Conversion of existing farm ponds to wetlands in agricultural landscapes for mitigation, land use treatment and conservation with a perspective toward climate change.

Final Report

Donald Huggins, Jude Kastens, Debbie Baker, Craig Freeman

Kansas Biological Survey

June 2017

Kansas Water Office Contract 15-111, KUCR project 72845

TABLE OF CONTENTS

APPENDICES

BACKGROUND

As described in numerous previous wetland program development grants (WPDGs) submitted by the Kansas Water Office (KWO, 2008 – 2013), Kansas has lost around 50% of its historic wetlands, and many of those remaining continue to be threatened by agricultural and urban development activities. Regulation of wetlands is governed by the Army Corps of Engineers 404 permitting process and the state's 401 certification program. There is no additional state regulatory oversight, and our influence to stop additional losses and restore wetland acres and functions of this valuable resource are limited to voluntary programs. The concept in this project is to try a unique and innovative approach to wetland restoration and to ultimately work with agricultural producers to transform existing farm ponds into more complex, functional wetlands that will provide more ecological goods and services than the original configuration. While these "pond to wetland" conversions most closely approximate constructed wetlands (CW) and do not represent original wetlands, they can still provide many wetland functions to a greater or lesser extent.

Almost half of the historic wetlands in Kansas have been reported to have been lost through drainage and agricultural land conversion. Of the approximately 435,400 acres of wetlands that remain, several well-known, large, important wetlands and wetland complexes have survived in some form. These include Cheyenne Bottoms, Quivira National Wildlife Refuge, Jamestown Wildlife Area, and McPherson Valley Wetlands Wildlife Area. Yet these and other federal- and state-owned wetlands (28,744 acres) account for a small fraction (about 7%) of the approximately 435,400 acres that remain of the originally estimated 841,000 acres (Dahl 1990). While other large wetlands or wetland complexes have been destroyed, it appears that most of the wetlands lost in Kansas were smaller, scattered ones in areas that experienced dramatic land use changes (e.g. cultivation, tile and surface drainage). According to the map coverage in the publication entitled "Classification of Wetland and Riparian Areas in Kansas" (Schenck *et al.* 1992), most wetlands on cropland occur in the eastern one-third of Kansas. Additionally, most wetland types listed for Kansas primarily occur in far eastern Kansas, and the south-central and west-central portions of the state. Assuming that the pattern of remaining wetlands is an indicator of regions where losses would have occurred within this natural distribution of Kansas wetlands, we focused on wetland loss, restoration, and mitigation in these same regions. These regions, in general, currently support a large population of "farm ponds," which are small to moderately sized, artificial ponds either excavated or dammed, created for a variety of rural uses. We believe that many of these existing ponds, most occurring in largely agricultural drainages, can be identified and used in mitigation or land use treatment if wetland functions currently exist or can be developed through inexpensive physical and biological modifications. This pond population consists of thousands of artificial ponds from a quarter of an acre to tens of acres, many dating back to the 1940s-1950s with some ponds already maturing into "wetlands" (≈ constructed wetlands) with many wetland functions. See Appendix A for a list of goods and services provided by artificial ponds.

Our concept recognizes that thousands of farm ponds have been constructed for various purposes on the agricultural landscape and that many may be suitable for repurposing to provide additional wetland and watershed functions. We evaluated the potential for converting

these existing ponds into functional wetlands through a Phase I, five-step process: 1) Identify and inventory existing ponds using GIS methodologies; 2) Delineate catchments that drain to each pond using LiDAR and quantification of land use/land cover (LULC) into at least crop, grassland, forest, and urban categories; 3) Classify ponds and their catchments using criteria such as pond size groupings, catchment drainage area, catchment LULC, local pond topography, presence of aquatic vegetation in the pond, and other potentially relevant selectors; 4) Field assess and inventory a selected number of classified ponds to determine basic physical and biological attributes to validate and improve pond classifications in Step 3; and 5) Field evaluate high priority ponds identified in Steps 3 and 4. This process provides the mechanisms to select, develop, and apply pond-to-wetland conversions based on most suitable use and conversion criteria developed by combining appropriate outcomes from Steps 3 – 5. Ultimately, this process will insure that pond-to-wetland conversions would occur in watersheds where lost wetland functions are most needed and where suitable candidates for conversion exist. Subsequently, this project addresses two activities in the Kansas Wetland Program Plan (KS WPP), Voluntary Restoration Core Element: 1) "Provide technical and financial assistance to private landowners for protecting, enhancing or restoring wetlands" and; 2) "Use a watershed approach to protect and restore wetlands by integrating wetland goals into WRAPS 9-element plans"

Farm ponds are most often located in the mid- to upper-portions of watersheds where they may impound water from overland runoff, gullies, grass waterways, and possibly first-order stream channels. Pond construction has been funded both privately and through public programs such as the Watershed Protection and Flood Prevention Act (PL-566). Pond uses and benefits include, but are not limited to, water storage (local uses such as water for concentrated animal feeding operation (CAFOs), orchards, fields, gardens), fire protection, wildlife habitat (e.g. birds, mammals, amphibians, reptiles), fishing, general recreation (e.g. boating, swimming, bird watching), aesthetics (e.g. landscaping features), and retention (e.g. to trap sediment and nutrients). Ponds provide many benefits but may also have negative impacts. We believe that many ponds, due to their morphology, landscape position, and developed biological attributes, have developed wetland features that support many wetland functions that are associated with constructed wetlands and even natural wetlands. This natural progression of some artificial ponds to assume wetland structure and function suggests that a program to identify, characterize, and classify members of this large pond population could produce a listing of pond candidates suitable for conversion to wetlands or enhancement of wetland function.

The pond community found throughout the Central and Southern Plains (> 2.5 million ponds) likely will be among the first aquatic ecosystem to respond to forecasted climate changes for this region. Our interpretation of projected climate change scenarios (see Covich *et al.* 1997, Coops *et al.* 2003, Melillo *et al.* 2014) suggests that ponds will begin to assume many features of wetlands including increases in hydroperiod variability and increases in extreme events (i.e. increases in deep, shallow, and dry water level conditions) with associated impacts on water storage, biota, and downstream water availability. Others have assessed the potential for the change or loss of specific biological services and functions associated with small ponds. For

example, Sorenson *et al.* (1998) found that populations of breeding waterfowl decreased with increases in the Palmer Drought Severity Index (PDSI), a climate change surrogate. This effort in inventorying and characterizing this community provides much needed biological, physical, and hydrological data that can be used to assess broad-scale responses to climate change in this region. Factors such as surface area, mean depth, volume, and spatial distribution are needed to estimate impacts due to changes in factors such as flood and drought frequency and intensity, transpiration, and evaporation. Identification of current abiotic and biotic conditions (e.g. macrophytes and wetland flora) will be necessary to assess future alterations in these communities as the climate continues to change (e.g. Williamson *et al.* 2009, Schindler 2009).

The project study area is the Delaware River Basin (approximately 750 square miles) in the Western Corn Belt Plains ecoregion of northeastern Kansas (Figure 1). The Delaware River watershed, a well-studied watershed with a very active Watershed Restoration and Protection Strategy (WRAPS) group, is an ideal location to develop this program effort and to assess its application to the larger Central and Southern Great Plains (http://www.delawarewraps.com). It is replete with farm ponds and larger impoundments, including Perry Lake, and has mixed agriculture use with both intensely cultivated regions and areas where grasslands and pastures dominate the land use. This landscape setting allows us to assess a broad variety of potential pond and catchment conditions to better develop this pond-to-wetland approach.

Outputs, Outcomes, and Results

Outputs. 1. Identification of pond populations in NE Kansas (Delaware Basin). 2. Delineation of catchments that drain to ponds. 3. Delineation of pond complexes (e.g., the wetted area if the pond was filled to, for example, one foot below the overflow spillway; this definition will be optimized during the research). 4. Pond complex suitability indexing scheme for the purpose of conversion to wetland or enhancement of wetland function. 5. Pond catchment suitability indexing scheme for the purpose of identifying level of need for wetland function of the associated pond complex. 6. Identification of pond complex features suggesting existing wetland functions. 7. Identification of local topographic features (*slope*) that might facilitate wetland conversion or wetland function enhancement. 8. Development of selection process based on opportunity (pond complex index, existing wetland function, facilitative topography) and need (pond catchment index) that would identify high priority areas for mitigation, land treatment or goal-orientated projects. 9. Identification of current wetland functions of selected ponds for potential conversion.

Outcomes. 1. Enhanced opportunities to work with the agricultural community to promote wetland restoration, enhancement, and creation with a program that is suited to their current needs. 2. Improved wetland restoration/protection efforts. 3. Enhanced tools to increase wetland acreage on the landscape. 4. Programs to assist with 9-element implementation plan especially for wildlife habitat and achievement of total maximum daily loads. 5. Ability to transform critical agricultural runoff through biologically active habitats. 6. Increased knowledge of potential climate change influences on ponds and their possible shift to wetland environments**.** 7. Implementation of two important activities in the WPP.

Link to EPA Strategic Plan. This project addresses the EPA Strategic Plan Objective *"… working with partners, achieve a net increase in wetlands nationwide with additional focus on coastal wetlands, and biological and functional measures, and assessment of wetland condition…"* by providing an additional strategy for working with private landowners to add wetland acreage to their properties. Landowners are often reticent to even discuss giving up any of their land currently in production for conservation practices. By providing assistance in improving an aquatic resource already on their property and not used for production, they may be more willing to participate in wetland enhancement activities.

POND SELECTION

GIS

Using a combination of LiDAR hydroflattened areas and USGS National Hydrography Dataset (NHD) waterbody polygons, an initial population (POP0) of farm ponds 1-5 acres in size was determined that consisted of 1148 elements (Appendix B; Figure 1 and Figure 2). Next, we computed watersheds and flood pools for elements of POP0 (Appendix C). These data layers were overlaid on high resolution aerial imagery to produce maps used to support and guide field data collection efforts, helping field workers to locate sampling sites within and around the ponds.

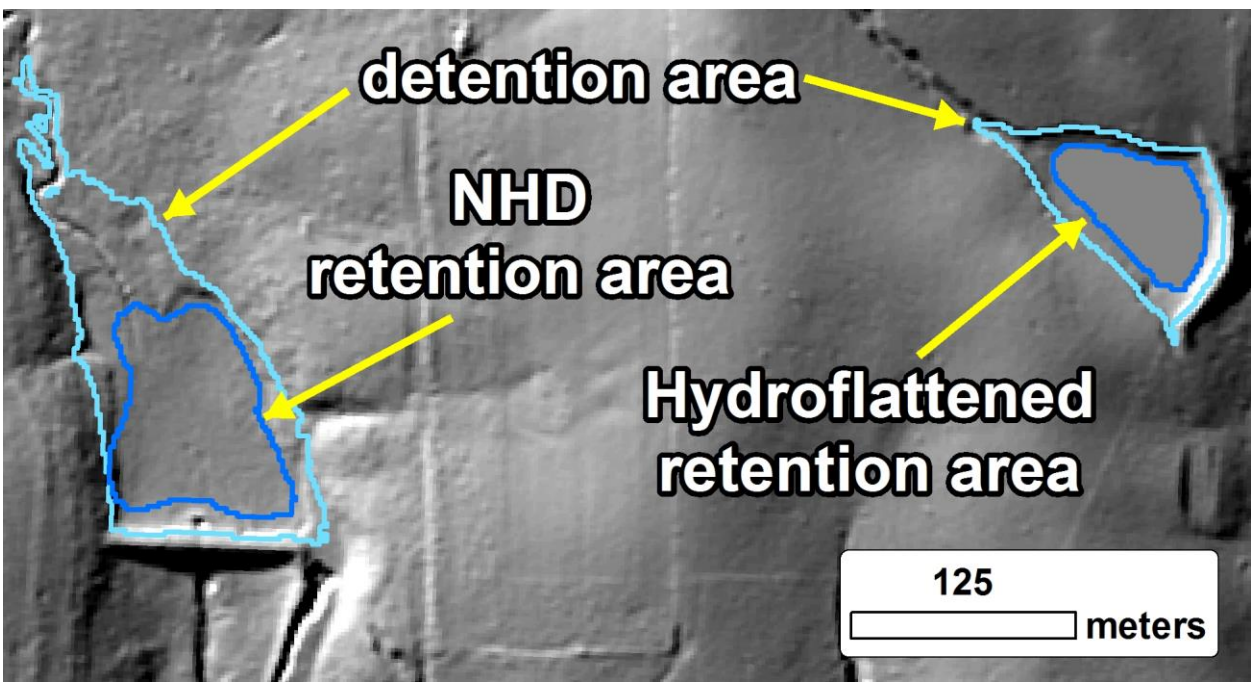


Figure 1. Retention area and detention area boundaries are shown for two elements of the initial population (POP0) of farm ponds 1-5 acres in size (See Appendix B for methods).

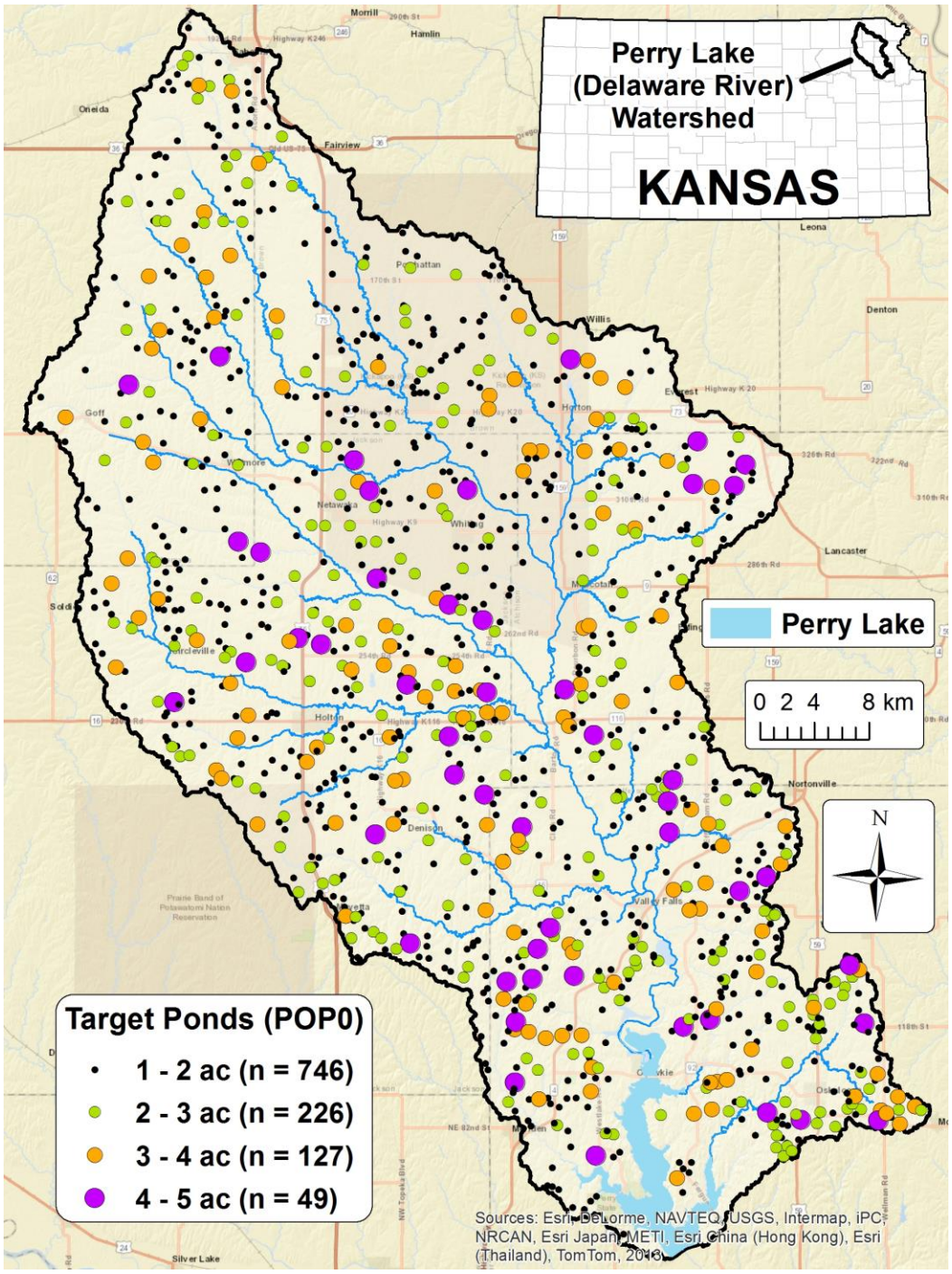


Figure 2. Perry Lake (Delaware River) watershed map, including locations for the 1148 farm ponds comprising the target sampling population (POP0).

GRTS Survey Design

From the resulting GIS layers containing 1148 ponds, we used a Generalized Random Tessellation Stratified (GRTS) survey design for characterization of moderately small (1 to 5 acre), man-made rural impoundments (i.e. farm ponds) in the Delaware River basin in northeast

Kansas. Probability-based sample surveys are increasingly being used to assess natural resources especially at large spatial scales; GRTS being a true probability design approach that allows an approximately spatially-balanced study design (e.g. Stevens 1997, Olsen *et al.* 1999, Stevens and Olson 2004, Stevens and Jensen 2007, Stevens *et al.* 2007).

This sampling regime allowed us spatially to balance the randomly selected ponds for sampling within the watershed population and *a prior* identified size categories. Our current and past GIS work in this watershed showed that pond surface areas were both continuous and were not evenly distributed across size categories identified (Table 1). For example, smaller ponds were more numerous than larger ponds, thus the sample design randomly selected ponds in numbers proportional to this size stratification. See Appendix D for details. A resulting list of 600 ponds was generated for evaluation in the office, with the goal of obtaining landowner permissions to sample 100 ponds. To maintain sample design integrity, ponds were used in the order they appeared in the list, i.e. all sites that occur prior to the last site sampled must have been evaluated for use and then either sampled or have a documented reason for not being used.

Permissions and Final Pond Tally

Office evaluation of ponds required sending permission request letters with maps to landowners of 283 ponds. We obtained permission to sample 100 ponds; however, at the start of the field season two of the landowners dropped out of the study. Thus we visited 98 ponds ranging in size from 1.03 acres to 4.89 acres, with 64 ponds less than 2 acres and only two ponds greater than 4 acres (Table 1, Figure 3). The final distribution of ponds that were retained in this study was as follows: 12 ponds in Atchison County; 10 in Brown County; 36 in Jackson County; 31 in Jefferson County; and 11 in Nemaha County. This final set of ponds provided excellent spatial coverage of the Delaware River watershed beginning near the upper end of Perry Reservoir, a large USACE reservoir in Jefferson County.

Table 1. Numbers of ponds by size category in total population (All), population for which landowners were contacted, population for which permissions were received, and sampled population. Size estimates were made using both LiDAR and the National Hydrology Database (NHD, Appendix B).

