Voluntary Clean Water Guidance for Agriculture Chapters

The purpose of this guidance is to describe best management practices (BMPs) that agricultural producers can use to protect water quality. It is intended to both support healthy farms while helping producers meet clean water standards. The Voluntary Clean Water Guidance introduction[1] provides overall goals and objectives, as well as information on how the guidance will be used. Readers are encouraged to read the overall introduction before this chapter.

This report is available on the Department of Ecology's website at https://apps.ecology.wa.gov/publications/SummaryPages/2010008.html

[1] https://apps.ecology.wa.gov/publications/documents/2010008.pdf

Chapter 8—Draft

Subsurface Drainage Management: Tile Drainage Management, Saturated Riparian Buffers & Blind Inlets

Voluntary Clean Water Guidance for Agriculture

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[2] https://ecology.wa.gov/about-us/accessibility-equity/accessibility

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Introduction

Subsurface drainage (tile) is commonly used on agricultural fields to facilitate crop production and timely equipment access to fields. While improving field conditions for farming, subsurface drainage fundamentally alters how water moves through the landscape and into nearby streams and affects pollutant exports from cropland. In general, subsurface drainage tends to decrease surface runoff because it enhances infiltration but accelerates and increases the volume of water leaving fields by facilitating the dewatering of soil water and shallow groundwater. These shifts alter the way potential pollutants such as fertilizer and other agrochemicals are transported and can also affect streamflow patterns. For example, in watersheds with extensive tile drainage, streams often exhibit more rapid and pronounced responses to rainfall events because water moves more quickly through the system which can lead to more frequent and higher peak flows, increased risk of flooding, streambank erosion and habitat degradation. Subsurface drainage can also significantly increase the transport of soluble pollutants (e.g., fertilizer and other agrochemicals) that leach through the soil profile because they bypass natural filtering processes that occur in surface soil and transport pollutants directly to surface waters.

Since tile discharges are conveyed through a pipe, there are opportunities to treat this drainage before pollutants carried through the system are delivered to surface waters. Controlled tile drainage, saturated buffers and blind inlets are three best management practices that can be used to address water quality challenges associated with subsurface drainage. The goals of these practices are to control the timing and amount of water drained from agricultural fields to match crop and farm operation needs while also limiting excess drainage, reducing pollutant transport and promoting infiltration in lieu discharging subsurface drainage water directly to surface waters.

Scope of Guidance

The purpose of this chapter is to outline best management practices (BMPs) that can be used to control subsurface drainage and treat it before entering surface waters. This guidance is intended to address localized areas where drainage may be needed and to enhance existing subsurface drainage systems. However, it is not intended to promote or provide guidance for the installation of new subsurface (tile) drainage systems.

While these best management practices can help limit pollutants from entering and leaving these drainage systems, they cannot fully eliminate all potential negative impacts associated with tile drainage. Therefore, it's critical that additional measures are taken to reduce the potential for nutrients and other pollutants (e.g., pesticides, herbicides, bacteria from manure) to enter subsurface drainage systems such as carefully managing the timing, source, place and amount of fertilizer and other agrochemicals. Common practices include nutrient management, pesticide management, irrigation management, and cover crops. Information about these and other practices can be found in companion chapters of the Clean Water Guidance for Agriculture.

Definitions as Used in the Guidance

Blind inlet: structures placed in farmed depressions used to drain water that collects in these areas while providing filtration to reduce fine sediments and soluble pollutants.

Controlled tile drainage: system used to manage the timing and amount of water leaving agricultural fields via subsurface (tile) drainage.

Denitrification: a process where nitrite and/or nitrate are converted to gaseous forms of nitrogen including nitrous oxide (N_2O), nitrogen oxides (NO and NO_x) and dinitrogen gas (N_2).

Saturated buffer: a vegetated buffer where a water control structure is used to direct water from a drainage system into a subsurface, perforated pipe that distributes water into the soils beneath and along a vegetative buffer.

Subsurface (tile) drainage: a network of underground pipes (called "tiles") to remove excess water from the soil, especially in fields with poor natural drainage.

Best Management Practices for Subsurface Drainage Management

The following are best management practices (BMPs) that can be used to control the timing and amount of water drained from agricultural fields through subsurface drainage. The goals of these practices are to prevent excess draining of soil water and shallow groundwater, reduce pollutant transport to surface waters via subsurface drainage, and promote infiltration in lieu discharging subsurface drainage directly to surface waters.

Controlled Tile Drainage

Controlled tile drainage (CTD), also known as drainage water management, is a system used to manage the timing and amount of water leaving agricultural fields via subsurface (tile) drainage. CTD works by adding adjustable water control structures (e.g., gates or weirs) to existing subsurface drainage networks to raise or lower the water table to a desired level, holding water in the field during times when drainage is not needed or likely to carry pollutants from the field to surface waters and releasing water when drainage is needed and the risk of pollutant transport is low.

Unlike conventional, free-draining systems that remove excess soil water or groundwater to the installed drain pipe depth, CTD allows water levels to be adjusted for better water retention and storage within the soil profile which can be beneficial for water quality and crops. CTD has been shown to reduce the amount of water drained from fields and limit pollutant loads from tile-drained agriculture, while also providing an opportunity to maintain optimal soil moisture levels. Raising the water table during dry periods or after the growing season and lowering it when fields need to dry out for planting or harvest can reduce water stress, improve crop development, support better nutrient uptake and more consistent yields, and reduce the potential for pollutant transport via subsurface drainage.

CTD is suitable for flat to gently sloping cropland with an existing and functional subsurface drainage system. Field-specific management and schedules for adjusting water levels will vary based on factors such as drainage system design (e.g., depth, spacing, outlet location), soil types, climate, weather, and seasonal crop needs. CTD requires active management to ensure crop needs are met and water quality outcomes are achieved.

Recommendations and Considerations

The following are recommendations for implementing drainage water management.

- CTD should be implemented in conjunction with saturated riparian buffers whenever feasible. Saturated buffers can reduce pollutants transported through subsurface drainage systems and promote infiltration which helps mitigate negative impacts to streams from tile drainage.
- Water control structures must be installed in a manner that allow for water elevations near the soil surface and below typical crop root depths.
- Develop a field, and/or crop-specific management calendar that identifies water control structure outlet elevations that will be maintained during key parts of the year to meet crop needs while limiting excess drainage and the potential pollutant transport.
    - The plan should identify times when outlet levels should be raised and lowered to support crop development and reduce the potential of pollutant loss via subsurface drainage.
    - Factors such as field conditions, weather, soil type, crops and the potential for nutrients and other agrochemicals to reach subsurface drainage systems should be used to establish outlet elevations at different times of the year.
    - The plan should also specify conditions where additional adjustments may be needed, such as significant rainfall events or when irrigation, fertilizer or other agrochemicals are scheduled to be applied.
- Manage water control structure outlet levels year-round according to a management plan.

