Quantifying the effectiveness of a saturated buffer to reduce tile NO₃-N concentrations in eastern Iowa

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Abstract Agricultural drainage tiles are primary contributors to NO₃-N export from Iowa croplands. Saturated buffers are a relatively new conservation practice that diverts tile water into a distribution tile installed in a riparian buffer parallel to a stream with the intent of enhancing NO₃-N processing within the buffer. In this study, tile NO₃-N concentration reductions were characterized through two different saturated buffers at a working farm site in eastern Iowa. Study objectives were to (1) evaluate the hydrogeology and water quality patterns in the saturated buffer and (2) quantify the reduction in tile NO₃-N concentration from the saturated buffer installation. Results showed that the two saturated buffers are reducing NO₃-N concentrations in tile drainage water from input concentrations of approximately 15 mg/l to levels < 1.5 mg/l at the streamside well locations. The reduction occurs rapidly in the fine-textured and organic-rich alluvial soils with most of the reduction occurring within 1.5 m of the distribution line. Denitrification is hypothesized as being primarily responsible for the concentration reductions based on soil and water chemistry conditions, completion of

a geophysical survey (quantifying low potential for N loss to deeper aquifers), and comparisons to other similar Iowa sites. The study provides more assurance to new adopters that this practice can be installed in many areas throughout the Midwestern Cornbelt region.

Keywords Water quality · Saturated buffer · Agricultural conservation · Drainage tile · Soil organic matter · Denitrification

Introduction

Impairment of stream water quality due to agricultural nutrient runoff and leaching is a significant and growing worldwide concern. Streams are often contaminated in the headwaters of watersheds where they originate and continue to accumulate excess nutrient until release into lakes and the world's oceans. In the USA, nitrogen (N) and phosphorus (P) losses contribute to significant eutrophication of streams and major rivers (Carpenter et al., 1998), which has led to seasonal hypoxia of the Gulf of Mexico (Turner et al., 2007) and even the Great Lakes (Biddanda et al., 2018; Scavia et al., 2014; Weinke & Biddanda, 2018). In Iowa, USA, agricultural fields are primary contributors of nonpoint source nutrient exports (Schilling et al., 2011). This is likely caused by extensive agricultural tile drainage beginning in earnest in the mid-nineteenth century (Pavelis, 1987), which in most

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cases, discharges directly to streams (Iowa Department of Natural Resources, 2018). Indeed, greater than 80% of some Iowa counties are tile drained (David et al., 2010). In an attempt to reduce concentrations of agricultural nutrients in streams, several states in the Midwestern US region have adopted nutrient reduction strategies that promote conservation practices to improve water quality (Christianson et al., 2018). These states include Minnesota (MNRS, 2014), Illinois (INLRS, 2014), Ohio (ONRS, 2013), and Iowa (INRS, 2013).

In 2014, saturated buffers were added to the growing list of approved conservation practices that improve water quality in Iowa (Iowa Nutrient Research Center, 2019). Saturated buffers work by intercepting tile water from a field before it enters a stream (Jaynes & Isenhardt, 2014). A secondary tile line runs perpendicular to the first and parallel to the stream within a riparian buffer near the stream. The tile water thus saturates the soil within the buffer, creating an anaerobic soil environment which promotes denitrification of water before it is exported from the field (Jaynes & Isenhardt, 2014). Appropriate locations for saturated buffers are vast since the requirements for successful implementation are limited. Vegetation does not need to be previously established (Tomer, Porter, James & Van Horn, 2020), and soils containing 1.2% soil organic matter (SOM) are sufficient for the denitrification process (Chandrasoma, Christianson & Christianson, 2019). Saturated buffers have been installed and monitored in several locations in Iowa and have been shown to significantly reduce tile $\text{NO}_3\text{-N}$ loads including up to 84% reductions over 17 site years of data in central Iowa (Jaynes & Isenhardt, 2019). The Iowa Nutrient Research Center estimates a 50% average load reduction in $\text{NO}_3\text{-N}$ via the implementation of saturated buffers (Iowa State University, 2020). Denitrification is considered the dominant $\text{NO}_3\text{-N}$ removal process in the buffers, removing between 27 and 96% of the diverted N load into the buffers (Groh et al., 2019). Some preliminary research in the Lake Erie basin has shown complete reductions of dissolved $\text{NO}_3\text{-N}$ and P over a 12-month period (Jacquemin, McGlinch, Dirksen & Clayton, 2020). Chandrasoma, Christianson & Christianson (2019) indicated that there is a potential to install more than 248,000 saturated buffers across the Midwest US region lining more than 75,000 km of stream banks.

Still, the practice of saturating riparian zones to promote denitrification is in its infancy and little work has been done to confirm their success in various soil types and landscapes. Since saturated buffers are dependent upon specific soil properties including particle size, hydraulic conductivity, and SOM, it is necessary to assess specific site characteristics at the inception of a practice and to test the effectiveness of saturated buffers in environments where these properties vary. The Iowan Surface of eastern Iowa is a region that provides significant variability in riparian soils which often include buried soils and deep post-settlement alluvial packages. This region has yet to be studied for its saturated buffer potential.