Group	Area category (acres)								Total Ponds
	1.0 - < 1.5		1.5 - < 2.5		2.5 - < 3.5		3.5 - 5		
	#	%	#	%	#	%	#	%	
All	500	44	387	34	153	13	108	9	1148
Contacted	123	43	95	34	38	13	27	10	283
Permission	42	42	33	33	14	14	11	11	100
Sampled	42	43	31	32	14	14	11	11	98

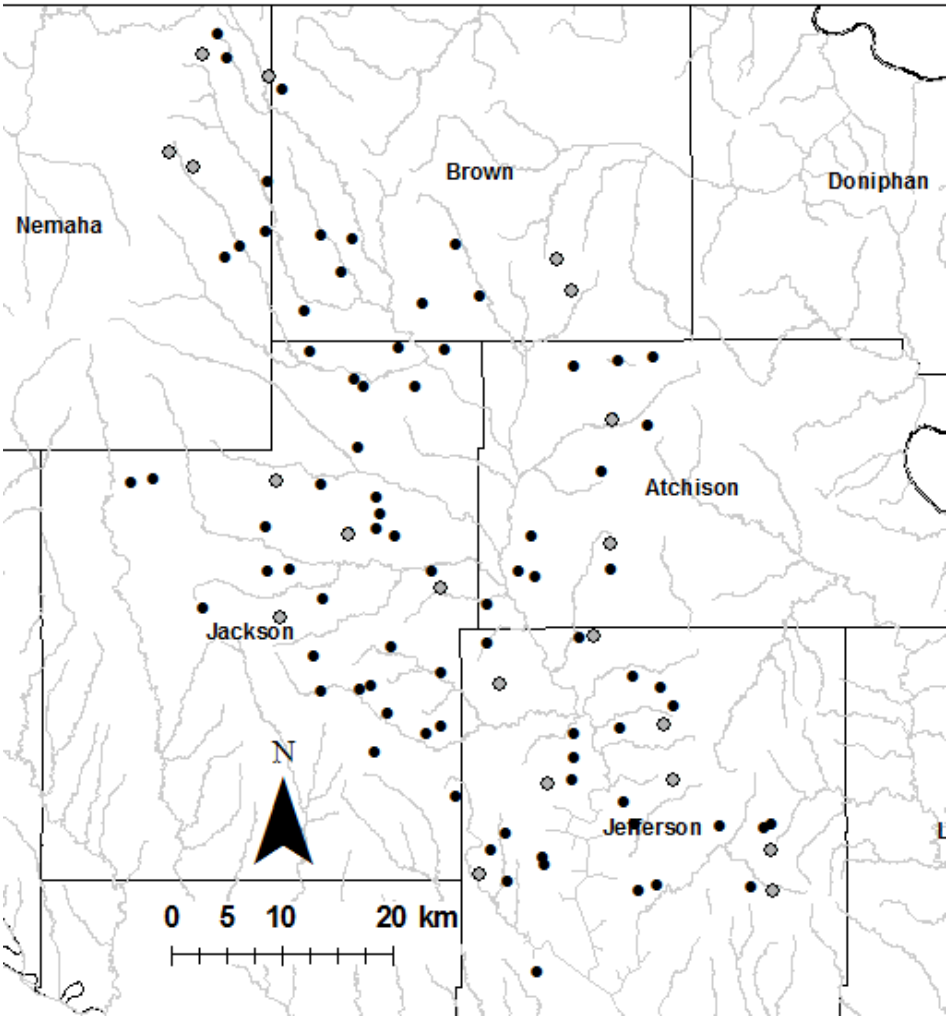


Figure 3. Map of the 98 ponds sampled, with the subset (gray circles) of 20 ponds re-visited for water quality and vegetation.

FIELD METHODS

The field season ran from November 2015 through June 2016 with 37 and 61 ponds sampled within two time periods (20 Nov. - 22 Dec. 2015, 5 May -27 June 2016), respectively. We typically were able to sample four ponds per day, but up to six per day could be done if they were in close proximity to one another. It generally took about two hours to sample a pond with a crew of two to four, depending pond size, complexity, and access. A larger crew did not impact sample time due to the large portion of time to access the site (up to 0.5 km) and the already optimum distribution of tasks but did lighten the burden of carrying equipment. The field data sheet (Appendix E) was composed of several measurement and assessment sections that corresponded with the different pond complex zone or regions that are explained below and illustrated in Figure 4 (A and B). The field forms were well illustrated with methodologies included with most sections to help maintain measurement consistencies.

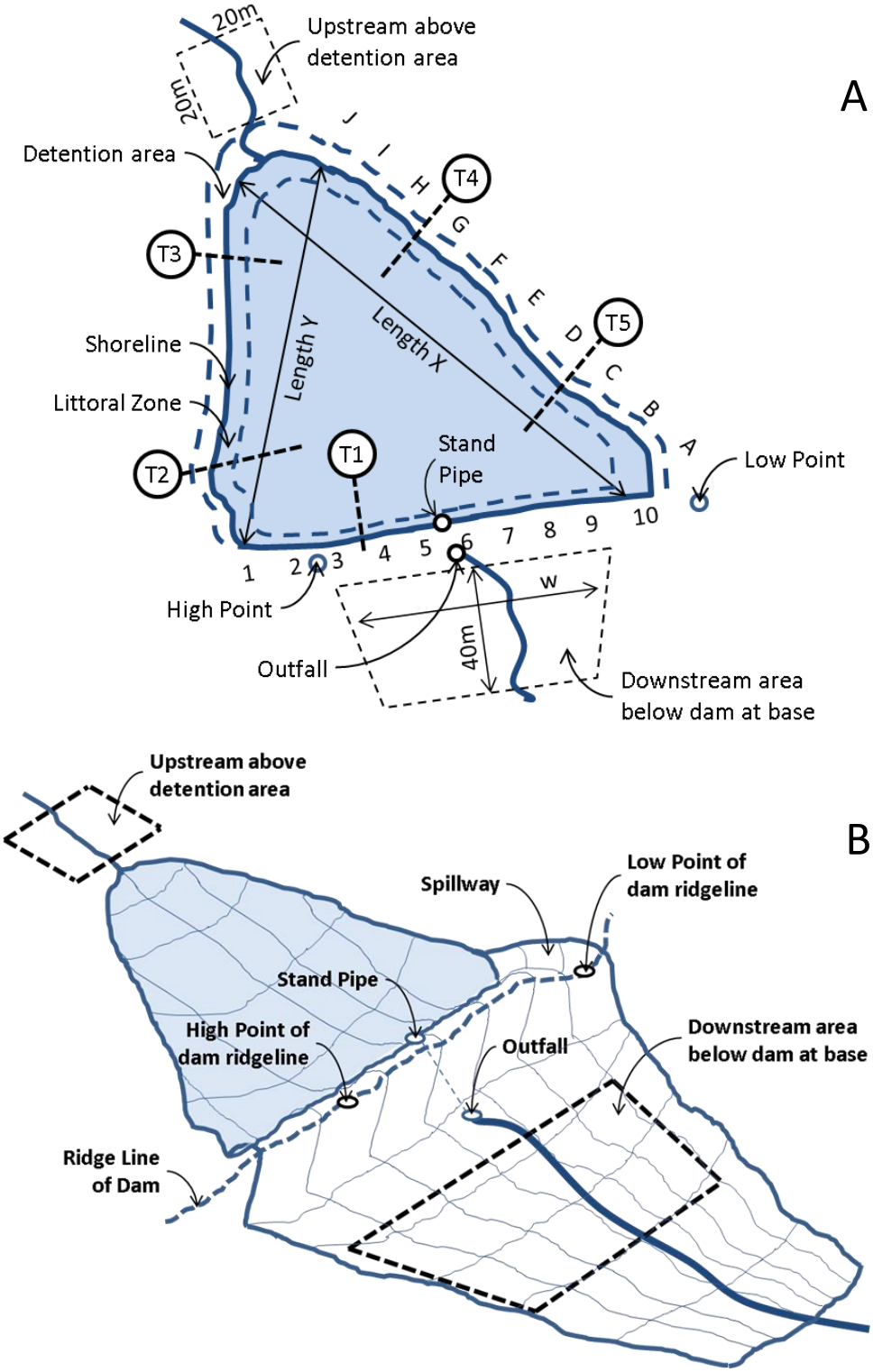


Figure 4. The plane (A) and oblique (B) illustrations of the pond complex sample areas and sampling regime. These include general distribution of lateral transects A - J, perpendicular transects 1 – 10, depth transects T1 – T5, longest axes X and Y, 20m x 20m upstream plot, and 40m x 40m downstream plot with width transects w.

A number of field elevation points were taken in relation to the prior determined LiDAR high and low points located on the dam and spillway, respectively. Three critical field survey elevations were taken: 1) the elevation of the current water level; 2) the elevation of the bottom of the stand pipe (Figure 5); and 3) the elevation of the mapped LiDAR high and low points. These survey points allowed us to calculate retention and detention areas that are discussed below regardless of current water levels. Attribute measurements of the immediate upstream areas (i.e. main inflow regions) and below dam outlet areas were also included in the field assessment both to further characterize the pond complex and to assess wetland characteristics and the potential of for wetland development.

Figure 5. Standing on top of the dam at pond 257 to measure stand pipe height (left) and dam high point (right) relative to water level.

The initial field surveys were designed to capture and record a number of key physical and biological features associated with each pond complex. After the completion of all initial surveys, a subset of 20 ponds was selected for further study of floristic and water quality (these results are discussed in the WATER QUALITY AND FLORISTIC ASSESSMENT section of this report). Ponds were considered to be just one part of the pond complex that is created when an impoundment is put in place. The pond itself is defined by the water retained within the impoundment and usually controlled by an outlet or drainage device such as drainage tube (Figure 6) while the immediate pond environment located at and below the emergency spillway constitutes an terrestrial/aquatic interface region (i.e. detention zone) that will flood during high runoff events that may be frequent enough to impart wetland features to this flooded, water retention region (Figure 7). The last pond complex zone or area is small watershed area that drains to the pond and thus provides water, nutrients, sediment and contaminates that influence the ecological nature of the pond environment and its potential human uses. A field sampling protocol and pond complex assessment document was constructed to identify and quantify various attributes of these zones that we thought to be of importance in determining the basic structure and function of the pond environments. Not all landscape and pond features of importance could be assessed within the time and economic constraints of this study. Therefore, a subset of 20 ponds was intensely examined beyond the initial field assessment and remote sensing efforts.

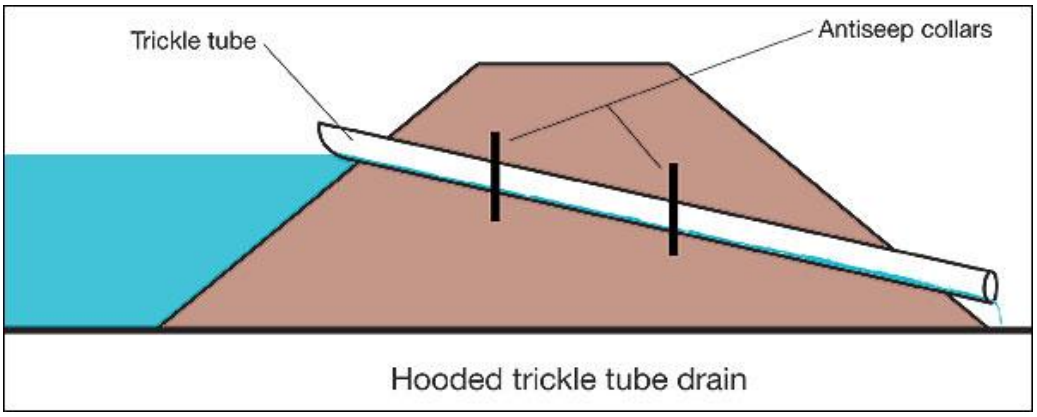


Figure 6. The hooded trickle tube drains and removes excess water from a pond during periods of normal runoff (Hicks and Pierce 2014).

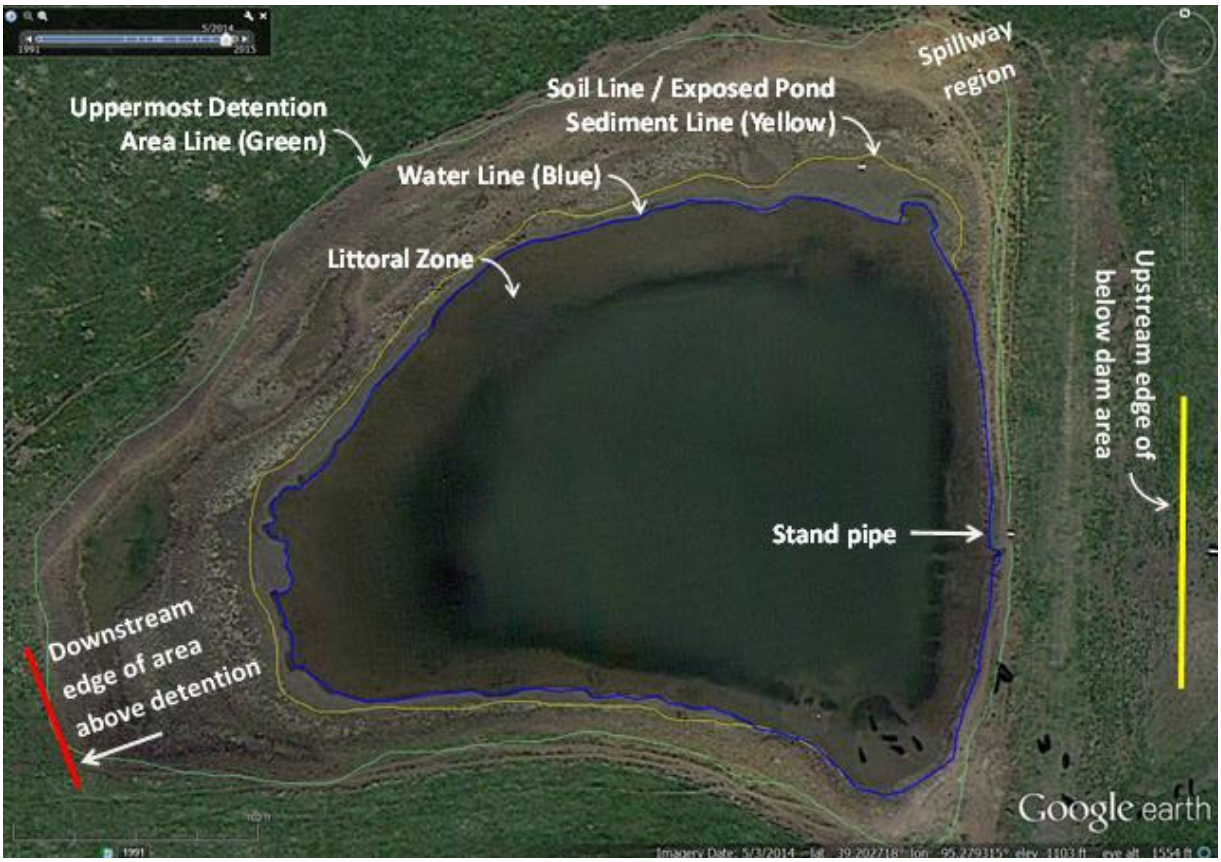


Figure 7. Schematic of two of the three pond complex zones recognized and studied in this project. The pond environment is define by the yellow line that denotes retention area created by the stand pipe discharge point while the innermost blue line is the actual water shoreline at date of photo. The outer green line delineates the maximum flood fill thus defining the detention zone as the area between the yellow and green lines. Additionally, the yellow linear line is the 40 m width of the below dam plot, red linear line is the 20 m width of the above pond plot. This 2015 pond photograph is of a non-study pond in the Delaware River basin.

Findings from the field surveys are presented in the following field measurement sections as well as their use in the LiDAR and GIS analyses of the detention and retention zones of the various pond complexes. These field-derived pond complex attributes along with remotely sensed measurements help define the structure and to a certain extent the function of the assessed pond population randomly chosen for study. Similarly the integrated assessment landscape-level and pond-level factors allows us to target those pond complexes that have developed wetland functions and what conditions might be altered to develop wetland conditions within or near these artificial ponds.

DATA ANALYSES

Some of the descriptive statistics and graphics in this report were calculated and constructed using Number Cruncher Statistical Software 9 (NCSS 2013). In addition R statistical package (R Core Team 2012) was used to calculate empirical Cumulative Distribution Functions for a number of pond complex attributes and characteristics.

One way to examine the relative condition of waterbodies is to look at how given values of a parameter fit in to the overall range of a larger sample. Cumulative distribution functions (CDFs) provide an excellent tool for doing just that. Similar to histograms, CDFs describe the magnitude of observed values for all of the observations in a sample. However, a CDF is essentially a curve that describes what proportion of a sample is expected to have a given value, based on the data observed. By plotting CDFs of various parameters, it is possible to generalize the characteristics of a group of observations in terms of proportions.

While a CDF can be extended to describe an entire population (e.g., all farm ponds in Kansas) under certain assumptions, an *empirical* CDF is based only on the actual values observed in a given sample (e.g., farm ponds in the Delaware River basin visited during this study). Empirical CDFs are made by ordering the directly observed values of a given parameter (e.g., stand pipe diameter) from least to greatest (e.g., 0 to 150 cm), then plotting those values versus the cumulative proportion of observations that have at least that value. Cumulative proportion is determined by finding the individual proportion of one observation (e.g., 1 out of 96 ponds observed or 1/96), then adding an individual proportion for each observation that is less than or equal to a given value of the given parameter (e.g., if 72 of 96 ponds have at least a value of 40, then the cumulative proportion is 72/96 or 0.75).

Empirical CDFs provide a quick but informative way to characterize a group of observations. For example, Figure 8 below shows the empirical CDF for the stand pipe diameter of the farm ponds sampled in this study. Note that in this figure, the proportion of the population with a stand pipe diameter of at least 40 cm is 0.75. In other words, 75% of the farm ponds observed had stand pipes that were 40 cm or less in diameter. Moreover, this also means that 25% of the farm ponds observed had stand pipes that were larger in diameter than 40 cm.

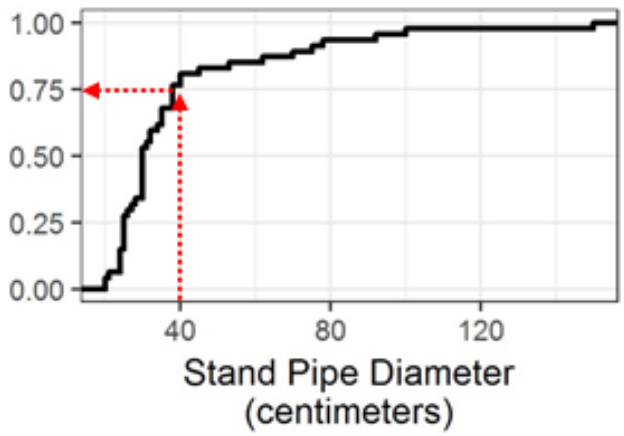


Figure 8. Empirical cumulative distribution function of stand pipe diameter for the farm ponds examined in this study, illustrating that the proportion of the population with a stand pipe diameter of at least 40 cm is 0.75.

By using CDFs, important thresholds in data may become immediately apparent. Sharp breaks in the curve tend to suggest abrupt changes in the observed parameter, whereas more smooth curves suggest gradual changes. This information may be valuable in understanding how and why certain ponds function differently from the majority of others.

FIELD MEASUREMENTS

Of the 98 ponds visited, 96 had water. The two that were dry had intentionally breached dams (Figure 9). Further statistics and discussion are about the 96 ponds with water.

Figure 9. Breached dam allows pond 212 to begin the transition to an emergent palustrine wetland.

Physical Structure

Inflow area

In order to provide a basic assessment of the physical and hydrological conditions associated with the area immediately above the normal pool area of the pond (i.e. inlet area), we measured or estimated a number of variables within a 20m x 20m subsample plot (area). In this upstream inflow area we recorded presence of livestock, and aerial coverage of wetland plants, woody vegetation, grasses and annuals, bare soil, and crops (Table 2). When a defined channel was present, we measured channel width, depth, and noted channel stability (sedimentation, erosion, retreating, etc.). Of 93 ponds for which we examined inflow area conditions, 39% of these areas showed evidence of livestock use with some areas showing signs of heavy vegetation damaging, soil erosion, trailing, and buildup of waste materials.

Table 2. Cover types in the 20m x 20m upstream inflow area.

cover type	# ponds surveyed	% ponds with cover type	max % area in cover type	% ponds with over 50% area in cover type
crops	92	8	20	0
wetland plants	91	76	98	14
woody	93	87	100	41
grasses & annuals	92	99	100	55
bare soil	92	57	40	0

The majority of ponds (72%) had inflow channels, of which most were in stable condition (90%). When inflow channels were present they were typically less than 5 meters in width (i.e. bank full width) and somewhat shallow (<0.8 m) (Figure 10 and Figure 11). The presence of inlet channels suggests that sediment and contaminate transport to these ponds is enhanced by open channel flows as opposed to sheet, rill, and non-channel overland flows (e.g. Prosser *et al.* 1995, Meritt *et al.* 2003, Aksoy and Kavvas 2005). The predominant sediment transportation mechanism is often drainage or watershed scale dependent with sediment yield (area-specific yield) in watersheds smaller than 0.05 km^2 coming mainly from rill and inter-rill flows (DeVente and Poesen 2005). The majority of pond watersheds in this study were greater than 0.168km^2 or 74.6 acres in size (the 25th percentile) suggesting gully erosion as the dominate source of sediment (yield). Many other factors contribute to both erosion and sediment transportation, but the occurrences of these large upland channels suggest that these ponds receive considerable amounts of sediment from both overland flows and the channels themselves. Smith and Wilcock (2015) observed that study ponds with inlet channels were also experiencing head cutting, which was also determined to be another source of pond sediment. Of the 7 inflow channels showing signs of unstable channel conditions, all but one showed evidence of livestock use in the plot. Conversely, over a one-third (34%) of all stable inflow channels assessed were noted to have past or present livestock use in the plot.

