different scales (within 50m, 125m, and entire subcatchment of stream section), instream habitat characteristics	Land use immediate to tributaries predicted the macroinvertebrate community condition better than larger scales. However, instream habitats explained more of the variance. Fish showed a stronger relationship to flow variability and immediate land use.

Appendix 4. Continued

Study	System	Sample Size	Dependent Variable	Independent Variable	Results
Mensing *et al.* 1998	Riparian wetlands in Minnesota	15 riparian wetlands associated with low-order streams	Abundance, species richness, and Shannon diversity for plants, birds, fish, amphibians, and macroinvertebrates	Land use within catchment at 4 scales (500m, 1000m, 2500m, 5000m)	Vegetation, bird and fish richness and diversity had a negative relationship with agriculture. Macroinvertebrates, vegetation, amphibians, and birds responded more to local scales (500m and 100m) of human disturbance, while fish responded to more to the more regional scales (2500m and 5000m) of human disturbance
Meyer 2012	Wetlands in north-central Oklahoma	58 depressional wetlands	Invertebrate communities metrics	Local (predominant land use surrounding wetlands, soil, slope, plant cover) and landscape (land use of either cropland, range, or pasture within 1km and 2km)	Local factors (within wetland) explained more of macroinvertebrate communities than landscape factors. However, sampling date explained most of the variation.
Rooney *et al.* 2012	Wetlands in Alberta's Beaverhills watershed	45 wetlands	Plant and bird-based indices of biotic integrity (IBI)	Land use within different buffering areas (100m, 300m, 500m, 1000m, 1500m, 2000m, 3000m.	IBI scores were significantly predicted by every spatial scale. Plant based IBI scores were best predicted by data from within 100m buffers while bird-based IBI scores were best predicted from land use within 500m. Road cover and proportion of disturbed land were consistent with the predictors of the IBI scores.
Schäfer *et al.* 2006	Wetlands in Sweden	9 wetlands	Culicidae and Dytiscidae	Landscape variables a 5 different spatial scales (from 100m to 3000m)	Mosquito species assemblages were mainly influenced by forest cover at a large spatial scale, whereas the amount of water bodies was more important at local scales. Dytiscid species assemblages were mainly influenced by water permanence, especially at intermediate spatial scales.

Appendix 4. Continued

Study	System	Sample Size	Dependent Variable	Independent Variable	Results
Tsai *et al.* 2012	Playa wetlands in Southern High Plains of Texas	80 playa wetlands	Plant community metrics	Landscape variables within 3 km (# of playas, percentage of urban area, percent area in CRP program), Local factors (water depth, sediment depth, playa area)	Water depth had a negative relationship with all plant community metrics. Wetlands with more cropland had more exotic species. However, wetland size had a very weak relationship with species richness.
Whited *et al.* 2000	Wetlands in Minnesota	40 wetlands	Wetland bird assemblages	Landscape variables for 3 spatial scales (500m, 1000m, and 2500m)	Roads had the highest impact on bird assemblages at the 500m scale. Species richness was lowest in the urbanizing ecoregion, but the community patterns were not correlated to any landscape variables.

Appendix 5. Percent of the land within 100m of each individual wetland (n = 22) within the different land use categories from the 2012 CropScape data. This table also contains the LDI and LUS values for each wetland sampled.

