WETLAND INVERTEBRATE COMMUNITY RESPONSE TO LAND-USE AND OTHER LOCAL AND LANDSCAPE FACTORS

By

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WETLAND INVERTEBRATE COMMUNITY RESPONSE TO LAND-USE AND OTHER LOCAL AND LANDSCAPE FACTORS

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CHAPTER I

RESPONSE OF INVERTEBRATE COMMUNITIES TO LAND-USE PRACTICES SURROUNDING DEPRESSIONAL WETLANDS IN NORTH CENTRAL OKLAHOMA

Abstract Wetlands provide many services, however, land-use practices may impact those services. In particular, wetlands imbedded in highly modified agricultural landscapes (i.e., annually tilled crop systems) may become severely degraded such that services may be impaired. Because invertebrates play an important role in nutrient cycling and also serve as critical food sources for waterbirds, they can be used to assess impacts on some services. Wetlands in Oklahoma occur within highly modified agricultural landscapes, and therefore, there is a need to assess the impact of these landscapes on invertebrate communities within these wetlands. My objectives were to determine the effects of different land-use practices on invertebrate diversity and biomass of invertebrate taxonomic and functional groups within depressional wetlands of north central Oklahoma. In 2009 and 2010, I sampled invertebrates from 58 wetlands located in cropland, pastureland, and native rangeland. Invertebrate taxa richness was higher in range wetlands than crop wetlands both years. Total biomass did not differ among land-use practices during both years, while the biomass of collector filterers and shredders was lowest in crop wetlands and herbivore biomass was higher in range than crop wetlands during 2010. The taxa Naididae, *Callibaetis*, *Enallagma*, and *Haliplus* larvae all had higher biomasses in range wetlands than crop wetlands during 2009, while in 2010 Cladocera and *Tropisternus* larvae biomasses were higher in range than crop wetlands. Also in 2010, Calanoida biomass was higher in range wetlands than pasture and crop wetlands, while Ostracoda biomass was higher in range and pasture wetlands than crop wetlands. The leech, *Mooreobdella*, had higher biomasses in crop wetlands than pasture and range wetlands during both years. Invertebrate taxa composition also differed among land-use practices during both years. However, the majority of functional feeding groups and taxa were similar among land-use practices. This suggests that the majority of the invertebrate groups were not affected by the changes in vegetation and water quality within the different land-use practices, or that high variability of invertebrate populations within land-use practices prevented finding more significant results. My results indicate some degradation of wetlands from land-use practices based on lower richness and lower biomass of some taxa in crop wetlands; however, the responses within the invertebrate community were limited.

Keywords Depressional wetlands · Invertebrates · Land-use · Oklahoma · Wetland degradation

Introduction

Wetlands provide many services including groundwater recharge, flood storage, and wildlife habitat that make them an integral part of the landscape (Mitsch and Gosselink 2007, Smith et al. 2011). Moreover, wetlands increase the biodiversity of an area by providing unique habitat for countless wildlife (Bolen et al. 1989). However, over 50% of the wetlands in the conterminous United States have been lost since European settlement (Dahl 1990). While wetland loss has moderated in recent decades (Dahl 2011), many wetlands are still at risk of degradation. Dominant causes of continued degradation are modification (e.g., pit construction) and changes in land-use practices (e.g., from native vegetation to annual tillage). In many regions of the United States, the dominant land-use practice is production agriculture, however, the full extent to which this land-use impacts wetland services is still unknown.

Several studies have indicated that changes in land-use to annual cropland systems has led to wetland degradation. In particular, cropland land-use has been shown to increase runoff as well as sedimentation into wetlands (Euliss and Mushet 1996, Luo et al. 1997). Increased runoff from uplands can increase wetland hydroperiods, while increased sedimentation can shorten a wetland’s hydroperiod by filling the wetland basin. Both of these factors can result in concomitant changes in the biological community. For example, increased sedimentation can impact plant communities by burying seeds and impact invertebrate communities by burying diapausing adults and aestivating eggs (Gleason et al. 2003). Agrichemicals used in surrounding cropland, pastureland, and rangeland may also impact wetlands. Fertilizers may cause eutrophication of wetlands (Brinson and Malvárez 2002), while pesticides can cause direct mortality of invertebrates and decrease plant diversity (Grue et al. 1986). Additionally, some wetlands are further impacted by tillage within the wetland when surface water is absent (Euliss and Mushet 1999). Tillage can have similar impacts as sedimentation.

Invertebrates are crucial components of wetland systems. They facilitate nutrient cycling within wetlands by processing living and dead plant material, and also provide a link between primary production and consumers (Wissinger 1999). Invertebrates are especially critical as food resources for many waterbirds. Waterfowl depend on invertebrates during all stages of their life cycle (Baldassarre and Bolen 2006). Similarly, shorebirds depend heavily on invertebrates as a food source (Skagen and Oman 1996). Additionally, the use of wetlands by waterbirds can be affected by the presence of invertebrate food resources (Davis and Smith 1998, Andrei et al. 2008). Because invertebrates are so critical for waterbirds as well other wetland services, there certainly is a need to understand how land-use practices impact invertebrates.

The impact of land-use on invertebrates has been studied in some regions of United States. Cropland land-use has been shown to decrease diversity (Hall et al. 2004, Euliss and Mushet 1999), but has also been shown to increase Naididae biomass (Davis and Bidwell 2008) and decrease the abundance of amphipods (Anteau et al 2011). Therefore wetland invertebrate communities in Oklahoma may also be impacted by changes in land-use practices. Sixty-seven percent of the original wetlands have been lost since European settlement in Oklahoma, and those wetlands remaining are still at risk for degradation (Dahl 1990). Depressional wetlands in Oklahoma are imbedded in highly modified agricultural landscapes. Yet, these wetlands provide critical habitat for breeding and wintering waterfowl (Heitmeyer and Vohs 1984a, Heitmeyer and Vohs 1984b), migrating shorebirds (Albanese 2011), and other wildlife (Henley and Harrison 2000).

Because of the importance of these wetlands for waterbirds and the importance of invertebrates in waterbird diets and wetland services, it is imperative that we understand the impacts of land-use practices on this important group of biota. Therefore, my objective was to determine the impact of land-use practices, specifically cropland, pastureland, and native rangeland on wetland invertebrate diversity, taxa and functional group biomass, in depressional wetlands of north central Oklahoma.

Methods

Study Area

This study was conducted in Garfield, Grant, Kingfisher, and Major counties encompassing an area of 365 km^2 in north central Oklahoma (Fig. 1). This region contains a diverse group of wetlands that include closed depressional, palustrine wetlands with temporary, seasonal, and semipermanent hydroperiods (Cowardin et al. 1979, Brinson 1993). These wetlands are located within the terraces of the Cimarron River and Salt Fork of the Arkansas River. Wetland size ranges from < 1 ha to 20 ha. Dominant hydrologic influences are rainfall and runoff, with some groundwater influences. Flooding of wetlands generally occurs from fall and winter precipitation, but summer storms may also contribute to flooding (Henley and Harrison 2000). Wetland vegetation is dominated by herbaceous emergent plants that include water knotweed (*Polygonum amphibium*, L.), chairmaker's bulrush [*Schoenoplectus americanus*, (Pers.) Volkart ex

Schinz & R. Keller], and upright burhead [*Echinodorus berteroi* (Spreng.) Fassett] and submergent aquatic plants that include southern waternymph [*Najas guadalupensis*, (Spreng.) Magnus] and waterthread pondweed (*Potamogeton diversifolius*, Raf.). The upland soils surrounding depressions are dominated by Meno loamy fine sand, Lovedale fine sandy loam, and Nobscot fine sand, while soils within depressions are dominated by Carwile loam. Most depressional soils have clayey subsoil (NRCS 2008).

Land-use practices surrounding wetlands are predominantly croplands and livestock grazing of native rangelands and improved pasturelands. Croplands are primarily winter wheat or rye, and are commonly grazed during the winter. Native rangelands consist of mixedgrass and tallgrass prairie that include sand bluestem (*Andropogon hallii* Hack.), little bluestem [*Scizachyrium scoparium*, (Michx.) Nash], switchgrass (*Panicum virgatum* L.), sideoats grama [*Bouteloua curtipendula* (Michx.) Torr.], and Chickasaw plum (*Prunus angustifolia* Marsh.), while improved pasturelands consist primarily of Bermudagrass [*Cynodon dactylon* (L.) Pers.]. Both rangelands and pasturelands are commonly grazed throughout the year. Wetlands located within croplands and pasturelands are usually tilled and replanted with crops or forage during dry years.

Average annual precipitation in the study area ranges from 74.7 to 85.7 cm, with most of the precipitation occurring from April through September (Oklahoma Climatological Survey 2005). Average annual temperatures range from 15.0 to 15.6° C with average minimum temperatures during January ranging from -6.1 to -3.8° C and average maximum temperatures during July ranging from 34.8 to 35.7° C. The average growing season for the study area ranges from 201 to 208 days (Oklahoma Climatological Survey 2005).

Data Collection

During the two year study, I selected 58 wetlands for inclusion in the study. Wetlands were selected from a larger group of available wetlands to encompass a gradient of hydroperiods from the most temporary to the most permanent within the three land-use practices (crop, pasture, and range). Land-use type was classified by the dominant landuse immediately surrounding each wetland. Hydroperiods were determined based on the dominance of certain plant communities (e.g., certain species tend to dominate temporary wetlands), observations of changes in water levels from several site visits, and observation of the changes in hydrological conditions between several years as determined from orthophotography. Thirty-seven wetlands were sampled during 2009 (11 pasture, 13 range, and 13 crop), while 40 wetlands were sampled during 2010 (13 pasture, 14 range, and 13 crop).

I collected invertebrates from April through July in 2009 and from March through July in 2010. Invertebrates were collected every three weeks, for a total of four sampling periods during 2009 and five sampling periods during 2010. Because some of the wetlands dried before the end of each season, I was unable to sample all wetlands during every sampling period (Table 1). Prior to collecting invertebrates, I divided each wetland into strata based on dominant plant types. Invertebrate communities are strongly affected by plant type, and this ensured a more complete coverage of the wetland and increased the probability that the majority of invertebrate communities were represented within the wetland (Voigts 1976). A stratum consisted of areas dominated by submergent or emergent vegetation or areas devoid of vegetation. Within each stratum, I located random points to sample invertebrates. Random locations were determined by entering the wetland from a random direction and proceeding a random distance. For wetlands with one stratum, five points were sampled, while for wetlands with two strata, three points were sampled in each stratum. At each sample point, one 50 × 50 cm quadrat of clipped vegetation (DeCoster and Persoone 1970, Anderson and Smith 1996), two 5.2 cm diameter water column samples (Swanson 1978, Anderson and Smith 1996), and one 5.2 cm diameter benthic core sample were collected (Swanson 1983).

Following collection of invertebrates, each benthic core sample was placed in a sampling jar containing 75% ethanol, and clipped vegetation samples were bagged and refrigerated (4° C) for processing in the lab within 5 days of collection (Anderson and Smith 1998). The two water column samples from each site were combined and processed in the field by sieving through a 500 µm (No. 35) sieve, and all retained invertebrates were preserved in 75% ethanol. During the field processing of water column samples and benthic core samples, I recorded the volume of water that was sieved as well as the length of the core sample. I determined the volume of vegetation sampled by multiplying the area of the quadrat by the water depth at the sampling site. Benthic core samples were processed in the lab by washing through a 500 µm sieve and retained invertebrates were preserved in 75% ethanol. Epiphytic samples were also processed in the lab and collected invertebrates were preserved in 75% ethanol. All invertebrates were identified to genera, when possible, according to Merritt et al. (2008)

for insects and Smith (2001) for other invertebrates. All invertebrates were counted, dried at 65°C for 48 hrs, and weighed to determine dry biomass. Voucher specimens are stored at the Department of Natural Resource Ecology and Management at Oklahoma State University.

To determine if environmental variables may be affecting responses within land-use practices, I measured a variety of environmental factors in the wetland. At each sampling point, I recorded water depth, dissolved oxygen, water temperature, pH, conductivity, plant taxa richness, percent emergent and submergent plant cover, and vegetation complexity. I used a YSI multiparameter water quality meter (YSI Incorporated, Yellow Springs, Ohio) to determine dissolved oxygen, water temperature, pH, and conductivity. I calculated percent emergent and submergent cover by digitally photographing (Olympus 1030SW, 314 dpi) the 50 × 50 cm quadrat. The digital photo was viewed on a 39.1 cm monitor at full screen under a 1 cm dot grid and percent cover was determined by the percent of points that covered vegetation within the quadrat (de Szalay and Resh 2000). For vegetation complexity, I used an index ranging from high to low complexity to estimate structural complexity of plants. I rated each sample's vegetation complexity as 1, 2, or 3 based on the taxa present within the quadrat. For example, southern waternymph, an aquatic submergent with diffuse branching and many leaves, was rated high complexity (3), while chairmaker's bulrush, an emergent with no branching leaves, was rated low complexity (1). Assessing structural complexity is important because it can affect invertebrate colonization and production (Hinojosa-Garro et al. 2010). To determine ammonia and nitrate-nitrite nitrogen, orthophosphate, and turbidity, I collected two 1-L water samples at each wetland during each sampling period.

Nutrients were analyzed within 24 hrs using a Hach 850 Colorimeter and turbidity was assessed using a Hach Turbidimeter (Hach Company, Loveland, Colorado).

Data Analyses

Prior to analysis, data from samples at each site were combined to provide an estimate of invertebrate biomass (g/m^3). All samples from the wetland were averaged to provide an estimate for the wetland during each sampling period. I used different transformations (natural log, squareroot, and fourthroot) to meet the assumptions of parametric tests (Quinn and Keough 2002). I used separate multivariate analysis of covariance (MANCOVA) tests to evaluate the effects of different landuses on invertebrate communities during each year (Kostecke et al. 2005, Davis and Bidwell 2008). Because responses between invertebrate groups are highly correlated, a MANCOVA allows simultaneous analyses across the entire community. Years were analyzed separately because of differences between wetlands and hydrological conditions between years. I used the covariate model of multivariate analysis to partition the effects of sampling date and hydroperiod (Quinn and Keough 2002). Sampling date was categorized as the number of days between the initial sampling date and the date of collection. Hydroperiod was assessed as a scale from least to most permanent and coded as 1-5.

I conducted two separate MANCOVAs during on each year’s data. One for taxa richness, Shannon index (*H’*), and Pielou’s evenness index (*J’*) and one on total biomass of nonmicrocrustaceans, total biomass of microcrustaceans, biomass of functional feeding groups (FFGs), and biomass of the most common taxa (i.e., those occurring in ≥

10% of the total samples). The taxa Cladocera, Copepoda (Calanoida and Cyclopida during the second year), and Ostracoda were included in the microcrustacean group, while collector-filterer, collector-gatherer, omnivore, piercer-herbivore, scraper, shredder, herbivore, and predator were included in the FFGs. Taxa were assigned to FFGs based on published accounts (Smith 2001, Merritt et al. 2008, Thorp and Covich 2009). Land-use practice was used as the predictor, and hydroperiod and sampling days as covariates in the MANCOVA model. Wilk's λ was used as the test criterion for the MANCOVA. Following a significant MANCOVA ($P \leq 0.05$), analysis of covariance (ANCOVA), using the same above MANCOVA model was used separately for each response variable (Barker and Barker 1984). I then used pairwise comparisons using the Tukey method to determine differences among land-use practices.

To determine if any environmental variables differed among land-use practices, I used the same above MANCOVA and ANCOVA models. Prior to conducting the analyses, I transformed the variables (natural log and squareroot) to meet the assumptions for parametric tests. I conducted all statistical analyses using Minitab® 16.2.1 (Minitab Inc., State College, Pennsylvania).

Results

Environmental Variables

Overall, environmental variables differed among land-use practices during 2009 (Wilks' $\lambda = 0.514$, $P < 0.001$). Water depth was higher in range than crop wetlands, while plant richness, vegetation complexity, and emergent plant cover were higher in range wetlands than in pasture and crop wetlands (Table 2).Turbidity and nitrate nitrogen were higher in crop wetlands than in range wetlands, while orthophosphate was higher in pasture wetlands than in range wetlands, and ammonia nitrogen was higher in crop and pasture wetlands than in range wetlands.

During 2010, environmental variables differed overall among land-use practices (Wilks' $\lambda = 0.730$, $P = 0.001$). pH was higher in crop wetlands than in range wetlands while dissolved oxygen was higher in crop and pasture wetlands than in range wetlands (Table 2). Vegetation complexity and plant richness were both higher in range and pasture wetlands than in crop wetlands, while emergent plant cover was higher in range wetlands than in crop wetlands. Turbidity, orthophosphate, ammonia nitrogen, and nitrate nitrogen were higher in crop wetlands than in pasture and range wetlands.

Invertebrate Composition

I identified 177 and 216 taxa in 2009 and 2010, respectively, with 138 of those taxa occurring during both years. Within the different land-use practices, 157 taxa were identified in crop wetlands (2009: 121 taxa, 2010: 119 taxa), 160 taxa in pasture wetlands (2009: 116 taxa, 2010: 121 taxa), and 174 taxa in range wetlands (2009: 129 taxa, 2010: 142 taxa). Twenty-two, 26, and 25 taxa were unique to crop, pasture, and range wetlands, respectively. In 2009, invertebrate biomass within crop wetlands was dominated by

Helisoma (27.9%), *Physa* (22.4%), and *Mooreobdella* (11.7%). In pasture wetlands, *Physa* (28.3%), *Helisoma* (20.1%), and Chironomidae (10.2%) were the dominant taxa, while in range wetlands, *Helisoma* (34.3%), *Physa* (19.7%), and *Tropisternus* adult (9.3%) were the dominant taxa. In 2010, the dominant invertebrate taxa in crop wetlands were Chironomidae (16.8%), *Mooreobdella* (16.2%), *Helisoma* (12.7%), and *Physa* (8.3%) and the dominant taxa in pasture wetlands were *Helisoma* (33.2%), Chironomidae (17.2%), and *Physa* (7.1%). *Helisoma* (38.9%), *Physa* (9.9%), and Cladocera (9.7%) were the dominant taxa in range wetlands.

Response of Invertebrate Taxa to Land-use Practices

Overall invertebrate diversity (i.e., taxa richness, Shannon index, and evenness combined) differed among land-use practices in 2009 (Wilks' $\lambda = 0.807$, $P = 0.002$). Taxa richness was higher in range than pasture and crop wetlands (Table 3). The Shannon index also differed among land-use practices, but means were not separated by pairwise comparisons. In 2010, overall diversity also differed among land-use practices (Wilks' $\lambda = 0.848$, $P < 0.001$). Taxa richness was highest in range wetlands and lowest in crop wetlands (Table 3).

Overall invertebrate biomass (i.e., FFGs and the individual taxa combined) differed among land-use practices in 2009 (Wilks' $\lambda = 0.257$, $P < 0.001$). However, none of the FFGs differed in biomass among land-use practices (Table 4). Total nonmicrocrustacean and total microcrustacean biomass also did not differ among land-use practices (Table 5). The biomass of 5 of the 23 most common taxa did differ among land-use practices. Naididae, *Callibaetis*, *Enallagma*, and *Haliplus* larvae biomasses were higher in range wetlands than in crop wetlands, but similar between range and pasture wetlands (Table 5). In contrast, the biomass of the leech taxon, *Mooreobdella*, was higher in crop wetlands than in range and pasture wetlands.

In 2010, overall invertebrate biomass differed among land-use practices (Wilks' $\lambda = 0.502$, $P < 0.001$). Collector-filterer and shredder biomasses were higher in range and pasture wetlands than in crop wetlands, while herbivore biomass was higher in range wetlands than in crop wetlands (Table 4). Total nonmicrocrustacean and total microcrustacean biomass did not differ among land-use practices. Five of the 23 most common taxa differed among land-use practices. Calanoida biomass was higher in range wetlands than pasture and cropland wetlands, while Ostracoda biomass was higher in range and pasture wetlands than in crop wetlands (Table 6). Cladocera biomass was higher in range wetlands than in crop wetlands, but was similar between range and pasture wetlands. *Tropisternus* larvae biomass was also higher in range wetlands than in crop wetlands and similar between range and pasture wetlands. In contrast, *Mooreobdella* biomass was higher in crop wetlands than in range and pasture wetlands.

Discussion

The goal of my research was to evaluate the influence of land-use practice on invertebrate communities in depressional wetlands of north central Oklahoma. Although many of the invertebrate metrics did not differ among land-use practices, I did observe differences among land-use practices for taxa richness, three FFGs (herbivore, collector-filterer, and shredder), and nine taxa (*Mooreobdella*, Naididae, *Callibaetis*, *Enallagma*, *Haliplus* larvae, *Tropisternus* larvae, Calanoida, Cladocera, and Ostracoda). The main assumption for my research was that wetlands within native rangeland would have been in better condition (i.e., reference wetlands) because of fewer modifications to the landscape with regard to landscape change.

