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Agricultural conversion of floodplain ecosystems: Implications for groundwater quality

Keith E. Schilling [a, *], Peter J. Jacobson [b], Jason A. Vogelgesang [a]

[a] Iowa Geological Survey, 340 Trowbridge Hall, Iowa City, IA 52242-1319, USA
[b] Department of Biology, Grinnell College, Grinnell, IA 50112, USA

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ABSTRACT

With current trends of converting grasslands to row crop agriculture in vulnerable areas, there is a critical need to evaluate the effects of land use on groundwater quality in large river floodplain systems. In this study, groundwater hydrology and nutrient dynamics associated with three land cover types (grassland, floodplain forest and cropland) were assessed at the Cedar River floodplain in southeastern Iowa. The cropland site consisted of newly-converted grassland, done specifically for our study. Our objectives were to evaluate spatial and temporal variations in groundwater hydrology and quality, and quantify changes in groundwater quality following land conversion from grassland to row crop in a floodplain. We installed five shallow and one deep monitoring wells in each of the three land cover types and recorded water levels and quality over a three year period. Crop rotations included soybeans in year 1, corn in year 2 and fallow with cover crops during year 3 due to river flooding. Water table levels behaved nearly identically among the sites but during the second and third years of our study, NO_3–N concentrations in shallow floodplain groundwater beneath the cropped site increased from 0.5 mg/l to more than 25 mg/l (maximum of 70 mg/l). The increase in concentration was primarily associated with application of liquid N during June of the second year (corn rotation), although site flooding may have exacerbated NO_3–N leaching. Geophysical investigation revealed differences in ground conductivity among the land cover sites that related significantly to variations in groundwater quality. Study results provide much-needed information on the effects of different land covers on floodplain groundwater and point to challenges ahead for meeting nutrient reduction goals if row crop land use expands into floodplains.

1. Introduction

Floodplains provide an abundance of ecosystem services to society (Opperman et al., 2010), including conservation of biodiversity (Tockner and Stanford, 2002), floodplain fisheries (Costanza et al., 1997; Bayley, 1991), floodwater storage (Opperman et al., 2009), water supply enhancements (Fleckenstein et al., 2004), recreation (Golet et al., 2006) and nutrient retention (Vidon and Hill, 2004; Van Der Lee et al., 2004; Krause et al., 2008; Natho et al., 2013). Denitrification is considered the main process associated with N losses in floodplains (e.g., Pinay et al., 2007; Saunders and Kalff, 2001), whereas sedimentation is a dominant process for phosphorus retention (Van Der Lee et al., 2004). Despite the services they provide, floodplains are among the most threatened ecosystems in the world (Tockner and Stanford, 2002). River regulation (e.g., levees) and intensive agricultural use have disconnected the interactions of rivers with their floodplains and homogenized floodplain environments (Schilling and Jacobson, 2011; Antheunisse et al., 2006; Hohensinner et al., 2004).

Encroachment of row crop land use into perennially-vegetated floodplains is occurring throughout the U.S. Midwest as demands from the biofuel industry are driving expansion of corn and soybean production into marginal areas (Secchi et al., 2011), and perennial grasslands, forest and pastures are increasingly being converted to row crops (Schilling et al., 2010). Approximately one-half of the corn grown in the US is now used for ethanol production and there is economic pressure for still more production (Mehaffrey et al., 2012). Effects of this expansion on hydrology (Xu

* Corresponding author. University of Iowa, Iowa Geological Survey, 340 Trowbridge Hall, Iowa City, IA 52242-1319, USA.
E-mail addresses: keith-schilling@uiowa.edu (K.E. Schilling), jacobsop@grinnell.edu (P.J. Jacobson).

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et al., 2013) and nutrient delivery to receiving waters (Jha et al., 2010; Donner and Kucharik, 2008) are being increasingly recognized. Fertilizer applied to newly converted corn will increase nitrogen export (Raymond et al., 2012) and make nutrient reductions more difficult to achieve (INRS, 2013).

Studies have shown that converting perennial vegetation to row crops leads to water quality deterioration, particularly with respect to nitrate. While many studies have used modeling to quantify the effects (e.g., Johnes, 1996; Donner et al., 2004; Schilling et al., 2008; Costello et al., 2009), fewer field monitoring studies have been conducted to directly measure this change. Huggins et al. (2001) found that residual soil nitrate increased 125% the first year following conversion of brome grass to corn. Schilling and Spooner (2006) reported nitrate concentrations in surface water increasing by more than 10 mg/l over a 4 year period in a small Iowa watershed following conversion of Conservation Reserve Program (CRP) grassland to row crop. Likewise, Zhou et al. (2010) observed that nitrate levels in the vadose zone and groundwater significantly increased following grassland to cropland conversion in their study of perennial filter strips. Nitrate concentrations increased from <2 mg/l to more than 11 mg/l at a toeslope landscape position following land use change to row crops (Zhou et al., 2010). During a riparian zone restoration, Schilling and Jacobson (2008) observed nitrate concentrations increasing from <1 to 40 mg/l when the overlying grass cover was removed.

With current trends of converting grasslands to row crop agriculture, there is a critical need to evaluate the effects of land use change on groundwater quality in a large river floodplain system. Our field study focused on comparing groundwater hydrology and nutrient dynamics associated with three land cover types (grassland, floodplain forest and cropland) commonly found on floodplains. Since the cropland site consisted of newly converted grassland, we were also able to document effects of land use conversion on groundwater quality. The specific objectives of our study were to: 1) evaluate spatial and temporal variations in groundwater hydrology and quality patterns associated with three floodplain land cover types; and 2) quantify changes in groundwater quality following land conversion from grassland to row crop in a floodplain. Study results provide much-needed information on the effects of different land covers on floodplain groundwater and point to challenges ahead for meeting nutrient reduction goals if row crop land use continues to expand into floodplains.

2. Materials and methods

2.1. Study area

The study was conducted at The Nature Conservancy (TNC) property located on the floodplain of the Cedar River in Muscatine County, Iowa (lat 41°23′21″, long 91°19′09″) (Fig. 1). The climate of the region is humid, continental with average annual precipitation of about 864 mm. The average summer temperature is 25 °C whereas the winter temperatures can reach −26 °C. The average growing season is about 170 days in a typical year. A U.S. Geological Survey (USGS) stream gage is located on the Cedar River approximately 1 km north of the site (Cedar River near Conesville, station number 05465000) (Fig. 1). The Cedar River watershed draining to the Conesville gage encompasses 20,163 km^2 (7785 mi^2), an area that includes much of eastern Iowa that is dominated by agricultural land use. The long-term mean discharge in the river is approximately 5200 cfs.