Site	Developed, High Intensity	Developed, Medium Intensity	Developed, Low Intensity	Developed, Open Space	Cultivated Crops	Pasture/Hay	Grassland/ Herbaceous	Evergreen Forest
Ravia	0.0	0.0	0.0	0.0	0.5	0.0	8.6	0.0
Wister	0.0	0.0	2.9	48.5	0.0	0.0	0.0	1.5
Muldrow	0.0	0.0	0.0	3.8	12.0	41.5	0.0	0.0
Grassy Slough	0.0	0.0	0.0	0.0	0.0	1.6	0.8	0.8
Red River	0.0	0.0	0.0	1.1	59.4	11.4	1.1	0.0
Eagleton	0.0	0.0	15.9	20.7	0.0	13.4	0.0	6.1
Boynton	0.0	0.0	0.0	30.8	0.0	18.3	31.7	0.0
Oologah North	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Oologah South	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
TNC Nickel	0.0	0.0	0.0	6.1	0.0	14.9	16.7	0.9
Tahlequah	0.0	1.4	4.1	0.0	5.4	47.3	0.0	0.0
Rt51 West	0.0	0.0	1.6	21.9	0.0	12.5	3.1	0.0
Boheler Seeps	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hulah	0.0	0.0	0.0	0.0	0.0	6.9	44.8	0.0
DFWMA Oxbow	0.0	0.0	0.0	0.0	0.5	0.0	7.3	0.0
Drummond Flats	0.0	0.0	0.0	0.0	0.0	0.0	100.0	0.0
Heyburn	0.0	0.0	0.0	17.4	0.0	0.0	4.7	0.0
McClellan	0.0	0.0	0.0	0.0	0.0	0.6	6.3	0.0
DFNWR1	0.0	0.0	0.0	10.7	0.0	0.7	2.7	0.0
DFNWR2	0.0	0.0	0.0	0.0	0.0	1.0	13.0	0.0
DF-Storage	1.6	1.6	3.3	3.3	0.0	0.0	21.3	0.0
Hugo	0.0	0.0	10.5	9.9	0.0	1.8	0.0	0.6

Appendix 5. Continued

Site	Mixed Forest	Deciduous Forest	Shrub/Scrub	Wetlands	Open Water	LDI Value	LUS Value
Ravia	0.0	90.0	0.0	1.0	0.0	1.17	1.00
Wister	2.9	42.6	0.0	1.5	0.0	4.07	0.83
Muldrow	1.1	31.7	0.0	8.7	1.1	2.79	0.78
Grassy Slough	0.8	66.4	0.0	29.7	0.0	1.06	1.00
Red River	0.6	20.0	0.0	6.3	0.0	3.50	0.55
Eagleton	1.2	40.2	0.0	2.4	0.0	3.66	0.77
Boynton	0.0	19.2	0.0	0.0	0.0	3.88	0.85
Oologah North	0.0	100.0	0.0	0.0	0.0	1.00	1.00
Oologah South	0.0	100.0	0.0	0.0	0.0	1.00	1.00
TNC Nickel	0.0	50.9	6.1	4.4	0.0	2.07	0.94
Tahlequah	0.0	35.1	0.0	0.0	6.8	2.85	0.77
Rt51 West	0.0	57.8	0.0	3.1	0.0	2.79	0.88
Boheler Seeps	0.0	100.0	0.0	0.0	0.0	1.00	1.00
Hulah	0.0	48.3	0.0	0.0	0.0	1.98	0.98
DFWMA Oxbow	0.0	92.2	0.0	0.0	0.0	1.15	1.00
Drummond Flats	0.0	0.0	0.0	0.0	0.0	2.77	100
Heyburn	0.0	77.9	0.0	0.0	0.0	2.11	0.95
McClellan	0.0	90.3	0.0	0.0	2.8	1.13	1.00
DFNWR1	0.0	85.9	0.0	0.0	0.0	1.70	0.97
DFNWR2	0.0	82.5	0.0	0.0	3.5	1.26	1.00
DF-Storage	0.0	65.6	0.0	0.0	3.2	2.06	0.93
Hugo	0.0	66.1	0.0	11.1	0.0	2.32	0.88

Appendix 6. List of invertebrate taxa collected during the summer of 2013 from the 22 Oklahoma wetlands.

Order	Family	Genus
Collembola		
	Isotomidae	
	Sminthuridae	
Ephemeroptera		
	Baetidae	
		Callibaetis
	Caenidae	
		Caenis
Odonata		
	Gomphidae	
		Phyllogomphoides
	Aeshnidae	
		Anax
		Coryphaeschna
		Nasiaeshna
	Libellulidae	
		Erythemis
		Libellula
		Pachydiplax
		Perithemis
		Plathemis
		Pseudoleon
		Sympetrum
		TUSA-RAMea
	Lestidae	
		Lestes
	Coenagrionidae	
		Argia
		Enallagma
Hemiptera		
	Hydrometridae	
		Hydrometra
	Macroveliidae	
		Macrovelia
	Veliidae	
		Microvelia
	Gerridae	
		Gerris
		Limnoporus