The differences I found with taxa richness, the nine taxa, may be a result of the changes found in environmental conditions among the land-use practices. For example, changes in plant structure and plant community have been known to affect invertebrate abundance and diversity (Voigts 1976, Olson et al. 1995). Increases in vegetation complexity in range wetlands likely increased available structure for invertebrates and periphyton food sources, which can lead to increasing invertebrate production and colonization rates (Hinojosa-Garro et al. 2010). Similarly, emergent plant cover has been shown to impact colonization rates of invertebrates (de Szalay and Resh 2000). I did find that taxa richness was lower in crop wetlands which had the lowest emergent plant cover and vegetation complexity. Similarly, of the taxa that were different among land-use practices, all except one taxa had higher biomasses in range or pasture wetlands, coinciding with higher emergent cover and vegetation complexity. I also observed increased turbidity in crop wetlands which may have had a similar impact on the invertebrate community through its effects on plant communities. Increased turbidity can suppress plant growth in wetlands (Zimmer et al. 2000, Scheffer 2004, Hentges and Stewart 2010). Additionally, increased turbidity reduces periphyton and phytoplankton production which decreases important food sources for invertebrates (Euliss and Mushet

1999, Kalff 2002, Scheffer 2004), both which could have led to the reduction in biomass that I found in some taxa (e.g., Cladocera).

Changes in water quality may have also influenced the invertebrate communities among the different land-use practices. Within cropland land-use practices, increased nitrogen and phosphorus loading in wetlands can be a problem (Brinson and Malvárez 2002), and it appeared that crop wetlands in my study did have higher levels of both nitrogen and phosphorus than range wetlands. Another study has reported lower invertebrate taxa richness in wetlands with increased nitrogen levels (Hentges and Stewart 2010). Similarly, I found lower taxa richness on average in crop wetlands that contained higher nitrogen levels than range wetlands on average during both years. However, despite these changes in the plant community and water quality which indicate greater degradation in crop wetlands, I only found a minority of the taxa responded significantly to land-use practices.

Similar to my study, several of the studies which have assessed land-use impacts on wetland invertebrates reported few significant results. For example, one of the studies found differences limited to a few taxa (Davis and Bidwell 2008), while other studies only examined the impact of land-use on a few taxa (Euliss and Mushet 1999, Brose 2003, Anteau et al. 2011). In one case, no significant differences in the invertebrate community were found among land-use practices (Tangen et al. 2003). It is possible that while we found differences in emergent plant cover, vegetation complexity, and water quality, these differences were not biologically relevant to the majority of invertebrate taxa found in wetlands. It is reported that many wetland invertebrates are able to tolerate and exploit a wide variety of habitat conditions (Williams 1996). This may have led to the high variability of invertebrate communities that I found among the different land-use practices.

Other wetland studies have also found few significant factors that explain invertebrate variation between wetlands (Zimmer et al. 2000, Batzer et al. 2004, Hanson et al. 2009). Two studies in north central Minnesota attributed few significant findings to the high variability of invertebrate communities that inhabit depressional wetlands (Batzer et al. 2004, Hanson et al. 2009). Other studies have also attributed the high variability of invertebrate communities to stochastic variation (Zimmer et al. 2000, Batzer et al. 2004), which certainly could have been occurring in my study as well. A factor that may have further increased the variability of invertebrate populations in my study was the variability of habitat conditions within the same land-use practices. While it was more likely that range wetlands had diverse plant communities with clear water, there were range wetlands that appeared to be in poorer condition than other range wetlands with fewer plants and excessive turbidity. Similarly, crop wetlands were more likely to be turbid, but some had clear water with abundant and diverse plant communities. Consequently, habitat variability, the ability of wetland invertebrates to tolerate a wide range of conditions, and the inherent variability of invertebrate populations may have prevented this study from finding many significant results.

Other unmeasured variables may have also impacted our results. In a study similar to mine, they also found weak relationships between land-use practices and the invertebrate community in prairie pothole wetlands (Tangen et al. 2003). They attributed the lack of relationship with land-use to the effects of other factors. In their study, fish presence seemed to explain the most variation among wetlands; however, they were still not able to explain a majority of the variation among wetlands using any variable or set of variables. In my study, occurrence of fish in some of the wetlands could have also impacted invertebrates. Mosquitofish (*Gambusia* sp.) were observed in a few of my wetlands, but fish populations were not assessed, so it was not taken into account in any analysis. Mosquitofish are opportunistic omnivores, and can reduce some invertebrate taxa (Peck and Walton 2008). Another factor which may have affected my results was the impact of grazing. The majority of the wetlands in my study were grazed by cattle at least for a portion of the year. Grazing has been shown to have some effects on invertebrate communities (Steinman et al. 2003, Silver and Vimosi 2012), but the effects may be different depending on timing of grazing and whether grazing occurred in the wetlands or on the land surrounding wetlands. In my study, wetlands within the same grazing unit were often seemingly affected differently by cattle grazing. Nonetheless, it may have attributed to the high variability of invertebrate populations between wetlands. Additionally, pesticides and fertilizers applied to rangeland may have impacted invertebrate communities as well effectively reducing the difference between “reference” wetlands and pasture and crop wetlands.

One taxa of note is, *Mooreobdella*, an erpobdellid leech, which responded differently compared to all other taxa and functional groups. *Mooreobdella* was found in higher biomasses in crop wetlands during each year. Other studies have indicated that other annelids (i.e., oligochaetes) increase with environmental pollution in aquatic systems (Howmiller and Scott 1977, Lauritsen et al. 1985). However, I know of no other studies that report this taxa or erpobdellid leeches increasing in crop wetlands. In my study, *Mooreobdella*, dominated the invertebrate community in turbid crop wetlands, but was also found in high numbers in crop wetlands with diverse plant communities. Another study reported naidid oligochaetes were found with higher biomasses in crop wetlands of the Rainwater Basin Region in Nebraska (Davis and Bidwell 2008); however, I did not find this response. In fact, I found that Naididae had lower biomasses in crop wetlands during one season. In my study, numerous Naididae were found in vegetation and water column samples, as well as benthic samples, however, it was found that Naididae biomass was only higher in benthic samples of crop wetlands of the Rainwater Basin Region (Davis and Bidwell 2008). It is possible that regional variation within Naididae may be attributing to the opposite results that I found.

Conclusion

I found some impacts of land-use practices on wetland invertebrate communities. Nonetheless, the majority of invertebrates did not differ significantly between land-use practices. This may have been because the inherent variability of invertebrate populations limits our ability to determine any significant responses from different land-use practices. Based on the differences in plant communities and water quality parameters among land-use practices, more changes in the invertebrate community were expected. Even so, the level of change may have not met the threshold needed to have an effect on all invertebrate taxonomic groups. Future impact could also be greater in crop wetlands, as I observed an increase in the use of central pivot irrigation in this region of Oklahoma. Irrigation allows for additional crops, such as corn and cotton, to be planted as well as double cropping small grains with soybeans. These changes in cropping practices could put additional stresses on the wetlands and increase wetland degradation, further impacting invertebrate communities.

Both the differences found in invertebrate communities and the concomitant changes in vegetation and water quality suggest some degradation in crop wetlands compared to range wetlands. Therefore, protecting crop wetlands from tillage by advocating no-till cropping or utilizing permanent buffers may benefit invertebrate communities and the waterbirds utilizing these wetlands as well. Even though my results showed crop wetland had lower diversity than range wetlands, wetlands in all land-use practices attributed to higher regional biodiversity by adding unique taxa within each land-use practice.

Other studies have indicated that in addition to land-use, the location of a wetland compared to other wetlands within a landscape may also affect invertebrate communities (Batzer et al. 2004, Hall et al. 2004, Wissinger 1999). Therefore, further research is needed to understand impacts at the landscape scale so that we can better understand the role land-use plays in influencing invertebrate communities relative to other important factors.

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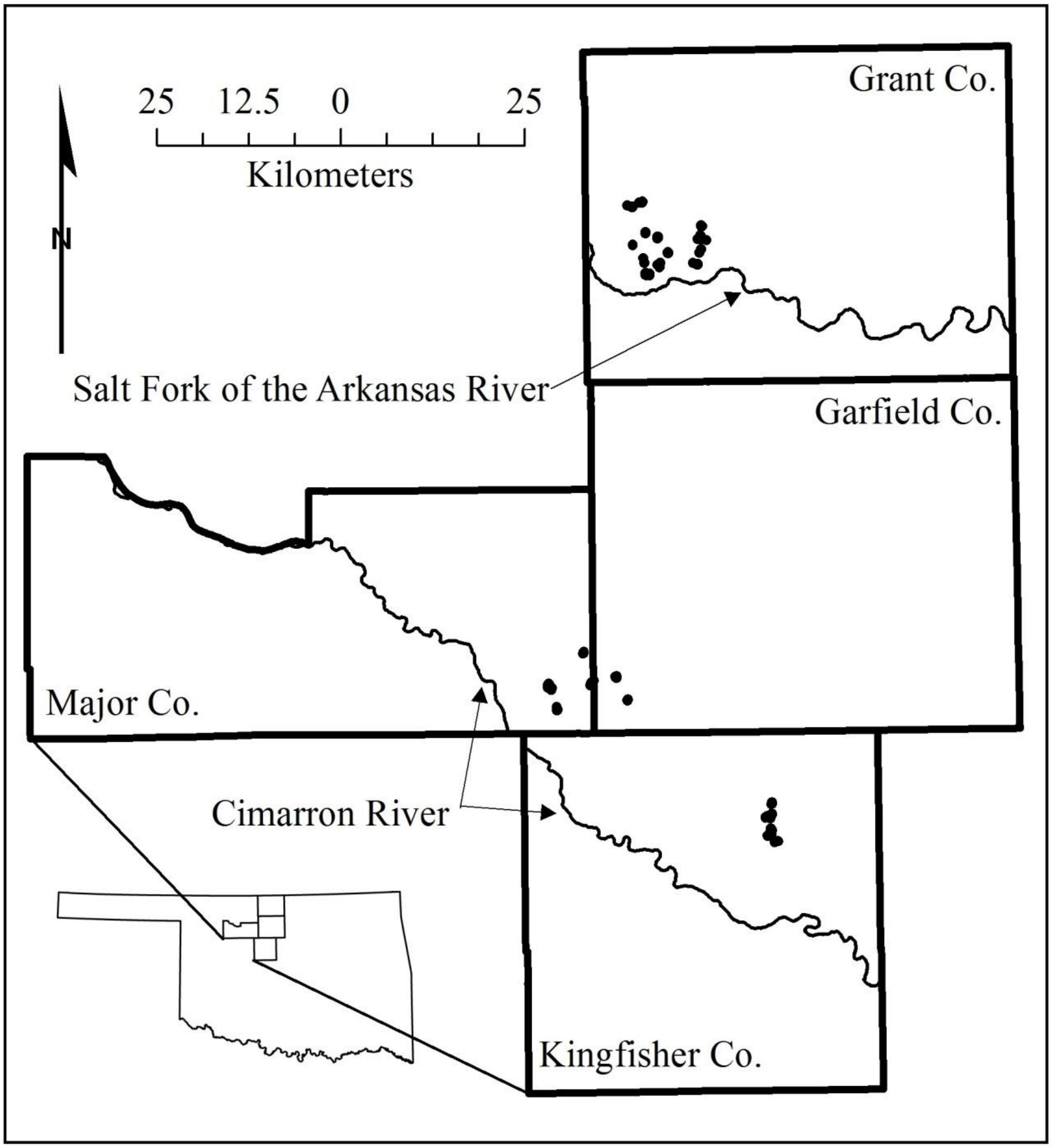
Figures

Fig. 1 Locations of study wetlands in Oklahoma, USA



Tables

Table 1 Number of depressional wetlands in north central Oklahoma sampled within each land-use practice during sampling periods in 2009 and 2010

Sampling Period	Range	Pasture	Crop
2009			
1 (April 20 - May 8)	2	2	4
2 (May 12 - June 1)	13	11	13
3 (June 3 - June 22)	12	10	12
4 (June 23 - July 11)	11	6	10
2010			
1 (March 17 - April 19)	11	12	10
2 (April 21 - May 17)	12	12	13
3 (May 18 - June 8)	14	13	13
4 (June 10 - July 1)	13	13	13
5 (July 5 - July 26)	13	13	13

Table 2 Mean (SE) and F and *P*-values of environmental variables for depressional wetlands surrounded by rangeland, pastureland, and cropland land-use practices in north central Oklahoma during 2009 and 2010. Different letters following means denote significant differences between land-use practices ($P < 0.05$)

Variable	F	*P*	Range	Pasture	Crop
2009 $F_{2,105}$					
Water Depth (cm)	6.99	0.001	26.5 (2.1)^{a}	26.3 (3.6)^{ab}	20.0 (2.2)^{b}
Water Temperature (°C)	2.02	0.138	27.5 (0.9)^{a}	25.5 (0.9)^{a}	25.6 (0.9)^{a}
Conductivity (µS/cm)	2.13	0.124	477.7 (61.8)^{a}	762.0 (141.8)^{a}	419.1 (82.7)^{a}
pH	0.45	0.636	7.56 (0.11)^{a}	7.87 (0.13)^{a}	7.75 (0.17)^{a}
Dissolved Oxygen (mg/L)	0.45	0.639	8.37 (0.69)^{a}	8.37 (0.87)^{a}	7.50 (0.73)^{a}
Plant Richness	10.15	< 0.001	4.7 (0.4)^{a}	3.1 (0.5)^{b}	2.6 (0.3)^{b}
Vegetation Complexity	7.81	0.001	1.9 (0.1)^{a}	1.4 (0.2)^{b}	1.2 (0.2)^{b}
Emergent Plant Cover (%)	4.73	0.011	11.6 (2.4)^{a}	4.4 (1.6)^{b}	2.9 (1.5)^{b}
Submergent Plant Cover (%)	1.71	0.187	5.7 (2.1)^{a}	3.5 (2.0)^{a}	5.3 (2.7)^{a}
Turbidity (NTU)	10.41	< 0.001	17.0 (3.7)^{b}	32.5 (11.0)^{ab}	75.0 (20.7)^{a}
Orthophosphate (mg/L)	3.54	0.033	1.50 (0.32)^{b}	2.71 (0.73)^{a}	2.55 (0.48)^{ab}
Ammonia Nitrogen (mg/L)	6.09	0.003	0.21 (0.07)^{b}	0.78 (0.32)^{a}	0.85 (0.21)^{a}
Nitrate Nitrogen (mg/L)	5.31	0.006	< 0.01 (< 0.01)^{b}	0.01 (0.01)^{ab}	0.04 (0.02)^{b}
2010 $F_{2,187}$					
Water Depth (cm)	1.38	0.254	27.3 (1.7)^{a}	32.3 (2.4)^{a}	24.6 (1.9)^{a}
Water Temperature (°C)	0.15	0.861	23.1 (0.7)^{a}	23.3 (0.7)^{a}	23.2 (0.8)^{a}
Conductivity (µS/cm)	0.18	0.831	279.9 (27.4)^{a}	370.2 (43.4)^{a}	297.4 (38.5)^{a}

Table 2 (continued)

pH	3.52	0.032	7.11 (0.11)^{b}	7.55 (0.10)^{ab}	7.46 (0.12)^{a}
Dissolved Oxygen (mg/L)	6.93	0.001	5.34 (0.44)^{b}	6.74 (0.48)^{a}	7.30 (0.52)^{a}
Plant Richness	5.80	0.004	3.9 (0.3)^{a}	3.6 (0.3)^{a}	2.5 (0.3)^{b}
Vegetation Complexity	5.77	0.004	1.2 (0.1)^{a}	1.3 (0.1)^{a}	0.7 (0.1)^{b}
Emergent Plant Cover (%)	6.74	0.002	16.1 (3.9)^{a}	7.3 (1.7)^{ab}	4.3 (4.1)^{b}
Submergent Plant Cover (%)	0.58	0.562	2.4 (1.1)^{a}	6.8 (2.5)^{a}	3.8 (1.9)^{a}
Turbidity (NTU)	7.57	0.001	21.4 (4.4)^{b}	14.9 (3.1)^{b}	51.4 (11.3)^{a}
Orthophosphate (mg/L)	12.53	< 0.001	1.05 (0.14)^{b}	0.83 (0.09)^{b}	2.01 (0.28)^{a}
Ammonia Nitrogen (mg/L)	9.90	< 0.001	0.24 (0.06)^{b}	0.18 (0.04)^{b}	0.82 (0.19)^{a}
Nitrate Nitrogen (mg/L)	6.22	0.002	< 0.01 (< 0.01)^{b}	< 0.01 (< 0.01)^{b}	0.11 (0.4)^{a}

Table 3 Mean (SE) taxa richness, Pielou's evenness (*J'*), and Shannon index (*H'*) for and F and *P*-values for the invertebrate community within different land-use practices in depressional wetlands in north central Oklahoma during 2009 and 2010. Different letters following means denote significant differences between land-use practices ($P < 0.05$)

Diversity Metric	F	*P*	Range	Pasture	Crop
2009	$_{2,105}$				
Richness	9.39	< 0.001	22.0 (1.5)^{a}	18.4 (2.2)^{b}	14.6 (1.5)^{b}
Evenness (*J'*)	0.15	0.864	0.51 (0.03)^{a}	0.51 (0.03)^{a}	0.52 (0.03)^{a}
Shannon (*H'*)	3.26	0.042	1.56 (0.09)^{a}	1.37 (0.09)^{a}	1.34 (0.09)^{a}
2010	$_{2,187}$				
Richness	14.85	< 0.001	24.8 (1.2)^{a}	22.0 (1.3)^{b}	16.5 (1.5)^{c}
Evenness (*J'*)	1.30	0.275	0.48 (0.02)^{a}	0.53 (0.02)^{a}	0.50 (0.02)^{a}
Shannon (*H'*)	2.80	0.063	1.55 (0.07)^{a}	1.60 (0.06)^{a}	1.39 (0.08)^{a}

Table 4 Mean (SE) of biomass (mg/m^{3}) and F and *P*-values of functional feeding groups within depressional wetlands surrounded by different land-use practices in north central Oklahoma during 2009 and 2010. Different letters following means denote significant differences between land-use practices ($P < 0.05$)

Functional Feeding Group	F	*P*	Range	Pasture	Crop
2009	$_{2,105}$				
Collector-Gatherer	2.10	0.128	9552.1 (1639.9)^{a}	8505.5 (2428.1)^{a}	4607.0 (1043.0)^{a}
Collector-Filterer	0.49	0.611	2465.8 (405.8)^{a}	2770.6 (754.4)^{a}	2258.0 (520.9)^{a}
Omnivore	2.42	0.094	12643.2 (1982.4)^{a}	12806.3 (3126.0)^{a}	8065.2 (1450.5)^{a}
Piercer-Herbivore	1.45	0.239	328.6 (153.6)^{a}	94.0 (61.0)^{a}	317.5 (142.0)^{a}
Scraper	1.87	0.159	5802.8 (2390.6)^{a}	2398.0 (1729.5)^{a}	1938.5 (1355.5)^{a}
Shredder	1.22	0.299	708.4 (252.4)^{a}	629.5 (311.2)^{a}	288.9 (156.9)^{a}
Herbivore	1.33	0.270	12463.1 (3255.4)^{a}	6876.3 (3164.1)^{a}	7303.7 (2886.4)^{a}
Predator	0.37	0.689	2700.8 (681.1)^{a}	3717.6 (1289.6)^{a}	3825.4 (1381.1)^{a}
2010	$_{2,187}$				
Collector-Filterer	3.95	0.021	2016.4 (335.2)^{a}	1840.5 (247.7)^{a}	1151.4 (206.8)^{b}
Collector-Gatherer	0.34	0.712	5092.8 (554.4)^{a}	5346.7 (660.5)^{a}	4840.9 (686.2)^{a}
Omnivore	0.93	0.398	7531.6 (815.4)^{a}	7613.2 (851.6)^{a}	6476.8 (834.5)^{a}
Piercer-Herbivore	2.00	0.139	240.4 (77.8)^{a}	77.1 (29.9)^{a}	104.6 (50.1)^{a}
Scraper	2.28	0.105	1230.3 (538.8)^{a}	1054.4 (495.2)^{a}	310.2 (180.0)^{a}
Shredder	9.94	< 0.001	293.5 (92.4)^{a}	292.9 (96.0)^{a}	34.0 (18.0)^{b}
Herbivore	3.52	0.032	4192.3 (1099.0)^{a}	3866.9 (1032.7)^{ab}	1537.5 (517.8)^{b}
Predator	1.27	0.283	2014.5 (428.5)^{a}	1662.1 (388.5)^{a}	2543.0 (699.8)^{a}

Table 5 Mean (SE) of biomass (mg/m^3) and F and P-values of nonmicrocrustacean, microcrustacean, and the most common taxa (i.e., taxa occurring in ≥ 10% of samples) within depressional wetlands surrounded by different land-use practices in north central Oklahoma during 2009. Different letters following means denote significant differences between land-use practices ($P < 0.05$)