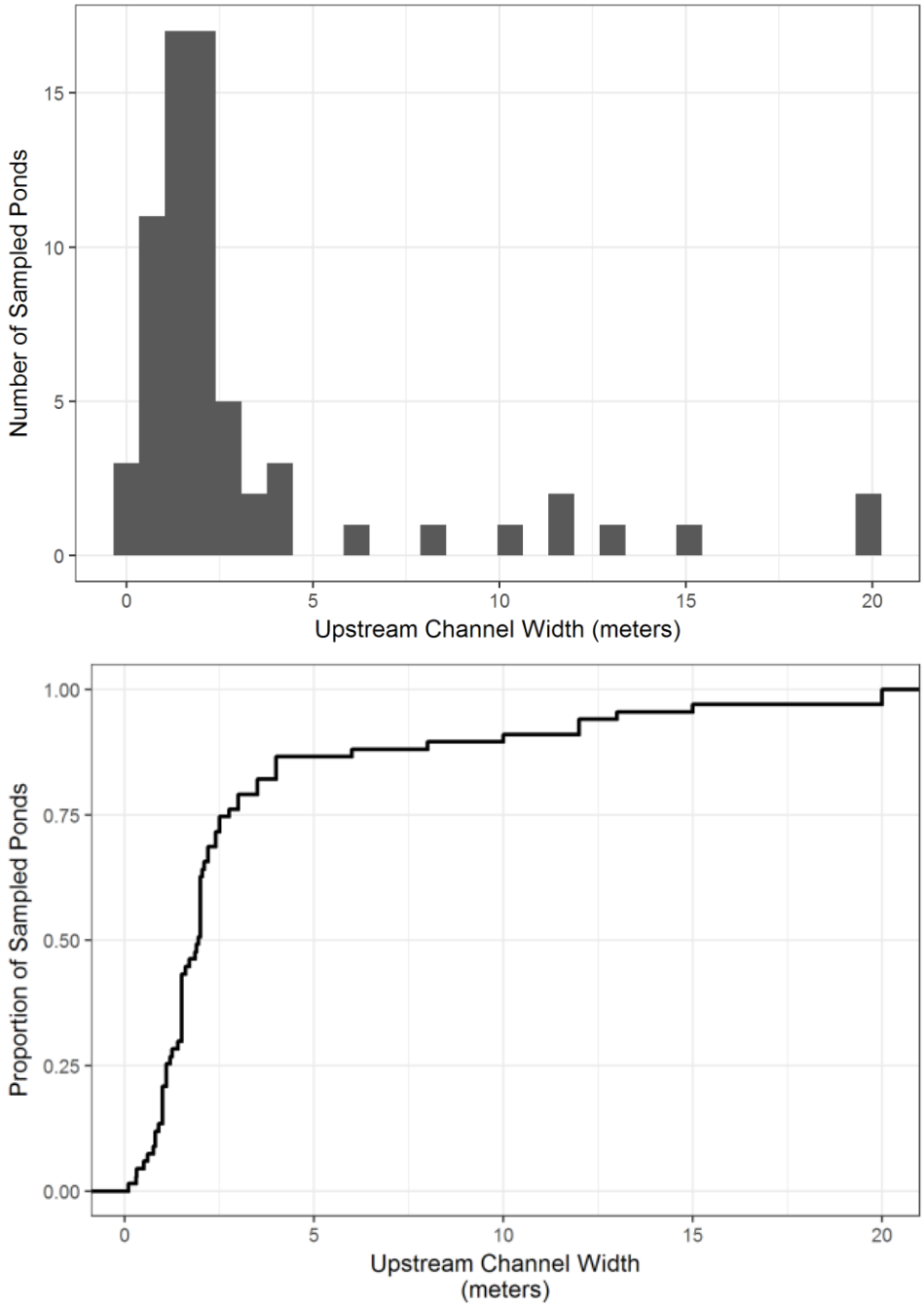


Figure 10. Histogram and CDF of upstream channel width showing that most upstream channels are < 5 m wide.

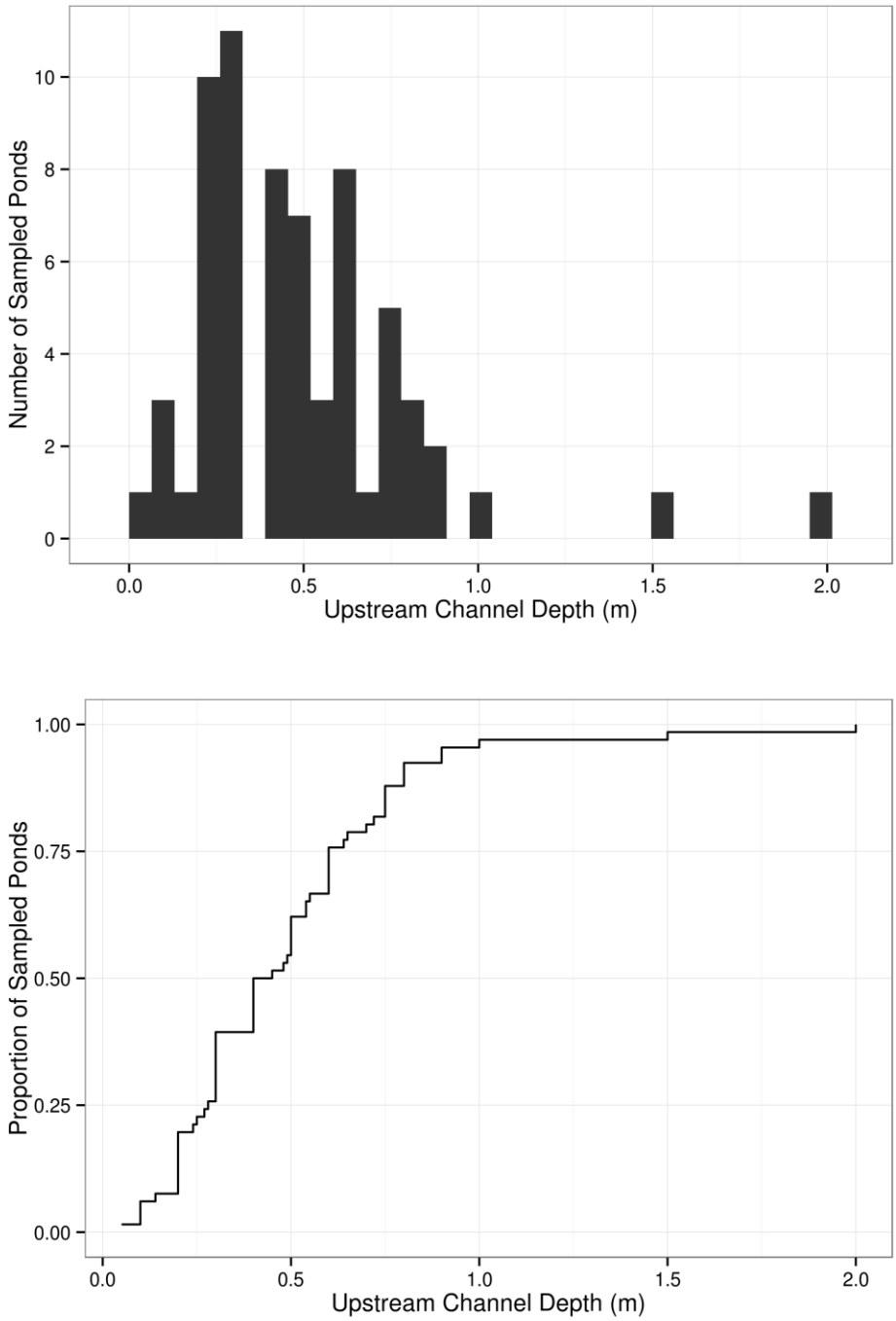


Figure 11. Histogram and CDF of upstream channel depth showing that most upstream channels are < 0.8 m deep.

While few inlet channels were noted to be highly unstable as indicated by the extent of bare erosional channel banks observed, it should be noted that the channels themselves are erosion features. It seems reasonable to assume that these ponds are intercepting sediment and water borne contaminates from their watersheds, but little is known of the trapping efficiencies of such small impoundments. Most research indicates that small pond and reservoir trapping efficiency rates are difficult to estimate and are highly variable, ranging from 20 – 80 percent (Verstraeten and Poesen 2000, 2001a, 2001b). Smith and Wilcock (2015) reported more consistent and typically higher sediment trapping efficiencies (85%) for six ponds in the piedmont region of Maryland USA.

Verstraeten and Poesen (2002) later examined nutrient trapping efficiencies in small Belgium ponds (n=13) and found that the mean nutrient trap efficiency varied between 4 and 31 percent, whereas sediment trap efficiency varies between 10 and 72 percent. Our finding of inlet channels to most ponds only suggest that sediments are entering these pond systems, but without direct knowledge of actual sediment yields and trapping efficiencies of these small ponds little else can be said regarding their ability to control downstream sediment delivery or their infill rates.

Typical, dominant vegetation in these inflow areas was grassy ground cover and an overstory of woody vegetation (i.e. trees and shrubs, Figure 12). Few inflow areas had crop cover (8%), and those that did had 20% or less of the area in crop. Nearly all inlet areas surveyed had some grass and annual cover with some sample sites having 100% of their area in grasses with over half (55%) of these areas being dominated by grasses and annuals (> 50% cover). A very high percentage of inflow sample areas (76%) had some sort of wetland vegetation cover with about 14% of these areas dominated by wetland vegetation cover (> 50% cover).

Figure 12. The channel of the upstream inflow area of pond 145. Vegetation was categorized as 35% wetland plants, 5% woody, 70% grasses and annuals, 0% crop, and 5% bare soil. A channel was present and measured 2 m wide and 0.75 m deep, and deemed stable.

<u>Dam structure</u>

All dams were earthen dams with graded, unarmored spillways to release flood inflows that cannot be controlled by the outlet system (e.g. trickle tube). Spillways were generally well vegetated and in good condition despite some used by livestock to access the ponds. Pond water retention was controlled almost exclusively by simple outlet pipes of various diameters that were installed during dam construction. These outlet systems will be discussed in more detail later. Dam heights were calculated as the difference between the dam high point (i.e. highest LiDAR determined dam elevation) and the spillway low point, which was also Lidar determined and represented the spillway crest or overflow point.

Elevational differences between these high and low LiDAR-determined points ranged from 0.175 to 2.785 m with the minimal elevation difference of 0.175 m representing the one pond configuration that did not have a spillway. That pond's dam and several others constituted a private road where the lowest elevation of the road serves as the emergency spillway or overflow area (Figure 13). Over 75% of all elevational differences between the spillway crest (i.e. LiDAR low point) and the dam high point were less than 1.5 m (Figure 14). As mentioned, two dams had breaches that were sufficient to drain their ponds, while an additional four ponds (136, 186, 221, 277) had shallow breaches that did not interfere with the pond's normal water retention capabilities (i.e. water level with bottom of trickle tube). Ponds 268 and 277 were Kansas Watershed District structures (Jean and Sanderson 2015, Figure 15). These two Kansas Water District dams are part of the approximately 1,539 PL 83-566 and other public funded dams that normally follow earth dam planning and design procedures outline by NRCS in a number of technical publications (e.g. NRCS 2005).

Figure 13. Few dams also function as a road such as pond 172 (high point elevation 2.505 m above water level, 0.635 above spillway low point).

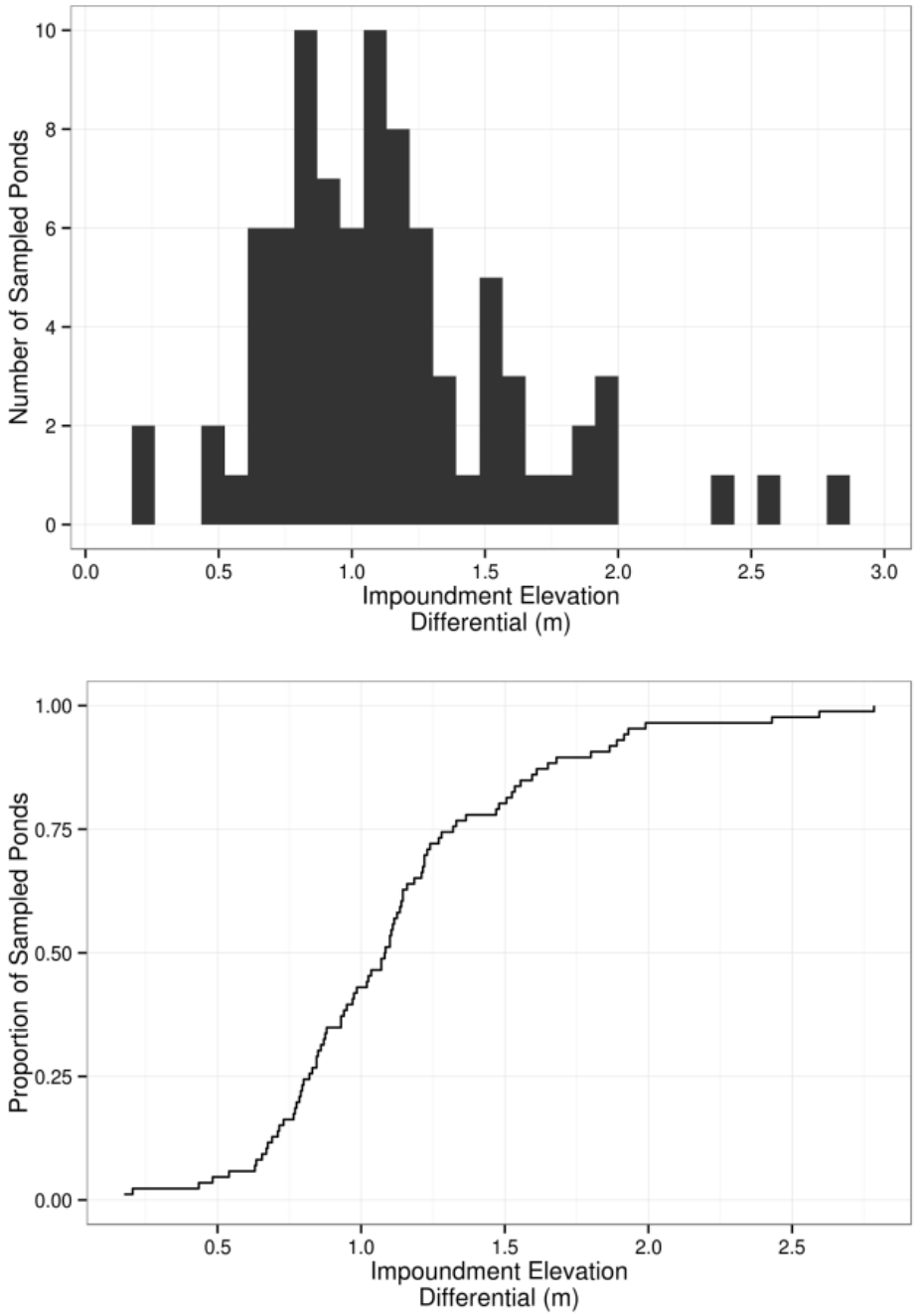


Figure 14. Histogram and CDF of the elevational differences (m) between the spillway low point and dam high point.

Figure 15.Standing at the pond-side base of a large dam (high point elevation 6.2 m above water level, 1.99 m above spillway low point) which is the watershed dam of pond 268.

Trickle tubes (e.g. overflow pipes, drain pipes, stand pipes)

Three ponds had no trickle tubes; excess water drained over the spillway. At two ponds the inlet end of the trickle tube was not visible, but water discharge was observed from the outlet end of the tube indicating that the tubes were functioning. The majority of the standpipes on the remaining 91 ponds (> 90%) were simple pass-through tubes or pipes without valves (Figure 16), while four were bottom withdrawal stand pipes without values, and two were bottom withdrawal stand pipes with valves.

Pipe diameter measurements were not part of the original sampling regime but were added later so that only 47 ponds were assessed for this parameter. The diameter of trickle tubes and pipes ranged from 20 to 150 cm (8 to 60 in.) with the majority of diameters less than 40 cm (≈16 in, Figure 17).

Figure 16. A beveled cut in the stand pipe opening may have been the landowner's attempt to reduce debris buildup on stand pipe at pond 279. The opening of the 30 cm diameter pipe was 2 mm above the surface of the water.

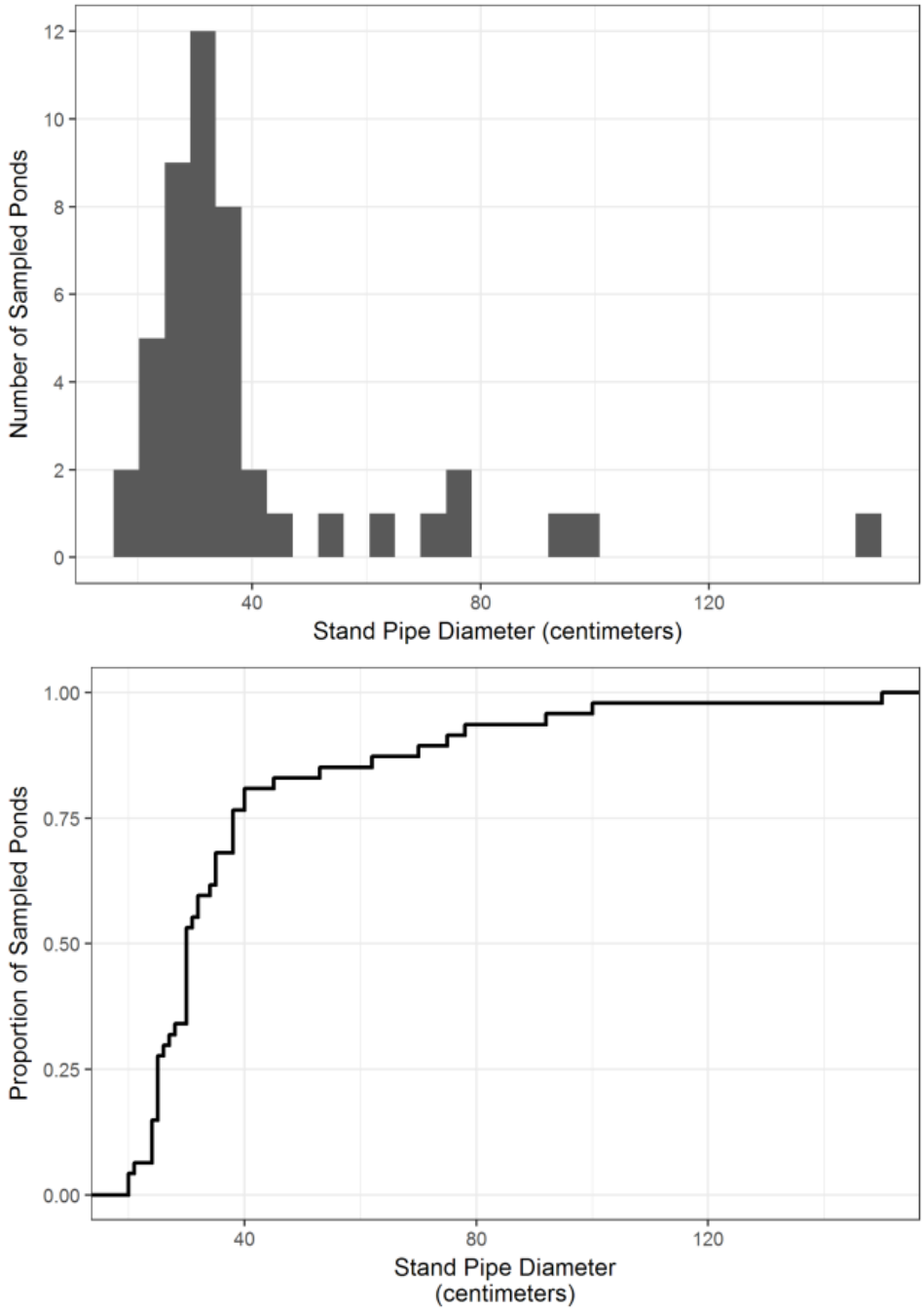


Figure 17. Histogram and CDF of stand pipe diameters (cm).

