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Potential for managing pool levels in a flood-control reservoir to increase nitrate-nitrogen load reductions

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Abstract

Few strategies are available to reduce nitrate-nitrogen (NO_3-N) loads at larger landscape scales, but flood control reservoirs are known to reduce riverine loads. In this study, we evaluated the potential to increase nitrogen (N) loss at Lake Red Rock, a large reservoir located in central Iowa, by evaluating the inundation of sediments deposited at the reservoir inflow. Sediment samples were collected at 51 locations in the lower delta region and analyzed for particle size and nutrient content. Nitrogen loss rates in delta sediments were determined from laboratory assays, and satellite imagery was used to develop a rating curve to quantify land area inundated within the delta. The daily mass of NO_3-N reduced with delta inundation was estimated by applying the mean N 24-h loss rate (0.66 g N m^2 day^{-1}) by the area of inundation (m^2). Results indicated that raising pool elevations to inundate more of the delta would result in greater N losses, ranging from 2 to 377 Mg per year. Potential N loss of 102 Mg achieved by increasing pool stage by 0.5 m would be equivalent to installing nearly 650 edge-of-field practices in the watershed. Although more work is needed to integrate with an existing environmental pool management plan, study results indicate that reservoir management could achieve N reductions at a novel landscape scale.

Plain Language Summary

Water pool levels behind large Midwestern reservoirs can be managed to inundate delta sediments and achieve large nitrate load reductions.

1 | INTRODUCTION

Nitrate-nitrogen (NO_3-N) export from the Corn Belt of the U.S. Midwest continues to impair local and regional ecosystem health (Dodds & Welch, 2000; McLellan et al., 2015; Turner et al., 2008) and threatens drinking water supplies that utilize rivers as source water (Jones et al., 2020; Schilling &

Abbreviations: TN, total nitrogen; USACE, U.S. Army Corps of Engineers; WRTDS, Weighted Regressions on Time, Discharge, and Season.

Wolter, 2009). Although new conservation practices such as blind inlets (Smith & Livingston, 2013), prairie filter strips (Zhou et al., 2014), multipurpose oxbows (Schilling et al., 2019), and saturated buffers (Jaynes & Isenhart, 2014; 2019) are being developed for field scale deployment, there are few strategies available to reduce NO_3-N loads at larger landscape scales. Nitrogen (N) loss rates in rivers and streams decline rapidly with increasing channel size (Alexander et al., 2000), and there has been less attention given to nutrient processing that may occur within the river systems themselves (Jones et al., 2018).

Within many Midwestern river systems, large reservoirs were constructed by the US Army Corps of Engineers (USACE) for the primary purposes of flood risk reduction, hydropower, recreation, and water supply (USACE, 2020). Many of these reservoirs, with loads drained from highly productive agricultural landscapes, receive input flows containing high NO_3-N concentrations. Although not constructed for water quality benefits, mass balance studies of several Midwest reservoirs have shown NO_3-N reductions ranging from 4.9% (Stenback et al., 2014) to 58% (David et al., 2006). Schilling et al. (2023) utilized a novel 42-year record to report that Lake Red Rock in south-central Iowa is reducing inflow NO_3-N loads in the Des Moines River by a long-term average of 12.4% per year. Annual NO_3-N reductions in Midwestern flood-control reservoirs were found to be related to water residence time (Schilling et al., 2023).

Less documented is the potential influence of reservoir deltas on riverine NO_3-N load reduction. There have been numerous studies describing the sedimentology and geomorphology of coastal and marine delta environments (Choi et al., 2004; Day et al., 2007; Innocent & Pranzini, 1993; Twilley et al., 2019, 2016), and these settings are considered hot spots for processing surface water NO_3-N (Sendrowski et al., 2021). Variations in hydrological and biogeochemical variables (Sendrowski et al., 2021), ecogeomorphology (Knights et al., 2020), channel-island connectivity (Hiatt et al., 2018), and organic matter content (Li et al., 2020) are among many factors found to contribute to spatial and temporal dynamics of coastal delta NO_3-N reduction. However, unlike reservoir deltas, coastal deltas are influenced by tides, salinity, and vegetation unique to their local environments (White et al., 2019), and many are heavily leveed (Day et al., 2007). River deltas forming at the mouths of reservoirs draining agricultural watersheds receive regular inundation and sediment discharge from river inflows and pool levels are routinely managed. Most work on the relation of reservoir sediment to river NO_3-N concentrations has focused on denitrification at the sediment-water interface (David et al., 2006; Shaughnessy et al., 2019).

In this study, we focused on the delta that has formed at Lake Red Rock, a large flood-control reservoir on the Des Moines River in central Iowa located about 30 km downstream of Des Moines, IA (Figure 1). Upstream of the reservoir, both the Des Moines River and major tributary (Raccoon River) are utilized for regional water supply and impaired for NO_3-N (Jha et al., 2010; Schilling & Wolter, 2009), and downstream, the City of Ottumwa water utility is severely threatened by high NO_3-N levels (Jones et al., 2020). Although the reservoir was found to reduce NO_3-N loads in the Des Moines River (Schilling et al., 2023), the goal of this study was to assess the potential for achieving additional NO_3-N reductions by inundating delta sediments. Specifically, our objectives were to (1) evaluate the sedimentology and ecogeomorphology of the reservoir delta; (2) quantify the rate of NO_3-N loss within the upper 10 cm of delta sediments; (3) develop a rating curve to relate the degree of delta inundation to pool stage; and (4) quantify the additional mass of NO_3-N reduced when various delta areas are inundated, and contextualize this amount in terms of historical NO_3-N loads in the river. Study results provide a perspective on a potential nutrient reduction strategy that could be pursued at a regional landscape scale for NO_3-N reductions in the U.S. Midwest.

Core Ideas

- Reservoirs are known to reduce riverine nitrate loads.
- Nitrate reductions occur in fine-textured delta sediments.
- Managing reservoir pool levels to inundate delta sediments increases nitrate loss.
- Reservoir pool management could achieve large nitrogen reductions at a novel landscape scale.

2 | MATERIALS AND METHODS

2.1 | Site description

Constructed in 1969, Lake Red Rock is located on the Des Moines River in Marion County, south-central Iowa, approximately 230 km upstream from its confluence with the Mississippi River (Figure 1). The original normal conservation pool elevation was 220.98 m in 1969, but levels have since increased to 221.89 m in 1979 and 223.72 m in 1988 due to loss of water storage caused by sedimentation. Since 1991, the normal conservation pool has resided at 226.16 m (U.S. Army Corps of Engineers, 2011). The major inflow (approximately 95%) to the reservoir is the Des Moines River, although there are additional small rivers and creeks that discharge directly to the reservoir. Land use in the watershed is predominantly agricultural, consisting of approximately 67% row crops of corn and soybeans.

