

Potential for saturated riparian buffers to treat tile drainage among 32 watersheds representing Iowa landscapes

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Abstract: The saturated riparian buffer (SRB) is a new and cost-effective conservation practice that diverts agricultural tile drainage toward subsurface discharge within riparian buffers to achieve nitrate ($\text{NO}_3\text{-N}$) removal. Conservation planners want to understand the potential role of the SRB practice for reducing $\text{NO}_3\text{-N}$ loads from tile-drained agricultural watersheds. The Agricultural Conservation Planning Framework (ACPF) includes a tool for identifying riparian zones where the SRB practice can be installed with minimal risks of unintended consequences (i.e., crop inundation and streambank failure). Watershed assessment of the potential role for SRBs, however, must identify where SRB-suited sites can actually receive drainage from tile-drained fields. This study compared the extent of SRB-suited riparian sites among 32 Iowa watersheds, and estimated the proportion of each watershed that was tile drained and located above SRB-suited riparian zones. Results showed the extent of sites suited for SRBs did not significantly differ among three Major Land Resource Areas (MLRAs) in Iowa, from which the selected watersheds were randomly chosen. Most watersheds had suitable sites along 30% to 70% of streambank lengths, where tile drainage from 15% to 40% of the watershed areas could be diverted, based on estimated extents of tile drainage above suitable sites. Therefore, the SRB has an important potential role for water quality improvement in many tile-drained watersheds in Iowa. However, the SRB practice is not readily designed for treating drainage from headwater catchments, which frequently comprised more than 30% of watershed areas in headwater streams of north central Iowa (MLRA 103), where tile drainage is extensive.

Key words: Agricultural Conservation Planning Framework—saturated buffers—tile drainage—watershed analysis

Hydrologic modification, including dams, stream straightening, and artificial drainage, has impacted the hydrology and water quality of many watersheds (Carlisle et al. 2010; Simon and Rinaldi 2006). In the US Midwest, artificial (tile) drainage has enabled agricultural production, yet carries a substantial portion of the nitrate ($\text{NO}_3\text{-N}$) loads found in the upper Mississippi River and its tributaries (Amado et al. 2017). Several practices can be placed to intercept tile drainage and reduce $\text{NO}_3\text{-N}$ loads via denitrification, including nutrient removal wetlands (Hefting et al. 2013) and woodchip bioreactors (Schipper et al. 2010).

Woodchip bioreactors are an edge-of-field practice and are typically installed to treat <40 ha of drainage. Performance of woodchip bioreactors for $\text{NO}_3\text{-N}$ removal is affected by temperature and hydraulic residence time (Hoover et al. 2015). Watershed-wide installation of bioreactors could potentially reduce $\text{NO}_3\text{-N}$ loads in streams by 20% to 30% (Moorman et al. 2015), but excavation and woodchip transportation costs must be considered (Christianson et al. 2013). Nitrate removal rates in wetlands receiving tile drainage often exceed 45% (Kovacic et al. 2000; Lenhart et al. 2016; Groh et al. 2015); a properly sized wetland can treat drainage from

large areas (i.e., hundreds of hectares), but land acquisition comprises a major cost for wetland installation (Christianson et al. 2013).

Riparian soils can also reduce $\text{NO}_3\text{-N}$ in subsurface waters (Mayer et al. 2007), but tile drainage discharge typically bypasses riparian zones via subsurface pipes. However, the saturated riparian buffer (SRB) is a new conservation practice, recently approved for USDA cost sharing, which diverts tile drainage to be treated through riparian-soil processes (Jaynes and Isenhardt 2014, 2019). A gated water level control box, installed along the tile line within the riparian buffer, is used to divert a portion of the tile flow along lateral distribution lines laid parallel to and 10 to 15 m from the stream. The control box raises the water table to within about 30 cm of the surface, causing the diverted tile flow to interact with carbon (C)-rich A horizon soil (topsoil), where denitrification can occur. When tile discharge rates exceed the hydraulic capacity of riparian soils to accept drainage water, discharge can pass through the control box to the original tile outfall along the ditch/stream (Jaynes and Isenhardt 2014). This practice involves small land acquisition and installation costs, compared to wetland and bioreactor practices, and hence are a viable and cost-effective option for tile drainage treatment (Jaynes and Isenhardt 2019). However, not all riparian sites are suited to SRB installation. Optimal $\text{NO}_3\text{-N}$ removal performance should occur where soil characteristics encourage discharged tile water to flow through riparian soils at shallow depth, where soil organic carbon (SOC) is most readily available to enhance denitrification. It is also important to reduce risks of unintended consequences; particularly inun-