Three land covers representative of common floodplain uses were evaluated in this study (Fig. 1). The grass site consists of a monotypic stand of *Phalaris arundinacea* (reed canary grass), a common perennial grass found throughout humid areas of northern United States and Canada (Galatowitsch et al., 1999) that is considered among the most invasive species found in wetlands and other lowland areas (Zedler and Kercher, 2004). The woods site was dominated by typical floodplain species, including Swamp white oak (*Quercus bicolor*), swamp hickory (*Carya cordiformis*), American elm (*Ulmus Americana*), hawthorns (*Crataegus* sp.) and scattered occurences of Osage orange (*Maclura pomifera*) and Honey locust (*Gleditsia triacanthos*), with a weedy understory including abundant nettles (*Urticacaeae*) and scattered sedges (*Cyperacaea*).

Unlike the existing grass and woods sites, the cropped site was carved out of the grass area especially for this study. Prior to land cover conversion, the cropped area was in reed canary grass although historical photographs of the area indicate that the land was cropped in the past as recently as the early 2000's. In 2011, a local farmer was retained by TNC to cultivate the floodplain. In April 2011, the grass was burned and the field was planted in soybeans. In June 2011, field applications included phosphorus in the form of monoammonium phosphate $(NH_4)H_2PO_4$ (40 lbs/ac) and potassium from potash (~95% KCl) at a rate of 70 lbs/ac. Glyphosate was applied for weed suppression at this time. In March 2012, granular application of 11-52-60 NPK (lbs/ac) was applied to the cropped field in preparation for corn planting. In June 2012, the corn was side-dressed with 32% liquid N (urea ammonium nitrate solution) at a rate of 220 lbs/ac (70.4 lbs/ac as N). In 2013, the field was not planted in crops due to wet conditions and flooding of local access roads. Instead, in August 2013, the field was planted with rye and radishes as a preventative cover crop.

2.2. Methods

Monitoring wells were located in a crossing pattern at each of the three land covers targeted for investigation (Fig. 1). Nested shallow and deep wells were installed in the center of the grid. All shallow wells were installed using a truck-mounted Geoprobe™ hydraulic percussion system to a depth of 2.4 m below ground surface with the well screen placed at a depth of 0.9–2.4 m. A 1.5 m riser attached to the screen extended the well above the land surface. At the deep well in the middle of each land cover plot, the well was installed to a depth of 5.2 m below ground surface.

A borehole geophysical log of ground conductivity was collected during well installation using the Geoprobe™ at the center of each plot (location of center well). In March 2011, a surface geophysical survey of the monitoring well area was conducted using a Geonics EM-31 unit. The EM-31 maps changes in ground conductivity (inverse of resistivity) using an electromagnetic induction technique with an effective depth of penetration of approximately 6 m (www.geonics.com). The EM-31 survey consisted of walking survey lines oriented east–west across the area. Values were recorded with coordinate locations in a continuous manner and appended and recording values in a continuous mode that were stamped with the coordinate locations using a high-precision GPS. The survey points were contoured with the kriging routine in ArcGIS.

Following well installation, the wells were located with GPS and the top of the casings were surveyed to a site-established benchmark. The wells were developed by surging and overpumping using a Waterra sampling system. The 18 monitoring wells were sampled on 12 occasions during the 2011 to 2013 study period. Water levels in wells were measured to the nearest millimeter at the time of sampling. Water samples from wells were collected using a peristaltic pump and analyzed in the field for temperature, specific conductance (SC), pH, dissolved oxygen (DO) and oxidation–reduction potential (ORP) using a YSI Model 556 water quality meter. Accuracy of the measurements was ±0.10 C for temperature, ±0.2 pH units for pH, ±0.1% for SC, ±0.2 mg/l for DO and ±20 mv for