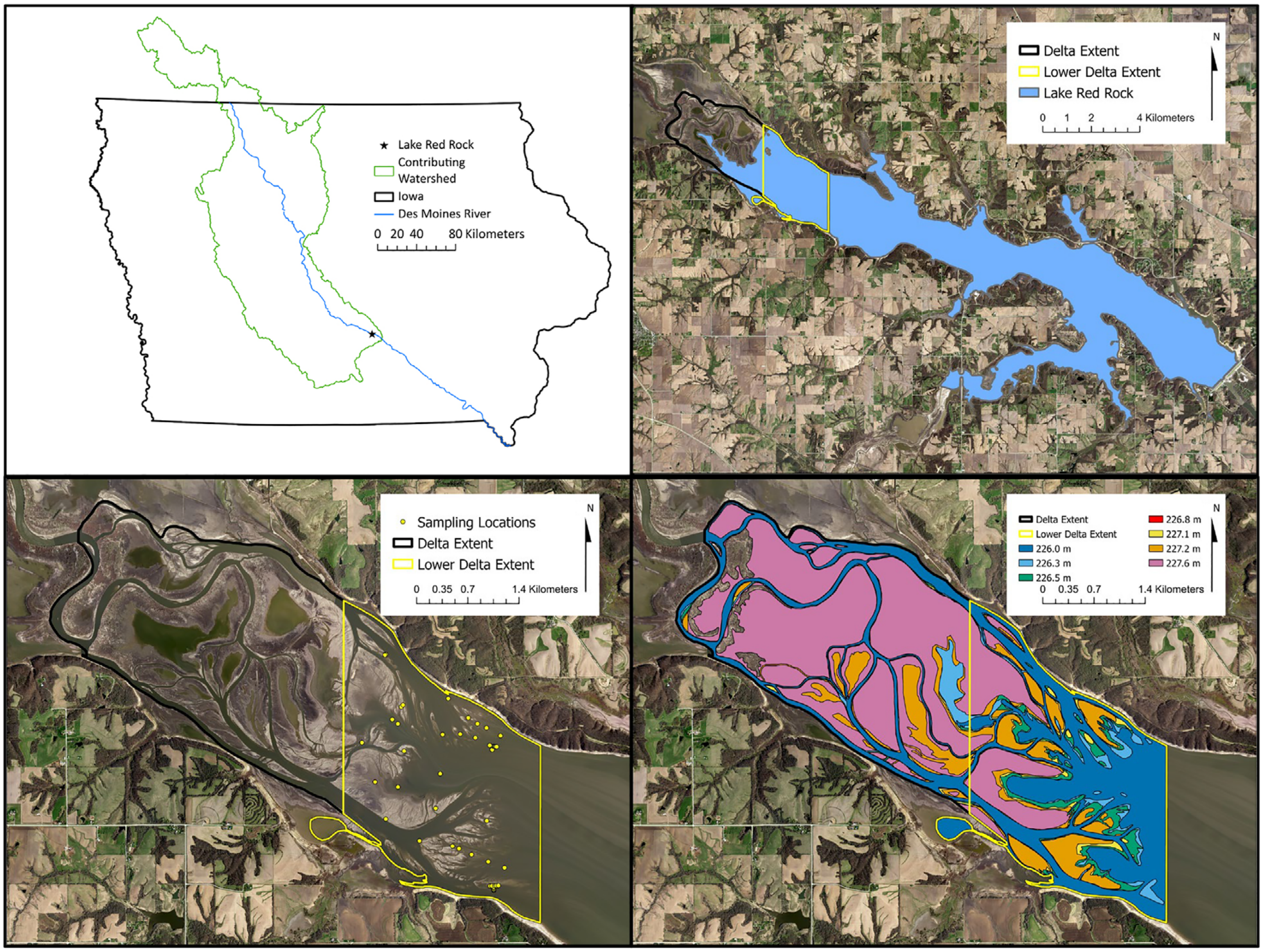

FIGURE 1 Top left: Location of Lake Red Rock within Des Moines River watershed; top right: Delineation of sediment delta and lower delta extent evaluated in this study; lower left: Sediment sampling locations within lower delta region; lower right: Lower delta inundation area associated with various reservoir pool levels.

Over a half-century since construction, a large 1718-ha delta formed where the Des Moines River flows into Lake Red Rock (Figure 1). In this study, we focused on a smaller subset of the delta that we spatially defined as the region from where the upstream Des Moines River channel braiding begins (93.2510311˚W) to the downstream point where delta sediments are always inundated (93.1753220˚W). We termed this 755-ha region the "lower delta," where changes in reservoir pool elevation cause regular inundation or exposure of delta sediments. The lower delta was defined as the area with a surface elevation between 225.8 and 227.2 m.

2.2 | Field investigation

In Fall 2022, field sampling was conducted to characterize upper sediments present within the lower delta. The study area was accessed when lake elevations were approximately 226.8 m or 0.6 m above normal conservation pool. At that level, most areas of the lower delta were accessible with a flat-bottom boat equipped with a mud propeller. A total of 51 sample locations were accessed across a range of topographic and vegetative conditions (Figure 1). Sample locations were sampled to assess delta topographic variations, including the active distributary channels, the delta plains, and the transition from the subaqueous to the exposed lower delta. Vegetation consisted of various emergent plants that grow at the interface of occasional water submergence and land, including four sites dominated by bidens (family *Asteraceae*), four in cocklebur (*Xanthium*), one in cottonwood (*Populus*), seven in smartweed (*Persicaria*), eight in wild millet (*Poaceae*), and four in willow (*Salix*).

At each sample location, surface sediments were collected to a depth of 10 cm using a hand auger. Field descriptions were recorded, and representative samples were collected for laboratory analyses of soil texture (Kimble et al., 1993). Total carbon and total nitrogen (TN) concentrations were determined following dry combustion by elemental analysis via chromatography (Costech ECS 4010 CHNSO Analyzer; Costech Analytical Tech Inc.).

2.3 | N loss measurements

To characterize the range of NO_3 removal capacity, soil cores were collected from 51 locations in the exposed delta or in areas of shallow water inundation. Locations corresponded with samples collected for assessment of soil physical characteristics. Cores were collected on September 15 and 29, 2022. Polycarbonate cylinders (5.1 cm inside diameter, 30.5 cm long) were pushed into the sediment to a depth of approximately 10 cm, the top closed with a rubber stopper, columns pulled out with the intact core of sediment, and the lower end stoppered. Upon return to the laboratory, approximately 300 mL of water collected from the reservoir at the time of core collection was added to each core. Air was introduced via a cannula inserted through a top rubber stopper to create gas bubbles close to the sediment-water interface in order to create some turbulence and simulate reservoir conditions. Cores were initially equilibrated for 8–12 h, allowing water to saturate the soil. After soil saturation, cores were refilled with reservoir water to approximately 300 mL of overlying water.