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dation of cropland adjacent to the SRB, and slumping of saturated stream/ditch banks. The Agricultural Conservation Planning Framework (ACPF; Tomer et al. 2013b; Porter et al. 2018) ArcGIS toolbox includes an SRB siting tool to identify riparian zones where an SRB can be installed with minimal risk of unintended consequences, by avoiding high streambanks potentially subject to bank failure, and flat fields where the practice could inundate adjacent crops (Porter et al. 2018).

In the context of the US Midwest and efforts to reduce $\text{NO}_3\text{-N}$ loads, data are needed to characterize how sites suited to the SRB practice are distributed relative to the extent of tile drainage in watersheds. The objective of this paper is to determine whether there is an association between extents of tile-drained agricultural land and of riparian sites suited to SRBs among 32 watersheds that represent Iowa landscape regions with varying extents of tile drainage.

Materials and Methods

Watershed Selection and Discretization.

Thirty-two hydrologic unit code (HUC)12 headwater watersheds were randomly selected for analysis, as described and listed by Tomer et al. (2020b) (figure 1). Watersheds were selected to represent three Major Land Resource Areas (MLRAs; Norton 1937; Olmernik and Griffith 2014; USDA NRCS 2006) and four Agro-Hydrologic Landscape classes (AHLs; Schilling et al. 2015). The MLRAs cover about two-thirds of Iowa (figure 1); MLRA 103 is an area of recent glaciation that has limited stream development and extensive cover of agricultural row crops that are artificially drained; MLRA 104 exhibits somewhat older glacial landscapes with greater stream development with similar wide extent of crop cover; and MLRA 108C is an older, more incised landscape with a greater mix of crop, pasture, and hardwood forest land cover. The AHL designations summarize soil drainage and slope classes of dominant soil map units of each watershed, abbreviated as either poorly drained (PD) or well drained (WD), followed by a range of slopes, in percentage (i.e., PD < 2, PD2-5, PD > 5, WD < 5, and WD > 5). These five AHL classes were designated by Schilling et al. (2015) and are meant to link landscape hydrology to conservation practice options for regional and watershed planning. The selected 32 watersheds included four watersheds from each of eight combined

classes of MLRA and AHL designations (figure 1; Tomer et al. 2020b).

The ACPF input databases (Tomer et al. 2017) and 2 m grid digital elevation models (DEMs) were obtained for these 32 watersheds, derived from Light Detection and Ranging (LiDAR) surveys of Iowa (University of Northern Iowa 2016). The DEMs were processed (hydro-modified) to correct overland flow paths where bridges, roads, etc., created “false impoundments” in the DEM (Tomer et al. 2013a), using ACPF tools described by Porter et al. (2018). This process enables the user to make edits (usually cuts through false impoundments) to the DEM, review effects of the edits in correcting flow paths, then adjust the edits in an iterative approach. These edits were made along flow paths with a minimum threshold of 2 ha contributing area. Perennial streams were then designated from among these flow paths by interpreting aerial photography and shaded-relief imagery for each watershed. Land use and soils data were assembled (Tomer et al. 2017) with the edited DEM and stream designation data to complete an ACPF input database for each watershed. The extent of tile drainage in each watershed was estimated as all agricultural fields that were dominated (>90%) by low (<5%) slopes, and/or were substantially covered (>40%) by dual soil hydrologic groups, e.g., B/D (Porter et al. 2018).