Of 86 measured trickle tube elevations, 50 were at or above the water level indicating that over half of the sampled ponds were at or below the designed water retention levels. Some pond water levels were fairly low with the water level in one pond being 0.56 meters below the discharge height of the trickle tube. The remain 36 ponds with recorded water levels were discharging water via functioning trickle tubes with the highest water elevation occurring 1.21

m above the normal retention level. However, most of these discharging ponds (56%) had water levels at or below 10 cm of the trickle tube bottom (i.e. the retention area). About 88% of the 81 pipes checked were found to be functional while 9 were determined to be non-functional. Trickle tubes were considered non-functional if it was determined that they no longer could regulate designed retention levels. Non-functional tubes were normally the result of tube blockage from debris or leaking pipes that were thought to compromise of water regulation. Most instances of debris blocking trickle tubes were the result of beaver or muskrat activities or lack of screening devices such as trash racks.

Spillway

All but one of the spillways assessed were placed in grass/rocked category, and eventually all those assigned this category were noted to have only grass cover and no rocked spillways were observed (Figure 18). The single spillway coverage not assigned the grass/rocked category was identified as a bare soil spillway condition. This particular spillway was in fact mostly bare soil because of extensive cattle trailing and use but did retain some vegetated areas. No signs of erosion were noted on this bare soil spillway, but about 6% of the spillways surveyed did show some limited signs of erosion with most of the erosion occurring in limited areas along the course of the spillway length (Figure 19Figure 18).

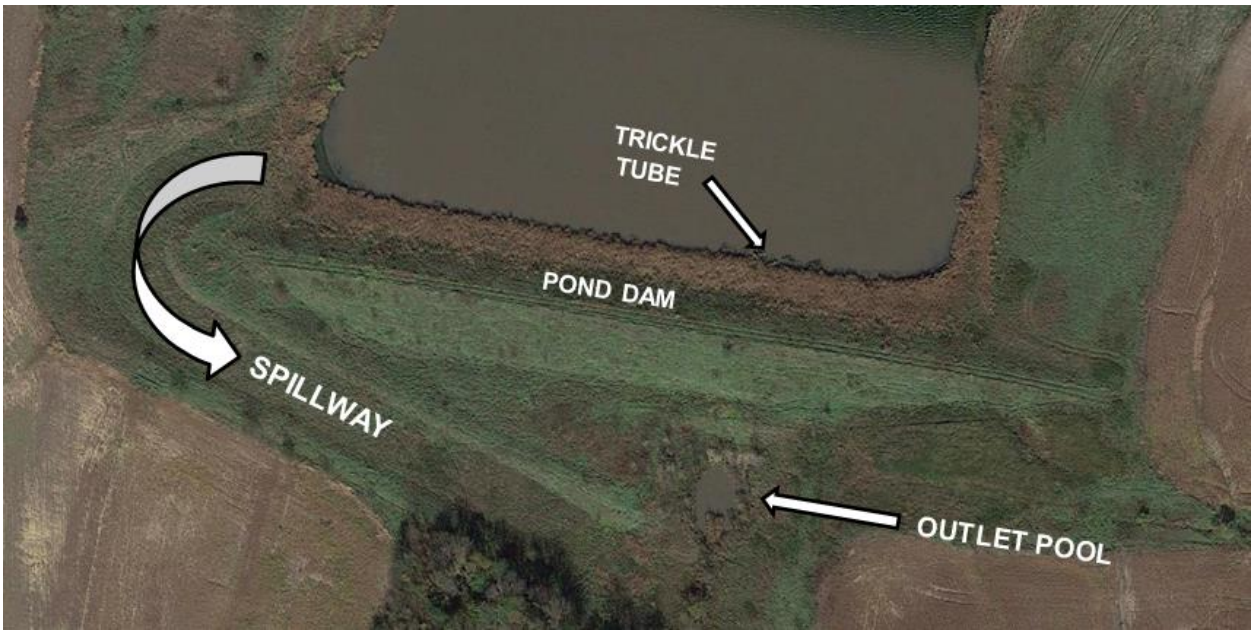


Figure 18. Delaware River Basin farm pond showing spillway system, pond dam, and outlet structures.

Figure 19. At pond 262, the eroded spillway is seen in the left foreground.

Outlet structures

The outlet ends of the trickle tubes or stand pipes varied in their condition, with many having various length of the pipes exposed due to dam erosion around and above the original exit site of the pipe. In a few cases pipes had collapsed and cracked open letting water leak from the pipe, thus exacerbating the erosion that had apparently allowed the pipes to originally shift and crack (Figure 20, Appendix F). In nearly all cases the outlet ends of the trickle tubes were perched such that the ends were suspended above the base of the dam and the channel bottom creating a splash drop and pool area. This practice creates small to large (≈ 1 – 5 m diameter) splash pools that often retain water past the discharge events and thus facilitate the development of small, temporary, aquatic and wetland environments. These are discussed as part of the next section on outlet areas. Pipe diameters at the outlet end of the trickle tubes were not measured but assumed to be the same diameter noted for the inlet end of the pipes.

Figure 20. Pond 183 standpipe, the underside of which is cracked allowing water to pass. Also note the lack of a trash guard that would reduce debris from plugging the outlet.

Outflow area

As with the upper pond area (i.e. inflow area) assessments, we selected a 40m x 40m subsample plot located immediately below the dam structure to characterize general physical environmental conditions associate with this pond zone (Figure 21). Within the subsample plot which typically brackets the outlet structure and often occurs in the drainage channel, we recorded presence/absence of livestock, and areal estimates of wetland plants, woody vegetation, grasses and annuals, bare soil, and crops. As with the inflow areas, channel width and depth and stability conditions were measured and assessed (e.g. sedimentation, erosion, retreating).

Figure 21. Pond 184, the area below the dam as photographed from on top of the dam.

At 35 of 96 ponds (36%), livestock use was evident in the downstream plot, which was just slightly less than that noted for the inflow areas (39%). This was expected as when livestock access to a pond was observed, access was unrestricted such that all areas comprising these pond complexes were typically exposed to livestock use and damage. Most of these (31) showing recent livestock presence had an outlet channel, but only 26% of these channels were characterized as unstable (i.e. highly eroding banks or head cutting).

Of 95 ponds, the majority (87%) had well developed outlet channels, 20 (24%) of which were classified as unstable. As noted above this number compares favorably with the 26% of outlet channels that were both unstable and exposed to livestock use. Downstream bankfull channel widths ranged 0.10m to 15m, while channel depths ranged from 0.08m to 8m (Figure 22 and Figure 23). Over 80% of these outlet channels had bankfull widths greater than 5 m and were 2 m or more in depth which approximates the channel size of many headwater streams in Kansas. Contrast this with the upstream (inflow) channels, which were smaller and with fewer (7%) examples of unstable channel conditions. We believe that pond outlet structures (e.g. trickle tubes, stand pipes, spillways) concentrate the volume and rate of discharge which accelerates the process of stream head cutting and down cutting.

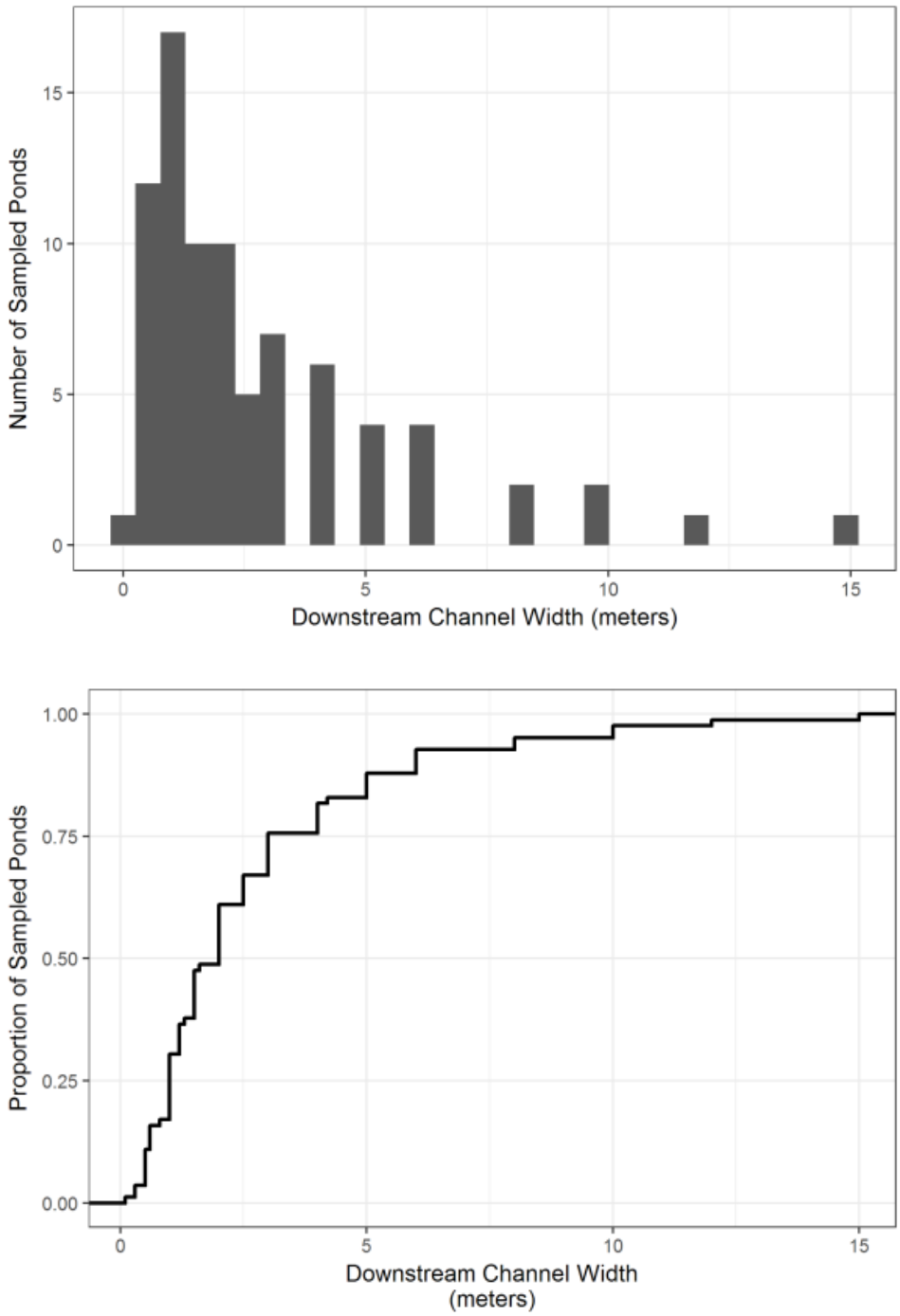


Figure 22. Histogram and CDF of downstream channel width showing that most downstream channels are < 4 m wide.

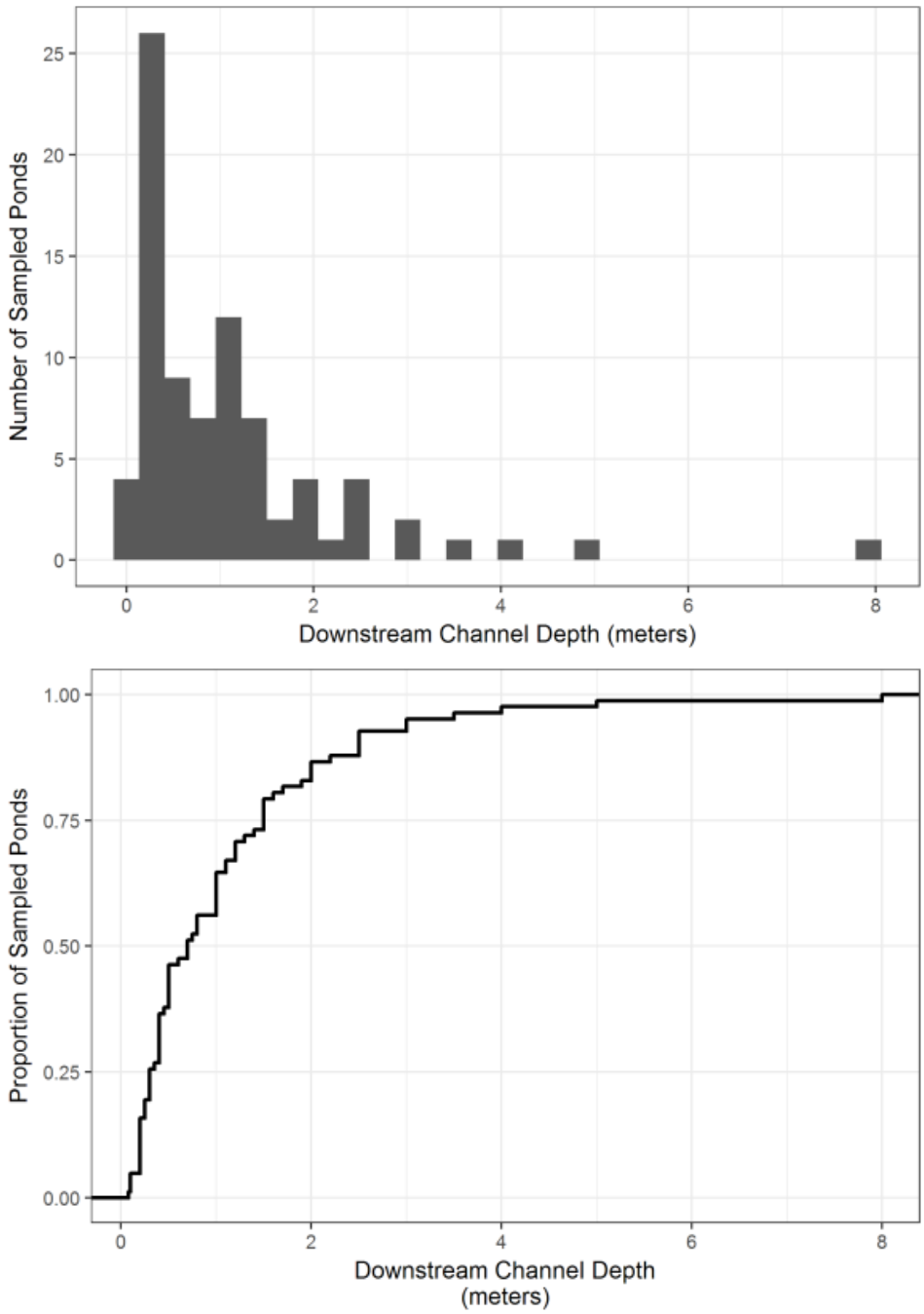


Figure 23. Histogram and CDF of downstream channel depth showing that most downstream channels are < 2 m deep.

Cover types in the outlet plots (Table 3) was similar to the cover types in the upstream plots (Table 2), though more ponds had wetland cover in the upstream plots (76%) than in the downstream plots (56%). In addition, in very few instances was wetland coverage a significant component of the vegetation of these outlet areas, with only 2% of these plots having 50% or more wetland plant cover. In contrast, 14 % of the inflow plots exhibited wetland plant cover of at least 50% which in part may be the result of more frequent wetting as water levels rise

with runoff conditions and these areas tend to occur on lower slope areas immediately adjacent to the pond retention extents.

Table 3. Cover types and estimated coverage in the 40m x 40m outlet plot area.

cover type	# ponds surveyed	% ponds with cover type	max % area in cover type	% ponds with over 50% area in cover type
crops	96	8	80	2
wetland plants	91	56	60	2
woody	96	91	100	50
grasses & annuals	92	98	100	60
bare soil	95	51	35	0
water	84	67	90	6

The least common ground cover in outlet area plots was crop, with just 8% of all plots with any crop. However, one of the two plots dominated with crop cover had 80% of its area in crop. Woody (i.e. trees and shrubs) and grass/annual cover types were very common, occurring in over 90% of all plots surveyed, with 50% or more of all plots being dominated by these cover types. Bare soil was noted in over half the sites but seldom accounted for large areas. The highest coverage of bare soil in any plot was 35%, but amounts this high were rare.

About 67% of all outflow plots had surface water with some cover estimates as high as 90%, but most water coverages were much less. In fact, only 7% of the plots examined had 50% or more areal water coverage, and these high coverages were the result of upstream dams and ponds. More often percent water coverages were much lower and represented outlet pools and water-filled drainage channels associated with the pond discharge events. Wetland and water coverages often co-occurred while other plots had one or the other coverages. The occurrence of wetland plants without measurable amounts of water coverage could merely indicate dry, non-discharging periods in which plants persist. Conversely, sometimes water coverage was noted but wetland plant cover was not observed, suggesting the prevailing hydrological conditions in the outlet plots could not support wetland plant species or enough wetland plant coverage to record.

<u>Pond retention area</u>

The pond retention area is essentially the pond area at the water level that is equal to the bottom of the inside diameter of the trickle tube or stand pipe. During low water periods this retention area would be the surface area of the pond plus whatever “freeboard” shoreline area occurs below the bottom of the trickle tube. Sometimes during extensive runoff events the pond surface area can equal the retention area plus whatever detention area that is flooded with excess runoff and pond water. Because large runoff events occur infrequently in this area of Kansas, most sampling events occurred when water levels equaled or were less than retention levels so that the persistent shoreline conditions could be evaluated. As previously noted most of the ponds sampled had water levels at, below, or slightly above (≤ 10 cm) the

normal retention levels making assessment of the shoreline retention feature easier to visually assess and measure. We assumed that the detention area would normally include a dry or saturated soil zone since high runoff events that might flood the detention areas would be infrequent on an annual basis. In part, this assumption is based on the fact that annual heavy rainfall frequency (and by inference runoff frequencies) is very limited in this region with the annual maximum rainfall frequencies by month for eastern Kansas varying from about 20% to near zero (Perry 2008, Villarini *et al.* 2011). Assuming all heavy rains produced some runoff to these small ponds and that with net evaporation rates of about 6 – 8 inches (KDA 1996) pond levels would normally recede during non-runoff periods, it seems reasonable to assume that long-term water retention levels varied from full to less than full levels.

It should be noted that in Missouri, evaporation losses from ponds could be as high as 61 to 91 cm during the summer, suggesting even more water loss between runoff events (Hicks and Pierce 2014). The other assumption we made was that when pond levels did reach detention levels (flood levels), outlet structures were capable of draining pond levels back to retention levels quickly (e.g. hours – days). When these terrestrial/aquatic interface zones within the retention zones were de-watered they retained a unique physical signature that included the presence of facultative wetland plant species, saturated to dry pond sediments (e.g. bare soils), and the accumulation of aquatic debris and detritus. Based on these assumptions and noted field conditions, identifying these portions of the retention areas that were de-water (i.e. shoreline zone) was easy and repeatable.

We measured a variety of features in the pond retention area – that region normally occurring between terrestrial vegetation and the water's edge when the water levels were at or below the retention level. We noted if the water was low and exposing the pond retention area, or if the water was high and covering this retention area. We also recorded livestock and gully erosion presence in this retention area, and percent bare soil, grasses and annuals, wetland vegetation and woody vegetation when they did occur within these shoreline zones.

The pond retention area was full or nearly full of water for most of the ponds for which this was noted (61 or 75 ponds, Figure 24). As mentioned previously, livestock was present in the pond retention area of 34 ponds.

Figure 24. The pond retention area was full and water high into the terrestrial vegetation at pond 179.