Taxa	$F_{2,105}$	P	Range	Pasture	Crop
Total	0.46	0.634	27797.1 (4560.7)^{a}	22105.8 (5842.8)^{a}	22704.1 (5239.9)^{a}
Microcrustacean	0.78	0.462	2844.0 (395.4)^{a}	3563.3 (952.6)^{a}	3169.4 (746.7)^{a}
Hirudinea					
Mooreobdella	14.38	< 0.001	0.4 (1.1)^{b}	0.2 (0.6)^{b}	555.0 (407.7)^{a}
Oligocheata					
Naididae	7.44	0.001	145.7 (71.9)^{a}	61.4 (43.3)^{ab}	4.4 (4.1)^{b}
Gastropoda					
Helisoma	2.68	0.073	647.2 (516.7)^{a}	51.7 (82.7)^{a}	73.6 (89.3)^{a}
Physa	2.21	0.115	2745.2 (1047.6)^{a}	1864.1 (1301.4)^{a}	697.4 (528.8)^{a}
Copepoda	1.47	0.234	507.6 (82.1)^{a}	667.2 (245.3)^{a}	832.1 (212.7)^{a}
Cladocera	0.69	0.506	1370.4 (175.0)^{a}	1619.1 (410.0)^{a}	1236.0 (334.1)^{a}
Ostracoda	1.05	0.354	448.3 (162.1)^{a}	825.9 (341.6)^{a}	646.4 (268.2)^{a}
Ephemeroptera					
Callibaetis	9.19	< 0.001	136.5 (75.0)^{a}	32.0 (37.6)^{ab}	1.4 (2.1)^{b}
Odonata					
Anax	1.25	0.29	1.4 (1.5)^{a}	7.2 (12.1)^{a}	0.2 (0.6)^{a}
Enallagma	4.90	0.009	164.4 (78.8)^{a}	70.6 (54.5)^{ab}	27.3 (25.2)^{b}
Hemiptera					
Corixidae nymph	1.08	0.344	44.5 (29.4)^{a}	34.8 (26.7)^{a}	114.4 (68.3)^{a}

Table 5 (continued)

Notectidae nymph	1.23	0.296	20.2 (17.1)^{a}	34.6 (30.7)^{a}	5.4 (6.4)^{a}
Coleoptera					
Berosus adult	1.59	0.208	1.5 (2.1)^{a}	14.4 (18.5)^{a}	0.5 (1.1)^{a}
Berosus larvae	0.97	0.384	159.0 (88.5)^{a}	104.6 (76.0)^{a}	47.2 (36.4)^{a}
Curculionidae adult	0.30	0.745	0.4 (0.5)^{a}	0.3 (0.4)^{a}	0.1 (0.2)^{a}
Haliplus larvae	5.04	0.008	6.1 (4.8)^{a}	1.1 (1.6)^{ab}	< 0.1 (0.1)^{b}
Helophorus adult	2.15	0.121	4.0 (3.6)^{a}	1.1 (1.4)^{a}	0.2 (0.4)^{a}
Hygrotus adult	1.04	0.358	5.7 (7.5)^{a}	26.0 (36.3)^{a}	7.7 (7.4)^{a}
Liodessus adult	1.37	0.260	0.7 (0.8)^{a}	1.5 (2.1)^{a}	< 0.1 (0.2)^{a}
Paracymus adult	2.86	0.062	0.5 (0.6)^{a}	0.2 (0.3)^{ab}	< 0.1 (< 0.1)^{b}
Tropisternus larvae	0.02	0.981	1.0 (0.9)^{a}	1.0 (1.2)^{a}	0.9 (1.1)^{a}
Diptera					
Chironomidae	1.78	0.174	3370.9 (575.7)^{a}	3893.1 (1137.0)^{a}	1998.6 (649.7)^{a}
Arachnida					
Hydrachnidia	0.460	0.631	12.9 (9.1)^{a}	5.9 (6.2)^{a}	7.7 (7.4)^{a}

Table 6 Mean (SE) of biomass (mg/m^3) and F and *P*-values of nonmicrocrustacean, microcrustacean, and the most common taxa (i.e., taxa occurring in ≥ 10% of samples) within depressional wetlands surrounded by different land-use practices in north central Oklahoma during 2010. Different letters following means denote significant differences between land-use practices ($P < 0.05$)

Invertebrate Group	$F_{2,187}$	P	Range	Pasture	Crop
Nonmicrocrustacean	0.97	0.388	13625.7 (1884.8)^{a}	15533.3 (1899.6)^{a}	11772.3 (1841.0)^{a}
Microcrustacean	1.83	0.164	2757.4 (349.9)^{a}	2269.1 (261.9)^{a}	1987.0 (250.6)^{a}
Hirudinea					
Mooreobdella	4.59	0.011	2.6 (2.6)^{b}	1.8 (2.1)^{b}	52.9 (47.6)^{a}
Oligocheata					
Naididae	1.76	0.176	58.1 (24.5)^{a}	22.5 (11.3)^{a}	26.4 (14.4)^{a}
Tubificidae	0.23	0.793	327.5 (105.6)^{a}	268.5 (107.8)^{a}	227.4 (94.8)^{a}
Gastropoda					
Helisoma	3.02	0.051	222.1 (151.8)^{a}	135.1 (98.9)^{a}	16.5 (16.5)^{a}
Physa	3.00	0.052	252.9 (111.4)^{a}	170.5 (101.7)^{a}	72.1 (50.9)^{a}
Calanoida	5.88	0.003	89.3 (33.6)^{a}	5.8 (3.9)^{b}	19.0 (11.7)^{b}
Cyclopida	2.13	0.122	306.3 (63.1)^{a}	260.0 (52.4)^{a}	173.8 (47.6)^{a}
Cladocera	3.38	0.036	1050.5 (243.1)^{a}	788.3 (136.1)^{ab}	474.7 (121)^{b}
Ostracoda	4.84	0.009	465.5 (75.3)^{a}	501.2 (108.6)^{a}	209.5 (66.2)^{b}
Ephemeroptera					
Callibaetis	1.77	0.173	80.3 (34.4)^{a}	57.9 (28.8)^{a}	23.3 (14.7)^{a}
Odonata					
Libellulidae	1.72	0.182	4.6 (3.3)^{a}	5.2 (3.2)^{a}	0.9 (0.8)^{a}
Enallagma	0.36	0.696	46.2 (23.5)^{a}	89.7 (41.2)^{a}	76.4 (43.5)^{a}

Table 6 (continued)

Hemiptera					
Corixidae nymph	0.54	0.585	47.4 (22.5)^{a}	21.1 (11.7)^{a}	21.2 (14.1)^{a}
Notectidae nymph	1.25	0.288	16.6 (10.2)^{a}	5.7 (4.0)^{a}	3.6 (3.4)^{a}
Coleoptera					
Berosus larvae	1.19	0.307	14.1 (9.1)^{a}	13.1 (8.1)^{a}	3.8 (2.8)^{a}
Haliplus larvae	1.59	0.208	0.3 (0.3)^{a}	0.2 (0.2)^{a}	< 0.1 (< 0.1)^{a}
Tropisternus larvae	4.46	0.013	4.4 (2.7)^{a}	0.8 (0.7)^{ab}	0.2 (0.2)^{b}
Diptera					
Chironomidae	2.12	0.123	881.6 (244.4)^{a}	1553.9 (417.3)^{a}	618.2 (248.8)^{a}
Dasyhelea	0.99	0.375	0.6 (0.6)^{a}	0.4 (0.4)^{a}	0.1 (0.1)^{a}
Stratiomyidae	0.55	0.577	2.8 (1.9)^{a}	2.5 (1.6)^{a}	0.8 (1.2)^{a}
Tanypodinae	0.88	0.417	20.0 (11.9)^{a}	26.3 (14.7)^{a}	39.7 (19.5)^{a}
Arachnida					
Hydrachnidia	1.31	0.272	0.4 (0.4)^{a}	4.7 (3.0)^{a}	0.9 (0.7)^{a}
Nematoda	1.65	0.194	0.7 (0.7)^{a}	0.6 (0.6)^{a}	0.1 (0.1)^{a}

CHAPTER II

RESPONSE OF WETLAND INVERTEBRATE COMMUNITIES TO LOCAL AND LANDSCAPE FACTORS IN NORTH CENTRAL OKLAHOMA

Abstract Wetland invertebrates are crucial components of wetland systems and provide important food sources for many species of waterbirds. To better manage and conserve wetland habitat, it is imperative that we know the important factors which affect invertebrate communities at both local and landscape scales. My objective was to determine the effects of local and landscape factors on invertebrate diversity, invertebrate taxa composition, and the biomass of common invertebrate taxa. I sampled 58 depressional wetlands in north central Oklahoma during 2009 and 2010. Local factors were those which varied within the wetland or had an immediate effect on the wetland and included variables such as plant cover, water quality, and land-use adjacent to the wetland. Landscape factors were those which varied outside the wetland and included variables such land-use and wetland density within 1 km and 2 km buffers of the wetlands. The amount of variation in invertebrate data explained by environmental variables varied from 7% to 70%. I found that local factors explained more variation in invertebrate diversity than landscape variables. However, sampling date explained more variation than local or landscape variables for the Shannon index during both years and evenness during one year. Of the 46 taxa biomasses analyzed during either year, 24 taxa had the most variation explained with local factors, seven taxa had the most variation explained with landscape factors, while 15 taxa had the most variation explained by sampling date. More variation in taxa composition also coincided with changes in local factors. Dominant local variables overall were vegetation complexity, plant richness, plant cover, and water depth, and while no landscape factors were dominant overall, land-use practices, hydroperiod diversity, wetland density, and wetland area within the surrounding landscape were important for some invertebrate taxa as well as overall taxa composition. My results indicated that while variation in local variables may explain larger amounts of variation for more taxa, landscape variables are still important for some taxa and overall taxa composition. Because hydroperiod diversity, wetland area, and wetland density were important factors for overall invertebrate taxa composition as well as some invertebrate taxa biomasses, it may indicate that some invertebrate taxa do utilize wetland complexes as habitat patches. My results indicate that management and conservation efforts within wetlands should be providing high quality habitat within wetlands, but should also consider available wetland habitat in the surrounding landscape as well, supporting the current paradigm of wetland management for wildlife.

Keywords Depressional wetlands · Invertebrates · Landscape ecology · Oklahoma

Introduction

Invertebrates play an important role in influencing the function of wetland ecosystems. In particular, invertebrates provide an important link between primary production and secondary consumers, play an important role in nutrient cycling, and facilitate decomposition of organic matter by consuming and breaking down plant and animal tissues (Wissinger 1999). From a conservation and management perspective, invertebrates are of great importance to a wide variety of waterbirds. Specifically, invertebrates are an important component of many waterfowl and shorebird diets throughout their annual cycle (Skagen and Oman 1996, Davis and Smith 1998, Baldassarre and Bolen 2006). For example, invertebrates provide critical resources for migrant shorebirds to replenish depleted energy and nutrient reserves that allow them to continue migration and successfully reproduce. Because of these important roles that invertebrates play, it is imperative that we understand the factors that influence the diversity and biomass of wetland invertebrates. Invertebrates in wetlands are influenced by a variety of factors ranging from localized factors such as plant composition and water chemistry parameters to landscape factors such as proximity to other wetlands and the type of land-use practice surrounding a wetland (Voigts 1976, Euliss and Mushet 1999, Wissinger 1999, de Szalay and Resh 2000, Hall et al. 2004). Information about the role these factors play in influencing invertebrate communities would be useful in the development of conservation and management strategies for many waterbirds that rely on invertebrates as an important food source during different periods of their annual cycle.

Several local factors have been shown to influence the structure and composition of wetland invertebrate communities. An increase in plant cover and vegetation types can increase invertebrate abundance and diversity (Voigts 1976, Olson et al. 1995, de Szalay and Resh 2000, Hinojosa-Garro et al. 2010). However, some studies have found that increased plant cover and increased detritus build-up can negatively impact some taxonomic groups (de Szalay and Resh 2000, Christensen and Crumpton 2010). To a lesser extent, abiotic factors within the wetland have been shown to influence invertebrate communities. For example, increases in salinity have been shown to decrease invertebrate diversity and abundance and change invertebrate community composition (Euliss et al. 1999). Increased nitrogen in wetlands has been related to decreases in invertebrate taxa richness (Hentges and Stewart 2010), but it has also been shown to correspond with higher abundances of some taxonomic groups (Bazter et al. 2004). Additionally, the duration and seasonality of flooding (i.e., hydroperiod) may also affect the composition and structure of wetland invertebrate communities. Generally, longer hydroperiods increase invertebrate abundance and diversity (Wissinger 1999); however, invertebrate diversity can decrease in permanently flooded wetlands (Whiles and Goldowitz 2005).

In contrast to local factors, the influence of landscape factors on wetland invertebrate communities has not been as well studied. Landscape factors such as changes in land-use practices, proximity to other wetlands, type of wetlands, and amount of wetland area within the surrounding landscape could have profound effects on wetland invertebrate communities. Several studies have examined the influence of land-use practices on wetland invertebrates, but the results have been varied and equivocal. For example, cropland land-use surrounding wetlands can negatively impact invertebrate taxa richness (Euliss and Mushet 1999, Hall et al. 2004). However, it has also been found that taxa diversity can be positively correlated with the percentage of some crop types in the watershed (Hall et al. 2004). Other studies have found responses of single taxa to different land-use practices For example, increased Naididae biomass in farmed wetlands (Davis and Bidwell 2008), and decreased amphipod abundance with increased land-use intensity (Anteau et al. 2011). Another study found taxa composition of carabid beetles was affected by land-use intensity as well as the distance to cropland (Brose 2003), while yet another found no significant correlations between land-use and wetland invertebrate taxa composition(Tangen et al. 2003). A few studies have also assessed the impacts of other landscape factors on wetland invertebrates. One found that the size of the nearest wetland was positively correlated with taxa richness, but this relationship only occurred during a portion of the season (Hall et al. 2004). Another found that the density of wetlands and the proximity to other wetlands was correlated with changes in taxa composition (Brose 2003). Additionally, others have found differences in richness and taxa composition between different glacial landforms that contained different wetland types and densities. (Batzer et al. 2004) As indicated by these studies, landscape factors certainly may affect wetland invertebrate communities, but they may result in different effects depending on the taxa. Moreover, these factors likely interact to affect invertebrate diversity and taxa composition.

Because many factors affect wetland invertebrates at both local and landscape scales, some of these factors are likely interacting to impact individual taxa, diversity, and taxa composition. Depressional wetlands occur as naturally isolated patches in many landscapes, and therefore, invertebrate communities within this type of wetland are likely not only to be affected by local factors, but also landscape factors as suggested by island biogeography theory (MacArthur and Wilson 1967). In Oklahoma, depressional wetlands are geographically isolated (i.e., upland areas between wetlands), but likely function as wetland complexes, connected biologically and possibly hydrologically across the landscape (Henley and Harrison 2000). Furthermore, these wetlands occur throughout an agriculturally modified landscape with land-use practices ranging from relatively unmodified rangeland to annually tilled cropland. Therefore, local and landscape factors may play a role in influencing invertebrate communities in these wetlands. Information on wetland invertebrates in this region is scant (Cosyleõn 2003), and information from this study will assist conservation and management decisions for depressional wetlands in the region by indicating factors that impact invertebrate abundance and diversity, as well as provide management and conservation implications for waterfowl and shorebirds that utilize the region (Heitmeyer and Vohs 1984, Albanese 2011). Therefore, my objectives were to determine the effects of local and landscape factors on invertebrate diversity, invertebrate taxa biomass, and invertebrate taxa composition in depressional wetlands of north central Oklahoma.

Methods

Study Area

This study was located in Garfield, Grant, Kingfisher, and Major counties in north central Oklahoma and encompassed an area of 365 km^2 (Fig. 1). The region contains a diverse group of wetlands that include closed depressional, palustrine wetlands with temporary, seasonal, and semipermanent hydroperiods (Cowardin et al. 1979, Brinson 1993). These wetlands are located within the terraces of the Cimarron River and Salt Fork of the Arkansas River. Wetland size ranges from < 1 ha to 20 ha. Dominant hydrologic influences are rainfall and runoff, with some groundwater influences. Flooding of wetlands generally occurs from fall and winter precipitation, but summer storms may also contribute to flooding (Henley and Harrison 2000). Wetland vegetation is dominated by herbaceous emergent plants that include water knotweed (*Polygonum amphibium*, L.), chairmaker's bulrush [*Schoenoplectus americanus*, (Pers.) Volkart ex Schinz & R. Keller], and upright burhead [*Echinodorus berteroi* (Spreng.) Fassett] and submergent aquatic plants that include southern waternymph [*Najas guadalupensis*, (Spreng.) Magnus] and waterthread pondweed (*Potamogeton diversifolius*, Raf.). The upland soils surrounding depressions are dominated by Meno loamy fine sand, Lovedale fine sandy loam, and Nobscot fine sand, while soils within depressions are dominated by Carwile loam. Most depressional soils have clayey subsoil (NRCS 2008).

Land-use practices surrounding the wetlands are agricultural croplands and livestock grazing of native rangelands and improved pasturelands. Agricultural croplands are primarily winter wheat or rye, and are commonly grazed during the winter. Native rangelands consist of mixedgrass and tallgrass prairie that include sand bluestem (*Andropogon hallii* Hack.), little bluestem [*Scizachyrium scoparium*, (Michx.) Nash], switchgrass (*Panicum virgatum* L.), sideoats grama [*Bouteloua curtipendula* (Michx.) Torr.], and Chickasaw plum (*Prunus angustifolia* Marsh.),, while improved pasturelands consist primarily of Bermudagrass [*Cynodon dactylon* (L.) Pers.]. Both rangelands and pasturelands are commonly grazed throughout the year. Wetlands located within croplands and pasturelands are commonly tilled and replanted with crops or forage during dry years.

Average annual precipitation in the study area ranges from 74.7 to 85.7 cm, with most of the precipitation occurring from April through September (Oklahoma

Climatological Survey 2005). Average annual temperatures range from 15.0 to 15.6° C with average minimum temperatures during January ranging from -6.1 to -3.8° C and average maximum temperatures during July ranging from 34.8 to 35.7° C. The average growing season for the study area ranges from 201 to 208 days (Oklahoma Climatological Survey 2005).

Data Collection

I conducted the study on 58 wetlands. Wetlands were selected from a larger group of available wetlands to cover a gradient of hydroperiods that ranged from temporary to semipermanent within the three land-use practices (designated as crop, pasture, and range). Land-use was classified by the dominant landuse immediately surrounding each wetland. Hydroperiods were determined based on the dominance of certain plant communities (e.g., annuals and spikerushes [*Eleocharis* spp.] tend to dominate wetlands with shorter hydroperiods), changes in water levels from several site visits, and the changes in hydrological conditions based on several years of orthophotography. Thirty-seven wetlands were sampled during 2009 (11 pasture, 13 range, and 13 crop), while 40 wetlands were sampled during 2010 (13 pasture, 14 range, and 13 crop).

I collected invertebrates from April through July in 2009 and from March through July in 2010. Invertebrates were collected every three weeks, for a total of four sampling periods during 2009 and five sampling periods during 2010. Because some of the wetlands dried before the end of each season, I was unable to sample all wetlands during every sampling period (Table 1). Prior to collecting invertebrates, I divided each wetland into strata based on dominant plant communities. Invertebrate communities can be strongly influenced by different plant types and this ensured that multiple invertebrate communities were better represented from the wetland (Voigts 1976). Dominant strata consisted of areas dominated by either submergent or emergent vegetation or areas with no vegetation. Within each stratum, I sampled invertebrates at random points. For wetlands with one stratum, five points were sampled, while for wetlands with two strata, three points were sampled in each stratum. At each sample point, one 50 × 50 cm quadrat of clipped vegetation (DeCoster and Persoone 1970, Anderson and Smith 1996), two 5.2 cm diameter water column samples (Swanson 1978, Anderson and Smith 1996), and one 5.2 cm diameter benthic core sample were collected (Swanson 1983).

Following collection of invertebrates, each benthic core sample was placed in a sampling jar containing 75% ethanol, and clipped vegetation samples were bagged and refrigerated (4° C) for processing in the lab within 5 days of collection (Anderson and Smith 1998). The two water column samples from each site were combined and processed in the field by sieving through a 500 µm (No. 35) sieve, and all retained invertebrates were preserved in 75% ethanol. During the field processing of water column samples and benthic core samples, I recorded the volume of water that was sieved as well as the length of the core sample. I determined the volume of vegetation sampled by multiplying the area of the quadrat by the water depth at the sampling site. Benthic core samples were processed in the lab by washing through a 500 µm sieve and retained invertebrates were preserved in 75% ethanol. Epiphytic samples were also processed in the lab and collected invertebrates were preserved in 75% ethanol. All invertebrates were identified to genera, when possible, according to Merritt et al. (2008)

for insects and Smith (2001) for other invertebrates. All invertebrates were counted, dried at 65°C for 48 hrs, and weighed to determine dry biomass. Voucher specimens are stored at the Department of Natural Resource Ecology and Management at Oklahoma State University.

To assess the influence of local factors on invertebrates, I measured a variety of environmental variables in the wetland. At each sampling point, I recorded water depth, dissolved oxygen, water temperature, pH, conductivity, plant taxa richness, percent emergent and submergent plant cover, and vegetation complexity. I used a YSI multiparameter water quality meter (YSI Incorporated, Yellow Springs, Ohio) to determine dissolved oxygen, water temperature, pH, and conductivity. I calculated percent emergent and submergent cover by digitally photographing (Olympus 1030SW, 314 dpi) the 50 × 50 cm quadrat of collected vegetation. The digital photo was then viewed on a 39.1 cm monitor at full screen under a 1 cm transparent dot grid and percent cover was determined by calculating the percent of points that covered vegetation within the quadrat (de Szalay and Resh 2000). For vegetation complexity, I used an index ranging from high to low complexity to estimate structural complexity of plants. I rated each sample's vegetation complexity a 1, 2, or 3 based on the plants present within the quadrat. For example, southern waternymph, an aquatic submergent with diffuse branching and many leaves, was rated high complexity and assigned a 3, while chairmaker's bulrush, an emergent with no branching leaves, was rated low complexity and assigned a 1. Assessing structural complexity is important because it can affect invertebrate colonization as well as production (Hinojosa-Garro et al. 2010). To determine ammonia and nitrate-nitrite nitrogen, orthophosphate, and turbidity, I collected two-1 L water samples at each wetland during each sampling period. Nutrients were analyzed within 24 hrs using a Hach 850 Colorimeter and turbidity was assessed using a Hach Turbidimeter (Hach Company, Loveland, Colorado).

