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AN HGM APPROACH FOR ASSESSING WETLAND FUNCTIONS IN CENTRAL OKLAHOMA: HYDROGEOMORPHIC CLASSIFICATION AND FUNCTIONAL ATTRIBUTES

FINAL REPORT

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[3]Oklahoma Conservation Commission
2800 N. Lincoln Blvd. Suite 160,
Oklahoma City, OK 73105

Prepared by:
Daniel Dvorett, M.S.[1], Joseph R. Bidwell, Ph.D.[1],
Craig A. Davis, Ph.D.[2], and Chris DuBois, M.S.[3]

[1]Department of Zoology
501 Life Sciences West
Stillwater, OK 74078

[2]Department of Natural Resource Ecology and Management,
Oklahoma State University, 008C
Agricultural Hall, Stillwater, OK 74078

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EXECUTIVE SUMMARY

By 2014, states should have the capacity to monitor and report on wetland status and trends. Of primary importance is assessing the ecological health of wetlands and identifying where systems have been degraded by anthropogenic disturbance. The Hydrogeomorphic Approach (HGM) is a rapid assessment protocol that has been implemented regionally throughout the United States to assess wetland health and meet state and federal monitoring objectives. System health is determined by calculating function using assessment models or algorithms that combine a number of ecological measures and indicators called assessment variables. The output of the assessment model or the functional capacity of a study site is then compared to the functional capacity at a pristine or least disturbed reference standard site. The degree of deviation of function from the reference standard condition can then attributed to anthropogenic disturbance. National wetland classes and regional wetland subclasses, based on hydrology and geomorphology, are employed in HGM to reduce the natural variability of grouped wetlands and to strengthen the relationship between assessment model output and anthropogenic disturbance.

The study described in this report represents a collaborative effort between Oklahoma State University and the Oklahoma Conservation Commission to develop and assess the utility of a regional sub-classification system for the Cross Timbers and Central Great Plains Ecoregions of Oklahoma. The actual field work and data analyses were undertaken by Mr. Daniel Dvorett as part of the requirements for a Masters of Science degree in the Department of Zoology at Oklahoma State University. His thesis titled "An HGM Approach for Assessing Wetland Functions in Central Oklahoma: Hydrogeomorphic Classification and Functional Attributes" was successfully defended and submitted in final form to the Graduate College in December 2010. This thesis forms the body of this final report, with the data presented in chapter format.
Chapter 1 presents the development of a regional sub-classification system and a spatially referenced inventory of wetlands within HGM subclasses. The study presented in Chapter 2 attempts to calibrate assessment variables to disturbance to facilitate the development of robust assessment models responsive to anthropogenic stress for the two most common HGM subclasses in the ecoregions.

Summary of Chapter 1: Developing a hydrogeomorphic wetland inventory: Reclassifying National Wetlands Inventory polygons in geographic information systems.

The objectives of the study presented in Chapter 1 were to create an HGM regional sub-classification system and to develop a spatially-referenced wetland inventory based on the HGM subclasses for the Cross Timbers and Central Great Plains Ecoregions of Oklahoma. The National Wetlands Inventory (NWI), a digital, spatially-referenced inventory of wetland polygons was used to identify an initial population of wetlands within the study area. NWI polygons were mapped from aerial photographs using the Cowardin classification system, which was not designed to evaluate wetland function. In Geographic Information Systems (GIS), NWI polygons were reclassified into HGM classes (depressional, lacustrine fringe, riverine, and impounded depressional) based on attribute queries of NWI classes and spatial queries using collateral datasets that provided information on hydrology and geomorphology.

Once classified, 190 wetlands were assessed in the field to develop HGM subclasses. Within three focal study counties (Okfuskee, Logan and Garfield Counties), 49 to 50 wetlands were selected using a Generalized Random Tessellation Stratified design. An additional 41 wetlands were selected from throughout the study area to supplement the HGM sub-classification scheme and ensure that functional variability was captured. Field assessments used in the development of subclasses included visual inspection of hydrology (e.g., drift lines, high water marks, and soil saturation) and geomorphology (e.g., topography and proximity to other surface water bodies). Within the study area, we documented wetlands within four of the national HGM classes, depressional, lacustrine fringe, riverine and slope.

Based on field assessments of hydrology and geomorphology as well as guidance from previously developed HGM sub-classification systems, 16 subclasses were created for central Oklahoma. Riverine wetlands include connected oxbows, beaver complexes, riparian, in-channel, floodplain and floodplain depressions. We divided slope wetlands into headwater slopes and low-gradient slopes. Depressional wetlands were sorted into three natural subclasses and two human-created subclasses. The three natural classes include groundwater depressions, closed surface water depressions and open surface water depressions. Closed and open impounded depressions were included as subclasses because farm ponds which take on the attributes of wetlands are common throughout the study area. Lacustrine fringe wetlands include one natural subclass, disconnected oxbow wetlands, and two subclasses of anthropogenic origin, reservoir fringe and pond fringe. A dichotomous key to facilitate sub-classification of future study wetlands was developed during this study.

During field assessments of the 149 randomly-selected sites in the three focal counties, the HGM class assigned in GIS was field verified. This field verification provided information on the accuracy of the classification, which was used to develop an estimate of the number of wetlands in each HGM class. Based on field verification, the overall accuracy of the GIS classification was 59.8%. Inherent issues with NWI due to attribute accuracy, spatial accuracy, and original age of mapping accounted for the incorrect classification of 20.8% of all field verified sites and more than half of the misclassified sites in this analyses. An inventory of wetlands in each HGM class and subclass was calculated based on user's accuracy metrics and HGM class populations in GIS. The most common natural HGM class within the study area was riverine and the most common HGM subclasses were riparian and connected oxbows.

Despite a potential for high error rate, reclassifying NWI polygons into HGM classes helped develop an understanding of the spatial distribution and relative abundance of specific wetland classes and subclasses within the study area. The inventory can help provide insight into rare wetland types in need of protection, focus restoration efforts, and identify local and systemic wetland degradation. Updating NWI maps can be a useful step in assessing statewide resources as long as those implementing reclassification understand the potential methodological limitations and assess accuracy using field verification.

Summary of Chapter 2: Assessing variability among hydrogeomorphic riverine subclasses

The ability of HGM assessment models to accurately predict disturbance has limited verification. Calibrating assessment variables to anthropogenic disturbance is an essential preliminary step to developing assessment models that have an output indicative of disturbance. In order to calibrate the response of assessment variables to disturbance, reference wetlands that range from highly altered to least disturbed reference standard conditions were selected. Our objectives were to:

1. Identify 20 reference wetlands within each of the two most common natural HGM subclasses, riparian and connected oxbows, from the most common natural HGM class, riverine, and
2. Collect data from reference wetlands to begin development of functional assessment models that are responsive to anthropogenic disturbance.

At 20 reference sites for each subclass (riparian and oxbow) we collected data for 21 site metrics that could be used as assessment variables in May and June 2010. Site metrics included vegetation physiognomy (e.g., canopy cover and coarse woody debris volume), water chemistry (e.g., hardness and conductivity) and soil variables (e.g., texture and organic matter). Anthropogenic disturbance at each of the reference sites was calculated as landscape disturbance based on National Land Cover Dataset land uses and using 100 m and 1000 m buffer widths. Landscape disturbance was calculated two ways, as a % of human altered land (cultivated crops, developed land, pasture/hay and barren land) and as a land use score, where each land use was assigned a coefficient based on its potential to induce anthropogenic stress to the wetland.

Prior to developing specific functional assessment models, we wanted to determine if the subclasses we developed indeed reduce natural variability among assessment variables, if assessment variables could be calibrated to disturbance, and if natural variability within subclasses could potentially be confounding the relationship between disturbance and the assessment variables. Using redundancy analysis (RDA), we determined that subclass accounted for 14.2% of the variance for the selected metrics, which suggests subclass can aid in reducing natural variability among wetlands. However, there were limited relationships between landscape disturbance metrics and assessment variables within each subclass. Percent human-altered land within 100 m of the wetland only had significant effects on soil texture in riparian sites and organic matter among oxbows.

The high degree of natural variability from climatic and hydrologic factors within both subclasses may be masking the impact of landscape disturbance on the other measured site metrics. Precipitation had significant effects within each of the subclasses, indicating that the reference domain (e.g., ecoregions), as currently defined, may not be appropriate and may need to be further subdivided. There is an east to west precipitation gradient in the study area with eastern portions receiving more than 110 cm of average annual rainfall and western portions receiving less than 60 cm of average annual rainfall. Precipitation explained 12.8% of the variance among site metrics for riparian sites using RDA. Soil texture, water hardness, water alkalinity, and herbaceous cover of riparian sites were all affected by precipitation as indicated by forward stepwise regressions. Among oxbow sites, precipitation affected herbaceous cover, coarse woody debris stem count and tree density. Using principal components analysis and RDA, we identified additional natural hydrologic factors that appear to be driving variation within the subclasses. Variation among oxbow sites seems to be explained by water source (groundwater vs. surface water) and degree of isolation from the river of origin, while Strahler stream order was important in explaining variation among water chemistry variables for riparian sites. Further subdivision of subclasses based on these hydrological and climatic factors may help reduce natural variability within a subclass, aid in identification of relationships between disturbance and assessment models, and ultimately increase the responsiveness of assessment models to disturbance.

A high degree of natural variability within subclasses is a potential explanation for why landscape disturbance metrics were not well correlated with site metrics. Alternatively, site metrics may be more responsive to on-site disturbance factors or severe landscape degradation. Developing correlations between disturbance and assessment variables could be improved by expanding collection of disturbance metrics to include on-site disturbance such as hydrologic alterations, invasive species colonization or any other factors that may be regionally important. Observing how site metrics respond to a variety of disturbance metrics using RDA and forward stepwise regressions may reveal patterns missed by solely using landscape disturbance.

We found a limited relationship between landscape disturbance metrics and the measured assessment variables within each subclass. If functional assessment models were developed for the riparian and connected oxbow subclasses, there would be little evidence that model output was in any way related to impairment from anthropogenic disturbance. Without establishing reliable trends between disturbance and assessment variables, HGM assessment tools cannot identify system health, and their value in wetland monitoring is severely reduced. The natural variability of assessment variables within a subclass may be masking subtle landscape disturbance effects, such that models can only reliably identify large-scale on-site disturbances or severe landscape degradation. The inability to relate disturbance and assessment variables may not be just confined to riverine wetlands in Oklahoma, and those who develop HGM models without calibration need to be aware that assessment model output may not indicate wetland health.

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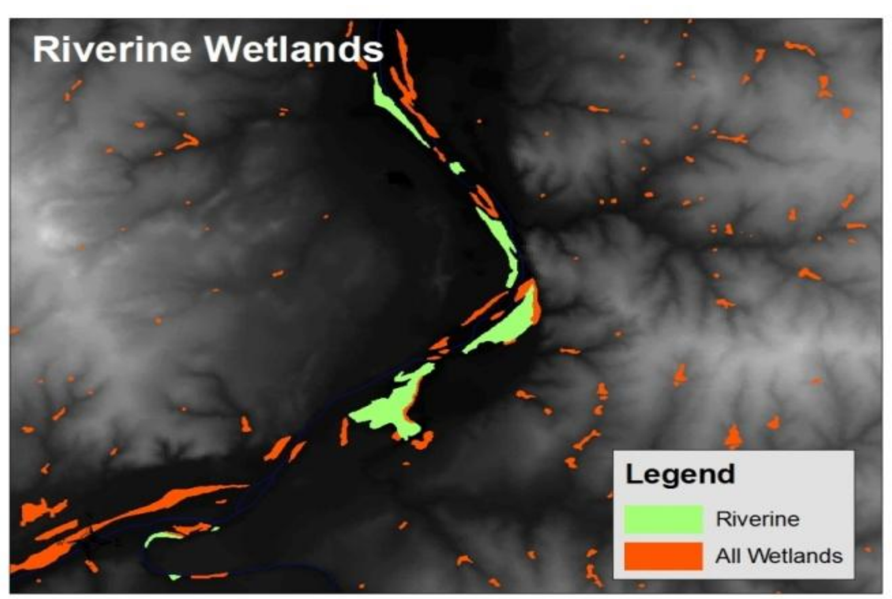

CHAPTER I

DEVELOPING A HYDROGEOMOPRHIC WETLAND INVENTORY: RECLASSIFYING NATIONAL WETLANDS INVENTORY POLYGONS IN GEOGRAPHIC INFORMATION SYSTEMS

INTRODUCTION

Conversion of wetlands for agriculture and development has reduced wetland area in the conterminous United States from approximately 89.4 million ha in 1780 to approximately 43.6 million ha in 2004 (Dahl and Johnson 1991, Dahl 2006). In Midwestern states, wetland losses have ranged from 67% in Oklahoma to 89% in Iowa (Dahl 1990). Although the remaining wetlands in the United States only represent 5% of the land area (Wilen and Bates 1995), these systems provide unique habitats for diverse plant and animal communities as well as a variety of ecosystem services that are beneficial to humans (Brinson 1993a, b, Smith et al. 1995, Euliss et al. 2008, Smith et al. 2008).

To maintain services that wetlands provide, wetland scientists, managers, and regulators need rapid wetland assessment methods that can identify sources of human disturbance and the resulting loss of ecosystem function (Hruby 1999, Stein et al. 2009). Accurate wetland assessment techniques and monitoring programs can help limit future degradation of wetlands and aid in targeting focused restoration efforts by identifying systemic or local losses of wetland function. However, assessing wetland functions is confounded by natural variability, making it difficult to distinguish between natural variation in function and changes in functions due to degradation. Natural variation in wetland function can be caused by interrelated variability in climatic conditions, hydrology (e.g., water source and movement), and geomorphology (e.g., topography and proximity to other surface waters) (Brinson 1993a). For example, precipitation-fed bogs have lower primary productivity, lower nutrient loads, and lower sediment loads than floodplain wetlands that receive water from overbank flow. The difference is simply because water traveling overland has a much higher potential to carry nutrients, and sediments than precipitation (Moore and Bellamy 1974, Brinson 1993b).

Resolution of wetland assessment can be improved by employing wetland classification to create groups of systems with similar structure and function (Cowardin and Golet 1995). The Hydrogeomorphic (HGM) Approach, developed for the U.S. Army Corps of Engineers (USACE) (Brinson 1993a), includes a functional classification system based on three factors (landscape position, water source and hydrodynamics) that have been demonstrated to influence wetland function (Brinson 1993b, Cole et al. 1997, Shaffer et al. 1999, Cole et al. 2002). HGM includes seven national wetland classes that can be further modified into subclasses based on regional conditions. Subclasses are primarily derived from localized hydrological processes and landscape conditions, further reducing functional variability among wetlands (Brinson 1993a). As such, HGM is typically applied at more localized scales such as ecoregions or watersheds (Klimas et al. 2004, Stutheit et al. 2004).

HGM functional assessments, within each subclass, use selected functional attributes characterized across a series of reference wetlands that represent a range of conditions from pristine (if possible) to highly degraded. Basic models of wetland function can then be developed and used for the actual wetland assessments. Sub-classification is an essential preliminary step to the development of HGM assessment models. Because wetlands of similar condition with an HGM subclass should function similarly, deviation from pristine conditions can be attributed to disturbance (Brinson 1993a).

Development of an inventory of wetlands classified using HGM or another functional classification system can provide a potential tool for tracking gains and losses of specific wetland functions. The National Wetlands Inventory (NWI) is the most extensive database of wetland resources in the United States but has limited ability to connect mapped wetlands with specific functions (Brinson 1993a). NWI wetlands are classified using Cowardin classification (Cowardin et al. 1979), which is a hierarchical system based primarily on vegetation and hydroperiod. The Cowardin classification system was applied to NWI because the classification criteria, hydroperiod, and vegetation community are easily discernable from aerial photography (Cowardin and Golet 1995, Wilen and Bates 1995). These classification criteria served the goal of NWI in tracking area of wetland loss and gain, but the Cowardin classes were not designed to estimate wetland functions within groupings. There is the potential to infer wetland function from Cowardin classes. For example, forested wetlands have the potential to support wildlife species that require trees for nesting, while palustrine wetlands do not (Brinson 1993a). However, currently there are no assessment tools derived for Cowardin classes to estimate potential function.