The most commonly occurring vegetation cover class in the pond retention areas was wetland vegetation (63%) followed by the grasses/annual class (60%, Table 4). It should be noted that the grass/annual vegetation category or class may have included some plant species listed as FAC (Facultative), indicating that these species are equally likely to occur in wetlands or non-wetlands (Lichvar 2012). It appears that wetland vegetation is a common component of the vegetation cover in the shoreline zones of most pond retention areas. In some pond retention areas, either the wetland or grasses/annual coverages were as high as 95 -100% while 23% of all sites had more than 50% of the area in wetland vegetation compared to 15% of sites dominated by grass and annual cover. Woody vegetation was found along the margins of 34% of all retention areas but seldom made up a large percent of the vegetation cover for any one pond. The limited extents of woody vegetation could be attributed to several facts including control and removal by land owners, limited species tolerance to infrequent but saturated soil conditions, and presence of beavers in some ponds. The amount of bare soil coverage was highly dependent of current and past water levels with dry periods resulting in water level losses due to evaporation, transpiration, infiltration and dam leakage thus exposing pond sediments that have not become vegetated (Figure 25). Wave action and sediment movement may also have prevented or slowed establishment of vegetation in these bare soil areas.

Table 4. Cover types in the shoreline zones of the pond retention area.

cover type	# ponds surveyed	% ponds with cover type	max % area in cover type	% ponds with over 50% area in cover type
wetland plants	83	63	100	23
woody	82	34	50	1
grasses & annuals	84	60	95	15
bare soil	86	45	100	21

Figure 25. The pond retention area of pond 268 was not full as evidenced by exposed mud flats (i.e. bare soil class), wetland vegetation, and water elevation below bottom of trickle tube.

Littoral and deep water zone depths

At each pond, five transects were selected and spaced along the dam and the two pond shores originating from the dam. Transect water depths were measured with a survey rod at 1, 2, and 3m from the water's edge (Figure 26). These measurements were considered to be reasonable estimate of the near shore or littoral zone areas of the ponds. Also at these transects three casts of a wireless depth transducer were made with a fishing rod and reel to measure depths at greater distances from shore. The mean cast length was about 16 m (≈ 52 ft) so that depths were typically taken at 16 m then 12 m and a final at about 8 m as the transducer was reeled in along the transect.


Figure 26. Measuring pond depths around the perimeter with a survey rod (pond 279 left) and remote depth transducer on fishing rod (pond 200 right).

Depth statistics for both the littoral zones (1 – 3 m measurements) and deeper zones (i.e. transducer casts) are provided in Table 5. This gave a maximum of 30 depth measurements per pond; however, occasionally we were not able to obtain all deeper water measurements, most often because aquatic vegetation blocked the sonar transducer signal. The range of mean pond depth determined using all transect measurements was found to vary from 0.52 to 1.99 m. These mean pond depths may to be conservative estimates of mean depth since half of the measurements were restricted to the first three meters of the pond margin. However, these near shore and shallower depth measurement biases, if any, may well of been offset by the transect measures from the dam face that is almost always the area of deepest depths.

Table 5. Pond depths (m) measured at various intervals from the water's edge.

Distance from water's edge	Count	Minimum	Mean	Median	Maximum
1 m pole	475	0.04	0.25	0.22	1.10
2 m pole	475	0.08	0.41	0.38	1.40
3 m pole	461	0.10	0.57	0.53	2.30
Ave. littoral zone (1-3 m) depths	1411	0.04	0.40	0.35	2.3
closest cast with depth finder	434	0.09	1.15	1.10	3.20
next farthest cast	456	0.22	1.56	1.40	5.00
farthest cast	470	0.35	2.01	1.80	5.60
Ave. deep water zone depths	1360	0.09	1.58	1.40	5.6
Average of all depths	2271	0.04	0.98	0.74	5.6

The littoral zone was operationally defined as occurring in the first three meters of water from the shoreline at each transect. This probably was an overestimate of the littoral zone along the dam (i.e. single transect) because of the high slope and rapid transition to deeper water created by the dam structure. The mean littoral value depth for our study ponds was 0.40 m, and the deep water pond zone average (> than 3 meters from shore (typically 8 meters) was 1.58 m. Median depth of littoral zones for the study ponds was about 0.35 m, which represents rather shallow zones where light penetration can reach the bottom under low to moderate water clarity conditions (see water quality results for turbidity discussion). These one to five-acre study ponds were shallow with a mean and median value for the deep-water zone of 1.58 and 1.4 m. The deepest depths (maximum depth = 5.6 meters) were measured nearer the dam and often at or near the middle of the pond. Most average depths were much shallow than land owner estimates, which tended to focus on just the deepest depth and not an estimate of the mean or median water depths. Examination of the CDF in Figure 27 shows that the average pond depths for over 90% of the study ponds were less than 1.5 m (< 5.0 ft).

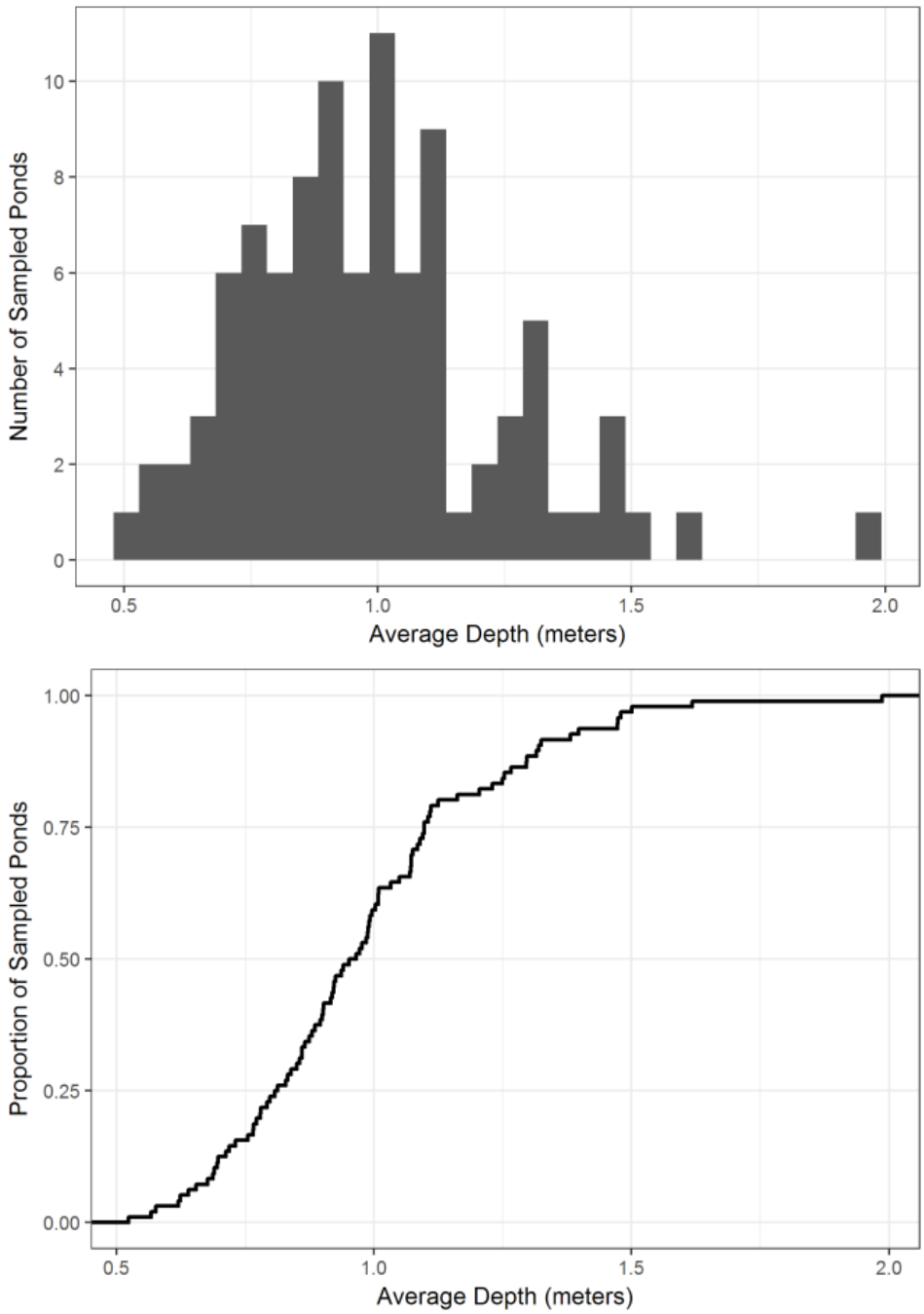


Figure 27. Histogram and CDF of pond depths (up to 16 m from shoreline).

The University of Missouri (Columbia MO) extension service bulletin entitled "Managing Ponds and Lakes for Aquaculture and Fisheries in Missouri Ponds: Pond Construction and Management Considerations" suggests that ponds should have a mean depth of about 1.5 to 1.8 m (5 – 6 ft.) but not exceed 3.0 m or more (≈10 ft.) anywhere in the pond (Hicks and Pierce 2014). In addition, Missouri recommends that half or more of the pond area should be about 1.2 to 1.5 m deep (4 – 5 ft) deep to allow for bottom foraging and feeding even in summer months when low dissolved oxygen levels are prevalent in deep water habitats. The authors

indicate that these depths will most likely sustain fish populations under most drought conditions. Estimates of summer evaporation rates indicated that Missouri ponds could experience evaporative water losses of up to 1.27 cm (0.5 inches) per day with cumulative summer time losses of 60 to 91 cm (2 – 3ft). Similar pond depth recommendations are found in the online publication by the Nebraska Game and Parks entitled "Nebraska Pond Guide" (https://outdoornebraska.gov/wp-content/uploads/2015/11/PMGS_Construction.pdf). As with the Missouri extension pond construction bulletin, Nebraska suggested that ponds from 1 to 5 acres should support multiple uses. The Nebraska guide also suggested that about 25% or more of the pond should be at least 3.0 to 3.7 m (≈10 – 12 ft) deep in depending on location within Nebraska. Additionally, the Nebraska guide recommended that no more that 25% of the pond area be less than 1.2 m (≈4 ft) deep and about 50% should be 2.4 m (≈8 ft) or greater in deep to help control extensive macrophyte growth.

Wildlife

We noted evidence of wildlife at or near each pond, as follows:

- 46 ponds had fish (dead, alive, or owner-reported). It should be noted that fish accounts were determined by direct observation, which was difficult in many ponds due to water clarity and macrophyte cover. We assume our estimation of ponds with fish is low as typically these rural ponds are all stocked with some combination of species from the sunfish and catfish families (Centrarchidae and Ictaluridae, respectively).
- 25 ponds had muskrats and 22 ponds had beavers (of these 6 had both). Beaver and muskrat signs were very apparent when these mammals were noted to occur in these pond environments. Beaver bank lodges were sometimes found along the face of the dam where these excavations could lead to dam leakage and other impacts at the dam structures.
- 73 ponds had reptiles and/or amphibians (Figure 28). Amphibians were more often noted than reptiles (i.e. snake species). Frogs (i.e. Ranidae and Hylidae) and turtles mostly those belonging to the families Chelydridae and Emydidae were the commonly occurring amphibians. No salamanders were observed either as adults or larvae. While no attempt was made to identify each species of reptiles and amphibians found in our surveys, in southeastern Minnesota as many as 10 amphibian species have been found to use small, artificial agriculture ponds (Knutson *et al.* 2004).
- 47 ponds had semi-aquatic and/or shoreline birds. A variety of waterfowl and shorebirds were observed including ducks,
- 6 ponds had freshwater mussels (Bivalva). Only mussels occurring in the family Unionidae were tallied and no attempt was made to identify the very small but more widely occurring fingernail clams (Sphaeriidae). Prior research on Sphaeriidae in Kansas found fingernail clams to be common in waterbodies throughout the state including small artificial ponds (Mackie and Huggins 1983).

Figure 28. Northern banded water snake (*Nerodia sipedon*) at pond 241.

Livestock

We noted evidence of livestock presence (past and current) in the upstream and downstream plots and within the pond retention areas (Table 6). Cattle were the only livestock observed having access to the study ponds and the immediate pond areas. There was no sign of livestock occurrence near the pond environment at 52 ponds, while at 26 ponds we observed livestock at or signs of past presence in all three of sample areas. Presence of cattle in any one area of the ponds usually indicated their presence in all other areas (36 upstream, 34 pond retention area, 35 downstream). Continuous livestock access to the pond environment along with large numbers of cattle had notable impacts on the shoreline, general water clarity, and other areas within the retention areas. Trailing and cattle aggregation along the shoreline and into the shallow zones of ponds resulted in heavy tramping and elimination of vegetation and subsequent erosion and disturbance to the littoral zones. Tramping near and within the pond resulted in bottom substrate and soil disturbances with deep hoof depressions, muddy water, and suppressed vegetation growth that contributed to erosional processes within the pond complexes.

Table 6. Presence and absence of cattle with presence and percent of woody vegetation found in three areas of the ponds: upstream 20m x 20m plot, pond retention area, and downstream 40m x 40m plot.

	cattle present			**cattle absent**		
Area	**# ponds**	**% ponds with woody veg**	**ave % woody veg in area**	**# ponds**	**% ponds with woody veg**	**ave % woody veg in area**
Upstream plot	36	78	33.9	57	91	42.5
Retention area	34	29	3.1	62	29	6.5
Downstream plot	35	86	37.8	61	93	53.3

More ponds without cattle than with had woody vegetation in the upstream and downstream areas. In the retention area, 29% of ponds had woody vegetation regardless of cattle presence. Where cattle were absent, the percentage of woody vegetation in these three areas was higher than if cattle were present. These trends suggest that cattle contribute to woody growth control.

In most cases where cattle were observed to have access to the pond environments the resulting tramping and shoreline erosion was limited both in severity and in extent of shoreline accessed (Figure 29 and Figure 30). Smaller herd size and additional water sources other than the study pond may have helped limit pond impacts. The total time cattle were allowed access to the ponds for drinking was also considered a fact that could limit observed erosional impacts to some ponds. Current research suggest limiting livestock access to stock ponds to help maintain both water quality and structural integrity of the pond itself (Deal *et al.* 1997, Wolinksy 2006, Blocksome and Powell 2007, Figure 31).

Figure 29. Pond 230 with cattle in the retention area, and a dredging pile (to the left).

Figure 30. Livestock watering area in the downstream plot of pond 107.

Figure 31. Pond 215 landowner uses best management practices to fence livestock out of pond (left) while a solar-powered pump (right) provides water to the cattle.

Macrophytes (Aquatic Vegetation)

We visually estimated the percent of shoreline length that had macrophytes in the accompanying littoral zone at or below the water level. In this phase of our study no attempt was made to identify macrophytes to genus or species for two reasons; first, not all field crew members were qualified to identify macrophytes, and second, time constraints prevented the detailed effort it would have taken to survey the whole pond perimeter to identify all taxa present. Therefore identifications were only taken to the lowest taxonomic level necessary to ensure that the observed vascular plants were macrophytes.

In order for a shoreline area to be added as part of the percent estimates, rooted macrophyte densities had to equal or be greater than one macrophyte plant or plant clump per $0.5m^2$ of bottom area. Only nine ponds had no measurable macrophyte abundance (Figure 32); almost half (41) had 50% or more of their littoral zones (i.e. shoreline length) with macrophyte densities large enough to measure (Figure 33). Nearly 20% (17) had over 90% of their littoral zone with macrophytes, and 75% of all ponds had at least 10% of their shoreline with macrophytes (Figure 34). Of the nine ponds with no observed macrophytes, none had gully erosion in the pond retention area, and only two had evidence of livestock. In the 80 ponds with macrophytes, 11 had gully erosion in the pond retention area, while 32 had evidence of livestock.

Figure 32. No macrophyte densities extensive enough to count as a percentage were observed along the perimeter of pond 145.

Figure 33. Macrophytes densities occurred along 100% of the perimeter of pond 136.

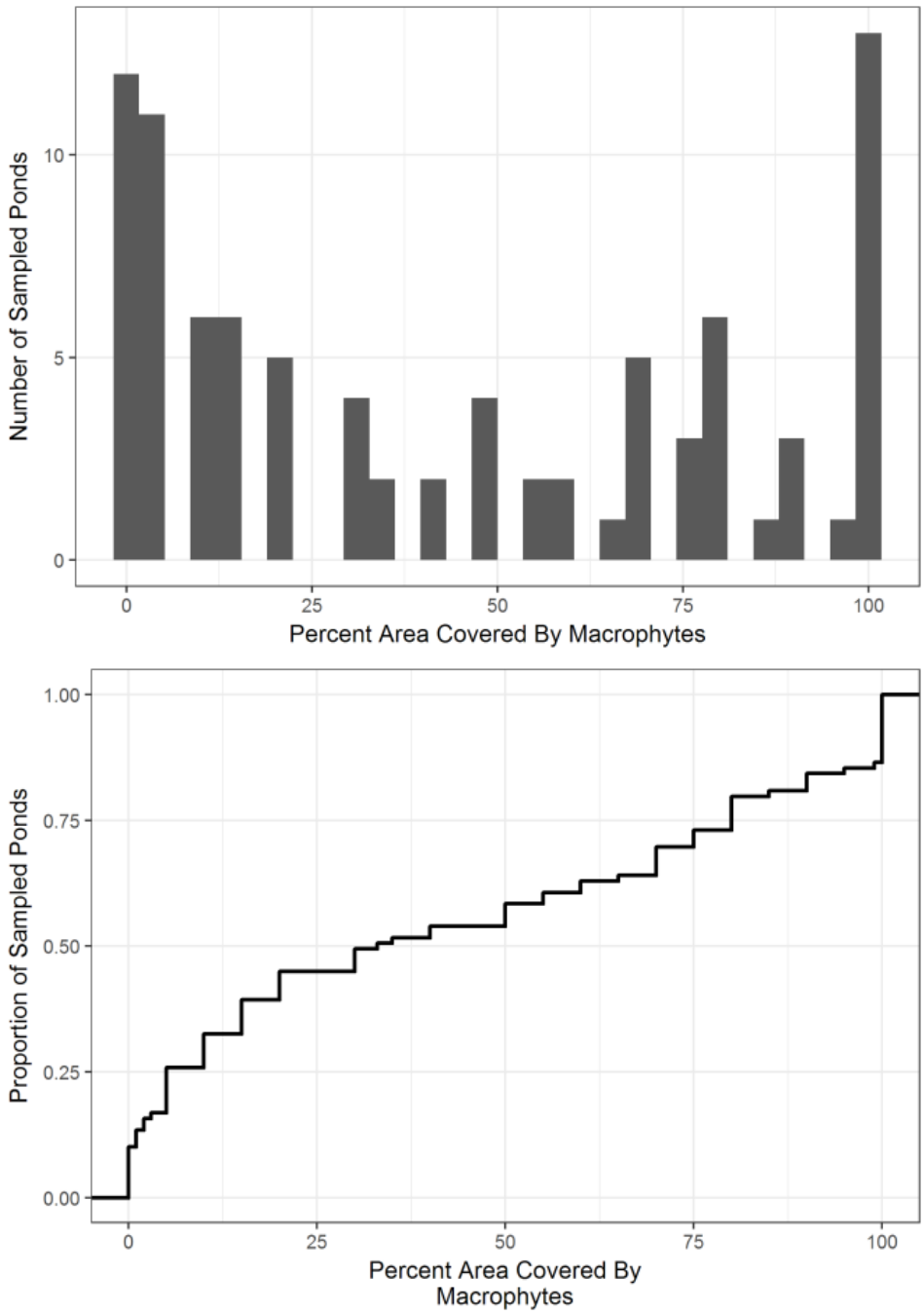


Figure 34. Histogram and CDF of percent shoreline covered by macrophytes.

WATER QUALITY AND FLORISTIC ASSESSMENT

In August 2016 we revisited 20 ponds considered among the best candidates for wetland conversion. At these ponds we measured *in situ* water chemistry and performed a detailed assessment of vegetation communities associated with the pond complexes aside from their watersheds. As part of the original survey of the nominal 100 randomly selected ponds, field crews determined 44 ponds to have above average potential for wetland conversion based on

attributes within the downstream 40m x 40m wide plot that extended from the base of the dam. This outflow area was selected for overall evaluation of conditions that might favor wetland development either in its current form or with some additional structural work because it was located downstream of the pond. This downstream location provides two important features that could promote the development of and use of a wetland or wetland functioning environment. First, being located downstream would allow for water availability to the wetland to promote its long-term existence. Secondly, these areas are often in a position to intercept spillway releases that are often high in sediment and other contaminates. Desirable attributes of these outflow areas were flat, low gradient terrain; no or little channel or gully development; presence of some wetland vegetation or propagules; evidence of persistent soil moisture; and absence of livestock watering areas.