		Trepobates
	Belostomatidae	
		Belostoma
	Nepidae	
		Ranatra
	Pleidae	
		Neoplea
	Naucoridae	
		Pelocoris
	Corixidae	
		Hesperocorixa
		Rhamphocorixa
		Sigara
		Trichocorixa
	Notonectidae	
		Buenoa
		Notonecta
	Mesoveliidae	
		Mesovelia
	Hebridae	
		Lipogomphus
	Saldidae	
		Saldoida
	Unknown 1	
Megaloptera		
	Sialidae	
		Sialis
	Corydalidae	
		Chauliodes
Trichoptera		
	Hydroptilidae	
		Orchrotrichia
		Oxyethira
	Leptoceridae	
		Oecetis
Lepidoptera		
	Crambidae	
	Noctuidae	
Coleoptera		
	Gyrinidae	
		Dineutus $_3$
		Gyrinus $_1$
	Carabidae $_2$	

Haliplidae

- *Haliplus* $_{3}$
- *Peltodytes* $_{3}$

Dytiscidae

- *Acilius* $_{1}$
- *Agabetes* $_{1}$
- *Agabinus* $_{1}$
- *Agabus* $_{1}$
- *Celina* $_{2}$
- *Copelatus* $_{3}$
- *Coptotomus* $_{3}$
- *Cybister* $_{1}$
- *Desmopachria* $_{3}$
- *Graphoderus* $_{3}$
- *Hydaticus* $_{1}$
- *Hydroporus* $_{2}$
- *Laccophilus* $_{3}$
- *Liodessus* $_{2}$
- *Oreodytes* $_{3}$
- *Thermonectus* $_{2}$
- *Neoporus* $_{3}$

Noteridae

- *Hydrocanthus* $_{3}$
- *Suphisellus* $_{2}$

Histeridae $_{2}$

Hydrophilidae

- *Berosus* $_{3}$
- *Derralus* $_{3}$
- *Enochrus* $_{3}$
- *Epimetopus* $_{1}$
- *Helochares* $_{3}$
- *Hydrochara* $_{3}$
- *Hydrochus* $_{3}$
- *Hydrophilis* $_{2}$
- *Laccobius* $_{3}$
- *Paracymus* $_{2}$
- *Tropisternus* $_{3}$

Staphylinidae $_{2}$

Tenebrionidae $_{2}$

Scirtidae

- *Cyphon* $_{1}$
- *Prionocyphon* $_{2}$
- *Scirtes* $_{3}$

	Elmidae	
		Ancronyx [1]
		Stenelmis [1]
	Ptilodactylidae	
		Anchytarsus [2]
	Chrysomelidae [3]	
	Curculionidae [3]	
	Anthicidae [3]	
	Scarabaeidae [2]	
	Unknown 1 [1]	
	Unknown 2 [2]	
Diptera		
	Ceratopogonidae	
		Atrichopogon
		Forcipomyia
		Alluaudomyia
		Bezzia
		Culicoides
		Probezzia
		Serromyia
		Spaeromias
	Chaoboridae	
		Chaoborus
	Chironomidae	
	Culicidae	
		Anopheles
		Culex
		Mansonia
		Orthopodomyia
		Uranotaenia
	Tipulidae	
		Limonia
	Stratiomyidae	
		Odontomyia
		Stratiomys
	Tabanidae	
		Chlorotabanus
		Tabanus
		Hybomitra
	Ephyridae	
		Brachydeutera
		Setacera

		Nostima
	Sciomyzidae	
		Tetanocera

1 - larvae only, 2 - adult only, 3 - larvae and adult

Appendix 7. List of plant taxa identified and their nativity to Oklahoma during 2012 and 2013 from 22 Oklahoma wetlands. For nativity, N= native A= alien, as listed by the Oklahoma Invasive Species Council. Also listed is the Coefficient of Conservatism (CoC) used for assessing the Floristic Quality Index. Lower CoC values represent species that are more tolerable to disturbances. Alien species are given a CoC value of 0.