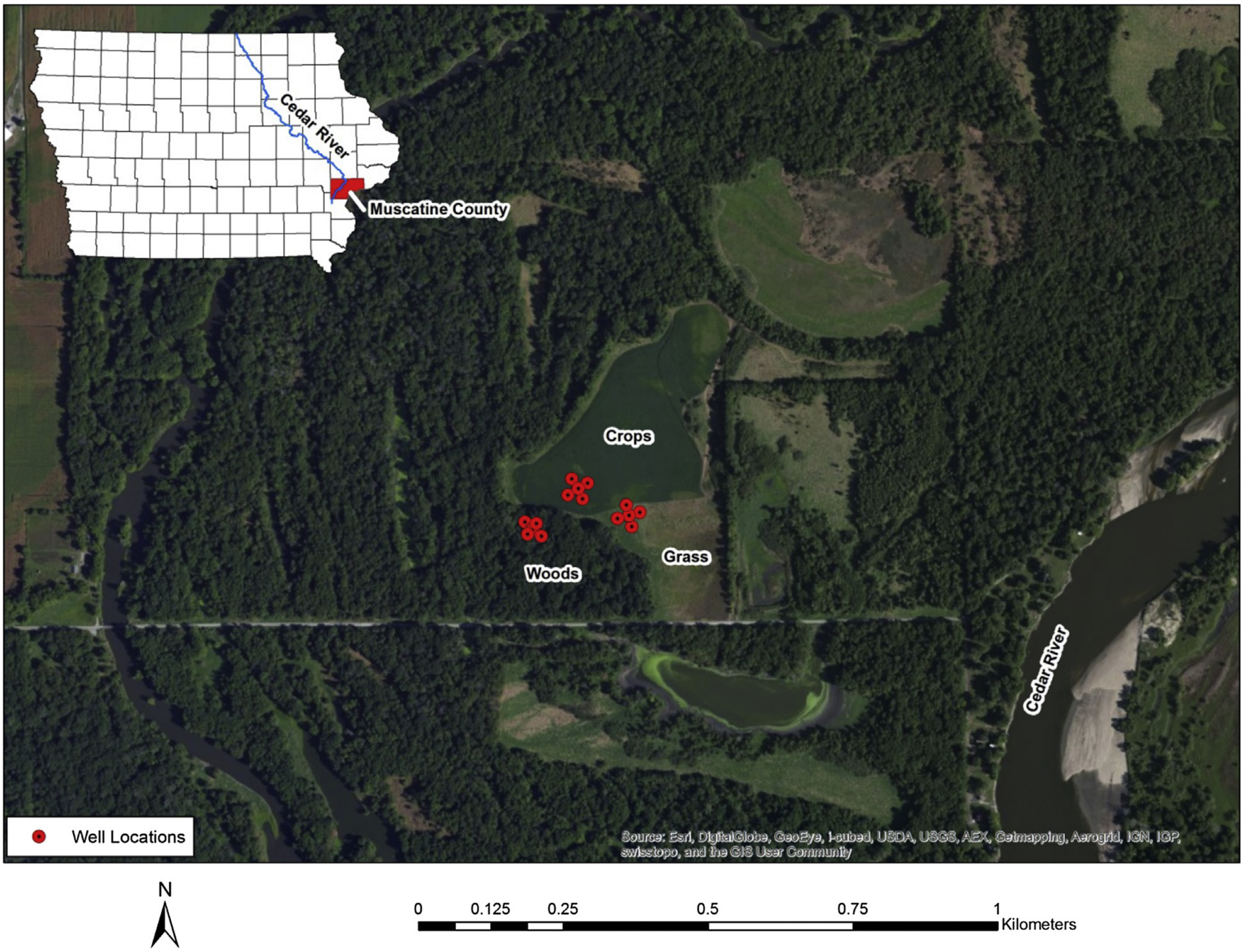

Fig. 1. Location of the monitoring sites along the Cedar River floodplain in Muscatine County, Iowa.

ORP. An In-Situ TROLL pressure transducer was installed in the shallow center well at each treatment and programmed to measure variations in pressure and temperature at 0.5-h intervals during the study period.

Water samples for laboratory analysis were field filtered through a 0.45 micron glass fiber filter, transported on ice and analyzed within 12 h of collection. Water samples were analyzed for ammonium-N (phenol-hypochlorite spectrophotometric analysis), nitrate-N (cadmium reduction) and soluble reactive phosphorus (SRP) (modified molybdenum blue ascorbic acid method) by flow injection analysis (QuickChem 8000, Lachat Instruments). Chloride was measured by ion chromatography (Lachat Instruments) and dissolved organic carbon by Pt-catalyzed, high temperature oxidation (TOC-V Total Organic Carbon Analyzer, Shimadzu Scientific Instruments, Inc., Columbia, MD, USA, for total non-purgeable organic content from acidified water samples; APHA, 1995).

We used the correlation of geochemical variables among well sites, along with ground conductivity measurements using the EM31, to assess the relation of groundwater patterns to lithology. We focused on the pre-NO_3–N change period (2011 to June 2012) to lessen any effect of land use changes on the correlation results. We investigated the influence of lithologic variations on post-cropping period NO_3–N concentrations by regressing peak NO_3–N concentration detected in the five cropped wells against ground conductivity measured at the well site. Statistical comparisons among land cover classifications were conducted using Sigma-Stat.

3. Results

3.1. Site characterization

The geology of the floodplain monitoring site was characterized vertically and spatially using geophysical methods. Borehole conductivity logs measured at the center wells in each land cover treatment revealed similar stratigraphy (Fig. 2). Within a vertical matrix consisting mostly of sand, a zone of higher ground conductivity (silt and clay materials) was present at all three sites at a depth of approximately 6.1–7.6 m; thicker at the woods site and thinnest and slightly deeper at the grass site. At a depth of approximately 15 m below ground surface, a hard zone of higher ground conductivity was encountered that we interpret to be the bedrock surface beneath the floodplain.

Results from the surface geophysical survey with the EM-31 showed spatial variations in ground conductivity ranging from 6.2 to 23.8 milliSiemens per meter (mS/m) (Fig. 3). Higher conductivity values (>15 mS/m) indicative of a greater proportion of silt and clay were measured in the northern portion of the study area in the crop field. The zone of higher conductivity extended toward the southeast near the grass wells and followed subtle topographic variations (Fig. 3). In contrast, the wells at the woods site and several located at the grass site were located in a region of lower ground conductivity (<15 mS/m) that probably indicates greater sand content in the upper 6 m.

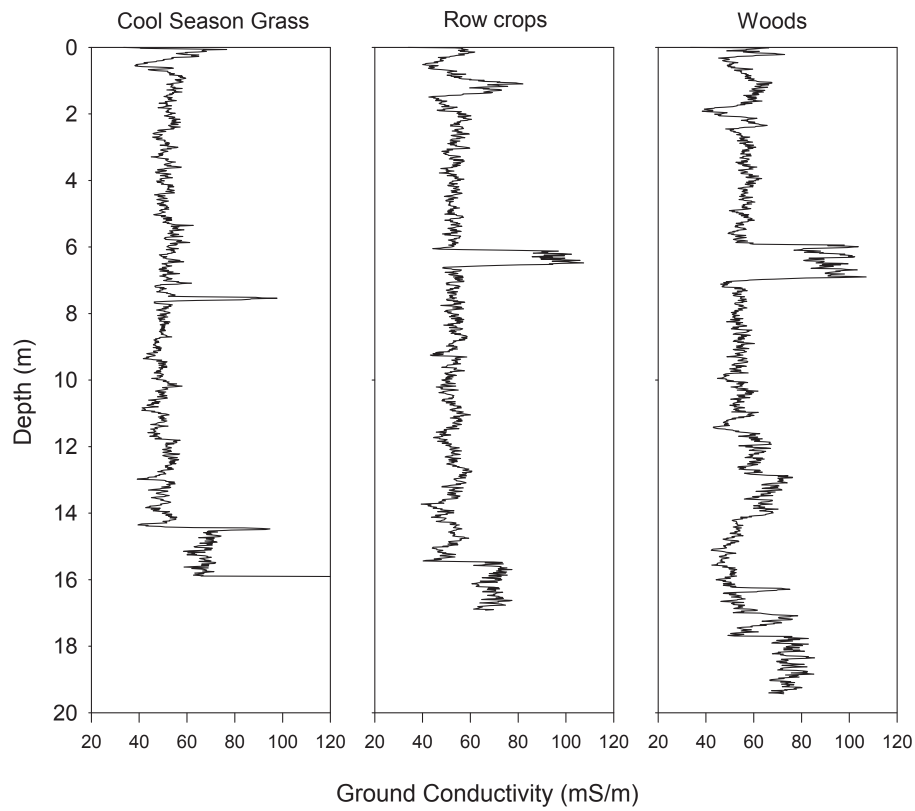

Fig. 2. Borehole geophysical log of ground conductivity measured at the center well of the three land cover sites using the Geoprobe™ hydraulic probe.