Background NO_3N concentration (mg L^{-1}) of reservoir water was assayed using second-derivative spectroscopy (Crumpton et al., 1992). The overlying water of each core was then spiked with a concentrated NO_3 solution to achieve an average starting concentration of 13.2 mg L^{-1} NO_3N. The initial target starting concentration was the 99th percentile NO_3N concentration of samples collected at the Red Rock Dam from 1978–2022 (Schilling et al., 2023). Cores were incubated at 22°C in the light. A 5 mL aliquot was sampled from each core at time zero and then approximately every 8 h until concentrations approached zero in some cores. Samples were acidified with 50 μL of sulfuric acid and assayed for NO_3-N as above. The 24-h nitrate loss rate, calculated as g N m^2 day^{-1}, was used to scale up to mapped delta inundation areas.

2.4 | Inundation mapping

Satellite imagery was used to develop a rating curve to quantify land area inundation within the delta. Google Earth Engine was used to filter and download multispectral satellite images from the Copernicus Sentinel-2 mission (10 m spatial resolution) for days that corresponded to specific lake elevations. Imagery was selected from the months of October through May when vegetation did not mask water/land boundaries. Nine sentinel imagery dates from 2015 to 2021 were selected that represented various lake elevations across an elevation range from 226.0 to 232.7 m. Images with negligible cloud cover were used to create false color images and identify the upstream edge of the reservoir for multiple lake elevations. The boundary edge between water and land was then hand-digitized at various lake elevations within the lower delta. Contours were developed to quantify the area of the lower delta inundated for a given lake level elevation (Figure 1). An inundation rating curve was developed using linear interpolation. Using the historical record of lake level elevations for Red Rock, we estimated each day's area of inundation from 1991 to 2020. The 1991–2020 period was selected because it followed significant dredging by the USACE and the initiation of new Red Rock operational water levels.

2.5 | Quantifying potential NO_3-N reductions

We assumed most N loss in reservoir delta sediments was due to denitrification (i.e., David et al., 2006; Hansen et al., 2016). This reaction is temperature dependent and takes the following form:

$$k = k_{20} \times Q_{10}^{T-20}$$

where k is the temperature-dependent daily N loss rate, k_{20} is the N loss rate at 20°C, Q_{10} is a measure of temperature sensitivity, which describes the increase in reaction rate corresponding to a 10°C change in temperature (Mundim et al., 2020), and T is the water temperature.

We used daily water temperature measured in the Des Moines River at Swan (available from 2012 to 2022; USGS 05487520) to derive a seasonal model of water temperature used for the entire estimation period. The k_{20} N loss rate was determined experimentally using the N loss observed in shallow lower delta sediment cores. For most biological processes, Q_{10} ranges between 1 and 3. A conservative value of 1.1 (Crumpton et al., 2020) was assumed for Q_{10} as larger Q_{10} values result in a greater estimated N loss.

Our approach assumed N loss was a zero-order reaction, that is, the rate was unaffected by NO_3-N concentration. Nitrogen loss observed in sediment cores exhibited this behavior, with rates consistent across the measured 8-, 16-, and 24-h periods (Figure S1). Furthermore, the modeled concentrations used by Schilling et al. (2023) to estimate Red Rock's incoming NO_3-N were normally distributed, with 99% of values less than the starting concentration of 13.2 mg L^{-1} NO_3-N (Figure S2).

The daily NO_3-N mass reduced with delta inundation was estimated by applying the mean N 24-h loss rate (g N m^2 day^{-1}) by the area of inundation (m^2)

$$\text{Delta N loss} = k \times (\text{Area of inundation}).$$

We aggregated daily N losses across the entirety of our study period (1991–2020) and over various annual and monthly scales.

Nitrogen reductions estimated by delta inundation were compared to the monthly and annual inflow and outflow N loads reported by Schilling et al. (2023). NO_3-N concentrations were monitored upstream and downstream of the reservoir by the USACE through a cooperative agreement with Iowa State University (Des Moines River Water Quality Network). From 1978 to 2019, more than 1000 samples were collected from the Des Moines River near Swan (upstream; $n = 1059$; Site 7) and near Pella (downstream; $n = 1029$; Site 9). Daily NO_3-N loads at monitoring sites (Swan, Pella) were estimated using Weighted Regressions on Time, Discharge, and Season (WRTDS) and available discharge and concentration data (Hirsch et al., 2010). The WRTDS model was improved by accounting for the autocorrelation structure of model residuals with the addition of a Kalman filter (WRTDS-K; Zhang & Hirsch, 2019). Through these modeling efforts, Schilling et al. (2023) determined that approximately 1,858,000 Mg of NO_3-N entered Red Rock from 1991 to 2020 through the Swan site and minor tributaries. A net NO_3-N loss of 233,500 Mg for this same period was estimated by comparing inflow loads with those at the downstream Pella site.

To quantify potential NO_3-N reductions in delta sediments, we modified the records of Red Rock's daily water levels from 1991 to 2020, assuming the operators raised Red Rock water levels to achieve further N loss in the lower delta area. Stage elevations were raised by a set increment each day when the water level was between 226.16 and 227.2 m, representing the non-flood operating limits of Red Rock. These are the pool stages where USACE operators could potentially raise water level elevations as restrictions due to drought or flood are not generally present under these conditions. Theoretical stages were kept equal to the original historical observations on days outside this range. We calculated theoretical N loss due to the larger inundation area resulting from increased water elevations.

Various stage increments were explored, ranging from a 0.01 to 2 m increase. The lower delta becomes completely inundated at an elevation of 227.2 m. Consequently, raising the stage beyond 2 m does not provide additional N loss. On days where the original water elevation was between 226.16 and 227.2 m, the stage was raised by 1 m. On days when the water elevation was outside this range, the stage values matched the original record.

3 | RESULTS AND DISCUSSION

3.1 | Sedimentology of delta deposits

Shallow (<10 cm) sediments within the lower delta area consisted largely of coarse silt (36.3%) and sand (27.7%), with lesser amounts of fine silt and clay (Table 1). Sand content was typically higher in distributary channel areas, whereas silt and clay content was higher in the interdistributary plain regions. Total N and carbon (C) contents were relatively consistent across the 51 sampling locations, averaging $0.09 \pm 0.02\%$ and $1.28 \pm 0.03\%$, respectively. Organic matter content ranged from 0.08% to 3.1% and averaged $1.6 \pm 0.4\%$ (Table 1). The mean bulk density of the delta sediment ($n = 8$) was 1.49 ± 0.08 g cm^{-3}, and all sampled sediments in the study area were leached of calcium carbonate.

