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Sediment and Nutrient Retention on a Reconnected Floodplain of an Upper Mississippi River Tributary, 2013–2018

Scientific Investigations Report 2022–5030

U.S. Department of the Interior
U.S. Geological Survey

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By Lynn A. Bartsch, Rebecca M. Kreiling, Lance R. Gruhn, Jessica D. Garrett, William B. Richardson, and Greg M. Nalley

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U.S. Geological Survey [USGS], 2019a, USGS water data for the Nation: U.S. Geological Survey National Water Information System database, accessed October 15, 2018, at https://doi.org/10.5066/F7P55KJN.

U.S. Geological Survey [USGS], 2019b, Long Term Resource Monitoring Program—Water Quality: U.S. Geological Survey Long Term Resource Monitoring Program web page, accessed May, 3 2019, at https://www.umesc.usgs.gov/data_library/water_quality/water_quality_data_page.html.

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Contents

Figures

Tables

Conversion Factors

U.S. customary units to International System of Units

Multiply	By	To obtain
	Length	
inch (in.)	2.54	centimeter (cm)
inch (in.)	25.4	millimeter (mm)
foot (ft)	0.3048	meter (m)

International System of Units to U.S. customary units

Multiply	By	To obtain
	Length	
millimeter (mm)	0.03937	inch (in.)
centimeter (cm)	0.3937	inch (in.)
meter (m)	3.281	foot (ft)
meter (m)	1.094	yard (yd)
kilometer (km)	0.6214	mile (mi)
	Area	
square meter (m^2)	0.0002471	acre
hectare (ha)	2.471	acre
square hectometer (hm^2)	2.471	acre
square kilometer (km^2)	247.1	acre
square centimeter (cm^2)	0.001076	square foot (ft^2)

Multiply	By	To obtain
square meter (m^2)	10.76	square foot (ft^2)
square centimeter (cm^2)	0.1550	square inch (ft^2)
square hectometer (hm^2)	0.003861	section (640 acres or 1 square mile)
hectare (ha)	0.003861	square mile (mi^2)
square kilometer (km^2)	0.3861	square mile (mi^2)
	Volume	
milliliter (mL)	0.03381402	ounce, fluid (fl. oz)
liter (L)	33.81402	ounce, fluid (fl. oz)
liter (L)	2.113	pint (pt)
liter (L)	1.057	quart (qt)
liter (L)	0.2642	gallon (gal)
cubic meter (m^3)	264.2	gallon (gal)
cubic meter (m^3)	0.0002642	million gallons (Mgal)
liter (L)	61.02	cubic inch (in^3)
cubic meter (m^3)	35.31	cubic foot (ft^3)
cubic meter (m^3)	1.308	cubic yard (yd^3)
cubic meter (m3)	0.0008107	acre-foot (acre-ft)
	Flow rate	
cubic meter per second (m^3/s)	70.07	acre-foot per day (acre-ft/d)
cubic meter per day (m^3/d)	35.31	cubic foot per day (ft^3/d)
cubic meter per day (m^3/d)	264.2	gallon per day (gal/d)
cubic meter per second (m^3/s)	22.83	million gallons per day (Mgal/d)
	Mass	
microgram (µg)	3.527×10^{-8}	ounce, avoirdupois (oz)
milligram (mg)	3.527×10^{-5}	ounce, avoirdupois (oz)
gram (g)	0.03527	ounce, avoirdupois (oz)
kilogram (kg)	2.205	pound avoirdupois (lb)
Megagram (Mg)	1.102	ton, short [2,000 lb]
Megagram (Mg)	0.9842	ton, long [2,240 lb]
	Application rate	
kilograms per hectare (kg/ha)	0.8921	pounds per acre (lb/acre)

Temperature in degrees Celsius (°C) may be converted to degrees Fahrenheit (°F) as °F = (1.8 × °C) + 32.

Temperature in degrees Fahrenheit (°F) may be converted to degrees Celsius (°C) as °C = (°F – 32) / 1.8.

Datum

Vertical coordinate information is referenced to the North American Vertical Datum of 1988 (NAVD 88).

Horizontal coordinate information is referenced to the North American Datum of 1983 (NAD 83).

Altitude, as used in this report, refers to distance above the vertical datum.

Abbreviations

APHA	American Public Health Association
DEA	denitrification enzyme activity ([micrograms nitrogen per square centimeter] per hour)
DEN	ambient denitrification rate ([micrograms nitrogen per square centimeter] per hour)
EPC_0	soil equilibrium phosphorus concentration (milligrams per liter) as phosphorus
$exNH_4^+$-N	potassium chloride-exchangeable ammonium (milligrams per gram dry sediment) as nitrogen
exSRP	magnesium chloride exchangeable soluble reactive phosphorus (milligrams per gram dry sediment) as phosphorus
HFV	high-frequency variability
HSD	honest significant difference
M3P	Mehlich-3 digested phosphorus (milligrams per kilogram) as phosphorus
NH_4^+-N	ammonium (milligrams per liter) as nitrogen
NO_3^- -N	nitrate is the sum of nitrate+nitrite (milligrams per liter) as nitrogen
NWIS	National Water Information System
$pwNO_3^-$ -N	sediment interstitial or porewater nitrate+nitrite (milligrams per gram dry sediment) as nitrogen
PSR	phosphorus saturation ratio
r_s	Spearman's rank-order correlation
R^2	coefficient of determination
RMSE	root mean square error
SE	standard error
SRP	soluble reactive phosphorus (milligrams per liter) as phosphorus
TP	total phosphorus (milligrams per liter) as phosphorus in water and (milligrams per kilogram) as phosphorus in sediment and soil
TSS	total suspended solids (milligrams per liter)
UMESC	U.S. Geological Survey, Upper Midwest Environmental Sciences Center
USGS	U.S. Geological Survey

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Abstract

The connection of rivers with their floodplains has been greatly reduced in agricultural drainage basins, especially in the Upper Mississippi River Basin. The restriction of the Mississippi River from its floodplain has reduced the sediment trapping and nutrient deposition capabilities of the floodplain, exacerbating water quality problems in the river and in downstream waterbodies. A small part of the Maquoketa River, a tributary to the Upper Mississippi River, was permanently reconnected to its floodplain in 2010 when a levee failure resulted in breaches in two locations. This study quantified the water quality benefits of that reconnection from October 2013 through September 2018. As part of the study, data from groundwater monitoring wells were used to determined hydraulic connectivity and surface-water/groundwater mixing; soil samples were collected in the floodplain to quantify floodplain sediment and nutrient retention potential during postflood and dry, interflood periods; and sensors were placed in the Maquoketa River to quantify total suspended solids, nitrogen, and phosphorus concentrations and loads.

The floodplain aquifer in the study area had low hydraulic gradients toward the Maquoketa (mean of 0.017) and Mississippi Rivers (mean of 0.0029) and reducing water-quality conditions (dissolved oxygen less than 1.0 milligram per liter [mg/L] and nitrate less than 0.04 mg/L as nitrogen) capable of denitrification. A specific conductance-based mixing indicated precipitation was the predominate source of groundwater; however, specific conductance-based mixing analysis was unable to distinguish between the river or direct precipitation as the source.

The floodplain was fully inundated five times during the study: in June–July 2014, March 2015, January 2017, February 2018, and September 2018. During the March 2015 flood (the only inundation event with sufficient duration to leave quantifiable sediment deposition in the study area), the equivalent of 0.91 percent of the nitrate load and 3.8 percent of the phosphorus load was deposited as sediment on the floodplain. Potential nitrogen losses on the floodplain because of denitrification ranged from 250 kilograms per day (kg/d) as nitrogen in March 2015 to 668 kg/d as nitrogen in October 2014. Potential denitrification rates indicate that when the soil is inundated, inorganic nitrogen present in the soil and in the water column is rapidly denitrified. Soil phosphorus measurements indicated that floodplain soils contain a mean of 365 milligrams per kilogram as phosphorus but still have the capacity to remove phosphorus from flood waters of the Maquoketa River depending on the surface water phosphorus concentration. Results from this study indicate that restoration of even small river-floodplain connections can improve water quality in the Upper Mississippi River.

Introduction

The annual flooding of floodplains is an important natural process in rivers, and it provides beneficial ecosystem services including the translocation of substantial amounts of nutrients, nitrogen (N) and phosphorus (P), and sediment carbon (C) from the river to the floodplain (Hupp and Bazemore, 1993; Tockner and Stanford, 2002; Noe and Hupp, 2005; Kiedrzyński and others, 2008). Historically, flooding replenished nutrients and carbon through deposition of sediments from the overlying flood waters onto the floodplain (Tockner and Stanford, 2002) and has resulted in uniquely productive and diverse biotic ecosystems adapted and reliant on seasonal flooding.

The connection of rivers with their floodplains has been greatly reduced in the past 100 years (Morton and Olson, 2014). Agricultural use and subsequent human settlement of the nutrient-rich floodplains placed people and property in the path of these beneficial annual floods. The beneficial nature of flooding that created these rich agricultural lands was overshadowed by the increasing human costs of property damage and loss of lives because of floods (Morton and Olson, 2014). Construction of flood barriers and control structures to protect human settlement and crops have interrupted many

natural benefits of ecosystem services historically provided through flooding and the connection between rivers and their floodplains (Loken and others, 2018).

Effectiveness of floodplains in providing ecosystem services is dependent on the extent and location of the connection between floodplain and river. Although small connections like levee breaches provide some benefit, expanded connections often allow increased flood volumes for extended periods of time (Opperman and others, 2009; Opperman and others, 2010). Duration of inundation is important for sediment settling and sediment-reactive biogeochemical processes such as reduction-oxidation dynamics, denitrification, and phosphorus sorption-desorption processes (Surridge and others, 2012). Extensive leveeing for flood protection throughout much of the Upper Mississippi River and associated tributaries has substantially limited the frequency of flooding of historical floodplains.

The restriction of the river from parts of its floodplain has reduced sediment trapping and nutrient deposition and removal in these systems, which has exacerbated water quality problems associated with downstream transport of sediment and nutrients. Tributary loading of sediments, nitrogen, and phosphorus to the Upper Mississippi River contribute to the development of river and coastal eutrophication and hypoxic conditions in the Gulf of Mexico (Jones and others, 2018). Nutrient abatement strategies during low flow periods, such as instream biological uptake and benthic denitrification, are largely ineffective at minimizing annual loads of sediments and nutrients (Royer and others, 2006). Instead, high streamflow periods—in which there is minimal instream processing because of short hydrologic retention times—often carry most of the annual yield of nutrients and sediments from their catchments (Hubbard and others, 2011). Recent analyses of changes in nutrient loading from Iowa tributaries indicate that despite substantial efforts by agricultural managers, progress toward limiting nutrient loss is lagging far behind published goals (Jones and others, 2018). Also, the interception of dissolved and particulate contaminants by existing coastal wetlands has not been as substantial as anticipated. Less than 1 percent of nutrients pass through diversions to coastal wetlands in Louisiana, and the percentage would still be less than 8 percent if the entire coastal wetlands were engineered for that purpose (Turner and others, 2007). As freshwater and coastal water quality concerns increase globally, managers continue looking for alternate management strategies to reduce downstream flux on sediments and nutrients.

Alternate management and reconnection of historical floodplains could potentially result in the substantial capture of transported sediments and nutrients, and permanently remove nitrogen via denitrification and store phosphorus through burial and biological capture in floodplain forest biomass. Recent research has determined that managing connectivity of river channels to floodplains effectively removes sediment, nitrogen, phosphorus, and carbon from floodwaters (for examples see Noe and Hupp, 2005; James and others, 2008; Kreiling and others, 2013). Reconnected floodplains also reduce downstream flooding (Remo and others, 2012) through increased capacity for storage. Thus, the reconnection of leveed floodplains may be a cost-effective alternative that may reduce a proportion of the nutrient loads (Mitsch and others, 2001; Hurst and others, 2016).

Purpose and Scope

This report quantifies the effects of increased river-floodplain connectivity on water quality in a 93-hectare (ha) parcel of the floodplain previously disconnected by levees from the Maquoketa River at Green Island, Iowa (U.S. Army Corps of Engineers, 2016) during the October 2013 through September 2018 study period. Specifically, the purpose is to (1) quantify the retention of flood-transported sediment, carbon, nitrogen, and phosphorus on the Maquoketa River floodplain; (2) quantify floodplain sediment denitrification and its relation to nitrate (NO_3^--N) removal on the floodplain postflood and during dry interflood periods; and (3) identify potential factors leading to sediment phosphorus retention or release.

Study Area

The Maquoketa River drains 4,843 square kilometers in northern Iowa (fig. 1). Land-use in the catchment consists of 54.3 percent row-crop agriculture, 25.8 percent pasture and hayland, 11.9 percent forest or natural areas, 7.7 percent urban, and 0.3 percent wetlands and open water (U.S. Department of Agriculture, 2011). The surficial geology in the drainage basin generally consists of soils and landforms developed in glacial and fluvial deposits laid down during the Pleistocene and Holocene (U.S. Department of Agriculture, 2011). Soil loss from the catchment in 2010 was estimated to be about 7,000,000 tons per year (Patrick Flanagan, U.S. Department of Agriculture, Natural Resources Conservation Service, written commun., June 2015). The Maquoketa River transports some of the highest loads of sediment and nutrients in Iowa (Garrett, 2012).