3.2. Hydrology

Annual precipitation measured nearby in Muscatine, Iowa, was greater in 2011 (860 mm) and 2013 (890 mm) compared to 2012 (796 mm). Lower amounts of local precipitation in 2012 mirrored regional patterns of moderate to severe drought that occurred in that the latter half of 2011 and all of 2012. Drought conditions were evident in Cedar River discharge patterns (Fig. 4). Average discharge in 2012 was 65 m^3/s, considerably less than average conditions in 2011 (196 m^3/s) and 2013 (262 m^3/s). In 2013, the drought was broken by abundant spring rainfall in April and May (187 and 223 mm, respectively) and flooding conditions observed in the Cedar River on several occasions (Fig. 4). Beginning in March 2013, the Cedar River experienced minor to moderate flooding on 48 non-consecutive days extending until July 4. Maximum flooding occurred on June 4, 2013 when average daily discharge in the river exceeded 1690 m^3/s. While minor flooding also occurred for a six day period in March 2011, when discharge peaked at 764 m^3/s, discharge in the Cedar River did not exceed flood stage for the next 748 days from March 2011 to March 2014.

The water table depth varied considerably throughout the monitoring period (Fig. 4). The water table was deeper at the woods site due to a higher land surface elevation (average of approximately 0.45 m deeper; Table 1) but the temporal patterns of water table fluctuations among the three continuously monitored sites were nearly identical. Missing values for the woods site were due to the water table depth dropping below the bottom of the well but the patterns observed during measurement periods were similarly identical to the grass and crops sites. Overall, there were no apparent differences in water table behavior among the three land covers.

The water table rose above the land surface on two occasions in 2011, none in 2012 and on numerous occasions in 2013 (Fig. 4). High water tables were associated with rising stage in the Cedar River and encroachment of floodwaters from the river into the floodplain site. In 2013, with the exception of a dry period in late March and early April, the water level was consistently above the land surface for much of a five month period between March and July. In contrast, for a 20 month period from mid-2012 to early 2013, the floodplain water table level was more than one meter below the ground surface. Hence our monitoring period was characterized by both wet and dry conditions during the three-year monitoring study.

3.3. Water quality

Viewed as three land treatment populations, there were distinct differences in groundwater conditions among the sites (Table 1, Fig. 5). Groundwater beneath the cropped area had significantly higher specific conductance (SC) than the grass and woods site, and significantly lower dissolved oxygen concentrations ($p < 0.01$; Fig. 5). In contrast, groundwater beneath the woods sites had significantly higher ($p < 0.01$) DOC and NH_4–N concentrations than the grass and cropped areas. Average SRP concentration beneath the woods was significantly higher than the cropped area, and neither woods nor crop sites were different than the grass, but the highest average SRP concentrations in a well were observed in the grass-north well (1.05 mg/l). No differences in water quality among the three land covers were observed for temperature, pH, and ORP.

The greatest difference in groundwater quality among the land covers was associated with increasing NO_3–N concentrations beneath the cropped field beginning in the June 2012 (Fig. 6). From concentrations less than 1 mg/l in 2011 and early 2012, NO_3–N levels increased to more than 25 mg/l in May 2013 with a maximum concentration of 70 mg/l in the crop-east well (Fig. 6). The trajectory of increasing NO_3–N was first evident in the late June

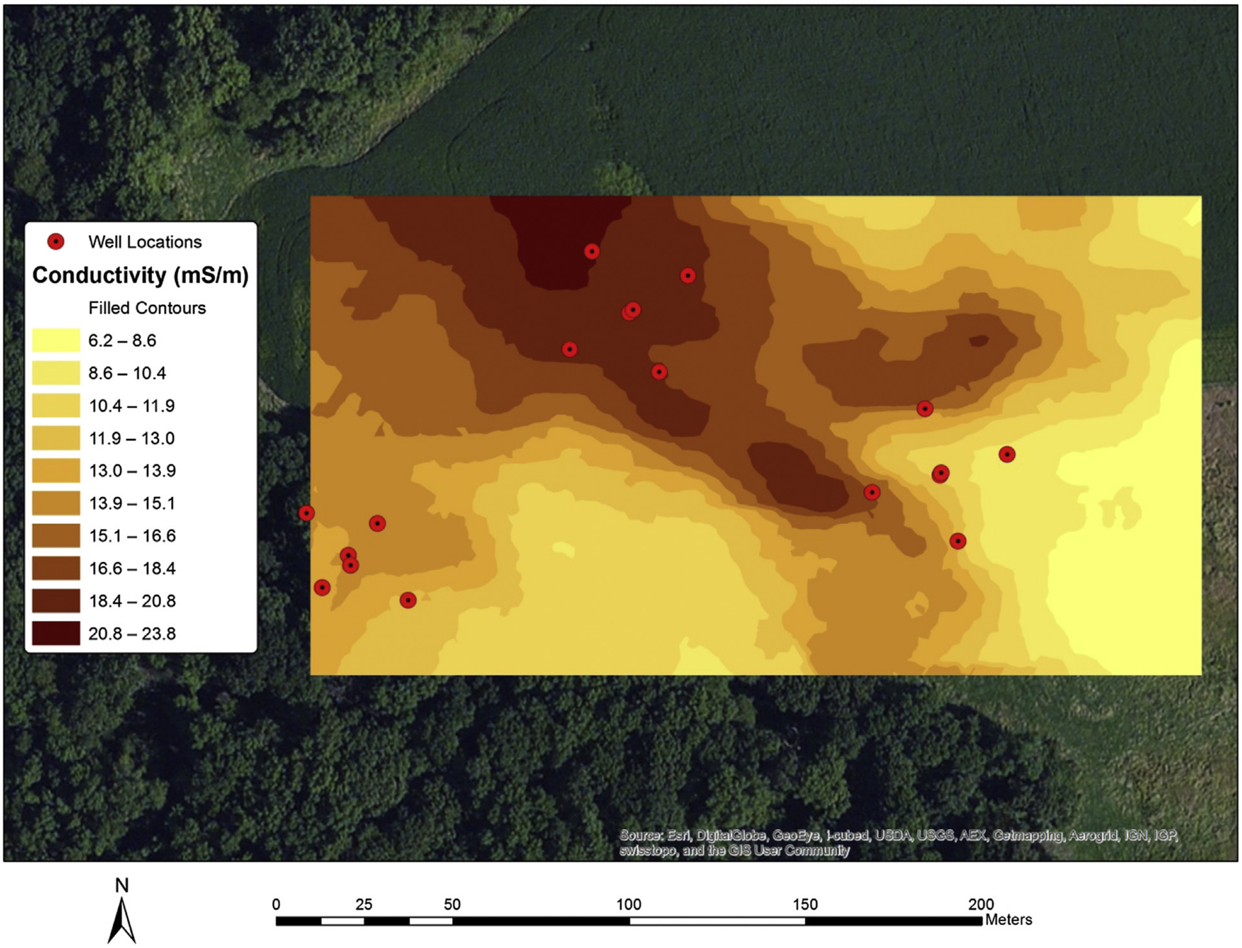

Fig. 3. Ground conductivity measured using surface geophysics (EM-31).