The dominant silt and sand fractions found in the lower delta surface sediments of Lake Red Rock reflect deposition and water sorting of watershed-derived materials eroded from the Southern Iowa Drift Plain (SIDP) landscape region of Iowa (Prior, 1991). Sediment contributions from the Des Moines Lobe landform region of Iowa are captured in Saylorville Lake reservoir on the Des Moines River (Hansen et al., 2016). The surficial geology of the SIDP region consists of variable loess (silt) overlying fine-textured pre-Illinoian till, and these source sediments, along with high C and N levels eroded from watershed soils, are deposited regularly in the delta during normal rainfall-runoff events and major floods. Particle size distribution in the lower delta is similar to average conditions in pre-Illinoian till (44% sand, 38% silt, and 18% clay; Schilling & Tassier-Surine, 2006). Some winnowing of coarse sediments likely occurs across the entire delta, as coarser sands are deposited at the river mouth and silt fractions increase in the lower delta (Shaughnessey et al., 2019).

3.2 | N loss rates

Based on laboratory assays conducted on 51 soil cores, aqueous NO_3-N concentrations within shallow lower delta sediment decreased by an average of 0.66 ± 0.22 g N m^2 day^{-1} (Table 1). The initial concentration of 13.2 mg L^{-1} NO_3-N for analyses represented a 99% exceedance probability for the reservoir (Figure S2). Nitrogen loss values ranged from 0.08 to 1.22, but most were clustered between 0.4 and 1 g N m^2 day^{-1}.

Nitrogen loss estimates were not strongly related to physical parameters describing delta sediments. Although 24-h N loss was positively related to percentage of sand content ($r^2 = 11.1\%$; $p = 0.019$; Figure 2) and negatively related to percentage of fine silt ($r^2 = 11.6\%$; $p = 0.019$) and percentage of clay ($r^2 = 15.6\%$; $p = 0.005$), the explanatory power was weak. Nitrogen loss was also weakly negatively correlated to percentage of TN ($r^2 = 5.7\%$; $p = 0.054$) but not significantly related to total C, C:N ratios, or organic matter content ($p > 0.1$). The lack of significant relation of N loss to spatial and textural variations within delta sediments suggests a mean value of N loss was appropriate to use for upscaling.

TABLE 1 Summary of sedimentology and nitrogen (N)-loss measurements in the upper 20 cm of delta sediment ($n = 51$ samples).

	Sand (%)	Coarse silt (%)	Fine silt (%)	Total silt (%)	Clay (%)	Total N (%)	Total C (%)	C:N	Organic matter (%)	N loss 8-h (g N m^2 day^{-1})	N loss 24 h (g N m^2 day^{-1})
Mean	27.7	36.3	18.7	53.9	18.4	0.09	1.28	14.5	1.6	0.436	0.659
Median	27.9	37.2	17.7	54.6	17.8	0.09	1.32	14.3	1.6	0.448	0.651
Standard deviation	13.3	6.6	6.4	10.3	4.0	0.02	0.30	1.7	0.4	0.310	0.215
Min	2.9	21.5	7.4	31.3	12.9	0.04	0.65	9.7	0.8	–0.755	0.077
Max	52.9	58.2	34.7	71.2	35.0	0.13	1.86	19.7	3.1	0.983	1.226

Note: N loss rates measured in the laboratory after 8-h and 24-h incubations.
Abbreviations: Max, maximum; Min, minimum.

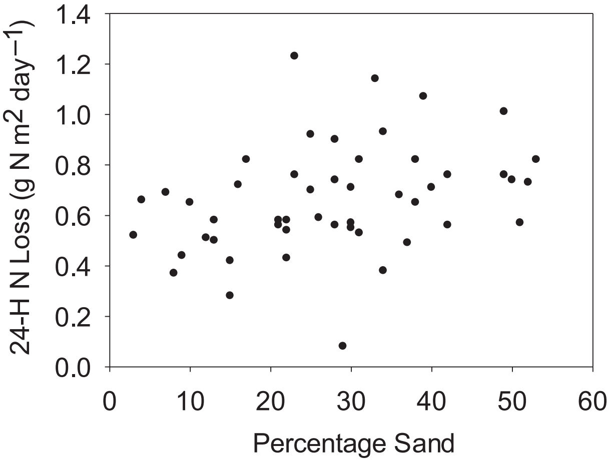

FIGURE 2 Relation of 24-h N loss measurements to sand content measured in surface soils collected from the lower delta at Lake Red Rock.

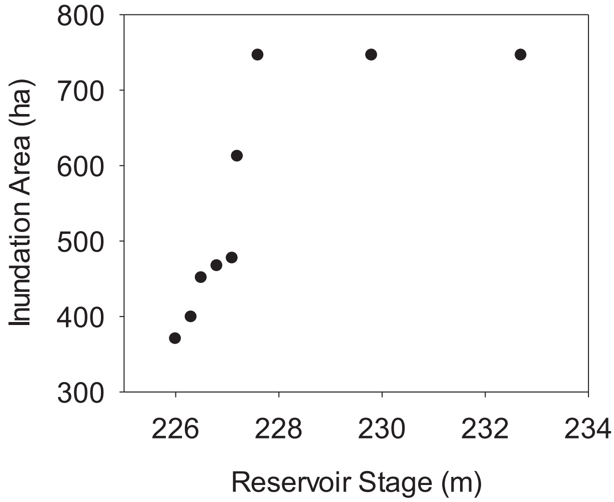

FIGURE 3 The relation between inundated area of the lower delta region at Lake Red Rock as a function of reservoir stage elevation.

3.3 | Reservoir inundation

Utilizing satellite imagery corresponding to specific lake levels, nine lake-level contours were delineated, ranging in elevation from 226.0 to 232.7 m (Figure 1). Area inundated within boundaries of the lower delta ranged from 370 ha (49% of the lower delta) at a reservoir stage of 226.0 m to 746 ha (98.8%) at a stage of 227.6 m and higher (Table 2). As reservoir levels rise, lower delta land is inundated. However, the increase in inundated area was not linear with stage. Inundation area increased substantially with lake level rise from 226.0 to 226.5 m but then nearly leveled off at stage levels above 227.1 m (Figure 3). The low rate of increase in inundation area from 226.5 to 227.1 m was attributable to the development of vertical banks within the lower delta at this elevation range that confined the extent of the reservoir until the bank threshold was exceeded and widespread inundation could occur again. Linear interpolation was used to develop a continuous relation between reservoir stage and area of delta inundation for a daily timescale (Figure 4).