The goal of this study was to characterize tile $\text{NO}_3\text{-N}$ concentration reductions through two different saturated buffers at a site in eastern Iowa. Study objectives were to evaluate the hydrogeology and water quality patterns in the saturated buffer and quantify the reduction in tile $\text{NO}_3\text{-N}$ concentration from the saturated buffer installation. This study expands the extent of monitoring beyond those studies concentrated in north-central Iowa (e.g., Jaynes & Isenhardt, (2019)) and provides additional empirical data quantifying the impacts of this rather new conservation practices in a different environmental setting.

Methods

Study area

Our study area included 40 ha of agricultural lands on both sides of an approximately 275-m section of Hickory Creek in Scott County, IA (Fig. 1). Hickory Creek is a tributary of Mud Creek which drains into the Wapsipinicon River approximately 20 km to the northeast. The Wapsipinicon River drains into the Mississippi River approximately 30 km east of where Mud Creek enters. The saturated buffer site is located in the Southern Iowa Drift Plain landform region of Iowa (Prior, 1991), a region dominated by a rolling pre-Illinoian glacial subsurface capped with Wisconsin-age loess. Floodplains in the region are typically composed of fine-textured alluvium sourced from silty upland glacial sediments. While the landscape is generally defined as

Fig. 1 Location of study area, saturated buffers, and sampling sites in Scott County, IA

relic or well-developed, extensive soil erosion and deposition has been enhanced by post-settlement agricultural practices (Cruse et al., 2006) and post-settlement deposition of floodplain soils (Schilling & Wolter, 2009).

In 2003, riparian buffer strips approximately 20 m wide were installed along the stream. The buffer vegetation consisted largely of cool-season grasses including 50% *Lolium perenne* (ryegrass), 20% *Festuca* (fescue), 15% *Poa* (bluegrass), and

15% *Phleum pratense* (timothy). Buffer vegetation was generally healthy and well managed by the landowner. Generally, two cuttings of hay were harvested from the buffer area annually. Agricultural fields bordering the buffer were managed under a 2-year *Zea mays* L. (corn) and *Glycine max* L. (soybean) rotation. The agricultural fields were fertilized prior to planting corn using a combination of manure N from nearby hog confinement facilities applied fall or spring and commercial N (urea ammonium nitrate) applied at planting at a combined rate of approximately 190 kg/ha. The fields were underlain by subsurface drains installed approximately 1.2 m below the ground surface that join and discharge into the stream at regular tile outlet points (Fig. 1). Subsurface drainage tiles bypass the potential N processing available in buffers (Schilling et al., 2015a, b), and these were the focus of the saturated buffer installation.

In 2017, two saturated buffers were installed within the riparian zones on both sides of the stream. The buffers were installed by digging down to the existing 12-cm-diameter tile outlet, installing the control structure, and then trenching in a 15-cm-diameter perforated tile parallel to the creek. On the south side of the stream, the saturated buffer tile intercepted flow from the SW tile outlet and paralleled the stream for approximately 162 m. It received tile water from an area of approximately 10 ha (Fig. 1). On the north side, the saturated buffer tile was shorter (87 m) and intercepted discharge from the NW tile. That buffer received tile water from an area of approximately 6 ha (Fig. 1). Both buffer systems were equipped with a tile gate control box that allowed for manual adjustment of the amount of tile water entering the buffer. Tile gates were occasionally adjusted by the landowner throughout the year (unrecorded) to accommodate for wetter conditions in the spring (lowered) and drier conditions in the summer and fall (raised). Tile gates were lowered to allow for heavy equipment traffic in the buffer during field planting and harvest. In those cases, tile gates were adjusted approximately 2 days prior to fieldwork commencing and readjusted as soon as fieldwork was completed. Unlike saturated buffer monitoring sites established for research (e.g., Jaynes & Isenhardt, (2019)), the current site was first installed by a landowner and then later retrofitted with wells for monitoring. Hence, the current site was built with two-chamber control boxes (not three-chamber

boxes used for measuring flow) and the focus of this research was on N concentration reductions and not load reductions.

Site characterization

A geophysical survey of the buffer sites was conducted using electrical resistivity (ER) and electromagnetic terrain conductivity (EM) methods to characterize the subsurface geology of the buffers. With the ER method, an electrical current is induced into the subsurface through stainless steel electrodes at locations along a transect and the resulting electric potential is used to map the resistivity of the subsurface materials to depths of > 100 m. After processing and inversion of the ER data, results are presented as a two-dimensional model representing bulk electrical resistivity values (in ohm-meters) at depth along a surface transect. For EM mapping, a Geonics EM-31 MK2 ground conductivity meter was used. The EM-31 maps change in ground conductivity (inverse of resistivity) using an electromagnetic induction technique with an effective depth of penetration of approximately 6 m (Geonics, 2020b). The EM-31 was affixed to an all-terrain vehicle via a wooden platform, separated from the vehicle by approximately 2 m, to facilitate rapid data collection in the field (~19,000 measurements). Vertical dipole conductivity values were recorded at a rate of five readings per second and affixed with the coordinate locations using a high-precision GPS. Site conductivity was measured in millisiemens per meter (mS/m) to a depth of 6 m. The survey points were contoured by kriging in ArcGIS (Environmental Systems Research Institute, 2012).