The levee separating the Maquoketa River from its floodplain was repeatedly breached by floods and repaired by the U.S. Army Corps of Engineers until a major flood in 2010 (resulting from a 17-inch rainfall in the drainage basin) resulted in two major breaches (fig. 1; U.S. Army Corps of Engineers, 2016). Levee breaches resulted in the inundation of 151 ha of cropped floodplain by the Maquoketa River. The levee was not repaired, and the U.S. Department of Agriculture, Iowa Department of Natural Resources, and Iowa Natural Heritage foundation purchased the flood-prone land for wildlife benefit (Hodges, 2017).

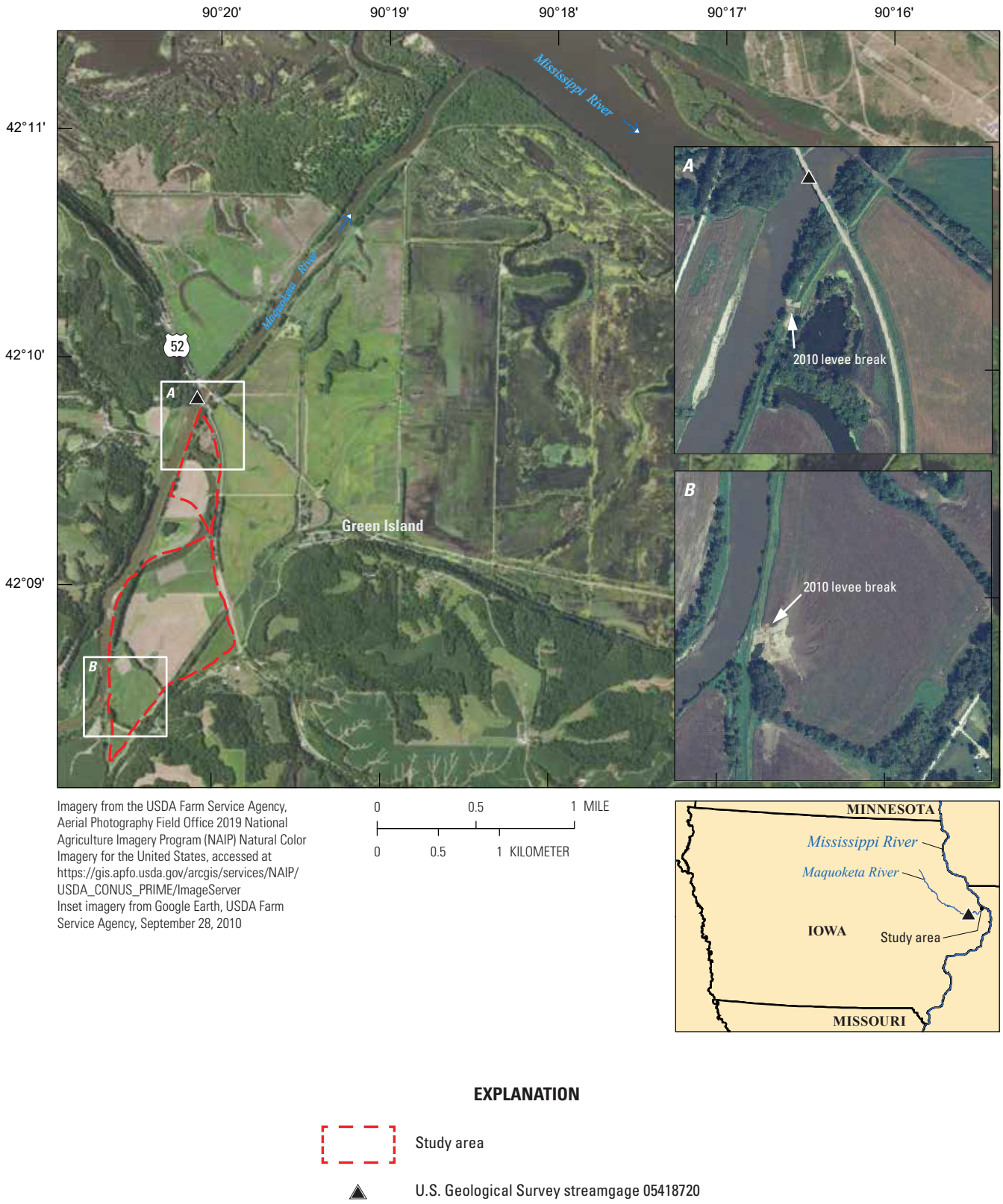


Figure 1. Site locations on the Maquoketa River and Maquoketa River floodplain near the confluence with the Mississippi River at Green Island, Iowa.

This study was conducted on 93 ha of the 151-ha reconnected Maquoketa River floodplain area. Management activities on the research site are complementary to nearby Mississippi River floodplain areas and include seeding a mixture of perennial wet grasses, periodic burning and mowing, and shallow pond scrapes in support of amphibian habitat. Grassy habitat and other early successional vegetation benefit from frequent disturbances, such as flooding, mowing, or burning, to prevent woody species from becoming dominant. During this study, the 93-ha study site contained grasses, prairie forbs, and small trees. Geology in this part of the drainage basin is generally characterized by well-drained and moderately well-drained silty soils over bedrock residuum (U.S. Department of Agriculture, 2011). A former oxbow channel (fig. 2) on the site forms a permanent pond and is inundated during flooding by the Maquoketa River.

Sampling Design and Methods

The primary goal for discrete floodplain sampling was to characterize an open water, Maquoketa River Basin event that resulted in full floodplain inundation. Samples would then be collected in the surface water above the floodplain, and subsequently the wells would be periodically sampled after the event to assess for mixing and potential denitrification in the aquifer. Unfortunately, the desired inundation event never occurred after the clay markers (described in the “Sediment Deposition” section) were emplaced in the fall of 2014 and all but the first inundation event lasted less than 24 hours. However, a secondary goal of the sampling efforts was to target a variety of hydrologic conditions, which was successful. The U.S. Geological Survey (USGS) Central Midwest Water Science Center was responsible for collecting hydrologic data in the Maquoketa River and in groundwater wells on the adjacent reconnected floodplain. Collected surface-water data included continuous (15-minute intervals) water levels, nitrate, turbidity, and temperature. Collected groundwater data included continuous groundwater levels and discrete water quality samples for nutrients. Instantaneous water-quality field measurements were made while pumping the wells before collection of discrete groundwater quality samples. The USGS Upper Midwest Environmental Sciences Center (UMESC) was responsible for collecting data to analyze surficial sediment and nutrient dynamics in the floodplain.

River Streamflow, Concentrations, and Loads

Sensors were used to record river water levels, nitrate, turbidity, and temperature at 15-minute intervals. Streamflow and sensor data are available via the National Water Information System (NWIS) database (streamgages 05418500, 05418600, 05418720; U.S. Geological Survey, 2019a; table 1). Maintenance and quality assurance procedures followed USGS protocols described by Turnipseed and Sauer (2010) for streamflow gaging, Anderson (2005) for turbidity sensors, Wagner and others (2006) for continuous water-quality sensors, and Pellerin and others (2014) for optical nitrate sensors.

A stage-discharge rating was developed for streamgage 05418720; however, it was difficult to maintain because of variation in control conditions at the site. The proximity of the streamgage to the railroad bridge and the confluence with the Mississippi River often led to backwater conditions at the location of the gage (fig. 1). During basin events, the railroad bridge downstream from the streamgage can often catch debris and ice, leading to constricted flow. The confluence with the Mississippi River, about 2.1 miles downstream from the streamgage can also act as a control dependent on Mississippi River stage in relation to stage on the Maquoketa River. Backwater conditions were verified through discharge measurements and hydrologic comparison with the upstream streamgage on the Maquoketa River at the city of Maquoketa (USGS streamgage 05418500). Monitoring of streamflow at the site began on April 10, 2014, and finished on September 30, 2018 (USGS streamgage 05418720). In total there were 1,625 days of monitoring and 68 percent of those days were determined to be affected by backwater from one of the controls mentioned previously (U.S. Geological Survey, 2019a).

Total suspended solids (TSS), nitrate, and phosphorus concentrations and loads were computed from streamflow, nitrate sensor concentration data, and surrogate models (regression) for TSS and phosphorus concentrations based on turbidity and discrete water samples. Approximately monthly river water sample data were accessed from the U.S. Army Corps of Engineer’s Upper Mississippi River Restoration Long Term Resource Monitoring element sample site on the Maquoketa River (Soballe and Fischer, 2004; data available at https://www.umesc.usgs.gov/data_library/water_quality/water_quality_data_page.html), with additional sample data available in the NWIS database (USGS streamgage 05418720; U.S. Geological Survey, 2019a). Continuous (typically 15-minute interval) and daily mean nitrate and phosphorus concentrations and loads are published in NWIS (USGS streamgage 05418720; U.S. Geological Survey, 2019a). TSS concentrations were computed using surrogate models, following procedures used for phosphorus surrogate models and computation of loads described by Garrett (2019). To obtain complete annual load estimates, any remaining gaps in TSS or phosphorus loads were filled using streamflow-based models with streamflow estimated based on the upstream streamgage at Maquoketa (USGS streamgage 05418500) adjusted by drainage area ratio. TSS sample data were used as the basis for sediment loads because these data were available, but TSS data are generally considered less reliable and biased lower than suspended-sediment concentrations (Gray and others, 2000).

Table 1. Site information on surface water and groundwater data-collection sites, Green Island, Iowa, 2013–18.

[--, not applicable]

U.S. Geological Survey Site identifier	Site name	Field identifier	Latitude	Longitude	Land-surface elevation in feet above North American Vertical Datum of 1988	Total depth (feet below land surface)	Measuring point height above land surface (feet)
05418720	Maquoketa River near Green Island, Iowa	Maquoketa River at Green Island	42°09'49.9	90°20'06.3	—	—	—
05418500	Maquoketa River near Maquoketa, Iowa	Maquoketa River at Maquoketa	42°05'00.0	90°37'58.0	—	—	—
05418600	Maquoketa River near Spragueville, Iowa	Maquoketa River near Spragueville	42°06'04.5	90°31'03.9	—	—	—
420931090200102	085N05E24CBCA 2014USGS Green Island W2	W2	42°09'31.0	90°20'01.0	595.18	22.9	2.10
420932090201003	085N05E23ADAC 2014USGS Green Island W3	W3	42°09'32.6	90°20'10.7	595.73	17.5	2.51
420924090200904	085N05E23DAAA 2014USGS Green Island W4	W4	42°09'24.9	90°20'09.6	595.70	17.6	2.40
420919090201705	085N05E23DABD 2014USGS Green Island W5	W5	42°09'19.8	90°20'17.8	597.31	18.0	2.00
420912090200406	085N05E24CCBB 2014USGS Green Island W6	W6	42°09'12.1	90°20'04.4	596.59	17.9	2.10
420902090202007	085N05E23DDCC 2014USGS Green Island W7	W7	42°09'02.3	90°20'20.9	597.21	17.9	2.10
420850090201608	085N05E26AACA 2014USGS Green Island W8	W8	42°08'50.7	90°20'16.8	597.50	17.9	2.13
420855090195709	085N05E25AAAA 2014USGS Green Island W9	W9	42°08'55.9	90°19'57.9	595.63	18.3	1.70
420836090201910	085N05E26ADCD 2014USGS Green Island W10	W10	42°08'36.3	90°20'19.1	596.06	18.1	1.90
420842090203311	085N05E26ACBD 2014USGS Green Island W11	W11	42°08'42.0	90°20'33.9	597.60	17.6	2.15
420824090202812	085N05E26DBDA 2014USGS Green Island W12	W12	42°08'24.1	90°20'28.2	598.95	23.2	1.79

The surrogate regression models for TSS concentration at the Maquoketa River near Green Island, Iowa, were based on turbidity and streamflow (Garrett, 2019). The primary turbidity-based model was used to compute continuous (15-minute interval) TSS concentrations:

$$\ln(TSS) = \text{for } TURB > 20,\ 0.966 \times \ln(TURB) + 0.524 \quad (1)$$

or

$$TSS = 1.891 \times TURB^{0.966} \quad (2)$$

where

TSS is total suspended-solids concentration, unfiltered, in milligrams per liter (mg/L); and

$TURB$ is turbidity, in formazin nephelometric units.

Turbidity-surrogate model calibration samples included 29 observations for a range of conditions during the study period. An alternate streamflow-based model (eqs. 3 and 4) was applied at a daily step during gaps to obtain a more complete record of loads despite periods of fragmentary sensor data.