3.4 | Estimating N loss due to delta inundation

Based on an average 24-h N loss rate measured for delta sediments (0.66 g N m^2 day^{-1}), we estimated that a total of 27,440 Mg of N were removed via delta inundation from 1991 to 2020. This value is equivalent to 1.5% of the total N entering Red Rock across this same period. It also represents 11.8% of the total N total removed throughout the entirety of the lake, implying inundation of the lower delta sediments was responsible for nearly 12% of the observed N loss in the reservoir.

Raising pool elevations to inundate more of the lower delta would result in greater N losses (Table 3; Figure 5). An increase of 0.1 m in pool stage would provide an additional 575 Mg N loss. A 0.5 and 1 m increase yielded further losses of 2968 Mg and 7726 Mg, respectively. The maximum additional N loss gained by inundating the lower delta was 10,950 Mg. At this point, the lower delta would be continuously and completely submerged, so further increasing Red Rock's

TABLE 2 Area and percent of lower delta inundation corresponding to each lake level and relative area increase based on lake level rise. Elevation in feet represents actual measurements collected by the US Army Corps of Engineers.

Date	Elevation (m)	Elevation (ft)	Inundation area (ha)	Inundation area (%)	Relative area increase (%)
September 8, 2021	226.0	741.59	370	49.0	
May 1, 2021	226.3	742.57	399	52.8	7.8
April 26, 2021	226.5	743.05	451	59.7	13.0
October 18, 2021	226.8	744.02	467	61.9	3.5
December 2, 2020	227.1	744.94	477	63.2	2.1
March 2, 2021	227.2	745.5	612	81.1	28.3
December 4, 2015	227.6	746.88	746	98.8	21.9
December 8, 2018	229.8	754.07	746	98.8	0.0
May 22, 2019	232.7	763.52	746	98.8	0.0

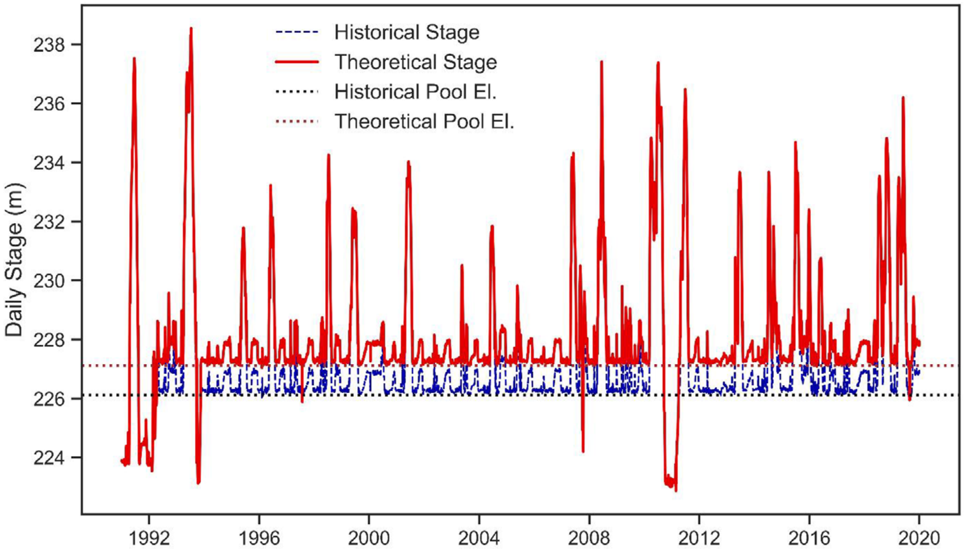

FIGURE 4 Red Rock's historical stages and an example of theoretically increasing the original daily stage elevations (EL) by 1 m during normal hydrologic conditions.

elevation fails to provide additional N loss. This maximum value represents 2.1% of the total inflow N and 16.4% of the lake's total N loss (Table 3). On an annual basis, increasing the reservoir stage would result in an increase of annual N loss from 2 to 377 Mg per year. Due to the nonlinearity between lake elevation and inundation area (Figure 3), the relationship between stage increase and delta N loss is not linear. The greatest relative gains in N loss occur between stage increases of 0.75–1.25 m, when annual N mass loss would increase nearly 2.4 times from 142 to 340 Mg (Table 3).

There is also a strong seasonal component to N loss due to warmer summer temperatures facilitating higher N removal rates. From June through September, when water temperatures exceed 20°C, more N loss occurs (Figure 6). This time period coincides with seasonal high NO_3-N concentrations in the Des Moines River (Schilling et al., 2023), suggesting that raising lake elevations during the summer period has potential for greater N reductions.

3.5 | Further considerations

This study highlights the potential for water stage manipulation to increase N loss in river water flowing through a flood control reservoir. However, for a control scheme to be implemented by the USACE at Lake Red Rock, refinement of the simplistic control scheme is needed. In our study, we had the benefit of theoretically manipulating the reservoir stage to achieve N load reductions, whereas in actuality, the USACE would be forced to balance potential N reductions with other objectives of the reservoir, such as flood control, recreation, and other factors. Although we allowed pool levels to fluctuate during floods and droughts, Lake Red Rock is operating under an environmental pool management plan (Warner et al., 2014) that will constrain the manipulation of base pool levels. The plan mandates water levels are held higher in the spring to support fish spawning, slowly lowered during summer to promote wetland expansion, raised during fall waterfowl migration,

TABLE 3 Nitrogen (N) loss is associated with increasing Red Rock's stage (1991–2020). Losses are contextualized by dividing the amount of N entering Red Rock (percentage of N inflow) and the total N loss observed throughout the reservoir (percentage of total Red Rock N loss). The N entering Red Rock (1,858,000 Mg) and N loss throughout the reservoir (233,500 Mg) are provided in Schilling et al. (2023).

Stage increase (m)	Delta N loss (Mg)	Annual delta N loss (Mg)	Increased N loss (Mg)	Annual increased N loss (Mg year^{-1})	Percentage of N inflow (%)	Percentage of total Red Rock N loss (%)	Equivalent edge of field practices per year[a]
0	27,440	946	–	–	1.48	11.8	
0.01	27,486	948	46	2	1.48	11.8	13
0.05	27,686	955	247	9	1.49	11.9	56
0.1	28,015	966	575	20	1.51	12.0	125
0.2	28,909	997	1469	51	1.56	12.4	319
0.3	29,575	1020	2135	74	1.59	12.7	463
0.4	30,010	1035	2570	89	1.61	12.9	556
0.5	30,407	1049	2968	102	1.64	13.0	638
0.75	31,548	1088	4108	142	1.70	13.5	888
1	35,166	1213	7726	266	1.89	15.1	1663
1.25	37,308	1286	9868	340	2.01	16.0	2125
1.5	38,369	1323	10,930	377	2.06	16.4	2356
2	38,377	1323	10,938	377	2.06	16.4	2356

[a]Assumes 20 kg/ha N loss in field × 16 ha of field area × 50% N reduction = 160 kg/year, consistent with edge-of-field N load reductions reported for bioreactors (Christianson et al., 2021), saturated buffers (Isenhart & Jaynes, 2019; Streeter & Schilling, 2021), and multipurpose oxbows (Schilling et al., 2019; Pierce & Schilling, 2023).