In the fall of 2018, shallow groundwater monitoring wells were installed in transects crossing the saturated buffers. In all, 12 wells were installed in four transects (Fig. 1). Each well was installed using a truck-mounted drill rig equipped with 11.5-cm augers. Initially, a 5.7-cm continuous soil core was collected from each well location followed by over-digging with the augers to widen the hole for well installation. Each well was 3.8 cm diameter and 2.1 m deep consisting of a 1.5-m screen and a 0.6-m riser that extended the well to the soil surface. A sorted gravel pack was placed around the screen section, and a bentonite hole plug was used to seal the well and prevent surface water contamination. Wells were

developed by surging and overpumping in the fall of 2018 and the spring of 2019 and 2020. Non-vented pressure transducers were installed in four wells (W1, W3, W7, and W9) to measure continuous water table fluctuations from March to December in 2019 and 2020. Furthermore, a pressure transducer was installed in the stream during the same time intervals to monitor stream stage.

Soil cores were collected from well locations in order to calibrate the EM31 data and gather information about soil morphological properties at the site. Each soil core was characterized according to methods of Schoeneberger (2012). Soils samples were air-dried and ground to pass through a 2-mm sieve for particle size and elemental analysis. Soil texture was determined by x-ray absorption using a SediGraph® (Olivier et al., 1971). Elemental analysis via chromatography was performed following dry combustion to measure total carbon (TC) and total nitrogen (TN). SOM was measured by weight loss on ignition (Walkley & Black, 1934) as performed by Schulte (1995).

Water sampling and analyses

Monitoring wells, stream water, and tiles were sampled on 13 occasions from April to November in 2019 and 8 occasions from June to November in 2020. Travel restrictions due to COVID-19 prevented spring water sampling in the spring of 2020. Water levels were determined to the nearest 0.3 cm at the time of sampling. Control boxes were sampled (NW and SW tiles) as well as two additional discharge tiles (NE and SE tiles) that were not included in the saturated buffer system (Fig. 1). When the tiles were above the stream water line, flow rates were estimated for each tile using a bucket and stopwatch at the time of sampling. Water samples were analyzed for multiple in-field parameters at the time of sampling including dissolved oxygen (DO), oxidation–reduction potential (ORP), pH, specific conductance (SC), and temperature using a YSI ProDSS water quality meter (Xylem, 2020). All samples were transported to the lab and analyzed for NO₃-N using a Nitratax (Hach, 2020).

NO₃-N reduction calculations

Comparisons of tile water N concentrations to groundwater buffer concentrations along with changes in N concentrations across the buffer were used to

characterize the effectiveness of the saturated buffer system. The N removal rate was computed based on the changes in concentration per meter of the buffer (mg N/l/m) according to Mayer, Reynolds Jr, McCutchen & Canfield (2007) and Jaynes & Isenhardt (2014). The rate of groundwater flow across the buffers was estimated from Darcy's law:

$$V = -Ki/n \quad (1)$$

where K is the hydraulic conductivity (m/s), i is the hydraulic gradient, and n is effective porosity. The mean hydraulic gradient across the saturated buffer was derived using the measured water table levels. The average K of the aquifer was estimated using slug tests and assuming an effective porosity of 0.25 for the alluvium. The rate of groundwater flow (V) was multiplied by the upgradient saturated thickness of the buffer (2-m depth) to estimate the daily groundwater N loads into the buffer. N loading rates into the buffer were compared with in situ denitrification rates measured in similar saturated buffers found in north-central Iowa (Groh et al., 2019) to assess whether the rate of denitrification could account for measured concentration reductions.

Water data were summarized as means and standard deviation for each sample location. Water levels were summarized as hourly means. Statistical comparisons of all data were made using analysis of variance. Conductivity data were processed using the Geonics Ltd. DAT31W software, version 2.06 (Geonics, 2020a). The data files were then compiled as spreadsheets and uploaded into ESRI ArcMap (version 10.7) to generate raster images of the data showing spatial variations in the apparent conductivity (mS/m) (Environmental Systems Research Institute, 2012).

Results

Site characterization

The geology of the two saturated buffer sites was characterized using geophysical methods and confirmed with soil borings (Fig. 2). The ER survey was conducted along four lines extending from the stream and across the buffer and into the neighboring fields. The resistivity of the sediments ranged from approximately 11 to 58 Ω m, and based on correlations

Fig. 2 Map showing results of geophysical conductivity survey

established at other sites (Vogelgesang, Holt, Schilling, Gannon & Tassier-Surine, 2019), the variations were interpreted to reflect different geologic materials (Fig. 2). Although the field resistivity measurements ranged from 11 to 58 Ω m, ER graphics were adjusted to a scale of 15 to 40 Ω m so all lines could be comparable. In both buffer areas, approximately 4.0 m of loamy sediment overlies a denser subsurface deposit (likely pre-Illinoian till).