Land areas contributing to riparian zones along perennial stream reaches in each watershed were discretized into riparian catchments using a 250 m riparian segment length. In this process, perennial stream reaches are defined from stream initiation points to upper stream confluences, and then successively between stream confluences down to the watershed outlet. Each reach is then divided into that number of equal-length sections of channel that is as close as possible to the selected (250 m) segment length (Porter et al. 2018). There is an adjustment of these sections within each reach to reduce differences in their straight-line lengths, which lengthens the segments where the channel is sinuous and shortens segments that are straight. This step avoids biasing toward delineation of small riparian catchments above sinuous stream lengths. Contributing areas are then defined along each riparian length and split by the channel itself to be delineated as riparian catchments. Headwater catchments that contribute to

stream initiation points are also delineated. Porter et al. (2018) and Tomer et al. (2020a) provide further details.

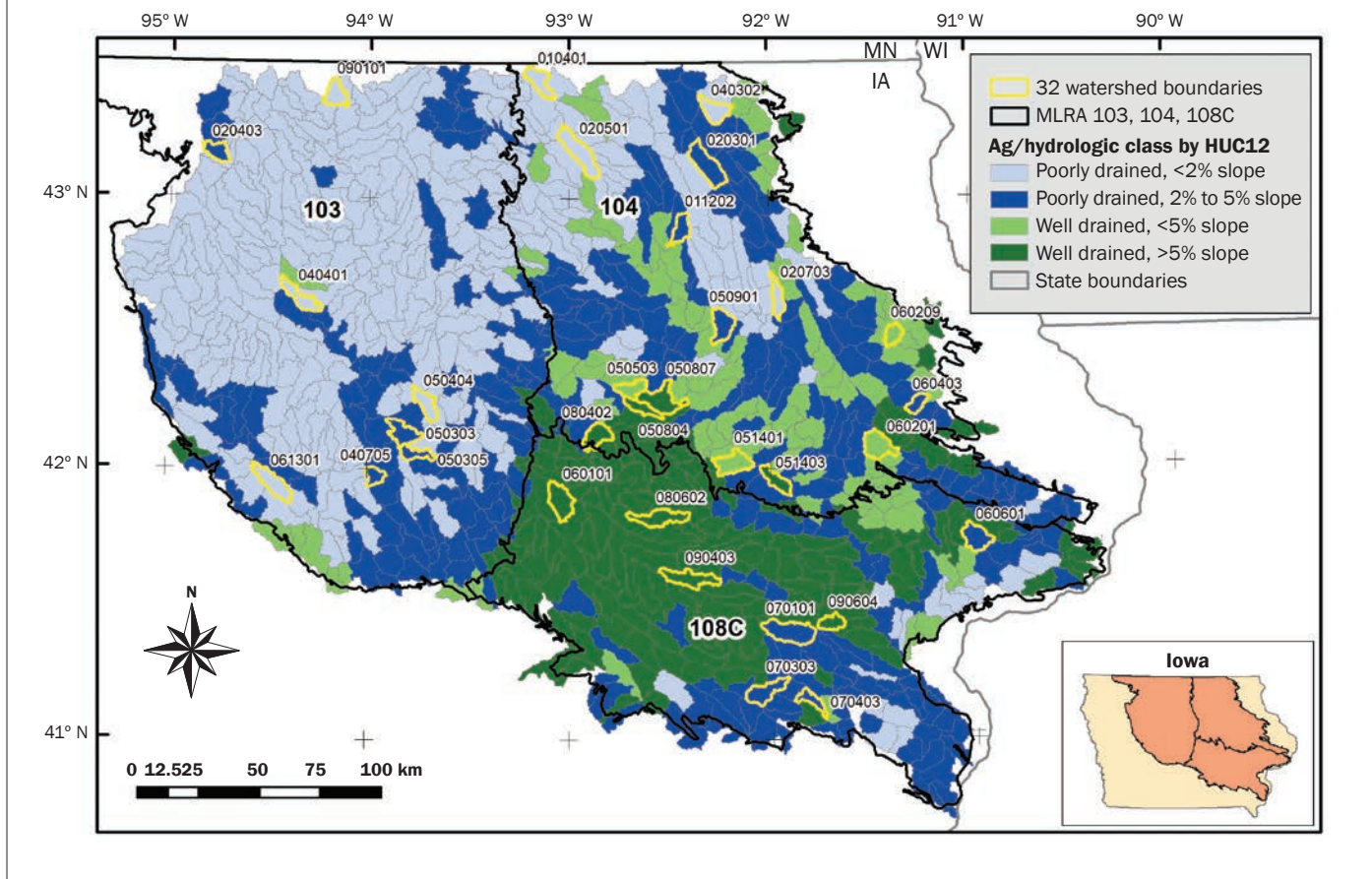
Determining Suitability for Saturated Riparian Buffers.