- Subsurface drainage should not be free draining except when the water table must be lowered for trafficability for field work and crop needs, adverse weather conditions, or system maintenance.
- Unrestricted drainage may be necessary in the spring to allow for field work and trafficability for machinery or livestock.
    - Change the system to free drainage approximately 2 weeks before the planned commencement of field operation.
    - Continue to evaluate the need for unrestricted drainage and raise outlet levels once field operations are complete or once optimal water levels are reached.
- Water levels should be as high as possible during the growing season. Water levels should be set near the root zone of the crop.
- Ensure the water control structure outlet elevation is raised prior to applications of liquid or solid manure, soluble fertilizers or other agrochemicals, or concentrated organic fertilizers.
    - The water level should be approximately 0.5 feet below the upper control elevation or just below the root zone of an actively growing crop.
    - Maintain the raised outlet elevation for at least 15 days following manure, fertilizer or soluble chemical applications or until the next precipitation event that produces drain flow. This will help limit the transport of soluble nutrients or other chemical from reaching drainage pipes through soil infiltration or macropores.
- Set water levels as high as possible after harvest and/or when fields are uncropped or fallow.
    - Raise the outlet within 2 weeks after final field operations are completed following harvest to reduce the potential nutrient leaching.
    - When fields are not cropped or fallow, raise the outlet elevation of the water control structure to within foot or less of the surface.
    - For perennial crops, keep the outlet elevation has high as possible without negatively affecting the crop or soil conditions.
    - In fields with winter cover crops, lower the outlet elevation during the winter to within 0.5 feet of the expected cover crop rooting depth.
- In cold climates, lower the outlet elevation during winter after drain flow has stopped. This will avoid freezing damage to the water control structures. Raise the water to the planned elevation when flow resumes.
- Raising the outlet elevation of subsurface drainage can increase surface runoff. It’s important to use best management practices such as tillage and residual management, nutrient and pesticide management, irrigation management and in-field and edge-of-field practices to reduce erosion, facilitate infiltration, and reduce the potential for polluted runoff.

- CTD can affect crop growth, water availability, nutrient use efficiency, and crop nutrient uptake. The effects of CTD should be considered when developing and implementing nutrient management as it may affect nutrient application rates, timing and placement considerations.
- Consider soliciting planning and design assistance from an engineer or other technical service providers such as those from a local conservation district and/or Natural Resource Conservation Service (NRCS) office.
- Use material and structural specifications outlined in NRCS Practice Standard 554 (Drainage Water Management) when planning and designing a controlled tile drainage system.
- Manage the drainage discharges and water levels in a manner that does not cause adverse impacts to other properties or the drainage system.

Saturated Riparian Buffers

A saturated riparian buffer is a vegetated buffer where a water control structure is used to direct water from a drainage system into a subsurface, perforated pipe that distributes water into the soils beneath and along a vegetative buffer. The primary purposes of saturated riparian buffers are to reduce nutrient and other agrochemical loads from subsurface drain outlets, reduce peak flows, and enhance or restore saturated conditions in riverine, lacustrine fringe, or wetland soils.

Recommendations and Considerations

The following are recommendations for implementing saturated riparian buffers.

- Saturated riparian buffers should meet Ecology's guidelines for riparian buffers as outlined in Chapter 12 of the Clean Water Guidance for Agriculture.
- Saturated buffers should be implemented in conjunction with controlled tile drainage and blind inlets.
    - Water leaving a subsurface drainage system or blind inlet should be routed to a saturated buffer.
    - Additional water control structures may need to be added to the saturated buffer distribution pipe to facilitate infiltration.
    - Water control structures should be configured to allow drainage water to bypass the saturated riparian buffer and discharge through a conventional tile outlet when the flow rate exceeds the hydraulic capacity of the buffer to avoid soil saturation in adjacent cropland areas.
- Adjoining field drainage should be gently sloped (approximately 2%–8%) and slightly above the distribution pipe for the saturated buffer.
    - If the elevation difference is insufficient, the water level within the saturated buffer must be adjusted seasonally to avoid soil saturation with the adjacent field.

- Saturated buffers should be as long as practicable. Length influences the amount of water that can infiltrate into the buffer.
    - The treatment area of saturated buffers may be increased by increasing the length or number of the distribution pipes and/or modifying distribution pipes.
        - Including fewer or smaller openings near the water control structure or incorporating a geotextile or fill can increase the distance water travels down the distribution pipe.
- Underground distribution pipes should be installed parallel to the adjacent surface waterbody and gently slope away from the water control structure.
    - For valleys with higher gradients, the distribution pipe will need to be installed deeper in the upstream direction and shallower in the downstream direction to keep the distribution pipes level.
- Soils within the buffer will affect infiltration rates and the potential to reduce pollutants.
    - Loam or clay loam soil are ideal for infiltration and pollutant removal.
    - Deep soils with high organic matter (e.g., at 1.2%) best support nitrogen denitrification, water storage and attenuation of other nutrients or agrochemicals.
    - Gravelly or sandy soils within the top 4 feet of the surface will promote infiltration but are less suitable for pollutant reductions due to their high permeability.
    - Soils with high clay content (low permeability) may limit when and how much water that can be infiltrated.
    - Soils with a seasonally or a continuously high water table are best suited for nitrogen reductions.
    - There should be no soil voids that allow preferential flow.
- Streambanks should be less than 8 ft. high to prevent bank instability and sloughing.
    - Channels incised more than 8 ft. may not be suitable for a saturated buffer unless a slope stability analysis shows a low risk of saturated streambank failure.
    - Deeply incised channels may need additional restoration such as bank reshaping, riparian planting, and floodplain reconnection prior implementing saturated buffers.
- Saturated buffer configuration may need be adjusted based on property boundaries or physical barriers such as ditches, gullies, or roads that limit distribution pipe installation.
- Woody vegetation should be kept as far from the distribution pipe as possible to prevent possible plugging.
- Saturated riparian buffers must not be used with septic system or animal waste effluent.