Subjecting NWI polygons to a series of queries based on their attributes and spatial location in the context of Geographic Information Systems (GIS) could allow for reclassification of Cowardin classes into HGM classes. Because NWI maps are completed for entire states, this would allow for the development of an HGM inventory, done primarily in the office without additional re-mapping efforts. Others have developed such a reclassification of NWI polygons into functional classes. Tiner (2003, 2005) developed

HGM-like attributes for NWI polygons for application in GIS. The additional attribution of NWI polygons includes landscape position, landform, water flow path and waterbody types (LLWW). These attributes, while not precisely aligned with HGM classes, provide information on the hydrogeomorphology of wetlands. As a result, LLWW attribution allowed wetland experts to predict the level with which wetland classes are expected to perform specific functions and track loss of wetland function (Tiner 2003, 2005). Tiner (2005) used LLWW attribution of NWI polygons in the Nanticoke Watershed in Delaware and Maryland to estimate cumulative loss of functional capacity for nine wetland functions. The Montana National Heritage Program has used an LLWW approach to track changes in wetland function at the watershed and ecoregion scale (Newlon and Burns 2010a, b). In both cases, LLWW attributes were applied to NWI polygons and wetlands were predicted to perform specific functions due to their new LLWW designations. By comparing historic wetland extent to current wetland extent, LLWW users are able to assess the degree of wetland function lost (Tiner 2005, Newlon and Burns 2010a, b).

LLWW attributes are assigned to NWI polygons utilizing attribute queries of NWI polygons in GIS, and certain LLWW attributions require manual observations of individual NWI polygons (Kevin McGuckin, Virginia Tech University, Blacksburg, VA, USA, personal communication). Over large areas, manual manipulations can become time consuming. As such, LLWW has generally been applied at the watershed scale. As of 2008, the largest area attributed with LLWW modifiers was the state of New Jersey (22,608 km^2) (Tiner 2010). Additionally, to our knowledge the accuracy of the LLWW attributes has not been verified in the field.

The objectives of this project were to create an HGM sub-classification for central Oklahoma wetlands, develop an inventory based on that sub-classification, and assess the accuracy of the inventory with field verification. We developed a methodology that eliminated manual manipulations of individual NWI polygons to facilitate assignment of HGM classes to a large area (i.e., central Oklahoma, 107,610 km^2), more rapidly than LLWW. We also wanted to test the accuracy of our classification to determine if additional tools could be developed to effectively track losses in wetland function. If NWI polygons cannot be assigned to an HGM class effectively, estimates of the distribution and loss and gain of wetland functions become inaccurate.

METHODS

Study Area

The study area consisted of the Central Great Plains and Cross Timbers Ecoregions of Oklahoma (Fig. 1). The Central Great Plains Ecoregion extends from Texas to Nebraska including a large portion of central Oklahoma (73,251 km^2), while the Cross Timbers Ecoregion extends from Texas to southern Kansas and is directly east of the Central Great Plains Ecoregion (Omernik 1987). In Oklahoma, the Cross Timbers encompasses 34,359 km^2. The topography includes hills, salt plains, karst formations, flats and sand dunes. Elevation ranges from approximately 200 m to approximately 800 m (Woods et al. 2005). In the Central Great Plains, native vegetation is dominated by mixed grass prairie species, such as big bluestem (*Andropogon gerardii*), little blue stem (*Schizachyrium scoparium*), switch grass (*Panicum virgatum*) and indiangrass (*Sorghastrum nugans*), with wooded riparian corridors (Woods et al. 2005).

In the Cross Timbers Ecoregion, where soils were formed from sandstone, post oak (*Quercus stellata*) and blackjack oak (*Quercus marilandica*) woodlands and savannas are common. On finer textured soils derived from limestone and shale, tall grasses form the native vegetation community. Rangeland and cropland are common in both ecoregions (Woods et al. 2005). The study area has an east-west precipitation gradient with the easternmost portion in the Cross Timbers receiving on average 110 cm of annual rainfall, while the westernmost portion of the study area in the Central Great Plains receives on average 60 cm of annual rainfall (Taylor et al. 1995).

GIS Classification

HGM divides wetlands into 7 national classes: depressional, lacustrine fringe, tidal fringe, slope, riverine, mineral soil flats and organic soil flats. Classification is based on water source and transport, hydrodynamics and geomorphology, which have been demonstrated to influence wetland function (Brinson 1993b). For a more complete review of HGM classification see Brinson (1993a) or Smith et al. (1995). Of the 7 national classes, depressional, lacustrine fringe, riverine and slope occur within the study area.

NWI wetlands were reclassified into riverine, depressional and lacustrine HGM classes based on spatial queries using GIS (Genet and Olsen 2008). Depressional wetlands were further subdivided into natural and human-made systems. Human-made systems included any wetland that was diked or excavated to facilitate water storage. Collateral data layers provided information on the geomorphology and hydrology of individual wetland polygons to facilitate our reclassifications. These data included county soil survey geographic datasets (SSURGO) originated by Natural Resources Conservation Service (NRCS) and national hydrography datasets (NHD). SSURGO datasets include information about the flooding frequency and the drainage characteristics of soils and NHD provides digital locations of river center lines.

Cowardin deep water habitats (i.e., river channels and reservoirs) were removed from the target population of NWI wetlands because they are not wetlands. Impounded depressions were defined as all NWI polygons with an impounded or excavated designation in NWI and < 0.4 ha. Riverine wetlands included all NWI polygons that occurred on occasionally or frequently flooded soils according to NRCS soil maps or intersected NHD river center lines. All wetlands abutting polygons that were mapped as lacustrine in NWI were placed into the lacustrine class. NWI polygons with an impounded or excavated designation > 0.4 ha were also included in the lacustrine class. Depressional wetlands were defined as any NWI wetland that was not classified as riverine or lacustrine. Slope wetlands were not included in the GIS classification because there were no characteristics of NWI or collateral datasets that could reliably identify them remotely in GIS.

HGM subclass development

For development of HGM regional subclasses, field assessments of 190 wetlands were conducted throughout the study area. Field assessments of wetland hydrology and landscape condition along with guidance of previous regional guidebooks (e.g., Klimas et al. 2004, Stutheit et al. 2004) were used to develop our subclasses. For assessment of wetland hydrology, hydrological indicators such as high water marks, drift lines and sediment accretion were primarily used. To assess landscape condition, topographic position of the wetland and its proximity to surface waters such as streams, lakes and ponds were determined during assessment.

All NWI polygons in the study area were reclassified into HGM classes using the queries outlined in the previous section. Between 8 and 18 wetlands were selected from each class in three study counties (Okfuskee, Garfield and Logan Counties) for a total of 50 wetlands in each county. Only 49 sites were visited in Okfuskee County due to access restrictions. The program S-Draw (Western Ecosystems Technology, Cheyenne, WY, USA) was run to randomly select sites for field verification. S-Draw uses a probabilistic sampling methodology similar to generalized random-tesselation stratified design (GRTS) (Stevens and Olsen 2004). Each wetland polygon was a discrete sampling unit with their location designated by a polygon centroid (Genet and Olsen 2008). An overdraw of wetland sites per HGM class was selected to account for landowner denial or other access issues. Access permission was obtained from landowners via phone calls or mailings. When access was denied, the next wetland was selected sequentially from the S-Draw output.

Our wetland selection was focused in Okfuskee, Logan and Garfield Counties because they are located in both study ecoregions as well as occurring in the transition region between the two ecoregions (Fig. 1). An additional 41 wetlands (20 in the Cross Timbers and 21 in the Central Great Plains) were selected throughout the study area to identify additional subclasses that may not have been observed among the initial 149 sites. These wetlands were not randomly selected, but were targeted to identify additional slope wetlands and to reduce the chance of visiting human-made ponds, which make up the majority of NWI polygons in the study area.

The assignment of subclasses was an iterative process involving the reassignment of wetlands to new subclasses as field assessments were conducted. After wetland assessments for one county were completed, a list of subclasses was created. Once additional wetlands in a second county were assessed, the preliminary subclasses were reviewed and revisions made based on the additional assessments. This process continued for the third county and then the 41 additional study-area-wide assessments. Once subclasses were finalized, a dichotomous key was developed to facilitate future classification of wetlands (Appendix A).

Field Verification

We field verified the GIS classification of NWI polygons into HGM classes by using the 149 randomly selected wetlands. Field verification was based on USACE wetland delineation procedures, but was designed for a more rapid assessment. General soil texture along with hydric soil indicators were recorded at each site (Environmental Laboratory 1987). Soil cores were taken to a 23 cm depth if hydric indicators were observed and to a 46 cm depth if no hydric indicators were observed in the first 23 cm. The dominant plant species (>20%) were recorded based on visual estimates within each vegetation stratum (submergent, emergent, shrub, or tree), and their wetland indicator status was documented (Godfrey and Wooten 1981, Environmental Laboratory 1987, Haukos and Smith 1997). Standing water, saturated soils, channelization, high water marks and drift lines provided indicators of wetland hydrology during visual assessments (Environmental Laboratory 1987).

Misclassification was attributed to misattribution of NWI polygons, inclusion of upland sites within NWI, and inaccuracies within the GIS queries. Misattribution of NWI polygons included classification errors that resulted from inaccurate Cowardin class assignments. Inclusion of non-target upland sites or commission errors included non-wetland sites that were delineated in NWI and wetlands that were lost after mapping was completed. Inaccuracies within the GIS queries included sites that were placed into the wrong HGM class based on the developed queries.

Accuracy Metric Calculations

Upon completion of field verification, accuracy assessments were conducted using error matrices and standard equations within each of the three study counties (Story and Congalton 1986, Lunetta et al. 1991). User's and producer's accuracy were calculated for each wetland class as well as overall classification accuracy. User's accuracy is a measure of errors of commission or the inclusion of non-target wetlands within a class. User's accuracy was calculated by dividing the total number of correct classifications within a class by the total number of sites classified as a particular class in GIS. Producer's accuracy is a measure of the errors of omission or the exclusion of target wetlands from the correct class. Producer's accuracy was determined by dividing the number of correctly classified sites by the total number of sites field verified as a class (Story and Congalton 1986, Lunetta et al. 1991). The user's accuracy for each ecoregion was calculated using all the sites in the two counties that fall within the ecoregions. Cross Timbers accuracy metrics were derived from Logan County and Okfuskee County calculations, while Central Great Plains accuracy metrics were derived from Logan and Garfield Counties.

After field verification was complete and initial accuracy measures were calculated, we attempted to improve accuracy of classification by using the hydroperiod designation within NWI to distinguish between impounded depressions and lacustrine fringe. All those sites included within the impounded depression class with a permanent hydroperiod were moved to the lacustrine class.

Inventory Calculations

An estimate of the number of wetlands in each HGM class was calculated for both ecoregions. Each individual wetland included in the inventory calculations represents an individual NWI polygon. The minimum estimate of wetlands within each class was calculated by multiplying the user's accuracy of a particular class by the number of wetlands classified within that class using the GIS queries outlined above. The maximum estimate was calculated by including all wetlands within a class that were previously misclassified using GIS queries. For example, wetlands initially classified as depressional but determined to be riverine upon field verification were included in the maximum estimate for riverine wetlands. The number of riverine wetlands initially included in the depressional class in GIS was divided by the total number of field-verified wetlands initially classified as depressional. The quotient was multiplied by the total number of wetlands classified as depressional using GIS queries. The product was then added to the minimum estimate for riverine wetlands. This series of calculations was repeated for riverine wetlands classified as lacustrine fringe and impounded depressional. The same series of calculations was repeated to provide an estimated range of the number of wetlands in all HGM classes and subclasses.

RESULTS

Development of Subclasses

We identified wetlands within four (riverine, depressional, lacustrine fringe, and slope) of the seven HGM national classes. After completion of the field assessments, the four classes were further divided into 16 regional subclasses (Table 1). Riverine wetlands include connected oxbows, beaver complexes, riparian, in-channel, floodplain and floodplain depressions. We divided slope wetlands into headwater slopes and low-gradient slopes. Depressional wetlands were sorted into three natural subclasses and two human-created subclasses. The three natural classes include groundwater depressions, closed surface water depressions and open surface water depressions. Closed and open impounded depressions were included as a subclass because farm ponds which take on the attributes of wetlands are common throughout the study area. Lacustrine fringe wetlands include one natural subclass, disconnected oxbow wetlands, and two subclasses of anthropogenic origin, reservoir fringe and pond fringe.

Accuracy of Classification

Overall accuracy of the classification was 59.8% for the entire study area, 58.6% for the Cross Timbers and 59% for the Central Great Plains (Table 2). User's accuracy in both ecoregions was highest for the lacustrine fringe class with 89% accuracy in the Cross Timbers and 77% in the Central Great Plains. User's accuracy was the lowest for the depressional class in both ecoregions with 12% accuracy in the Cross Timbers and 44% accuracy in the Central Great Plains.

In all three study counties, a total of 60 wetlands were misclassified using GIS for an error rate of 40.2%. Seventeen of the misclassified wetlands were due to misattribution of NWI polygons. All errors were due to the omission of the impounded or excavated designation from farm ponds. Misattribution accounted for 11.4% of the total error of the classification. Fourteen of the misclassified wetlands were due to inclusion of upland sites within NWI. These fourteen sites included both non-wetland sites that were included in NWI mapping and wetlands that have been lost since NWI mapping. Inclusion of upland sites within NWI accounted for 9.4% of the total error of the classification. The remaining

29 misclassified sites were due to errors inherent in the GIS classification methodology. Of these 29 sites, 17 of the errors were due to the inability to distinguish between impounded depression and lacustrine fringe wetlands. Inaccuracies of the GIS queries accounted for 19.4% of the total error (Table 3).

After using hydroperiod to refine queries, accuracy was reduced. All sites initially classified as impounded depression that had permanent hydroperiods attributed in NWI were moved to the lacustrine class. Sites in a basin with more than 2 m of permanent water should be considered lacustrine sites according to HGM (Smith et al. 1995). Therefore, sites with a permanent hydroperiod, as long as the depth requirement is met, should be considered lacustrine. Inclusion of these permanently flooded sites that were initially placed into impounded depressions into the lacustrine class led to the misclassification of two additional study sites (Table 3). Many of the field verified created depressions with a permanent hydroperiod designation in NWI were found to have seasonal or temporary hydroperiods.

HGM Inventory

Riverine wetlands were the dominant natural HGM class (i.e., excludes anthropogenic modified wetlands such as impounded depressions, reservoir fringe and pond fringe wetlands) in both ecoregions (Table 4). The number of riverine wetlands in the Cross Timbers ranged between 5,847 and 7,300 wetlands, while in the Central Great Plains the number of riverine wetlands ranged between 10,368 and 13,002 wetlands. Riparian wetlands were the dominant subclass in both ecoregions with between 3,248 and 3,884 riparian wetlands occurring in the Cross Timbers and between 8,985 and 11,223 riparian wetlands occurring in the Central Great Plains.

DISCUSSION

Overall accuracy of the classification was about 59.8%. Of the 41.2% error, more than half (20.8%) was due to inaccuracies inherent in NWI. The inaccuracies of NWI were attributed to incorrect attribution of NWI polygons (Cowardin and Golet 1995), upland sites that were included in the original maps and wetlands that were lost since the maps were produced. In an attempt to improve the accuracy of the classification, wetlands originally included in the impounded depression class were reclassified into the lacustrine class based on the NWI hydroperiod attributes. After alterations to the queries, accuracy was actually reduced (error rate of 41.6%). Hydroperiod, if accurately designated, should provide a good characteristic to distinguish between impounded depressions and created lacustrine wetlands. However, we found that the NWI hydroperiod designation did not correspond with what was observed in the field for small created ponds. This suggests that reclassification of NWI based on queries of Cowardin class attributes such as hydroperiod has the potential to misclassify a large number of wetland polygons. LLWW may be susceptible to these errors because attribute queries, including those based on water regime modifiers, are primarily used to classify NWI polygons (Tiner 2005; Kevin McGuckin, Virginia Tech University, Blacksburg, VA, USA personal communication).

Cowardin and Golet (1995) acknowledged that placing artificial boundaries between continua of wetland characteristics, such as hydroperiod, was problematic during the development of an NWI classification. The problem can be exacerbated due to the constraints of available remote sensing technology and resulting inability to reliably identify desired classes. Of all the attributes applied to NWI polygons, water regime modifiers, which indicate length of flooding for freshwater systems, are among the most inaccurate (Cowardin and Golet 1995). Graves (1991) found incorrect classification of 13% of NWI wetland area at South Slough in Oregon. The majority of classification errors were due to incorrect vegetation class and water regime modifiers.

While our analysis was conducted by wetland polygon and not area, similar error rates for NWI attribution were found prior to altering the queries. When the analysis relied more heavily on water regime modifiers, the outcome became less reliable. Attribute inaccuracies may not only result from incorrect initial mapping, but could also arise from alterations to wetlands since mapping was completed. For example, sedimentation can rapidly change the hydroperiod of depressional wetlands in agricultural landscapes (Gleason 1998). Luo et al. (1997) found the hydroperiod of playa wetlands in the Southern High Plains of Texas was dramatically shortened due to sedimentation over the last 60 years. In Oklahoma, much of the NWI mapping was completed over 30 years ago (USFWS 2002). Consequently, sedimentation in farm ponds may in part account for the discrepancy between NWI hydroperiod attributes and that observed during field verification.