After completing the field work and initial examination of the data we decided that in selecting ponds for further study we needed to also consider potential wetland development opportunities within the pond itself, and also the inflow areas. However, it was noted that inflow areas that had considerable macrophytes and wetland plants were nearly always (> 90%) associated with ponds with similar macrophyte/wetland plant conditions. Because inlet conditions were reflected in the overall observed pond conditions, we decided to look at only pond condition, and not inflow areas, to select likely candidates for further study. Prior to the study, we believed the pond and pond retention areas had potential for wetland development; that supposition was supported by observations of shoreline vegetation, macrophyte abundance, and pond depth measurements. In fact, over half of the ponds surveyed already had extensive macrophyte communities (≥ 50% shoreline coverage) and fairly extensive shallow zones. Additionally, we observed that some of the inflow areas had wetland features that could be enhance with minimal physical efforts.

Therefore, we decided that all three areas of the pond complex should be considered for potential wetland enhancement or development opportunities because each area including the pond itself had separate attributes and conditions that were related to wetland function development. To be more inclusive in our in depth studies we selected 10 ponds based on outflow plot attributes, as well as 10 ponds chosen for attributes and conditions associated with the pond retention areas. The presence of macrophytes was weighted heavily in the selection of sites (pond areas), as presence of facultative and obligate wetland plants indicates that wetland conditions were being met and suggest a ready source of plant seeds and propagules. Shallow water conditions were another factor that was given more selection weight because shallow zones more frequently have light penetration to the bottom substrate to support photosynthesis in rooted macrophytes. Shallow areas are also prone to more frequent hydrological variability that is often associated with some wetland types. The selection process for these two categories of high potential wetland areas follows.

Selection Based on Downstream Plot

Forty-four ponds were identified on the field evaluation and assessment sheets as having high potential for wetland conversion or modification based on outflow plot characteristics (Figure 35). These ponds were then evaluated for both macrophyte abundance and overall depth.

Seventeen ponds with over 70% of their shoreline in macrophyte growth were then selected from the original 44 ponds with high downstream wetland potential. From these remaining 17 ponds we selected the shallowest ponds based on the values found for the littoral and deeper water areas and selected the 10 with overall deep-water zone depths less than 1.75 meters. One of the initial 10 ponds (pond 221) was slated for removal by the land owner, thus replaced with the next shallowest pond (pond 218 at 1.82 m).

Figure 35. Pond 215, marked as a potential site for wetland conversion, showing the area below the dam as photographed from on top of the dam (left). A close up of *Equisetum hyemale* (scouring-rush), a highly recognizable wetland plant (right).

Selection Based on Overall Retention Area Attributes

The remaining 52 ponds not marked for potential conversion based on outlet plot attributes were the assessed for the 10 best candidates for additional study. In selecting from this group we again relied heavily on the macrophytes and wetland plant abundances that we already had observed as well as the overall shallow nature of the pond environment. Of the initial 52 ponds considered in this selection process we selected 15 that had at least 70% of their littoral margin in macrophytes/wetland plants. From these 15 we selected the 10 ponds that were less than 2.25 m at their deepest measured points. We did not attempt to include ponds that had inflow plots that were observed to have strong wetland characteristics because of the limited sample size selected for further study. Sample size was partly based on estimated adequacy (20% subsample), economics, and time.

Pond Water Chemistry

At each pond, *in situ* water chemistry was measured at three sites – from the center of the dam, and the left and right shorelines (as facing the pond from the dam, Figure 36). In the resulting data, we first examined the entire dataset of 60 measurements (20 ponds x 3 sites). All *in situ* measurements were normally distributed except for turbidity (even with the highest value removed), oxidation reduction potential, and pH. The strongest correlations were among parameters that closely related: conductivity, salinity, and total dissolved solids (TDS) (Table 7). Salinity and TDS values are actually derived variables based on conductivity. Oxidation Reduction Potential (ORP) was negatively correlated with these three parameters as was turbidity. This relationship between ORP and turbidity might have been related to limited light

penetration, which would reduce photosynthesis, allow a more reducing environment to develop, and cause ORP values to decrease. Low and negative ORP measurements are commonly found in wetlands because of the high amount of bacterial activity and the anaerobic reduction of decomposing organ matter. Turbidity was also significantly correlated with TDS (0.36), suggesting that conditions promoting changes in turbidity were similarly affecting dissolved solids or merely that TDS was contributing to the overall turbidity measures. Turbidity was one of the water quality parameter that varied greatly both between ponds and within ponds (Table 7). For example, in one pond the differences between the three *in situ* turbidity measurements was nearly 500 NTUs. Therefore, to better characterize overall *in situ* pond conditions, parameter measurements were averaged for each pond (Table 8, Appendix G).

Figure 36. Using a pole to extend the Horiba U-52 water quality monitor beyond the aquatic vegetation at pond 23.

Table 7. Descriptive statistics of in situ water chemistry measurements taken at 3 sites at each of 20 ponds.

Parameter	Minimum	Mean	Median	Maximum
water temperature C°	23.94	27.57	27.63	30.17
pH	7.51	9.02	9.20	10.87
oxidation reduction potential (ORP)	-88.00	134.72	149.50	270.00
Specific conductivity mS/cm	0.04	0.21	0.20	0.45
turbidity NTU	2.00	42.26	16.35	685.00
dissolved oxygen mg/l	0.87	6.71	6.82	16.68
total dissolved solids g/l	0.03	0.13	0.13	0.29
salinity %	0.00	0.01	0.01	0.02

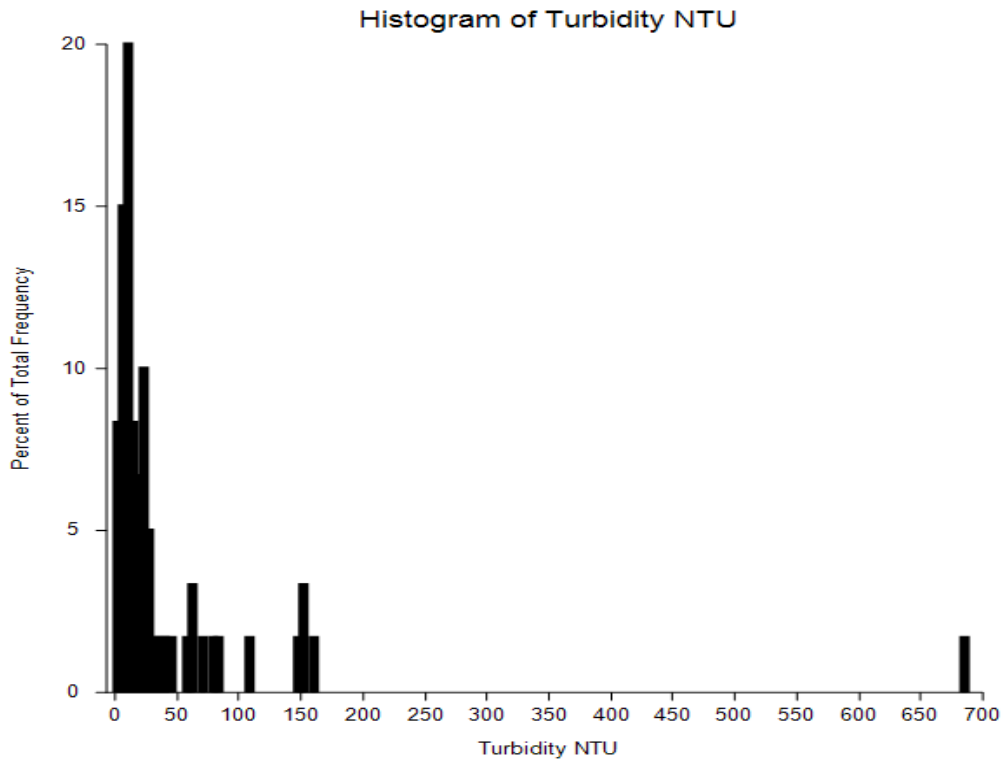


Figure 37. Histogram of individual turbidity values for all ponds.

Table 8. Spearman-Rank Correlation matrix of average per pond in situ water chemistry variables (retaining the highest turbidity value). *Significant at p<0.05.

Parameter	Salinity	TDS	DO	Turbidity	Cond	ORP	pH
water temperature C	0.01	0.06	0.82*	0.13	0.05	-0.21	0.76*
pH	0.23	0.11	0.80*	0.12	0.10	-0.21	
oxidation reduction potential (ORP)	-0.79*	-0.72*	-0.13	-0.41	-0.71*		
conductivity mS/cm	0.88*	1.00*	-0.03	0.41			
turbidity NTU	0.50*	0.40	0.15				
dissolved oxygen mg/l	-0.01	-0.02					
total dissolved solids g/l	0.88*						

Vegetation Survey

A vegetation survey was done at each pond, with 190 plant species recorded across all sites(Figure 38, Appendix I). The two most common species, each found at 18 of 20 ponds, were *Persicaria punctata* and *Salix nigra*. Fifty-eight species were rare—represented each by a single occurrence. Plant species were classified by provenance (native/introduced), wetland indicator status (obligate through facultative), coefficient of conservatism (0—10), longevity (annual, biennial, perennial), and habit (woody, perennial herbs, annual herbs, aquatic herbs, etc.).

Figure 38. Surveying vegetation communities at pond 257.

These classifications were used to calculate the following indices for each pond (based on all species and also natives only, Table 9):

- species richness – total number of species
- floristic quality index (FQI) – mean coefficient of conservatism × square root of number of species (either all species or native species only)
- mean conservatism coefficient – higher number represents more species restricted to higher quality areas
- mean wetland quality coefficient – lower number represents more obligate wetland species.

Table 9. Descriptive statistics of the vegetation indices calculated for the 20 ponds selected for further study.

Metric	Minimum	Mean	Median	Maximum
species richness (all)	23.00	41.80	41.50	72.00
species richness (native)	17.00	31.05	30.00	54.00
percent introduced species	13.04	25.00	26.06	34.62
FQI (all)	14.71	23.66	24.16	29.70
FQI (native)	18.19	27.36	27.34	34.58
mean conservatism (all)	2.88	3.74	3.67	4.74
mean conservatism (native)	4.00	4.98	4.96	5.80
mean wetland quality coefficients (all)	-1.47	-0.77	-0.66	-0.35
mean wetland quality coefficients (native)	-1.79	-1.01	-0.91	-0.41

The floristic quality index used in this study used coefficient of conservatism values and other assigned plant metric values for Kansas as some values are regionally specific (Freeman and

Morse 2002, Appendix H). Wetland indicator status values were obtained from the National Conservation Service at https://plants.usda.gov/core/wetlandSearch.

It has been shown that the Floristic Quality Index, also called the Floristic Quality Assessment Index (FQAI), does have value in identifying wetland condition (Herman 2001, Lopez and Fennessy 2002, Miller and Wardrop 2006). Lopez and Fennessy (2002) found that wetlands and the immediate zone around them tended to have lower FQAI scores and were dominated by plants associated with heavy agriculture and urban watersheds. For example, these authors found that for their study wetlands (n=20) in Ohio, FQAI (=FQI) scores for fairly high quality wetlands were typically 25 or higher. Our mean FQI score for our 20 selected ponds was 23.7 for the FQI (all species) and 27.4 for the FQI (native species only) (Table 9). The FQI also performed well in discriminating wetland condition in the Great Lakes coastal wetlands and consistently outperformed coefficient of conservatism indices in this capacity (Bourdaghs *et al.* 2006). These authors found that the performance results of indices that included or excluded introduced species (our all and native only FQIs) were nearly the same but they thought conceptually that introduced species should be included in the FQI. They reasoned that "introduced species are simultaneously a source of and a response to anthropogenic stress."

We examined relationships and patterns among these vegetation indices and other data. As expected, FQI (native) correlated with native species richness (0.76), while there was not a relationship between FQI (native) and the percent of introduced species (0.37, Figure 39). The wetland quality coefficient was examined with visual estimates (pond perimeter macrophytes, wetland plants in upstream and downstream plots) of wetland plants recorded during the first visit to each pond. The highest correlation (-0.48, p=0.0505) was that between percent of the pond perimeter with macrophytes and mean wetland quality coefficient (all species). The lower this coefficient, the more wetland species present, so the negative relationship indicates that there were more obligate wetland species in ponds that had a higher percentage of the perimeter covered in macrophytes. There was some correlation between average pond depth and FQI (native, 0.55) and FQI (all, 0.57); however only FQI (all) was significant (p=0.049).

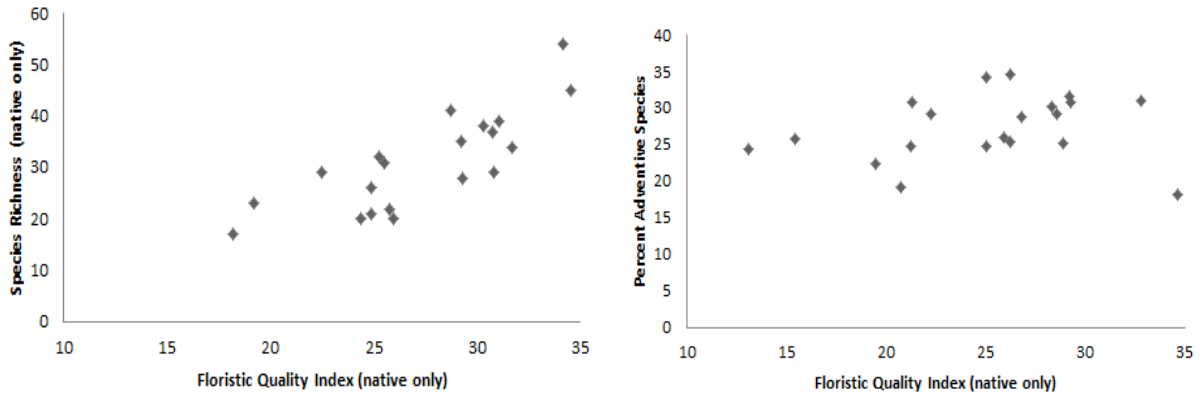


Figure 39. Relationships between floristic quality index (FQI, native only) and species richness (native only, left) or percent adventive species (right).

Pond vegetation types were compared to the Kansas native plant communities outlined in Lauver *et al.* (1999). Pond vegetation types did not precisely into the classification of Lauver *et al.* (1999) because they are essentially created wetlands—mostly herbaceous but a few substantially forested—in riparian zones. However, as expected, pond vegetation types appeared to be influenced strongly by the immediate riparian vegetation, as well as pond size and age. Despite the abundance of species found at just one pond each, most species found in this study are relatively common and widespread in wetland communities across eastern Kansas. Herbaceous-dominated vegetation types combined aspects of wet prairie (*Spartina pectinata-Eleocharis* sp.-Carex sp. Herbaceous Vegetation), freshwater marsh (*Scirpus validus-Typha* sp.-(*Sparganium* sp., *Juncus* sp.) Herbaceous Vegetation), and eastern cattail marsh (*Typha* sp. Midwest Herbaceous Vegetation). Woody-dominated types are most like the *Populus deltoides*-(*Salix amygdaloides*)/*Salix exigua* Woodland.

Relationships among plant cover variables

Significant correlations (Spearman-Rank, $p < 0.05$) among cover types in all three evaluated categories were those between percent woody vegetation and wetland plant or grasses/annuals cover (r provided), as well as the positive correlations in the pond retention area between soil, wetland plants, and grasses/annuals.

Upstream woody cover was negatively correlated with wetland plant cover ($r = -0.44$) and grasses/annuals ($r = -0.39$). One interpretation of the fairly weak relations between woody vegetation and the other cover classes is that canopy shading by trees and shrubs is reducing coverage of the ground cover classes (i.e. wetland plants, grasses and annuals). This seems logical as most of the woody vegetation classes assessed were dominated by trees at least 5 meters in height and had a mostly shrubby understory unless heavily influenced by livestock. Downstream woody cover also showed negative correlations with wetland plant cover ($r = -0.54$) and grasses/annuals ($r = -0.55$). Again the most logical biological explanation of these relationships is canopy shading by the woody vegetation resulting in lower percentages of ground cover classes. These correlation coefficients were somewhat stronger but still only suggestive of a strong relationship.

More problematic interpretations come when trying to evaluate possible causal relationships (positive) observed between woody cover in the pond retention area and wetland plant cover($r = +0.46$) and grasses/annuals ($r = +0.49$). As with the downstream correlations these positive correlation coefficients are weak and may or may not represent a causal relationship between the cover types. Why increases in woody vegetation cover would result in increases in both the wetland plant and grasses/annual vegetation coverages is uncertain. Possible explanations might include differences in management and/or hydrology between retention areas as compared to upstream/downstream areas .

Pond retention area bare soil was positively correlated with wetland plant ($r = +0.35$) and grasses/annuals cover ($r = +0.46$). Low water conditions that exposed bare soil (i.e. pond sediment) may have persisted long enough for both wetland and terrestrial plants to establish

along exposed shoreline. This may also explain why wetland plant cover and grass/annuals cover were also positively correlated (r = +0.62). This relationship, which was the strongest correlation coefficient found for vegetation cover relationships, was still fairly modest and if still significant as a regression coefficient would be an R^2 of +0.38. While of limited strength, these pond retention relationships among vegetation categories do suggest that these relationships are causal in nature—low pond water levels allowing colonizing of vegetation in some areas and bare sediment in others along the dewatered shore extent.

There were no statistically significant relationships between littoral zone depth (mean depth) and percent macrophytes; however this analysis may be problematic because percent shoreline cover was assessed for the entire shoreline while littoral mean depths were calculated only for the five transect sites. No significant correlations were found between turbidity and either percent macrophytes, cattle presence, or mean depth. This may not be surprising because this study was based on a single sampling event for all variables, including water chemistry, which can change significantly diurnally, seasonally, and with runoff and drought events (e.g. Hill *et. al.* 1962, Murdoch *et al.* 2000, Brainwood *et al.* 2004). The presence of cattle may or may not mean that the animals were actually in the pond disturbing the water and sediment, while macrophyte occurrence and densities are the results of longer-term conditions not measures in this study. On a few occasions we did see cattle in ponds creating highly turbid conditions at least locally. A number of studies report that small ponds heavily accessed by cattle can negatively impact shoreline and littoral vegetation, water quality, and aquatic life (Hubbard *et al.* 2004, Bilotta *et al*. 2007, Schmutzer *et al.* 2008, McDowell and Wilcock 2008).

WETLAND CONVERSION

Pilot Wetland Conversion Suitability Index (WCI)

To choose the 20 ponds for further assessment for conversion to wetlands, we sorted and compiled study data to select the shallowest ponds or downstream plots with the highest macrophyte or wetland plant coverage. We attempted to develop an integrated approach (i.e. numerical index) that factored characteristics from all three areas of the pond complex (pond, upstream, and downstream plots) that could aid in selection of pond complexes best suited for potential wetland conversion. This initial additive index would “score” or rank macrophyte coverage, pond depth, channel depths, and landscape slope within the upstream and downstream plots to obtain an index score. The pond complexes with the highest wetland conversion suitability index scores (WCI) would be those most likely to have a higher potential for development of wetland function and wetland conversion if this was desired. The index would also be an indirect measure of ponds with wetland features (e.g. shallow, well vegetated ponds).

To assign scores, we parsed the variable values into thirds, giving the best condition a score of 3 and worst condition a score of 1. We considered the trisection method to parse the data into thirds to assign the deepest most macrophyte-rich areas the highest score. Trisection has been used by USEPA to designate reference condition of lakes by assigning the best third for a particular parameter as reference state for that parameter (USEPA 1998, USEPA RTAG 2011). For pond depth we trisected ponds based on the average depth at farthest cast. The deepest

third of ponds were deeper than 2.2m at farthest cast and assigned the lowest score of 1, while the shallowest ponds were no deeper than 1.7 m and assigned the best score of 3. The middle third were assigned a value of 2 for this parameter (Table 10).

Table 10. Wetland conversion suitability index (WCI) scoring matrix for seven factors measured at ponds in the Delaware River Basin, Kansas. Scores are summed to create an index. Highest index possible is 21.