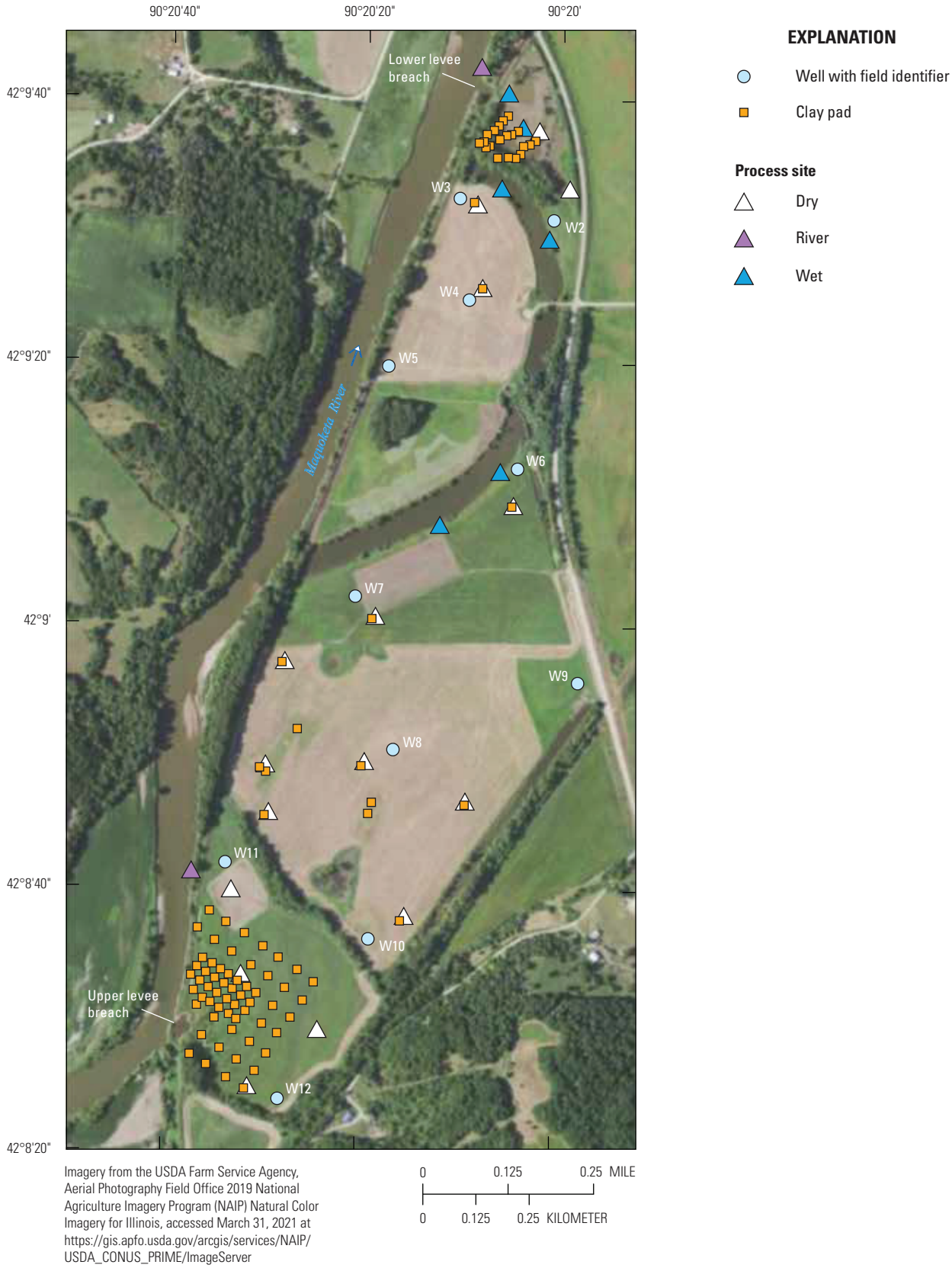
$$\ln(TSS) = 1.13 \times \ln(Q) - 8.198 \quad (3)$$

or

$$TSS = 0.0003524 \times Q^{1.13} \quad (4)$$

where

Q is streamflow in cubic meters per second (m^3/s).



Imagery from the USDA Farm Service Agency, Aerial Photography Field Office 2019 National Agriculture Imagery Program (NAIP) Natural Color Imagery for Illinois, accessed March 31, 2021 at https://gis.apfo.usda.gov/arcgis/services/NAIP/USDA_CONUS_PRIME/ImageServer

Figure 2. Maquoketa River floodplain study area detail.

Because calibration samples were collected throughout the study, in all seasons, and covered most of the range of predictor variables, models were extrapolated beyond the sampled range except where outliers were excluded or the calibration data were purposefully constrained (such as in the turbidity-based model, which excluded samples with turbidity less than 20 units). In these cases, models were applied only to the reduced range of data.

Gaps in nitrate loads were filled based on regression with streamflow (including estimated streamflow) using LOADEST in the "rloadest" package in R (Runkel and others, 2004; Runkel and De Cicco, 2017: R Core Team, 2019). Calibration data for the nitrate LOADEST model were flow-stratified random observations selected from discrete water sample results and daily mean sensor values from October 2012 through September 2019. Samples were selected a minimum of 7 days apart drawn first from observations with streamflow greater than 60 m^3/s. Equations 5 and 6 described a LOADEST model used to compute nitrate loads on days without sensor data at the Maquoketa River near Green Island, Iowa.

$$\ln(L) = 1.2992 \times \ln(Q) - 0.5082 \times \mathrm{dQ7d} - 1.6887, \text{ or} \quad (5)$$

$$L = 0.1848 \times Q^{1.2992} \div \exp(0.5082 \times \mathrm{dQ7d}) \quad (6)$$

where

- L is nitrate load in megagrams per day,
- Q is mean daily streamflow in cubic meters per second, and
- $dQ7d$ is difference between mean daily streamflow and the mean streamflow of the previous 7 days (Garrett, 2012)

The surrogate regression models (eqs. 7 and 8) for total phosphorus (TP) concentrations were based either on turbidity and streamflow or streamflow only (Garrett, 2019).

$$TP = 0.01608 \times TURB^{0.631} \div \exp(0.000443 \times dTime) \text{ or} \quad (7)$$

$$TP = 0.000355 \times Q^{0.894} \quad (8)$$

where

- TP is total phosphorus concentration, unfiltered, in milligrams per liter,
- $TURB$ is turbidity, in formazin nephelometric units,
- $dTime$ is centered date in decimal days, and
- Q is streamflow in cubic meters per second.

Surface Deposition and Nutrient Processing

The mass of sediment depositions during flooding was measured using feldspar clay marker horizons (Cahoon and Turner, 1989; Hupp and Bazemore, 1993; Noe and Hupp, 2009). One hundred feldspar clay marker horizons (clay pads) were established throughout the floodplain in fall 2014 in preparation for spring floods of the following year (fig. 2). Dense grids of clay pads were deployed adjacent to the upper and lower levee breaks and randomly throughout the rest of the floodplain. Each point was surveyed for geospatial location. After flooding, sediment was sampled by coring through the sediment accumulated on top of the clay pad. Deposited sediment was sampled on May 13, 2015, after an early spring flood (fig. 3).

In October 2014, 24 sampling sites were randomly assigned across the 93-ha site using geographic information system ArcMap software. These locations were repeatedly sampled for the duration of the study. Included in these fixed sites were 16 dry sites distributed throughout the central part of the site; 6 wet sites in water-saturated floodplains including 4 sites in the oxbow lake and 2 sites in the pond associated with the downstream levee breach; and 2 sites in the Maquoketa River channel. Sediment or soil cores (5-centimeter [cm] depth, 7.6-cm diameter) were collected at each location for determination of denitrification and variables known to affect denitrification in fall 2014 (before spring floods; October 28, 2014), early spring 2015 (March 18, 2015), and late spring 2015 (May 6, 2015; fig. 3).

Sediment denitrification was measured within 24 hours of sample collection. Denitrification rate was determined in the laboratory at UMESC using the acetylene block method (Groffman and others, 1999; Richardson and others, 2004). Homogenized sediment slurries were incubated under anoxic conditions at ambient temperatures with acetylene inhibition techniques to determine potential rates (denitrification enzyme activity [DEA], with the addition of nitrate [14 mg/L as NO_3-N] and carbon [12 mg/L as glucose-C]) and ambient denitrification rates (DEN, no additions). Slurries for denitrification incubations were created from either deionized water at the dry sites or surface water at the wet sites.

In early summer 2016 (June 21, 2016) during a long period without floods (greater than 365 days; fig. 3), denitrification rates were measured in incubations of dry floodplain soils and floodplain soils amended with river water to simulate a summer flood. Soils were collected by coring from all 24 fixed sites to measure DEN and DEA as in the previous sampling periods. A flooding event was simulated by incubating all soil/sediment samples with Maquoketa River water (concentration of 11.53 mg/L NO_3-N). Also, the dry sites were incubated without water additions to determine anoxic soil denitrification rates (ambient porewater nitrate concentration of 1.0 mg/L NO_3-N).

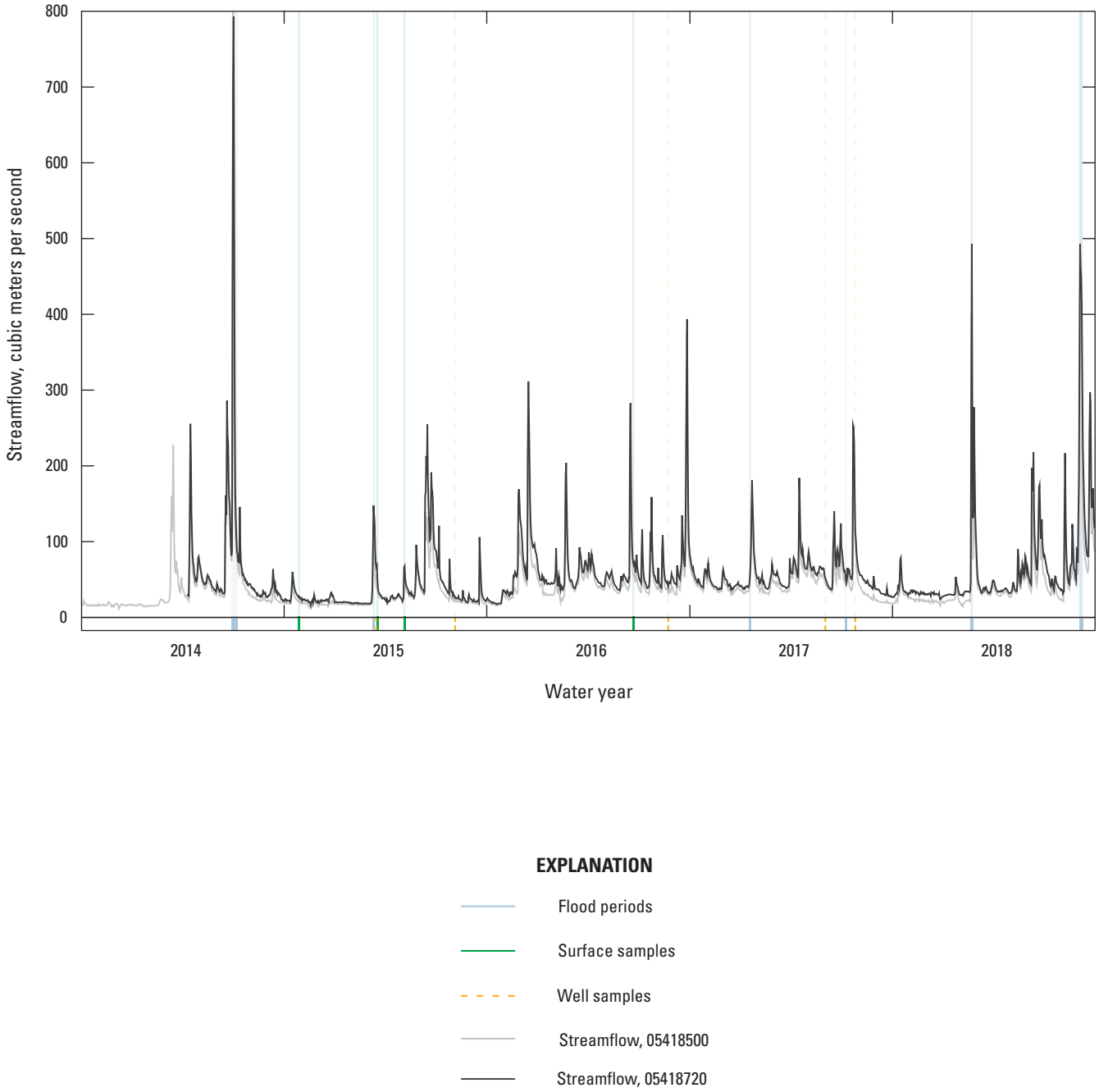


Figure 3. Streamflow of the Maquoketa River at Green Island (U.S. Geological Survey [USGS] streamgage 0518720; black line) and at Maquoketa (USGS streamgage 05418500; gray line), Iowa, flooding events indicated with light blue vertical lines, groundwater well sampling indicated with tan dashed vertical lines. Surface soil sites were sampled on October 28, 2014; March 18, 2015; May 6, 2015; and June 21, 2016; and are indicated with green vertical lines. [Streamflow data from USGS, 2019a.]

Soil and Sediment Nutrient and Physical Properties

Percent total carbon and percent nitrogen from sediment and soil were determined with an Elementar VarioMax CN analyzer (Hanau, Germany) at UMESC. Sediment total phosphorus concentration was determined using standard methods (APHA, 1998) at RTI Laboratory (Livonia, Michigan). Potassium chloride exchangeable ammonium ($exNH_4^+$-N) concentration, porewater nitrate ($pwNO_3$-N) concentration, bulk density, ash-free dry weight, percentage total porosity, percentage water-filled pore space, and percentage water were also measured at UMESC from the sediment cores following Robertson and others (1999).