The ACPF riparian practice tools include an SRB placement tool, which was applied to the 32 selected watersheds and their riparian catchments. In developing the SRB tool, there were considerations to identify locations where an SRB should function effectively to reduce $\text{NO}_3\text{-N}$ from tile drainage discharged into riparian soils, while avoiding sites with potential for unintended consequences of bank failure (due to saturation of high banks) and/or inundation of adjacent crops (Porter et al. 2018). Briefly, $\text{NO}_3\text{-N}$ reduction in riparian soils requires SOC to facilitate denitrification and soil texture and seasonal water table conditions that should encourage shallow, lateral flows toward the stream through saturated riparian soils. The ACPF database includes soils data with derived soil survey information (Tomer et al. 2017) to test whether these soil conditions are likely present in the near-stream environment. To ensure a sufficient C source, the concentration of SOC between 0 and 100 cm depth should average 1.0% (equivalent to 1.7% soil organic matter). To identify riparian sites where discharged tile water should remain at a shallow soil depth, the seasonal water table depth must be less than 1 m, and the sand plus gravel content at 50 to 150 cm cannot exceed 65% in any horizon.

In addition to these soils criteria, topographic criteria were used to check that (1) bank heights were <3.7 m (default) to reduce the risk of stream bank collapse, and (2) “moderate” slopes of 2% to 8% were extensive across the riparian zone (>35% as default). A slope range of 2% to 8% is preferred because if riparian slopes are <2%, an SRB could raise the chance of crop inundation above the buffer, and where those slopes are >8%, there may be a risk of return flows (seepage) across the surface, potentially causing soil erosion within the buffer. In a review of buffer literature, Liu et al. (2008) noted that where riparian soils have >10% slope, buffer design should consider risks of soil erosion within the buffer. The ACPF criteria limiting extent of soils with >8% as an SRB placement criteria recognizes this, but also provides a safety margin considering the enhanced erosion risk involved with controlling for a shallow water table on sloping soils.

Figure 1

Map figure showing three Major Land Resource Areas (MLRAs) and the Agro-Hydrologic Landscape (AHL) designations of HUC12 watersheds found within those MLRAs. Locations of 32 watersheds selected for this study are also shown.



The SRB tool results identify riparian lengths where all these criteria are met, and for those that do not, the reason/s for failure. This emphasizes that ACPF results should be viewed flexibly, and that criteria most limiting for the placement of SRBs will vary among watersheds. The C requirement is an important case in point because there is evidence that riparian buffer vegetation, through rooting activity, can enhance the free C available to drive denitrification (Dosskey et al. 2010; Jaynes and Isenhardt 2019). To allow the SRB practice to be proposed in watersheds where high C soils are not common, sites that only fail the SOC criteria are deemed “suitable with C enhancement” because SOC could be augmented through rooting activity of buffer vegetation, or even by direct addition of C, e.g., using denitrification walls (Schipper et al. 2010). Herein, we report riparian sites that are suited for SRBs with those that are suited with C enhancement, summed together. As we will show, riparian sites that only fail on

the soil C criterion were absent or rare in most of these 32 Iowa watersheds.

Assembling Watershed Data and Statistical Analysis. After classifying riparian catchments by SRB suitability, riparian catchment data were tabulated including the size of each riparian catchment and extent of tile drainage within each riparian catchment (including headwater catchments above stream initiation points). These results were aggregated by watershed to provide the proportion of riparian lengths suited to SRBs, and the extent of tile drainage in riparian catchments that drain to suitable SRB sites (proportion of watershed area). A one-way analysis of variance was conducted to determine if the MLRA-AHL landscape groupings significantly explained variation in the proportion of streambank suited to SRBs among watersheds. Data were log-transformed prior to analysis. If the ANOVA result was significant ($p < 0.05$), contrasts were planned to determine which differences in landscape designation were responsible for the signifi-