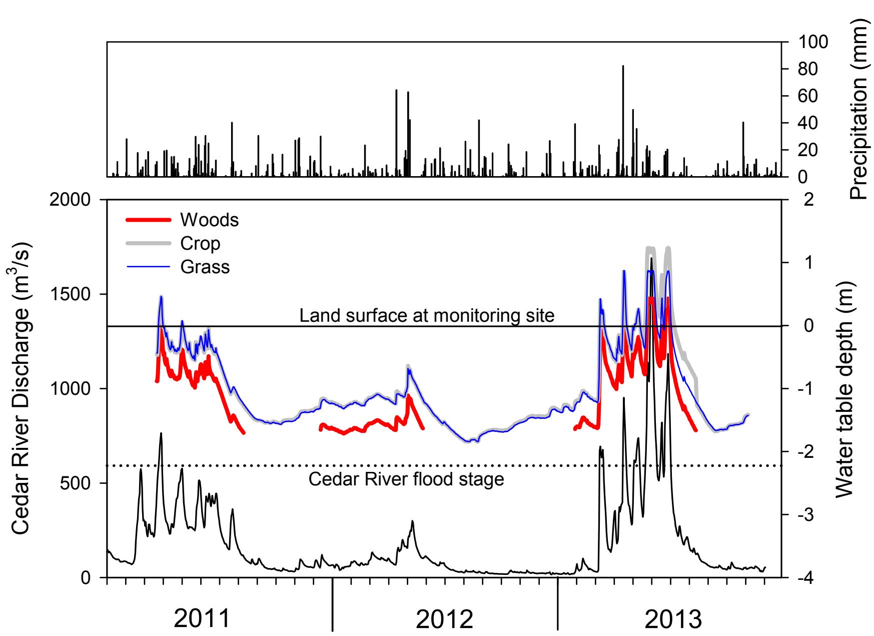

Fig. 4. Time-series of precipitation, Cedar River discharge and water table depth during the 2011–2013 study period. Cedar River discharge data and flood stage information is from the USGS gage near Conesville (station number 05465000) located approximately one km north of the site.

Table 1
Summary of average depth to water and concentrations at land cover type (water table wells only) and by individual well during the 2011 to 2013 monitoring period.

Site	n	Depth to water (m)	NH_4–N (mg/l)	NO_3–N (mg/l)	Soluble reactive phosphorus (mg/l)	Dissolved organic carbon (mg/l)	Total nitrogen (mg/l)	Temp (°C)	pH	Spedifc cond. (umhos/m)	Dissolved oxygen (mg/l)	Oxidation–reduction potential (mv)
Grass	60	2.018	0.047	0.14	0.246	2.47	0.34	13.2	5.93	300	2.50	82
Crop	60	2.014	0.030	12.30	0.041	1.68	10.01	14.0	6.16	451	1.63	70
Woods	51	2.464	0.131	0.74	0.120	3.97	1.04	13.1	5.90	230	3.08	37
Grass-south	12	2.078	0.021	0.19	0.047	2.04	0.32	12.8	5.96	263	2.72	114
Grass-west	12	1.955	0.084	0.04	0.045	2.73	0.33	12.9	5.99	324	1.81	70
Grass-north	12	1.868	0.095	0.13	1.050	4.09	0.39	13.6	5.84	387	1.81	38
Grass-east	12	2.154	0.012	0.16	0.045	1.56	0.36	13.4	5.91	247	3.39	102
Grass-center	12	2.033	0.011	0.17	0.042	1.61	0.32	13.2	5.94	280	2.76	88
Grass-deep	12	2.107	0.011	0.00	0.085	1.57	0.12	12.1	5.77	282	1.38	74
Crop-south	12	2.052	0.018	13.15	0.044	1.46	9.26	14.1	6.12	437	1.95	74
Crop-west	12	2.063	0.016	13.38	0.043	1.58	11.69	13.9	6.16	415	1.37	75
Crop-north	12	1.904	0.070	5.65	0.051	1.75	4.04	14.1	6.20	431	1.52	70
Crop-east	12	2.048	0.024	16.79	0.039	1.57	17.57	13.9	6.16	498	2.04	73
Crop-center	12	2.004	0.029	10.16	0.028	2.01	7.49	14.0	6.17	473	1.31	57
Crop-deep	12	2.120	0.017	1.38	0.052	1.71	0.63	12.8	5.98	415	1.12	49
Woods-south	8	2.473	0.175	0.96	0.210	5.34	1.42	13.8	5.86	209	3.43	13
Woods-west	11	2.564	0.210	0.77	0.138	3.85	0.88	12.3	5.69	195	2.87	50
Woods-north	10	2.472	0.083	0.66	0.107	4.16	1.02	12.3	5.98	164	4.97	81
Woods-east	11	2.340	0.027	0.30	0.048	2.65	0.49	13.1	6.03	337	1.77	60
Woods-center	11	2.474	0.163	1.04	0.122	3.86	1.35	13.5	5.89	223	2.65	−6
Woods-deep	12	2.555	0.020	0.00	0.062	1.34	0.14	12.1	5.60	222	1.02	57

2012 sampling event, and the increase continued in most wells up to May the following year. After the peak NO_3–N concentration in May 2012, in all wells but the crop-south well, concentrations decreased during the latter portion of 2012 with decreasing concentrations measured during sampling events conducted in August and November. During this same time period, SC values also increased in groundwater beneath the cropped site, while decreasing in groundwater beneath the grass and woods sites (Fig. 6). Prior to June 2012, there was no correlation of groundwater SC with NO_3–N at the five crop wells, but after June 2012, increasing groundwater NO_3–N and SC were highly correlated ($r = 0.90$; $p < 0.01$).