- Consider soliciting planning and design assistance from an engineer or other technical service providers such as those from a local conservation district and/or Natural Resource Conservation Service (NRCS) office.
- Use material, design and construction specifications outlined in NRCS Practice Standard 604 (Drainage Water Management) when implementing a saturated buffer.
- The effectiveness of saturated buffers can be improved through complementary practices that reduce or regulate tile drain discharge and the potential for pollutants to enter subsurface drainage systems. Examples of complementary practices include cover crops, nutrient, irrigation and pesticide management.
- Saturated riparian buffers may be less effective at removing certain pollutants from surface runoff in comparison to conventional riparian buffers because saturated soils will have a reduced infiltration capacity. Use best management practices such as tillage and residual management, nutrient and pesticide management, irrigation management, and in-field and edge-of-field practices to reduce erosion, facilitate infiltration, and reduce the potential for polluted runoff.

Blind Inlet

Blind inlets are structures placed in farmed depressions used to drain water that collects in these areas while providing filtration to reduce fine sediments and soluble pollutants. Blind inlets are often implemented to replace tiles risers which are a vertical perforated pipes connected to a subsurface drain network (tile drains). Tile risers are commonly used to drain these depressional areas; however, they provide a direct conduit to the tile drainage system and can convey surface runoff, sediment, and both particulate and dissolved pollutants directly into field drainage without treatment. Blind inlets can be implemented on any cropland or pasture where removal of surface water is necessary or an outlet for a terrace or other water-detention structure is needed.

Recommendations and Considerations

The following are recommendations for implementing blind inlets.

- Use blind inlets in lieu of tile risers. Blind inlets can be installed anywhere a tile riser is recommended and eliminate field obstructions that need to be farmed around.
- Blind inlets can also be used in depressional areas created behind other BMPs such as terraces (NRCS code 600) and Water and Sediment Control Basins (NRCS code 638).
- Blind inlets should be used in conjunction with controlled tile drainage and saturated buffers.
- Blind inlets may not be installed to drain wetlands, seasonal or perennials streams.
- Blind inlets are engineered structures and must be designed to fit each site. Principal considerations include:
    - Sizing: based on precipitation patterns and contributing area.

  - Aggregate selection: materials, mix of fine and coarse materials to provide filtration, choice of amendments if desired.
  - Hydraulics: permeability of filter media, pipe sizing, slope of piping, connection to tile drain system or outlet.
- Use material, design and construction specifications outlined in NRCS Practice Standard 620 (Underground Outlet) or land grant university guidance such as Extension Bulletin E3454[3] when implementing a blind inlets.
- Blind inlets require occasional maintenance to ensure continuing effectiveness. Regular inspection and cleaning are necessary to maintain their performance and prolong their service life.
  - Sediment and vegetation should be periodically removed to a depth that exposes clean gravel/sand at the surface. Any filter materials removed along with the sediment should be replaced to maintain a filter layer of at least one foot thickness.
- Care is required while operating farm equipment near the blind inlet to avoid mixing soil and sediment into the sand filter layer. Operators should minimize driving over the inlet and lift implements while passing over the gravel/sand surface.
- Tillage and soil disturbance in the vicinity of a blind inlet can reduce its lifespan due to soil erosion and sedimentation clogging the filter layer. Use practices such as reduced or no-tillage or filter strips to limit sedimentation of the blind inlet.
- Blind inlets should be used in conjunction with additional field BMPs to increase water quality benefits and decrease the amount and frequency of maintenance, and lengthen design life.
  - Examples of complementary practices include filter strips, cover crops , no-till or other tillage management practices, nutrient management, pest management and irrigation management.
  - Crop management practice that reduces erosion, soil loss and the transport of nutrients or agrochemicals toward a blind outlet will help limit negative impacts to water quality and help maintain the inlet's drainage capacity.

Related NRCS Practices[4]

The following are NRCS practices that directly apply to controlled tile drainage, saturated buffers and blind inlets.

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[3] https://www.canr.msu.edu/field_crops/uploads/files/E3454_BlindInlet_Rev2.pdf
[4] NRCS practice codes are listed in parentheses.

Controlled Tile Drainage

- Drainage Water Management Underground Outlet (554)
- Subsurface Drain (606)
- Structure for Water Control (587)

Saturated Riparian Buffers

- Saturated Buffer (604)
- Structure for Water Control (587)
- Subsurface Drain (606)

Blind Inlet

- Underground Outlet (620)

Commonly Associate Practices

Numerous additional conservation practices can be associated with controlled tile drainage, saturated buffers, and blind outlets including the following NRCS practices.

Controlled Tile Drainage

- Constructed Wetland (656)
- Contour Farming (330)
- Contour Buffer Strips (332)
- Cover Crop (340)
- Denitrifying Bioreactor (605)
- Filter Strip (393)
- Nutrient Management (590)
- Residue and Tillage Management, No Till (329)
- Residue and Tillage Management, Reduced Till (345)
- Residue and Tillage Management, Ridge Till (346)
- Riparian Forest Buffer (391)
- Riparian Herbaceous Buffer (390)
- Saturated Buffer (604)
- Underground Outlet (620)

Saturated Riparian Buffers

- Blind Inlet (Underground Outlet (620))

- Cover Crop (340)
- Critical Area Planting (342)
- Drainage Water Management (554)
- Nutrient Management (590)
- Residue and Tillage Management, No Till (329)
- Residue and Tillage Management, Reduced Till (345)
- Riparian Forest Buffer (391)
- Riparian Herbaceous Cover (390)
- Streambank and Shoreline Protection (Code 580)

Blind Inlets

- Constructed Wetland (656)
- Contour Farming (330)
- Contour Buffer Strips (332)
- Cover Crop (340)
- Denitrifying Bioreactor (605)
- Drainage Water Management (554)
- Filter Strip (393)
- Grassed Waterway (412)
- Nutrient Management (590)
- Residue and Tillage Management, No Till (329)
- Residue and Tillage Management, Reduced Till (345)
- Residue and Tillage Management, Ridge Till (346)
- Riparian Forest Buffer (391)
- Riparian Herbaceous Buffer (390)
- Saturated Buffer (604)
- Subsurface Drain (606)
- Terrace (600)
- Underground Outlet (620)
- Water and Sediment Control Basin (638)

Chapter 8 Appendix Part A: Effectiveness Synthesis Controlled Tile Drain Management

Controlled Tile Drainage

Introduction

Controlled tile drainage (CTD) involves the control of agricultural subsurface drainage systems with adjustable structures to establish a desired water table level. The water table is typically maintained at a higher level than free drainage would allow during critical periods of the growing season, especially during rainfall events. This reduces the rate at which water, and consequently nutrients, pathogens, or agricultural chemicals, are leached from the soil into the groundwater or nearby water bodies. By regulating the water table height, CTD can also minimize the amount of drainage water that leaves the field, reducing pollutant losses. Drainage water management is also referred to as controlled, managed, or conservation drainage.