Inclusion of upland sites also contributed to misclassification. NWI was developed to limit errors of commission or inclusion of upland sites (Tiner 1999). Relatively low errors of commission have been verified by researchers throughout the United States. Kudray and Gale (2000) found that 93.7% of NWI sites field verified in the Upper Peninsula of Michigan were wetlands. In the Blue Ridge Region of Virginia, 91% of field verified palustrine wetlands were delineated as jurisdictional wetlands, leaving a remaining 9% of sites incorrectly included in NWI (Stolt and Baker 1995). The errors of commission

(9.4%) for our study were similar. While 9% error is not egregious, taken in conjunction with attribute inaccuracies, significant error can be introduced into reclassification.

Inclusion of non-target upland sites might not be a result of errors during NWI mapping, but due to wetland loss since mapping was conducted (USFWS 2002). As NWI maps age, the potential for the inclusion of non-target upland area increases as more wetlands are lost due to anthropogenic activities (Dahl 1990, Dahl and Johnson 1991, Luo 1997, Dahl 2006). While overall wetland area has been estimated to have increased between 1998 and 2004, much of the rise in area is from ponds and non-vegetated wetlands (Dahl 2006). During this time period, it has been estimated that 57,500 ha of freshwater emergent and 364,000 ha of freshwater shrub wetlands were lost (Dahl 2006).

More than half of the commission errors in this study were due to sites that were incorrectly classified as riverine in GIS. Wetlands associated with river systems are susceptible to loss from hydrological alterations to stream flow. Along meandering prairie streams, reservoir construction can alter downstream river morphology and flood events (Friedman et al. 1998). Channels become susceptible to incision and widening due to low sediment loads as well as reduced downstream flows (Friedman et al. 1998). Riverine wetland areas typically subjected to frequent floods can become hydrologically separated from the river channel and undergo succession to a less flood tolerant upland community (Nilsson and Berggren 2000).

Natural processes of river migration could also have contributed to errors associated with classifying riverine wetlands during this analysis. Migration of river meanders over the course of 30 years could result in channel movement of hundreds of meters (Shields et al. 2000, Micheli et al. 2004). Agricultural activities adjacent to river systems can increase channel migration, even in the context of reduced flows from impoundments (Micheli et al. 2004). Channel migration of this magnitude could cause NWI polygons to become disconnected from river flow since mapping.

Collateral datasets such as SSURGO that were used in this wetland classification can introduce additional error to classification due to spatial resolution of the maps. SSURGO minimum mapping units can be between 2 to 4 ha (Bowen et al. 2010), but mapping units can be as large as 40 ha in rangeland or forested areas (Tiner 1999). These mapping units can include multiple soil types, which can make it difficult to relate specific soil characteristics to wetlands that occur at smaller spatial scales or span multiple large soil mapping units. As such, the spatial queries used for this analysis may be establishing inaccurate relationships between NWI polygons and the soil characteristics derived from SSURGO.

Any efforts to track spatial changes in wetland function based on the methods provided herein would be diminished by the relatively low accuracy of this classification. Much of the inaccuracy cannot be eliminated because it is due to spatial, temporal and attribute accuracy limitations of NWI. It would not be appropriate to calculate gains and losses of wetland functions on acreages of individual polygons that are misclassified approximately 40% of the time. To our knowledge the attributes applied during LLWW classification have not yet been field verified. We suggest that users of LLWW and other reattribution efforts need to be aware of errors resulting from the limitations of NWI.

The accuracy of this type of assessment potentially could be improved by using more recent wetland inventory maps. For example, the Montana Natural Heritage Program has used 2005 National Agricultural Imagery Program (NAIP) images to develop NWI coverages for Montana (Newlon and Burns 2010a, b). Using newer NWI coverages should reduce errors associated with loss or movement of wetlands over time. Furthermore, mapping at finer spatial scales has the potential to reduce errors of commission and increase classification accuracy. Utilizing finer spatial scales during mapping could potentially allow for better discrimination between upland and wetland areas. While historic NWI maps were created at 1:58,000 or 1:24,000 scale imagery, Montana Natural Heritage Program is using 1:12,000 scale imagery (Newlon and Burns 2010a, b). However, without field verification the degree to which accuracy is improved is unknown.

The wetland inventory for the Cross Timbers and Central Great Plains of Oklahoma provides an estimated range for the number of NWI polygons within each HGM class and subclass. The classification makes use of the high error rate by including originally misclassified polygons in the maximum end of the range. But, one issue that arises from this estimation is that individual wetlands may be comprised of multiple NWI polygons. Because the Cowardin classification separates wetland polygons based on vegetation, an individual wetland with a forested area and an emergent area may be designated by two polygons (Cowardin and Golet 1995). As a result, a wetland that occurs as a connected ecosystem on the landscape may be mapped as multiple polygons.

Despite this potential for overestimation of individual wetlands, our wetland inventory range provides information on the classes and subclasses that can be expected within Oklahoma. Prior to this analysis there was limited understanding of the types of wetlands that exist within the state and the functions they provide. This inventory can help guide future monitoring strategies by aiding in the identification of rare wetland types and development of restoration priorities, as well as serving as a preliminary step in assessing specific functional attributes of wetland subclasses.

While this methodology was tailored to the hydrogeomorphic conditions of central Oklahoma, we believe this GIS approach could provide guidance to others attempting to develop a wetland inventory of HGM classes and subclasses. Those using NWI to create inventories of wetlands by HGM class should be aware of the limitations of the dataset from spatial, attribute and temporal inaccuracies. Because of the potential for misclassification, accuracy assessments are an essential component of developing GIS-based HGM wetland inventories. If accuracy is high, additional tools can be developed to track changes in specific wetland functions. If accuracy is low, applying wetland functions to specific wetland classes and tracking the spatial distribution of those functions may be inappropriate. However, the rate of misclassification can be utilized to provide a range of the number of wetlands within each HGM class and subclass for a study area that still can be used to facilitate future conservation and management priorities.

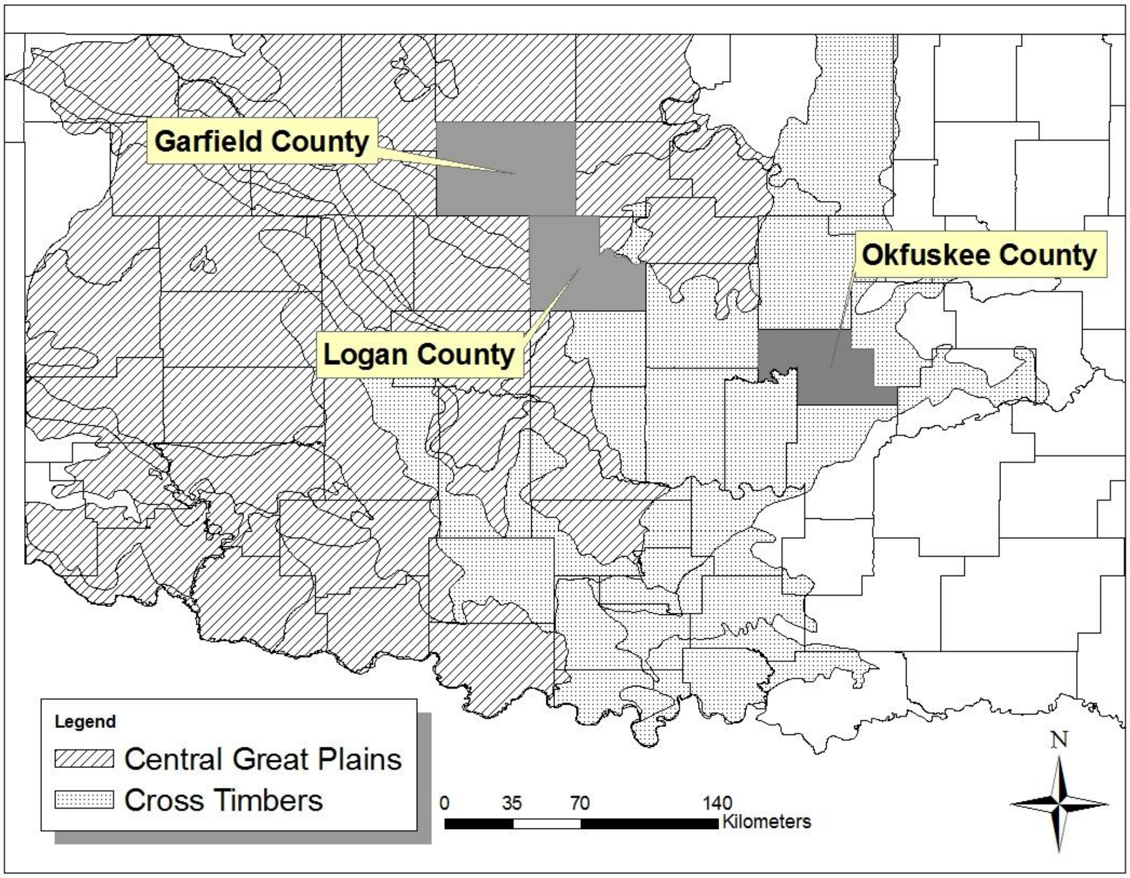

Fig. 1. Map of the study area within the Cross Timbers and Central Great Plains Ecoregions of Oklahoma.

Table 1. List of all HGM subclasses within the Central Great Plains and Cross Timbers Ecoregions of Oklahoma. Each subclass is listed with the typical geomorphic setting, dominant water source and dominant hydrodynamics.

Class	**Subclass**	**Typical Geomorphic setting**	**Dominant Water Sources**	**Dominant Hydrodynamics**
Riverine	connected oxbow	remnant river channel within 5 year floodplain of river/stream	overbank flow, precipitation	unidirectional, vertical
Riverine	beaver complex	area flooded by beaver impounded	overbank flow, precipitation	unidirectional, vertical
Riverine	riparian	natural levee directly adjacent to river/stream	overbank flow, lateral subsurface flow	unidirectional
Riverine	in-channel	sand and gravel bars within river/stream channel	channelized flow	unidirectional
Riverine	floodplain	flat, backwater area within 5 year floodplain of river/stream	overbank flow	unidirectional
Riverine	floodplain depression	basin within 5 year floodplain of river/stream	overbank flow, precipitation	unidirectional, vertical
Slope	headwater slope	steeply sloping area adjacent to low order (Strahler 1-4) streams	groundwater	unidirectional
Slope	low-gradient slope	gradually sloping area near high order (Strahler >4) rivers	groundwater	unidirectional
Depression	groundwater depression	basin in sandy soils where the water table is close to the surface	groundwater	vertical
Depression	open surface water depression	basin with a confining layer and with a water outlet	precipitation, overland flow	vertical, unidirectional
Depression	closed surface water depression	closed contour basin with a confining layer	precipitation, overland flow	vertical
Depression	open impounded depression	basin created by impounding a small stream or draw and with a water outlet	precipitation, overland flow	veridical, unidirectional
Depression	closed impounded depression	basin created by impounding a small stream or draw with no water outlet	precipitation, overland flow	vertical
Lacustrine	disconnected oxbow	remnant river channels outside the 5 year floodplain of a river/stream	precipitation, overland flow	vertical, bidirectional
Lacustrine	reservoir fringe	lakes created by impounding high order, permanent rivers	precipitation, overland flow, channelized flow	bidirectional, vertical
Lacustrine	pond fringe	human impounded basins with at least 2 m of semi-permanent water depth	precipitation, overland flow	vertical

Table 2. GIS Classification Accuracy Error Matrices for classification of four wetland classes in the Cross Timbers and Central Great Plains Ecoregions. Each row represents the class of the wetland after the GIS classification of NWI polygons was completed. Each column represents the number of wetlands field verified as a class. User's accuracy was calculated by dividing the number of correctly field verified sites by all sites initially classified in GIS as that class. Producer's accuracy was calculated by dividing the number of correctly field verified sites by all sites field verified into a class.

Cross Timbers	Depressional	Impounded Depression	Riverine	Lacustrine	Not a Wetland	**Total**	*User's Accuracy*
Depressional	2	8	2	1	4	17	11.8%
Impounded Depression	0	13	0	10	1	24	54.2%
Riverine	2	2	18	1	7	30	60.0%
Lacustrine	0	0	3	25	0	28	89.3%
Total	4	23	23	37	12	99	**Total Accuracy**
Producer's Accuracy	50.0%	56.5%	78.2%	67.6%			**58.6%**
Central Great Plains	Depressional	Impounded Depression	Riverine	Lacustrine	Not a Wetland	**Total**	*User's Accuracy*
Depressional	7	4	1	1	3	16	43.8%
Impounded Depression	0	15	0	10	1	26	57.7%
Riverine	3	5	14	1	5	28	50.0%
Lacustrine	0	3	4	23	0	30	76.7%
Total	10	27	19	35	9	100	**Total Accuracy**
Producer's Accuracy	70.0%	55.6%	73.7%	65.7%			**59.0%**

Table 3. (a) Causes of misclassification of wetlands according to HGM class for wetlands in the Cross Timbers and Central Great Plains Ecoregions in Oklahoma. Columns represent the number of misclassified sites from the GIS queries by class. Percent error is calculated by dividing the number of misclassified sites by all 149 sites (b) Causes of misclassification of HGM class if water permanence is included as a query to separate impounded depressions and lacustrine wetlands. All wetlands that were initially classified as impounded depressions with a permanent hydroperiod attributed in NWI were moved to the lacustrine class

(a)

Cause of classification error	**Depression**	**Impounded Depression**	**Lacustrine**	**Riverine**	**Total**	**% of Error**
Inaccurate attribution of NWI	10	0	0	7	17	11.4
Commission errors and lost wetlands	5	1	0	8	14	9.4
Incorrect placement from query	2	14	8	5	29	19.4
				Total Error		**40.2**

(b)

Cause of classification error	**Depression**	**Impounded Depression**	**Lacustrine**	**Riverine**	**Total**	**% of Error**
Inaccurate attribution of NWI	10	16	0	7	33	22.1
Commission errors and lost wetlands	5	1	0	8	14	9.4
Incorrect placement from query	2	0	8	5	15	10.1
				Total Error		**41.6**

Table 4. The HGM inventory is presented as an estimated range for each subclass by ecoregion, Cross Timbers (CT) and Central Great Plains (CGP). Minimum (min) estimates are calculated by multiplying the user's accuracy for each class by the total number of polygons in that class from GIS queries. The total number in each class is then multiplied by the percentage of field verified sites within that subclass. Maximum (max) estimates include sites misclassified in GIS.

Class*	Subclass	CT Min #	CT Max #	CGP Min #	CGP Max #	Total Min	Total Max
Riverine	Beaver Complex	650	1285	1595	1595	2245	2880
	Riparian	1949	1949	4785	4785	6734	6734
	Riparian/In Channel	1299	1935	3190	4199	4489	6133
	Floodplain	325	507	0	617	325	1124
	Floodplain Depression**	0	0	0	0	0	0
	Connected Oxbow	1624	1624	798	1806	2422	3430
Total		5847	7300	10368	13002	16215	20301
Depressional	SW/open	0	650	0	798	0	1447
	SW/closed	183	183	3084	4679	3267	4862
	GW/closed	0	0	1234	1234	1234	1234
Total		183	833	4318	6711	4501	7543
Lacustrine	Reservoir/Pond Fringe	15249	59030	23199	64337	38448	123367
	Disconnected Oxbow	635	635	0	0	635	635
Total		15884	59665	23199	64337	39083	124002
Impounded Depression	Closed	56374	57338	51641	59527	108015	116865
	Open	0	416	7945	9540	7945	9956
Total		56374	57754	59586	69067	115960	126821
					Total	175758	278667

* Slope wetlands were not included in the GIS classification because of the difficulty in successfully identifying them using spatial queries. As a result, they are under-represented in the table.

** Floodplain depressions were observed in the study area, but not during the accuracy assessment and as a result are under-represented in the table.

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CHAPTER II

ASSESSING VARIABILITY AMONG HYDROGEOMORPHIC RIVERINE WETLAND SUBCLASSES

INTRODUCTION

A number of wetland assessment methods have been developed over the last few decades to support wetland monitoring, management, restoration and regulatory programs (e.g., Adamus et al. 1987, Brinson 1993a, Hruby 1999, Fennessy et al. 2004, Stein 2009b). The efficacy of these programs is in part dependent on the extent to which the chosen assessment method can rapidly, accurately and replicably identify impairment (Fennessy et al. 2004). The hydrogeomorphic approach (HGM), which has been applied regionally throughout the country (Brinson 1993a, Brinson et al. 1995, Smith et al. 1995, Klimas et al. 2004, Stutheit et al. 2004) is a wetland assessment method that identifies impairment through assessments of wetland functions. Preston and Bedford (1988:570) define wetland functions as "the ecosystem properties that derive from the spatially structured interactions among many processes, and the biological, physical and chemical components of the system." These ecosystem properties can include biogeochemical processes such as carbon export and nutrient cycling, hydrological processes such as flood retention and groundwater recharge, and the ability to support biotic communities.