I also considered slope and soil texture as local scale factors because of the proximate effects on the wetland. I determined the maximum slope and soil texture surrounding each sample wetland from Natural Resources Conservation Service soil survey data (NRCS 2008). Similarly, land-use practices immediately surrounding the wetlands (within 15 m), wetland size, and wetland shape were also considered local factors. Wetland shape index was calculated in ArcMap with the add-in V-LATE 2.0 beta (Z_GIS, Centre for Geoinformatics, University of Salzburg, Austria). The shape index calculates the complexity of the wetland shape based on perimeter-area ratios. Wetland size and land-use practices were calculated in ArcMap. The description of all variables and variable abbreviations are presented in Table 2. The median and range of local scale variables are presented in Table 3.

To assess the effects of landscape level factors on invertebrates, I created a GIS using ArcGIS 10 (ESRI, Redlands, California) for the study area by delineating wetlands and land-use within 2 km of study wetlands. Wetlands within the surrounding landscape were delineated using Farm Service Agency National Agricultural Imagery Program Aerial photography at 1 m resolution for 2008 and 2010 combined with observations from ground surveys. Hydroperiods of other wetlands surrounding study wetlands were determined by similarity in aerial photography to wetlands with known hydroperiods. Land-use within 1 and 2 km of study wetlands was delineated from a combination of ground surveys and aerial photography.

Hydroperiod diversity was calculated in ArcMap with the add-in V-LATE 2.0 beta. Hydroperiod diversity was calculated using the Shannon index, which provides an index for the variety of wetland types within an area. Other landscape metrics calculated included area of each land-use practice (crop, pasture, and range), area of wetlands, number of wetlands, area of semipermanent wetlands (wetlands that contained surface water for the longest duration), and number of semipermanent wetlands within 1 and 2 km buffers surrounding study wetlands. To assess the influence of proximity of crops and wetlands on invertebrate communities, I calculated the distance to the nearest crop, nearest wetland, and nearest semipermanent wetland for each study wetland. The aforementioned metrics were all calculated in ArcMap 10. The median and range of landscape scale variables are presented for both years in Table 3.

Data Analyses

Data from each sampling device at each site were combined to provide an estimate of invertebrate biomass (g/m^3). Invertebrate taxa richness, Shannon index (H'), total nonmicrocrustacean biomass, total microcrustacean biomass, and biomass of the most common taxa (i.e., those occurring in $\geq$ 10% of total samples) were calculated as response variables to local and landscape factors. Prior to conducting analyses, both invertebrate data and independent variables were transformed (squareroot, fourthroot, natural log, and arcsin) to improve distribution of residuals during linear regression (Quinn and Keough 2003)

To determine the factors impacting invertebrate biomass and diversity, I used stepwise linear regression (Quinn and Keough 2003, Hall et al. 2004). Independent variables were included in the model if alpha values were ≤ 0.05. I used partial r^2 values to assess the strength of associations between invertebrate variables and individual environmental variables. Because seasonality can exert a significant influence on invertebrate communities (Wissinger 1999, Kratzer and Batzer 2007), I also included the variable sampling days, which was the number of days since the first sample was collected, as a possible independent variable in the model. I conducted the analyses in Minitab 16.2.1 (Minitab Inc., State College, Pennsylvania).

In addition to multiple linear regressions, I also used partial canonical correspondence analysis (pCCA) to determine the factors that are associated with changes in the composition of the invertebrate communities (ter Braak 1988, ter Braak and Verdonschot 1995, Lepš and Šmilauer 2003). I performed separate analyses for each year with all local and landscape variables included as possible environmental variables. Sampling days were included as a covariable to reduce the effects of seasonality. During 2009, 175 taxa were included in the analysis and during 2010, 212 taxa were include in the analysis. I used a stepwise utility to select variables that had an alpha of ≤ 0.05. Individual variables were tested with a Monte Carlo permutation test with 499 permutations. After all significant variables were added, the significance of the first axis and all axes combined were tested with a Monte Carlo permutation test with 999 permutations. In all analyses, rare taxa were down weighted and biomass values were squareroot transformed. When rare taxa are down weighted, each taxa is given a weight based on the number of occurrences in the data set. I conducted the analyses in Canoco 4.5 (Biometris, Wageningen, The Netherlands).

Results

Relationship Between Diversity Metrics and Local and Landscape Variables

During both years, over 70% of the variation in taxa richness was accounted for by environmental variables (Table 4). According to partial r^2 values, the majority of the variation in 2009 was positively attributed to vegetation complexity, while in 2010, a positive relationship between taxa richness and sampling date and plant richness explained nearly 50% of the variation with both variables explaining twice the variation attributed to vegetation complexity. Almost 50% of the variation in the Shannon index was explained during both years by environmental variables (Table 4). The strongest relationship with Shannon index during both years was sampling days and ammonia nitrogen. The Shannon index increased with sampling date, but decreased with increased ammonia nitrogen. Less than 25% of the variation in Pielou’s evenness index was explained by environmental variables during each year, with sampling days accounting for the most variation in each year.

Relationship Between Invertebrate Taxa and Local and Landscape Variables

In 2009, 10 taxa had a least 30% of the variation in biomass explained by environmental variables (Table 5). For the leech genera, *Mooreobdella*, nearly 34% of the variation was accounted for by environmental variables. *Mooreobdella* was positively associated with the amount of cropland occurring within 15 m of the wetlands and turbidity. While only 25% of the variation in Naididae biomass was explained, over 20% of the variation was attributed negatively to the amount of cropland within 1 km of the wetland. Over 30% of the variation was explained with environmental variables in both gastropod taxa. According to partial r^2 values, the majority of the variation in *Helisoma* was positively attributed to hydroperiod diversity within 2 km, while vegetation complexity positively attributed to the majority of the variation in *Physa* (Table 5).

Environmental variables explained 49.6% of the variation in *Callibaetis* biomass during 2009 (Table 5). Almost half of this variation was positively associated with sampling date, while smaller portions were positively associated with vegetation complexity and negatively associated to the amount of cropland within 15 m of the wetland. Of the odonates, 63.2% of the variation in *Enallagma* biomass was explained by environmental variables, with the majority of the variation positively attributed to vegetation complexity and hydroperiod.

The remaining taxa in which at least 30% of the biomass variation was explained during 2009 were coleopterans (Table 5). Increases in *Haliplus* larvae were most strongly related to vegetation complexity and wetland density within 1 km. Emergent plant cover and sampling date explained the majority of the variation and were both positively related to the biomass of *Helophorus* adults. The majority of the variation in *Hygrotus* adult biomass was negatively attributed to water depth and positively attributed to orthophosphate levels. According to partial r^2 values, sampling date accounted for the most variation in both *Liodessus* adult and *Paracymus* adult. Additional variation in *Liodessus* was negatively attributed to water depth and the amount of cropland within 1 km of the wetland and positively attributed to conductivity, while additional variation in *Paracymus* was negatively attributed to ammonia nitrogen and hydroperiod diversity within 2 km.

In 2010, 8 taxa had a least 30% of the variation in biomass explained by environmental variables (Table 6). Only one gastropod taxon, *Physa*, had more than 30% of the variation explained. Partial r^2 values indicated strong positive relationships with water depth as well as vegetation complexity for *Physa*. Almost 40% of the biomass variation in Cyclopidae, a microcrustacean, was explained with environmental variables. The majority of the variation was negatively associated to sampling date. *Callibaetis* had more than 30% of the variation in biomass explained by environmental variables. Similar to 2009, the strongest relationship for *Callibaetis* was with sampling date, with vegetation complexity still explaining a smaller portion of the variation.

Both odonate taxa had > 30% of the variation in biomass explained by environmental variables (Table 6). According to partial r^2 values, sampling date had the strongest relationship for both taxa, with both *Enallagma* and Libellulidae being positively related to sampling date. *Enallagma* biomass was also positively associated to vegetation complexity, while Libellulidae biomass was also positively associated to plant richness and submergent plant cover. Only two taxa within Diptera had > 30% of the variation explained by environmental variables. The majority of the variation in Stratiomyidae was positively attributed to sampling date and plant richness, and negatively attributed to dissolved oxygen. In contrast, the majority of the variation in Tanypodinae was solely attributed to sampling date. The remaining arachnid taxon, Hydrachnidia, had 42.6% of the biomass variation explained by environmental variables. The strongest relationship occurred with conductivity, while both vegetation complexity and slope had weaker but still important relationships (Table 6).

Relationship of Invertebrate Taxa Composition to Local and Landscape Variables

The pCCA analysis on 2009 data showed that 19 variables explained a significant amount of taxa composition (Table 7). The first four axes of the pCCA explained 18.6% of the variance in taxa data (Axis 1[F = 6.51, P = 0.001], all axes [F = 2.47, P = 0.001]). Strong correlations within the first four axes indicate relationships between environmental variables and taxa. The five variables crop within 1 km, hydroperiod diversity within 2 km, vegetation complexity, orthophosphate, and turbidity accounted for 48.5% of the explained variation. Axes one and two explained 11.5% of the taxa variation. The biplot of axes one and two shows four groupings of invertebrate taxa split among areas with (1): increased crop (*Mooreobdella*), (2): increased submergent plant cover, vegetation complexity, and hydroperiod diversity (*Physa*, *Enallagma*, *Anax*, Tanypodinae, and *Haliplus* larvae), (3): increased range (*Helisoma*, *Tropisternus* adult, and Naididae), and (4): increased turbidity and orthophosphate (*Hygrotus* adult, Tubificidae, and Ostracoda) (Fig. 2).

The pCCA analysis showed that 22 variables explained significant variation in taxa composition during 2010 (Table 8). The first four axes of the pCCA explained

12.1% of the taxa variation (Axis 1[F = 6.64, P = 0.001], all axes [F = 2.32, P = 0.001]). Again, correlations within the first four axes indicated relationships between environmental variables and taxa. Forty-six percent of the explained variation was accounted for by the variables pasture within 1 km, crop within 1 km, semipermanent wetland density within 2 km, water depth, pH, vegetation complexity, and emergent plant cover. Axes one and two explained 7.5% of the taxa variation. The biplot of axes one and two shows four grouping of invertebrate taxa (Fig. 3). These are split into areas with (1): increased crop (*Mooreobdella*), (2): increased range, pasture, emergent cover, and wetland density (Scarabaeidae larvae, *Tropisternus* larvae, Curculionidae larvae, Curculionidae adult, Stratiomyidae, *Helisoma*, and *Dasyhelea*), (3): increased vegetation complexity, semipermanent wetland density and area, water depth, and hydroperiod (*Physa*, *Tramea*, and Hydrachnidia), and (4) increased nearest semipermanent wetland (*Sigara*).

Discussion

The purpose of this study was to analyze the effects of local and landscape factors on wetland invertebrate communities. However, besides local and landscape factors, sampling date had a large impact on many invertebrate taxa as well as richness, diversity, and evenness. In fact, sampling date explained the majority of the variation for some taxa. In both years of the study, sampling date explained the majority of variation for more taxa than landscape variables. Except for some of the microcrustaceans, the relationship with sampling date was positive. Other studies have suggested invertebrate communities and invertebrate production changes throughout the year as habitat conditions and food sources change (Wissinger 1999, Kratzer and Batzer 2007). In my study, taxa richness, diversity, and evenness increased as the season progressed. These metrics were likely affected by immigration of taxa into the wetland and increases in populations of rarer taxa. Increased populations of less abundant taxa would have increased the likelihood of capturing those taxa. Biomass also increased as the season progressed, likely as invertebrates utilized increased habitat as well as food sources. A negative relationship occurred with sampling days for most microcrustaceans. This likely occurred as many microcrustaceans serve as food sources for larger predatory invertebrates and standing biomass likely decreased as predatory invertebrate populations increased (Wissinger 1999, Merritt et al. 2008).

Local Factors

Besides sampling date, the majority of variation in both taxa richness and the Shannon index was explained by local factors during both years. Furthermore, more taxa had the majority of variation explained by local factors than landscape factors. Although land-use practices seemingly influence wetlands in the region, local land-use practices (i.e., immediately surrounding wetland) only occurred in a few of the models and only explained ≥ 10% of the variation for two taxa (*Mooreobdella* and *Callibaetis*). While land-use practices can affect invertebrate communities, the effects are reported for or restricted to a few taxa or to diversity (Euliss and Mushet 1999, Hall et al. 2004, Davis and Bidwell 2008), and sometimes relationships are weak (Tangen et al. 2003). Additionally, it would be expected that land-use practices alter other habitat conditions within the wetland allowing variation to be more accurately described by other factors (e.g., vegetation, turbidity, and nutrients). Therefore, it is less surprising that vegetation factors (i.e., vegetation complexity, plant richness, plant cover) were present in more models than land-use. In fact, 15 taxa had ≥ 10% of the biomass variation explained by at least one vegetation factor.

Vegetation complexity explained a large portion of the variation for several taxa as well as invertebrate taxa richness. Vegetation complexity also explained the most variation in taxa composition during both years. Plant richness and plant cover (both emergent and submergent) were in several models as well, but tended to explain less variation than vegetation complexity. Vegetation complexity, vegetation type, and plant cover have all been identified to have large impacts on the invertebrate community (Voigts 1976, Olson et al. 1995, de Szalay and Resh 2000, Hinojosa-Garro et al. 2010). Despite the perceived importance of vegetation structure and type, and while it did explain more variation than any other variables in my study, it did not explain any significant variation in some of the invertebrate taxa. Possibly a more refined measurement and quantification of vegetation complexity would have resulted in more variation being explained, or perhaps some invertebrate taxa are not influenced by vegetation factors.

Beyond changes in vegetation metrics, water depth accounted for the majority of the variation attributed to local variables. With the exception of *Physa*, invertebrate biomass was negatively correlated with water depth for all taxa in which water depth occurred in the model. Increasing water depth may decrease productivity because of limited light in deeper areas, especially in wetlands with increased turbidity. Others have also indicated that water depth may be an important factor affecting invertebrate communities (Zimmer et al. 2000). It was suggested that shallower depths led to an increase in aquatic plants and increased primary production, allowing some invertebrate taxa to increase populations.

Hydroperiod only explained a large portion of the variation in one taxa biomass (*Enallagma*) during one year. Across all taxa, hydroperiod was both negatively and positively associated with biomass. Perhaps what is most surprising is that hydroperiod did not have larger effects on the invertebrate community. Hydroperiod is often thought as one of the most important factors affecting wetland invertebrate diversity as well as invertebrate abundance (Wissinger 1999, Whiles and Goldowitz 2005). However, it has been suggested by others, that while hydroperiod can be important in determining rare species, common taxa may be able to utilize a wide variety of hydroperiods because of their ability to rapidly colonize a wetland after flooding (Williams 1996, Batzer et al. 2004). Because the pCCA analysis did include less common taxa, this may explain why hydroperiod explained a portion of the variation during the second year. Perhaps the rarer taxa, although down weighted during the analysis, were partially explained by differences in hydroperiod. Other possibilities are, that while hydroperiod may have an impact on invertebrates, the impact of hydroperiod may be larger on other habitat characteristics (e.g., plant community and water chemistry) and therefore, any effect of hydroperiod on the invertebrate community is more accurately explained through variation in these variables (Wissinger 1999).

Turbidity only explained a larger portion of the variation in two taxa biomasses (*Berosus* larvae and *Mooreobdella*). It was also one of the more important variables in explaining taxa composition during one year. Changes in turbidity can affect system production as well as plant structure in lentic systems (Kalff 2002, Scheffer 2004). Increases in turbidity may have been attributed to land-use intensity and the resulting increase in sedimentation (Anteau et al. 2011), or other biological factors (e.g., increases in phytoplankton production). In my study, most increases in turbidity seemed to occur as a result of suspended sediment, and often decreased submergent plant cover.

The remaining local factor that explained a large portion of the variation was ammonia nitrogen. Other nutrients had minor effects on some taxa and orthophosphate was important in one year for taxa composition. Ammonia nitrogen had a strong negative relationship with the Shannon index during both years. Another study reported a negative relationship of total nitrogen with taxa richness (Hentges and Stewart 2010). While I also found taxa richness negatively correlated with ammonia nitrogen, ammonia nitrogen had a larger role in explaining the Shannon index. It is possible that high levels of ammonia were toxic to certain taxa (Hentges and Stewart 2010), however increased plant growth may have also lowered ammonia levels.

Landscape Factors

Only one taxon, *Helisoma*, had more variation explained by landscape factors than local factors during both years. However, some taxa had a majority of the variation explained by landscape factors during at least one year. Compared to local factors, there was not one single landscape factor that occurred in a majority of the models. Furthermore, while some of the variables explained a large portion of the variation (e.g., land-use composition, wetland area, wetland density, and hydroperiod diversity), they often explained varied amounts of variation, or did not explain significant variation during both years for the same taxa. While it would be expected that different taxa would be affected differently by some landscape factors because of the variation in life history strategies (Williams 1996, Williams 2006, Verberk et al. 2008), I would have expected more similarity in relationships between years. However, because landscape variables may influence local variables, it may be possible that local variation would always explain more of the variation in invertebrate data. It has been shown that landscape factors can also impact plant communities (Houlahan et al. 2006, Boughton et al. 2010, Tsai et al. 2012). Two studies have shown that land-use in the surrounding landscape can reduce plant diversity in wetlands (Houglahan et al. 2006, Tsai et al. 2012), while another showed that isolation of wetlands can alter plant diversity in wetlands (Boughton et al. 2010). Therefore, the variation between wetlands caused by landscape factors may have already been explained by local factors. Since vegetation and other habitat factors are important at a local scale, perhaps quantifying vegetation and as well as other factors at the landscape scale may lead to a greater importance of landscape factors.

While there was variation in landscape factors between study wetlands, many wetlands shared landscape characteristics because of the close proximity of some study wetlands. For example, some study wetlands were within 100 m of each other and would have shared many of the landscape characteristics within both the 1 km and 2 km buffer. This may have led to some wetlands with very different invertebrate communities being tested against very similar landscape factors. In this case, it would be expected that local factors would then explain more variation, as vegetation metrics or other local variables may have differed with the change in invertebrate populations. Furthermore, while some local factors varied during the season (e.g., vegetation complexity) along with invertebrate populations, landscape factors remained constant. It is possible that landscape factors may be more important during different times of the year depending on habitat conditions within the landscape (Hall et al. 2004). Sources of colonizers from semipermanent wetlands may be more important after inundation of seasonal and temporary wetlands, but become less important as the season progresses. Moreover, extreme drought may lead to a higher importance of permanent waterbodies on the landscape as sources for immigration after the drought ends.

Despite the lack of a significant effect of landscape factors on many invertebrate taxa, some of the taxa composition variation was attributed to landscape factors. One aspect, land-use practice within 1 km, was significant during both years indicating that land-use beyond what is adjacent to the wetland may have an effect on taxa composition in wetlands. Many adult invertebrates diapause in the vegetation surrounding wetlands and permanent grass may support higher numbers of adults leading to greater reproduction efforts in those wetlands (Wissinger 1999). In addition to land-use practices, hydroperiod diversity, wetland density, and wetland area explained additional variation within taxa composition. This indicates that these wetland systems may act as complexes of habitat rather than individual patches of habitat for some invertebrates. Research within this region of Oklahoma has shown that wetland use by shorebirds changes with different landscape conditions (Albansese 2011), and research in other regions supports other waterbirds utilize wetland complexes differently as well (Naugle et al. 2000). Since wetland dependent vertebrates utilize wetland complexes as habitat, it would also be expected that some wetland invertebrates utilize suitable habitat in a similar manner.

Weak Relationships with Local and Landscape Factors

Despite significant relationships among many of the factors, numerous variables explained ≤ 10% of the variance. Furthermore, of the 30 invertebrate taxa tested, only two of them had ≥ 50% of the variation explained in at least one year, and an additional six taxa had ≥ 40% of the variation explained in at least one year. In addition to explaining less than half the variation, the majority of the metrics were explained by different variables during each year. Additionally, while significant, only a small portion of taxa composition was explained. Therefore, the most significant finding of the study may be the lack of definite strong relationships between many invertebrates and local and landscape factors.

One possible explanation for explaining small amounts of variation is the lack of taxonomic resolution (Batzer et al. 2004). Several of the taxa analyzed were only identified to family or higher. Different species within these groups may be responding differently to the same factors. Indeed, the taxa that were identified to genera had the highest explained variation. Other reasons for large amounts of unexplained variation are that invertebrate populations in wetlands tend to be dominated by generalists, and populations may be changing stochastically (Tangen et al. 2003, Batzer et al. 2004, Hanson et al. 2009). Many wetland invertebrates are tolerable of a wide range of habitat characteristics and can often exploit wide ranges of habitat conditions (Williams 2006). The most common taxa in my study were present in a wide variety of habitats, and while the variation in some taxa was explained by measured variables, other taxa were seemingly unexplainable, indicating random variation or other causes of variation.