The EM-31 mapping showed that the two riparian buffer zones were slightly different. The conductivity of the north saturated buffer (buffer 2) averaged 49 ± 3 mS/m, whereas the average conductivity in the south saturated buffer (buffer 1) was lower, 45 ± 2 mS/m. Soil particle size analysis confirmed the variability found during the geophysical investigation (Table 1). Both buffers were similar in terms of average clay content (41%), but buffer 1 had significantly higher ($p < 0.05$) sand content than buffer 2 (14 ± 10 compared to 10 ± 8 , respectively) (Table 1). Soil characterization at the site confirmed that the higher ground conductivity on the north side of the stream and within buffer 2 corresponded to a finer textured silty clay loess parent material (Tama soil series, fine-silty, mixed, superactive, mesic Typic Argiudolls). The parent material in buffer 1 is a silty alluvium with a higher average sand content (Ackmore soil series, fine-silty, mixed, superactive, nonacid, mesic Mollic Fluvaquents). Buffer 1 had less SOM than buffer 2 (5.3 ± 1.1 compared to 6.2 ± 1.0 , respectively) and also had lower concentrations of TC (1.4 ± 0.6 compared to 2.0 ± 0.6 , respectively) (Table 1). TN concentrations were not significantly different between the two buffers. Due to difference in TC between buffers, the C:N ratio varied from 24 to 15 in buffers 1 and 2, respectively.

Hydrology

Precipitation estimates for the study area were determined based on radar data analyzed by the Daily Erosion Project (Iowa State University Department of Agronomy, 2020). In 2019 and 2020, the total precipitation for the study area was 1155 and 861 mm, respectively. The annual totals were higher than the 10-year average for the watershed (1089 mm) in 2019 and much lower in 2020. During the periods in 2019 and 2020 when water data was being collected (April to November in 2019 and June to November in 2020),

859 and 458 mm of precipitation were reported. This resulted in approximately 8.59×10^6 and 4.58×10^6 l/ha of precipitation. In 2019 and 2020, the basin draining to the saturated buffer on the north side of the stream received approximately 51.54×10^6 and 27.48×10^6 total liters of precipitation, respectively, whereas the basin draining to the south saturated buffer received 85.90×10^6 and 45.80×10^6 total liters of precipitation, respectively.

Water table depths varied throughout the monitoring period as measured using both transducers and periodic manual measurements (Fig. 3 and Table 2). The water table was measured continuously in wells W1–W3, and W7–W9 with the well pairs focused on an upgradient and downgradient well within each saturated buffer (Fig. 3). At the south transect, water tables at W1 and W3 wells tracked closely together and averaged approximately 130 and 129 cm below the ground surface, respectively, whereas at the north transect, the upgradient well (W7) was 45 cm higher than the near-stream well (W9) (145 vs 100 cm, respectively). At both transects, the groundwater depths were highly correlated with the surface water stage ($r = 0.64$ to 0.94), with a higher degree of 2-year correlation between the near-stream wells and the stream ($r = 84$) than with the upgradient wells at stream ($r = 0.77$). At both transects, hydraulic heads decreased from the buffer edge to stream, indicating groundwater flow across the buffer. Differences in water table depths between the buffers were also reflected in mean values of manual measurements for all wells collected at the time of sampling (Table 2). When all 12 wells were considered, the mean depth of water was deeper in buffer 1 compared to that in buffer 2 (124 ± 13 compared to 103 ± 17 cm, respectively).

During the spring of 2019, several rain events caused a rise in water table levels, but after July, wells decreased steadily until rain events resumed in late September. In 2020, few significant rain events were recorded at the site and a general decline in groundwater levels was detected with only one significant rain event in mid-September. In most cases, the wells responded quickly to precipitation recharge.

The average daily horizontal hydraulic gradient averaged 0.02 across the four transects but was much higher across the north side buffer (0.03) than the south side (0.01). Based on slug tests, the hydraulic conductivity of the fine-textured

Table 1 Soil properties, texture and nutrient content with depth found in the middle well of each transect. FeMn con. color is the color of redoximorphic concentrations, TC is total carbon, TN is total nitrogen, C:N is the ratio of total carbon to total nitrogen, and SOM is soil organic matter