Within HGM, the potential of a wetland to perform a function is determined using assessment models or algorithms that combine a number of ecological measures and indicators called assessment variables. The measures utilized for each functional algorithm are chosen by a group of wetland experts because of the hypothesized relationship between the measure and the resulting function. Directly measuring functions can be time consuming, difficult and counter to the goals of a rapid assessment (Hruby 1999). Ideally, data collection and processing efforts for rapid assessments can be conducted in a day (Fennessy et al. 2007), precluding the direct measurement of processes that occur at longer timescales such as flood frequency or nutrient cycling. Therefore, best professional judgment is usually utilized to determine what variables may indicate the potential for a wetland to perform a specific function (Hruby 1999).

The goal of HGM assessments is to attribute deviation from expected functional condition to anthropogenic disturbance so impaired systems can be identified. Assessing wetland functions is confounded by the natural variability of wetland systems, making it difficult to distinguish between natural variation in function and changes in functions due to degradation (Brinson and Rheinhardt 1996). The term wetland includes ecosystems with vastly different climate, hydrology (e.g., ground water vs. surface water fed) and geomorphology (e.g., basin vs. flat). These factors can drive the structure and processes of wetland systems and therefore can strongly influence wetland function (Brinson 1993a). For example, precipitation-fed bogs have lower primary productivity, lower nutrient loads and lower sediment loads than floodplain wetlands that receive water from overbank flow. The difference is simply because water traveling overland has a much higher potential to carry nutrients and sediments than precipitation (Moore and Bellamy 1974, Brinson 1993b).

The resolution of HGM assessments is improved by comparing wetlands within the context of a classification system which can reduce natural variability among grouped sites. HGM relies on seven national classes that are subdivided into subclasses at the regional level. The classes and subclasses are based on landscape position, water source, and hydrodynamics (Brinson 1993a, Smith et al. 1995). Because hydrology and location on the landscape should influence how wetlands function, sub-classification should reduce natural variability in function for wetlands in the same subclass (Brinson 1993a, Brinson and Rheinhardt 1996). Within each subclass, pristine or least disturbed reference standard sites are identified and assigned the highest level of functional capacity. Deviation of study wetlands from the functional capacity of the reference standard can then be attributed to anthropogenic alteration because natural variability of a function for wetlands within a subclass should be low (Smith et al. 1995).

Developing subclasses can be extremely time consuming, requiring site visits to hundreds of wetlands to ensure that overall variability is observed and appropriately organized into regional subclasses. If national classes are effective in capturing variability among sites for a variety of assessment variables, sub-classification is either unnecessary or inappropriately applied. On the other hand, if variability within a subclass is great and cannot be attributed to disturbance, then subclasses may need to be further subdivided. Validation of the assumption that subclasses are in fact reducing natural variability among wetland sites is crucial to ensuring that assessment models are identifying anthropogenic impacts on function. However, validation is limited in the primary literature (Cole et al. 1997, Shaffer et al. 1999).

During the development of HGM assessment models, data are collected from reference sites that range from highly altered to pristine. By collecting assessment variables under a range of conditions, the relationship between wetland function and disturbance can be established (Brinson 1993a). Trends in how disturbance impacts assessment variables can help to calibrate assessment models and ensure that deviation from pristine functional capacity is related to anthropogenic disturbance. While correlating the functional capacity of wetlands with disturbance factors is essential to developing assessment models that reliably identify impairment, documentation of the methods utilized during the process are limited (Hruby 2001, Hill et al. 2006, Wardrop et al. 2007). If deviation from expected conditions cannot be attributed reliably to anthropogenic disturbance, the value of HGM assessment tools for determining ecosystem health is diminished.

HGM subclasses are applied at a regional scale because factors such as climate and geology can influence the formation and function of wetlands (Brinson 1993a). Defining a reference domain or study area that encompasses a relatively homogenous biogeography can help to reduce the natural variability within a subclass as well (Cole et al. 2002, Merkey 2006). If variability within a subclass results from climatic or geologic factors, the ability of the functional assessment models to detect disturbance will be reduced. Variation in assessment model output from the expected reference standard condition may result more from natural abiotic factors rather than anthropogenic alteration. Alternatively, a reference domain that is constrained unnecessarily will reduce the applicability of the assessment models (Smith et al. 1995). This could potentially be time consuming and a costly mistake in the development of regional wetland programs.

Oklahoma is in the process of developing wetland assessment tools based on a recently developed HGM sub-classification system for the Central Great Plains and Cross Timbers Ecoregions (Chapter 2). Prior to developing functional assessment tools for HGM subclasses, we wanted to validate that natural variability within subclasses was low and determine if we could identify if disturbance leads to quantifiable changes in assessment variables used to calculate function. We collected 21 vegetation physiognomy, soil structure, and water chemistry metrics that represent assessment variables from two riverine wetland subclasses (riparian wetlands and connected oxbows). Redundancy analysis (RDA), principal components analysis (PCA) and forward stepwise regressions were used to assess the following four assumptions of the HGM sub-classification system in central Oklahoma:

1. The subclasses within the study area reduce the variability between sites for a suite of assessment variables.
2. Disturbance within a subclass causes quantifiable patterns of changes among the measured variables.
3. Natural variability within a subclass is low and does not confound identifying relationships between disturbance and assessment variables.
4. The reference domain defined is appropriate for the development of subclasses. The precipitation gradient in the reference domain does not introduce significant variability to the assessment variables.

METHODS

Study Area

The study area encompasses the Central Great Plains and Cross Timbers Ecoregions of Oklahoma (Fig. 1). The Central Great Plains Ecoregion extends from Texas to Nebraska including a large portion of central Oklahoma (73,251 km^2), while the Cross Timbers Ecoregion extends from Texas to southern Kansas and is directly east of the Central Great Plains (Omernik 1987). In Oklahoma, the Cross Timbers encompasses

34,359 km^2. The topography in the study area includes hills, salt plains, karst formations, flats and sand dunes. Elevation ranges from approximately 200 m to approximately 800 m (Woods et al. 2005). In the Central Great Plains Ecoregion, native vegetation is dominated by mixed grass prairie species, such as big bluestem (*Andropogon gerardii*), little blue stem (*Schizachyrium scoparium*), switch grass (*Panicum virgatum*) and indiangrass (*Sorghastrum nugans*), with wooded riparian corridors (Woods et al. 2005).

In the Cross Timbers Ecoregion, where soils were formed from sandstone, post oak (*Quercus stellata*) and blackjack oak (*Quercus marilandica*) woodlands and savannas are common. On finer textured soils derived from limestone and shale, tall grasses form the native vegetation community. Rangeland and cropland are common in both ecoregions (Woods et al. 2005). The study area has an east-west precipitation gradient with the easternmost portion in the Cross Timbers receiving more than 110 cm of average annual rainfall. The westernmost portion of the study area in the Central Great Plains receives less than 60 cm of average annual rainfall (Taylor et al. 1995).

Study Sites

The study included 40 wetlands, of which 20 were riparian wetlands and 20 were connected oxbow wetlands (Fig. 1). In a previous study, we identified riverine wetlands as the dominant natural HGM class and riparian wetlands and connected oxbows as the most common subclasses for the riverine class within the study area (Dvorett unpublished). Riparian wetlands are directly adjacent to stream and river channels. They are inundated through lateral subsurface flow from the channel bank and from overbank flow. Flood events are usually high energy and the water recedes with the river stage. Connected oxbows are remnant river channels within the 5 year floodplain of a stream or river. They receive water from flooding events from the river or stream of origin. Precipitation and groundwater influences may also impact the site hydrology (Dvorett unpublished).

Wetlands within each subclass were selected to cover a broad range of precipitation conditions. Average annual precipitation for each wetland was derived from parameter-elevation regressions on independent slopes model (PRISM) developed by the PRISM group (Taylor et al. 1995). Each site was assigned a Strahler stream order (Strahler 1952) using the National Hydrography Dataset in GIS (ARC View 9.2, ESRI, Redlands, CA). Oxbow wetlands were assigned a stream order based on the order of the stream of origin.

Landscape disturbance was calculated for each wetland using 100 m and 1000 m buffers. Land use and land cover (LULC) data was obtained from the National Land Cover Dataset (NLCD) (USGS Land Cover Institute, Sioux Falls, SD) in GIS. At each buffer width, landscape disturbance was calculated two ways, as a percentage of human altered land and using a land use score. Human altered land use types included developed open space, developed low intensity, developed mid-intensity, developed high intensity, crops, pasture/hay, and barren land.

Land use scores were calculated by multiplying a land use weight coefficient by the percentage of land use within the buffer. The products were summed for all land uses to create a score between 0 (most degraded) and 1 (pristine). Weight coefficients were applied to each land use based on their potential negative impact to the wetland (VIMS 2005). The coefficients applied to each land use were based on best professional judgment and do not represent direct quantifiable impacts to wetland function. For example, developed high intensity was weighted with the lowest possible score of 0, while deciduous forest was weighted with the highest possible score of 1. Table 1 lists coefficients used for each land use category. A list of study sites with all environmental variables is presented in Table 2. Environmental variables are the independent variables used in the multivariate analyses and include precipitation, stream order, and the four landscape disturbance scores.

At each wetland a water sample, soil samples and vegetation physiognomy data were collected. The data collected represents a range of site metrics or assessment variables that have been used in previously developed assessment models (Klimas et al. 2004) or could be used in the development of new assessment models. All data were collected between 10 May and 23 June 2010.

Water chemistry

A water sample was taken at each wetland from the middle of the water column by placing a sealed 1 L polyethylene bottle below the water surface and then removing the lid. At riparian sites where no water was present within the wetland, the water sample was collected from the river channel. At connected oxbows with no water within the wetland, no sample was collected. Conductivity and hardness were both measured on-site with a field water quality meter (EC 400, Extech, Waltham MA) and a Model CM-1 Hach Kit (Hach, Loveland, Colorado), respectively. Most samples had suspended solid concentrations that were too high to accurately measure alkalinity in the field, so samples were placed on ice and taken back to the laboratory, where they were filtered through a 1.6 μm filter to remove suspended solids and then titrated to a pH of 4.8 with H_2SO_4. Connected oxbows with no water on site during the study were assigned median conductivity, alkalinity and hardness values for statistical analyses.

Soil Sampling

At each wetland, four soil samples were collected at 0-5 cm and 15-20 cm depths from the top of the O horizon. Samples were collected along a transect located perpendicularly to the vegetation community zonation, with each sample collected from a distinct morphologic feature (e.g., natural levee, terrace, top of bankfull channel) or vegetative zone. Morphologic and vegetative zones were delineated by visual assessment. The four samples from each depth were composited in a polypropylene bag for particle size distribution and organic content analysis. Samples were homogenized in the laboratory by placing soil in a grinder (Pulverizer Type UA, Bico, Burbank, CA, USA). Particle size distribution was determined by the hydrometer method (Bouyoucos 1962) and organic matter was determined by loss on ignition of oven dried samples at 450 ^{o}C (Storer 1984). The specific soil sample metrics included percent clay, percent sand, percent silt, and percent organic matter. Each metric was determined for surface (0-5 cm depth) and subsurface (15-20 cm depth) sample depths.

Vegetation and Woody Debris Sampling

A point-intercept method was used to collect data that described the vegetation community for each wetland (Goodall 1952). Transects were traversed from the upland edge to the opposite upland edge or from the upland edge to deep water habitat in a direction that traversed all cover types (Smith and Haukos 2002). The upland edge of the wetland was determined based on a visual assessment of vegetation community, hydrology and soils based on USACE delineation protocols (Environmental Laboratory 1987). Deep water habitat was visually assessed as the inundated portion of the wetland where vegetation was precluded.

The first transect within the wetland was randomly assigned using GIS. An assessment region was defined for each wetland by delineating a square with 200 m sides and using a random number generator to select a starting point from 1 to 200 m. The starting point was identified in the field using a global positioning system (GPS) unit. Additional transects were randomly assigned with a minimum distance of 30 m and a maximum distance of 50 m from other transects until at least 100 m of transect was traversed. Transect length varied with wetland width so it was not possible to sample exactly 100 m without stopping mid-transect. In order to correct for this potential sampling bias, 100 m of transect was randomly selected for inclusion in statistical analyses.

Presence/absence of the herbaceous layer, shrub/sapling layer, vine layer, and ground litter was measured every meter. Canopy cover was measured four times along each transect using a convex spherical densiometer. Densiometer readings were taken at intervals determined by dividing the transect length by four. These data were used to calculate percent cover of the herbaceous layer, shrub/sapling layer, vine layer, litter and canopy.

Line-intercept transects, which were placed onto the point-intercept transects, were used for measurement of coarse woody debris (CWD). All down debris > 0.25 cm in diameter and <5 cm in diameter were included for stem counts. Stem count is presented as number of stems per meter of transect. All woody debris >5 cm in diameter on either end of the debris piece were included for CWD volume measurements. The diameter at each end and the total length of each CWD were measured. Volume is presented as cubic meters of CWD per meter of transect.

Ten-meter belt-transects were also located onto point-intercept transects and used to measure tree diameter breast height (DBH) for all trees ≥ 8 cm DBH. From these data, tree density measured as trees/ha, tree basal area measured as m^2/ha and snag density measured as dead trees/ha were calculated.

Statistical Analysis

Redundancy Analysis- Redundancy Analysis (RDA) was used to determine how much variation in soil, vegetation physiognomy, woody debris, and water chemistry metrics was explained by wetland subclass, average annual precipitation, Strahler stream order and landscape disturbance. All RDA analyses were conducted in CANOCO (Plant Research International, Wagingen, The Netherlands). Within CANOCO, all site metrics were centered and standardized by subtracting the mean and dividing by the standard deviation to account for scaling differences among the site metrics. The significance of the analyses was tested using 499 Monte Carlo permutations. Results were considered significant at $\alpha = 0.05$.

Principal Components Analysis- Principal components analysis (PCA) was used to visualize how wetland sites grouped and to determine if additional factors not explicitly used in the RDA could potentially explain the variation within each wetland subclass. All PCA analyses were also conducted in CANOCO, and all variables were centered and standardized by subtracting the mean and dividing by the standard deviation. Prior to conducting PCA analyses, all response variables were tested for normality using the Shapiro-Wilk test (Shapiro and Wilk 1965). Site metrics were transformed where necessary to meet normality assumptions (Table 3).

Ability of subclasses to reduce variance- To determine how much variability in all the site metrics are attributable to the HGM sub-classification, RDA was run on all 40 sites with subclass as the environmental variable. Stream order, precipitation and landscape disturbance were used as co-variables.

Impact of disturbance on site metrics- RDA with landscape disturbance as the environmental variable was run for oxbow and riparian sites separately to assess what, if any, effects disturbance had on the measured site metrics. Stream order and precipitation were included as co-variables.

To determine if landscape disturbance could explain variability in assessment models for specific wetland functions, RDA analyses were conducted on a subset of site metrics. The site metrics chosen for each analysis represent groupings that have been used in assessment models for specific wetland functions (Klimas et al. 2004). This was done for both oxbow and riparian wetlands for nutrient cycling, carbon export, and flood detention functions. Landscape disturbance was the environmental variable and precipitation and stream order were co-variables in the analyses.

The analyses were based on assessment models developed for the regional guidebook for the Delta Region of Arkansas, Lower Mississippi River Alluvial Valley (Klimas et al. 2004). The RDA for nutrient cycling included tree basal area, shrub/sapling cover, herb cover, surface organic matter, CWD volume, CWD stem count and snag density as site metrics. The amount of organic matter in the top 5 cm of soil was used as a surrogate for A horizon biomass that was used by Klimas et al. (2004). For carbon export, RDA was run on litter cover, surface organic matter, CWD volume, CWD stem count, snag density, tree basal area, shrub/sapling cover, and herb cover. For flood detention, CWD volume, herb cover, shrub/sapling cover, tree basal area and tree density were included in the RDA. The Arkansas assessment models for flood detention and carbon export also included flood frequency, but it was omitted from our analyses due to the lack of a reliable methodology to measure flood frequency for the short time period of this study.