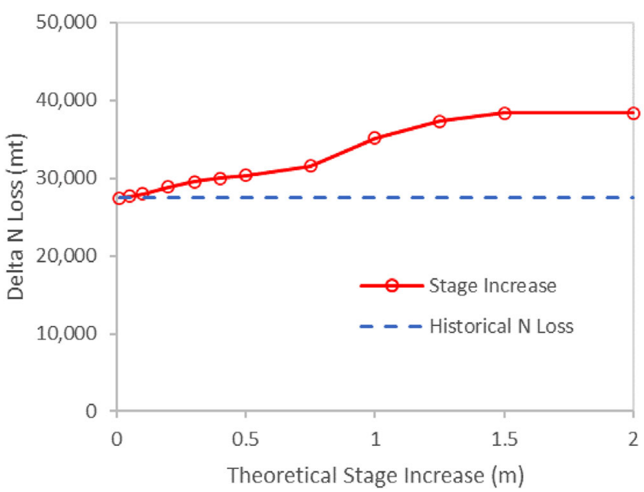

FIGURE 5 The total estimated delta N loss in metric tons (mt) (1991–2020) corresponding to various theoretical increases in reservoir stage. Additional N removal was determined for each theoretical stage increase achieved by raising Red Rock's recorded stages by a specified increment during normal hydrologic conditions. The historical N loss (27,400 mt) estimated using the recorded stage elevations was added as a reference.

and held high through winter to support reptile overwintering. It is clear that additional modeling is needed to assess how adjusting pool levels for greater N loss could be integrated with the existing environmental pool management plan.

In addition, there is ongoing sedimentation in Lake Red Rock that will impact future lake levels and management operations. The 2011 sedimentation report documented a sediment deposition rate of approximately 85 ha-m (288 acre-ft) for the conservation pool that has reduced the pool capacity by 44% since 1969 (U.S. Army Corps of Engineers, 2011). At the time, it was estimated that the current conservation pool may be filled with sediment around 2065. It is conceivable that continued sedimentation may spur another pool operation shift in the next few decades that would inundate sediments within the upper portion of the delta. However, given the relative similarity of sediment N loss measurements in this study, we would anticipate similar NO_3-N mass reductions should the area of sediment inundation was equivalent.

There are few opportunities to achieve landscape-scale NO_3-N reductions in Midwestern rivers beyond edge-of-field practices such as bioreactors (Addy et al., 2016; Christianson et al., 2021), saturated buffers (Jaynes & Isenhart, 2019), multi-purpose oxbows (Schilling et al., 2019), and larger NO_3-N-removal wetlands (Crumpton et al., 2020). The potential to manipulate reservoir pool elevations to achieve N load reductions could be developed as a new N reduction practice operating at a novel landscape scale. Study results suggest that achieving even a small percent reduction in N in a large river such as the Des Moines River results in a large N load reduction compared to edge-of-field practices. Increasing pool stage in Lake Red Rock by 0.1 m would be the N removal equivalent to installing 125 bioreactors or saturated buffers in the watershed (Table 3). Increasing pool levels by 0.5 and 1.0 m would be equivalent to installing 638 and 1663 practices, respectively. Hence,

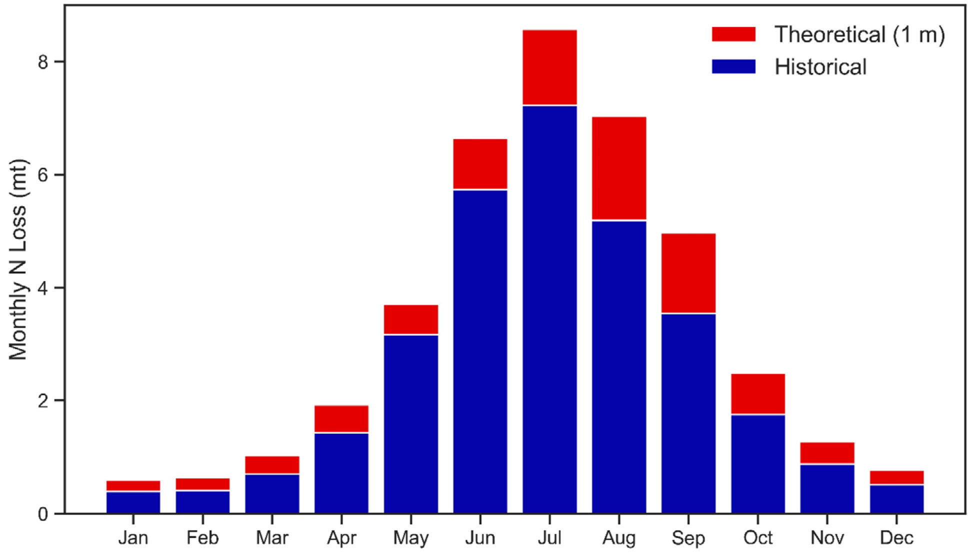

FIGURE 6 The monthly estimated delta N loss in metric tons (1991–2020). Historical losses were estimated based on Red Rock's recorded stage elevations and theoretical losses represent additional N removal that could have been achieved by raising the stage 1 m during normal hydrologic conditions.

focusing on reservoir pool management for N reductions could achieve results that greatly surpass the rate of current conservation practice implementation at the edge-of-field scale.

AUTHOR CONTRIBUTIONS

Keith E. Schilling: Conceptualization; formal analysis; funding acquisition; investigation; methodology; project administration; supervision; writing—original draft; writing—review and editing. **Matthew T. Streeter**: Conceptualization; data curation; formal analysis; investigation; methodology. **Elliot Anderson**: Formal analysis; methodology. **Jennifer Merryman**: Formal analysis; investigation; methodology; writing—original draft. **Thomas Isenhart**: Conceptualization; methodology; writing—review and editing. **Antonio Arenas-Amado**: Formal analysis; investigation; methodology. **Chuck Theiling**: Conceptualization; funding acquisition; project administration; resources.

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CONFLICT OF INTEREST STATEMENT

The authors declare no conflicts of interest.