cant ANOVA result. A regression was run to determine if the extent of SRB suitability in a watershed could predict the extent of tile-drained land that could benefit from SRB treatment. The SRB practice is not available for tile-drained lands found in headwater catchments because this drainage is discharged to a stream initiation point rather than through a streamside riparian zone. We compared the proportion of all tile drainage in each watershed to the proportion of tile drainage found in headwater catchments. Results indicate how much of the tile drainage in each watershed cannot be treated by an SRB without significantly modifying the practice and/or incorporating other treatment designs (e.g., denitrifying bioreactors, surface or subsurface flow wetlands).

Results and Discussion

Characterization of Watersheds and Riparian Catchments. The 32 watersheds (table 1) varied in size from about 4,200 to nearly 15,000 ha, based on summed areas of the riparian

Table 1
Riparian catchment and saturated riparian buffer (SRB) suitability results for 32 watersheds.

Watershed ID	MLRA-AHL	Area (ha)	Streambank length (km)	Riparian catchments (count)	Headwater catchments (count)	Headwater catchments (ha)	Fraction tile drained	SRB suitability		Most common reason for failure
								Fraction of bank length suited*	Suited with added carbon	
50303	103-PD2-5	9,091	92.2	550	33	5,603	0.873	0.560	0.000	Topography
50305		4,964	63.9	332	14	2,325	0.808	0.766	0.000	Topography
20403		6,363	48.6	224	7	2,478	0.910	0.700	0.000	Topography
40705		4,547	32.4	196	6	2,860	0.758	0.133	0.000	Topography and land use
90101	103-PD < 2	6,957	31.4	136	2	2,209	0.871	0.612	0.000	Topography
40401		8,376	54.3	270	7	2,859	0.917	0.415	0.000	Topography
61301		10,934	63.6	329	20	6,639	0.923	0.592	0.000	Topography
50404		7,081	21.0	104	1	2,218	0.920	0.141	0.000	Soils
60403	104-PD2-5	4,320	78.3	382	29	1,262	0.618	0.240	0.048	Topography and soils
11202		6,962	73.5	376	14	2,234	0.794	0.391	0.000	Soils
50901		9,561	102.9	514	25	3,413	0.768	0.346	0.002	Soils
20301		13,488	142.5	750	19	4,247	0.880	0.499	0.000	Soils
40302	104-PD < 2	8,424	79.7	388	15	2,525	0.935	0.317	0.000	Soils
10401		8,294	58.4	252	6	1,681	0.915	0.015	0.000	Topography and soils
20501		13,861	118.7	577	17	4,382	0.913	0.377	0.003	Topography
20703		6,290	74.5	374	11	1,510	0.863	0.003	0.000	Soils
50804	104-WD > 5	5,567	55.7	258	15	1,891	0.710	0.505	0.009	Topography
51403		6,643	90.3	440	26	2,020	0.871	0.703	0.000	Topography
80402		8,777	93.1	444	25	3,330	0.421	0.709	0.121	Topography
50807		12,330	107.4	494	16	3,082	0.678	0.502	0.012	Topography
60209	104-WD < 5	4,862	62.6	302	22	1,618	0.431	0.000	0.000	Soils
50503		6,482	57.4	294	17	2,442	0.781	0.647	0.000	Topography
51401		9,833	114.2	540	34	3,255	0.901	0.702	0.002	Topography
60201		10,687	104.8	550	31	3,581	0.680	0.458	0.000	Soils
70101	108C-PD2-5	14,952	180.7	886	62	5,561	0.827	0.740	0.025	Topography
70303		9,597	216.6	1,132	117	3,050	0.505	0.311	0.110	Topography and soils
70403		5,489	100.7	558	50	2,027	0.426	0.511	0.217	Topography and soils
60601		9,458	91.1	468	22	3,956	0.568	0.732	0.000	Topography
80602	108C-WD > 5	10,544	174.1	818	75	3,311	0.500	0.513	0.034	Topography
90403		10,760	201.4	958	75	2,694	0.508	0.483	0.016	Topography
90604		5,107	76.8	378	25	1,541	0.530	0.676	0.180	Topography and soils
60101		10,803	153.7	746	40	3,162	0.331	0.432	0.085	Topography

Notes: MLRA = Major Land Resource Area. AHL = Agro-Hydrologic Landscape class. PD = poorly drained. WD = well drained.