Variability within subclasses- Oxbow sites and riparian sites were analyzed separately using RDA with stream order as the environmental variable. These analyses were run to determine if variation within each subclass could be explained based on the size of the stream providing water to the wetland. Landscape disturbance and precipitation were used as co-variables. For oxbow analyses, all water chemistry metrics were removed because seven oxbows did not contain water during the time of sampling. PCA was run separately on oxbow sites and riparian sites to determine if the variation in site metrics could be attributed to any other variable not explicitly measured.

Other environmental factors that were used to interpret the PCA biplots include isolation from source river and groundwater influence for oxbow wetlands and distance to downstream reservoir for riparian sites. Isolation from source river and groundwater influence were not quantified, but based on indicators of hydrology in the field (Table 4). Any wetland where a spring or seep was observed was considered to have groundwater influence. Isolation from a source river was considered possible when the source river was deeply incised, overbank flooding indicators (e.g., drift lines and high water marks) were lacking, or if landowners indicated that flooding was infrequent. Distance to downstream reservoir was assessed using GIS by calculating National Hydrography Dataset (NHD) distance from the wetland to where the river widens into the reservoir (Table 4). This distance does not represent an ecotone between lotic and lentic systems, but is rather an approximation of how close the downstream reservoir is to the study wetland.

Verification of the assessment region- RDA was run for oxbow and riparian sites separately with precipitation as the environmental variable and stream order and landscape disturbance as co-variables. This analysis was used to determine if the reference domain was appropriate or if there was a significant effect on site metrics based on climatic factors. Since there is a precipitation gradient based on longitude within the study area, this analysis was conducted to provide insight into how site metrics change based on geographic location.

Impacts of environmental variables on individual site metrics- Each of the site metrics were analyzed using forward stepwise regressions to determine how much variation could be explained for each site metric. Analyses for oxbow and riparian sites were run separately. Independent variables used in the regressions included average annual precipitation, Strahler stream order, % human altered landscape within a 100 meter buffer, % human altered landscape within a 1000 meter buffer, land use score for a 100 meter buffer, and land use score for a 1000 meter buffer. These analyses, including tests for normality (Shapiro and Wilk 1965) and heteroscedacity, were conducted with SigmaPlot 11 (Systat Software, San Jose, CA, USA). Where necessary, site metrics were transformed to meet normality assumptions (Table 3).

Bonferroni corrections were not included for regressions due to the unacceptably high rate of committing a Type II error ($\beta > 0.999$) (Nakagawa 2004). Moreover, application of Bonferroni corrections can make it difficult to identify significance when studying complex systems like wetland ecosystems (Moran 2003). For our study, the use of regressions is meant to be exploratory to identify potential trends in how site metrics respond to disturbance, precipitation and stream order (Nakagawa 2004).

RESULTS

Ability of subclasses to reduce variance

The first RDA axis was significant ($p = 0.002$), had an eigen value of 0.121, and indicated that subclass explained 14.2% of the total variation among all site metrics (Fig. 2). This indicates that a variable with average explainability along the first axis has at least 14.2% of the variability in its values explained by subclass (Leps and Smilauer 2003). Because subclass was the only environmental variable, only the first axis is related to variation explained by subclass. The second axis explains variability among wetlands that is not related to subclass or any other measured environmental gradient. Soil structure, water chemistry, and vegetation physiognomy were important in explaining the difference between oxbows and riparian sites.

Response variables that were most correlated with the first RDA axis were hardness, % surface clay, % surface sand, canopy cover, tree density and % subsurface clay (Table 5). Other variables that were correlated with the first RDA axis included tree basal area, % subsurface sand, % surface organic matter, conductivity, % shrub/sapling cover and % surface silt. The component loadings along the first RDA axis indicate, that oxbows had finer grained soils that were dominated by clays and silts and contained more organic matter, while riparian sites had coarser grained soils. Conductivity and water hardness were also lower in oxbows than in riparian sites. Oxbows had lower tree density, canopy cover, tree basal area, and shrub/sapling cover when compared to riparian sites.

Impact of disturbance on site metrics

For the oxbow RDA analysis, there was no effect ($p = 0.424$) of landscape disturbance, measured as % human alteration within a 100 meter buffer, on the site metrics. Landscape disturbance also had no significant impact on the suites of site metrics included in the specific functional assessment models for carbon export ($p = 0.774$), nutrient cycling ($p = 0.708$) and flood detention ($p = 0.806$).

There was also no effect ($p = 0.124$) of landscape disturbance, measured as % human alteration within a 100 meter buffer in the RDA analysis for riparian sites. Landscape disturbance did not significantly impact the suite of site metrics included in specific functional assessment models for carbon export ($p = 0.808$), nutrient cycling ($p = 0.718$) and flood detention ($p = 0.382$).

Variation within subclasses

Stream order did not explain variation among site metrics for oxbow sites ($p = 0.876$). The first two principal component axes of the PCA analysis on the oxbow sites explained over 55% of the variance among the site metrics (Fig. 3). The first axis had an eigen value of 0.315 and accounted for 31.5% of the variance among site metrics. This axis seems to be explained in part by water source of the oxbow, with sites with groundwater influences occurring closely together. Sites O1, O3, O9, O12, O13 and O14 were all associated with slope wetlands or springs that fed groundwater into the oxbow basin. The first axis is also partially explained by precipitation. O14 and three additional sites, O16, O17, and O19, had the four lowest average annual precipitation and grouped closely together (Table 2). Groundwater-fed and low precipitation sites generally had coarser soils, less organic matter and more herbaceous cover (Table 5).

The second PCA axis had an eigen value of 0.242 and explained 24.2% of the variance among site metrics. This axis may be explained in part by frequency of inundation events. Oxbows that appeared to be more isolated from the source river and flood events had greater canopy cover, tree density, tree basal area, vine cover and CWD stem count than sites that were more hydrologically linked to the source river (Table 5). Site O4 plotted more closely with the more isolated oxbows, although it appeared to be hydrologically linked with the source river. This outlier may be explained by difficulty in delineating the upland edge of the wetland or due to our inability to correctly assess degree of isolation from the river.

Stream order did not explain variation ($p = 0.146$) among the riparian site metrics in the RDA analysis. But the PCA analysis explained 35.9% of the variation on the first axis, with an eigen value of 0.359 (Fig. 4). This axis seems to be a combination of disturbance and precipitation effects. Six sites (R3, R6, R9, R13 R15 and R19) grouped closely because of higher percentages of clays and silts than the other riparian sites. These sites also had greater litter cover, greater organic matter, particularly in the subsurface and greater CWD volume (Table 5). Site R6 and R13 had large agricultural fields directly adjacent to them and relatively high % human alteration scores within 100 m (28.5% and 66.7%, respectively). Four of the six sites (R3, R9, R15 and R19) had % human alteration scores within the 100 m buffer that ranged from 0 to 16.5%. However, these sites were relatively close to downstream reservoirs and may have experienced altered hydrology as a result. R10 and R11 were relatively close to a downstream reservoir, but they did not plot closely with R3, R9, R15 and R19. These sites were both located along fairly high gradient tributaries (Strahler stream order of 1 and 2, respectively) that flowed into impounded rivers. As a result, elevation differences between the reservoir and the riparian zone may limit hydrological connectivity.

Three sites, R7, R8 and R20, grouped together on the other end of the first axis. These three sites were all along the North Fork of the Red River and had higher conductivity and hardness and coarser grained soils as well as lower canopy cover, tree basal area, and CWD volume (Table 5). While R7, R8, and R20 were among the lowest precipitation sites, other low precipitation sites did not plot closely together on the triplot (Fig. 4). This indicates that while precipitation explains some variability among the site metrics, it may be that the sites along the North Fork of the Red River plot together because of similarity in water chemistry and bed load.

The second PCA axis had an eigen value of 0.165 and explained 16.5% of the variance among site metrics. Given that the first axis explained more than twice the variability of the second, the effect along the second axis is much less important. This axis also appears in part to explain differences among sites due to landscape disturbance and hydrological disturbance from downstream reservoirs. The six riparian sites (R3, R6, R9, R13 R15 and R19) that grouped together due to proximity to impoundment and high % human altered land within a 100 m buffer, plotted on the lower half of the biplot.

Verification of the reference domain

For the RDA analysis, there was no effect ($p = 0.146$) of precipitation on the site metrics for oxbow wetlands, but precipitation did have a significant effect ($p= 0.026$) on the site metrics for riparian sites (Fig. 5). The first axis had an eigen value of 0.109 and explained 12.8% of the variance among site metrics. Differences among sites with different average annual precipitation were explained by vegetation physiognomy, water chemistry and soil texture metrics. Lower precipitation sites had greater herbaceous cover, lower shrub/sapling and vine cover, greater water hardness, and coarser grained soils than higher precipitation sites (Table 5). Canopy cover and snag density were also lower in low precipitation sites.

Impacts of environmental variables on individual site metrics

Forward stepwise regressions were run on all site metrics except water chemistry variables for oxbow sites. Only herbaceous cover, tree density, CWD stem count, and subsurface organic matter varied with the measured environmental variables (Table 6). Herbaceous cover decreased with precipitation (p = 0.03, R^2 = 0.235), while tree density (p = 0.028, R^2 = 0.240) and CWD stem count increased (p = 0.031, R^2 = 0.226). Subsurface organic matter increased with increased % human altered land within 100 m buffer (p = 0.016, R^2 = 0.282).

Stepwise regressions were also conducted on all site metrics for riparian sites. Subsurface sand, surface silt, subsurface silt, subsurface organic matter, herbaceous cover, tree density, conductivity, alkalinity and hardness varied with the measured environmental variables (Table 6). Subsurface sand decreased with % human alteration within a 100 meter buffer (p = 0.017, delta R^2 = 0.237) and precipitation (p = 0.021, delta R^2 = 0.211). Surface silt increased with % human alteration within a 100 meter buffer (p = 0.016, delta R^2 = 0.258) and precipitation (p = 0.05, delta R^2 = 0.154), and subsurface silt increased with % human alteration within 100 meter buffer (p = 0.003, delta R^2 = 0.346) and precipitation (p = 0.02, delta R^2 = 0.183). Organic matter in the subsurface decreased with increasing stream order (p = 0.031, R^2 = 0.234). Herbaceous cover decreased with precipitation (p = 0.05, R^2 = 0.198), and tree density increased with % human alteration within a 1,000 meter buffer ($p = 0.05$, $R^2 = 0.197$). Conductivity increased with stream order ($p < 0.001$, delta $R^2 = 0.431$) and decreased with precipitation ($p = 0.034$, delta $R^2 = 0.135$). Hardness increased with stream order ($p < 0.001$, delta $R^2 = 0.415$) and decreased with precipitation ($p = 0.012$, delta $R^2 = 0.185$), and alkalinity decreased with stream order ($p = 0.054$, $R^2 = 0.191$), albeit marginally significant.

DISCUSSION

Ability of subclasses to reduce variance

The utility of HGM-based assessments of wetland function for monitoring wetland condition depends on the responsiveness of assessment models to disturbance (Hruby 2001, Fennessy et al. 2007). Natural variation among variables can make it difficult to correlate model output with disturbance. Regional sub-classification is employed to reduce variation resulting from natural hydrological and geomorphological factors that impact function and improve the accuracy of assessment models (Brinson and Rheinhardt 1996). However, without determining if subclasses do in fact explain variation among assessment variables, it is difficult to validate the appropriateness of subclasses.

We sought to determine the amount of variation that two riverine subclasses could explain among 21 site metrics and found that subclass designation (oxbow vs. riparian) accounted for 14.2% of the variance. The variance explained by subclasses was related to differences between oxbows and riparian wetlands for vegetation physiognomy, soil structure and water chemistry metrics. Oxbows have finer grained soils comprised of clay and silt particles, which should hold water for longer periods of time than riparian sites that tend to have coarser grained soils. The soil texture of these two subclasses may differ due to water retention time in them. Oxbows have generally deeper basin morphology which facilitates longer water retention that can allow more time for finer grained particles entrained in the floodwater to drop out of suspension (Kirschner et al. 2001). In contrast, riparian sites, due to their generally sloping morphology towards the river channel, should drain fairly quickly as flood waters recede. As a result, only coarser grained sediments would have time to settle out in these systems (Hupp and Bazemore 1993, Brueske and Barrett 1994). Oxbows tend also to have more organic matter than riparian sites likely because of longer hydroperiods resulting from the more clay rich soils. Longer hydroperiods should create anoxic conditions under which organic matter can accumulate in the soil rather than being mineralized by microbial activity (Craft 2001, Collins and Kuehl 2001).

Riparian sites had greater canopy cover, tree density, tree basal area and shrub/sapling cover than oxbow sites. This difference makes sense in the context of expected hydrological differences between wetlands in each subclass. The longer hydroperiod in oxbows should preclude the growth of trees throughout a large portion of the wetland (Craft 2001), whereas riparian sites with rapid flood pulses and relatively short hydroperiods provide good habitat for tree species adapted to high energy flood events (Freidman et al. 1998, Steiger et al. 2005). Herbaceous cover was lower in riparian sites, likely resulting from shade effects from the denser canopy. Furthermore, the more proximate location of the riparian sites to the stream channel relative to oxbow wetlands likely causes higher energy and more frequent flood events that can eliminate the herbaceous cover during the growing season (Steiger et al. 2005). Relatively higher levels of herbaceous cover in oxbows could also explain the higher levels of organic matter in these systems, as herbaceous material has a higher net primary productivity than woody vegetation (Craft 2001).

Conductivity and hardness were lower in oxbows, likely due to the contribution of direct precipitation to the basin and groundwater influences. Riparian sites tended to have greater conductivity and hardness, likely due to the accumulation of salts during overland flow to the river channel (Harrel and Dorris 1968). There is also a temporal component to conductivity and alkalinity measurements in basins. This study was conducted during the spring and early summer, during high rainfall events. As such, evaporative processes later in the season may reverse the trend that was observed for conductivity and hardness. Evaporation of surface waters concentrates ions in closed basins such as oxbows, which likely have higher conductivity and hardness at the end of the summer (Liebowitz and Vining 2003).

We found sub-classification to be useful in explaining variability for a wide range of site metrics among oxbow and riparian wetlands in our study area. There is limited validation in the primary literature that classification indeed reduces variation. However, those that have validated classification have found that wetlands of different classes and subclasses have different hydrologic attributes. Shaffer et al. (1999) found hydrologic differences in mean water level, range of water level, duration of inundation, and extent of inundation among HGM slope, riverine and depressional classes. Cole et al. (1997) found differences among four HGM subclasses for hydrologic and water chemistry parameters in central Pennsylvania. Wetlands of different subclasses had different median water depth, pH, and specific conductance. Our data confirm that applying regional subclasses can in fact reduce natural variability of wetlands. However, 85.8% of the overall variability of site metrics was unaccounted for by subclass alone. This variability could potentially be associated with other natural factors not considered in the subclasses or from anthropogenic disturbance effects.

Impact of disturbance on site metrics

If the variability for site metrics within a subclass is related to natural variability and not disturbance, the ability to relate assessment model output with impairment becomes impossible. We assessed how two landscape disturbance factors at two different buffer scales impacted site metrics to determine if significant relationships could be observed. If landscape disturbance significantly impacted the ability of wetlands within a subclass to perform a function, sites with high landscape disturbance scores should plot closely together in the RDA analyses for specific functions (nutrient cycling, carbon export and flood detention). In these analyses there was no significant impact of disturbance on site metrics. These results can be attributed to two causes; the disturbance metrics used do not control variability among the measured site metrics or the natural variability within the subclass is too great to identify patterns of disturbance on site metrics.

There is some indication that the measured landscape disturbance factors do influence wetland structure. When site metrics were analyzed individually, there were some significant trends. Among riparian sites, silt in both the surface and subsurface increased with disturbance. The higher amount of silt in disturbed sites may be a result of sedimentation from the surrounding landscape due to increased surface runoff from impervious surfaces and from agricultural activities (Daniels and Gilliam 1996, Mensing et al. 1998). For oxbow sites, organic matter in the subsurface significantly increased with landscape disturbance, which may indicate increased productivity of wetlands due to nutrient rich runoff (Mensing et al. 1998). While eutrophication can also enhance decomposition rates in wetlands, it tends to increase net primary productivity to a greater extent, which fosters organic matter accumulation (Craft 2001).

Identifying patterns of disturbance on one or two assessment variables for a subclass is not sufficient to develop an array of robust assessment models that can relate functional capacity to disturbance. There has been limited research conducted attempting to relate wetland function with anthropogenic disturbance. Hruby (2001) found that for two depressional and two riverine HGM subclasses in western Washington, disturbance did not correlate well with the potential of a wetland to perform functions. As a result, he concluded the outputs of the HGM assessment models would not be reliable indicators of the ecological health or degree of disturbance of the study wetlands. However, without verifying that the subclasses limit natural variability the conclusion may be hasty. The inability to relate disturbance to function may be caused by subclass designations that are cast too broadly and introduce too much natural variability.