Well	Horizon	Lower depth (cm)	Matrix color	FeMn con. color	Sand (%)	Silt (%)	Clay (%)	TC (%)	TN (%)	C:N	SOM (%)
W2	Ap	18	10YR 2/2	-	5.4	43.8	50.8	2.06	0.19	11	6.0
W2	A	43	2.5Y 2.5/1	5YR 4/6	11.5	43.4	45.1	1.37	0.10	14	5.6
W2	C1	64	10YR 2/1	-	6.4	49.9	43.7	1.63	0.15	11	5.3
W2	C2	86	10YR 2/1	-	6.1	52.0	41.9	1.81	0.12	16	5.5
W2	2Ab	103	N 5/1	5YR 2.5/2	10.2	29.0	60.8	1.83	0.20	9	6.5
W2	2Bg1	123+	N 5/1	5YR 2.5/2	18.5	38.3	43.2	0.92	0.04	22	4.5
W2	2Bg2	153	2.5Y 2.5/1	5YR 2.5/2	18.1	39.5	42.4	1.02	0.11	9	4.5
W2	2Cg	184+	2.5Y 5/2	5YR 4/6	25.9	36.3	37.8	0.37	0.01	31	3.1
W5	Ap	18	10YR 3/2	-	3.9	56.0	40.1	2.09	0.17	13	5.8
W5	A	44	10YR 3/2	-	2.0	58.6	39.4	1.71	0.10	17	5.7
W5	C	63	10YR 2/1	-	2.0	53.9	44.1	1.88	0.18	10	5.8
W5	2Ab1	85	2.5Y 2.5/1	-	19.3	39.1	41.7	2.00	0.10	21	7.7
W5	2Ab2	123+	2.5Y 2.5/1	-	20.7	37.7	41.6	1.55	0.05	30	5.9
W5	2Abg	144	2.5Y 2.5/1	5YR 2.5/2	32.0	55.1	12.9	0.63	0.01	48	4.5
W5	2Cg	161+	2.5Y 5/2	5YR 2.5/2	28.0	32.7	39.2	0.31	0.00	104	3.6
W8	Ap	21	10YR 2/2	-	10.0	46.7	43.3	2.03	0.14	15	6.2
W8	A	34	10YR 2/2	-	4.6	57.2	38.1	1.74	0.12	14	5.5
W8	AB	61	10YR 3/2	-	5.4	60.3	34.3	1.60	0.12	14	5.9
W8	Bt	107	10YR 2/2	-	5.8	54.4	39.7	1.98	0.18	11	4.8
W8	BC	153	10YR 2/2	2.5YR 2.5/4	5.4	53.1	41.5	2.02	0.37	5	4.9
W8	2Cg	201	2.5Y 5/2	7.5YR 4/6	13.4	43.4	43.2	0.59	0.17	3	3.2
W11	Ap	20	10YR 2/2	-	7.3	54.1	38.7	1.98	0.27	7	6.4
W11	C1	47	10YR 2/2	5YR 4/6	5.5	53.3	41.2	1.60	0.35	5	5.0
W11	C2	66	10YR 2/1	2.5YR 4/6	3.5	52.8	43.7	1.93	0.24	8	5.2
W11	2Ab	88	N 2.5/1	5YR 4/6	9.7	44.7	45.6	2.34	0.49	5	6.8
W11	2Abg1	122+	10YR 2/1	2.5YR 4/6	16.6	39.4	44.0	1.26	0.58	2	6.9
W11	2Abg2	149	2.5Y 3/1	2.5YR 4/6	17.1	40.9	41.9	0.79	0.12	7	6.0
W11	2Cg	210+	2.5Y 4/1	5YR 4/6	31.7	27.8	40.5	0.20	0.01	28	4.1

Fig. 3 Continuous water table measurements for wells W1, W3, W7, and W9 and stream stage

alluvium was estimated to be 0.03 m/day at both buffers. Assuming a porosity of 0.25, the groundwater flow velocity across the buffer is approximately 0.2 m/day. Considering groundwater flow across a unit 1-m width of the buffer and a depth

of 2 m (equal to the well transect depth), approximately 2.5 m³ of groundwater flows through a 1-m width of the north buffer every year and 0.9 m³ of groundwater flows through a representative section of the south buffer.

Table 2 Summary of water quality analysis results where N is nitrate, DTW is depth to water, DO is dissolved oxygen, ORP is oxidation reduction potential, SC is specific conductance, Temp is temperature, and flow is the rate of flow

Location	N (mg/l)	DTW (cm)	DO (%)	pH	ORP (mV)	SC (μS/cm)	Temp (°C)	Flow (l/min)
W1	0.7±0.6	116±16	3.5±0.9	6.1±0.8	13±82	563±74	15.9±6.3	
W2	0.4±0.2	107±25	3.9±1.5	6.1±0.8	33±59	473±60	14.8±6.8	
W3	0.5±0.3	112±29	3.4±1	6.2±0.8	4±70	459±134	14.5±6.9	
W4	1.4±2.8	141±20	5.5±1.7	6.3±0.9	63±53	470±92	14.9±7.4	
W5	1.1±1.4	138±28	4.3±1.4	6.3±0.9	63±51	433±135	14.2±6.6	
W6	3.5±2.7	128±37	4.1±1.5	6.1±0.9	75±44	455±120	14.0±6.6	
W7	1.5±1.6	133±16	4.2±1.9	6.1±0.9	21±66	528±71	14.9±7.6	
W8	0.8±0.8	94±26	3.9±1.5	6.2±1.0	-2±49	685±124	14.6±7.4	
W9	1.7±2.2	91±46	3.6±1.4	6.3±1.0	3±58	612±67	14.5±7.3	
W10	0.9±1.2	110±18	3.8±1.3	6.5±1.0	20±62	636±89	14.4±6.7	
W11	4.2±3.1	109±22	4.4±1.6	6.4±0.9	31±57	607±166	14.8±6.9	
W12	2.4±3.7	82±41	3.4±1.5	6.3±1.0	-51±84	900±227	14.4±6.9	
NW tile	15.1±4.5		8.1±0.7	6.4±0.9	72±55	594±88	14.7±8.9	68±60
NE tile	12.0±5.0		9.4±1.1	6.8±0.8	61±36	656±79	13.6±8.8	69±45
SW tile	15.6±6.0		7.8±1.4	6.2±0.7	72±57	590±45	14.7±8.4	73±100
SE tile	15.7±6.1		9.5±0.9	6.7±0.7	50±51	601±20	11.5±8.3	72±41
Surface	11.3±6.5		9.9±1.9	7.0±1.0	46±53	565±77	14.7±8.2	