Additionally, there are possible variations in unmeasured variables that may be affecting invertebrate populations. In the past, it was assumed that many wetland food webs were based on macrophyte sources of carbon, however, research has suggested algae may play a more prominent role in prairie wetlands (Euliss et al. 1999, Wissinger 1999). Therefore, it is possible that unmeasured changes in algae and other microflora and microfauna may have impacted the invertebrate community. While the majority of the wetlands were grazed by cattle at some point in the year, I did not assess the effects of cattle grazing, as most perceived impacts would have also affected measured local factors. Cattle grazing has been shown to have some impacts on wetland invertebrate communities, primarily from its effects on vegetation (Steinman et al. 2003, Silver and Vimosi 2012). In my study, the impacts of cattle grazing were usually apparent in its effects on vegetation cover and structure as well as turbidity; however, soil compaction, which was unmeasured, may have occurred during other periods of the year and impacted invertebrate communities by preventing emergence of, or causing damage to diapausing invertebrates or by changing soil structure and the habitat of invertebrate burrowers.

Another factor that may have influenced my results is the natural variability associated with invertebrate populations. Invertebrate populations are known to differ significantly between years, and in many cases the same wetland can have strikingly different invertebrate communities from year to year (Zimmer et al. 2000). Still, I would expect similar factors to affect individual taxa as well as overall taxa composition, and that variation between years would be a result of changes in environmental conditions. Some taxa and taxa composition were explained by similar variables between years. However, when the variation is explained by a different variable each year, it may indicate a false relationship, as the variable only explains noise or random variation in the data. Furthermore, some landscape variables had both positive and negative relationships at the 1 and 2 km buffers. For example, I would not expect the amount of rangeland within 1 km to be positively related with taxa biomass, but then negatively related to the same taxa biomass at 2 km. It would be expected that the taxon would be related similarly at both landscape levels since the two landscape levels themselves were correlated. It is also possible that variables may be correlated with important, but unmeasured variables. For example, the amount of coarse and fine particulate organic matter, important food sources for some invertebrates, was likely correlated to plant cover. However, measuring the actual amount of particulate organic matter may have led to more explained variation for some invertebrate taxa.

Conclusion

These wetlands possess diverse invertebrate communities and are important regionally for many species of waterbirds (Heitmeyer and Vohs 1984, Albanese 2011). My results indicate that local and landscape factors can explain variation in invertebrate communities, but these relationships often varied between years and in some cases, only a small portion of the variation could be explained. However, my results identified some important factors that should be taken into consideration for management and conservation of wetlands depending on management objectives (e.g., vegetation, water depth, land-use, wetland composition within the landscape). While in many cases, local variables were more important in determining variation in the invertebrate community, landscape factors also need to be considered; especially when other research has indicated landscape factors can impact local factors within a wetland. Furthermore, because some landscape factors seemed to be important, further studies may be able to better pinpoint landscape relationships. Studies should be undertaken where wetland replications across a landscape or region each possess individually unique landscapes to better untangle the relationships of local and landscape factors. Moreover, taking into account the importance of seasonal variability, future studies may also be able to elucidate more information if sampling efforts are concentrated into a shorter period of time to reduce temporal variability.

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Figures

Fig. 1 Locations of study wetlands in Oklahoma, USA

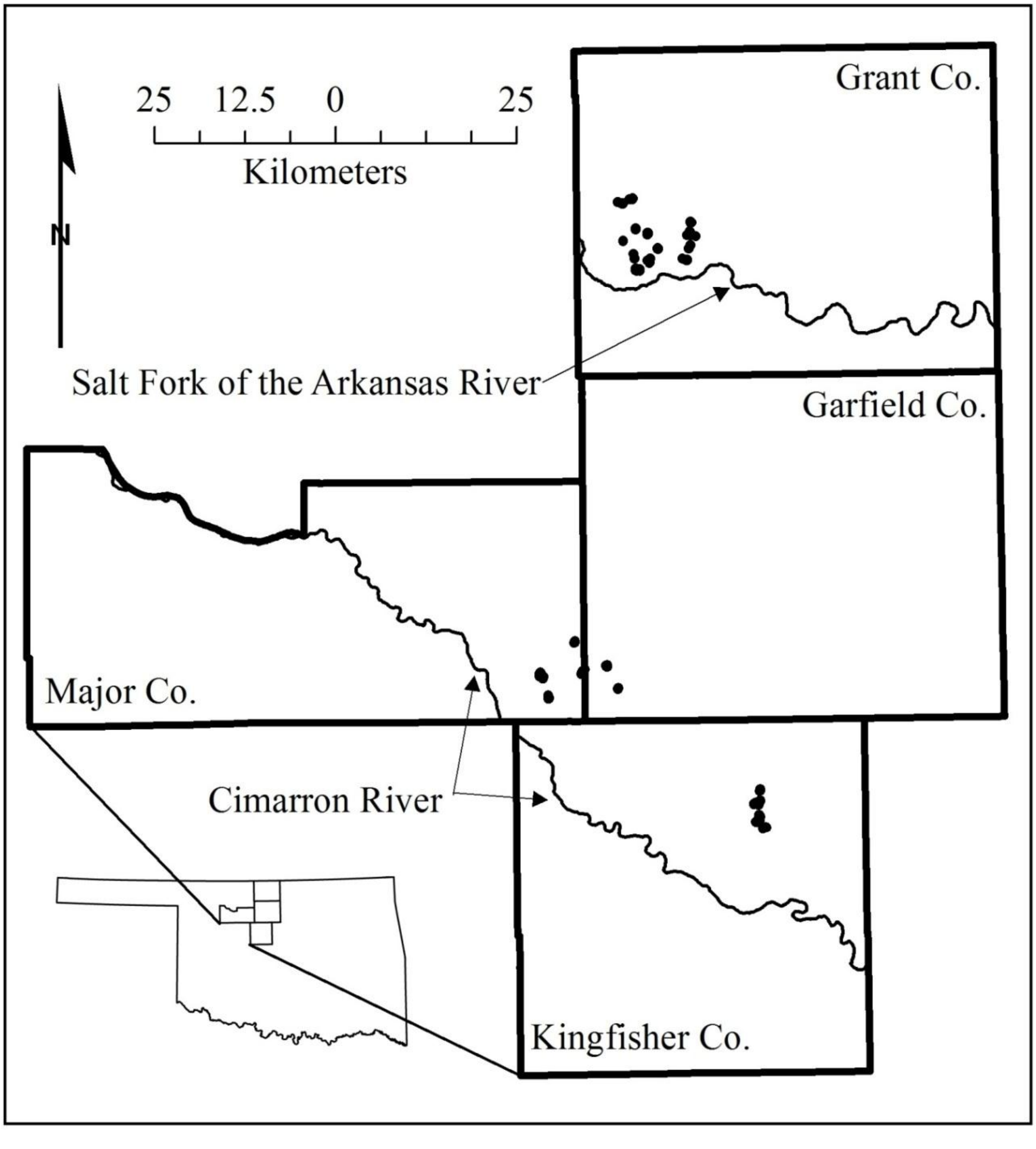


Fig. 2 Biplot of the first and second partial canonical correspondence analysis axes for invertebrate taxa during 2009 with significant environmental variables ($P < 0.05$). Taxa shown are those with a > 4% weight and >1% fit. For environmental variable abbreviations see Table 2

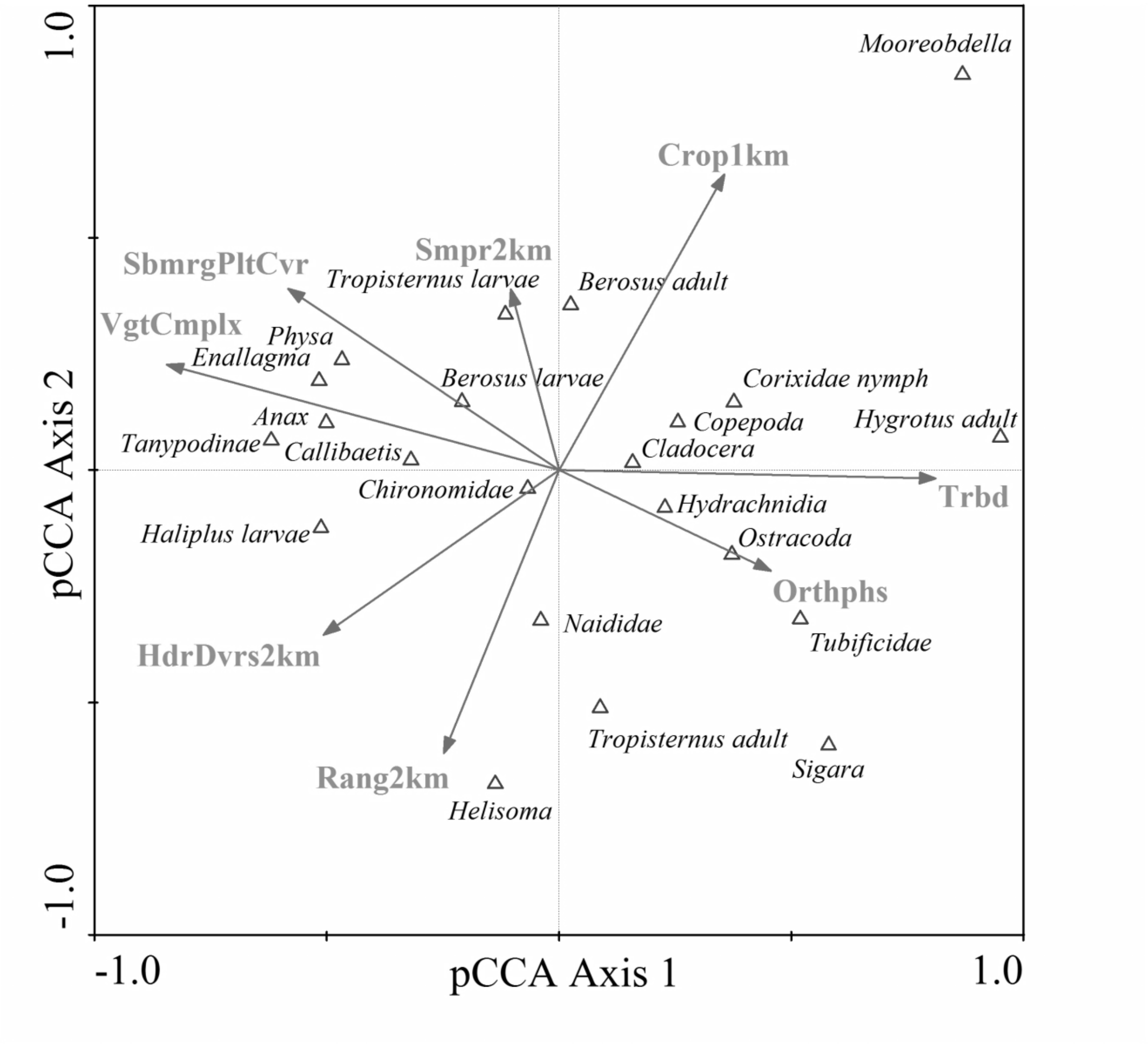
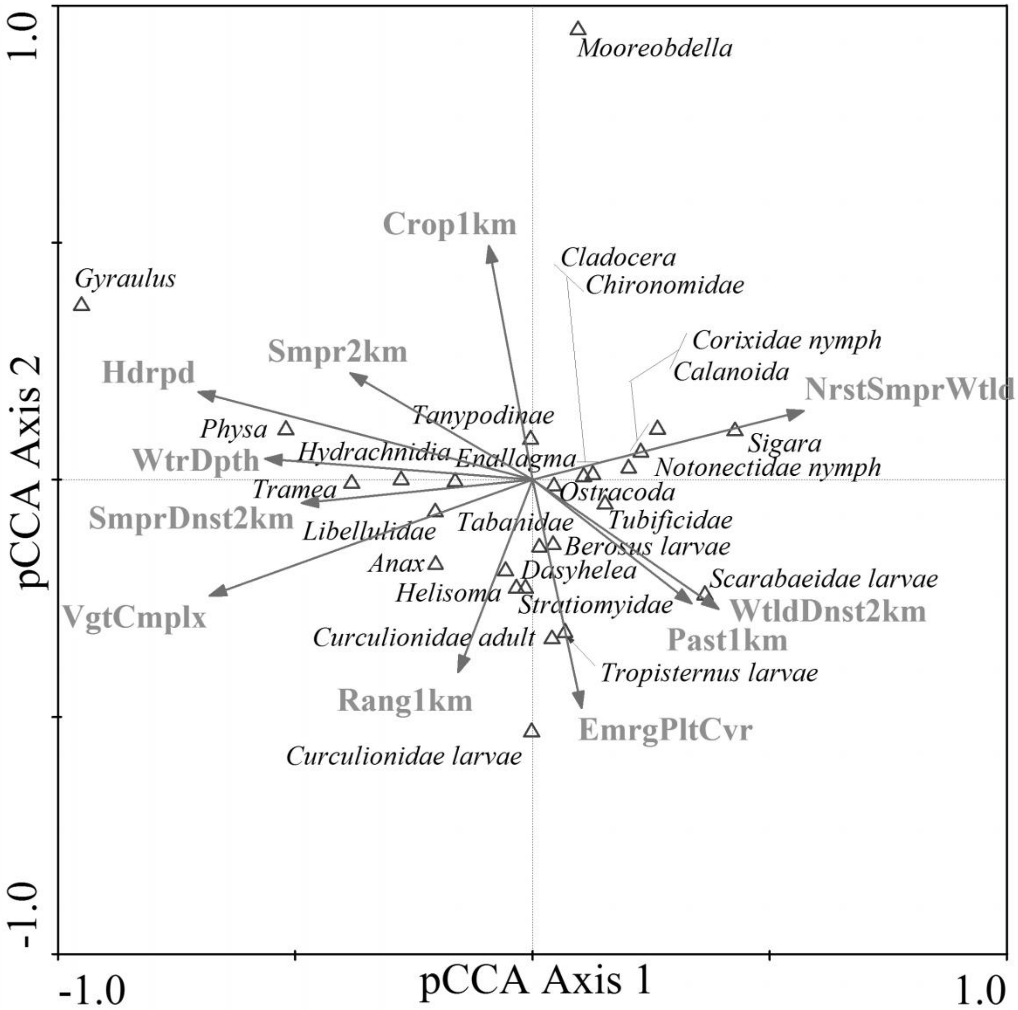


Fig. 3 Biplot of the first and second partial canonical correspondence analysis axes for invertebrate taxa during 2010 with significant environmental variables ($P < 0.05$). Taxa shown are those with a > 4% weight and >1% fit. For environmental variable abbreviations see Table 2



Tables

Table 1 Number of depressional wetlands in north central Oklahoma sampled within each land-use practice during sampling periods in 2009 and 2010

Sampling Period	Range	Pasture	Crop
2009			
1 (April 20 - May 8)	2	2	4
2 (May 12 - June 1)	13	11	13
3 (June 3 - June 22)	12	10	12
4 (June 23 - July 11)	11	6	10
2010			
1 (March 17 - April 19)	11	12	10
2 (April 21 - May 17)	12	12	13
3 (May 18 - June 8)	14	13	13
4 (June 10 - July 1)	13	13	13
5 (July 5 - July 26)	13	13	13

Table 2 Abbreviations and descriptions of environmental variables used in analyses of invertebrate communities

Abbreviation	Description
SmpDys	Number of days since first sampling day
Local	
Crop15m	Percent cropland land-use within 15 m
Past15m	Percent pastureland land-use within 15 m
Rang15m	Percent rangeland land-use within 15 m
SoilCrsns	Coarseness of predominant soils around wetland
Slope	Predominant slope around wetland
Hdrpd	Wetland hydroperiod
WtldSz	Wetland size
ShpIndx	Wetland shape index
WtrDpth	Average water depth from sampled points
WtrTmp	Average water temperature from sampled points
Cndct	Average conductivity from sampled points
pH	Average pH level from sampled
DslvdOxgn	Average dissolved oxygen level from sampled points
PltRchns	Plant taxa richness of wetland
VgtCmplx	Average vegetation complexity from sampled points
EmrgPltCvr	Average emergent cover from sampled points
SbmrgPltCvr	Average sumbergent cover from sampled points
Trbd	Average turbidity from sampled points
Orthphs	Orthophoshpate on sampling date
AmmnNtrgn	Ammonia nitrogen level on sampling date
NtrtNtrgn	Nitrate nitrite nitgrogen level on sampling date
Landscape	
Crop1km	Percent cropland land-use within 1 km
Past1km	Percent pastureland land-use within 1 km
Rang1km	Percent rangeland land-use within 1 km
Crop2km	Percent cropland land-use within 2 km
Past2km	Percent pastureland land-use within 2 km
Rang2km	Percent rangeland land-use within 2km
NrstCrop	Nearest cropland land-use to study wetland
NrstSmprWtld	Nearest semipermanent wetland to to study wetland
NrstWtld	Nearest wetland to study wetland
WtldDnst1km	Wetland density within 1 km
SmprDnst1km	Semipermanent wetland denisty within 1 km
WtldAr1km	Wetland area within 1 km
SmprAr1km	Semipermanent wetland area within 1 km

Table 2 (Continued)

Smpr1km	Percent wetland area semipermanent wetlands within 1 km
HdrDvrs1km	Shannon index of hydroperiods within 1 km
WtldDnst2km	Wetland density within 2 km
SmprDnst2km	Semipermanent wetland denisty within 2 km
WtldAr2km	Wetland area within 2 km
SmprAr2km	Semipermanent wetland area within 2 km
Smpr2km	Percent wetland area semipermanent wetlands within 2 km
HdrDvrs2km	Shannon index of hydroperiods within 2 km

Table 3 Median and range of local and landscape scale variables during 2009 and 2010 for 37 and 40, respectively, depressional wetlands in north central Oklahoma. See Table 2 for variable abbreviations

	2009		2010	
Variable	Median	Range	Median	Range
Local				
Crop15m	0.0	0.0-100.0	0.0	0.0-100.0
Past15m	0.0	0.0-100.0	0.0	0.0-100.0
Rang15m	0.0	0.0-100.0	0.0	0.0-100.0
SoilCrsns	3.2	1.3-5.0	3.0	1.4-5.0
Slope	5.0	1.0-15.0	3.0	1.0-15.0
Hdrpd	3.0	1.0-5.0	3.0	1.0-5.0
WtldSz	0.5	0.1-8.1	0.7	0.1-8.1
ShpIndx	1.42	1.05-2.82	1.34	1.05-2.82
WtrDpth	25.0	1.0-68.2	29.1	1.0-83.0
WtrTmp	26.93	16.61-41.71	24.59	6.06-33.50
Cndct	538.6	13.8-4428.0	257.0	21.7-4606.0
pH	7.69	5.53-10.04	7.25	5.25-9.85
DslvdOxgn	8.72	0.98-22.28	6.65	0.82-20.67
PltRchns	4	0-9	3	0-10
VgtCmplx	1.4	0-3	1.0	0-3
EmrgPltCvr	4.3	0.0-82.5	4.8	0.0-97.6
SbmrgPltCvr	0.5	0.0-91.0	0.0	0.0-100.0
Trbd	27.1	2.5-400.0	17.3	0.0-400.0
Orthphs	1.70	0.00-51.20	0.97	0.00-13.90
AmmnNtrgn	0.10	0.01-75.00	0.09	0.00-22.80
NtrtNtrgn	0.00	0.00-0.49	0.00	0.00-3.45
Landscape				
Crop1km	30.2	0.0-99.6	45.5	0.0-100.0
Past1km	22.1	0.0-71.4	28.6	0.0-56.5
Rang1km	35.5	0.0-100.0	20.1	0.0-100.0
Crop2km	35.3	2.7-76.2	46.3	5.4-82.8
Past2km	22.3	7.8-38.5	24.7	3.6-42.4
Rang2km	37.5	11.9-85.2	21.4	11.9-85.2
NrstCrop	119.0	0.0-1600.6	88.8	0.0-1396.3
NrstSmprWtld	211.6	19.2-1084.3	200.7	19.2-1218.9
NrstWtld	36.1	13.8-228.2	30.3	11.6-228.2
WtldDnst1km	11.1	4.6-28.2	13.6	5.4-31.1
SmprDnst1km	1.8	0.0-4.1	1.5	0.0-4.1
WtldAr1km	20.6	5.1-63.1	26.8	5.1-63.1

Table 3 (Continued)

SmprAr1km	2.6	0.0-25.3	4.2	0.0-25.6
Smpr1km	19.1	0.0-89.4	18.9	0.0-89.4
HdrDvrs1km	1.36	0.84-1.60	1.31	0.84-1.57
WtldDnst2km	8.8	4.9-21.0	11.6	4.1-18.2
SmprDnst2km	1.6	0.6-2.7	1.2	0.5-2.6
WtldAr2km	52.3	20.2-134.7	71.3	21.4-134.7
SmprAr2km	9.8	3.1-74.3	13.6	3.1-74.3
Smpr2km	23.5	5.6-45.7	18.1	3.3-45.7
HdrDvrs2km	1.28	0.98-1.50	1.28	0.82-1.50

Table 4 Results of multiple regression analyses between diversity metrics and environmental variables for invertebrate communities in depressional wetlands in north central Oklahoma during 2009 and 2010. See Table 2 for independent variable abbreviations