*Includes riparian catchments suited to SRBs with carbon addition/enhancement.

and headwater catchments. The combined length of streambanks in these watersheds varied by nearly an order of magnitude, from about 32 to 220 km (note 1 km of stream has 2 km of streambank). Stream order at the watershed outlets varied from first to fourth order, as one watershed in MLRA 103 only had one first-order stream reach, while one watershed in MLRA 108C had 117 first order reaches, with headwater catchments above each initiation point (table 1). The

differences in stream order and streambank lengths also led to a wide number of riparian catchments among the 32 watersheds, from 104 to 1,132.

The extent of tile drainage in each watershed was estimated from the area of agricultural fields that had >90% cover of <5% slopes, or, were >40% covered by soil map units with dual soil hydrologic groups, e.g., B/D (Porter et al. 2018; Tomer et al. 2020b). This query provides an estimate of

the likely maximum extent of tile-drained fields in each watershed. This extent of tile drainage varied from 92% in three watersheds found in MLRA 103 with a PD < 2 AHL class (i.e., dominated by low slopes and poorly drained soils), down to 33% in one watershed in MLRA 108C with a WD > 5 AHL class (i.e., dominated by sloping and well drained soils; table 1).

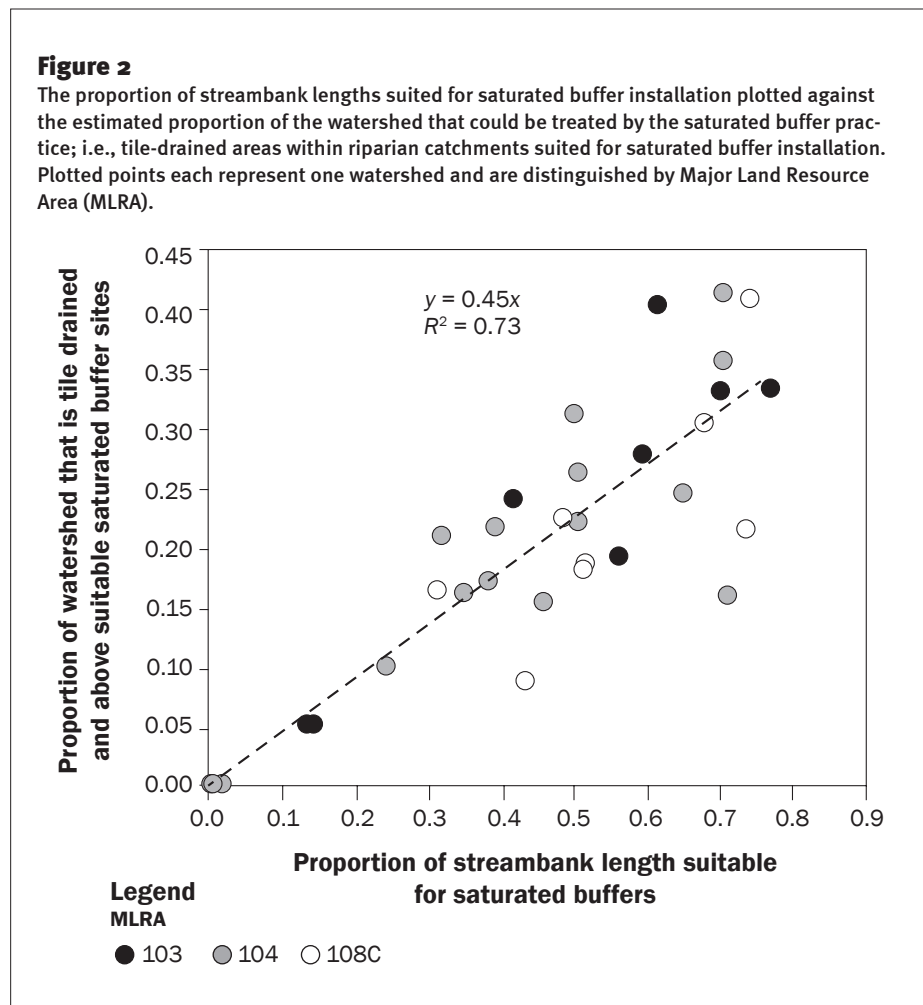
Statistical Results. The proportions of riparian lengths suited to SRB installation