Water quality

Water samples were collected and analyzed from twelve wells, four drainage tiles, and the stream on 21 occasions in 2019 and 2020 (Table 2 and appendix 1). $\text{NO}_3\text{-N}$ concentrations were significantly higher in the tiles and the stream compared to those in wells throughout the study period. Mean $\text{NO}_3\text{-N}$ levels in wells ranged from 0.4 ± 0.2 to 4.2 ± 3.1 mg/l whereas they ranged from 12 ± 5 to 15.7 ± 6.1 mg/l in the tiles and 11.3 ± 6.5 mg/l in the stream. These analyses include both corn and soybean crops in the 2-year rotation. In 2019, corn was grown next to buffer 2 and next to buffer 1 in 2020. Since trends in $\text{NO}_3\text{-N}$ levels were consistent among the two crop years, both years of data were combined for the current analyses. Overall, $\text{NO}_3\text{-N}$ levels were found to be much lower within the groundwater of the saturated buffer compared to the water discharging from the tiles or in the stream.

Within the four well transects, $\text{NO}_3\text{-N}$ concentrations were higher in north transect wells W10–W12 with several measurement days exceeding 6 mg/l (Fig. 4). In contrast, concentrations were much lower in south transect wells W1–W3 where $\text{NO}_3\text{-N}$ was nearly always less than 1.5 mg/l. Concentrations tended to be much higher in the spring and early summer at all transects and lower in the late summer and fall. Likewise, a similar temporal pattern was evident in concentrations measured in tiles and in the stream (Fig. 4). $\text{NO}_3\text{-N}$ concentrations measured in the two tiles feeding the north and south saturated buffers occasionally exceeded 20 mg/l, values considerably higher than concentrations measured in any of the groundwater monitoring wells. The highest groundwater $\text{NO}_3\text{-N}$ measured in the transects was observed in a middle well in the northern buffer (W11), in close proximity to the saturated buffer tile line (as expected). Interestingly, the middle wells of the other transects did not show noticeable impacts of higher $\text{NO}_3\text{-N}$ from the distribution tile into the saturated buffer.

In-field water analyses were completed immediately upon extraction of water samples (Table 2). The mean temperature ranged from 11.5 ± 8.3 to 15.9 ± 6.3 °C for all water samples over the study period. DO varied only slightly within the wells ranging from 3.4 ± 1.0 to $5.5 \pm 1.7\%$ and was considerably higher in the tiles and stream ranging from 7.8 ± 1.4 to $9.9 \pm 1.9\%$. pH was consistent through the study

site ranging only slightly from 6.1 in wells W1, W2, W6, and W7 to 7.0 in the stream. SC was mostly consistent ranging from 433 ± 135 to 685 ± 124 mS/m except in well W12, where SC averaged 900 ± 227 .

Discussion

Study results showed that saturated buffers installed on both sides of a small stream in eastern Iowa are reducing $\text{NO}_3\text{-N}$ concentrations in tile drainage water. Tile concentrations averaged 15 mg/l at the two control boxes. This concentration was assumed to reflect the input concentration of water into the saturated buffer distribution lines. Based on groundwater monitoring in the buffer at four transects, input tile $\text{NO}_3\text{-N}$ concentrations of 15 mg/l into buffer groundwater were reduced to levels < 1.5 mg/l at the streamside well locations. This reduction in tile $\text{NO}_3\text{-N}$ concentration through the buffer represents $\text{NO}_3\text{-N}$ not directly discharged to the stream. Although water fluxes were not able to be measured and N load reductions were not calculated, the concentration reductions are consistent with monitoring conducted at other central Iowa saturated buffer monitoring sites (Jaynes & Isenhardt, 2019; Jaynes & Isenhardt, 2014). The current study expands the geographical extent of saturated buffer monitoring to a pre-Illinoian glacial landscape region not previously monitored (Southern Iowa Drift Plain) and indicates even greater utility of this practice in the glaciated Midwest beyond the intensely tile-drained regions of the Des Moines Lobe (Schilling et al., 2015a, b).

Much of the $\text{NO}_3\text{-N}$ concentration reductions observed at the transects appeared to occur very rapidly downgradient of the distribution tile. With an input tile concentration of 15 mg/l, the mean $\text{NO}_3\text{-N}$ concentration reduction computed was approximately 0.11 mg/l/m of buffer from the middle well to the near-stream well (distance of 5 m). However, this mean value was greatly influenced by conditions at the W10–W12 transect along the north buffer where the mean concentration reduction was 0.65 mg/l/m of buffer (values were < 0.05 at the other three transects). Overall, three of the four well transects were substantially less than a value of 1.01 ± 0.07 mg/l/m reported by Jaynes & Isenhardt (2014) for a buffer in north-central Iowa but the W10–W12 concentration reduction was similar. The lower values more closely

Fig. 4 Nitrate nitrogen concentrations by time for 2019 and 2020

resemble values reported by Mayer, Reynolds Jr, McCutchen & Canfield (2007) in a meta-analysis of 53 different buffer sites. In the current study, most of the groundwater concentrations measured in the buffer

were very low, so the concentration reductions were correspondingly also low. However, where higher initial concentrations (W10–W12) were observed, N concentration reductions were reliably measured. In

a later publication, Jaynes & Isenhardt (2019) showed that $\text{NO}_3\text{-N}$ decreased approximately 50–60% from the distribution tile to the first set of wells 1.8 m downgradient at a saturated buffer site over 5 years of monitoring. At the saturated buffer site, concentrations decreased approximately 93–97% at well transects W1–W3, W4–W6, and W7–W9 and 72% at well transect W10–W12 from the distribution pipe to the first set of wells located 1.5 m downgradient. These concentration reductions suggest there is a large capacity for N concentration removal in the buffer at the eastern Iowa site.