Dependent Variable		Independent Variable	R^2	Partial r^2	P-value
2009	Taxa Richness		0.732		< 0.001
		SmpDys		+0.112	
		Cndct		+0.049	
		VgtCmplx		+0.530	
		EmrgPltCvr		+0.029	
		AmmnNtrgn		-0.011	
	Shannon Index		0.478		< 0.001
		SmpDys		+0.229	
		SbmrgPltCvr		+0.092	
		AmmnNtrgn		-0.119	
		SmprAr2km		-0.037	
	Evenness		0.204		< 0.001
		SmpDys		+0.157	
		Cndct		-0.062	
2010	Taxa Richness		0.782		< 0.001
		SmpDys		+0.257	
		Rang15m		+0.078	
		WtrTmp		-0.001	
		PltRchns		+0.230	
		VgtCmplx		+0.120	
		EmrgPltCvr		+0.020	
		AmmnNtrgn		-0.042	
		SmprDnst1km		+0.035	
	Shannon Index		0.488		< 0.001
		SmpDys		+0.291	
		Hdrpd		+0.033	
		ErmgPltCvr		+0.023	
		AmmnNtrgn		-0.129	
		WtldDnst1km		-0.013	
	Evenness		0.224		< 0.001
		SmpDys		+0.082	
		Rang15m		-0.017	
		PltRchns		-0.043	
		AmmnNtrgn		-0.063	
		WtldAr1km		-0.019	

Table 5 Results of multiple regression analyses between invertebrate taxa biomass and environmental variables for invertebrate communities in depressional wetlands in north central Oklahoma during 2009. See Table 2 for independent variable abbreviations

Dependent Variable	Independent Variable	R^2	Partial r^2	P
Annelida				
Mooreobdella		0.337		< 0.001
	Crop15m		+0.215	
	VgtCmplx		+0.002	
	Trbd		+0.120	
Naididae		0.248		< 0.001
	Crop1km		-0.232	
	Hdrpd		+0.016	
Gastropoda				
Helisoma		0.350		< 0.001
	WtrDpth		-0.003	
	SbmrgPltCvr		+0.072	
	AmmnNtrgn		-0.074	
	HdrDvrs2km		+0.201	
Physa		0.524		< 0.001
	VgtCmplx		+0.493	
	SbmrgPltCvr		+0.030	
Crustacea				
Cladocera		0.271		< 0.001
	SmpDys		-0.165	
	WtrDpth		-0.048	
	WtldDnst2km		+0.058	
Copepoda		0.158		< 0.001
	SmpDys		-0.099	
	WtrDpth		-0.059	
Ostracoda		0.295		< 0.001
	WtrDpth		-0.165	
	Past2km		+0.088	
	WtldAr2km		+0.041	
Ephemeroptera				
Callibaetis		0.496		< 0.001
	SmpDys		+0.196	
	Crop15m		-0.108	
	VgtCmplx		+0.137	
	Orthphs		-0.024	
	SmprDnst2km		-0.032	

Table 5 (Continued)

Odonata				
Anax		0.213		< 0.001
	SmpDys		+0.029	
	PltRchns		+0.137	
	Past1km		+0.047	
Enallagma		0.632		< 0.001
	SmpDys		+0.088	
	Hrdpd		+0.226	
	WtrDpth		-0.021	
	VgtCmplx		+0.297	
Hemiptera				
Corixidae nymph		0.113		0.002
	SmpDys		-0.065	
	Hdrpd		-0.048	
Notonectidae nymph		0.295		< 0.001
	SmpDys		+0.217	
	HdrDvrs2km		+0.078	
Coleoptera				
Berosus adult		0.199		< 0.001
	SmpDys		+0.102	
	Orthphs		-0.059	
	WtldDnst2km		+0.039	
Berosus larvae		0.264		< 0.001
	WtrTmp		+0.065	
	DslvdOxgn		+0.038	
	Trbd		-0.160	
Curculiondae adult		0.253		< 0.001
	WtrDpth		-0.045	
	EmrgPltCvr		+0.208	
Haliplus larvae		0.456		< 0.001
	Hdrpd		+0.028	
	WtrDpth		-0.008	
	VgtCmplx		+0.221	
	EmrgPltCvr		+0.028	
	NrstWtld		+0.001	
	WtldDnst1km		+0.171	
Helophorus adult		0.443		< 0.001
	SmpDys		+0.179	
	EmrgPltCvr		+0.193	
	SbmrgPltCvr		+0.036	

Table 5 (Continued)

	SmprAr2km		-0.035	
Hygrotus adult		0.364		< 0.001
	SoilCrsns		+0.001	
	WtrDpth		-0.183	
	Orthphs		+0.112	
Liodessus adult		0.457		< 0.001
	SmpDys		+0.157	
	WtrDpth		-0.109	
	Cndct		+0.062	
	Crop1km		-0.063	
	Rang2km		-0.039	
	SmprAr1km		-0.027	
Paracymus adult		0.388		< 0.001
	SmpDys		+0.142	
	Cndct		+0.036	
	AmmnNtrgn		-0.084	
	NrstWtld		+0.042	
	HdrDvrs2km		-0.083	
Tropisternus adult		0.093		0.002
	SbmrgPltCvr		+0.093	
Diptera				
Chironomidae		0.134		< 0.001
	SoilCrsns		+0.082	
	Crop1km		-0.052	
Arachnida				
Hydracnidia		0.263		< 0.001
	SmpDys		+0.163	
	NrstSmprWtld		-0.071	
	HdrDvrs1km		-0.029	

Table 6 Results of multiple regression analyses between invertebrate taxa biomass and environmental variables for invertebrate communities in depressional wetlands in north central Oklahoma during 2010. See Table 2 for independent variable abbreviations

Dependent Variable	Independent Variable	R^2	Partial r^2	P
Nematoda		0.271		< 0.001
	SmpDys		+0.023	
	WtrDpth		-0.077	
	EmrgPltCvr		+0.034	
	Past1km		-0.020	
	Past2km		+0.093	
	HdrDvrs1km		+0.025	
Annelida				
Mooreobdela		0.137		< 0.001
	Slope		+0.008	
	VgtCmplx		-0.030	
	Crop1km		+0.099	
Naididae		0.259		< 0.001
	SmpDys		+0.119	
	pH		-0.001	
	VgtCmplx		+0.108	
	SmprDnst1km		+0.030	
Tubificidae		0.283		< 0.001
	SmpDys		+0.051	
	WtrDpth		-0.012	
	Cndct		-0.002	
	Past1km		+0.055	
	SmprAr1km		+0.112	
	HdrDvrs2km		+0.048	
Gastropoda				
Helisoma		0.185		< 0.001
	SmpDys		+0.094	
	SoilCrsns		+0.001	
	Slope		-0.041	
	NrstCrop		+0.050	
Physa		0.464		< 0.001
	WtrDpth		+0.178	
	VgtCmplx		+0.159	
	Past2km		-0.086	
	HdrDvrs1km		+0.041	
Crustacea				

Table 6 (Continued)

Calanoida		0.076		< 0.001
	SbmrgPltCvr		-0.076	
Cladocera		0.277		< 0.001
	SmpDys		-0.043	
	WtrDpth		-0.036	
	VgtCmplx		+0.077	
	NtrtNtrgn		-0.039	
	WtldDnst1km		+0.046	
	Smpr2km		+0.036	
Cyclopidae		0.396		< 0.001
	SmpDys		-0.228	
	Rang15m		+0.028	
	WtrDpth		-0.016	
	VgtCmplx		+0.033	
	Rang1km		+0.019	
	Rank2km		-0.054	
	WtldAr1km		+0.019	
Ostracoda		0.207		< 0.001
	SmpDys		+0.126	
	Crop15m		-0.028	
	WtrDpth		-0.005	
	VgtCmplx		+0.048	
Ephemeroptera				
Callibaetis		0.351		< 0.001
	SmpDys		+0.230	
	VgtCmplx		+0.085	
	Rang2km		+0.036	
Odonata				
Enallagma		0.527		< 0.001
	SmpDys		+0.229	
	DslvdOxgn		+0.034	
	VgtCmplx		+0.199	
	EmrgPltCvr		+0.022	
	SbmrgPltCvr		+0.042	
Libellulidae		0.478		< 0.001
	SmpDys		+0.245	
	PltRchns		+0.101	
	SbmrgPltCvr		+0.113	
	Rang1km		+0.018	
Hemiptera				

Table 6 (Continued)

Corixidae nymph		0.222		< 0.001
	pH		+0.008	
	SbmrgPltCvr		-0.049	
	NtrtNtrgn		-0.013	
	Crop1km		+0.005	
	WltdDnst1km		+0.112	
	HdrDvrs2km		+0.048	
Notonectidae nymph		0.287		< 0.001
	SmpDys		+0.168	
	WtrDpth		-0.011	
	SbmrgPltCvr		+0.022	
	WtldDnst2km		+0.086	
Coleoptera				
Berosus larvae		0.260		< 0.001
	SmpDys		+0.097	
	Hdrpd		-0.002	
	DslvdOxgn		+0.025	
	Trbd		-0.137	
Haliplus larvae		0.271		< 0.001
	SmpDys		+0.050	
	Past15m		+0.002	
	VgtCmplx		+0.111	
Tropisternus larvae		0.182		< 0.001
	WtldSz		-0.029	
	pH		+0.016	
	EmrgPltCvr		+0.136	
Diptera				
Chironomidae		0.249		< 0.001
	VgtCmplx		+0.021	
	Past1km		+0.156	
	Past2km		-0.043	
	WtldDnst1km		+0.029	
Dasyhelea		0.218		< 0.001
	EmrgPltCvr		+0.053	
	SbmrgPltCvr		+0.165	
Stratiomyidae		0.319		< 0.001
	SmpDys		+0.086	
	WtldSz		+0.003	
	DslvdOxgn		-0.059	
	PltRchns		+0.078	

Table 6 (Continued)

	NrstWtld		+0.049	
	SmprDnst1km		+0.044	
Tanypodinae		0.399		< 0.001
	SmpDys		+0.282	
	Hdrpd		+0.052	
	SbmrgPltCvr		+0.032	
	Crop1km		-0.003	
	Crop 2km		+0.030	
Arachnida				
Hydrachnidia		0.426		< 0.001
	Slope		+0.090	
	Cndct		+0.175	
	VgtCmplx		+0.102	
	EmrgPltCvr		-0.018	
	Crop1km		-0.028	
	Crop2km		+0.014	

Table 7 Cumulative percentage variance of species data and species-environment relation, and correlation of significant environmental variables ($P < 0.05$) within the first four axes of the partial canonical correspondence analysis (pCCA) during 2009. See Table 2 for environmental variable abbreviations

	pCCA Axis			
	1	2	3	4
Taxa-environment correlations	0.855	0.829	0.763	0.741
Cumulative percentage variance				
of taxa data	6.9	11.5	15.2	18.6
of taxa-environment relation	22.2	37.0	49.1	60.0
Correlation Coefficients				
Crop1km	0.306	0.530	0.092	-0.061
Past1km	-0.182	-0.184	0.102	0.133
Rang1km	-0.189	-0.393	-0.136	-0.011
Crop2km	0.204	0.566	0.063	-0.069
Past2km	0.028	-0.200	0.010	0.055
Rang2km	-0.212	-0.505	-0.066	0.053
NrstWtld	-0.031	0.199	-0.096	0.129
HdrDvrs1km	-0.364	0.171	-0.106	-0.076
SmprDnst2km	-0.322	0.061	0.025	-0.169
HdrDvrs2km	-0.436	-0.294	0.030	-0.346
Smpr2km	-0.088	0.322	0.030	-0.040
Crop15m	0.264	0.425	0.180	-0.232
Past15m	-0.104	-0.029	0.052	0.210
Rang15m	-0.162	-0.385	-0.222	0.035
VgtCmplx	-0.722	0.189	-0.102	-0.027
EmrgPltCvr	-0.167	-0.158	-0.268	0.227
SbmrgPltCvr	-0.507	0.329	0.083	-0.074
Trbd	0.711	-0.015	0.088	-0.167
Orthphs	0.393	-0.181	-0.353	-0.087

Table 8 Cumulative percentage variance of species data and species-environment relation, and correlation of significant environmental variables ($P < 0.05$) within the first four axes of the partial canonical correspondence analysis (pCCA) during 2010. See Table 2 for environmental variable abbreviations

	pCCA Axis			
	1	2	3	4
Taxa-environment correlations	0.853	0.717	0.713	0.726
Cumulative percentage variance				
of taxa data	3.9	7.5	10.0	12.1
of taxa-environment relation	17.0	32.8	43.7	53.1
Correlation Coefficients				
Crop1km	-0.078	0.354	0.144	0.006
Past1km	0.287	-0.187	-0.196	0.246
Rang1km	-0.133	-0.291	-0.022	-0.205
Past2km	0.314	-0.151	-0.194	0.063
NrstSmprWtld	0.489	0.104	-0.016	-0.079
NrstWtld	0.178	-0.134	-0.030	-0.064
SmprDnst1km	-0.362	-0.116	-0.134	0.111
SmprAr1km	-0.351	-0.019	0.109	0.152
Smpr1km	-0.439	0.082	0.038	0.101
WtldDnst2km	0.335	-0.195	-0.126	0.083
SmprDnst2km	-0.416	-0.036	0.099	-0.012
WtldAr2km	-0.023	-0.059	-0.061	0.090
SmprAr2km	-0.317	0.133	0.008	0.068
Smpr2km	-0.329	0.162	0.033	0.046
Crop15m	0.029	0.323	0.011	0.006
SoilCrsns	-0.366	0.093	-0.108	0.044
Hdrpd	-0.602	0.133	-0.109	0.105
WtrDpth	-0.491	0.033	-0.135	0.119
WtrTmp	-0.190	-0.039	0.119	-0.057
pH	-0.421	0.083	-0.016	-0.123
VgtCmplx	-0.592	-0.179	-0.265	-0.021
EmrgPltCvr	0.092	-0.358	0.108	-0.411

CHAPTER III

ASSESSMENT OF TWO METHODS FOR SAMPLING INVERTEBRATES IN SHALLOW VEGETATED WETLANDS

Abstract Invertebrates are often used as indicators of wetland health and habitat quality for waterbirds. However, collecting accurate data to assess invertebrate populations may be hampered by the sampling technique used. I compared the accuracy of two commonly used invertebrate sampling methods, the aquatic D-frame net method and the vegetation quadrat, water column, and benthic core method (QCC method), in vegetated depressional wetlands of north central Oklahoma. I assessed the differences in the sampling methods by comparing diversity and densities and biomasses of total invertebrates, functional feeding groups, and the most common taxa collected. In general, the QCC method was more effective at collecting invertebrates than the D-frame net method. The QCC method resulted in higher densities and biomasses for total invertebrates, seven of the eight functional feeding groups and nearly half of the 49 taxa collected than the D-frame net method, while the D-frame net method resulted in higher taxa richness and diversity as well as higher densities and biomasses for five taxa than the QCC method. Therefore, the QCC method should provide more accurate abundance estimates, while the D-net should provide more accurate diversity estimates. However, because invertebrate communities and wetland characteristics may vary regionally, sampling methods should be pre-tested to provide accurate answers to research goals.

Keywords Benthic core · D-net · Invertebrate · Quadrat · Sampling methods · Water column · Wetland

Introduction

Aquatic invertebrates are critical components of wetlands because they influence nutrient cycling and organic matter decomposition and provide an important linkage between primary production and consumers (Wissinger 1999). In fact, invertebrates are crucial food items for breeding (Baldassarre and Bolen 2006) and wintering waterfowl (Anderson and Smith 1998), and migrating shorebirds (Skagen and Oman 1996). Therefore, invertebrates are often used as indicators for wetland health and habitat productivity for waterbirds. However, sampling methods must accurately assess invertebrate populations to answer the questions desired by researchers. Additionally, methods used must be able to efficiently and accurately sample the diversity of habitats found within a study area. Currently, there are several methods available to sample invertebrate populations within vegetated wetlands (Batzer et al. 2001, Merritt et al. 2008), and many comparisons between methods have been published (Murkin et al. 1983, Downing and Cyr 1985, Cheal et al. 1993, Anderson and Smith 1996, Brinkman and Duffy 1996, Hyvönen and Nummi 2000, Meyer et al. 2011). However, none have assessed the difference between aquatic D-frame (hereafter, D-net) type nets and a combination method (the vegetation quadrat, benthic core, and water column method [hereafter, QCC]).

The QCC method combines several sampling methods (vegetation quadrat [DeCoster and Persoone 1970], water column [Swanson 1978], and benthic core [Swanson 1983]) that are used separately to collect epiphytic, nektonic, and benthic invertebrates. This method has been used in the past by Anderson and Smith (2000) and Kostecke et al. (2005) to obtain accurate quantitative results in a variety of wetland habitats. The D-net has been used by many studies and is still a very common method used for estimating aquatic invertebrate abundances and diversity (e.g., Voigts 1976, Olson et al. 1995, Hall et al. 2004, Davis and Bidwell 2008, Anteau et al. 2011, Silver and Vamosi 2012). D-nets are also commonly used for wetland monitoring by state agencies (Genet 2012). The D-net is often used because of its simplicity and ease of use. This method has also been used in conjunction with other methods (e.g., benthic corer [Hall et al. 2004, Davis and Bidwell 2008]) to allow for better spatial coverage of the wetland.

Although D-nets are widely used for aquatic invertebrate studies, they are often criticized as being less quantitative than quadrat or column methods (Meyer et al. 2011). Additionally, the effectiveness of D-nets may be affected by vegetation (Anteau et al. 2011, Meyer et al. 2011). Submergent vegetation can block the net opening, while robust emergent vegetation can impede movement of the net. Furthermore, because of differences in substrate (i.e., soft vs. firm or muck vs. soil), it is not always known if, or how much of the benthic community is actually being sampled. Nonetheless, the QCC method may have drawbacks as well. Large active swimmers (e.g., some adult Coleoptera) may be under-sampled because of their size and their ability to evade capture. Additionally, because of the patchy nature of invertebrate populations, some taxa may be missed by the relatively small size of water column and benthic core samples. Because each of these sampling techniques may produce biased results, information is needed on the accuracy of these methods when researchers choose a sampling method. Furthermore, because no information on the effectiveness of the QCC method compared to other methods, except box samplers, has been published (Anderson and Smith 1996), there is a need for the QCC method to be tested against the very common D-net sampling method. Therefore, the objective of my study was to evaluate the QCC and D-net sampling methods by comparing diversity, density, and biomass of aquatic invertebrates inhabiting depressional wetlands in north central Oklahoma.

Methods

Study Area

This study was located in Grant and Major counties in north central Oklahoma (Figure 1). This region contains a diverse group of wetlands that include closed depressional palustrine wetlands with temporary, seasonal, and semipermanent hydroperiods (Cowardin et al. 1979, Brinson 1993). These wetlands are located within the terraces of the Cimarron River and Salt Fork of the Arkansas River. Wetland size ranges from < 1 ha to 20 ha. Dominant hydrologic influences are rainfall and runoff, with some groundwater influences. Flooding of wetlands generally occurs from fall and winter precipitation but summer storms may also contribute to flooding (Henley and Harrison 2000).

Wetland vegetation is dominated by herbaceous emergent plants that include water knotweed (*Polygonum amphibium*, L.), chairmaker's bulrush [*Schoenoplectus americanus*, (Pers.) Volkart ex Schinz & R. Keller], and upright burhead [*Echinodorus berteroi* (Spreng.) Fassett] and submergent aquatic plants that include southern waternymph [*Najas guadalupensis*, (Spreng.) Magnus] and waterthread pondweed (*Potamogeton diversifolius*, Raf.). Soils within depressions are dominated by Carwile loam. Most depressional soils have clayey subsoil (NRCS 2008).

Average annual precipitation in the study area ranges from 74.7 to 85.7 cm, with most of the precipitation occurring from April through September (Oklahoma Climatological Survey 2005). Average annual temperatures range from 15.0 to 15.6° C with average minimum temperatures during January ranging from -6.1 to -3.8° C and average maximum temperatures during July ranging from 34.8 to 35.7° C. The average growing season for the study area ranges from 201 to 208 days (Oklahoma Climatological Survey 2005).

Data Collection

Invertebrate sampling occurred from June through July in 2009 on nine wetlands. I collected invertebrates from wetlands every three weeks, during two sampling periods.

Because two wetlands dried before the end of the study I only sampled those wetlands once.

Prior to collecting invertebrates, each wetland was divided into strata based on dominant vegetation communities. Dominant strata consisted of areas dominated by either submergent or emergent vegetation or areas devoid of vegetation. Because wetland invertebrates are affected by vegetative characteristics (e.g, different plants support different invertebrates), stratified sampling ensures different habitat types were sampled (Wissinger 1999). Within each stratum, I located random points to sample invertebrates. For wetlands with one stratum, five points were sampled, while for wetlands with two strata, three points were sampled in each stratum. At each sample point, the QCC approach was employed by collecting one 50 × 50 cm quadrat of clipped vegetation sample (DeCoster and PerSoone 1970, Anderson and Smith 1996), two 5.2 cm diameter water column samples (Swanson 1978), and one 5.2 cm diameter by 10 cm depth benthic core sample (Swanson 1983). Immediately adjacent to each QCC sample, a 50 × 50 cm area was swept once using a 500 µm mesh D-net. The same size area was sampled with the D-net as the QCC quadrat to reduce variability caused by sampling a larger area. Within each quadrat, I recorded the water depth for determination of volume sampled.

Following collection of invertebrates, each benthic core sample was initially placed in a sampling jar containing 75% ethanol, and clipped vegetation samples were bagged and refrigerated at 4° C for later processing in the lab (within 5 days of collection)(Anderson and Smith 1998). The two water column samples from each site were combined and processed in the field by sieving through a 500 µm (No. 35) sieve, and all retained invertebrates were preserved in 75% ethanol. During the processing of water column samples, I recorded the volume of water that was sieved. D-net samples were placed in sample jars in the field and preserved with 75% ethanol. Benthic core samples were processed in lab by washing through a 500 µm sieve and retained invertebrates were preserved in 75% ethanol. Epiphytic samples and D-net samples were also sorted in lab and collected invertebrates were preserved in 75% ethanol. All invertebrates were identified to genera, when possible, according to Merritt et al. (2008) for insects and Smith (2001) for other invertebrates. All invertebrates were counted, dried at 65°C to a constant mass, and weighed to determine dry biomass. Voucher specimens are stored at the Department of Natural Resource Ecology and Management at Oklahoma State University.