varied from 0% to 74% (table 1) among the 32 watersheds. Note the single zero value was reassigned to 0.001 for log-transformation and statistical analysis. The analysis of variance results showed landscape grouping did not explain significant variation in the proportion of streambanks deemed suitable for SRBs among watersheds ($p = 0.23$), nor in extent of tile-drained land above streambanks suited to SRBs ($p = 0.16$). This suggests regional landscape classifications provide little information for planners seeking to predict where SRBs can most contribute to $\text{NO}_3\text{-N}$ load reductions.

Installation of SRBs will typically be most beneficial where suitable riparian sites and tile-drained cropland are found in the same riparian catchments. We plotted the extent of riparian sites suited to SRBs (proportion of streambank length) against the extent of tile-drained land in riparian catchments above riparian sites suited to SRBs (proportion of watershed area). The resulting plot (figure 2) suggests extent of suitable SRB sites in a watershed is related ($R^2 = 0.73$) to the extent of tile-drained lands that could be readily treated by those SRBs upon installation. However, as the extent of suitable sites increased by watershed, so did the apparent uncertainty around the estimated extent of treatable area (i.e., results appear heteroscedastic). The plot (figure 2) shows that in some watersheds it may be possible to treat tile drainage from 40% of the watershed using SRBs. More typically among these 32 watersheds, suitable SRB sites were indicated on 30% to 70% of streambanks, and areas of tile drainage above the suitable sites covered 15% to 25% of the watersheds.

The most common reason that riparian sites failed the SRB suitability assessment is listed by watershed (table 1). Common reasons for failure included flat topography in MLRA 103 and steep topography in MLRA 108C; a mix of soils and topographic criteria limited site suitability among MLRA 104 watersheds. In three of four watersheds with <15% of SRB-suited riparian lengths, soils criteria were the most common reason for failure.

Riparian lengths suited to SRBs with C enhancement, where SOC averaged <1% to a 1 m depth, were only common in a few watersheds (table 1). That is, they were absent in 18 of the 32 watersheds, and comprised <3% of riparian lengths in 7 others. However, in five watersheds, sites suggested for C



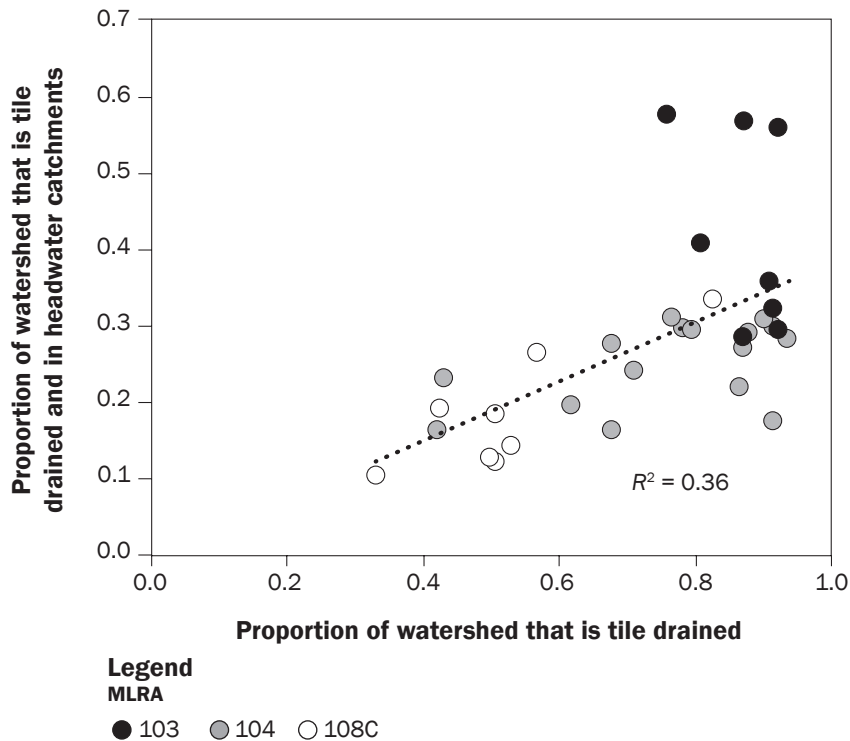
enhanced SRBs were found along 8% to 22% of riparian lengths. These five watersheds were found in MLRA 108C and/or had a WD > 5 AHL designation, i.e., in landscapes with well drained and/or sloping soils where tile drainage is least extensive (table 1). It may be appropriate to include high-biomass riparian species and/or denitrification walls (Schipper et al. 2010) in the SRB design where SOC stores are marginal. However, we note the USDA Natural Resources Conservation Service (NRCS) SRB practice standard has a more lenient SOC criterion than the ACPF tool, requiring only 0.75% SOC averaged to a 0.76 m depth (USDA NRCS 2018), rather than an average 1% SOC to a 1 m depth. The ACPF SRB siting tool was developed in 2016 for ACPF Version 2, two years before the NRCS practice standard was released in 2018. The SRB tool was edited for ACPF version 3; subsurface texture (sand/gravel) and streambank height requirements were relaxed to improve the SRB tool's match with the NRCS practice standard. However, matching the 0.76 m SOC depth would