Consistent with other studies of Iowa saturated buffers (Groh et al., 2019; Jaynes & Isenhardt, 2019; Jaynes & Isenhardt, 2014), denitrification is believed to be primarily responsible for the N concentration reductions. The ER data collected from the site did not indicate the presence of any coarse-textured sediments at a depth that could result in N loss due to downward leakage. Hence, the majority of N is likely transported laterally across the buffer to the stream where interaction occurs with saturated and organic-rich alluvial soils. For $\text{NO}_3\text{-N}$ reduction via denitrification to occur, the requisite environmental conditions are considered to be sufficient organic carbon, anaerobic conditions, and a nitrogen supply (e.g., Burt et al. (1999); Cey et al. (1999); Clément et al. (2002); Kellogg et al. (2010)). The soils characterized at the study site typify a post-settlement landscape in Iowa where extensive soil erosion from agricultural tillage in the uplands has placed a large depositional package on top of the original floodplain soils. Indeed, buried surface horizons were identified beginning from 85 to 103 cm below the ground surface in three of the four soil cores (W2, W5, and W11). This was above the depth of tile installation in the buffers (approximately 1.2 m). The present-day fine-textured alluvial soil contained $5.6 \pm 0.5\%$ SOM and $0.19 \pm 0.09\%$ TN on average (Table 1). Interestingly, the buried soils contained a similar amount of SOM and TN ($5.2 \pm 1.5\%$ SOM and $0.15 \pm 0.18\%$ TN). Based on continuous water table measurements in 2019 (full year), the water table resided in the organic-rich A and B horizons of the post-settlement soils approximately 6–16% of the time near the tile distribution line (W3, W9; Fig. 3). However, considering that the pre-settlement soil surface also provides similar SOM, the water table resided within this buried soil profile approximately 79–89% of the time

(39–70% of the time was perched above the gleyed soil horizons).

Groh et al. (2019) noted that denitrification can be maximized when tile water is brought closer to the soil surface. This is based on a typical soil column where SOM decreases with increasing depth. Since sufficient carbon is necessary for effective denitrification, it is appropriate to raise the water table to areas near the soil surface to include as much of the organic-rich soil profile as possible. However, at the current study site, a significant increase in SOM beneath the post-settlement soil profile and above the tile depth was identified (Fig. 5). Indeed, SOM was well above the minimum requirement of 1.2% (Chandrasoma, Christianson & Christianson, 2019) throughout the saturated buffers. Furthermore, both post-settlement and buried soils were found to have sufficient soil carbon levels to support denitrification as compared to soils with similar textures and soil carbon content from north-central Iowa where conditions were judged to be appropriate (Jaynes & Isenhardt, 2014). During the current study, the farm manager actively monitored and adjusted the depth of water in the saturated buffer to keep the water table as high as possible to maximize denitrification potential while still being able to use the buffer as a path for heavy implement traffic. However, the high SOM content within the buried soil may be sufficient for denitrification without attempting to perch the water table within the upper A horizon, thereby benefiting the manager and reducing the amount of tile gate management. Indeed, even though the average water table depth stayed well below the average depth of the post-settlement soil profile, significant denitrification occurred.

Groundwater DO concentrations were low (< 4 mg/l) but were not severely anaerobic, and redox conditions were < 100 mV. Denitrification primarily occurs in settings with DO concentrations < 2 mg/l (Rivett et al., 2008) and redox conditions < 200 mV (Anderson et al., 2005). Hence, soil and groundwater conditions in the saturated buffers appear to be suitable, but not necessarily ideal for denitrification. Denitrification may be occurring primarily in small microenvironments where DO has been depleted and reducing conditions are maintained (Jacinthe et al., 1998). Groh et al. (2019) noted that saturated buffer soils can often display periods of high and low denitrification potentials at various hotspots and

Fig. 5 Soil profile graph showing average soil organic matter and sand content and range of buried soils

hot moments (McClain et al., 2003; Vidon et al., 2010). The current observations of rapidly decreasing $\text{NO}_3\text{-N}$ concentrations downgradient of the tile distribution line suggest the buffer soils are providing some degree of N processing from denitrification. However, it is well understood that N cycling among various forms occurs in the environment and the processes driving these cycles include mineralization, nitrification, and immobilization (e.g., Booth et al. (2005); Gómez-Rey et al. (2012); Guo et al. (2014)).