Data Analyses

Data from each QCC sample were combined to create one sample. Both QCC samples and D-net samples were converted to volumetric measures for density (number of individuals/m^3) and biomass (g/m^3). Invertebrate taxa richness, Shannon index (*H’*), Pielou’s evenness (*J’*), total nonmicrocrustacean density and biomass, total microcrustacean density and biomass, density and biomass of functional feeding groups (FFGs), and density and biomass of the most common taxa (i.e., those occurring in ≥ 10% of total samples) were calculated as response variables. FFGs were collector filterer, collector gatherer, scraper, shredder, piercer herbivore, predator, omnivore, and herbivore. I assigned taxa to FFGs based on Smith (2001), Merritt et al. (2008), and Thorp and Covich (2010). I compared both density and biomass because the sampling methods may be biased towards larger or smaller individuals within taxa and FFGs. Additionally, larval and adult stages within Coleoptera were considered separate taxa because of differences in FFGs and motility (Merritt et al. 2008). Because I was only able to identify nymphs within Hemiptera to the family level, I also considered them separate taxa.

Response variables were tested for differences between sampling methods (i.e., QCC vs. D-net) using paired t-tests. Prior to analysis, I transformed density and biomass data to meet assumptions for parametric analyses (Quinn and Keough 2003). I performed all analyses in Minitab 16.2.1 (Minitab Inc., State College, Pennsylvania).

Results

Overall, I identified 98 taxa from QCC samples and 86 taxa from D-net samples. Thirty-one taxa were unique to QCC samples, while 19 taxa were unique to D-net samples. Within the QCC samples, overall density was dominated by Ostracoda (37.3%), Chironomidae (14.3%), *Physa* (11.6%), *Callibaetis* (10.2%), and Cladocera (8.4%), while overall biomass was dominated by *Physa* (39.7%), *Helisoma* (31. 3%), and Chironomidae (4.5%). The overall density in D-net samples was dominated by Ostracoda (36.4%), *Physa* (15.1%), *Callibaetis* (12.4%), Cladocera (7.6%), and *Enallagma* (5.7%), with overall biomass being dominated by *Physa* (50.8%), and *Helisoma* (24.5%).

Taxa richness ($t_{1,74} = 6.29$, $P < 0.001$) and Shannon index were higher using the D-net, but evenness did not differ between sampling methods (Table 1). The densities and biomasses of seven of the eight FFGs differed between sampling methods. Greater densities and biomasses of collector filterers, collector gatherers, omnivores, scrapers, shredders, herbivores, and predators were collected using the QCC sampling method than the D-net sampling method (Tables 2 and 3).

Greater densities and biomasses of nonmicrocrustacean and microcrustacean were collected with the QCC sampling method than the D-net sampling method (Tables 4 and 5). Higher densities and biomasses were collected with the QCC method than the D-net method for the annelid taxa Naididae, *Helobdella*, and *Mooreobdella*. Three of the gastropod genera (*Gyraulus, Helisoma*, and *Physa*) had higher densities and biomasses using the QCC method than the D-net method. Individual crustacean taxa differed between sampling methods with Copepoda, Cladocera, and Ostracoda having higher densities and biomasses in QCC samples.

The mayfly genus, *Callibaetis* was collected in higher densities and biomasses with the QCC sampler (Tables 4 and 5). For Odonata genera, *Enallagma* densities and biomasses were higher in QCC samples than in D-net samples, while *Lestes* densities and biomasses were higher in D-net samples than in QCC samples. Within Hemiptera, *Buenoa* and *Sigara* densities and biomasses were higher using the D-net sampling method than the QCC sampling method. Corixidae nymph biomass was higher using the D-net sampling method than using the QCC sampling method (Table 5), but Corixidae nymph densities did not differ between sampling methods (Table 4).

Of the Coleoptera genera, *Berosus* larvae, *Enochrus* adults, *Haliplus* larvae, and *Helophorus* adults were collected in greater densities and biomasses using the QCC method, but *Laccophilus* adults were collected in greater densities and biomasses using the D-net (Tables 4 and 5). For Diptera, Chironomidae and Stratiomyidae densities and biomasses were higher using the QCC method than using the D-net method.

Discussion

The purpose of both the D-net and QCC method is to provide an accurate estimation of invertebrate abundance and diversity. Both methods have been proven to be accurate when compared against other methods (Cheal et al. 1993, Anderson and Smith 1996). However, because collection of higher densities and biomasses can be judged as an indicator of increased accuracy (Downing and Cyr 1985), my results suggest that the QCC method is more accurate for many taxa that may be encountered in vegetated depressional wetlands. Higher densities and biomasses were collected with the QCC method for nonmicrocrustacean, microcrustacean, all of the FFGs except piercer herbivore, and in 22 of the 49 taxa. In contrast, higher densities and biomasses were collected with the D-net sampler for only four taxa and in one taxa, only biomass was collected higher with the D-net. Stovepipe corer samplers, a similar method to the QCC method, have also been reported to collect higher densities and biomasses than a D-net sampler (Meyer et al. 2011).

For small taxa, such as microcrustaceans, the D-net significantly underestimated density and biomass compared to the QCC sampler. Density and biomass values from the D-net were as much as 200 times lower than from the QCC. This difference between the QCC and the D-net may be due to dense vegetation impeding water flow through the net which likely reduced the number of microcrustaceans collected. When sampling in dense vegetation, the opening of the D-net would often become blocked by vegetation, which likely prevented vegetation, litter, invertebrates, and water from consistently entering the net. Additionally, the mesh of the net would sometimes become clogged with litter and other debris, preventing water flow through the net. In a study examining invertebrate communities of seasonal wetland in north central Minnesota, it was reported that leaf litter and detritus may have clogged net openings as well (Batzer et al. 2004). Both of these issues could result in water being pushed away in front of the net instead of flowing through net. Many samples collected with the D-net also contained many damaged soft bodied invertebrates. Some larger invertebrates were still recognizable, but some small invertebrates (e.g., Cladocera) may have become unrecognizable, and therefore, were not counted and as a result lost from the sample.

Another group of invertebrates that were collected in higher densities and biomasses using the QCC method were invertebrates that are typically associated with being attached to vegetation (e.g., Gastropoda, *Enallagma*, *Helophorus* adult, and *Haliplus* larvae) (Merritt et al. 2008). While the magnitude in differences were not as large as the microcrustaceans (only about 10 ×), the differences in density and biomass for these taxa indicate that D-net sampling may be less accurate at sampling these taxa relative to the QCC method. These taxa were likely collected in lower amounts with the D-net simply because they were not dislodged from the vegetation, or as mentioned above, vegetation blocked the opening and prevented capture. The QCC sampler may more efficiently capture these taxa because the actual vegetation and invertebrates are collected simultaneously, instead of relying on the sweeping action to dislodge invertebrates from the vegetation.

Two other invertebrate taxa that were collected at much higher densities and biomasses with the QCC method were Naididae and Chironomidae. While some individuals within these taxa occur on vegetation and within the water column, these taxa are also found within the sediment. Therefore, it is not surprising that the D-net collected fewer of these taxa, as the D-net is not well designed to sample the benthos. Another study also reported that D-net samplers missed benthic organisms when compared against benthic core samplers (Cheal et al. 1993). Although the sediments are often stirred up during sampling and some benthic invertebrates are captured, the exact amount of sediment sampled is often unknown. Furthermore, depending on the type of sediment (i.e., firm vs. soft, muck vs. soil) found at a site, different amounts of the benthos may be sampled. Therefore, some studies have utilized benthic corers in addition to D-nets to account for benthic taxa (Hall et al. 2004, Davis and Bidwell 2008). However, this can make analyses more complex and results ambiguous. Separate analyses are often done for each sampling method and different sampling methods may produce different results for the same taxa (Davis and Bidwell 2008). Combining the data from benthic with epiphytic and nektonic data within the QCC method provides one consistent density and/or biomass and reduces the number of data analyses needed.

For a few taxa, the D-net appeared to be more effective than the QCC method. The taxa that were collected in higher densities and biomasses were larger sized (e.g., *Lestes*) or better swimmers (e.g., *Buenoa* and *Sigara*). These taxa may have evaded capture, or were missed with the smaller diameter water column sampler. Others have also suggested that rare and large taxa were missed by small diameter benthic core samplers when compared to D-net samplers (Cheal et al. 1993). However, I observed no differences between sampling methods for several other taxa, including large-sized taxa (e.g., *Anax* and *Tramea*) and those that are excellent swimmers (e.g., Belastomatidae, Notonectidae, *Berosus* adult, and *Hygrotus* adult). My data suggests that the size of the individuals within a taxonomic group may also affect sampling accuracy. For, example, the biomass, but not density, of corixid nymphs collected was higher with the D-net sampler than the QCC sampler. It was possible that the QCC method sampled similar densities to the D-net, but of smaller individuals such that the biomass collected was lower. Studies of terrestrial invertebrates have indicated that sampler type may also affect the sizes collected. For example, it has been found that vacuum samplers collected small invertebrates more accurately, while a sweep net collected larger invertebrates more accurately (Doxon et al. 2011).

Both richness and the Shannon index differed between sampling methods with the QCC sampler providing lower estimates for both measures than the D-net. This may be a result of the small diameter size of the water column sampler and benthic core sampler, which may miss larger and less abundant taxa, or rare taxa (Cheal et al. 1993, Meyer et al. 2011). In contrast, the D-net would have a greater chance of collecting these invertebrates simply because of the larger area sampled. Similarly, another study has reported that D-nets consistently produced higher richness than small diameter corers (Cheal et al. 1993). However, another study found that both richness and the Shannon index did not differ between D-net and stovepipe core samplers (Meyer et al. 2011). In their study the diameter of the core sampler was almost 10 times larger than my water column and benthic core samplers which may explain why they were able to obtain similar richness and Shannon index values between the D-net and stovepipe corer sampler.

While I did not assess the time or effort involved in using either sampling method, differences in time and cost should be considered when determining sampling methods. The QCC method involves collecting more samples so more time and effort will likely be required with the QCC method. Processing time for either method usually ranged between 30 min to 4 hrs and was directly related to the type of habitat sampled. Areas with dense submergent vegetation tended to require more time and effort for the QCC method, while areas with fine debris often required more time and effort with the D-net as the amount of debris collected with the D-net typically was greater than the amount collected with the water column or benthic core sampler. Another study did assess the differences in time and effort involved between a stovepipe corer sampler and a D-net, but found the average time for processing was similar between methods (Meyer et al. 2011).

Conclusion

My results indicated that QCC sampling collected higher densities and biomasses of a greater number of taxonomic groups than D-net sampling. If a study is largely interested in estimates of invertebrate abundance, QCC sampling is likely a better option than using a D-net, especially if dense vegetation is expected to be encountered. Additionally, if researchers are interested in estimating biomass production for waterbirds, it is essential to utilize a method that efficiently samples the multiple habitat types and substrates used by foraging waterbirds (Skagen and Oman 1996, Baldassarre and Bolen 2006). However, if a study is purely interested in measuring invertebrate diversity, the D-net may be a suitable option. Perhaps the best option is to pre-test the accuracy of sampling methods before a study is undertaken to determine if the method will allow research objectives to be met. Furthermore, pre-testing should be conducted within a variety of wetland habitat types to ensure accurate results across all habitat types encountered during a study.

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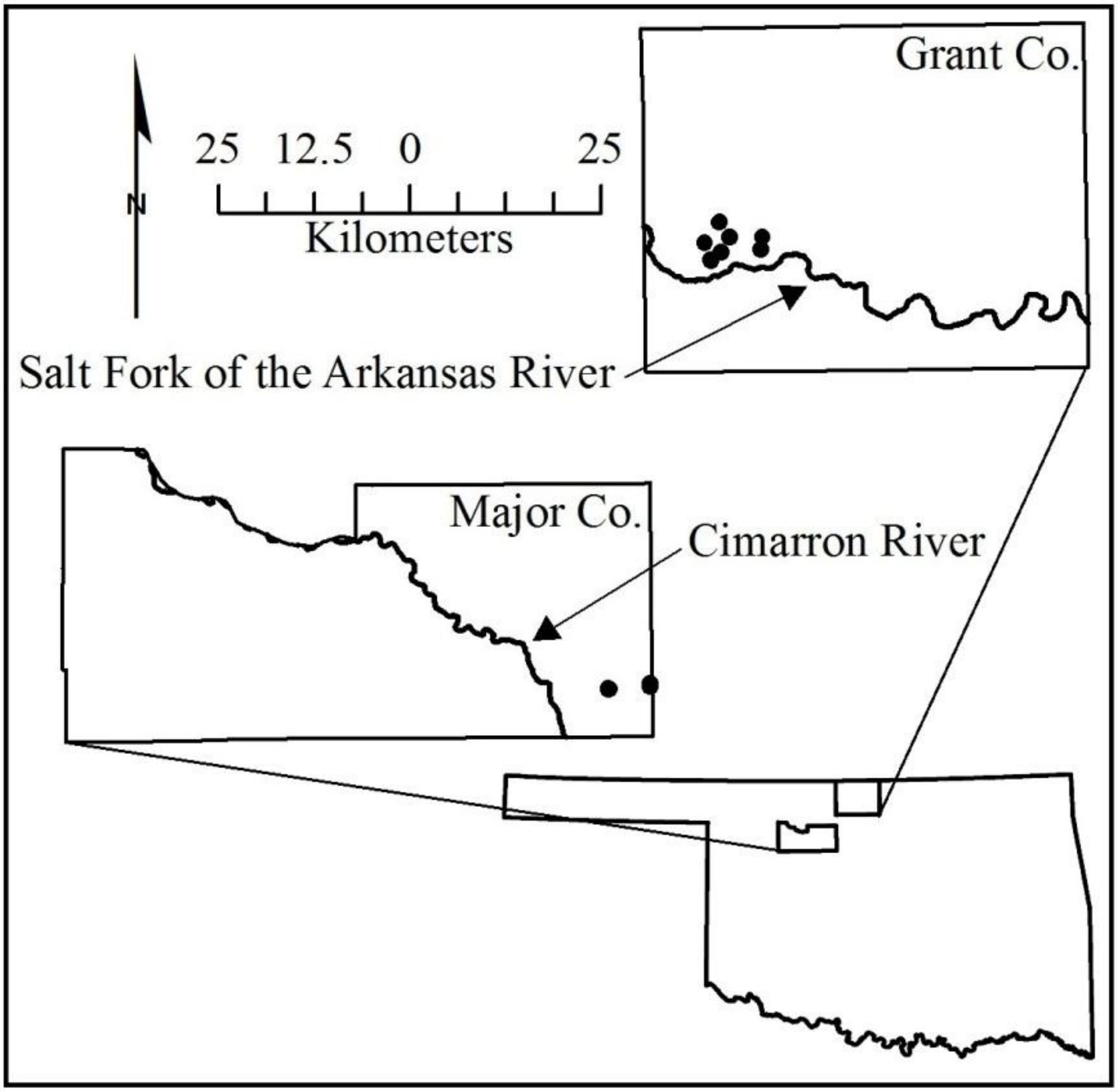
Figures

Fig. 1 Locations of sampled wetlands in Oklahoma, USA



Tables

Table 1 Mean and standard error (SE) of taxa richness, Pielou's evenness (*J'*), and Shannon index (*H'*) and t and *P* values for quadrat, water column, benthic core (QCC) and aquatic D-net sampling methods of invertebrates in depressional wetlands of north-central Oklahoma. Means with an asterisk are significantly different at the 0.05 level

	QCC		D-Net			
Variable	Mean	SE	Mean	SE	$t^{1,74}$	P
Richness	12.9*	0.9	16.6	1	6.29	< 0.001
H'	1.3*	0.6	1.6	0.6	4.49	< 0.001
J'	0.6	< 0.1	0.6	< 0.1	1.14	0.257

Table 2 Mean and standard error of functional feeding group (FFG) density (no./m^3) and and t and *P* values for quadrat, water column, benthic core (QCC) and aquatic D-net sampling methods of invertebrates in depressional wetlands of north-central Oklahoma. Means with an asterisk are significantly different at the 0.05 level

	QCC		D-Net			
FFG	Mean	SE	Mean	SE	$t^{1,74}$	P
Collector Filterer	6462.4*	2224.6	343.5	137.0	7.18	< 0.001
Collector Gatherer	10756.6*	2761.6	622.0	160.1	11.91	< 0.001
Omnivore	29612.2*	6356.8	1635.1	403.7	13.39	< 0.001
Piercer Herbivore	0.8	0.8	4.4	2.2	1.85	0.069
Scraper	4198.3*	1226.7	474.0	131.1	10.02	< 0.001
Shredder	328.7*	124.9	36.9	12.3	5.73	< 0.001
Herbivore	6024.3*	1549.0	732.3	160.4	9.41	< 0.001
Predator	5449.8*	1027.3	850.3	124.9	9.59	< 0.001

Table 3 Mean and standard error of functional feeding group (FFG) biomass (mg/m^3) and t and *P* values for quadrat, water column, benthic core (QCC) and aquatic D-net sampling methods of invertebrates in depressional wetlands of north-central Oklahoma. Means with an asterisk are significantly different at the 0.05 level

	QCC		D-Net			
FFG	Mean	SE	Mean	SE	$t^{1,74}$	*P*
Collector Filterer	730.2*	220.8	26.7	9.7	8.07	< 0.001
Collector Gatherer	3709.4*	972.8	219.8	62.8	11.25	< 0.001
Omnivore	6637.6*	1329.9	358.9	86.0	13.96	< 0.001
Piercer Herbivore	1.2	1.4	8.5	4.5	1.96	0.053
Scraper	12598.6*	4392.7	1925.7	611.9	7.23	< 0.001
Shredder	314.6*	128.1	29.0	10.0	5.54	< 0.001
Herbivore	16781.9*	5204.6	2464.9	679.9	7.66	< 0.001
Predator	4691.1*	1108.0	831.1	132.5	6.38	< 0.001

Table 4 Mean and standard error of nonmicrocrustacean, microcrustacean, and most common taxa (occurring in ≥ 10 % of the samples) density (no./m^3) and t and *P* values for quadrat, water column, benthic core (QCC) and aquatic D-net sampling methods of invertebrates in depressional wetlands of north-central Oklahoma. Means with an asterisk are significantly different at the 0.05 level

	QCC		D-Net			
Taxonomic Group	Mean	SE	Mean	SE	$t^{1,74}$	*P*
Nonmicrocrustacean	34553.6*	5594.9	3050.9	440.6	15.43	< 0.001
Microcrustacean	6462.4*	2224.6	315.9	131.2	7.2	< 0.001
Oligochaeta						
Naididae	17.1*	15.6	2.1	1.6	2.39	0.019
Hirudinea						
Helobdella	< 0.1*	< 0.1	< 0.1	< 0.1	2.33	0.022
Mooreobdella	0.6*	1.0	< 0.1	< 0.1	2.74	0.008
Gastropoda						
Gyraulus	0.8*	1.2	0.1	0.1	2.67	0.009
Helisoma	23.9*	15.0	3.3	2.0	3.57	0.001
Physa	3308.2*	1019.7	411.9	116.1	8.88	< 0.001
Amphipoda						
Hyalella	0.4	0.6	0.5	0.6	0.62	0.539
Copepoda	89.9*	55.2	0.7	0.6	4.93	< 0.001
Cladocera	1141.0*	498.1	59.4	28.5	6.73	< 0.001
Ostracoda	615.1*	398.5	15.7	13.9	4.41	< 0.001
Ephemeroptera						
Caenis	0.6	0.7	0.2	0.2	1.19	0.238
Callibaetis	1407.4*	572.9	255.7	82.9	4.84	< 0.001
Odonata						
Anax	1.1	0.9	1.7	1.0	0.53	0.596
Enallagma	649.9*	262.1	131.9	41.2	5.03	< 0.001
Lestes	< 0.1*	< 0.1	0.2	0.1	2.34	0.022
Libellulidae	0.8	0.9	1.2	0.8	0.52	0.606
Sympetrum	< 0.1	< 0.1	< 0.1	< 0.1	1.26	0.213
Tramea	0.1	0.1	0.4	0.3	1.21	0.231
Hemiptera						
Belastomatidae nymph	0.1	0.1	0.3	0.2	1.25	0.215
Buenoa	< 0.1*	< 0.1	0.1	0.1	2.05	0.044
Corixidae nymph	< 0.1	< 0.1	0.3	0.2	1.65	0.103
Hesperocorixa	< 0.1	< 0.1	< 0.1	< 0.1	1.73	0.089

Table 4 (continued)