Biologically Available Soil Phosphorus

In June 2016, soil cores from the 24 fixed sites were analyzed for Mehlich-3 phosphorus (M3P) and cations. Soil cation composition was determined using the Mehlich-3 method on particles that had passed through a 2-millimeter sieve (Mehlich, 1984) at the University of Minnesota Research Analytical Laboratory. Cations measured were aluminum (Al), calcium, iron (Fe), magnesium, and manganese. From this information, phosphorus saturation ratios (PSR) were calculated (PSR = M3P / [Fe + Al]; Maguire and Sims, 2002). PSR indicates the extent to which phosphorus binding sites are available in soils. Additionally, the soil equilibrium phosphorus concentration (EPC_0; Froelich, 1988; Haggard and others, 2004; Tye and others, 2016) and $MgCl_2$-exchangeable soluble reactive phosphorus were determined within 24 hours of sample collection. The $MgCl_2$-exchangeable soluble reactive phosphorus (SRP) was extracted from the soil and analyzed on a discrete autoanalyzer with the ammonium molybdate method (APHA, 1998). Equilibrium phosphorus concentration zero represents no net uptake or release from the sediment to the water column (EPC_0) and was determined by placing 1.5 grams (g) of wet soil into six 50-milliliter (mL) centrifuge tubes along with a series of 30 mL of 0.01 molar potassium chloride solutions with one of the following phosphorus concentrations: 5.0, 2.0, 1.0, 0.1, 0.05, or 0 mg/L as phosphorus. The samples were shaken for 24 hours at room temperature, centrifuged, and the aliquot was filtered and analyzed for SRP on a discrete autoanalyzer (APHA, 1998). The EPC_0 was estimated by a simple linear regression of the amount of phosphorus sorbed (milligrams per kilogram [mg/kg] as P per dry sediment) against the final SRP concentration (mg/L as P) in the centrifuge tube (Pant and Reddy, 2001). The x-intercept was the EPC_0, where phosphorus uptake or release from the soil was negligible.

Extrapolation of Surficial Site Measurements to Study Area

Extrapolation of site-based measurements of carbon, nitrogen, and phosphorus from sediment deposition in the study area was conducted by kriging using the geostatistical package Surfer (Version 12, Golden Software, Golden Colorado; https://www.goldensoftware.com/). Kriging is a form of weighted local averaging. It is an optimal interpolation method where the interpolated values are unbiased and have minimum variance, because the interpolation weights are derived from a geostatistical spatial analysis based on the sample semivariogram. Also, the variance of the interpolated values can be estimated so that the estimation error is known (Burgess and Webster, 1980).

Statistical Analyses

Data on soil nutrient concentrations, physical properties, and nutrient processes were analyzed for normality and homogeneity of variances in R version 3.4.2 (R Core Team, 2019). Data not meeting these assumptions were natural-log transformed. If data still did not meet requirements after transformations, nonparametric tests were done. We did one-way analysis of variance tests (ANOVAs) on the denitrification data to determine differences among the time periods and sample location type. We also did ANOVAs on the June 2016 denitrification data to determine differences about the treatment means. We tested significant differences ($\alpha = 0.05$) among treatment means using Tukey Honest Significant Difference (HSD) test. Spearman's rank order correlations (r_s) were run to determine correlations between the denitrification predictor variables and denitrification rates and between the phosphorus predictor variables and PSR and EPC_0. Field and laboratory data for floodplain sample sites and clay marker horizons are available in a published U.S. Geological Survey data release Kreiling and others (2020) at https://doi.org/10.5066/P9A12PVX.

Groundwater Levels and Chemistry

Eleven 2-inch diameter wells (W2–W12) finished with flush-joint polyvinyl-chloride (PVC) casing were installed for monitoring groundwater levels and collecting discrete water quality samples. These wells were approximately equidistant from each other and spatially dispersed to cover almost the entire floodplain (fig. 2). Wells were drilled with a truck-mounted Simco 2800 HS HT drill rig using 3.25-inch hollow stem augers and hollow stem augering techniques. Saturated sands collapsed around well casings and the remaining voids were filled with silica sand and then a bentonite seal topped with concrete cap. Six-inch wide PVC was used for outer casings with a hole drilled at the bottom to create a pressure gradient preventing direct surface water and groundwater interaction during inundation.

Wells were set to a depth of 17–18 feet and constructed using a 10-foot screened casing and a 10-foot solid casing except for wells W2 and W12. W2 was at the lowest land-surface elevation in the study area, and well W12 is at the highest land-surface elevation. Wells W2 and W12 were set to depths of about 23 feet, and each consisted of 15 feet of screened casing and 10 feet of solid casing. The naming convention for the monitoring wells is the year the well was installed (2014), the agency identifier (USGS), and the local project identifier (Green Island W2–12). For the purpose of this report, only the last part of the local project identifier (W2–W12) will be used for well identification (table 1).

Groundwater Levels

Wells were instrumented with unvented submersible transducers from December 12, 2014, to October 23, 2018, with an additional sensor kept above ground for barometric compensation to the groundwater-level data. During visits, recorded groundwater levels were verified with manual measurements and data were periodically downloaded. Groundwater-level data are available in NWIS database (U.S. Geological Survey, 2019a).

Groundwater Samples

To characterize groundwater quality in the reconnected floodplain, discrete samples were collected from each of the wells on five different occasions during the study. Sampling was targeted to collect a variety of hydrologic conditions on the Maquoketa River near Green Island, Iowa (USGS streamgage 05418720). Before sampling, each well was pumped for 12 minutes then instantaneous readings for field properties of pH, dissolved oxygen, specific conductance, temperature, and nitrate were collected to determine a stable mean value for each property. After pumping, discrete samples of the aquifer water were collected and analyzed by the USGS National Water Quality Laboratory using methods described by Fishman (1993) and Patton and Kryskalla (2011) for concentrations of nitrogen as ammonia, nitrogen as nitrite, nitrogen as nitrite plus nitrate, and phosphorus as orthophosphate (ortho-P).

Mixing Model Analysis

A simple two end-member mixing model based on specific conductance was used to identify samples dominated by precipitation-event water or base groundwater. With groundwater level gradients, the groundwater mixing analysis was used to define potential hotspots for delivery of nitrate-laden waters to the floodplain groundwater area.

The event end-member was based on stream samples collected from the Maquoketa River. Event samples were classified based on streamflow anomaly variables computed in R (R Core Team, 2019; Lorenz and Diekoff, 2017) as described by Vecchia (2003). Because a long period of record is required to calculate anomalies, the upstream USGS streamgage on the Maquoketa River at Maquoketa (USGS streamgage 05418500) was used for streamflow. Data from three nearby sites were used to develop a larger calibration set with a longer period of record than was available from the streamgage adjacent to the study area (USGS streamgage 05418720). Sample data for the Maquoketa River near Green Island, Iowa (USGS streamgage 05418720), were accessed from the U.S. Army Corps of Engineer's Upper Mississippi River Restoration Long Term Resource Monitoring element (Soballe and Fischer, 2004; U.S. Geological Survey, 2019b). Sample data for Maquoketa River at Maquoketa, Iowa (USGS streamgage 05418500), and Maquoketa River near Spragueville, Iowa (USGS streamgage 05418600), were accessed from the NWIS database (U.S. Geological Survey, 2019a).

Streamflow anomalies decompose the time-series variable (streamflow) into deviation from the mean for user-selected time steps. High streamflow anomalies indicate high streamflow for the (user-specified) time step, after sequentially accounting for anomalies at longer time steps. For this analysis, anomalies were computed for a 30-day time step, and the remainder term (high-frequency variability, HFV) was used to assess periods of streamflow that were higher than the preceding 30 days. Because the event end-member is based on stream samples rather than precipitation, this end-member more closely represents surface overland flow rather than precipitation, Unlike precipitation, overland flow will have some contact time with the soils in the drainage basin before entering streams or shallow groundwater. Groundwater samples from the well most upgradient from the stream, based on water-level data, were used to describe the base groundwater end-member and were compared with stream base flow samples (based on low HFV streamflow anomaly).

Processes on the Reconnected Floodplain

During the 5-year duration of this study, the Maquoketa River flooded onto the floodplain through the upstream levee breach five times, in June–July 2014, March 2015, January 2017, February 2018, and September 2018. The 2014 event persisted above the level of the upper breach for 3 days, but other flooding events above the level of the upper breach were less than 24 hours in duration. Events in March 2015, January 2017, and February 2018 resulted from ice dams behind the Highway 52 bridge and railroad trestle that caused the river water to back up until the water level was high enough to enter the upper levee breach and onto the floodplain. For the March 2015 event, the soil had thawed (soil temperature was above 0 degrees Celsius [°C]),

which permitted soil sampling. During the March 2015 flood, receding waters left quantifiable fine silt and sand deposited throughout the study area on the clay marker horizons.

River Concentrations and Loads

Concentrations of TSS, nitrate, and phosphorus are summarized for years 2014–2018 and for flood-events during the study (table 2; fig. 4). Summarized concentrations for TSS and phosphorus are based on primary turbidity-surrogate models; alternate streamflow-based models are used for filling gaps in loads only. The primary turbidity-TSS model had an adjusted R^2 value of 0.77 and a root mean square error (RMSE) value of 0.521. The alternate TSS model was based on 62 observations using streamflow and had an adjusted R^2 value of 0.51 and RMSE value of 0.754. The LOADEST nitrate model based on 167 observations had an adjusted R^2 value of 0.922 and RMSE value of 0.2343 (Garrett, 2019). The primary total phosphorus model based on turbidity and streamflow had an adjusted R^2 value of 0.91 and RMSE value of 0.215. The alternate total phosphorus model using streamflow had an adjusted R^2 value of 0.46 and RMSE value of 0.499 (Garrett, 2019). Daily mean river concentrations based on sensor readings or surrogate-based values ranged from 5.92 to 2,350 mg/L for TSS, 3.0 to 15.6 mg/L nitrate as nitrogen, and 0.0258 to 1.66 mg/L as phosphorus. LOADEST models, which are fitted to loads, are not directly comparable with models fitted directly to concentration. Streamflow-based models for TSS and total phosphorus were fitted to concentration, whereas the nitrate streamflow-based model was a load model fitted using LOADEST.

Floodplain Removal of Material During Floods

We estimate that during the March 2015 flood, the net deposition over the upper and middle areas of the study site was 298,000 kilograms (kg) of sediment (3,230 kg/ha), 4,830 kg carbon (52 kg/ha as C), 486 kg nitrogen (5 kg/ha as N), and 100 kg phosphorus (1 kg/ha as P). Accretion of material was highest adjacent to the upper levee breach (fig. 5). The 1.1-ha area by the lower levee breach had substantial intrasite reworking of the sediments. Four of the clay pads were eroded and may have been deposited on adjacent pads. Gross sedimentation rates were extremely high (fig. 5) with 402,000 kg sediment deposited in the 1.1-ha area. Gross deposition of sediment-bound materials by the lower breach were 7,180 kg carbon, 676 kg nitrogen, and 410 kg phosphorus.

The suspended solids load in the Maquoketa River during the 2015 water year (October 2014 to September 2015) was 273,000 megagrams (table 2). Of that, 700 megagrams of sediment was deposited on the floodplain—or 0.26 percent of the annual suspended solids load or about 12.7 percent of the 2-day flood load. The 2015 water year Maquoketa River nitrate load was 8,020 megagrams as nitrogen. Event-deposited nitrogen totaled 1.16 megagram—or

Table 2. Summary of river suspended solids, nitrate, and phosphorus concentrations and loads.

[mg/L, milligrams per liter; Mg/d, megagrams per day; Mg, megagrams; --, not applicable]

	Suspended solids				Nitrate				Phosphorus			
	Concentration		Load		Concentration		Load		Concentration		Load	
Time period	Mean, mg/L	Days, percent	Mean, Mg/d	Total, Mg	Mean, mg/L	Days, percent	Mean, Mg/d	Total, Mg	Mean, mg/L	Days, percent	Mean, Mg/d	Total, Mg
Annual summaries, for days of available data												
October 2013–September 2014	84.5	11.5	1,490	495,000	6.45	34.5	32.6	11,900.0	0.235	11.5	1.29	472.00
October 2014–September 2015	159.0	54.8	823	273,000	6.20	58.4	22.0	8,020.0	0.273	54.5	1.23	450.00
October 2015–September 2016	217.0	57.1	1,210	401,000	7.53	62.8	42.4	15,500.0	0.286	57.1	2.47	904.00
October 2016–September 2017	159.0	64.4	1,000	332,000	8.05	69.3	39.4	14,400.0	0.205	64.4	1.50	549.00
October 2017–September 2018	193.0	61.1	1,830	605,000	7.51	61.1	42.3	15,500.0	0.076	15.6	3.35	1,220.00
Selected flood periods												
June 30–July 09, 2014	--	--	39,100	391,000	11.40	80.0	272.0	2,720.0	--	--	8.30	83.00
March 11–12, 2015	--	--	2,760	5,530	--	--	64.1	128.0	--	--	6.78	13.60
January 17–18, 2017	--	--	376	753	--	--	33.2	66.3	--	--	1.16	2.33
February 20–22, 2018	--	--	23,600	70,900	--	--	164.0	491.0	--	--	2.00	6.00
September 03–08, 2018	121.0	100.0	34,700	208,000	5.80	100.0	205.0	1,230.0	--	--	64.30	386.00