CTD can be implemented under the Washington NRCS conservation practice standard for Drainage Water Management (554) (NRCS, 2023) which defines drainage water management (DWM) as a process of managing the drainage volume and water table elevation by regulating the flow from a surface or subsurface agricultural drainage system. DWM is based on the premise that the same drainage intensity is not required at all times during the year. The practice is used to:

- Reduce nutrient, pathogen, and pesticide loading from drainage systems into downstream receiving waters.
- Retain water in fields that could be used for crop production later in the season when it is needed.
- Improve productivity, health, and vigor of plants.
- Reduce oxidation of soil organic matter.
- Reduce soil denitrification in saturated soil conditions, saving valuable nutrients needed for winter annual, biennial and perennial crop growth.

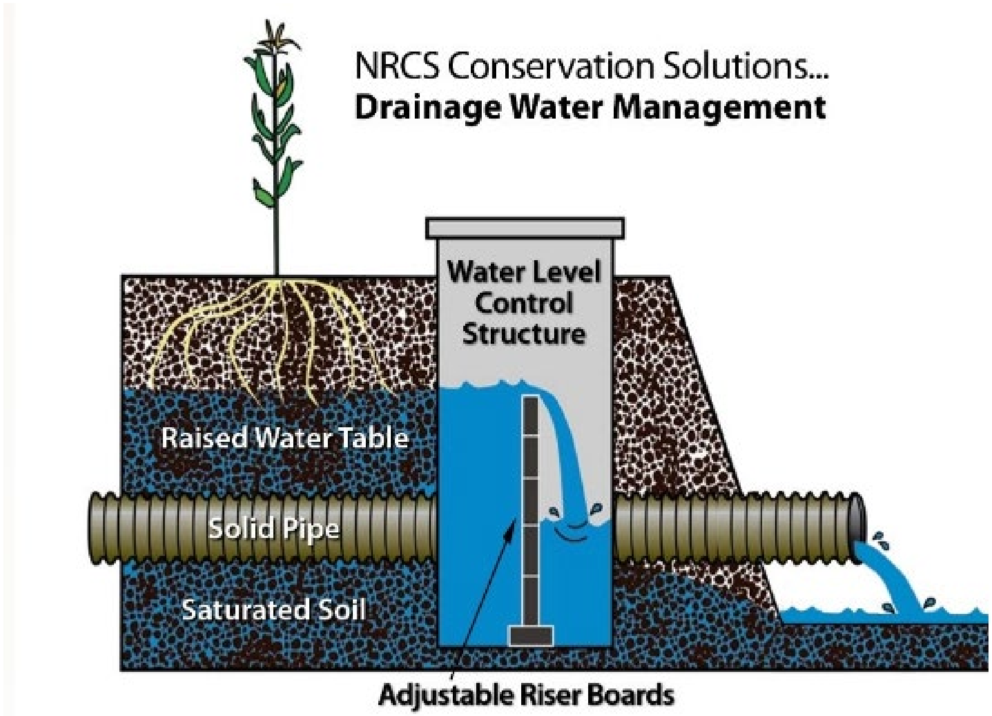

Figure 1. Water level control structure.

- DWM is applicable to flat to gently sloping cropland where a high natural water table exists and where a surface or subsurface drainage system exists that can be adapted to allow management of drainage volume and water table by changing the elevation of water level at the drainage outlet(s). In both surface and subsurface (tile) drainage systems, DWM is generally accomplished through water control structures that function as in-line dams where the discharge elevation can be manually or automatically adjusted (see figure 1).

This evaluation of CTD focuses on the use of water level control structure(s) (NRCS practice code 587) to manage water table elevations and drainage volumes from subsurface drain systems.

Relevance to Washington

Washington's diverse climate, ranging from the wet western side to the drier eastern region, means that water management is crucial for crop production. CTD enables farmers to adjust soil drainage to meet needs associated with planting and plant growth and allow more water to be held in soils outside of the crop growing season to protect the quality of surface waters receiving tile drainage. It may also permit the storage of water in the soil profile during summer that can be accessed by crops, potentially reducing drought stress and increasing crop yields. Although direct research data in Washington are scant, research elsewhere suggests that CTD could reduce nitrogen (N) and phosphorus (P) losses from tile drained cropland throughout the state, by lowering the amount of water released from the soil during times when nutrient leaching is most likely to occur. Research conducted in the Yakima Valley and other agricultural areas of the state has demonstrated that CTD can lead to significant increases in crop productivity. CTD is adaptable to most crops where subsurface drainage is used. Examples of crops known to be associated with subsurface drainage that can benefit from CTD include corn, beans, root vegetables, perennial cool-season grasses, grains and berries.

When and Where Controlled Tile Drainage is Most Appropriate

CTD is applicable to agricultural lands with subsurface agricultural drainage systems that can be adapted to allow management of drainage volume and water table by changing the elevation of water level at the outlets. This practice applies where a high natural water table exists or has existed, soils high in clay content and cation exchange capacity, and the topography is relatively smooth, uniform, and flat to gently sloping. CTD does not apply to the seasonal inundation of fields from overland surface runoff. Also, the practice does not apply to the management of irrigation water supplied through a subsurface drainage system (see other NRCS practice standards for Irrigation System (443) and Irrigation Water Management (449)).

CTD is most appropriate in areas of extensive tile drainage where nitrate (NO_3^-) loads are a concern. For example, CTD is a highly suitable practice for the Puget Sound lowlands, where high precipitation levels during fall, winter, and early spring facilitate NO_3^- leaching and losses to surface water from tile drained fields. Similarly, CTD is highly suitable for fields within lower landscape positions in the Palouse region where tile drainage is used to alleviate soil saturation during the spring resulting from natural drainage of adjacent hilly areas.

Water Quality Concerns Addressed by Controlled Tile Drainage

Uncontrolled tile drainage on cropland is a significant source of pollution in Washington. CTD is typically applied to address losses of dissolved N and P. CTD may also reduce loads of pathogens and pesticides in tile drain discharges. In some cases, enhancement of surface runoff under CTD may increase sediment and sediment-bound pollutant loads to surface waters.

CTD may also indirectly address water quality concerns such as residual soil N and elevated soil-test P by improving crop growth and nutrient use efficiency, reducing the need for additional fertilizer inputs and lowering post-harvest residual nutrient stocks in soils.