In the deeper wells installed in the central location of each land cover site, fewer differences among land covers were observed. SC was significantly different ($p < 0.01$) among the three deep wells, with higher values beneath the cropped field (mean = 415 umhos/m) and lowest values beneath the woods (222 umhos/m). DOC was significantly higher in deep woods groundwater compared to the cropped site, whereas SRP was significantly higher in deep groundwater beneath the grass site compared to the woods and crop sites, with all SRP concentrations generally considered low (less than 0.085 mg/l). NO_3–N concentrations were not detected in any of the deep wells until November 2013 when NO_3–N was detected at 5.5 mg/l in the deep well beneath the cropped field (Fig. 7). The detection of NO_3–N lagged approximately 2 months after increasing Cl concentrations were observed in the deep crop well. While chloride was only analyzed intermittently during our study, with a single value in 2012 and three measurements in 2013, the rising Cl concentration in the deep well beneath the crop site may likely have signaled the arrival of the conservative ion from KCl fertilizer additions in June 2011 (Fig. 7).

3.4. Effects of lithology on water quality

Although groundwater quality was impacted by agricultural activities at the cropped site, particularly with respect to increasing NO_3–N after June 2012, differences among the three land cover sites were also related to lithologic variations within the floodplain sediments. Based on the surface geophysical mapping, the cropped site is underlain by finer-textured sediments characterized by higher ground conductivity (>15 mS/m; Fig. 2). The extent of higher ground conductivity extends into the grass region (north and west grass wells), but most of the grass wells and all of the woods wells are located in a portion of the floodplain characterized by lower conductivity values (<15 mS/m) and greater sand content. Monitoring results indicated that groundwater within the finer-textured cropped site, as well as the grass north and west wells, had higher SC values than the other floodplain wells. Across the 15 wells over two sampling events ($n = 30$), correlation of groundwater SC with ground conductivity was statistically significant ($r = 0.52$; $p < 0.01$). Conversely, DO concentrations were higher in groundwater beneath the woods sites and lower beneath the fine-textured areas, and DO concentrations were negatively related to ground conductivity ($r = -0.43$; $p = 0.02$).

No significant relation ($p > 0.1$) was observed between the peak NO_3–N concentration detected in the five cropped wells and ground conductivity measured at the well site. There was, however, a marginally significant ($p = 0.06$) relationship between the amount of decrease in NO_3–N concentrations observed from the peak concentration (expressed in % decline) to ground conductivity (Fig. 8). A greater decline from peak NO_3–N concentrations in groundwater was observed in wells screening finer-textured sediments, even though the range in ground conductivity measured at the cropped wells was not particularly large (~3 mS/m).

4. Discussion

Our study of the Cedar River floodplain in southeast Iowa was focused on comparing groundwater hydrology and nutrient dynamics associated with three common floodplain land cover types, woods, crops and grass. In order to monitor a crop site, the land use in a cool season grass area was converted to row crop for this study. Study results demonstrated clearly that land use conversion from perennial grass to row crop production on the Cedar River floodplain impacted groundwater quality. Following the first year of soybean, the liquid N fertilizer additions in June of the second year resulted in a rapid increase of NO_3–N concentrations in shallow floodplain groundwater. Concentrations increased from less than

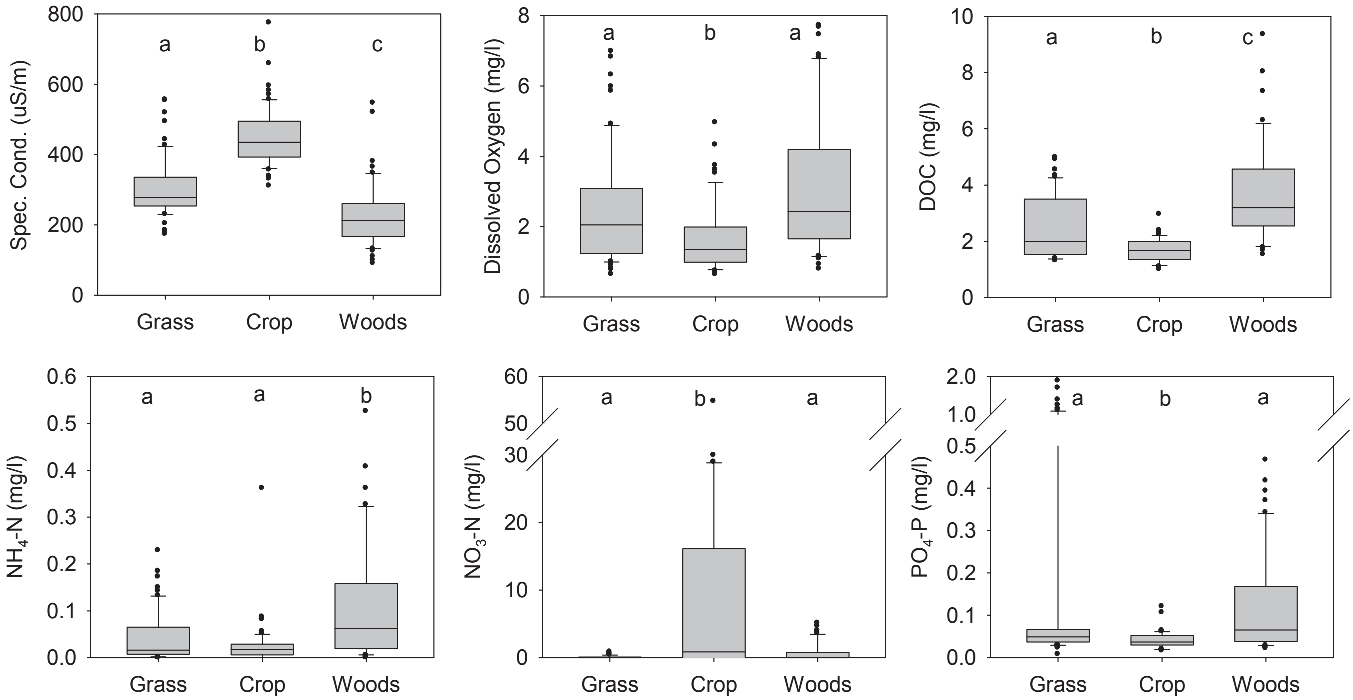

Fig. 5. Box plot of water quality parameters measured in monitoring wells sampled during this study. Letters denote significant differences ($p < 0.05$ for all comparisons).