Variability within subclasses

While there is some indication that subclasses can reduce natural variability for hydrologic metrics (Cole et al. 1997) as well as a variety of other metrics used in this analysis, variability is not necessarily sufficiently reduced to develop assessment models that are responsive to disturbance effects. To appropriately calibrate assessment models within subclasses it is important to have an understanding of natural variability for variables included in those models. To our knowledge, no study has attempted to quantify natural variation for a variety of site metrics within designated HGM subclasses. Wetland classification will always rely on best professional judgment because boundaries between ecosystems are artificial constructs used to differentiate systems along continua (Brinson 1993b, Cowardin and Golet 1995). Nevertheless, quantifying variation would allow managers to set criteria to determine when subclasses are acceptable or need refinement.

We wanted to identify any additional sources of natural variation within subclasses that may be confounding the relationship between landscape disturbance and site metrics. Stream order was strongly correlated with water conductivity and hardness at riparian sites. These changes in conductivity and hardness are potentially a result of larger streams draining a greater area and accumulating more salts from tributaries and a larger watershed (Harrel and Dorris 1968). Increasing stream order also decreased organic matter in the subsurface of the soil. Along riparian zones of small streams, the elevation of the riparian zone may only be slightly higher than the stream channel. As a result, lateral flow through stream banks may be more important at the subsurface depth (15-20 cm) included in this analysis for smaller streams. There may be more persistent anoxic conditions from lateral flow in the subsurface of riparian zones on small streams, allowing for the accumulation of organic matter (Jones and Holmes 1996, Craft 2001).

Other regional applications of HGM have used stream order to designate differences among floodplain and riparian riverine systems (Cole 1997, Klimas 2004). These results indicate that additional subdivisions within the riparian subclass based on stream order would further reduce variability. However, there is a tradeoff between the accuracy gained by dividing wetland subclasses into smaller groupings and the time needed to develop additional assessment tools. Additional subclasses require additional reference site identification, additional assessment model development, and more time spent calibrating models (Brinson and Rheinhardt 1996). In cases such as this, where only a few assessment variables significantly differ for an environmental variable, it may be just as effective to eliminate those variables from assessment models. This would allow for improved accuracy of models without the associated costs of further sub-classification.

There was no significant effect of stream order of the origin river on site metrics for oxbow wetlands. This may in part be a function of limited variability of stream order among study sites. Seventeen out of 20 study oxbows were associated with fifth and sixth order rivers. We were unable to find many oxbows that formed from streams smaller than fifth order. This limited distribution of oxbow systems for Strahler stream order may be an artifact of oxbow ontogeny, with low gradient rivers more prone to form oxbows. However, little has been documented correlating Strahler stream order with oxbow formation.

In addition to testing hypotheses on how site metrics respond to pre-determined environmental gradients, it may be useful to determine how wetlands group without being constrained by variables set *a priori*. PCA can help identify additional factors important in controlling variability within a subclass that were not included for hypothesis testing. For oxbows, the PCA analysis indicated that groundwater influence and degree of isolation from overbank flood events can influence site metrics. Since the variability among site metrics is not constrained by any environmental variable, it is not possible to directly quantify how much variability environmental factors explain. But the amount of variability explained by the axis and how well the sites group on the biplot can help determine how important the effect of the environmental variable is. For oxbows, the first axis explained close to a third of the variation. All of the sites with observed groundwater influence plotted separately from the remainder of the sites, with the exception of the three lowest precipitation sites. This indicates that groundwater influence may be an important driver of oxbow structure, process and function.

Water source is one of the parameters that is often used to create regional subclasses (Brinson 1993a, Smith et al. 1995, Klimas 2004). However, there are no guidelines on which classification parameters to apply when developing subclasses. Oxbow wetlands all form as river channel cut-offs but their hydrology can vary based on age, degree of isolation from the river channel and degree of connection to groundwater (Bornette et al. 1994). Directly quantifying variability for wetlands that exist under a variety of environmental conditions can help identify when HGM classification parameters should be applied to create regional subclasses. Oxbows with observed groundwater influences, even though they retained hydrologic connectivity to the river of origin, had sandier soils and less organic matter than oxbows with hydrology driven solely by surface water. These structural differences can represent differences in the ability of wetlands to perform functions as measured by assessment models. Soil texture and organic matter have been used in assessment models for functions such as aquifer recharge, surface water storage, and organic carbon export (Brinson et al. 1995).

In the PCA for oxbow wetlands, four sites that appeared to be more isolated from overbank flooding plotted fairly close together. Some of the oxbows included in this study may function more as depressional wetlands when flood frequency is reduced beyond the 5 year recurrence interval and precipitation is the dominant hydrologic force (Smith et al. 1995, Klimas 2004). HGM uses a 5 year flood return interval to divide riverine wetlands from other systems. As a result, it may be necessary to place oxbows that are more isolated from stream flow into a new subclass, or potentially into a different class.

Unconstrained ordination offers insights into ecological patterns not conceived prior to data collection and can be used as part of an iterative process to refine subclasses and reference domains. Additional data collection efforts from groundwater-fed, surface water-fed, isolated and connected oxbows could be re-run using RDA to see if a significant amount of variation could be explained by creating additional oxbow subclasses. The utility of PCA can be enhanced by collecting ancillary hydrologic and geomorphologic data in the field so trends in the biplot can be attributed to environmental factors. While PCA has typically been applied to assess how environmental gradients affect ecological communities (Gauch 1982), there is no reason why this type of analysis cannot be used to determine how environmental factors impact the structure of ecosystems.

Verification of the reference domain

Natural variability can be introduced to subclasses by broadly defining a reference domain that includes an area with variable climate. This natural variability associated with climate could also serve to confound the relationship between landscape disturbance and assessment variables. To assess if variability among site metrics was significantly explained by climatic variability within the study area, the effects of precipitation were evaluated. A variety of vegetation, soil and water chemistry variables were affected by precipitation, indicating that variability in climate across the reference domain is introducing natural variability within subclasses and may be obscuring the relationship between disturbance and function.

The accuracy of functional assessments could once again be improved by further subdividing the subclasses based on average annual rainfall to reduce natural variability among site metrics. However, there are tradeoffs between narrowly and broadly defining a reference domain. A small reference domain is more likely to reduce natural variability among wetlands that result from geographic and climatic factors. A large reference domain is more broadly applicable but wetlands within a subclass may have significant variation for a variety of site metrics and function (Smith et al. 1995). Smith et al. (1995) recommend using programmatic objectives to decide on how broadly to define a reference domain.

There is also the question as to where breaks in the reference domain should be placed in order to reduce natural variability associated with climate. The PCA analysis provided insight into this question because within the oxbow subclass the four lowest precipitation sites plotted fairly close together and more closely with the groundwater sites. This indicates that there may be an important break in the reference domain close to the western edge of the study area. Three site metrics had significant variation explained by precipitation using forward stepwise regressions, but this effect may be driven by those few lowest precipitation sites. This relationship could be clarified by collecting data from more oxbows at the western edge of the study area, and then using RDA to verify that the trend seen for the four lowest precipitation oxbows is important in explaining variation among site metrics.

A few other studies have attempted to determine if site metrics for specific classes of wetlands vary based on geographic location. Those that have evaluated reference domain effects on site metrics have found that wetlands can vary based on climate and geography. Merkey (2006) found that three HGM classes of wetlands had differences in hydrology and water chemistry for different ecoregions in Michigan. Cole et al. (1997)

assessed the hydrology of three HGM subclasses in Oregon and Pennsylvania and found that wetlands within a subclass significantly differed by region.

Including Additional Disturbance metrics

A high degree of natural variability within a subclass is a potential explanation for why landscape disturbance metrics were not well correlated with site metrics. Alternatively, site metrics may be more responsive to on-site disturbance factors, or severe landscape degradation. Developing correlations between disturbance and assessment variables could be improved by expanding collection of disturbance metrics to include on-site disturbance such as hydrologic alterations, invasive species colonization or any other factors predicted to be regionally important.

For riparian sites, it may be important to consider proximity of downstream reservoir as a disturbance factor. When PCA was run on riparian sites, several wetlands that were located close to downstream reservoirs plotted with two sites that had high landscape disturbance scores. These sites near reservoirs may have altered flood regimes, which can significantly affect sedimentation, organic matter input and accumulation, CWD dynamics and vegetation structure within riparian zones (Harmon et al. 1986, Craft 2001, Collins and Kuehl 2001, Steiger et al. 2005). So, altered flood dynamics can have impacts on the measured assessment variables. Observing how site metrics respond to a variety of disturbance metrics using RDA and forward stepwise regressions may reveal patterns missed by solely using landscape disturbance.

Furthermore, natural disturbance during flood events is an important driver of the structure and processes of riverine wetlands (Ward 1998). Floods can influence soil texture, nutrient loads, organic matter composition, coarse woody debris composition and the biotic community (Junk 1989, Harmon et al. 2004). As a result, it is important to consider temporal variability when measuring site metrics from wetlands that are hydrologically linked to rivers (Ward 1998, Smith et al. 2008). Evaluating a wetland directly before and after a flood event may yield significantly different results for a variety of site metrics. If flood events control more variability in a site metric than landscape disturbance, the impact of landscape disturbance may be masked. For example, a flood event in a riparian zone adjacent to a river with a sandy bed load may mask the effect of siltation resulting from landscape alteration. The influence of natural flood events may be driving the observed soil texture at riverine wetlands more than siltation from surrounding agricultural land. When soil texture is then measured, the relationship between disturbance from agricultural siltation is confounded by the input of coarse grained sediments during natural flood events. The inclusion of factors such as duration since last flood and flood frequency into the RDA analysis and regressions could help indicate if natural disturbance impacts site metrics more than anthropogenic disturbance.

Importance of Validation

Validation of HGM classification and assessment models is limited in the primary literature (Cole et al. 1997, Shaffer et al. 1999, Hruby 2001, Wakeley and Smith 2001, Cole et al. 2002, Merkey 2006, Wardrop 2007). We provide a methodology for testing the utility of the subclasses in reducing natural variation between wetlands. Wetland scientists and managers can use RDA and PCA to ensure that the developed subclasses are appropriate and assess if natural variability within a subclass may be confounding the relationship between disturbance and assessment variables. Since developers of regional HGM programs should be collecting data during the model calibration phase, this type of multivariate analysis could be conducted prior to formalization of models with little extra data collection.

This type of analysis could be expanded to include all subclasses within the reference domain. Natural variability within each subclass can be evaluated and site metrics correlated with disturbance. Assessment models for each subclass could then be developed to include only the assessment variables that vary with disturbance. Variables with variation that cannot be explained by disturbance should be omitted from inclusion in assessment models. This would greatly improve the ability of the assessment models to identify reduction of function from anthropogenic disturbance. Furthermore, the site metrics collected for this analysis are not meant to be exhaustive. Collecting additional site metrics including those based on site hydrology could help establish additional relationships between disturbance and the structure and process of wetlands.

We found a limited relationship between landscape disturbance metrics and the measured assessment variables within each subclass. If functional assessment models were developed for the riparian and connected oxbow subclasses, there would be little evidence that model output was in any way related to impairment from anthropogenic disturbance. Without establishing reliable trends between disturbance and assessment variables, HGM assessment tools cannot identify system health, and their value in wetland monitoring is severely reduced. The natural variability of assessment variables within a subclass may be masking subtle landscape disturbance effects, such that models can only reliably identify large-scale on-site disturbances or severe landscape degradation. The inability to relate disturbance and assessment variables may not be just confined to riverine wetlands in Oklahoma (Hruby 2001), and those who develop HGM models without calibration need to be aware that assessment model output may not indicate wetland health.

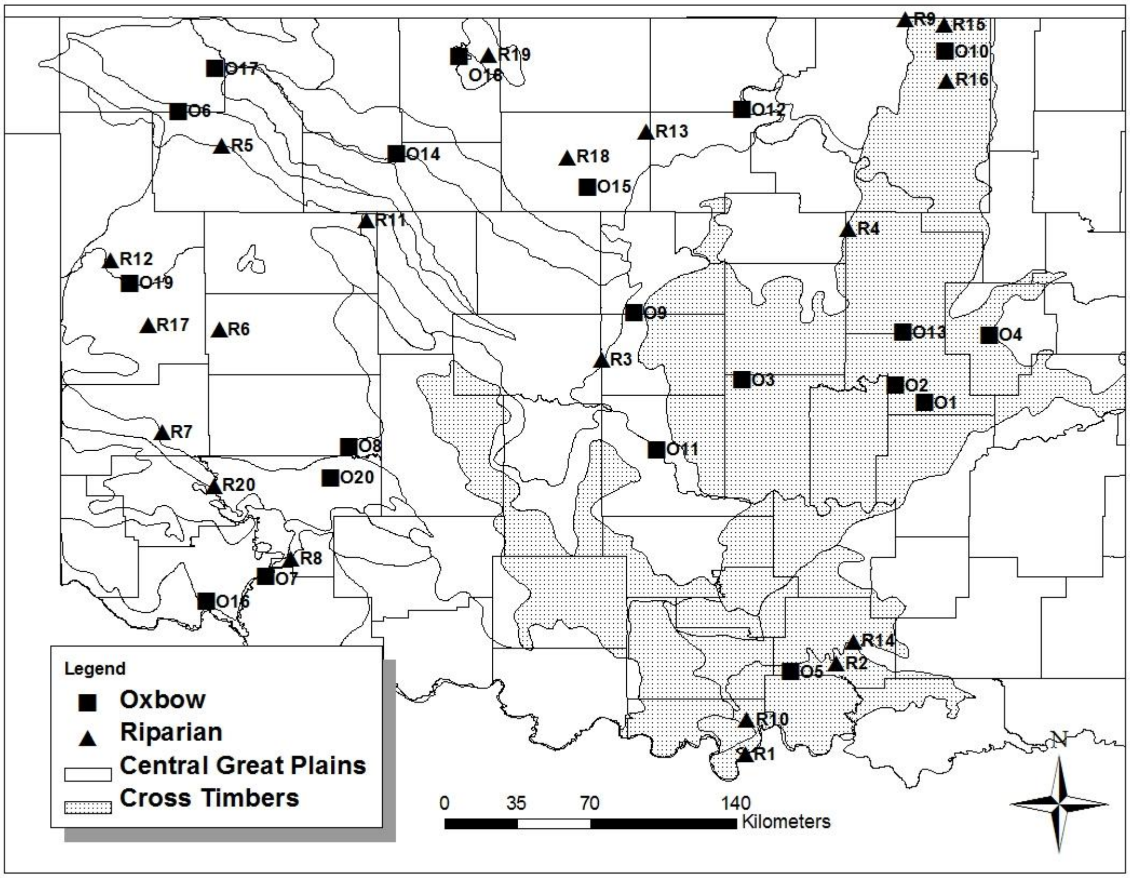

Fig 1. Location of study sites in the Cross Timbers and Central Great Plains Ecoregions of Oklahoma. Oxbow sites are denoted with solid squares and riparian wetlands are denoted with solid triangles.