Order	Family	Species	Nativity	CoC
Alismatales				
	Alismataceae			
		Alisma subcordatum	N	6
		Echinodorus cordifolius	N	8
		Sagittaria brevirostra	N	4
		Sagittaria graminea	N	8
		Sagittaria latifolia	N	5
		Sagittaria platyphylla	N	7
	Hydrocharitaceae			
		Limnobium spongia	N	8
	Lemnaceae			
		Lemna minuta	N	5
		Lemna valvidiana	N	7
		Spirodela polyrhiza	N	6
		Wolffia columbiana	N	5
	Potamogetonaceae			
		Stuckenia pectinata	N	7
		Potamogeton diversifolius	N	6
Apiales				
	Apiaceae			
		Cicuta maculata	N	4
		Hydrocotyle ranunculoides	N	4
		Hydrocotyle umbellata	N	6
		Limnosciadium pinnatum	N	6
		Ptilimnium capillaceum	N	4
		Torilis arvensis	A	0
Asparagales				
	Alliaceae			
		Allium canadense	N	3
Asterales				
	Asteraceae			
		Achillea millefolium	N	5
		Ageratina altissima	N	5
		Ambrosia artemisiifolia	N	6

		Ambrosia psilostachya	N	6
		Ambrosia trifida	N	2
		Bidens aristosa	N	6
		Cirsium horridulum	N	5
		Cirsium undulatum	N	4
		Conoclinium coelestinum	N	4
		Eclipta prostrata	N	3
		Erigeron canadensis	A	0
		Eupatorium perfoliatum	N	3
		Gamochaeta purpurea	N	3
		Iva annua	N	1
		Lactuca canadensis	N	2
		Mikania scandens	N	5
		Packera glabella	N	3
		Pluchea camphorata	N	4
		Pluchea odorata	N	4
		Senecio hieraciifolius	N	3
		Solidago canadensis	N	3
		Solidago gigantea	N	2
		Solidago rugosa	N	4
		Xanthium strumarium	N	0
Brassicales				
	Brassicaceae			
		Rorripa palustris	N	3
Caryophallales				
	Amaranthaceae			
		Alternanathera philoxeroides	A	0
	Polygonaceae			
		Brunnichia ovata	N	6
		Persicaria hydropiper	A	0
		Persicaria hyropiperoides	N	4
		Persicaria lapathifolia	N	4
		Persicaria pensylvanica	N	2
		Persicaria punctata	N	4
		Persicaria sagittata	N	4
		Rumex crispus	A	0
	Ceratophyllaceae			
		Ceratophyllum demersum	N	5
		Ceratophyllum echinatum	N	6

Commelinales				
	Commelinaceae			
		Commelina communis	A	0
		Commelina virginica	N	4
Cucurbitales				
	Cucurbitaceae			
		Melothria pendula	N	1
Dipsacales				
	Adoxaceae			
		Sambucus nigra	N	3
	Caprifoliaceae			
		Lonicera japonica	A	0
		Symphoricarpos orbiculatus	N	1
Ericales				
	Balsaminaceae			
		Impatiens capensis	N	5
	Ebenaceae			
		Diospyros virginiana	N	2
Fabales				
	Fabaceae			
		Amorpha fruticosa	N	6
		Amphicarpaea bracteata	N	3
		Apios americana	N	6
		Gleditsia triacanthos	N	2
		Lespedeza cuneata	A	0
		Sesbania vesicaria	N	2
	Betulaceae			4
		Alnus serrulata	N	3
		Betula nigra	N	3
	Fagaceae			
		Quercus alba	N	6
	Juglandaceae			
		Carya illinoinensis	N	6
Gentianales				
	Apocynaceae			
		Thyrsanthella difformis	N	6
	Asclepiadaceae			
		Asclepias incarnata	N	5
		Matelea cynanchoides	N	6
	Rubiaceae			
		Cephalanthus occidentalis	N	4