Given that $\text{NO}_3\text{-N}$ was rapidly reduced from tile levels (~ 15 mg/l) to < 3 mg/l to the first set of downgradient wells (1.5 m), would there be enough time for denitrification to occur? The groundwater

flow rate in the silt-dominated eastern Iowa saturated buffer (0.2 m/day) is less than flow rates measured in loamy central Iowa buffers (0.4–4.8 m/day; McEachran, Dickey, Rehmann, Groh, Isenhardt, Perez et al. (2020)). Groundwater flow velocities at the buffer site suggest that the groundwater travel time for 1.5 m is approximately 7 days. Groh et al. (2019) reported that the median denitrification rate for a saturated buffer overlain by grass (most similar to the current site) was approximately 9 mg N/m³/day. Assuming that this level of denitrification is occurring in a 1.5-m³ volume of soil between the distribution tile and the first set of wells, the median rate could potentially result in denitrification removing

approximately 13.5 mg N/day. Since the groundwater flow velocity could potentially yield approximately 4.5 mg N/day from the input tile, denitrification could potentially remove the tile N inputs in < 1 day, suggesting that the 7-day travel time would be more than sufficient for significant N reductions.

Although tile discharge data for the current study was not available to calculate N load reductions, it is possible to estimate the reductions based on local watershed data and literature values. The study site is located near the headwaters of the 14,840 ha Duck Creek watershed that drains to the Mississippi River. A stream gauging station located in the watershed (latitude 41° 32' 46", longitude 90° 31' 26" NAD27) was used to estimate daily discharge from the site. Based on measured streamflow data, approximately 359 mm and 330 mm of water were discharged from Duck Creek in 2019 and 2020, respectively. Applying this discharge to the north and south buffer catchments (approximately 6 and 10 ha, respectively) suggests that approximately 20,029 to 36,308 m³ of water discharged from the two areas in 2019 and 2020. Of this amount, research from Iowa suggests that tile flow comprises approximately 50% of the total flow in Iowa rivers (Schilling et al., 2019; Thomas et al., 2016); therefore, the total discharge was reduced by 50% to estimate total tile flow from the two buffer catchments. Jaynes & Isenhardt (2019) further observed, based on 19 site years of data, that tiles draining into saturated buffers typically diverted 21 to 94% of total tile flow into buffer tile lines and averaged 50% of total tile flow diverted. Hence, tile flow was reduced by 50% to estimate the amount of the water that may have flowed through the saturated buffers. If an average tile concentration of 15 mg/l was applied to the tile flow through the saturated buffer, it may be estimated that the two saturated buffers processed approximately 82 to 136 kg of N in 2019 and 75 to 125 kg of N in 2020. These N reductions are in the middle of the range of N reductions reported by Jaynes & Isenhardt (2019) (13–179 kg N) for other saturated buffer sites in Iowa. Further, based on the watershed areas contributing to the saturated buffer sites, the estimated load reductions were 12.3 to 13.5 kg N/ha. These load reductions are slightly higher than reductions measured by Jaynes & Isenhardt (2019) in central Iowa (3.2–11 kg N/ha). Overall, based on site estimates, the N load reductions observed in eastern Iowa during this study were

consistent with data reported from other saturated buffer sites in central Iowa.

Lastly, the current study evaluated the performance of a saturated buffer system installed by a landowner/producer who was interested in reducing the export of NO₃-N from field tiles to the local stream. The willingness of the landowner to adopt the practice indicates that saturated buffers are moving beyond research to broader acceptance by producers. Because N removal rates for saturated buffers (\$2.94/kg; Jaynes & Isenhardt, (2019)) are similar to other edge-of-field practices such as bioreactors (\$2.10/kg), controlled drainage (\$2.00/kg), and denitrification wetlands (\$2.90/kg) (Christianson et al., 2013), the relatively simple and low-maintenance practice is gaining rapid support by the farming community. Although the monitoring design of the saturated buffer used in this study did not facilitate quantification of water yields for N load reductions, a demonstration was made through simple concentration reductions that the buffer system was working as intended. Further, the location of the study site in the Southern Iowa Drift Plain landform region expands the potential range of this practice to areas of Iowa not previously considered. However, Jaynes & Isenhardt (2019) noted that the practice requires certain soil and landscape characteristics to function properly, so it is not necessarily applicable to all locations equally.

Conclusions

Groundwater monitoring was used to quantify tile NO₃-N concentration reductions through two saturated buffers at a site in eastern Iowa. Results showed that saturated buffers installed on both sides of a small stream are reducing NO₃-N concentrations in tile drainage water from input concentrations of approximately 15 mg/l to levels < 1.5 mg/l at the streamside well locations. The NO₃-N reduction appears to happen rapidly in the fine-textured and organic-rich alluvial soils, evidenced by low concentrations measured in the first set of downgradient wells located about 1.5 m from the tile distribution pipe. Although direct evidence is not available, denitrification is likely primarily responsible for the concentration reductions based on soil and water chemistry conditions, completion of a geophysical survey (quantifying low

potential for N loss to deeper aquifers), and comparisons to other similar Iowa sites. Overall, the saturated buffer system installed by the farmer-landowner appears to be working as intended. This study shows that the performance of the saturated buffer systems can be monitored relatively simply using concentration-based data, serving to provide more assurance to new adopters that this practice can be installed in many areas of the Midwestern Cornbelt region.

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Availability of data and materials Data may be accessed by contacting the authors.

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