	Macrophytes			**Pond depth**	**Landscape slope**	**Channel depth**	
	pond	**upstream plot**	**downstream plot**	**pond**	**downstream plot**	**upstream plot**	**downstream plot**
score	% shoreline	% of plot area	% of plot area	deepest depth m	low gradient	deepest depth m	deepest depth m
1	<50	<50	<50	>2.2	no	>0.6	>0.6
2	50 - 70	50 - 70	50 - 70	1.7 - 2.2	--	0.30 - 0.60	0.30 - 0.60
3	>70	>70	>70	<1.7	yes	none or <0.30	none or <0.30
factor code	WCI1_ vegpond	WCI2_ vegup	WCI3_ vegdown	WCI4_ gradpond	WCI5_ graddown	WCI6_ depthup	WCI7_ depthdown

Ponds with no channels received the highest score (3). Shallowest depths are most desirable as there is reduced drainage potential increasing time of soil contact with moisture. We applied the trisection method separately to downstream channels and upstream channels and used that which was most restrictive, which was the upstream channel depths with the shallowest third at less than 0.30 m deep (score 3) and deepest third at more than 0.60 m deep (score 1). Downstream channels were deeper, with shallowest third at less than 0.40 m deep and deepest third at more than 1.2 m deep.

Because we did not have numeric slope values for either upstream and downstream plots, we used a high/low estimate of slope taken from the field sheets. Gradient estimates within the plots were recorded only for the downstream area and were categorized in the field as 'yes' or 'no' for flat and low gradient. Therefore, we assigned 'yes' the score of 3, and 'no' the score of 1 indicating that it was some gradient or slope condition other than flat or nearly so.

The macrophyte data for both percent of shoreline and percent of plot area were skewed toward low coverage, so it did not make biological sense to use the trisection method. For instance, the top third of upstream plots (those with the most macrophytes) had a cutoff of greater than only 15% macrophyte coverage. Thus, we based scores on best professional judgment and assigned ponds with $\geq$70% coverage the highest score (3), 70 to 50% coverage a score of 2 and less than 50% coverage the lowest score (1).

This resulted in seven characteristic or condition categories and scores which we summed to create the wetland conversion suitability index (WCI). The highest index score possible was 21 and the lowest 7 creating a scoring spread of 14, which may or may not be sufficient to accommodate the potential continuum of conditions associated with pond complexes

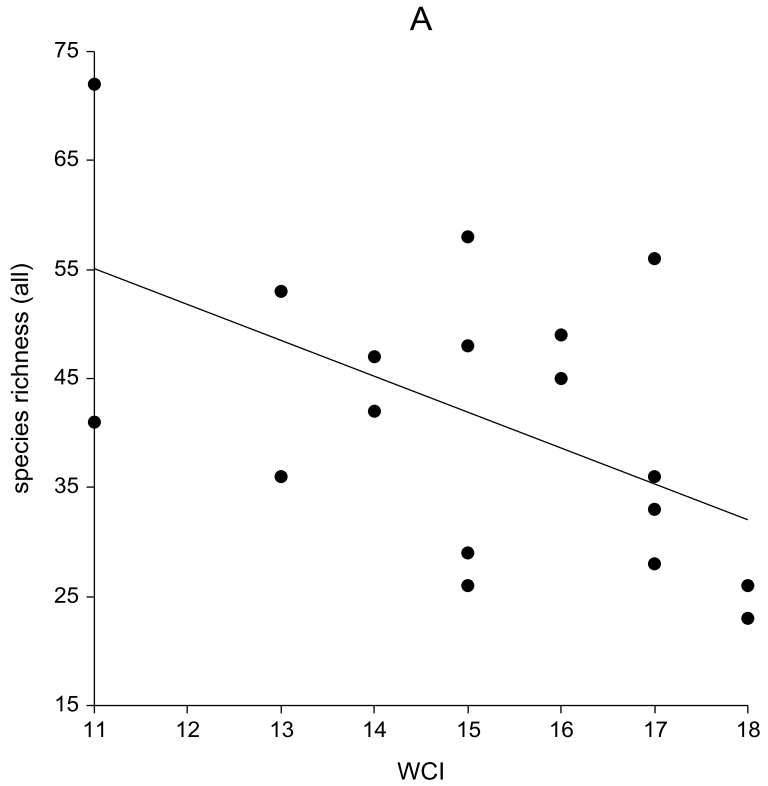
examined. Not all ponds had data for each category, so we applied the index to only the 77 ponds that had data for all seven factors. The WCI scores ranged from 7 to 18 such that a spread of 11 points was noted between all 77 ponds that were scored for the WCI.

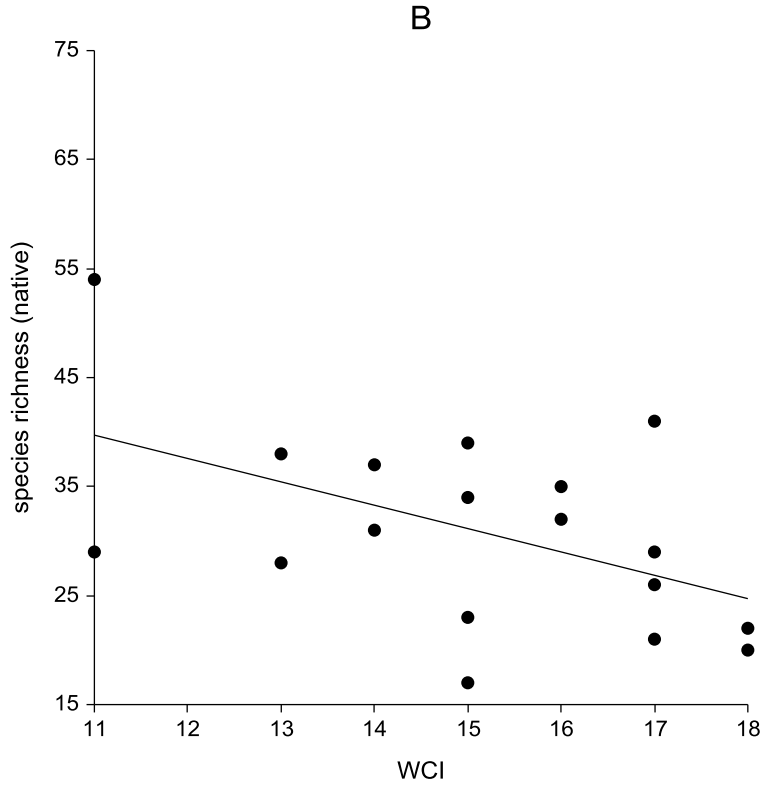
Of the 20 ponds that were selected for revisit based on depth and macrophyte coverage described previously, two could not be scored because of missing data. The remaining 18 scored 11 and higher for WCI. These revisited ponds had higher WCI scores than the entire suite of 77 scored ponds (Table 11). Acknowledging that four of the seven WCI factors were used to select the revisit sites, the higher WCI of the ponds revisits suggests the WCI approach is more robust than basing pond selection on only our original four initial screening parameters. This pilot WCI needs to be tested with an outgroup of ponds to see if the WCI does identify what we perceived to be pond complexes with a high potential for further improvement of wetland functions and conditions. Presently, all that can be inferred regarding the functionality of the WCI until further tested is that it encompasses all those factors and conditions we found during this probability-based study that were associated with wetland conditions observed.

Table 11. Comparison of wetland conversion suitability scores between all ponds scored across seven factors and the subset of ponds chosen only by pond and downstream depth or gradient and macrophyte coverage.

Ponds	**n**	**Minimum**	**Mean**	**Median**	**Maximum**
with WCI score	77	7	12	12	18
only revisited	18	11	15	15	18

We examined how *in situ* water quality and vegetation indices varied with WCI. The only significant ($p<0.05$) relationship with water quality parameters was a negative correlation with water temperature ($r = -0.31$). However, water temperatures varied with time of day when sampling occurred (i.e. air temperature), and this relationship was considered spurious and not biologically meaningful. The only significant relationships with vegetation were negative correlations with species richness (all species $r = +0.28$ and only natives $r= +0.25$), and FQI (only natives $r= +0.27$) (Figure 40). However all these correlation coefficients are small and relationships, if any, are very weak. At best these negative correlations might suggest that pond complexes with high WCI scores, and thus high macrophyte cover, have overall lower plant richness values. Because dense macrophyte cover is often the result of high numbers of one or more dominate species, macrophyte abundance is perhaps limiting the overall richness and FQI scores.


A
species richness (all)
75
65
55
45
35
25
15
11
12
13
14
15
16
17
18
WCI


B
species richness (native)
75
65
55
45
35
25
15
11
12
13
14
15
16
17
18
WCI

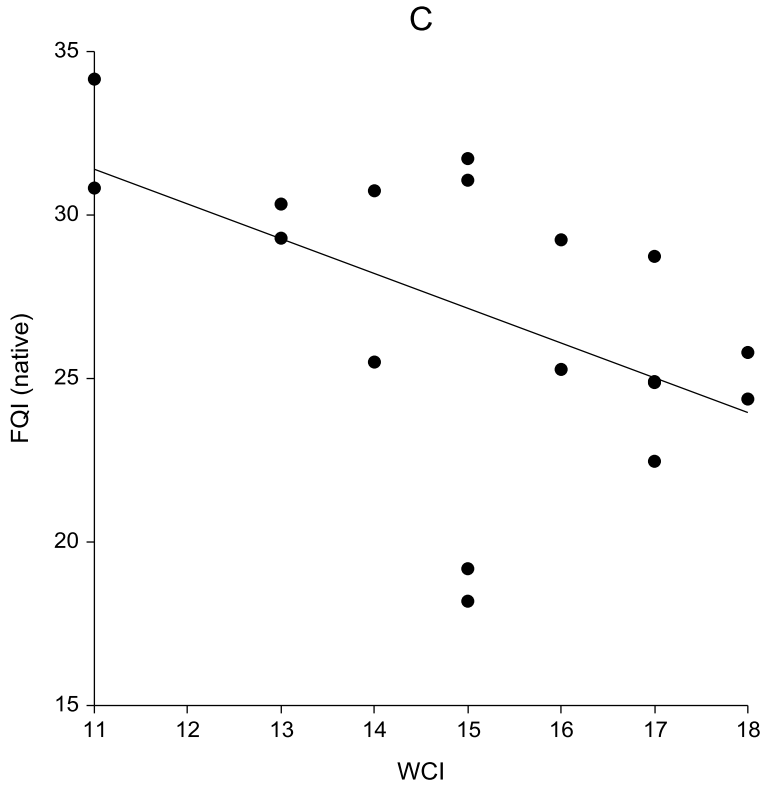


Figure 40. Scatter plots of wetland conversion suitability index (WCI) and vegetation indices for species richness of A. all species and B. only native species; and C. floristic quality index (native species). Linear regression lines are shown. All are significant ($p<0.05$).

GIS-Based Wetland Conversion Suitability Index (gWCI)

The WCI described earlier provides a ground-level indicator of wetland conversion site suitability based on *in situ* properties. In this section, we describe a synoptic, GIS-based approach to characterize conversion suitability for all of the elements in POP0. The resulting ranking index is called the gWCI and can be used to identify priority areas for conversion potential.

To develop the gWCI, first we use watershed topography and land cover to characterize erosion potential and thus the NEED for a downstream wetland to treat runoff. Next, we use topography between a pond extent and its floodpool extent (i.e. the detention area) to characterize the OPPORTUNITY for wetland conversion of the pond. To help link this activity to the nine Outcomes described in the Background section, we itemize the subsequent development with respect to the Outcomes.

1) [POP0] Target pond population, initially developed at 5m then later at 2m, incorporating new LiDAR data available for Jefferson County. The 2m version of POP0 was used for development of the gWCI.
2) [WSHED] Catchments for target ponds. Like POP0 delineations, first completed at 5m then later at 2m. For the 2m development, downstream obstructions (typically an

elevated roadway or a larger dam) higher than the upstream target pond impoundment were identified and breached to mitigate catchment mapping errors.

3) We define the *pond complex* to be the region contained within the *detention area* (DET), which is the lake footprint when the auxiliary spillway is overtopped (i.e. the LiDAR flood pool extent; Figure 1). This area is potentially inundated in high water conditions. Within each DET is the *retention area* (RET), which we define using the hydroflattened extent in the LiDAR (this accounted for 992 of the 1148 initial POP0 elements). Where there was no hydroflattening, we used the NHD extent clipped to its corresponding DET (Figure 1).
4) Opportunity: For "pond complex suitability indexing", we first identified the area outside RET but inside DET for each pond (denoted DET \ RET in set notation). Using a percent-slope map derived from 2m LiDAR, we determined the area within DET \ RET that exhibited at most 5% grade. This somewhat arbitrary threshold was selected to ensure near-flat terrain with some accommodation for minor variability and potential LiDAR data noise.

   OPP = pond complex suitability index
   = area in DET \ RET with 0-5% grade
5) Need (risk): For "pond catchment suitability indexing", we used slope and land cover (specifically, cropland; Kansas Applied Remote Sensing Program 2009) upstream from DET to determine the index, following Duley and Hays (1932) for slope-erosivity thresholds and weights.

   NEED = pond catchment suitability index
   = (cropland area in WSHED \ DET with 0-4% grade) +
   2*(cropland area in WSHED \ DET with 4-8% grade) +
   4*(cropland area in WSHED \ DET with >8% grade)
6) Available imagery and land cover data lacked the spatial, temporal, and class resolution to adequately identify vegetation indicative of pond complex wetland function. Further research is needed relating the field data to these and other geospatial data to determine the feasibility of remote identification of existing wetland function in the farm pond detention areas.
7) As described in #4, low-slope areas within DET \ RET were identified to provide an indicator of wetland development potential. Near flatness of the underlying terrain is a fundamental wetland characteristic.
8) GIS-based Wetland Conversion Suitability Index [gWCI]: The objective here was to determine a rank ordering for the ponds in the target population (POP0) reflective of their suitability for wetland establishment or conversion. Fourteen elements of POP0 were eliminated from the analysis due to irreconcilable mismatches between RET and DET in the LiDAR and NHD data (e.g. some dams had been breached at the time of LiDAR collection), leaving 1134 ponds to examine for the GIS-based suitability index evaluation. Step 1: Elements of POP0 were sorted by NEED, so that ponds with catchments containing substantial cropland, especially high-slope cropland, would take priority. The assumption is that these ponds receive the most sediment- and nutrient-laden runoff, and thus have the greatest potential need for wetland function for water treatment. 425 (of 1134) ponds contained no cropland in their watershed, leaving 709 ponds with some degree of need. The highest need watershed was assigned a rank of 1,

the second a rank of 2, and so on. Step 2: Elements of POP0 were sorted by OPP, so that ponds with the most low-relief detention area terrain would take priority. The assumption is that these ponds are most likely to exhibit existing wetland characteristics are would require minimal land modification for development of wetland characteristics. 11 (of 1134) ponds had detention areas containing no low-relief terrain, leaving 1123 ponds that exhibited some degree of opportunity. The highest opportunity watershed was assigned a rank of 1, the second a rank of 2, and so on. Step 3: NEED and OPP rankings were added together, and ponds were sorted on this combined ranking, with the lowest values indicating ponds with a high degree of need and a high degree of opportunity. This combined ranking is the GIS-based Wetland Conversion Suitability Index (gWCI). Six ponds had no need and no opportunity, leaving 704 (= 1134-425-11+6) with some need and some opportunity.

9) With the development of the gWCI, the target farm pond population has been rank ordered for rapid identification of sites where the need for wetland conversion is the greatest and where there is also the greatest potential opportunity for conversion. Should Phase II of the broader project design be undertaken, then gWCI can be used to prioritize ponds for detention area wetland establishment or development.

Summarized gWCI rankings are shown in Figure 41 for the study area. A close-up of some sites featuring relief and land cover is shown in Figure 42. While this development of the gWCI provides a sound starting point, in future studies its definition could be refined to take into account other relevant GIS-based factors that can influence wetland conversion suitability.

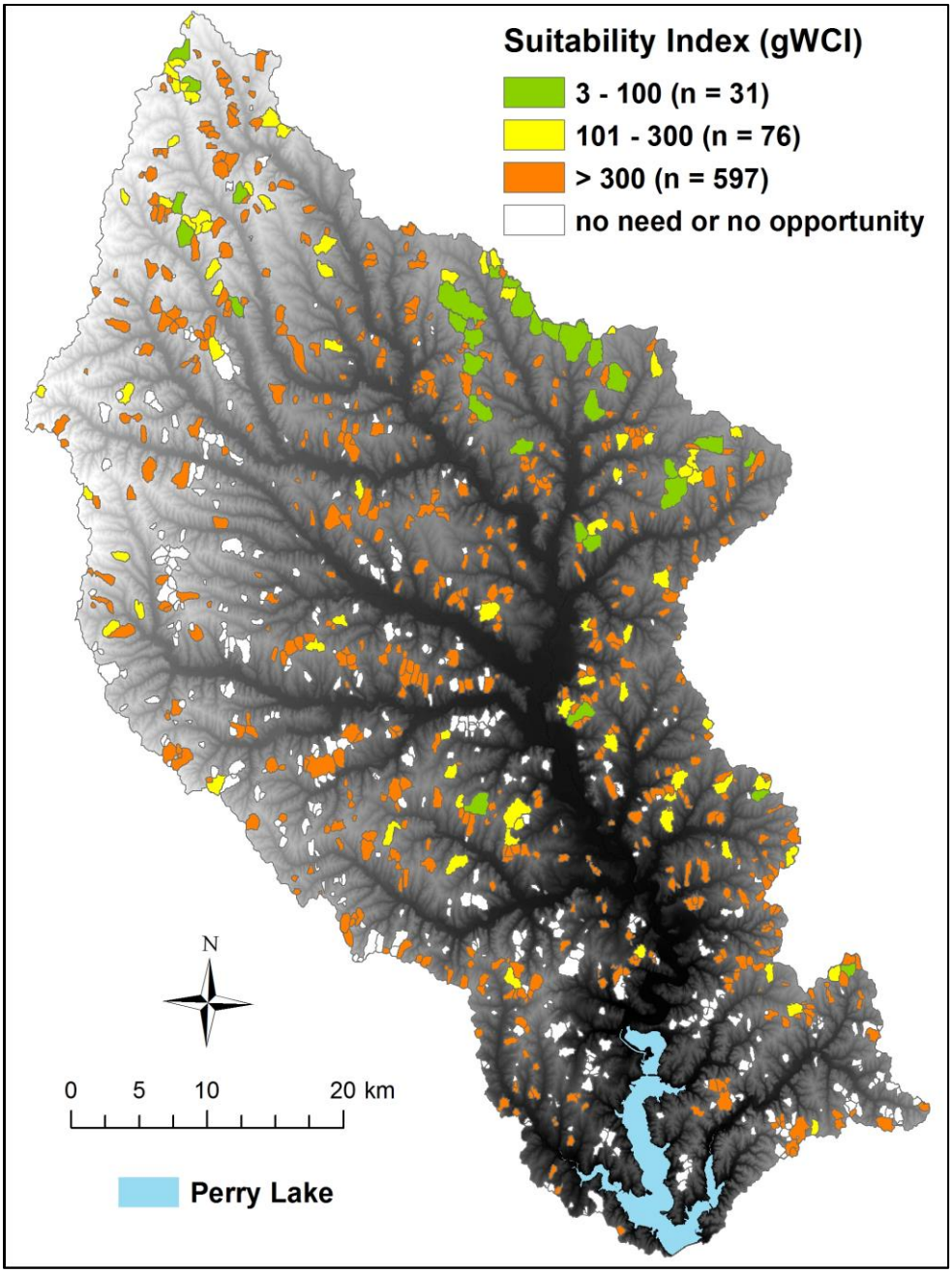


Figure 41. Summarized gWCI rankings for the Delaware River Watershed study area.