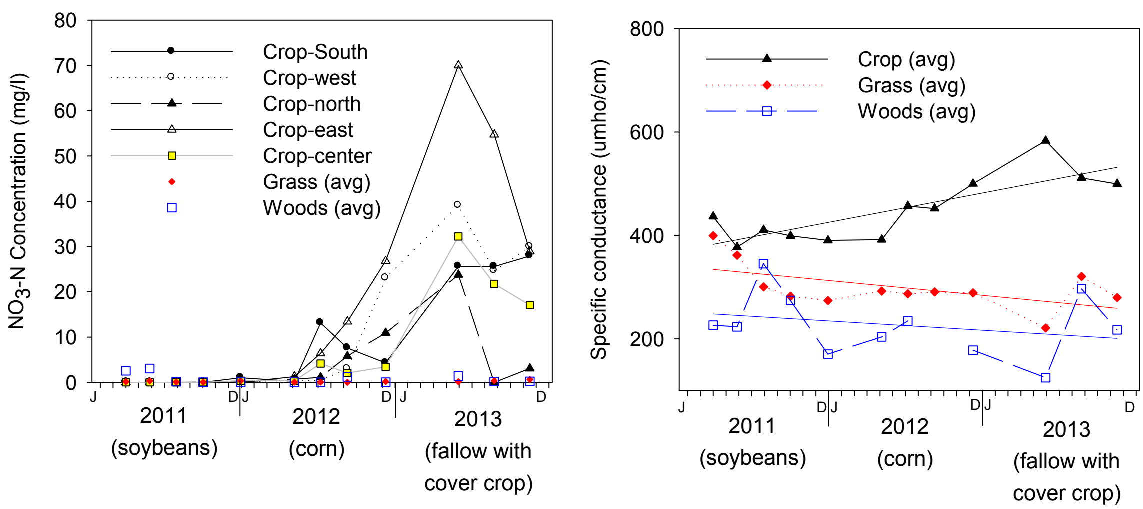

Fig. 6. Left: NO_3–N concentrations measured in shallow groundwater during the 2011–2013 study period. Right: Average specific conductance values measured in groundwater in the shallow wells at the three land cover types.

1 mg/l to 70 mg/l in less than one year in one water table well but all five wells in the new crop area demonstrated an increase of at least 25 mg/l during this time.

4.1. NO_3–N sources

We can trace the increase in groundwater NO_3–N concentrations to fertilizer applications since the increase began following application of 220 lbs/ac urea ammonium nitrate solution in June 2012. In April 2012, the five shallow crop wells had an average NO_3–N concentration of 0.5 mg/l, but in late June after application, average NO_3–N concentrations were 5 mg/l, increasing to 6.3 mg/l in August and 13.7 mg/l in November. By May the following year, average concentrations were 38.1 mg/l. However, while the source of the NO_3–N in groundwater was likely the fertilizer application, hydrologic conditions probably played an important role in enhancing NO_3–N leaching the following spring. In the spring of 2013, the floodplain was flooded on several occasions with the first event occurring in March 2013 (Fig. 4). Water level monitoring indicates that flooding depth exceeded 0.5 m at the site, thereby providing a hydraulic head gradient to mobilize NO_3–N downward through the soil profile during flood recession. Hence, when groundwater samples were collected in May 2013, NO_3–N concentrations exhibited a significant increase from the previous November 2012 levels. Flooding impacts were severe enough in the spring that no crop was planted in 2013 at the site. Although a preventative cover crop was planted in June 2013, the major loss of NO_3–N to groundwater had already occurred.

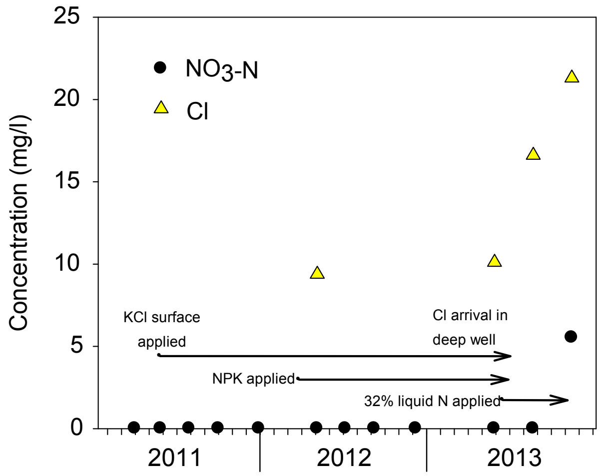

Fig. 7. NO_3–N and Cl concentrations measured in the deep well beneath the cropped site relative to the timing of fertilizer applications.

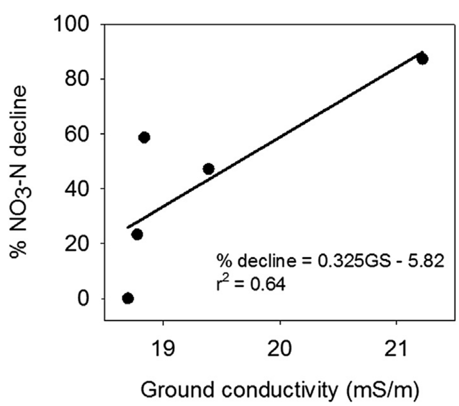

Fig. 8. Relation of NO_3–N concentration decline in shallow groundwater at the cropped site from their maximum value to last measured value (November 2013) (in percent) to ground conductivity measured at the individual well site.

Other sources of NO_3–N to groundwater at the crop site can be ruled out as major contributing factors to the increase. Prior to planting, the grass was burned off and the soil cover was left bare, possibly leaving the site vulnerable to mineralization losses (Schilling and Jacobson, 2008). Zhou et al. (2010) reported an increase of approximately 10 mg/l in groundwater following brome grass conversion to row crops at a toeslope landscape position. Similarly, at a floodplain site in south-central Iowa, reed canary grass was removed from the floodplain for a restoration project (burning and glyphosate treatment) and the ground was left bare for much of a year. No fertilizer additions were made yet groundwater NO_3–N concentrations increased from less 1 mg/l to 40 mg/l due to N mineralization and water leaching through bare floodplain soils (Schilling and Jacobson, 2008). The difference in effects was likely due to the amount of time allowed for mineralization to occur. At the restoration site, bare soil was left exposed for much of a year whereas at our floodplain crop site, soybeans were planted soon after burn-down. Other fertilizer applications had little impact on NO_3–N concentrations. During the first year of soybean, monoammonium phosphate was applied to the field and in the following March, granular NPK fertilizer was applied. However, groundwater NO_3–N concentrations beneath the crop site remained less than 0.5 mg/l. Hence, we conclude that the major impact from fertilizer was primarily from the liquid N application rather than earlier granular applications.