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REFERENCES

Addy, K., Gold, A. J., Christianson, L. E., David, M. B., Schipper, L. A., & Ratigan, N. A. (2016). Denitrifying bioreactors for nitrate removal: A meta-analysis. *Journal of Environmental Quality*, *45*(3), 873–881. https://doi.org/10.2134/jeq2015.07.0399

Alexander, R. B., Smith, R. A., & Schwarz, G. E. (2000). Effect of stream channel size on the delivery of nitrogen to the Gulf of Mexico. *Nature*, *403*(6771), 758–761. https://doi.org/10.1038/35001562

Choi, K. S., Dalrymple, R. W., Chun, S. S., & Kim, S.-P. (2004). Sedimentology of modern, inclined heterolithic stratification (IHS) in the macrotidal Han River delta, Korea. *Journal of Sedimentary Research*, *74*(5), 677–689. https://doi.org/10.1306/030804740677

Christianson, L. E., Cooke, R. A., Hay, C. H., Helmers, M. J., Feyereisen, G. W., Ranaivoson, A. Z., Mcmaine, J. T., Mcdaniel, R., Rosen, T. R., Pluer, W. T., Schipper, L. A., Dougherty, H., Robinson, R. J., Layden, I. A., Irvine-Brown, S. M., Manca, F., Dhaese, K., Nelissen, V., & Von Ahnen, M. (2021). Effectiveness of denitrifying bioreactors on water pollutant reduction from agricultural areas. *Transactions of the ASABE*, *64*(2), 641–658. https://doi.org/10.13031/trans.14011

Crumpton, W. G., Isenhart, T. M., & Mitchell, P. D. (1992). Nitrate and organic N analyses with second-derivative spectroscopy. *Limnology and Oceanography*, *37*(4), 907–913. https://doi.org/10.4319/lo.1992.37.4.0907

Crumpton, W. G., Stenback, G. A., Fisher, S. W., Stenback, J. Z., & Green, D. I. S. (2020). Water quality performance of wetlands receiving nonpoint-source nitrogen loads: Nitrate and total nitrogen removal efficiency and controlling factors. *Journal of Environmental Quality*, *49*, 735–744. https://doi.org/10.1002/jeq2.20061

David, M. B., Wall, L. G., Royer, T. V., & Tank, J. L. (2006). Denitrification and the nitrogen budget of a reservoir in an agricultural landscape. *Ecological Applications*, *16*(6), 2177–2190. https://doi.org/10.1890/1051-0761(2006)016%5b2177:DATNBO%5d2.0.CO;2

Day, J. W., Boesch, D. F., Clairain, E. J., Kemp, G. P., Laska, S. B., Mitsch, W. J., Orth, K., Mashriqui, H., Reed, D. J., Shabman, L., Simenstad, C. A., Streever, B. J., Twilley, R. R., Watson, C. C., Wells, J. T., & Whigham, D. F. (2007). Restoration of the Mississippi Delta: Lessons from Hurricanes Katrina and Rita. *Science*, *315*(5819), 1679–1684. https://doi.org/10.1126/science.1137030

Dodds, W. K. K., & Welch, E. B. (2000). Establishing nutrient criteria in streams. *Journal of the North American Benthological Society*, *19*(1), 186–196. https://doi.org/10.2307/1468291

Hansen, E., Chan, K.-S., Jones, C. S., & Schilling, K. (2016). Assessing the relative importance of nitrogen-retention processes in a large reservoir using time-series modeling. *Journal of Agricultural, Biological, and Environmental Statistics*, *21*, 152–169. https://doi.org/10.1007/s13253-015-0218-1

Hiatt, M., Castañeda-Moya, E., Twilley, R., Hodges, B. R., & Passalacqua, P. (2018). Channel-island connectivity affects water exposure time distributions in a coastal river delta. *Water Resources Research*, *54*(3), 2212–2232. https://doi.org/10.1002/2017WR021289

Hirsch, R. M., Moyer, D. L., & Archfield, S. A. (2010). Weighted regressions on time, discharge, and season (WRTDS), with an application to Chesapeake Bay river inputs. *JAWRA Journal of the American Water Resources Association*, *46*(5), 857–880. https://doi.org/10.1111/j.1752-1688.2010.00482.x

Innocent, L., & Pranzini, E. (1993). Geomorphological evolution and sedimentology of the Ombrone River delta, Italy. *Journal of Coastal Research*, *9*, 481–493.

Jaynes, D. B., & Isenhart, T. M. (2014). Reconnecting tile drainage to riparian buffer hydrology for enhanced nitrate removal. *Journal of Environmental Quality*, *43*(2), 631–638. https://doi.org/10.2134/jeq2013.08.0331

Jaynes, D. B., & Isenhart, T. M. (2019). Increasing infiltration in saturated buffers by adding additional distribution pipes. *Journal of Soil and Water Conservation*, *74*, 545–553. https://doi.org/10.2489/jswc.74.6.545

Jaynes, D. B., & Isenhart, T. M. (2019). Performance of saturated riparian buffers in Iowa, USA. *Journal of Environmental Quality*, *48*, 289–296. https://doi.org/10.2134/jeq2018.03.0115

Jha, M. K., Wolter, C. F., Schilling, K. E., & Gassman, P. W. (2010). Assessment of total maximum daily load implementation strategies for nitrate impairment of the Raccoon River, Iowa. *Journal of Environmental Quality*, *39*(4), 1317–1327. https://doi.org/10.2134/jeq2009.0392

Jones, C. S., Kim, S.-W., Wilton, T. F., Schilling, K. E., & Davis, C. A. (2018). Nitrate uptake in an agricultural stream estimated from high-frequency, in-situ sensors. *Environmental Monitoring and Assessment*, *190*, Article 226. https://doi.org/10.1007/s10661-018-6599-1

Jones, C. S., Li, T., Sukalski, A., Thompson, D. A., & Cwiertny, D. M. (2020). Use of real-time sensors for compliance monitoring of nitrate in finished drinking water. *Water Science and Technology*, *82*(12), 2725–2736. https://doi.org/10.2166/wst.2020.365

Kimble, J., Knox, E., & Holzhey, C. (1993). Soil survey laboratory methods for characterizing physical and chemical properties and mineralogy of soils. In O. Hoddinott (Ed.), *Application of agricultural analysis in environmental studies* (pp. 23–31). ASTM International.