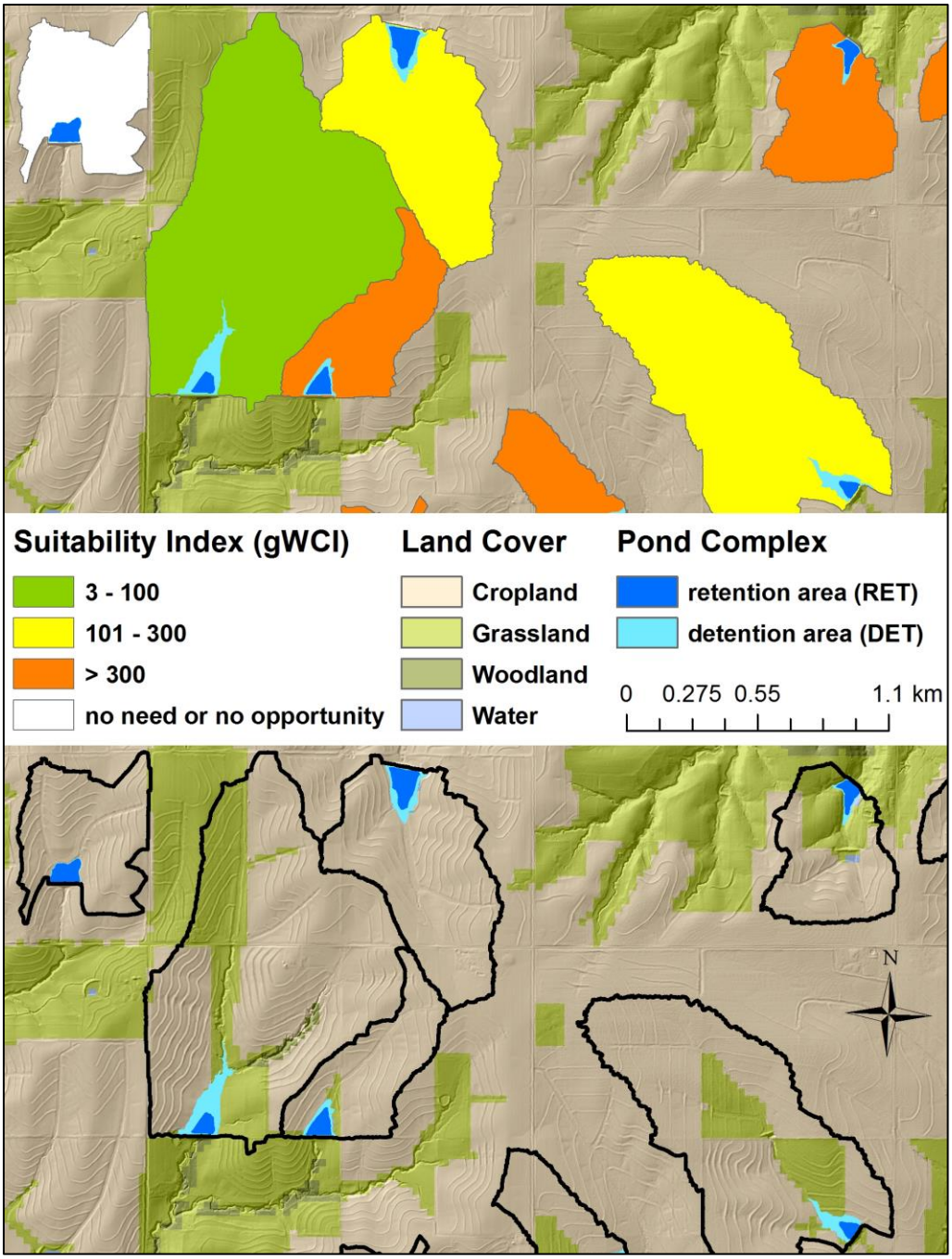


Figure 42. A close-up of some study area farm ponds, featuring relief and land cover.

PROJECT SUMMARY AND KEY OUTCOMES

This multifaceted project had many outputs and outcomes design to identify key characteristics and conditions associated with a random set of small (1-5 acres) artificial ponds located within the Delaware River Basin of northeast Kansas that would help identify wetland conditions occurring in some ponds and factors that assist in selecting ponds that could be altered to further develop wetland attributes. In addition, an effort was made to identify and characterize

the watersheds draining to all ponds within this basin within the same size range as our study ponds. This watershed effort and the spatial quantification of some pond complex areas were addressed using several remote sensing and GIS tools including LiDAR and NHD data. Site specific pond assessments and data collection was accomplished following a detailed field survey methodology and field evaluation forms (see Appendix E). The specific study findings and developed assessment tools are scattered throughout this document. The following list of outputs and outcomes is not exhaustive but does summarize key study achievements.

1. All but 10% of the pond complexes survey had limited or no pond macrophytes and little surrounding wetland plants (i.e. facultative and obligate only).
2. Over half of the pond complexes surveyed were already colonized with significant areas (≥ 50%) wetland plants and pond macrophytes, many appearing as artificial, palustrine, aquatic bed wetlands with unconsolidated bottoms.
3. Ponds were typically shallow with median littoral and deep water zone depths of 0.35 m and 1.4 m, respectively. **Note:** Sampling seasons had about average rainfall for this area for Kansas.
4. Over 75% of inflow zones had some wetland vegetation cover, but very few had over 50% cover (14%).
5. About a third of the inflow zones showed livestock use.
6. Most ponds have drainage channels in both the inlet and outflow zones.
7. 60% of outflow zones had wetland vegetation but only 2% had significant amounts (≥ 50%).
8. Over half of the outflow zones had standing water and in about 6% of the survey pond complexes water made up about half the sample plot.
9. All study pond had earthen dams with graded, unarmored emergency spillways except for one with no spillway.
10. 6% of the spillways surveyed showed some limited signs of erosion.
11. Over 90% of all ponds had simple pass-through tubes or pipes (i.e. trickle tubes) without valves for water control.
12. Average trickle tube diameter was 16 inches, and 22% of all ponds had trickle tubes that were NOT functioning for various reasons (e.g. blockages).
13. Over 60% of pond retention areas that exhibited a dewatered shoreline had wetland plant cover, while 23% of these shorelines had over 50% coverage in wetland plants
14. Just one-third of all study pond complexes had livestock present or signs of recent livestock accesses to the pond environments.
15. 66% of the pond complexes surveyed did not have livestock use associated their environment.
16. Nearly 20% of the ponds had over 90% of their littoral zone with macrophytes.
17. 75% of all ponds had at least 10% of their littoral zones with macrophytes.
18. The most common macrophytes forming large colonies were cattail (*Typha* sp.), pondweed (*Potamogeton* spp.), duckweed (*Lemna* spp.), and najas grass (*Najas* sp.).
19. In situ water quality parameters were measures for ponds selected for further study.

20. For selected ponds, pH values were slightly basic (median = 9.2) with a pH of 10.9 recorded at one pond. High daytime pH values were thought to be associated with high plant and algae production.
21. Selected pond water clarity was often good with a median turbidity level of just over 16 NTUs.
22. For selected ponds, daytime dissolved oxygen values varied greatly (0.9 to 16.7 mg/l) with both a mean and median concentration of 6.7 to 6.8, respectively.
23. A variety of wildlife including fish, amphibians, beaver, and muskrats were either observed or signs of their occurrence found at all most all ponds.
24. A number of vegetation indices, including FQI were calculated for ponds selected for further study (see text for specifics).
25. One hundred ninety plant species were founded associated with ponds selected for further study.
26. For selected ponds, mean wetland quality coefficients were calculated based on all species and only native species.
27. In general, wetland improvement opportunities were observed for all pond complex areas or zones (see text for specifics).
28. A pilot, site-specific wetland conversion index (WCI) was proposed for evaluation (see text for specifics).
29. A pilot watershed-specific wetland conversion index (gWCI) was also proposed for further study (see text for specifics).

In their current form, the WCI and gWCI could be used together or separately to select ponds that have wetland characteristics and could possibly be further developed as wetlands. The gWCI gives a broader approach to wetland selection sites as it includes a risk factor (i.e. opportunity) based on potential erosion within specific watersheds draining to an individual pond complex. The wetland conversion assessment element of this gWCI is the general gradient associated with the upper inlet zone within the detention area.

Future Work

A phase two of this project has already been proposed in the original proposal award and can be viewed in detail in that document. Several project findings have shifted some of our future focus on both nutrient and sediment retention (e.g. trapping and processing efficiencies) in individual pond complexes and nested ponds complexes under various hydrological conditions (high runoff and spillway over flow, and low runoff). While this project touched on possible climate change impacts on small ponds, little could be said quantitatively at this point other than the pond community examined was on average very shallow making them very susceptible to increased droughts and dewatering. An outline of some proposed new research focuses are as follows:

1. Test WCI and gWCI on an out-group (i.e. non-study ponds) of randomly selected ponds in the Delaware River Basin to assess veracity and transferability of these indices.
2. Modify WCI and gWCI if necessary.
3. Evaluate utility of converting gWCI to a risk (i.e. watershed condition) index and joining inlet conversion element to WCI to enhance WCI as a wetland conversion index.

4. Evaluate use of LiDAR to determine outlet zone slope conditions.
5. Expand use of both index approached to a different ecoregion pond setting.
6. Quantify pond retention times and sediment trapping efficiencies (TE) for single and nested pond complexes.
7. Evaluate nutrient processing and sediment retention efficiencies for a suite of single and nested ponds to determine overall downstream reduction potentials.
8. Eventually develop and adapt wetland restoration methods suitable for high priority ponds.
9. Identify pond/wetland enrollment criteria and incentives.
10. Apply approved pond-to-wetland modification and restoration methods to ponds that become enrolled in this "ponds to wetland" program.

ACKNOWLEDGMENTS
This study was funded with USEPA Wetland Program Development Grants money distributed by the Kansas Water Office, coordinated by KWO Project Officer Kirk 'TJ' Tjelmeland. We thank all the Delaware River Basin landowners who allowed us access to their ponds. This study would not be possible without their participation. We also thank those who assisted the authors in the field: Sharon Ashworth, Jennifer Delisle, Bob Everhart, and Kaitlyn Loeffler.

REFERENCES
Aksoy, H. and M.L. Kavvas. 2005. A review of hillslope and watershed scale erosion and sediment transport models. Catena 64(2):247-271.

Bilotta, G.S., R.E. Brazier and P.M. Haygarth. 2007. The impacts of grazing animals on the quality of soils, vegetation, and surface waters in intensively managed grasslands. Advances in Agronomy 94:237-280.

Blocksome, C.E. and G.M. Powell (eds). 2007. Waterers and watering systems: A handbook for livestock owners and landowners. Kansas State University Agricultural Experiment Station and Cooperative Extension Service, Manhattan, KS. Publ. No. S-147. 149pp.

Brainwood, M.A., S. Burgin and B. Maheshwari. 2004. Temporal variations in water quality of farm dams: impacts of land use and water sources. Ag. Water Manag. 70(2):151-175.

Bourdaghs, M., C.A. Johnston, and R.R. Regal. 2006. Properties and performance of the floristic quality index in Great Lakes coastal wetlands. Wetlands 26(3):718-735.

Coops, H., M. Beklioglu and T.L. Crisman. 2003. The role of water-level fluctuations in shallow lake ecosystems–workshop conclusions. Hydrobiologia, 506(1-3):23-27.

Covich, A.P., S.C. Fritz, P.J. Lamb, R.D. Marzolf, W.J. Matthew, K.A. Poiani, E.E. Prepas, M.B. Richman and T.C. Winter. 1997. Potential effects of climate change on aquatic ecosystems of the Great Plains of North America. Hydrological Processes 11(8):993-1021.

Dahl, T.E. 1990. Wetlands - Losses in the United States, 1780's to 1980's. U.S. Fish and Wildlife Service. Wash. D.C., 13pp.

Deal, C., J. Edwards, N. Pellmann, R.W. Tuttle, D. Woodward and G. Wells. 1997. Ponds: planning, design, construction. Agriculture Handbook No. 590.

De Vente, J. and J. Poesen. 2005. Predicting soil erosion and sediment yield at the basin scale: scale issues and semi-quantitative models. Earth-Science Reviews 71(1): 95-125.

Duley, F.L., and O.E. Hays. 1932. The Effect of the Degree of Slope on Run-off and Soil Erosion. Journal of Agricultural Research 45(6): 349-360.

Freeman, C.C. and C.A. Morse. 2002. Kansas floristic quality assessment: coefficients of conservatism for plants in southeast Kansas. Submitted to Kansas Dept. of Health and Environment. 15pp. + appendix.

Herman, K.D. 2001. Floristic quality assessment with wetland categories and examples of computer applications for the state of Michigan. Michigan Dept. of Nat. Res., Wildlife Division, Natural Heritage Program. 19pp + app.

Hicks, C.E. and R.A. Pierce II. 2014. Managing Ponds and Lakes for Aquaculture and Fisheries in Missouri: Pond Construction and Management Considerations. Bulletin g9474. http://extension.missouri.edu/p/G9474 Accessed 15 May 2017.

Hill, R.D., G.O. Schwab, G.W. Malaney, and H.H. Weiser. 1962. Quality of water in Ohio farm ponds. Research Bulletin 922, Ohio Ag. Exper. Station, Wooster, OH. 99pp.

Hubbard, R.K., G.L. Newton and G.M. Hill. 2004. Water quality and the grazing animal. J.Animal Science 82(13_suppl): E255-E263.

Jean, C.R. and M.R. Sanderson. 2015. The Kansas Watershed District Story. State Association of Watersheds, KS Dept. of Agriculture, Topeka KS. 29pp.

Kansas Applied Remote Sensing Program. 2009. Kansas Land Cover Patterns – Level I. Lawrence, Kansas: Kansas Biological Survey, Univ. of Kansas.

KDA. 1996. Map: Potential net evaporation, in inches, for Kansas. Div. of Water Resources, KS Dept. Agriculture. http://agriculture.ks.gov/docs/default-source/dwr-water-appropriation-documents/newevapomap.pdf Accessed 15 May 2017.

Knutson, M.G., W.B. Richardson, D.M. Reineke, B.R. Gray, J.R. Parmelee and S.E. Weick. 2004. Agricultural ponds support amphibian populations. Ecological Applications 14(3):669-684.

Lauver, C.L., K. Kindscher, D. Faber-Langendoen, and R. Schneider. 1999. A classification of the natural vegetation of Kansas. The Southwestern Naturalist 44(4):421-443.

Lichvar, R.W. 2012. The national wetland plant list. ERDC-CRREL-TR-12-11), Cold Regions Research and Engineering Laboratory, U.S. Army Engineer Research and Development Center, Hanover, NH. 9pp. with unpaged appendices. (approximately 224 pp. total)

Lopez, R.D. and M. Siobhan Fennessy. 2002. Testing the floristic quality assessment index as an indicator of wetland condition. Ecological Appl. 12(2):487-497.

Mackie, G.L. and D.G. Huggins. 1983. Sphaeriacean clams of Kansas. Tech. Publ. State Biol. Survey KS, No. 16. 92pp.

McDowell, R.W. and R.L. Wilcock. 2008. Water quality and the effects of different pastoral animals. New Zealand Veterinary Journal 56(6):289-296.

Melillo, J.M., T.T. Richmond and G. Yohe. 2014. Climate change impacts in the United States. Third National Climate Assessment.

Merritt, W.S., R.A. Letcher and A.J. Jakeman. 2003. A review of erosion and sediment transport models. Environmental Modelling and Software 18(8):761-799.

Miller, S.J. and D.H. Wardrop. 2006. Adapting the floristic quality assessment index to indicate anthropogenic disturbance in central Pennsylvania wetlands. Ecol. Indicators 6(2):313-326.

Murdoch, P.S., J.S. Baron and T.L. Miller. 2000. Potential effects of climate change on surface water quality in North America. J. Am. Water Res. Assoc. 36(2):347-366.

NCSS 9 Statistical Software. 2013. NCSS, LLC. Kaysville, Utah, USA, ncss.com/software/ncss.

NRCS (Natural Resources Conservation Service). 2005. Earth Dams and Reservoirs Conservation TR–60 (210-VI-TR60). Conservation Engineering Div., NRCS, USDA, Wash. D.C.

Olsen, A.R., J. Sedransk, D. Edwards, C.A. Gotway, W. Liggett, S. Rathbun, K.H. Reckhow, and L.J. Young. 1999. Statistical issues for monitoring ecological and natural resources in the United States. Environ. Monitoring Assess. 54(1):1-45.

Perry, C.A. 2008. Precipitation-frequency and Discharge-frequency Relations for Basins less than 32 Square Miles in Kansas, Sci. Invest. No. 2008-5112. United State Geological Survey, Reston, VA. 24pp.

Prosser, I.P., W.E. Dietrich, and J. Stevenson. 1995. Flow resistance and sediment transport by concentrated overland flow in a grassland valley. Geomorphology 13(1-4):71-86.

R Core Team. 2012. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. ISBN 3-900051-07-0, http://www.R-project.org/ accessed 15 May 2017.

Schenck, E., K.L. Wedel, and M.J. Monda. 1992. Classification of Wetland and Riparian Areas in Kansas. Kansas Wetland & Riparian Areas Project. Topeka, KS.

Schindler, D.W. 2009. Lakes as sentinels and integrators for the effects of climate change on watersheds, airsheds, and landscapes. Limnology and Oceanography, 54(6):2349.

Schmutzer, A.C., M.L. Gray, E.C. Burton, E.C., and D.L. Miller. 2008. Impacts of cattle on amphibian larvae and the aquatic environment. Freshwater Biology 53(12):2613-2625.

Smith, S.M.C. and P.R. Wilcock. 2015. Upland sediment supply and its relation to watershed sediment delivery in the contemporary mid-Atlantic Piedmont (USA). Geomorphology 232:33-46.

Sorenson, L., R. Goldberg, T. Root, and M. Anderson. 1998. Potential effects of global warming on waterfowl populations breeding in the northern Great Plains. Climatic Change 40:343–369.

Stevens, D.L. 1997. Variable density grid-based sampling designs for continuous spatial populations. Environmetrics 8(3):167-195.

Stevens Jr, D.L. and S.F. Jensen. 2007. Sample design, execution, and analysis for wetland assessment. Wetlands 27(3):515-523.

Stevens Jr, D.L., D.P. Larsen, and A.R. Olsen. 2007. In *Salmonid Field Protocols Handbook*, Johnson *et al.* (eds). The role of sample surveys: why should practitioners consider using a statistical sampling design? American Fisheries Society, Bethesda, MD. pp.11-23.

Stevens Jr, D.L. and A.R. Olsen. 2004. Spatially balanced sampling of natural resources. J. Am. Statistical Assoc. 99(465):262-278.

USEPA. 1998. Lake and Reservoir Bioassessment and Biocriteria Technical Guidance Document Office of Water, Office of Science and Technology. Washington, DC. USEPA-841-B-98-007.

USEPA Regional Technical Advisory Group (RTAG). 2011. Nutrient Reference Condition Identification and Ambient Water Quality Benchmark Development Process, Freshwater Lakes and Reservoirs within USEPA Region 7. Kansas Biological Survey, Lawrence, KS 42pp.

Verstraeten, G. and J. Poesen. 2000. Estimating trap efficiency of small reservoirs and ponds: methods and implications for the assessment of sediment yield. Progress in Physical Geography 24(2):219-251.

Verstraeten, G. and J. Poesen. 2001a. Factors controlling sediment yield from small intensively cultivated catchments in a temperate humid climate. Geomorphology 40(1):123-144.

Verstraeten, G. and J. Poesen. 2001b. Modelling the long-term sediment trap efficiency of small ponds. Hydrological Processes 15(14): 2797-2819.

Verstraeten, G. J. and J. Poesen. 2002. Regional scale variability in sediment and nutrient delivery from small agricultural watersheds. J. Environ. Quality 31(3):870-879.

Villarini, G., J.A. Smith, M.L. Baeck, R. Vitolo, D.B. Stephenson and W.F. Krajewski. 2011. On the frequency of heavy rainfall for the Midwest of the United States. J. Hydrology 400(1):103-120.

Williamson, C.E., J.E, Saros, W.F. Vincent and J.P. Smold. 2009. Lakes and reservoirs as sentinels, integrators, and regulators of climate change. Limnology and Oceanography, 54(6part2):2273-2282.

Wolinsky, S. 2006. Farm Pond Ecosystems. Fish and Wildlife Habitat Management Leaflet No. 29. 12pp. USDA NRCS. https://policy.nrcs.usda.gov/OpenNonWebContent.aspx?content=18500.wba Accessed 15 May 2017.