		Galium tinctorium	N	6
Lamiales				
	Acanthaceae			
		Justicia americana	N	5
	Bignoniaceae			
		Campsis radicans	N	3
	Lamiaceae			
		Lycopus americanus	N	4
		Teucrium canadense	N	3
	Lentibulariaceae			
		Utricularia gibba	N	6
		Utricularia macrorhiza	N	9
	Oleaceae			
		Forestiera acuminata	N	7
		Fraxinus americana	N	6
		Fraxinus pennsylvanica	N	3
	Plantaginaceae			
		Callitriche heterophylla	N	5
		Veronica peregrina	N	2
	Verbenaceae			
		Phyla lanceolata	N	3
		Verbena urticifolia	N	3
Liliales				
	Smilacaceae			
		Smilax bona-nox	N	5
		Smilax tamnoides	N	3
Malpighiales				
	Hypericaceae			
		Hypericum mutilum	N	4
		Hypericum virginicum	N	9
	Salicaceae			
		Populus deltoides	N	1
		Salix nigra	N	2
	Violaceae			
		Viola sororia	N	2
Malvales				
	Malvaceae			
		Hibiscus laevis	N	4
		Hibiscus moscheutos	N	4
	Lythraceae			
		Ammannia coccinea	N	6

		Didiplis diandra	N	7
		Rotala ramosior	N	4
	Melastomaceae			
		Rhexia mariana	N	7
	Onagraceae			
		Ludwigia alternifolia	N	5
		Ludwigia decurrens	N	5
		Ludwigia glandulosa	N	5
		Ludwigia peploides	N	6
Nymphaeales		*Ludwigia repens*	N	6
	Cabombaceae			
		Brasenia schreberi	N	5
	Nymphaeaceae			
		Nuphar advena	N	6
Piperales				
	Saururaceae			
		Saururus cernuus	N	6
Poales				
	Cyperaceae			
		Carex annectens	N	4
		Carex crus-corvi	N	7
		Carex frankii	N	5
		Carex gigantea	N	6
		Carex granularis	N	5
		Carex lupulina	N	6
		Carex scoparia	N	5
		Carex tribuloides	N	4
		Cyperus strigosus	N	4
		Dulichium arundinaceum	N	8
		Eleocharis compressa	N	6
		Eleocharis lanceolata	N	7
		Eleocharis obtusa	N	4
		Eleocharis quadrangulata	N	7
		Fimbristylis puberula	N	4
		Fimbristylis vahlii	N	6
		Rhynchospora corniculata	N	7
		Scirpus cyperinus	N	7
		Scleria oligantha	N	7
	Juncaceae			
		Juncus acuminatus	N	5
		Juncus diffusissimus	N	5

		Juncus effusus	N	5
	Poaceae			
		Agrostis hyemalis	N	3
		Chasmanthium latifolium	N	4
		Dichanthelium acuminatum	N	4
		Dichanthelium oligosanthes	N	5
		Distichlis spicata	N	4
		Echinochloa muricata	N	0
		Glyceria striata	N	6
		Leersia oryzoides	N	4
		Leersia virginica	N	4
		Setaria pumila	A	0
		Sorghum halepense	A	0
		Sphenopholis intermedia	N	5
		Sphenopholis obtusata	N	2
		Zizaniopsis miliacea	N	9
	Typhaceae			
		Typha angustifolia	A	0
		Typha latifolia	N	2
Polypodiales				
	Onocleaceae			
		Onoclea sensibilis	N	9
Proteales				
	Nelumbonaceae			
		Nelumbo lutea	N	6
Ranunculales				
	Menispermaceae			
		Menispermum canadense	N	4
	Ranunculaceae			
		Ranunculus sceleratus	N	3
Rosales				
	Cannabaceae			
		Celtis occidentalis	N	5
	Moraceae			
		Maclura pomifera	N	0
	Rosaceae			
		Geum canadense	N	2
	Ulmaceae			
		Ulmus americana	N	2
	Urticaceae			
		Boehmeria cylindrica	N	6

Salviniales				
	Salviniaceae			
		Azolla cristata	N	6
Sapindales				
	Anacardiaceae			
		Toxicodendron radicans	N	1
	Sapindaceae			
		Acer negundo	N	1
		Acer rubrum	N	6
Saxifragales				
	Haloragaceae			
		Myriophyllum heterophyllum	N	8
	Penthoraceae			
		Penthorum sedoides	N	5
Solonales				
	Hydroleaceae			
		Hydrolea ovata	N	8
Vitales				
	Vitaceae			
		Ampelopsis arborea	N	4
		Ampelopsis cordata	N	2
		Vitis riparia	N	4
Zingiberales				
	Marantaceae			
		Thalia dealbata	N	7

VITA

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