Effectiveness

The effects of CTD on tile drain discharge and nutrient losses have been fairly well studied in the eastern United States and Eastern Canada. In contrast, there has been little research on the effectiveness of CTD for reducing loads of other pollutants such as pesticides and pathogens in the more arid regions west of the Mississippi River. In interpreting the results, it is important to recognize differences in climate between Washington and the study locations. For example, in the eastern United States, precipitation is distributed relatively evenly throughout the year, while in Washington, most precipitation occurs between the months of October and May with summers being warm and dry. West of the Cascades, the climate is predominantly marine-influenced Mediterranean and receives abundant rainfall from fall through early spring due to its proximity to the Pacific Ocean. In contrast, Eastern Washington lies in a rain shadow caused by the Cascade Range and is much drier and has characteristics of continental and mediterranean climates depending on location. In this regard, tile drain discharges in Washington, particularly west of the Cascade Mountain Range, may display greater volumes than in the researched systems. The effect of this climatic difference upon the potential effectiveness of CTD in Washington is a source of uncertainty although the processes that influence pollutant reductions are similar.

Hydrologic Changes

By maintaining the water table at higher levels compared to those under free drainage during parts of the year, CTD can exert strong influence on the hydrologic balance of a crop field. Most studies have reported significant reductions in annual or storm event subsurface flow volumes under CTD, although some researchers have noted the potential for CTD to increase surface runoff from drained landscapes (King et al., 2015; Wesström et al., 2014). Hydrologic changes are important because much of reported change in pollutant export under CTD has been attributed to reduction of drain flow volume, although for nitrate reductions, denitrification and other mechanisms can also be important (Frankenberger et al., 2006; Mitchell et al., 2023; Nash et al., 2014; Povilaitis et al., 2018). The range of reported decreases in tile drainage discharge due to CTD is summarized in Table 1.

In an Ontario plot study, Drury et al. (2014) reported that relative to unrestricted drainage, CTD with subirrigation decreased total tile drainage volume by 9%–28%, mainly because controlled drainage increased near surface soil water content. In a review, Ross et al. (2016) reported that CTD was highly effective in reducing drainage water discharge and nutrient losses via drain tiles as tile discharge volumes were reduced by 46%, on average. Greater relative reductions in tile discharge were associated with narrower drain spacing, deeper drain depths, and when tiles were managed during the non-growing season. At multiple field sites in Ohio, Gunn et al. (2015) reported that DWM was effective in reducing daily subsurface drainage volume by 40%–100%. While the magnitude of the daily volume reductions was site-specific, the authors expressed the expectation that if DWM is instituted broadly, mean daily subsurface drainage volume would lessen on an annual basis. In Ontario, Sunohara et al. (2015) noted that CTD imposed widely at a watershed scale can significantly and consistently reduce streamflow; streamflow decreased by 20% under CTD.

Table 1. Decrease in tile drain discharge attributed to drainage water management.

Location	Conditions	Decrease in discharge (%)[1]	Source
Ontario	Corn-soybean rotation	ns	Drury et al. (2014)
Minnesota	Corn	63	Feser et al. (2010)
Ohio	Wheat-corn-soybean rotation	40–100	Gunn et al. (2015)
Midwest/South east	Various	28–49	Helmers et al. (2022)
Various	Various	20–95	King et al. (2015)
Indiana	(crop unspecified)	8–38	Lahdou et al. (2019)
Illinois	No-till corn-soybean rotation	10	Lavaire et al. (2017)
Ohio	Corn	ns	Miller et al. (2022)
Missouri	Continuous corn	52	Nash et al. (2014)
Missouri	Continuous corn	63	Nash et al. (2015)
North Carolina	Corn-wheat-soybean rotation	33	Poole et al. (2018)
Various	Various	46	Ross et al. (2016)
Various	Various	17–40	Skaggs et al. (2012)
Ontario, Canada	Corn-soybean-forage rotation	20	Sunohara et al. (2015)
Quebec, Canada	Corn plots	27	Valero et al. (2007)
Sweden	Potato-barley-oats-wheat	60–95	Wesström & Messing (2007)
Ohio	Corn-soybean rotation	8–34	Williams et al. (2015)

Notes:
[1] "ns" indicates no significant effect.

Increases in surface runoff following CTD have been reported in multiple studies. Drury et al. (2014) reported a 16%–35% increase in surface runoff under controlled drainage attributable to increased soil moisture levels. In Ohio field sites, mean event surface runoff was reported to be greater under CTD, while tile discharge was slightly reduced (King et al., 2022). In a NC modeling study, Deal et al. (1986) reported increased surface runoff under CTD, even as subsurface discharge was decreased. In an extensive literature review, Evans et al. (1995) noted a similar pattern of increased surface runoff with controlled drainage. The authors also reported that the effect of controlled drainage on peak outflow rates varied seasonally. During a dry period or season, drainage control typically reduced peak outflow rates, sometimes totally eliminating outflow for some storm events. During wetter periods, drainage control may have little effect, or in some cases may even increase peak outflows. In another review, Ross et al. (2016) surface runoff was approximately 32%–154% higher under DWM, based on monitored and modeled results.

Drainage water management can affect field hydrology in more ways than total subsurface drainage or surface runoff. From hydrologic monitoring of multiple drainage events in Indiana, Lahdou et al. (2019) reported that controlled drainage reduced event drainage volume and peak flows by an average of ~22% and ~29%, respectively, and increased the time to peak of drainage by approximately 98%. Saturation excess overland flow was more pronounced in controlled fields because water table levels rose higher than the water table levels of their free draining counterpart.

In Ohio, Miller et al. (2022) reported on the impact of automated CTD (ACTD) on groundwater, soil moisture, and tile outlet discharge following storm events. Average groundwater level, soil moisture, and percentage of surface flooding indicated by groundwater levels were not significantly different following precipitation events during periods of ACTD compared to free drainage. However, the time to achieve maximum soil moisture[5] following precipitation onset was significantly longer and lateral hydraulic gradients were greater during periods of free drainage compared to periods of ACTD, which potentially alters how agricultural fields contribute water through edge-of-field buffers and waterways. In addition, maximum discharge occurred significantly later for storm events during ACTD; however, total tile outlet discharge was surprisingly not significantly different for events that happened under ACTD compared to events that occurred during free drainage. ACTD for this study was thus more effective at retaining water in the field and reducing tile outlet to near zero discharge during low-intensity precipitation events and when dry antecedent conditions were present.

Management of DWM is critical. In Poland, Kęsicka et al. (2023) conducted modeling studies that showed the greatest reduction in tile drain outflows obtained at a March 1 start date for controlled drainage, whereas starting controlled drainage on April 1–15 showed much lower or no reduction in tile discharge.