However, we also observed increasing SC in groundwater during the monitoring period that would seemingly relate to application of granular amendments, most likely the application of 70 lb/ac KCl in June 2011. SC increased beneath the cropped site and was correlated with increasing NO_3–N levels during the latter portion of the study period. During the same time, SC levels were decreasing beneath the woods and grass sites (Fig. 6). The arrival of Cl in the deep well in the cropped site suggests that vertical migration of water table impacts to deeper strata had occurred. Two months after the detection of increasing Cl, NO_3–N was detected (5.5 mg/l). If we assume the source of Cl was KCl application in June 2011 and the arrival of Cl in the deep well was in August 2013 (~800 days), the vertical groundwater travel time through 2.8 m of aquifer is estimated to be approximately 0.0035 m/day. This vertical groundwater flow velocity is about one order of magnitude less than horizontal velocity measured within the Cedar River floodplain at a site located one mile north of the current site (Schilling and Jacobson, 2009). The ratio of horizontal to vertical groundwater flow rates is consistent with common understanding (Freeze and Cherry, 1979).

Finally, floodwaters flowing over the floodplain may be a potential source of N to groundwater. Floodwater NO_3–N concentrations in the Cedar River measured during spring 2008 flooding ranged from 2 to 6 mg/l (unpublished data) and this concentration range is similar in magnitude to shallow groundwater NO_3–N concentrations measured beneath the crop site in June and August 2012. However, since no concentration impacts were observed at either the grass or woods sites site this would seem to eliminate floodwater as a major N source.

Other studies have observed rapid increase in groundwater NO_3–N concentrations following land use conversion from perennial grass to row crop (Huggins et al., 2001; Schilling and Spooner, 2006). At a small watershed scale (118 ha), Schilling and Spooner (2006) reported an increase in stream water NO_3–N concentration of 11 mg/l in the span of 10 years, with most of the change concentrated within a span of four years following conversion of grassland to row crop. Thus, data from the floodplain land use conversion reported herein are consistent with previous studies and confirm the vulnerability of groundwater to increased NO_3–N losses following land use conversion from perennial vegetation, particularly in floodplain environments with shallow water tables and dynamic hydrology.

4.2. Spatial patterns in groundwater quality related to lithologic variations

Geophysical investigation revealed differences in ground conductivity among land cover, with higher conductivity beneath the cropped site and lower values at the woods and grass sites. Ground conductivity is proportional to soil texture, with lower values associated with greater sand content and higher values indicating greater silt and clay fraction (Schilling and Jacobson, 2011). Water quality variations in the floodplain were related to the lithologic patterns with groundwater within higher ground conductivity zones (finer textured) exhibiting higher SC and lower DO relative to groundwater within areas characterized by low ground conductivity. The relationship between lithology and water quality is consistent with results reported from another Cedar River floodplain site located approximately one mile north (Schilling and Jacobson, 2011). At the Swamp White Oak floodplain savanna site, groundwater nutrient concentrations were closely linked to ridge and swale topography and lithologic variations. Groundwater located beneath sand-dominated ridges had low SC and high DO whereas groundwater located beneath fine-textured swales was typified by high SC and low DO. Higher concentrations of NH_4–N, SRP and DOC were found in groundwater beneath swales. At the current floodplain monitoring site, although we found these same water quality variations related to lithology, the typical floodplain ridge and swale topography was not present. Instead we observed little topographic variation (with the wood site being about 0.4 m higher elevation) and there was no surface expression of topographic variations that may have provided clues to subsurface lithologic differences. LiDAR elevation mapping in the area suggests that macro-scale variations in floodplain topography could have resulted in lithologic variations, but these features are not observable at eye level. Hence, within two floodplain environments of the same river (Cedar River), water quality variations are clearly related to lithology but the surface expression of topographic and lithologic patterns are quite different.

More work is needed to understand the floodplain depositional environments in the lower Cedar River to identify where zones of high and low ground conductivity may be located because these areas are the focus of enhanced biogeochemical activity (Schilling and Jacobson, 2011). The positive relationship between the rate of NO_3–N concentration decline and ground conductivity (Fig. 8), suggests that denitrification may be occurring within the fine-grained sediments beneath the cropped field site as biogeochemical conditions, including low DO (average of 1.5 mg/l) and abundant organic carbon (1.7 mg/l), are favorable. Denitrification is often cited as an important biogeochemical process in floodplain soils (Pinay et al., 2007; Forshay and Stanley, 2005). Although denitrification is likely, other factors such as dilution or nitrogen uptake by the cover crop plantings (rye and radishes) may play a role, along with transfer to deeper portions of the aquifer.

The concentration patterns of other nutrients appear less affected by aquifer lithology. DOC concentrations were highest in shallow and deep groundwater beneath the woods site whereas SRP concentrations were statistically similar at the grass and woods sites and lower beneath the crop site. Low SRP concentrations at the crop site suggest that application of monoammonium phosphate in June 2011 did not impact groundwater quality during the study period. However, if we consider the average SRP concentrations measured during the entire study as indicative of typical background floodplain concentrations in the area (average 0.12 mg/l), concentrations would approach or exceed proposed nutrient criteria for streams in nearby ecoregions (0.18 and 0.092 mg/l for Ecoregions 47 and 40, respectively; USEPA, 2000). SRP concentrations measured in the grass north well would greatly exceed these criteria (1 mg/l; Table 1).

4.3. Risks of floodplain farming

Study results confirm that farming on floodplains comes with risk. During our three-year study, a crop was harvested for two of the three years but flooding prevented cropping for a third year. The one out of three failure rate (33%) may be low given the hydrologic history of the river. The long-term record of peak streamflow and stage in the Cedar River from 1940 to 2013 indicates that flood stage has been exceeded for at least one day per year during 52 of 74 years (70%) of the monitoring record. Obviously, the timing of year and duration of flooding would impact whether or not a crop would have been successfully planted or harvested if the land were farmed continuously during the gauging record. However, the fact that flooding prevented cropping during one year of our three year study is certainly not a surprise. A preventative cover crop was planted during the flood year, but NO_3–N concentrations remained elevated throughout the non-crop year. Our results indicate that the legacy of floodplain farming during the corn rotation carried over to the following year when the site was flooded and not cropped. Hence, the risk of floodplain farming involves not only loss of planting or a successful crop but also risks associated with carry-over NO_3–N loss to groundwater and rivers during both crop and non-crop years.

Overall, study results provide much-needed information on the effects of different land covers on floodplain groundwater and point to challenges ahead for meeting nutrient reduction goals if row crop land use expands into floodplains. Floodplain farming involves risk not only to the producer for losing a crop due to floods, but risk to the environment of increased NO_3–N loss to groundwater and rivers.

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