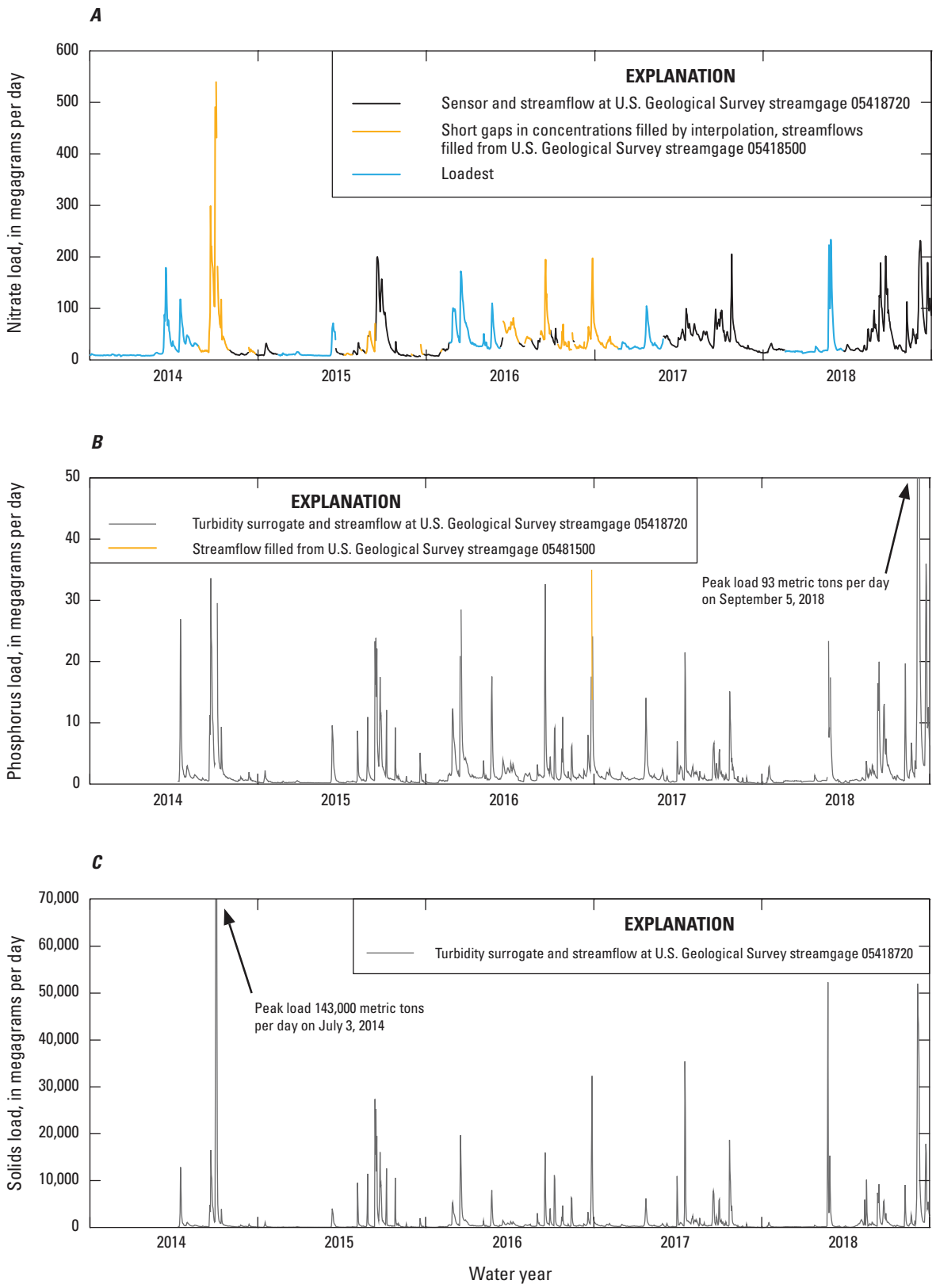


Figure 4. Observed and predicted nitrate, phosphorus, and suspended solids load during the years 2013–2018, measured at the U.S. Geological Survey streamgage 0518720 in the Maquoketa River at Green Island, Iowa.

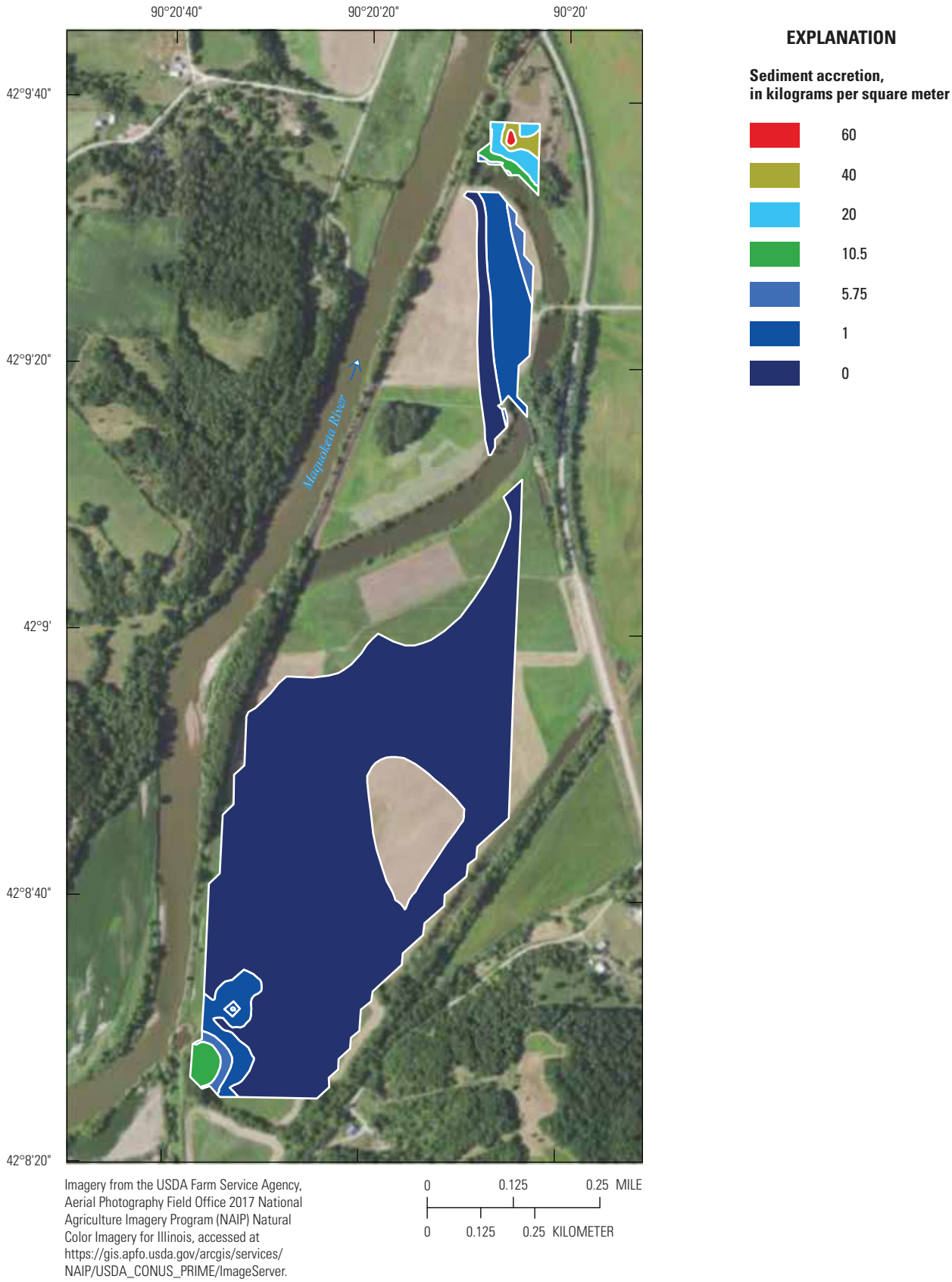


Figure 5. Spatial distribution of postflood sediment accretion on the Maquoketa River floodplain at Green Island, Iowa.

0.014 percent relative to the annual nitrate load or about 0.91 percent relative to the 2-day flood nitrate load. The 2015 water year Maquoketa River phosphorus load was 450 megagrams. Of that, 0.51 megagram phosphorus was deposited on the floodplain—or 0.11 percent of the annual phosphorus load or about 3.8 percent of the 2-day flood load.

Surficial Denitrification

Ambient rates of denitrification (October 2014 – June 2016) had a mean of 1.64 ± 0.24 SE micrograms as nitrogen per square centimeter per hour [µg N/cm^2]/h. DEA rates of denitrification (October 2014 – June 2016) had a mean of 5.06 ± 0.48 SE [µg N/cm^2]/h. Rates of DEA were positively correlated with percentage of soil nitrogen (Spearman rank order correlation, $r_s = 0.579$, $p < 0.0001$) and percentage of soil carbon ($r_s = 0.611$, $p < 0.0001$), indicating that DEN rate is potentially limited by nitrogen and carbon. The sediments underlying the nitrate-rich Maquoketa River (mean 7.09 ± 0.83 SE mg/L NO_3-N) exhibited mean ambient denitrification of 4.59 ± 1.43 SE) [µg N/cm^2]/h (fig. 6B) and potential denitrification with a mean of 6.57 ± 2.98 SE) [µg N/cm^2]/h (fig. 6D). Seasonal effects (fig. 6C) were evident such that rates of DEA were slightly higher and more variable in warmer months (October, May, and June) than in early spring (March). Ambient rates of denitrification were similar throughout the sampling periods (fig. 6A). Lowest ambient rates were measured in the wet sediments of the floodplain (fig. 6B). We estimated that nitrogen losses on the entire study site because of denitrification ranged from 250 kg/d as nitrogen in March 2015 to 668 kg/d as nitrogen in October 2014. During warm seasons, nearly half of the nitrogen deposited during small floods could potentially be denitrified. Our study did not measure nitrate production via mineralization and nitrification, which could be substantial (De Jager and others, 2015). Wet sediments were rich in exNH_4^+-N, likely because of mineralization of organic nitrogen and anoxic conditions preventing nitrification (fig. 7). Meanwhile, dry floodplain soils had lower exNH_4^+-N and higher pwNO_3^- -N compared to wet sites likely because of mineralization followed by nitrification (fig. 7).

Soils on the dry floodplain after a long period without floods had detectable ambient denitrification rates with a mean of 0.55 ± 0.13 SE [µg N/cm^2]/h without addition of any water (fig. 8). Incubation with deionized water more than doubled denitrification rates with a mean of 1.26 ± 0.23 SE [µg N/cm^2]/h above dry soil rates. Addition of Maquoketa River water quadrupled dry soil denitrification rates with a mean of 2.55 ± 1.07 SE [µg N/cm^2]/h and was nearly three-quarters of the DEA rate with a mean of 3.52 ± 1.31 SE [µg N/cm^2]/h (fig. 8). Maquoketa River water contained 11.53 mg/L nitrate, nearly the same amount of nitrogen used in our DEA incubations (14.0 mg/L NO_3^--N). Incubation with excess nitrate and carbon (DEA) further increased denitrification to more than 6 [µg N/cm^2]/h at some sites.

Soil Phosphorus Characteristics

Soils of the floodplain were rich in phosphorus, with a mean of 365 ± 26.1 SE mg/kg as phosphorus and range of 56–1037 mg/kg as phosphorus. Mehlich-3 (M3P) was also relatively high, with a mean of 111.2 ± 7.4 SE mg/kg as phosphorus, which was much higher than the 36–50 mg/kg as phosphorus suggested as optimal for crop growth (Mallarino and others, 2013). Mean soil PSR was high 0.12 ± 0.01 SE and likely high enough to risk phosphorus loss from soils (Maguire and Sims 2002). EPC_0 is a useful indicator of a soil's ability to adsorb or desorb phosphorus depending on the phosphorus concentration of overlying floodwater. EPC_0 had a mean of 0.18 ± 0.03 SE mg/L as phosphorus and a range of 0.01–0.58 mg/L as phosphorus, indicating flood waters with SRP concentrations less than 0.18 mg/L as phosphorus would result in phosphorus flux from soils to overlying water to sediments (James and Barko, 2004). Soil EPC_0 was strongly related to variation in soil moisture ($r_s = -0.619$, $p = 0.0016$; fig. 9) and soil iron concentrations ($r_s = -0.531$, $p = 0.0084$; fig. 10). Highest soil EPC_0 were associated with lower iron concentrations, whereas iron concentrations greater than approximately 500 mg/kg as iron resulted in very low EPC_0 (less than [<] 0.1 mg/L as P). This pattern likely derives from natural clay particles with a surficial armoring of reactive iron and aluminum hydroxides, which have a high capacity for absorbing phosphate and for maintaining low EPC_0 in solution (Froelich, 1988). Such iron-driven phosphorus dynamics are highly dependent on the soil reduction-oxidation environment, which will vary with soil moisture.

Concentrations of SRP lower than 0.18 mg/L as phosphorus in river water would likely result in a loss of phosphorus from soils into flood waters. Flood waters on the Maquoketa River are documented to carry phosphorus concentrations higher than 0.18 mg/L as phosphorus (ortho-P of 0.17 mg/L as P and TP of 1.10 mg/L as P; Hubbard and others, 2011). Mean annual concentrations (1979–2007) are much lower (ortho-P: 0.091 mg/L as P; TP: 0.296 mg/L as P; Hubbard and others, 2011). Hence flood water with high concentrations of SRP would likely lose phosphorus to floodplain soils through two processes: (1) soil sorption dynamics and (2) deposition of phosphorus-rich fine transported particles.