Water Quality Benefits

Nitrogen

Most studies of CTD report significant reductions in N loss, primarily for NO_3^- by reducing leaching from agricultural land under CTD compared to free drainage (Helmers et al., 2022; Mitchell et al., 2023; Ross et al., 2016). Most research has found that CTD does not dramatically change the nitrate concentration in drain flow; nitrate loads are typically reduced by about the same magnitude as drainage flow is reduced (Wesström et al., 2014; Wesström & Messing, 2007; Williams et al., 2015). In contrast, Nash et al. (2014) reported higher NO_3–N concentrations in tile flow under managed drainage (8.1 mg/L, ppm) than under free drainage (5.8 mg/L, ppm); however the effect of CTD on tile discharge was so large in this case that NO_3 –N export in tile flow was still reduced by 29%. The range of reported decreases in N load via leaching due to CTD is summarized in Table 2.

[5] https://www.sciencedirect.com/topics/agricultural-and-biological-sciences/soil-water

Table 2. Changes in leaching N load attributed to drainage water management.

Location	Conditions	Total Nitrogen (TN) Leaching pollutant load reduction (%)[1]	Nitrate nitrogen (NO_3-N) Leaching pollutant load reduction (%)[1]	Source
Manitoba	Corn plots	-	71	Cordeiro et al. (2014)
North Carolina	Modeling	-	34	Deal et al. (1986)
Ontario	Corn-soybean plots	-	38	Drury et al. (2014)
Various	Various	30–50	-	Evans et al. (1995)
Minnesota	Corn	-	61	Feser et al. (2010)
Various	Various	-	15–75	Frankenberger et al. (2006)
Ontario	Manure applied to barley	-	-	Frey et al. (2013)
Midwest, Southeast	Various	-	29–44	Helmers et al. (2022)
Poland	Model calibrated to field data	-	ns–98	Kęsicka et al. (2023)
Ohio	Corn-soybean rotation	-	38–62	King et al. (2022)
Illinois	No-till corn-soybeans	-	ns	Lavaire et al. (2017)
Midwest	Various	-	45	Mitchell et al. (2023)
Missouri	Continuous corn	-	29	Nash et al. (2014)
North Carolina	Corn-wheat-soybean plots	-	14–59	Poole et al. (2018)
Lithuania	Wheat	-	42–77	Povilaitis et al. (2018)
Various	Various	-	48	Ross et al. (2016)
Various	Various	-	-	Skaggs et al. (2012)
Ontario	Corn-soybeans-forage	-	25	Sunohara et al. (2015)
Ontario	Corn-soybeans-forage	-	51	Sunohara et al. (2016)
Sweden	Potato-sugar beet-cereal	-	40	Wesström et al. (2014)
Ohio	Corn-soybean rotation	-	(-8) - 44	Williams et al. (2015)

Notes:

[1] Negative values indicate increase; “ns” indicates no significant effect. “-“ indicates no data reported.

Nitrate reductions from drainage management systems can also result from denitrification within the soil profile and deep seepage, as well as from reduced drainage volume (Frankenberger et al., 2006). Poole et al. (2018) reported an average 30% reduction in NO_3-N losses and attributed part of this reduction to the effect of CTD on increasing yields and the additional N removed in the harvested grain. Wesström and Messing (2007) and Wesström et al. (2014) also found that yield and N uptake by crops in most measurements were higher in CTD due to greater water availability and higher N use efficiency during the growing season. Wesström and Messing (2007) reported that crop yields under CTD were 2%–18% larger and the crop uptake of N increased by 3 kg/ha–14 kg/ha (2.7 lb./ac – 12.6 lb./ac) compared to free drainage.

In an exception to the widely-reported lack of effect of CTD on N concentrations in drainage water, in Ontario, Canada, Drury et al. (2014) reported that CTD with subirrigation reduced NO_3-N concentration by 15%–33% and cumulative NO_3-N loss by 38%–39% relative to free drainage. The authors also reported that when cover cropping and CTD were combined, five-year cumulative NO_3-N concentrations and loss in tile drainage were decreased by 47% relative to no cover crop and unrestricted drainage. Also in Ontario, Sunohara et al. (2015) reported that CTD imposed widely at watershed scales can significantly and consistently reduce N fluxes to stream water; the authors estimated potential decreases in NH_4-N loads of 53% and NO_3-N decreases of 25%.

Much of the variability in the effects of CTD on N leaching shown in Table 2 likely derives from variability in climate, crop, and soil characteristics across the wide geographic range of reported studies. However, the effectiveness of CTD for N (and other pollutant) reduction is dependent on the precise management of the drainage system, e.g., the timing and magnitude of the drainage control relative to season and precipitation events, the specific elevation of the water table, crop need for moisture, and other factors. In a complex modeling study, Kęsicka et al. (2023) evaluated the effects of CTD on hydrology and N and P export under multiple scenarios of climate variation, water table elevation, and drainage timing. For the research site in Poland, the greatest reduction in tile drain outflows and for N and P loads were obtained at a March 1 start date for controlled drainage; the highest reduction close to 100% of NO_3–N loads was shown for the controlled drainage application on March 1 models. For another model, starting the withholding of outflow in mid-March showed NO_3–N load reductions of 85% or more for wet, normal, and dry years. Starting controlled drainage practice in April yielded a much lower reduction or no reduction.

As noted earlier under hydrologic effects of CTD, reduction in subsurface drainage discharge may come with increases in surface runoff. Deal et al. (1986) reported that controlled drainage increased runoff losses of N as a result of increased surface runoff volume. However, the reduced subsurface drainage N export more than offset the increase in N surface losses, resulting in up to 34% reductions in N loss. Frey et al. (2013) and Frey et al. (2016) also found that N concentrations were higher in tile flow from controlled drainage plots, but because of the lower tile discharge, there was no significant difference in N export between controlled and free drainage.

Phosphorus

Many studies also report significant reductions in P loss from agricultural land under CTD compared to free drainage, largely associated with the reductions in subsurface flow volume (Cordeiro et al., 2014; King et al., 2015; Williams et al., 2015). In Ontario, Zhang et al. (2017) reported that controlled drainage reduced dissolved P concentration in tile drain water by 19%. Nash et al. (2015) observed that dissolved P concentration in tile water was significantly lower with managed drainage compared with that of free drainage and dissolved P loss in tile water was reduced with CTD by 80%. The authors noted that the reduced dissolved P loss was not solely due to the reduced amount of water drained annually but also to the lower P concentrations in tile water and speculated that the ability of CTD to conserve water during dry summer months increased crop uptake of water and P, reducing the amount of P available for subsequent leaching loss.