Knights, D., Sawyer, A. H., Barnes, R. T., Piliouras, A., Schwenk, J., Edmonds, D. A., & Brown, A. M. (2020). Nitrate removal across ecogeomorphic zones in Wax Lake Delta, Louisiana (USA). *Water Resources Research*, *56*(8), e2019WR026867. https://doi.org/10.1029/2019WR026867

Li, S., Christensen, A., & Twilley, R. R. (2020). Benthic fluxes of dissolved oxygen and nutrients across hydrogeomorphic zones in a coastal deltaic floodplain within the Mississippi River delta plain. *Biogeochemistry*, *149*, 115–140. https://doi.org/10.1007/s10533-020-00665-8

Mclellan, E., Robertson, D., Schilling, K., Tomer, M., Kostel, J., Smith, D., & King, K. (2015). Reducing nitrogen export from the Corn Belt to the Gulf of Mexico: Agricultural strategies for remediating hypoxia. *Journal of the American Water Resources Association*, *51*(1), 263–289. https://doi.org/10.1111/jawr.12246

Mundim, K. C., Baraldi, S., Machado, H. G., & Vieira, F. M. C. (2020). Temperature coefficient (Q10) and its applications in biological systems: Beyond the Arrhenius theory. *Ecological Modelling*, *431*, 109127. https://doi.org/10.1016/j.ecolmodel.2020.109127

Pierce, S., & Schilling, K. E. (2023). Nutrient retention in tile-fed and non-tile reconstructed oxbows in north central Iowa. *Journal of the American Water Resources Association*, *59*(5), 913–928. https://doi.org/10.1111/1752-1688.13124

Prior, J. C. (1991). *Landforms of Iowa*. University of Iowa Press.

Schilling, K. E., Anderson, E., Streeter, M. T., & Theiling, C. (2023). Long-term nitrate-nitrogen reductions in a large flood control reservoir. *Journal of Hydrology*, *620*, 129533. https://doi.org/10.1016/j.jhydrol.2023.129533

Schilling, K. E., & Tassier-Surine, S. (2006). Groundwater flow and velocity in a 500 ka pre-Illinoian till, eastern Iowa. *Environmental Geology*, *50*(8), 1255–1264. https://doi.org/10.1007/s00254-006-0299-9

Schilling, K. E., Wilke, K., Pierce, C. L., Kult, K., & Kenny, A. (2019). Multipurpose oxbows as a nitrate export reduction practice in the agricultural Midwest. *Agricultural & Environmental Letters*, *4*(1), 190035.

Schilling, K. E., & Wolter, C. F. (2009). Modeling nitrate-nitrogen load reduction strategies for the Des Moines River, Iowa using SWAT. *Environmental Management*, *44*(4), 671–682. https://doi.org/10.1007/s00267-009-9364-y

Sendrowski, A., Castañeda-Moya, E., Twilley, R., & Passalacqua, P. (2021). Biogeochemical and hydrological variables synergistically influence nitrate variability in coastal deltaic wetlands. *Journal of Geophysical Research: Biogeosciences*, *126*(9), e2020JG005737. https://doi.org/10.1029/2020JG005737

Shaughnessy, A. R., Sloan, J. J., Corcoran, M. J., & Hasenmueller, E. A. (2019). Sediments in agricultural reservoirs act as sinks and sources for nutrients over various timescales. *Water Resources Research*, *55*(7), 5985–6000. https://doi.org/10.1029/2018WR024004

Smith, D. R., & Livingston, S. J. (2013). Managing farmed closed depressional areas using blind inlets to minimize phosphorus and nitrogen losses. *Soil Use and Management*, *29*, 94–102. https://doi.org/10.1111/j.1475-2743.2012.00441.x

Stenback, G. A., Crumpton, W. G., & Schilling, K. E. (2014). Nitrate loss in Saylorville Lake reservoir in Iowa. *Journal of Hydrology*, *513*, 1–6. https://doi.org/10.1016/j.jhydrol.2014.03.037

Streeter, M. T., & Schilling, K. E. (2021). Quantifying the effectiveness of a saturated buffer to reduce tile NO_3-N concentrations in eastern Iowa. *Environmental Monitoring and Assessment*, *193*, Article 500. https://doi.org/10.1007/s10661-021-09297-3

Turner, R. E., Rabalais, N. N., & Justic, D. (2008). Gulf of Mexico hypoxia: Alternate states and a legacy. *Environmental Science & Technology*, *42*(7), 2323–2327.

Twilley, R. R., Bentley, S. J., Chen, Q., Edmonds, D. A., Hagen, S. C., Lam, N. S.-N., Willson, C. S., Xu, K., Braud, D., Hampton Peele, R., & Mccall, A. (2016). Co-evolution of wetland landscapes, flooding, and human settlement in the Mississippi River Delta Plain. *Sustainability Science*, *11*, 711–731. https://doi.org/10.1007/s11625-016-0374-4

Twilley, R. R., Day, J. W., Bevington, A. E., Castañeda-Moya, E., Christensen, A., Holm, G., Heffner, L. R., Lane, R., Mccall, A., Aarons, A., Li, S., Freeman, A., & Rovai, A. S. (2019). Ecogeomorphology of coastal deltaic floodplains and estuaries in an active delta: Insights from the Atchafalaya Coastal Basin. *Estuarine, Coastal and Shelf Science*, *227*, 106341. https://doi.org/10.1016/j.ecss.2019.106341

U.S. Army Corps of Engineers. (2011). *Report of sedimentation 2011 resurvey, Red Rock Reservoir Des Moines River Basin, Des Moines River, Iowa*. US Army Corps of Engineers, Rock Island District.

U.S. Army Corps of Engineers. (2020). *National inventory of dams*. https://nid.usace.army.mil/#/

Warner, A. T., Bach, L. B., & Hickey, J. T. (2014). Restoring environmental flows through adaptive reservoir management: Planning, science, and implementation through the Sustainable Rivers Project. *Hydrological Sciences Journal*, *59*(3–4), 770–785. https://doi.org/10.1080/02626667.2013.843777

White, J. R., Delaune, R. D., Justic, D., Day, J. W., Pahl, J., Lane, R. R., Boynton, W. R., & Twilley, R. R. (2019). Consequences of Mississippi River diversions on nutrient dynamics of coastal wetland soils and estuarine sediments: A review. *Estuarine, Coastal and Shelf Science*, *224*, 209–216. https://doi.org/10.1016/j.ecss.2019.04.027

Zhang, Q., & Hirsch, R. M. (2019). River water-quality concentration and flux estimation can be improved by accounting for serial correlation through an autoregressive model. *Water Resources Research*, *55*(11), 9705–9723. https://doi.org/10.1029/2019WR025338

Zhou, X., Helmers, M. J., Asbjornsen, H., Kolka, R., Tomer, M. D., & Cruse, R. M. (2014). Nutrient removal by prairie filter strips in agricultural landscapes. *Journal of Soil and Water Conservation*, *69*(1), 54–64. https://doi.org/10.2489/jswc.69.1.54

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