Groundwater

Hydrologic gradients in the floodplain study area were typically lateral toward the Maquoketa River in the upstream two-thirds of the study area, transitioning near wells W4, W5, and W6 to a gradient parallel to the Maquoketa River and flowing toward the Mississippi River. Hydrologic gradients in the upstream two-thirds of the floodplain aquifer were low (mean of 0.0017) during base-flow periods (relative to the Maquoketa River), and the downstream part of the floodplain aquifer had a mean hydrologic gradient

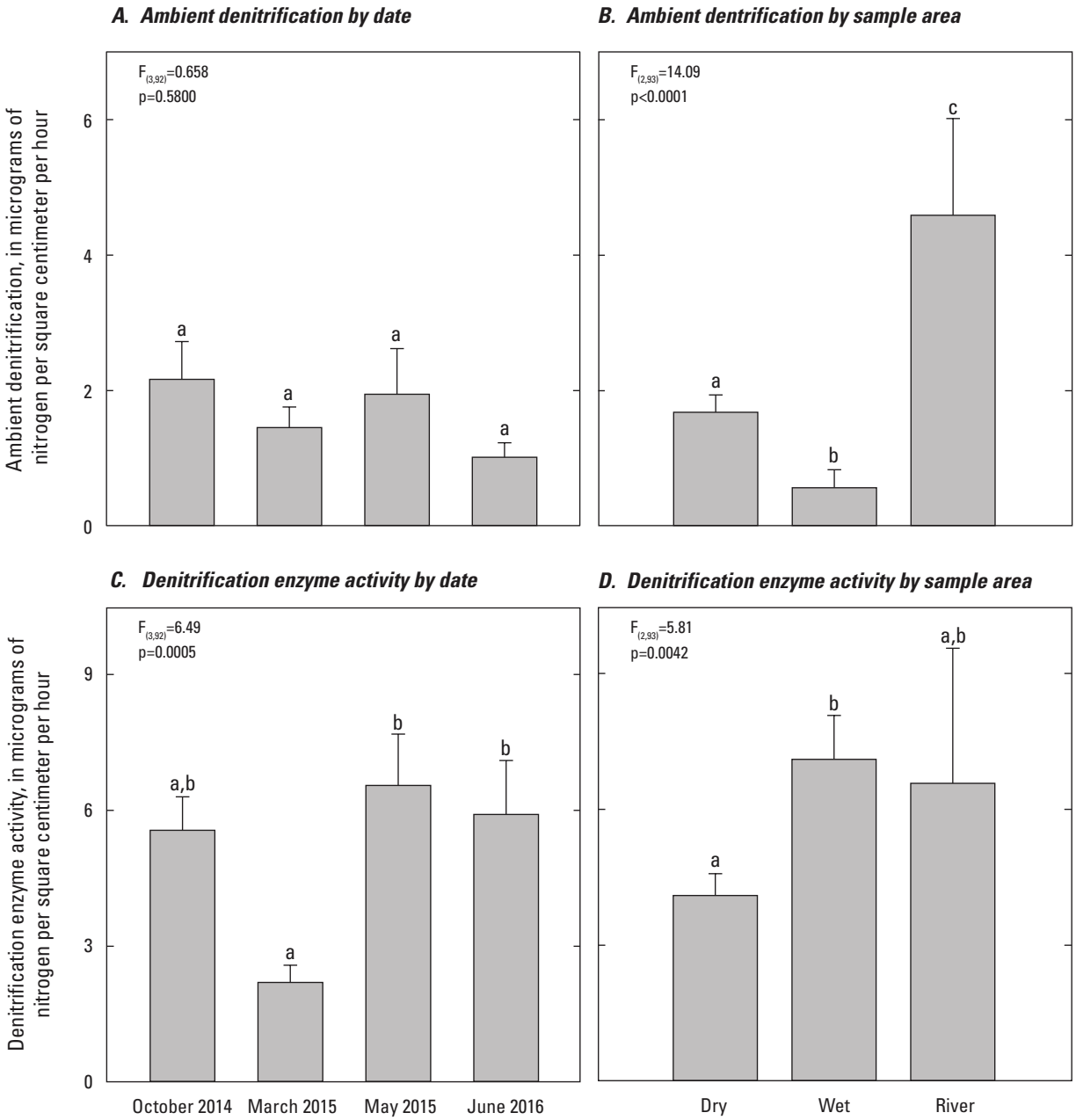


Figure 6. Rates of denitrification in micrograms as nitrogen per square centimeter per hour (mean + 1 standard error) on the Maquoketa River floodplain at Green Island, Iowa: *A,* ambient denitrification by date; *B,* ambient denitrification by sample area where dry and wet sites were on the floodplain in dry and wet areas, respectively, and river sites were in the Maquoketa River; *C,* denitrification enzyme activity by date; *D,* denitrification enzyme activity by sample area. F- and p-values refer to analysis of variance tests by dates (*A*, *C*) and sample area (*B*, *D*). Different letters indicate significant differences (α=0.05) among sample area or date identified during Tukey Honest Significant Difference (HSD) tests.

of 0.0029 during base-flow periods. Base-flow periods were defined by days with HFV streamflow anomaly less than –0.5. Short-duration reversals were observed during streamflow events, with water levels in the river and adjacent wells higher than in other nearby wells. Even maxima hydraulic gradients, however, were not particularly high, with a mean maxima for the upstream and downstream parts of the aquifer of 0.0054 and 0.0086, respectively.

Results of the discrete sampling varied across the floodplain and indicated three distinct groupings of wells based on groundwater chemistry: (1) groundwater that was similar to the chemistry of the river water (downstream floodplain wells [W2, W3, W4]); (2) groundwater chemistry indicative of the water residing in the floodplain (central floodplain wells [W5, W6, W7, W8, W9]); and (3) groundwater presumed to be more characteristic of the groundwater coming from the upland slopes (upstream floodplain wells [W10, W11, W12] table 3; fig. 2). In particular, specific conductance was statistically different for the three floodplain groupings (multiple pairwise comparison by Tukey HSD had p-values less than 0.05 for wells in different floodplain groupings, except W2 and W5, which were similar). Specific conductance in the three groupings ranged from 352 to 605 microsiemens per centimeter at 25 degrees Celsius (µS/cm at 25 °C), 250 to 379 µS/cm at 25 °C, and 592 to 850 µS/cm at 25 °C, respectively, for downstream, central, and upstream floodplain wells. Nitrate concentrations ranged from < 0.04 to 7.69 mg/L as nitrogen in the downstream floodplain wells, from < 0.04 to 12.9 mg/L as nitrogen in central floodplain wells, and from < 0.04 to 3.48 mg/L as nitrogen in upstream floodplain wells, with medians for each section, respectively, of 1.31, 0.506, and < 0.04 mg/L as nitrogen. Dissolved oxygen was low (< 1.0 mg/L) in all downstream and upstream floodplain well samples, but was as high as 3.2 mg/L in samples from the central floodplain wells.

Groundwater Mixing Model

Two characteristic sources are described for a simple groundwater mixing model with end-members for precipitation event water and base groundwater. Stream water samples were classified based on HFV streamflow anomaly with stream event and base-flow breakpoints chosen graphically and to include at least 20 samples in each group: event samples have an HVF greater than 1.35 and the base groundwater end-member (from W12) is compared to stream base-flow samples with an HFV less than –0.5 (fig. 11). The mean specific conductance for stream event samples was 271.7 ± 15.0 SE µS/cm at 25 °C. W12 had a mean specific conductance of 768.6 ± 25.2 SE µS/cm at 25 °C, which was higher than the mean stream base-flow specific conductance of 600.7 ± 6.8 SE µS/cm at 25 °C.

Apportioning between precipitation event water and base groundwater, based on specific conductance data, groundwater samples ranged from –8 to 116 percent base groundwater, with values outside the range 0–100 percent a result of using mean conditions for the end-members. For example, mean specific conductance for W12 was used to define base groundwater specific conductance, but individual sample data resulted in a range of 87–116 percent base groundwater at this site. Other wells in the upstream floodplain section (wells W10 and W11) were dominated by base groundwater, with median samples at least 73 percent base groundwater. Samples from wells in the central floodplain section (wells W5 through W9) were dominated by precipitation event water, ranging from –3 to 12 percent base groundwater. Wells sampled in the downstream section of the floodplain (wells W2 through W4) were mixed, with 31–55 percent base groundwater in samples.

Applicability to Reducing Nutrient and Sediment Transport

Substantial efforts to restrict the flux of sediments, nitrogen, and phosphorus from agricultural landscapes have been met with mixed success (Arnold and others, 2014; Jones and others, 2018). Tributaries to the upper Mississippi River that drain some of the most fertile and productive agricultural systems in the world have exhibited small reductions in phosphorus loads, but small increases in nitrogen loads during the last 20 years (Sprague and others, 2011; Kreiling and Houser, 2016). Agricultural landscapes, including grassed waterways installed as part of nutrient management strategies, are becoming phosphorus saturated so they now leak legacy phosphorus into drainage water (Williamson and others, 2014). Further, as tile drainage acreage continues to increase, nutrient-rich water fluxes from agricultural fields to drainage ditches and is carried to leveed streams and rivers that are no longer fully connected with biogeochemically active floodplains. Lack of lateral connectivity is known to inhibit natural removal of transported sediments and nutrients (Covino, 2017; Wohl and others, 2018). Potential load reduction and improvement of downstream water quality via lateral floodplain connection and capture of flood-transported sediments and nutrients have been quantified in rivers draining agricultural and developing catchments of Europe and the eastern United States (Noe and Hupp, 2005; Kiedrzyński and others, 2008; Noe and Hupp, 2009; Surridge and others, 2012). Floodplain rivers draining the midwestern United States, with large nutrient and sediment loads, are not well studied (but see Schilling and others, 2015).

Here we quantify material deposited onto a reconnected floodplain, and the potential fate of inorganic nitrogen and phosphorus deposited there. This site is relatively unique because levees were purposely kept unrepaired by the managing agency (U.S. Army Corps of Engineers) after repeated breaching as part of a strategy to reduce downstream flooding and limit construction costs of repeated levee repairs. The U.S. Army Corps of Engineers evaluates cost/benefit ratios of such repairs, and currently, water-quality benefits and other ecosystem service benefits do not weigh

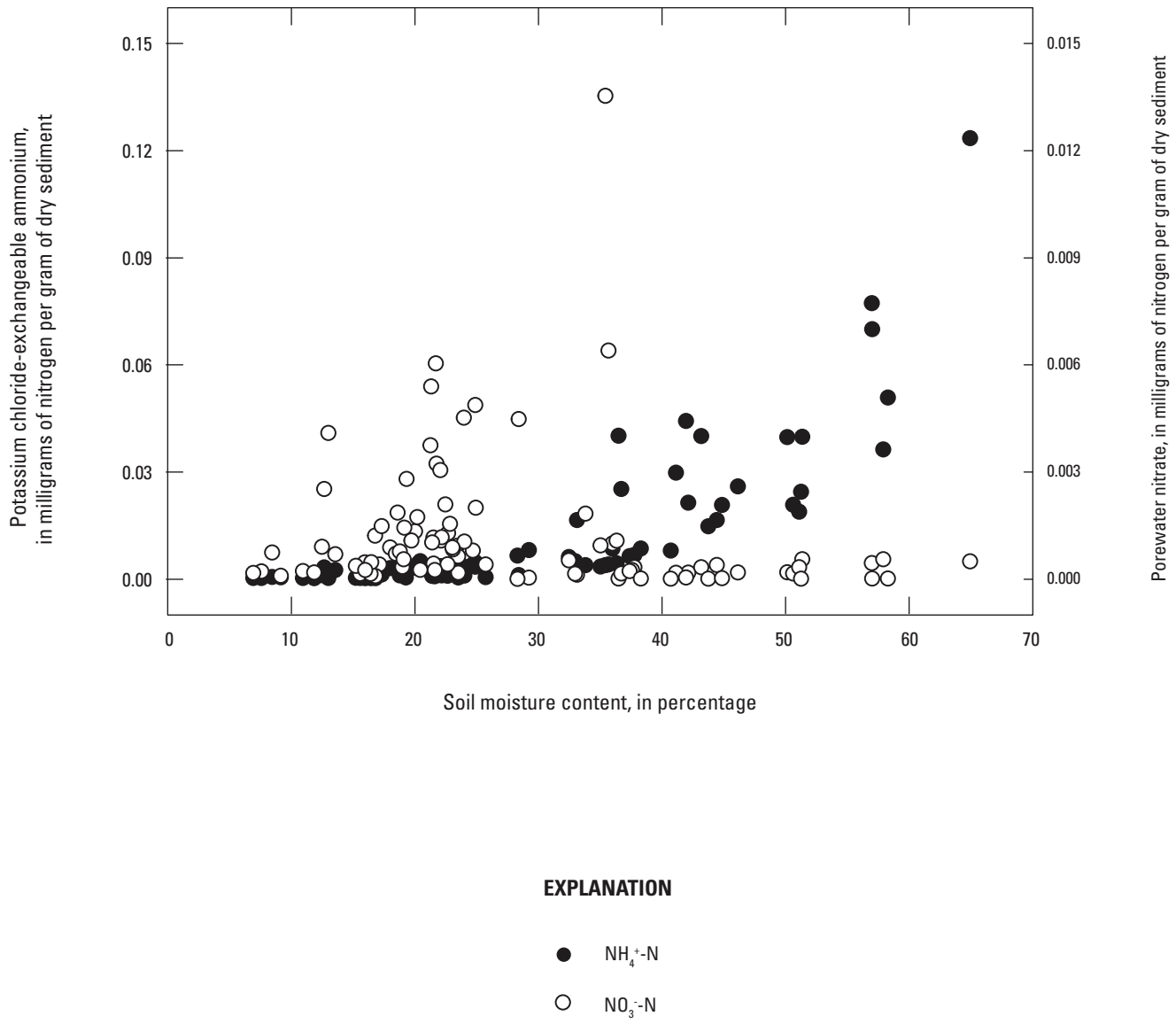


Figure 7. Plot of relation between potassium chloride-exchangeable ammonium nitrogen and soil moisture and between porewater nitrate nitrogen and soil moisture.