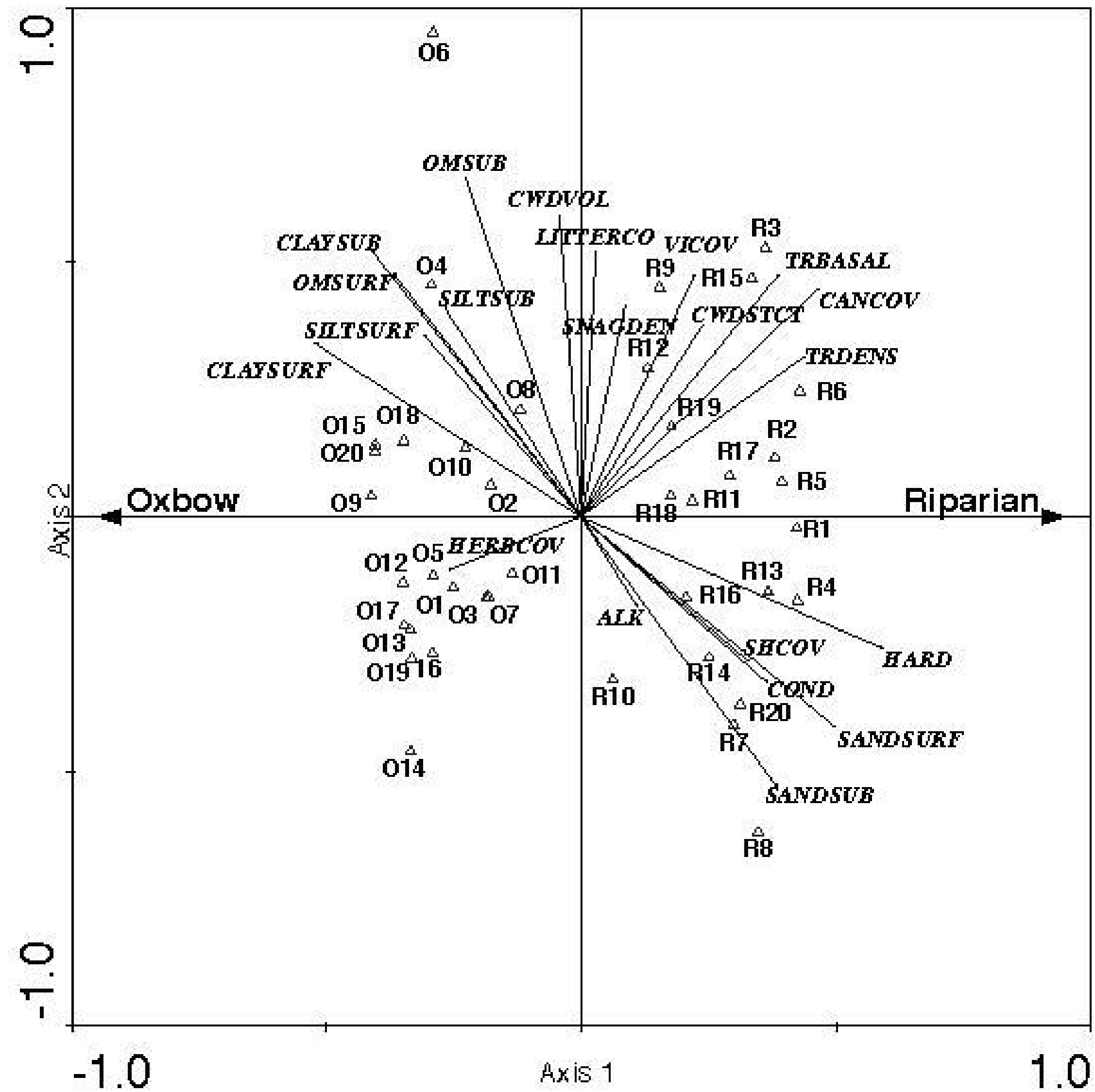

Fig 2. Triplot of first and second RDA axes for 20 riparian (R1-R20) and 20 oxbow (O1-O20) wetlands in the Cross Timbers and Central Great Plains Ecoregions of Oklahoma. Subclass (Oxbow and Riparian) is the environmental variable and precipitation, stream order and % human alteration within a 100 meter buffer were included as co-variables. Response variables were collected at all sites between 10 May and 23 June, 2010 and include % surface clay (CLAYSURF), % subsurface clay (CLAYSUB), % surface silt (SILTSURF), % subsurface silt (SILTSUB), % surface sand (SANDSURF), % subsurface sand (SANDSUB), % surface organic matter (OMSURF), % subsurface organic matter (OMSUB), coarse woody debris volume (CWDVOL), coarse woody debris stem count (CWDSTCT), litter cover (LITTERCO), snag density (SNAGDEN), herbaceous cover (HERBCOV), vine cover (VICOV), shrub/sapling cover (SHCOV), canopy cover (CANCOV), tree density (TRDEN), tree basal area (TRBASAL), water hardness (HARD), water conductivity (COND), and water alkalinity (ALK).

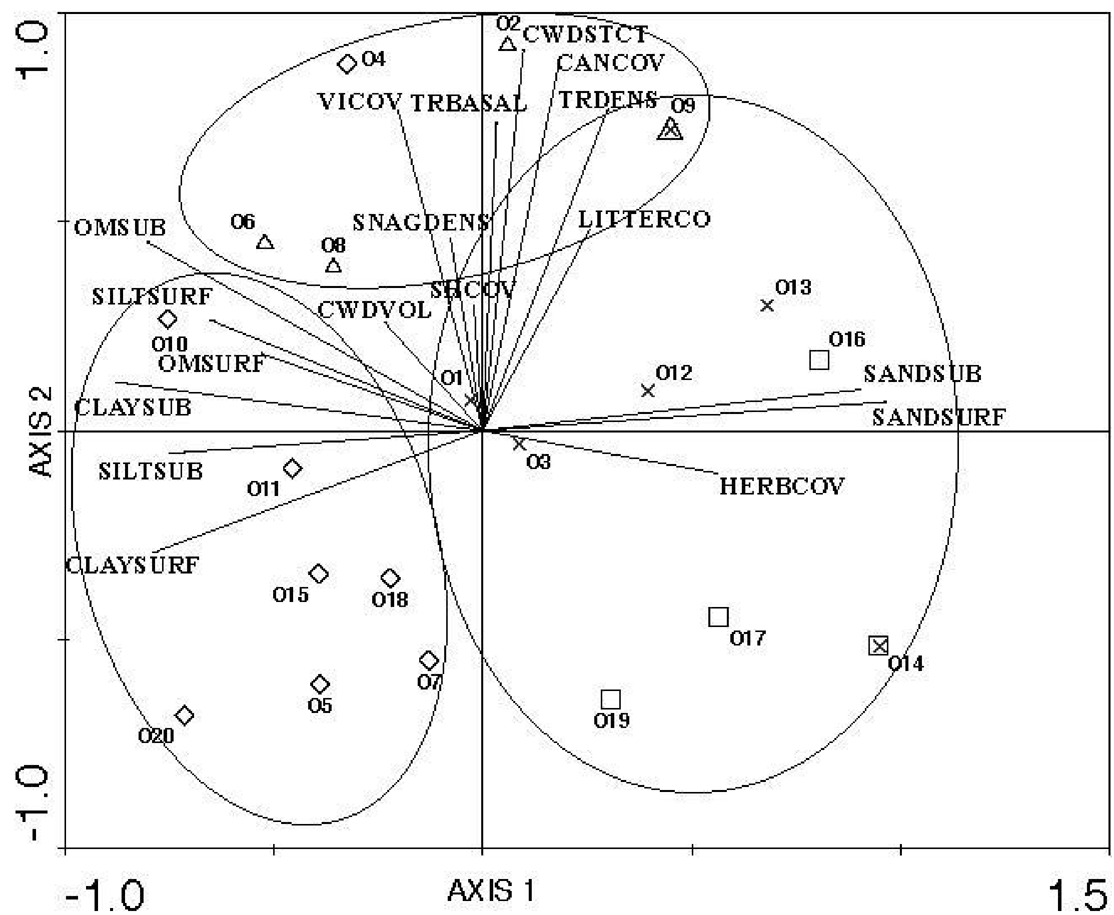

Fig 3. Biplot of first two PCA axes for oxbow sites (O1-20) in the Cross Timbers and Central Great Plains Ecoregions of central Oklahoma. X's are sites with observed groundwater influence. Diamonds are sites that are surface water fed which likely flood regularly from the stream of origin. Triangles are oxbows that are more disconnected from river flood events. Squares are the four oxbows with the lowest average annual precipitation. Circles denote groups of oxbows with similar characteristics. The rightmost group includes oxbows with groundwater influence and the four lowest precipitation sites. The leftmost group includes oxbows with only surface water influences. The topmost group includes oxbows that are relatively isolated from river flood events. Response variables were collected at all sites between 10 May and 23 June, 2010 and include % surface clay (CLAYSURF), % subsurface clay (CLAYSUB), % surface silt (SILTSURF), % subsurface silt (SILTSUB), % surface sand (SANDSURF), % subsurface sand (SANDSUB), % surface organic matter (OMSURF), % subsurface organic matter (OMSUB), coarse woody debris volume (CWDVOL), coarse woody debris stem count (CWDSTCT), litter cover (LITTERCO), snag density (SNAGDEN), herbaceous cover (HERBCOV), vine cover (VICOV), shrub/sapling cover (SHCOV), canopy cover (CANCOV), tree density (TRDEN) and tree basal area (TRBASAL).

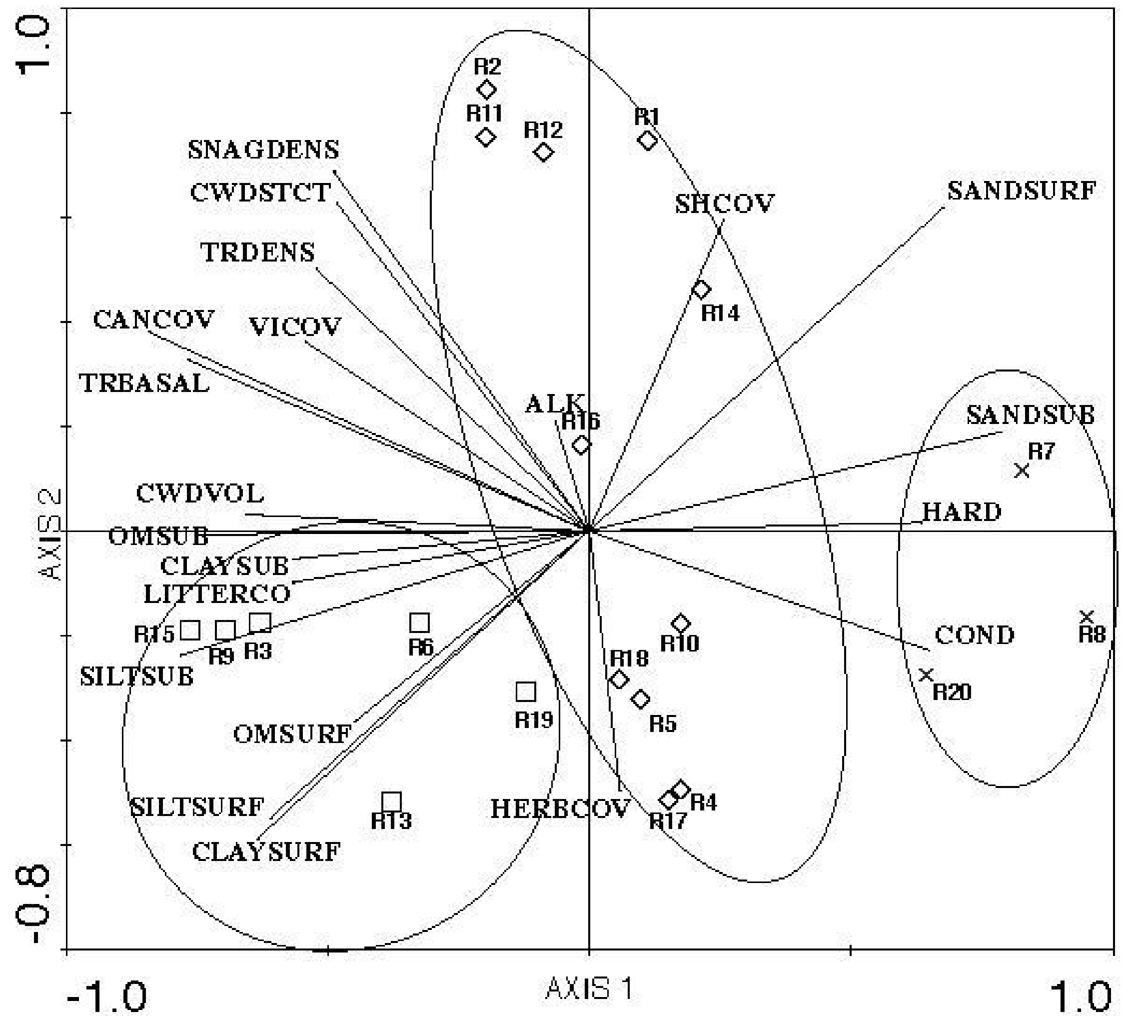

Fig 4: Biplot of first two PCA axes for riparian sites (R1-R20) in the Cross Timbers and Central Great Plains of central Oklahoma. X's are sites along the North Fork of the Red River. Squares are sites with relatively high landscape disturbance and sites that are close to reservoirs. Diamonds are all remaining riparian sites. Circles denote groups or riparian wetlands with similar characteristics. The rightmost group includes three wetlands on the North Fork of the Red River. The leftmost group includes sites with high landscape disturbance and sites close to a reservoir. The middle group includes all other sites. Response variables were collected at all sites between 10 May and 23 June, 2010 and include % surface clay (CLAYSURF), % subsurface clay (CLAYSUB), % surface silt (SILTSURF), % subsurface silt (SILTSUB), % surface sand (SANDSURF), % subsurface sand (SANDSUB), % surface organic matter (OMSURF), % subsurface organic matter (OMSUB), coarse woody debris volume (CWDVOL), coarse woody debris stem count (CWDSTCT), litter cover (LITTERCO), snag density (SNAGDEN), herbaceous cover (HERBCOV), vine cover (VICOV), shrub/sapling cover (SHCOV), canopy cover (CANCOV), tree density (TRDEN), tree basal area (TRBASAL), water hardness (HARD), water conductivity (COND), water alkalinity (ALK).

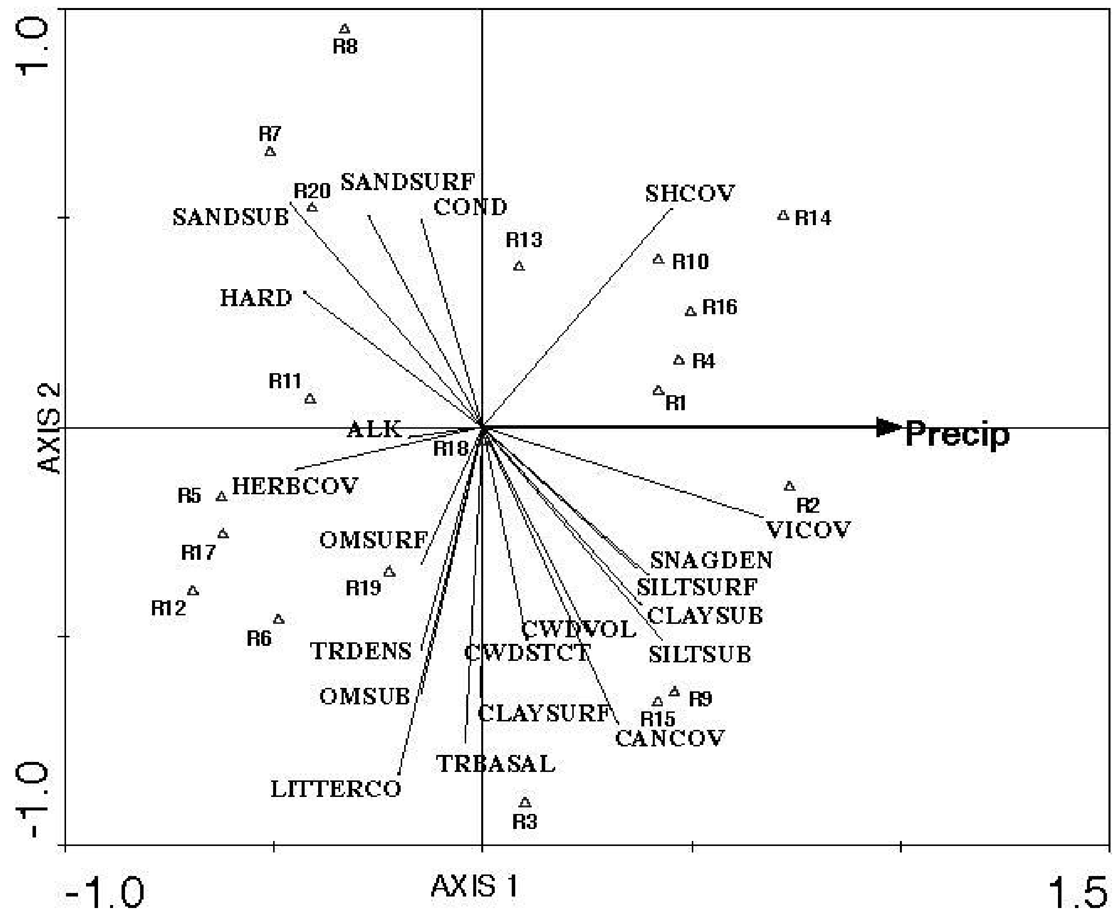

Fig 5. Triplot of first two RDA axes for riparian sites (R1-R20) in the Cross Timbers and Central Great Plains of central Oklahoma. Average annual precipitation is the environmental variable and stream order and % human alteration within 100 meter buffer were included as co-variables. Response variables were collected at all sites between 10 May and 23 June, 2010 and include % surface clay (CLAYSURF), % subsurface clay (CLAYSUB), % surface silt (SILTSURF), % subsurface silt (SILTSUB), % surface sand (SANDSURF), % subsurface sand (SANDSUB), % surface organic matter (OMSURF), % subsurface organic matter (OMSUB), coarse woody debris volume (CWDVOL), coarse woody debris stem count (CWDSTCT), litter cover (LITTERCO), snag density (SNAGDEN), herbaceous cover (HERBCOV), vine cover (VICOV), shrub/sapling cover (SHCOV), canopy cover (CANCOV), tree density (TRDEN), tree basal area (TRBASAL), water hardness (HARD), water conductivity (COND), water alkalinity (ALK).