Mesoveliidae nymph	0.2	0.3	< 0.1	< 0.1	1.41	0.162
Notonectidae nymph	28.1	18.9	22.8	9.8	0.39	0.695
Sigara	0.0*	0.0	0.1	0.1	4.09	< 0.001
Trichoptera						
Oecetis	< 0.1	< 0.1	< 0.1	< 0.1	0.67	0.502
Coleoptera						
Berosus adult	1.1	1.2	0.5	0.3	0.81	0.422
Berosus larvae	55.9*	30.9	11.3	4.8	2.85	0.006
Cybister larvae	< 0.1	< 0.1	< 0.1	< 0.1	1.11	0.273
*Enochrus*adult	< 0.1*	< 0.1	< 0.1	< 0.1	2.15	0.035
Haliplus larvae	4.6*	3.6	0.9	0.6	2.79	0.007
Helophorus adult	1.1*	0.8	< 0.1	< 0.1	3.60	0.001
Hygrotus adult	0.3	0.5	1.5	1.1	1.40	0.166
Laccophilus adult	0.0*	0.0	< 0.1	< 0.1	2.90	0.005
Liodessus adult	0.2	0.2	0.2	0.2	0.19	0.851
Paracymus adult	0.3	0.3	0.1	0.1	1.68	0.098
Peltodytes larvae	< 0.1	< 0.1	< 0.1	< 0.1	0.10	0.919
Tropisternus adult	< 0.1	0.1	0.7	0.5	1.86	0.067
Tropisternus larvae	0.3	0.4	1.3	0.8	1.49	0.140
Diptera						
Bezzia	< 0.1	< 0.1	< 0.1	< 0.1	0.35	0.725
Ceratopogonidae pupae	< 0.1	< 0.1	0.1	0.1	1.67	0.099
Chaborus	< 0.1	< 0.1	< 0.1	< 0.1	1.11	0.273
Chironomidae	3244.1*	1092.5	64.5	20.4	9.21	< 0.001
Dasyhelea	< 0.1	0.1	< 0.1	< 0.1	1.85	0.068
Stratiomyidae	0.3*	0.4	< 0.1	< 0.1	2.70	0.009
Tabanidae	0.1	0.1	< 0.1	< 0.1	1.28	0.204
Tanypodinae	49.5	33.2	41.0	12.3	0.31	0.757
Arachnida						
Aranae	< 0.1	< 0.1	< 0.1	< 0.1	0.25	0.804
Hydrachnidia	12.3	9.3	9.1	3.9	0.45	0.656

Table 5 Mean and standard error of nonmicrocrustacean, microcrustacean, and most common taxa (occurring in ≥ 10 % of the samples) biomass (mg/m^3) and t and *P* values for quadrat, water column, benthic core (QCC) and aquatic D-net sampling methods of invertebrates in depressional wetlands of north-central Oklahoma. Means with an asterisk are significantly different at the 0.05 level

	QCC		D-Net			
Taxonomic Group	Mean	SE	Mean	SE	$t^{1,74}$	*P*
Nonmicrocrustacean	43760.7*	8870.0	5103.1	966.4	11.43	< 0.001
Microcrustacean	730.2*	220.8	24.0	9.1	8.04	< 0.001
Oligochaeta						
Naididae	4.5*	4.1	0.3	0.2	2.85	0.006
Hirudinea						
Helobdella	< 0.1*	< 0.1	< 0.1	< 0.1	2.43	0.018
Mooreobdella	3.5*	5.3	< 0.1	< 0.1	2.79	0.007
Gastropoda						
Gyraulus	1.3*	1.9	0.1	0.1	2.58	0.012
Helisoma	340.7*	262.0	63.5	43.1	2.44	0.017
Physa	7448.0*	2542.9	1401.2	435.3	6.87	< 0.001
Amphipoda						
Hyalella	0.1	0.2	0.1	0.1	0.28	0.777
Copepoda	18.0*	11.2	0.1	0.1	5.03	< 0.001
Cladocera	123.2*	50.7	4.5	2.0	6.73	< 0.001
Ostracoda	67.0*	38.5	1.5	1.2	4.72	< 0.001
Ephemeroptera						
Caenis	0.3	0.3	0.1	< 0.1	1.52	0.133
Callibaetis	344.1*	136.8	54.6	18.2	5.23	< 0.001
Odonata						
Anax	9.0	8.9	7.0	4.4	0.26	0.797
Enallagma	251.8*	105.5	56.5	17.4	4.29	< 0.001
Lestes	< 0.1*	< 0.1	0.2	0.2	2.68	0.009
Libellulidae	0.4	0.4	0.3	0.2	0.26	0.795
Sympetrum	< 0.1	< 0.1	0.1	0.1	1.45	0.152
Tramea	0.5	0.6	0.9	0.7	0.53	0.595
Hemiptera						
Belastomatidae nymph	0.3	0.3	0.9	0.8	1.18	0.242
Buenoa	< 0.1*	< 0.1	0.2	0.3	2.05	0.044
Corixidae nymph	< 0.1*	< 0.1	0.2	0.2	2.12	0.037
Hesperocorixa	< 0.1	< 0.1	0.4	0.4	1.55	0.126

Table 5 (continued)

Mesoveliidae nymph	0.1	0.2	< 0.1	< 0.1	1.73	0.087
Notonectidae nymph	25.9	17.5	13.4	6.0	1.15	0.255
Sigara	0.0*	0.0	0.1	0.1	4.11	< 0.001
Trichoptera						
Oecetis	< 0.1	< 0.1	< 0.1	< 0.1	0.25	0.802
Coleoptera						
Berosus adult	3.4	3.7	1.3	0.9	0.91	0.365
Berosus larvae	47.4*	27.0	5.4	2.4	3.40	0.001
Cybister larvae	< 0.1	< 0.1	< 0.1	0.1	0.81	0.418
*Enochrus*adult	< 0.1*	< 0.1	< 0.1	< 0.1	2.23	0.029
Haliplus larvae	3.0*	2.4	0.5	0.4	2.81	0.006
Helophorus adult	0.6*	0.4	< 0.1	< 0.1	3.28	0.002
Hygrotus adult	0.6	1.0	3.1	2.3	1.52	0.133
Laccophilus adult	0.0*	0.0	< 0.1	< 0.1	2.90	0.005
Liodessus adult	< 0.1	< 0.1	0.1	0.1	0.67	0.503
Paracymus adult	0.1	0.1	< 0.1	< 0.1	1.49	0.141
Peltodytes larvae	< 0.1	< 0.1	< 0.1	< 0.1	0.24	0.814
Tropisternus adult	0.5	1.1	5.9	4.8	1.63	0.108
Tropisternus larvae	0.1	0.2	0.7	0.4	1.70	0.094
Diptera						
Bezzia	< 0.1	< 0.1	< 0.1	< 0.1	0.69	0.490
Ceratopogonidae pupae	< 0.1	< 0.1	< 0.1	< 0.1	1.08	0.285
Chaborus	< 0.1	< 0.1	< 0.1	< 0.1	0.25	0.803
Chironomidae	1151.1*	384.9	13.6	4.4	9.39	< 0.001
Dasyhelea	< 0.1	< 0.1	< 0.1	< 0.1	1.47	0.145
Stratiomyidae	0.4*	0.4	< 0.1	< 0.1	2.87	0.005
Tabanidae	0.1	0.2	< 0.1	< 0.1	1.44	0.154
Tanypodinae	17.1	11.5	5.9	1.7	1.48	0.144
Arachnida						
Aranae	< 0.1	< 0.1	< 0.1	< 0.1	0.16	0.870
Hydrachnidia	5.1	4.0	3.0	1.3	0.72	0.475

APPPENDICES

Appendix 1 List of invertebrate taxa collected during 2009 and 2010 from depressional wetlands in north central Oklahoma

Class/Order	Family	Subfamily/Genus
Platyhelminthes		
Hydroida		
	Hydrididae	
		Hydra
Nematoda		
Arhynchobdellida		
	Erpobdellidae	
		Erpobdella
		Mooreobdella
Rhynchobdellida		
	Glossiphoniidae	
		Desserobdella
		Helobdella
		Placobdella
Branchiodellida		
	Branchiobdellidae	
Tubificida		
	Naididae	
	Tubificidae	
Gastropoda		
	Ancylidae	
		Ferrissia
	Lymnaeidae	
		Fossaria
		Stagnicola
	Physidae	
		Physa

Appendix 1 (Continued)

	Planorbidae	
		Gyraulus
		Helisoma
		Promenetus
Veneroida		
	Sphaeriidae	
Cladocera		
Anostraca		
	Branchinecta	
		Branchinecta
	Chirocephalidae	
		Eubranchipus
	Streptocephalidae	
		Streptocephalus
Spinicaudata		
	Cyzicidae	
	Limnaidae	
		Eulimnadia
Amphipoda		
	Hyalellidae	
		Hyalella
Calanoida		
Cyclopida		
	Cyclopidae	
Ostracoda		
Decapoda		
	Cambaridae	
Isopoda		
Arachnida		
	Halacaridae	
	Hydrachnidae	
		Aranae
Collembola		
Ephemeroptera		
	Caenidae	
		Caenis
	Baetidae	
		Callibaetis
Odonata		
	Aeshnidae	

Appendix 1 (Continued)

		Anax
		Epiaeschna
	Libellulidae	
		Erythemis
		Erythrodiplax
		Nannothemis
		Pachydiplax
		Pantala
		Plathemis
		Perithemis
		Pseudoleon
		Sympetrum
		Tramea
	Coenagrionidae	
		Enallagma
	Lestidae	
		Lestes
Orthoptera		
	Gryllidae	
		Allonemobius
		Neonemobius
	Tettigoniidae	
		Conocephalus
Hemiptera[a]		
	Aradidae $_1$	
	Ciccadellidae $_2$	
	Coreidae $_1$	
	Corimelaenidae $_2$	
	Fulgoroidea $_3$	
	Lygaeidae	
	Pentatomidae	
	Reduviidae	
	Belastomatidae $_3$	
		Belastoma
	Corixidae $_3$	
		Arctocorisa
		Cenocorixa

Appendix 1 (Continued)

		Corisella
		Graptocorixa
		Ramphocorixa
		Hesperocorixa
		Sigara
		Trichocorixa
	Gerridae $_3$	
		Gerris
	Hebridae $_3$	
		Hebrus
		Lipogomphus
		Merragata
		Oravelia
	Hydrometridae $_1$	
		Hydrometra
	Mesoveliidae $_3$	
		Mesovelia
	Macroveliidae $_3$	
		Macrovelia
	Naucoridae $_1$	
	Nepidae $_1$	
		Ranatra
	Notonectidae $_3$	
		Buenoa
		Notonecta
	Pleidae $_3$	
		Neoplea
	Saldidae $_3$	
		Rupisalda
		Saldula
	Veliidae $_1$	
Thysanoptera		
Trichoptera		
	Hydroptilidae	
		Oxyethira
		Orthotrichia
	Leptoceridae	
		Oecetis

Appendix 1 (Continued)

Lepidoptera		
	Crambidae	
	Noctuidae	
Coleoptera[b]		
	Dytiscidae	
		Hydroporinae $_1$
		Agabinae $_1$
		Bidessini $_2$
		Agabus $_3$
		Biodessonotus $_2$
		Celina $_1$
		Copelatus $_2$
		Coptotomus $_2$
		Cybister $_1$
		Desmopachria $_2$
		Graphoderus $_2$
		Hydrovatus $_2$
		Hygrotus $_2$
		Ilybius $_1$
		Laccophilus $_3$
		Liodessus $_2$
		Neoporus $_2$
		Thermonectus $_2$
		Uvarus $_2$
	Gyrinidae	
		Dineutus $_1$
		Gyrinus $_3$
	Haliplidae	
		Haliplus $_3$
		Peltodytes $_1$
	Noteridae	
		Hydrocanthus $_3$
		Suphis $_1$
		Suphisellus $_3$

Appendix 1 (Continued)

	Hydrophilidae	
		Anacaena $_{3}$
		Berosus $_{3}$
		Crenitis $_{2}$
		Enochrus $_{3}$
		Helophorus $_{3}$
		Hydrobiomorhpa $_{2}$
		Hydrobius $_{3}$
		Hydrochus $_{2}$
		Hydrophilus $_{1}$
		Paracymus $_{3}$
		Tropisternus $_{3}$
	Anthicidae $_{3}$	
	Carabidae $_{3}$	
	Chrysomelidae $_{3}$	
	Coccinellidae $_{3}$	
	Curculionidae $_{3}$	
	Dryopidae $_{3}$	
	Elmidae $_{3}$	
	Heteroceridae $_{1}$	
	Hydraenidae $_{2}$	
	Hydroscaphidae $_{2}$	
	Scarabaeidae $_{3}$	
	Scirtidae $_{2}$	
	Staphylinidae $_{2}$	
	Unknown 1 $_{2}$	
	Unknown 2 $_{2}$	
	Unknown 3 $_{2}$	
	Unknown 4 $_{1}$	
Hymenoptera		
	Braconidae	
	Eulophidae	
	Figitidae	
	Formicidae	

Appendix 1 (Continued)

	Ichneumonidae	
	Scelionidae	
Diptera		
	Ceratopogonidae	
		Atrichopogon
		Bezzia
		Ceratopogon
		Culicoides
		Dasyhelea
		Forcipomyia
		Mallochelea
		Probezzia
		Serromyia
		Sphaeromias
	Chaoboridae	
		Chaoborus
	Chironomidae	
		Tanypodinae
	Culicidae	
		Aedes
		Anopheles
		Culiseta
		Culex
		Masonia
		Psorophora
		Toxorhynchites
	Tipulidae	
		Limoniinae
	Ephydridae	
		Hydrellia
		Notiphila
		Parydra
		Scatella
		Setacera
	Pelecorhynchidae	
		Glutops
	Dolichopodidae	
	Empididae	
	Muscidae	
	Phoridae	

Appendix 1 (Continued)

	Psychodidae
	Sciomyzidae
	Stratiomyidae
	Tabanidae

a) 1 - nymph only, 2 - adult only, 3 - nymph and adult

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

Apendix 2 Land-use practice, hydroperiod, size (hectares), and location of sampled depressional wetlands in north central Oklahoma during 2009 and 2010. Locations are X and Y coordinates in meters (UTM Zone 14N NAD83). In name, Kin is Kinfisher County, Gar is Garfield County, Maj is Major County, and Grt is Grant County

Name	Land-use Practice	Hydroperiod	Size	X	Y
Kin_2	Crop	3	0.7	605135	3987903
Kin_5	Crop	2	0.4	605561	3987995
Kin_42	Pasture	3	1.3	604509	3991103
Kin_44	Pasture	3	1.7	604088	3991317
Kin_45	Pasture	3	0.6	604581	3991515
Kin_52	Crop	3	1.5	604737	3993117
Kin_58	Pasture	3	0.6	604735	3989508
Kin_59	Pasture	2	0.9	604614	3989628
Kin_81	Range	3	0.7	604686	3988839
Kin_82	Range	2	0.2	604682	3989386
Kin_83	Range	4	0.8	604645	3989135
Kin_84	Range	4	0.9	604192	3988748
Kin_80	Pasture	2	0.5	604755	3991678
Kin_86	Range	1	0.1	604603	3989325
Kin_85	Crop	3	0.9	603915	3991221
Gar_5	Crop	2	1.3	585132	4007165
Gar_11	Range	3	0.3	583486	4010289
Gar_18	Range	2	0.1	583619	4010418
Gar_19	Range	2	0.1	583619	4010157
Maj_1	Pasture	5	8.1	574345	4009082
Maj_3	Crop	3	1.9	574746	4008776
Maj_4	Crop	3	0.2	575470	4006198
Maj_5	Crop	2	1.3	575600	4005821
Maj_8	Crop	5	2.2	580177	4009289
Maj_9	Crop	5	0.2	580032	4009152
Maj_16	Crop	2	0.3	574725	4008484
Maj_18	Crop	4	0.4	580513	4009714
Maj_19	Pasture	4	0.9	579182	4013616
Maj_20	Pasture	5	0.3	579090	4013443
Grt_1	Pasture	3	2.9	595154	4071435
Grt_4	Crop	3	0.5	595031	4070179
Grt_5	Crop	3	0.5	589208	4069755
Grt_6	Crop	3	0.2	589129	4069784
Grt_8	Crop	1	1.0	589184	4069962
Grt_11	Crop	4	1.8	590607	4067807

Appendix 2 (Continued)

Grt_12	Crop	3	0.8	589556	4066414
Grt_13	Pasture	5	1.4	589424	4066144
Grt_15	Pasture	2	0.3	589206	4066208
Grt_20	Pasture	3	1.0	589466	4065872
Grt_24	Pasture	3	0.5	585827	4068844
Grt_26	Range	3	1.4	587113	4074653
Grt_31	Crop	2	1.2	587557	4070501
Grt_35	Pasture	5	0.3	587270	4067074
Grt_37	Pasture	3	0.4	587418	4066450
Grt_39	Range	4	3.5	587613	4064894
Grt_41	Range	4	1.1	588112	4064844
Grt_46	Range	2	1.1	594600	4066226
Grt_48	Range	2	0.1	594001	4066449
Grt_49	Range	3	0.4	594854	4067814
Grt_53	Pasture	4	0.3	594660	4069677
Grt_54	Range	3	0.2	585115	4074225
Grt_56	Range	4	1.4	585750	4074005
Grt_58	Range	3	0.3	586712	4074589
Grt_63	Range	2	0.2	594843	4069739
Grt_66	Range	1	0.3	595335	4069649
Grt_67	Range	3	0.9	595062	4069646
Grt_69	Range	4	0.3	595102	4068283
Grt_72	Pasture	2	0.1	595838	4069531

Appendix 3 List of plant taxa and vegetation complexity rating for wetland plants recorded in depressional wetlands in north central Oklahoma, 2009-2010

Plant Taxon	Common Name	Complexity
Ammannia coccinea	valley redstem	1
Bacopa rotundifolia	disk waterhyssop	1
Carex sp.	sedge	1
Cicuta maculata	spotted water hemlock	1
Coreopsis tinctoria	golden tickseed	1
Cynodon dactylon	Bermudagrass	1
Cyperus sp.	flatsedge	1
Echinochloa sp.	barnyardgrass	1
Echinodorus bertoi	upright burhead	1
Eleocharis compressa	flatstem spikerush	1
Eleocharis palustris	common spikerush	1
Eleocharis quadrangulata	squarestem spikerush	1
Heteranthera limosa	blue mudplantain	1
Heteranthera rotundifolia	roundleaf mudplantain	1
Lemna minor	common duckweed	1
Ludwigia palustris	marsh seedbox	1
Marsilea vestita	hairy waterclover	1
Paspalum distichum	knotgrass	1
Poaceae (unknown)	grass	1
Polygonum amphibium	water knotweed	1
Polygonum lapathifolium	curlytop knotweed	1
Polygonum pennslyvanicum	Pennsylvania smartweed	1
Rorippa palustris	bog yellowcress	1
Rumex crispus	curly dock	1
Sagittaria sp.	arrowhead	1
Salix sp.	willow	1
Schoenoplectus americanus	chairmaker's bulrush	1
Schoenoplectus tabernaemontani	softstem bulrush	1
Schoenplectus fluviatilis	river bulrush	1
Typha domingensis	southern cattail	1
Xanthium strumarium	rough cocklebur	1
Potamogeton nodosus	longleaf pondweed	2
Stuckenia pectinata	sago pondweed	2
Zannichellia palustris	horned pondweed	2
Ceratophyllum demersum	coon's tail	3
Chara spp.	muskgrass	3
filamentous algae		3

Appendix 3 (Continued)		
Myriophyllum heterophyllum	twoleaf watermilfoil	3
Najas guadalupensis	southern waternymph	3
Nitella spp.	stonewort	3
Potamogeton diversifolious	waterthread pondweed	3

VITA

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Candidate for the Degree of

Master of Science

Thesis: WETLAND INVERTEBRATE COMMUNITY RESPONSE TO LAND-USE AND OTHER LOCAL AND LANDSCAPE FACTORS

Major Field: Natural Resource Ecology and Management

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Pages in Study: 128 Candidate for the Degree of Master of Science

Major Field: Natural Resource Ecology and Management

Scope and Method of Study: The objectives of this study were to determine the influence of land-use and other local and landscape factors on wetland invertebrate diversity, biomass, and taxa composition within depressional wetlands of Oklahoma. Fifty-eight wetlands in north central Oklahoma were sampled for invertebrates during 2009 and 2010. All wetlands were closed depressional, palustrine wetlands. Wetlands were located within annually tilled cropland, improved pastureland, or native rangeland. To determine the influence of land-use on the invertebrate community means were compared for invertebrate diversity, biomass of taxa and functional feeding groups among the land-use practices. Differences in taxa composition between land-use practices were also assessed. To determine the influence of local and landscape factors, linear regression and ordination was used to determine the variables explaining the most variation in invertebrate diversity, the biomass of the most common taxa, and taxa composition. Local factors included vegetation complexity and water quality, while landscape factors included land-use practices and wetland area within 1 and 2 km of study wetlands.

Findings and Conclusions: Land-use influenced nine invertebrate taxa, three functional feeding groups, as well as invertebrate taxa composition. The majority of the taxa did not significantly respond to land-use practices. However, differences in plant cover, vegetation complexity, and water quality among the land-use practices suggest some degradation of wetlands caused by land-use. High variability among sites may have prevented finding more significant results. Numerous local and landscape factors did significantly explain variation in invertebrate taxa biomass as well as taxa composition. Overall, 24 taxa had the most variation explained with local factors, while seven taxa had the most variation explained by landscape factors. The most variation in taxa composition was explained by local factors. Dominant local factors were vegetation complexity, plant cover, and water depth, while wetland density, hydroperiod diversity, and land-use practices were important landscape factors. The results indicate that management and conservation efforts should consider habitat quality within the wetland, but also wetland habitat availability within the landscape.

ADVISER'S APPROVAL: Craig A. Davis