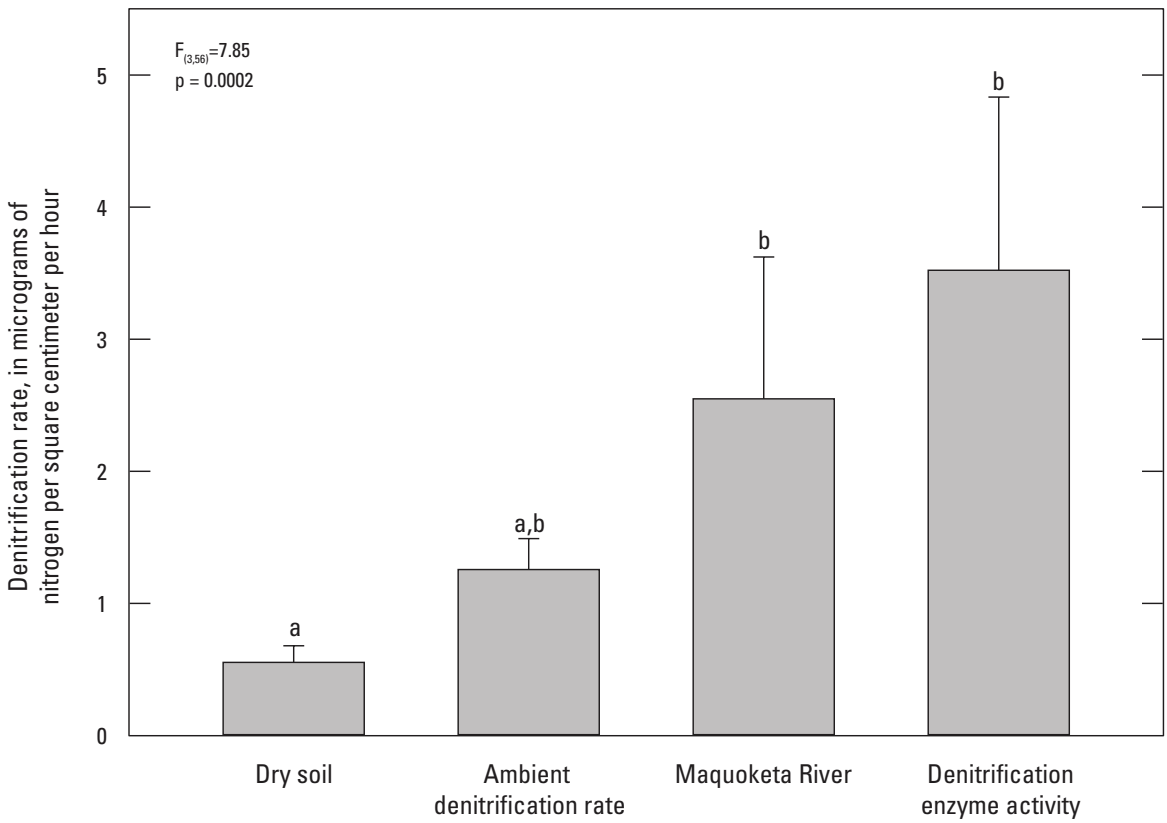


Figure 8. Rates of denitrification (mean + 1 standard error) from the interflood period during June 2016 on the dry sites on the Maquoketa River floodplain at Green Island, Iowa. The first bar shows denitrification in dry soils with no liquid added. The second bar shows ambient denitrification (DEN) from all sites. The third bar shows denitrification with Maquoketa River water amendments (11.53 nitrate nitrogen per liter NO_3^--N mg/L). The fourth bar shows denitrification enzyme activity (DEA) from all sites. Different letters indicate significant differences (α=0.05) among soil treatment means with the Tukey Honest Significant Difference (HSD) tests.

heavily in these decisions (Hintz, 2011). Information provided here may allow for inclusion of ecosystem services in the cost/benefit analysis of infrastructure repair (Opperman and others, 2009). Further, collaboration among State and Federal land management agencies at this site supported quantification of the environmental benefits accruing from the action not to repair the levee.

Subsequent flooding of the site resulted in the capture of sediment and nutrients, and promoted permanent removal of nitrogen by denitrification. As expected, rates of denitrification varied with season, soil organic matter, soil moisture, $pwNO_3^-$-N concentrations, and $exNH_4^+$-N concentrations. During interflood periods, potential denitrification had high rates. Ambient and potential denitrification on dry floodplain soils indicates resident inorganic nitrogen is accumulating and when conditions are favorable (such as, flood water and anoxic conditions) this nitrogen is rapidly denitrified. Comparison of potential denitrification across four dates indicates soil total nitrogen and total carbon are the best predictors of floodplain potential denitrification and that air temperature further affected rates (for example, warmer temperatures result in higher DEA rates). Relatively high concentrations of soil inorganic nitrogen also indicate both nitrification and mineralization were actively taking place on the floodplain (De Jager and others, 2015). Meanwhile, floodplain soil phosphorus flux equilibrium dynamics are controlled by a combination of sediment iron concentrations and water-filled pore space. The higher the percent water-filled pore space, the lower the likely phosphorus flux. Likewise, the higher the iron concentration, the lower the phosphorus flux.

These results indicate that during floods, the floodplain soils will rapidly denitrify and remove nitrate from flood waters, whereas the soils may release phosphorus into floodwaters depending on the concentration of SRP in overlying waters. Transported fine sediments likely will be deposited onto the floodplain, along with substantial quantities of sediment-bound phosphorus and carbon. These initial estimates of nitrogen, phosphorus, and sediment

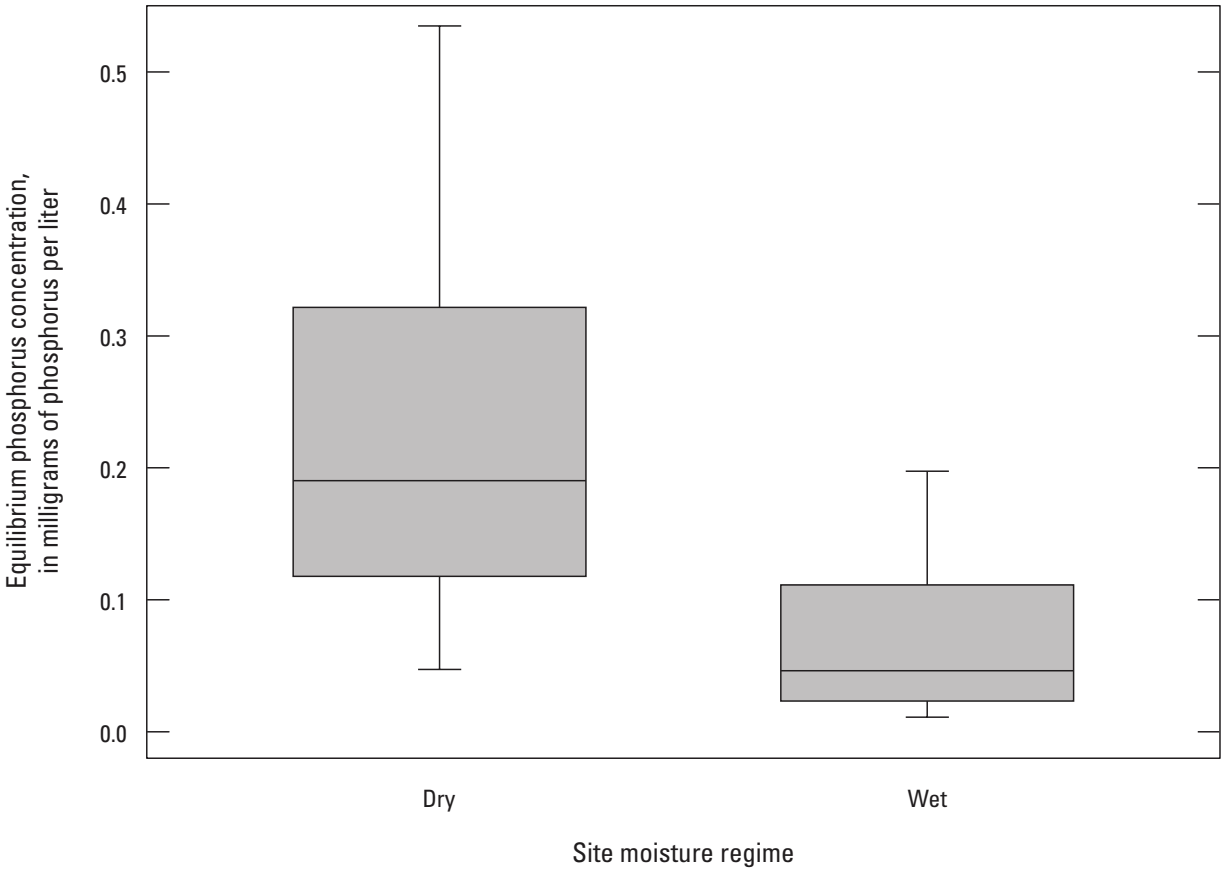


Figure 9. Median equilibrium phosphorus concentrations by site moisture regime (dry or wet) on the Maquoketa River floodplain and in the Maquoketa River at Green Island, Iowa. Boxes are bound by the 25th and 75th percentiles and the whiskers represent the 10th and 90th percentiles of rates, midline in percentile box is the median.

removal by fully connected floodplains are estimates that would benefit from additional data collections and analysis. Estimates are for removal of nitrate because the movement of solutes are likely more evenly distributed across a floodplain during floods relative to particulate material (Keizer and others, 2018). Particulate materials tend to deposit close to the channel border and in low spots on the floodplain (Asselman and Middelkoop, 1995; Walling and He, 1998; Keizer and others, 2018).

In addition to increased sediment and nutrient retention and processing on the surface of the floodplain, the increased connection between the stream and floodplain likely promotes nutrient processing in shallow groundwater under the floodplain. Low-oxygen and nitrate conditions in well samples indicate the aquifer has additional capacity for denitrification. Although short-term reversals (in the order of days to weeks) in groundwater gradients may drive nutrient-laden stream water into bank storage in a limited area of the floodplain, increased surface inundation (and infiltration) from the levee breaches provides a more direct pathway for nutrient delivery. The mixing model based on specific conductance was unable to distinguish the specific source of water in the floodplain aquifer, between the stream water during flooding events or direct precipitation on the floodplain and infiltration. Hydraulic gradients indicate infiltration is the more likely pathway and source of the floodplain aquifer's water.

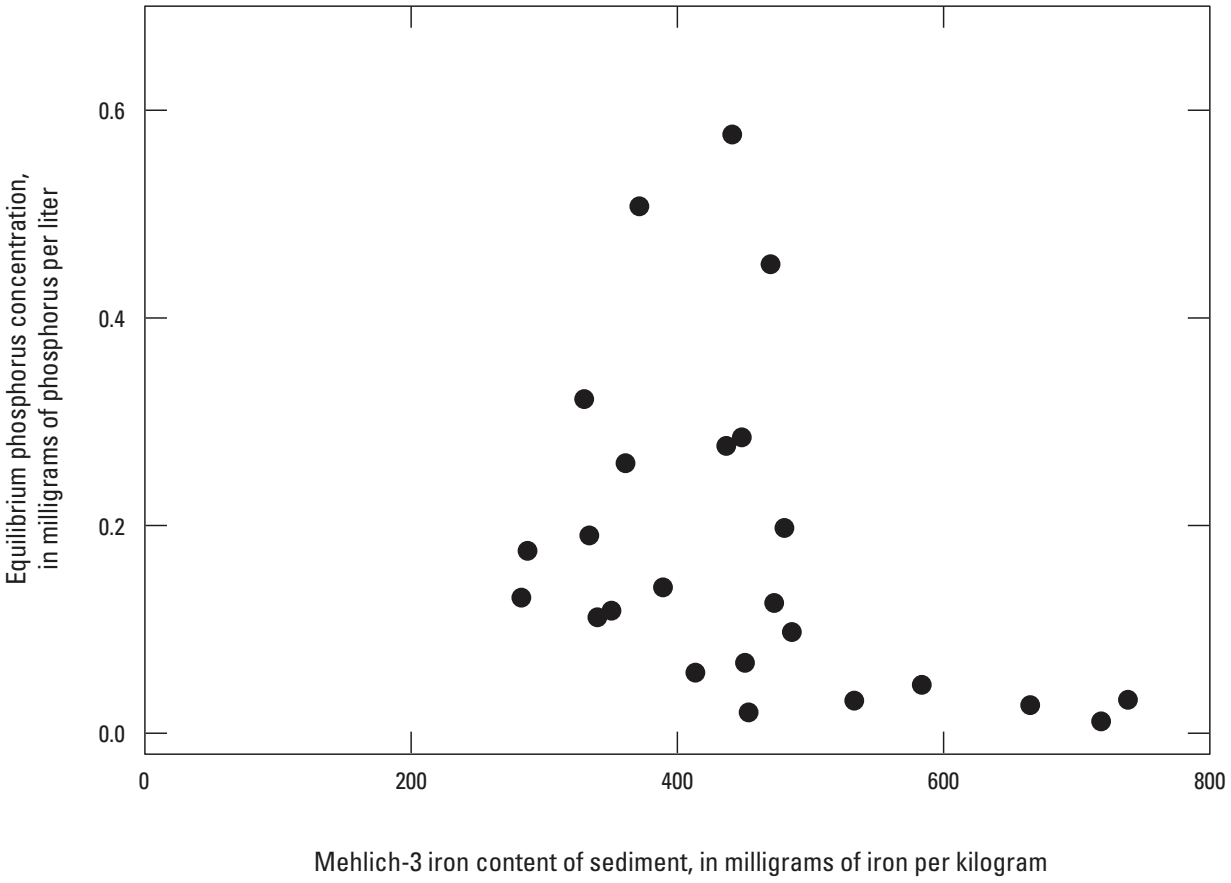


Figure 10. Soil equilibrium phosphorus concentrations compared to Mehlich-3 iron concentration of sediment at sites on the Maquoketa River floodplain and in the Maquoketa River at Green Island, Iowa.