Table 1. List of land use coefficients for calculating landscape disturbance scores for oxbow and riparian wetlands in central Oklahoma. Land use classes were obtained from the National Land Cover Dataset (NLCD) in GIS.

NLCD class	Land Use Score
Open Water	1.0
Developed, Open Space	0.7
Developed, Low Intensity	0.2
Developed, Medium Intensity	0.0
Developed, High Intensity	0.0
Barren Land	0.5
Deciduous Forest	1.0
Evergreen Forest	1.0
Mixed Forest	1.0
Scrub/Shrub	1.0
Grassland/Herbaceous	1.0
Pasture/Hay	0.7
Cultivated Crops	0.3
Woody Wetlands	1.0
Emergent Herbaceous Wetlands	1.0

Table 2. Environmental variables for oxbow (O1-O20) and riparian (R1-R20) sites from the Cross Timbers and Central Great Plains of central Oklahoma. Variables include average annual precipitation (PRECIP), Strahler stream order (STORD) and four landscape disturbance scores. Landscape disturbance scores were calculated by applying the landscape coefficients in Table 1 to 100 m (LD100m) and 1000 m (LD1000m) buffers. Percent human alteration was calculated by summing all of the human impacted land use in 100 m (%ALT100m) and 1000 m (%ALT1000m) buffers.

SITEID	PRECIP (cm)	ST.ORD.	LD1000m	LD100m	%Alt100m	%Alt1000m
O1	108.1	6	0.94	0.95	15	18.9
O2	106.0	6	0.79	0.87	40.7	56.9
O3	98.9	6	0.80	0.84	37.2	37.7
O4	110.1	6	0.96	1.00	0.0	13.6
O5	106.9	6	0.85	0.99	1.1	30.4
O6	63.7	6	1.00	1.00	0.0	1.51
O7	74.7	6	0.70	0.74	38.2	46.2
O8	78.0	6	0.48	0.59	60.0	75.8
O9	90.1	3	0.93	0.88	17.6	14.4
O10	98.0	5	0.75	0.71	42.7	40.3
O11	97.5	5	0.73	0.75	75.0	45.9
O12	93.8	5	0.77	1.00	0.0	36
O13	105.5	5	1.00	0.98	5.1	1.3

O14	73.4	5	0.91	0.98	4.5	14.7
O15	86.1	3	0.52	0.90	20.2	71.3
O16	73.7	5	0.74	0.86	19.7	39
O17	65.5	5	0.99	1.00	0.0	3.4
O18	78.1	5	0.95	1.00	0.0	4.0
O19	64.7	5	0.95	0.97	5.0	9.5
O20	76.7	4	1.00	1.00	0.0	0.0
R1	99.4	7	0.83	0.95	7.3	25.2
R2	109.8	6	0.89	0.96	12.6	24.8
R3	89.3	6	0.72	0.98	5.5	41.5
R4	101.0	7	0.96	0.97	7.8	11.5
R5	66.1	6	0.89	0.94	16.2	21.3
R6	70.7	6	0.49	0.8	28.5	74.9
R7	69.8	5	0.99	0.99	3.1	2.7
R8	75.3	6	0.89	1.00	0.0	17.9
R9	101.7	2	0.86	0.96	12.2	25.3
R10	100.4	1	1.00	1.00	0.0	0.0
R11	74.0	2	0.77	0.76	34.0	34.9
R12	64.4	2	0.96	0.99	3.0	11.7
R13	90.5	3	0.53	0.56	66.7	70.4
R14	109.6	4	0.91	0.98	7.0	28.7
R15	99.9	5	0.87	0.94	16.1	24.6
R16	102.7	3	0.97	0.97	10.8	10
R17	66.1	5	0.95	1.00	0.0	8.7
R18	86.6	3	0.58	1.00	0.0	54.5
R19	79.3	3	0.97	1.00	0.0	6.0
R20	73.1	5	0.82	0.95	7.7	26.7

Table 3. Response variables that were transformed to meet normality assumptions for oxbow and riparian wetlands in central Oklahoma. Response variables were collected in May and June, 2010 and include vine cover (VICOV), canopy cover (CANCOV), snag density (SNAGDEN), coarse woody debris volume (CWDVOL), % subsurface organic matter (OMSUB), % surface organic matter (OMSURF), water hardness (HARD), water conductivity (COND), herbaceous cover (HERBCOV), shrub/sapling cover (SHCOV), tree basal area (TRBASAL), coarse woody debris stem count (CWDSTCT), and % subsurface sand (SANDSUB).

Wetland Type	**Site metric**	**Transformation**
Riparian	VICOV	$-(\mathrm{VICOV}+1)^{-2}$
Riparian	CANCOV	CANCOV^{2}
Riparian	SNAGDEN	$-(\mathrm{SNAGDENS}+0.01)^{-2}$
Riparian	CWDVOL	$-(\mathrm{CWDVOL}+0.01)^{-2}$
Riparian	OMSUB	$\log_{10}(\mathrm{OMSUB})$
Riparian	OMSURF	$\log_{10}(\mathrm{OMSURF})$
Riparian	HARD	$\log_{10}(\mathrm{HARD})$
Riparian	COND	$\log_{10}(\mathrm{COND})$
Oxbow	HERBCOV	$\mathrm{HERBCOV}^{2}$
Oxbow	VICOV	$-(\mathrm{VICOV}+1)^{-2}$
Oxbow	SHCOV	$-(\mathrm{SHCOV}+1)^{-1}$
Oxbow	TRBASAL	$-(\mathrm{TRBASAL}+0.01)^{-2}$
Oxbow	SNAGDENS	$-(\mathrm{SNAGDENS}+0.01)\hat{}^{-2}$
Oxbow	CWDVOL	$-(\mathrm{CWDVOL}+0.01)^{-2}$
Oxbow	CWDSTCT	$\log_{10}(\mathrm{CWDSTCT}+1)$
Oxbow	OMSURF	$\log_{10}(\mathrm{OMSURF})$
Oxbow	SANDSUB	$\log_{10}(\mathrm{SANDSUB}+0.01)$

Table 4. List of study oxbow (O1-O20) and riparian (R1-R20) sites in central Oklahoma and their associated river. Groundwater influence and isolation from flooding were determined based on visual assessments conducted in the field in May and June, 2010. Distance to downstream reservoir was approximated using GIS.

Site ID	River	Groundwater influence	Isolation from flooding	Distance to downstream reservoir
O1	North Canadian River	yes	no	n/a
O2	North Canadian River	no	yes	n/a
O3	North Canadian River	yes	no	n/a
O4	Deep Fork River	no	no	n/a
O5	Washita River	no	no	n/a
O6	North Canadian River	no	yes	n/a
O7	North Fork of the Red River	no	no	n/a
O8	Washita River	no	yes	n/a
O9	Chisholm Creek	yes	yes	n/a
O10	Caney River	no	no	n/a
O11	Canadian River	no	no	n/a
O12	Salt Fork of the Arkansas River	yes	no	n/a
O13	Deep Fork River	yes	no	n/a
O14	Cimarron River	yes	no	n/a
O15	Bitter Creek	no	no	n/a
O16	Salt Fork of the Red River	no	no	n/a
O17	Buffalo Creek	no	no	n/a
O18	Salt Fork of the Arkansas River	no	no	n/a
O19	Canadian River	no	no	n/a
O20	Rainy Mountain Creek	no	no	n/a
R1	Red River	n/a	n/a	>25 km
R2	Washita River	n/a	n/a	>25 km
R3	North Canadian River	n/a	n/a	<5km
R4	Cimarron River	n/a	n/a	>25 km
R5	North Canadian River	n/a	n/a	>50 km
R6	Washita River	n/a	n/a	<10 km
R7	North Fork of the Red River	n/a	n/a	>50 km
R8	North Fork of the Red River	n/a	n/a	>50 km
R9	Buck Creek	n/a	n/a	<25 km
R10	Oil Creek	n/a	n/a	<10 km
R11	Unnamed	n/a	n/a	<5km
R12	Sand Creek	n/a	n/a	>50 km
R13	Red Rock Creek	n/a	n/a	>50 km
R14	Blue River	n/a	n/a	>50 km
R15	Caney River	n/a	n/a	<10 km
R16	Sand Creek	n/a	n/a	>50 km
R17	Washita River	n/a	n/a	>50 km
R18	Skeleton Creek	n/a	n/a	>50 km
R19	Sandy Creek	n/a	n/a	<10 km
R20	North Fork of the Red River	n/a	n/a	<5 km

Table 5. Component loadings for 21 response variables from 20 riparian and 20 oxbow wetlands in central Oklahoma collected in May and June, 2010. Component loadings are presented for all four ordination plots presented in Figs. 2-5. Only the component loadings for the first RDA axes are presented. Both RDA analyses presented only included one environmental variable, wetlands with subclass as the environmental variable (Fig. 2) and riparian sites with precipitation as the environmental variable (Fig. 5). The component loadings for the first two axes are presented for both PCA analyses on oxbows (Fig. 3) and riparian (Fig. 4) wetlands. Response variables include % surface clay (CLAYSURF), % subsurface clay (CLAYSUB), % surface silt (SILTSURF), % subsurface silt (SILTSUB), % surface sand (SANDSURF), % subsurface sand (SANDSUB), % surface organic matter (OMSURF), % subsurface organic matter (OMSUB), coarse woody debris volume (CWDVOL), coarse woody debris stem count (CWDSTCT), litter cover (LITTERCO), snag density (SNAGDEN), herbaceous cover (HERBCOV), vine cover (VICOV), shrub/sapling cover (SHCOV), canopy cover (CANCOV), tree density (TRDEN) and tree basal area (TRBASAL). X's denote variables that were not included in an analysis.

	RDA- ALL SITES SUBCLASS	**RDA-RIPARIAN PRECIPITATION**	**PCA- OXBOWS**		**PCA- RIPARIAN**	
VARIABLE	**AXIS 1**	**AXIS 1**	**AXIS 1**	**AXIS2**	**AXIS 1**	**AXIS 2**
HERBCOV	-0.260	-0.448	0.560	-0.102	0.059	-0.498
VICOV	0.222	0.672	-0.203	0.767	-0.544	0.363
SHCOV	0.329	0.451	-0.022	0.299	0.257	0.594
CANCOV	0.465	0.324	0.181	0.870	-0.841	0.380
LITTERCO	0.028	-0.201	0.256	0.483	-0.567	-0.098
TRDENS	0.438	-0.148	0.299	0.769	-0.521	0.502
TRBASAL	0.388	-0.042	0.032	0.737	-0.769	0.328
SNAGDEN	0.086	0.369	-0.078	0.462	-0.488	0.688
CWDVOL	-0.043	0.249	-0.234	0.259	-0.658	0.032
CWDSTCT	0.239	0.105	0.098	0.913	-0.483	0.628
OMSUB	-0.227	-0.148	-0.801	0.451	-0.741	-0.007
OMSURF	-0.367	-0.146	-0.531	0.185	-0.448	-0.365
SANDSUB	0.383	-0.461	0.905	0.098	0.787	0.189
SILTSUB	-0.268	0.430	-0.752	-0.055	-0.782	-0.238
CLAYSUB	-0.409	0.379	-0.879	0.116	-0.567	-0.053
SANDSURF	0.499	-0.274	0.964	0.068	0.679	0.618
SILTSURF	-0.309	0.396	-0.653	0.265	-0.611	-0.550
CLAYSURF	-0.523	-0.004	-0.789	-0.293	-0.635	-0.590
ALK	0.110	-0.180	x	x	-0.066	0.212
HARD	0.590	-0.428	x	x	0.635	0.016
COND	0.363	-0.147	x	x	0.650	-0.228

Table 6. Summary of significant forward stepwise regressions for 20 oxbow and 20 riparian wetlands in the Central Great Plains and Cross Timbers Ecoregions of central Oklahoma. Independent variables include average annual precipitation, stream order, % human altered land within a 100m buffer, % human altered land within a 1000m buffer, land use score for a 100m buffer, and land use score for a 1000m buffer.

Subclass	Dep. Variable	Ind. Variable 1	Delta R^2	p value	Ind. Variable 2	Delta R^2	p value	R^2
Oxbow	% herbaceous cover	precipitation	0.235	0.030	n/a	n/a	n/a	0.235
Oxbow	tree density	precipitation	0.240	0.028	n/a	n/a	n/a	0.240
Oxbow	coarse woody debris stem count	precipitation	0.235	0.030	n/a	n/a	n/a	0.235
Oxbow	% subsurface organic matter	% human Altered 100m	0.282	0.016	n/a	n/a	n/a	0.282
Riparian	% subsurface sand	% human Altered 100m	0.237	0.017	precipitation	0.211	0.021	0.448
Riparian	% surface silt	% human Altered 100m	0.258	0.016	precipitation	0.154	0.050	0.412
Riparian	% subsurface silt	% human Altered 100m	0.346	0.003	precipitation	0.183	0.020	0.529
Riparian	% subsurface organic matter	stream order	0.234	0.031	n/a	n/a	n/a	0.234
Riparian	% herbaceous cover	precipitation	0.198	0.050	n/a	n/a	n/a	0.198
Riparian	tree density	% human Altered 1000m	0.197	0.050	n/a	n/a	n/a	0.197
Riparian	conductivity	stream order	0.431	<0.001	precipitation	0.135	0.034	0.566
Riparian	hardness	stream order	0.415	<0.001	precipitation	0.185	0.012	0.415
Riparian	alkalinity	stream order	0.191	0.054	n/a	n/a	n/a	0.191

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APPPENDIX

Appendix: Key for classifying wetlands according to the Hydrogeomorphic Approach in the Cross Timbers and Central Great Plains Ecoregions of central Oklahoma.

1. Wetland is within the 5 year floodplain of a river but not fringing an impounded water body..**Riverine (4)**

1. Wetland is associated with a topographic depression or slope..2

2. Wetland is located on a topographic slope or relatively flat area and has groundwater as the primary water source. Wetland does not occur in a basin with closed contours..**Slope (15)**

2. Wetland is located in a natural or artificial (dammed/excavated) topographic depression..3

3. Topographic depression has permanent water greater than 2 meters deep..**Lacustrine Fringe (9)**

3. Topographic Depression does not contain permanent water greater than 2 meters deep..**Depression (11)**

4. The wetland is a remnant river channel that is periodically hydrologically connected to a river or stream every 5 years or more frequently..................................***Connected Oxbow***

4. The wetland is not an abandoned river channel...5

5. The hydrology of the wetland is impacted by beaver activity...................***Beaver Complex***

5. The hydrology of the wetland is not impacted by beaver activity.....................................6

6. The wetland occurs within the bankfull channel...***In-Channel***

6. The wetland occurs on the floodplain or is adjacent to the river channel...........................7

7. The wetland occurs within a depression on the floodplain.............***Floodplain Depression***

7. The wetland occurs on a flat area on the floodplain or is adjacent to the river channel......8

8. Wetland water source primarily from overbank flooding that falls with the stream water levels or lateral saturation from channel flow...***Riparian***

8. Wetland water source is primarily from overbank flooding that remain in the wetland due to impeded drainage after stream water level falls...***Floodplain***

9. Wetland is associated with a remnant river channel that is hydrologically disconnected from the stream or river of origin..***Disconnected Oxbow***

9. Wetland is associated with a reservoir or pond created by impounded or excavation......10

10. Wetland water source is primarily from a permanent river......................***Reservoir Fringe***

10. Wetland water source is primarily from a draw or overland flow...................***Pond Fringe***

11. Wetland was created by human activity...12

11. Wetland was not created by human activity...13

12. Wetland does not have discernible water outlets...............***Closed Impounded Depression***

12. Wetland has discernible water outlet....................................***Open Impounded Depression***

13. Wetland primary water source is groundwater............................***Groundwater Depression***

13. Wetland primary water source is surface water..14

14. Wetland does not have any discernible water outlets.....***Closed Surface Water Depression***

14. Wetland has discernible water outlets.............................***Open Surface Water Depression***

15. Wetland is hydrologically connected to a low order (Strahler <=4), high gradient, or ephemeral stream...***Headwater slope***

15. Wetland is hydrologically connected to a high order (Strahler >=5), low gradient river. Slope may be imperceptible or extremely gradual (includes wet meadows)...***Low Gradient Slope***