However, some contradictory results have been reported, where either no change or increases in P loss in both subsurface flow and surface runoff occurred under CTD (Frey et al., 2013; Sunohara et al., 2015; Valero et al., 2007). In Quebec, for example, Valero et al. (2007) reported that P concentrations were significantly greater (TP +131% and dissolved P +178%) in controlled drainage plots compared to free draining plots. Increased P loads in tile drainage from controlled drainage plots compared to free drainage plots occurred even though the total outflow volumes from controlled drainage plots were reduced by 27% compared to free drainage plots.

The range of reported decreases in P load via subsurface flow due to CTD is summarized in Table 3. Several researchers have reported on P losses in surface runoff as well. Evans et al. (1995) reported a 42% decrease in TP load in surface runoff from a CTD site. Combined with a 19% decline in TP leaching load, CTD yielded a net 35% decrease in TP export from the field. King et al. (2022) reported greater total P surface runoff losses from CTD sites compared to free drainage sites, suggesting that benefits from CTD with respect to tile P losses could be negated at least in part by increased surface losses.

Much of the difference in behavior of P vs. N under DWM can be attributed to differences in transport pathways and chemistry between the two elements. Whereas NO_3-N is soluble, moves with percolating water through the soil, is subject to denitrification under the right conditions, and is not often transported by surface runoff. While P tends to sorb to soil particles and is more readily transported with eroded sediment in surface runoff, dissolved phosphorus can also be transported via pathways of concentrated or preferential flow such as rills, gullies, soil macropores, artificial drainage channels and through tile lines and bedrock fractures. Further, phosphorus solubility is controlled by complex chemical processes and a variety of soil conditions such as pH, clay content and type of clay, moisture content, and temperature, with pH playing a key role in determining solubility. For example, phosphorus is fixed when pH is greater than 7.5 or lower than 5.5, and acidic soils are often associated with high soil water.

A second explanation for the difference in behavior of N and P under CTD is in biochemistry. While anaerobic conditions promoted by high water tables under CTD can encourage N loss through denitrification, the same anaerobic conditions can promote release of sorbed P from soil. This can lead to increased P concentrations in drainage water, especially in soils with high P saturation. Feser et al. (2010) noted a 33% increase in TP concentration in subsurface flow under controlled drainage. Frey et al. (2013) showed that both total and soluble P concentrations were greater in drainage tile discharge under CTD, but because of lower tile discharge from the controlled drainage plots, there was no significant difference in surface water nutrient loading between the CTD and free draining plots. During the entire monitoring period, total P concentrations were four times higher in effluent from the controlled drainage plots compared with effluent from the free drainage plots, corroborating previous work that has shown that controlled drainage can promote P losses. Mitchell et al. (2023) cautioned that a water quality disservice could occur if the anaerobic conditions generated in saturated soils with controlled drainage lead to pollutant swapping where N is removed but P in the soil is released. It's important to note these studies did not evaluate perennial crops which have different rooting patterns that may or may not affect subsurface P loads, and perennial, long-season crops are likely to have lower tile discharge than annual crops.

Table 3. Changes in subsurface flow P load attributed to drainage water management. Leaching pollutant load reductions.

		Total P (%)[1]	Dissolved P (%)[1]	
Manitoba	Corn	-	69	Cordeiro et al. (2014)
North Carolina	Modeling	26	-	Deal et al. (1986)
Various	Various	19	-	Evans et al. (1995)
Minnesota	Corn	50	63	Feser et al. (2010)
Ontario	Manure applied to barley	ns	ns	Frey et al. (2013)
Various	Various	30–83	-	King et al. (2015)
Various	Various	$\leq$80	-	Kleinman et al. (2015)
Various	Various	50	63	Kröger et al. (2013)
Various	Various	54	56	Mitchell et al. (2023)
Missouri	Continuous corn	-	80	Nash et al. (2015)
Lithuania	Wheat	34–72	-	Povilaitis et al. (2018)
Various	Various	50	57	Ross et al. (2016)
Ontario	Corn-soybeans-forage	18	(-61)	Sunohara et al. (2015)
Ontario	Corn-soybeans-forage	66	66	Sunohara et al. (2016)
Ontario	Corn-soybean plots	12		Tan and Zhang (2011)
Quebec	Corn plots	(-69)	(-98)	Valero et al. (2007)
Sweden	Potato-barley-oats-wheat	60–95	-	Wesström and Messing (2007)
Sweden	Potato-barley-oats-wheat	40	40	Wesström et al. (2014)
Ohio	Corn-soybean rotation	-	40–68	Williams et al. (2015)
Ontario	Corn-soybean rotation	ns	ns	Zhang et al. (2017)

Notes:

[1] Negative values indicate increase; "ns" indicates no significant effect. "-" indicates no data reported.

Where surface runoff increases as subsurface flow decreases under CTD, P transport in surface runoff may be enhanced. Deal et al. (1986) reported that modeled decreases in TP load in subsurface drainage under CTD water were small (26%) and more than offset by a 56% increase in TP loss in surface runoff, yielding a net increase of 21% in total P export. In Ontario, Tan and Zhang (2011) reported moderate reductions in TP loss (12%) from controlled drainage plots; partitioning of P loss between subsurface and surface flows differed between controlled and free drainage. Of the total P loss in free drainage, just 3% to 5% was accounted for in surface runoff water, while 95% to 97% was accounted for in tile drainage water. For controlled drainage, 29% to 35% of the total P loss was in surface runoff water, while 65% to 71% was in tile drainage water. In Ohio, King et al. (2022) reported that both total and dissolved P loss measured in surface runoff was greater under CTD, while no significant differences were noted in tile drainage dissolved P loss. Mitchell et al. (2023) noted potential water quality tradeoffs associated with CTD as flows shift from subsurface to surface paths. Large increases in surface runoff could result in increased losses of dissolved P, soil particles, and associated sorbed materials such as P or pesticides.

Finally, it is worth noting that seasonal performance of CTD may vary due to precipitation and crop growth factors. In Quebec, Valero et al. (2007) reported that most P losses from controlled drainage plots occurred in October due to wet weather and increased drain flow P concentrations after the growing season. P losses from controlled drainage plots in summer were reduced due to reduced outflows. Compared to free drainage plots, TP loads from controlled drainage plots were elevated in spring (April–June, +74%) and fall (October–December, +375%), but were reduced in the summer (July–September, -40%).