Table 3. Summary of Maquoketa River and groundwater sample results, Green Island, Iowa, 2013–2018.

[USGS, U.S. Geological Survey; µS/cm, microsiemens per centimeter; mg/L, milligrams per liter; <, less than, --, not available]

		Specific conductance, µS/cm at 25 degrees Celsius		Dissolved oxygen, mg/L		Orthophosphate, mg/L as phosphorus		Nitrate plus nitrite, mg/L as nitrogen		Nitrite, mg/L as nitrogen		Ammonia, mg/L as nitrogen	
Site	**USGS site number**	**Minimum**	**Maximum**	**Minimum**	**Maximum**	**Minimum**	**Maximum**	**Minimum**	**Maximum**	**Minimum**	**Maximum**	**Minimum**	**Maximum**
Maquoketa River													
Maquoketa River	05418500, 05418600, 05418720	132	700	4.5	25.0	< 0.001	0.727	0.20	15.000	--	--	< 0.01	1.86
Maquoketa River	05418720 (study period only)	367	617	5.9	16.4	0.013	0.294	4.18	9.420	0.019	0.032	< 0.01	0.34
Downstream floodplain wells													
W2*	420931090200102	352	479	0.3	0.6	0.029	0.137	< 0.04	< 0.040	< 0.002	0.003	0.30	1.05
W3	420932090201003	389	605	0.3	0.6	0.038	0.053	1.31	7.000	0.028	0.082	< 0.01	0.01
W4	420924090200904	494	561	0.2	0.7	0.036	0.043	1.00	7.690	0.037	0.075	< 0.01	< 0.01
Central floodplain wells													
W5*	420919090201705	315	379	0.1	1.2	0.139	0.211	< 0.04	0.429	< 0.001	0.002	0.03	0.05
W6	420912090200406	233	369	0.3	2.0	0.046	0.073	< 0.04	9.570	< 0.001	< 0.001	< 0.01	0.01
W7	420902090202007	328	368	0.1	0.6	0.058	0.080	1.70	8.220	0.015	0.022	< 0.01	< 0.01
W8	420850090201608	250	348	1.2	3.2	0.098	0.125	2.59	12.900	< 0.001	0.009	< 0.01	< 0.01
W9	420855090195709	268	334	0.3	3.0	0.026	0.299	< 0.04	0.197	< 0.001	0.003	0.05	1.22
Upstream floodplain wells													
W10	420836090201910	592	737	0.1	0.6	0.054	0.770	< 0.04	0.059	0.001	0.009	0.33	1.06
W11	420842090203311	607	718	0.2	0.7	0.030	0.040	< 0.04	3.480	< 0.001	0.111	< 0.01	0.17
W12	420824090202812	705	850	0.2	0.5	0.094	0.514	< 0.04	< 0.040	0.003	0.006	0.09	0.26

*Floodplain well groupings based on specific conductance are statistically different (Tukey's Honest Significant Difference (HSD), $p < 0.05$), except for W2 and W5, which are similar.

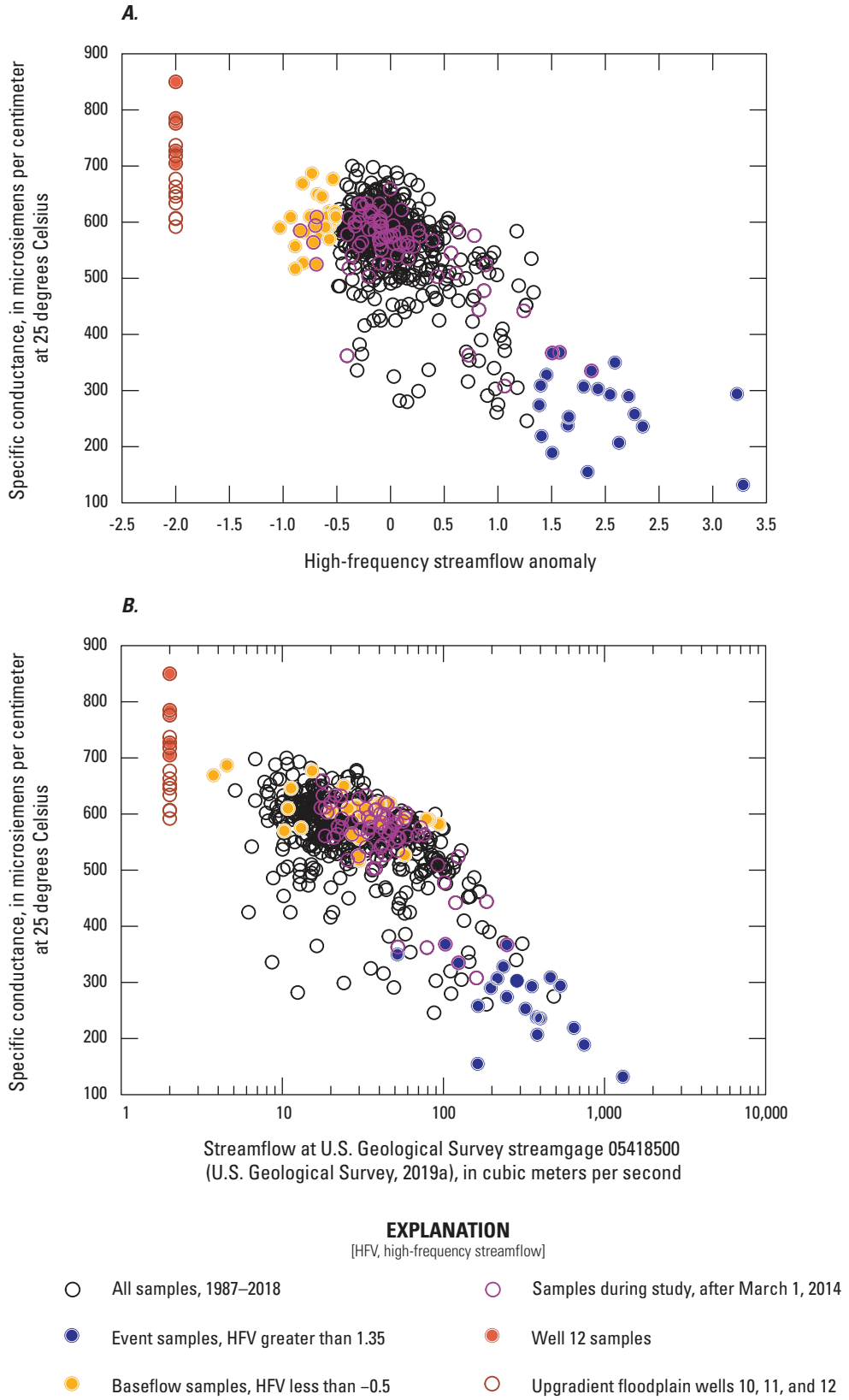


Figure 11. Event and base groundwater mixing end-members. Sample specific conductance in relation to, *A*, high-frequency streamflow anomaly provided more distinct event and base-flow end members than, *B*, streamflow.

Summary and Conclusions

The Maquoketa River transports some of the highest loads of sediment and nutrients in Iowa. In 2010, the river was reconnected to 151 hectares of its floodplain when a levee was breached in two locations. The formerly agricultural floodplain area is currently being managed for wildlife. The renewed connectivity between the river and the floodplain provides additional water-quality benefit to the riverine ecosystem. During floods, sediment and nutrients can be deposited onto the floodplain and nitrogen can be permanently removed through denitrification. This study was done to quantify the effects of increased river-floodplain connectivity on water quality. Specifically, we quantified the retention of flood-transported sediment, carbon, nitrogen, and phosphorus on the Maquoketa floodplain; floodplain sediment denitrification and its effect on nitrate removal on the floodplain; factors influencing sediment phosphorus retention or release; and factors affecting shallow groundwater nitrogen and phosphorus processing under the floodplain.

For this study, soil samples were collected in the floodplain, and sensors that measured water level, nitrate, and turbidity were placed in the Maquoketa River immediately downstream from the floodplain. Soil samples were used to quantify floodplain sediment and nutrient retention potential, and sensors were used to quantify total suspended solids, nitrogen, and phosphorus concentrations and loads. Streamflow was monitored from April 2014 to September 2018. In addition, clay markers were placed throughout the floodplain to capture sediments during the floods. Soil samples were collected four times throughout the study (October 2014, March 2015, May 2015, and June 2016) to measure denitrification rates and once (June 2016) to quantify phosphorus retention potential.

The floodplain was inundated in March 2015 and January 2017. During the March 2015 flood, the net deposition over the upper and middle areas of the study site was 298,000 kilograms (kg) sediment (3,230 kg/ha), 4,830 kg carbon (52 kg/ha as C), 486 kg nitrogen (5 kg/ha as N), and 100 kg phosphorus (1 kg/ha as P). The 1.1-ha area by the lower levee breach had considerable intrasite reworking of sediment resulting in extremely high gross sedimentation rates. Gross deposition in the lower section was estimated at 402,000 kg sediment, 7,180 kg carbon, 676 kg nitrogen, and 410 kg phosphorus. Comparing the deposition measurements to the Maquoketa River loads indicated that 0.91 percent of the nitrogen load and 3.8 percent of the phosphorus load were deposited on the floodplain during the 2-day March 2015 flood.

Nitrogen losses on the floodplain because of denitrification ranged from 250 kilogram per day (kg/d) as nitrogen in March 2015 to 668 kg/d as nitrogen in October 2014. Denitrification rates varied with season, soil organic matter, soil moisture, and soil inorganic N concentrations. Ambient and potential denitrification rates on dry floodplain soils indicate that resident inorganic nitrogen is accumulating in the soil and when conditions are favorable (for example, flood water and anoxic conditions) this nitrogen is rapidly denitrified.

Floodplain soils are rich in phosphorus but still have the capacity to remove phosphorus from flood waters of the Maquoketa River. When flooded, soils can absorb phosphorus from the water column if dissolved phosphorus concentrations are higher than 0.18 milligram per liter (mg/L) as phosphorus in the river water. Measured phosphorus concentrations of flood waters on the Maquoketa River are higher than 0.18 mg/L as phosphorus. Thus, the floodplain is capable of retaining phosphorus by deposition of phosphorus-rich sediment and through soil adsorption.

Groundwater hydraulic gradients indicated groundwater flows typically toward the Maquoketa River in the upstream (relative to the Maquoketa River) two-thirds of the study area or toward the Mississippi River in the lower one-third of the sturdy area. Gradients were low, with a base-flow mean of 0.0017 or 0.0029 in the upstream two-thirds and downstream one-third of the floodplain aquifer, respectively. Reversals in hydraulic gradients were observed during stream events, but even maxima hydraulic gradients were only 0.0054 for the upstream part and 0.0086 for the downstream part of the floodplain aquifer. Groundwater chemistry indicates the aquifer is reducing, with low dissolved oxygen (less than 1 mg/L) and low nitrate (below laboratory detection of 0.04 mg/L as NO_3-N) concentrations for many samples. A simple mixing model based on specific conductance indicates that water in much of the floodplain aquifer is primarily from precipitation events. Samples from the downstream most wells were a mix of base groundwater and precipitation event water, with 31–55 percent base groundwater in wells W2–W4. Samples from the central part of the floodplain were predominately precipitation event water, with as much as only 12 percent base groundwater in W5–W9 samples. Samples from the upstream most wells were dominated by base groundwater, with at least 73 percent base groundwater in W10–W12. Although the mixing model was unable to distinguish between the river (during streamflow events) and direct precipitation as the source of water in the floodplain aquifer, hydraulic gradients indicate infiltration is the more likely pathway.

This study infers that restoration of even small river-floodplain connections can result in removal of relatively large quantities of sediment, nitrogen, phosphorus, and carbon. Loads of transported sediments and nutrients are substantial in the midwestern United States. Thus, alternate management strategies like restoring river-floodplain connectivity could provide additional water-quality benefit at relatively low long-term costs. High rates of denitrification in the Maquoketa River floodplain indicate that much of the nitrogen deposited during floods or cycled internally in the soil may potentially be permanently removed. The nutrient removal benefits documented here may be important in future decisions regarding floodplain restoration especially for management agencies tasked with levee repair.

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