require a rebuild of ACPF soils databases for over 11,000 watersheds. We emphasize that regardless of the SOC decision point, the C enhancement class of SRB suitability is meant to suggest design and management options to enhance performance, not site qualification or exclusion. Jaynes and Isenhardt (2019) found evidence that soil drainage characteristics and presence of established buffer vegetation had the most influence on SRB performance for $\text{NO}_3\text{-N}$ removal.

The SRB practice cannot readily be applied to treat tile drainage from headwater catchments. The proportion of watershed areas that were tile drained and found in headwater catchments clearly varied among the three MLRAs (figure 3). Watersheds in MLRA 103 typically had >30% of total land area that was tile drained and in headwater catchments, whereas MLRAs 104 and 108C typically had less than 30%. Because these areas drain to stream initiation points rather than through riparian zones, this means the SRB practice would have to be substantially modified to provide denitrification services

Figure 3

Plot showing the extents of tile-drained lands among 32 watersheds (x-axis) plotted against the extents of tile-drained lands that are also within headwater catchments (y-axis); these tile-drained lands do not drain through streamside riparian zones. Result indicates that saturated riparian buffers placed along streams are a more limited treatment option in Major Land Resource Area (MLRA) 103 than MLRAs 104 and 108C.



in headwater areas of MLRA 103. While other denitrification practices, i.e., bioreactors and/or nutrient removal wetlands, will be suited to many of these areas, there will also be a need to continue the creative thinking that led to the development, testing, and availability of the SRB practice, in order to provide a full suite of options to treat tile drainage from many headwater catchments in Iowa.

Summary and Conclusions

This multiwatershed assessment evaluated the extent of SRB-suited riparian sites among riparian catchments delineated for 32 watersheds representing landscape regions dominant in central and eastern Iowa. Sites suitable for SRBs designed to accept tile-drainage discharge into riparian soils for $\text{NO}_3\text{-N}$ reduction were common in most watersheds, but no discernible differences were found among regional landform classes. A small number of watersheds had few suitable SRB sites (<20% of streambank

length), but most had >30% suitable sites into which tile drainage from 15% to 40% of the watersheds could be diverted. The design and management of SRBs in older, more incised, and steeper Iowa landscapes (i.e., watersheds in MLRA 108C and/or with a WD > 5 AHL designation) may need to consider options to increase C availability in riparian zone soils. Results indicate the SRB practice may have a substantial potential role for reducing $\text{NO}_3\text{-N}$ losses for many tile-drained midwestern watersheds. However, tile-drained areas that are in headwater catchments, particularly in MLRA 103, cover a substantial area and will require alternative treatment designs, including but not limited to wetlands, bioreactors, and novel practices suited for implementation near points of stream initiation.

Disclaimer

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