Other Water Quality Benefits

There have been few studies reporting on the effects of CTD on pollutants other than N and P. In Ontario, Frey et al. (2013) evaluated movement of *Escherichia coli* and other bacteria from surface manure application to tile discharge. Following manure application on free-drainage plots, bacteria moved rapidly via tiles to surface water; at the controlled drainage plots, tiles did not flow until the first post-manure rainfall, so the immediate risk of manure-induced contamination of surface water was reduced. Bacteria behaved differently from nutrients, with no significant difference in total coliform, *Escherichia coli*, fecal coliform, fecal streptococcus, and *Clostridium perfringens* concentrations between the controlled drainage and free draining tile effluent; however, for all but *C. perfringens*, hourly loading was higher from the free draining plots. Results indicate that controlled drainage has potential for mitigating bacteria movement to surface water.

Wilkes et al. (2014) reported significantly lower occurrences of human, ruminant, and livestock *Bacteroidales* markers in the controlled drainage watershed compared to the free-drainage watershed. There were also significantly lower occurrences of *Salmonella* spp. and *Arcobacter* spp. in the controlled drainage watershed. Overall, the authors suggested that fecal pollution from tile drained fields to streams could be reduced by CTD. Also in Ontario, Sunohara et al. (2016) reported significant reductions in flux of fecal indicator bacteria such as *E. coli* and *Enterococci* due to decreased drainage water discharge under CTD.

In a literature review, Evans et al. (1995) reported that there had been few studies of the effects of CTD on pesticide loss from cropland. The authors cited limited work reporting reduced subsurface transport of Aldicarb under CTD, but an increase in surface transport, leading to a 25% increase in combined Aldicarb transport under controlled drainage. The authors cited other work reporting lower soil-solution concentrations of Prometryn and Atraziine under controlled compared to conventional drainage.

The Washington NRCS conservation practice standard for Drainage Water Management (NRCS, 2023) cites improvement in productivity, health, and vigor of plants and reduced oxidation of organic matter in soils as additional environmental benefits of DWM.

Design and Management Considerations

CTD is an active management practice; the producer must regularly adjust drainage discharges and water levels to balance crop growth and environmental objectives. Kęsicka et al. (2023) modeled numerous alternative CTD management options where CTD was operated to achieve different groundwater table elevations, seasonal timing, etc. Specific management activities and schedules will depend on climate, soil conditions, crop(s) produced, and other factors.

The Washington NRCS conservation practice standard for Drainage Water Management (554) lists several design and management criteria for the practice (NRCS, 2023):

- Manage gravity drainage systems by adjusting the outlet elevations of the water control structures located within the drainage system, either manually or using automation technology.
- Raising the outlet elevation of a water control structure in a flowing drain must result in an elevated free water surface within the soil profile.
- Raise the outlet elevation of the water control structure to within 12 inches or less of the ground surface during non-cropped periods.
- Raise the outlet within 2 weeks after final field operations following harvest.
- Change to free drainage mode no more than 2 weeks before the planned commencement of the next season's field operations, except during system maintenance periods or to provide trafficability when field operations are necessary.
- Minimize drainage below that which is necessary to provide an adequate root zone for the crop.
- Maintain each water control structure outlet in managed drainage mode except when the water table must be lowered for trafficability for field work, adverse weather conditions, or system maintenance.
- Raise the outlet elevation of the water control structure to within 0.5 feet below the control elevation or just below the root zone of an actively growing crop prior to and during applications of liquid or solid manure, soluble fertilizers, or concentrated organic fertilizers in order to prevent direct leakage of manure and limit transport of soluble nutrients into drainage pipes through soil macropores.
- Maintain the raised outlet elevation for at least 15 days following manure or fertilizer application or until the next precipitation event that produces drain flow.
- In cold climates, lower the outlet elevation during winter after drain flow has stopped. This will avoid freezing damage to the water control structures. Raise the water to the planned elevation when flow resumes.
- In fields with winter cover crops, lower the outlet elevation during winter to within 0.5 feet of the expected cover crop rooting depth.
- Develop a management calendar that specifies the target water control structure outlet elevation throughout the year to meet the intended purpose. Adjust the water levels

throughout the year to allow for proper root zone development. Specify conditions where adjusting the outlet elevations may be required, such as significant rainfall events.

- Manage the drainage discharges and water levels in a manner that does not cause adverse impacts to other properties or drainage systems. Lavaire et al. (2017) reported that in some cases CTD on one field can force retained water to exit out of a neighboring free draining tile system, negating the overall effect of CTD at a larger scale.

CTD can be readily integrated with other conservation practices on appropriate fields. Several researchers have studied the combined benefits of cover cropping and CTD. In Ontario, Sunohara et al. (2014) evaluated combined effects of cover cropping and CTD. Cover cropping reduced NO_3-N concentration in tile drainage water by 21%–38% and cumulative NO_3-N loss by 14%–16% relative to free drainage. CTD with subirrigation reduced NO_3-N concentration by 15%–33% and cumulative NO_3-N loss by 38%–39% relative to free drainage. When cover cropping and CTD were combined, NO_3-N concentrations and loss in tile drainage were decreased by 47% relative to no cover crop and unrestricted drainage. Zhang et al. (2017) reported that cover crop alone reduced surface runoff flow volume by 32% relative to free drainage, regardless of drainage water treatment. In contrast, cover cropping increased tile drain flow volume by 57% and 9.4% with CTD and free drainage, respectively, compared to the corresponding CTD treatment without cover crop. The total (surface water + tile drain flow) field water discharge volumes were comparable among all the treatments. Total (surface runoff + tile drain flow) TP loss was the least with CTD + cover crop, followed by free-drainage + cover crop, CTD + no cover crop, and free-drainage + no cover crop. Compared with conventional drainage without cover crop, total TP loss was reduced by 23% with CTD + cover crop.

It is worth noting that CTD may interact with reduced tillage practices in a negative way to the extent that reduced tillage promotes the formation and maintenance of soil macropores. Macropores can rapidly transmit water, dissolved chemicals, and bacteria into soil water and drainage flow, bypassing the filtration properties of the soil matrix. This enhanced transmission of pollutants into groundwater and drainage water may reduce the effectiveness of CTD in pollutant reduction.

In addition to reducing water-driven losses of N and P, CTD can also affect crop growth, water availability, nutrient use efficiency, and nutrient removal by crop uptake. Thus, CTD will potentially interact with the NRCS Nutrient Management (590) practice by altering N and P requirements in fertilizers and/or manure to be applied.

Commonly Associate Practices

Numerous additional conservation practices can be associated with CTD, including the following common NRCS practices:

- Constructed Wetland (656)
- Contour Farming (330)
- Contour Buffer Strips (332)

- Cover Crop (340)
- Denitrifying Bioreactor (605)
- Filter Strip (393)
- Nutrient Management (590)
- Residue and Tillage Management, No Till (329)
- Residue and Tillage Management, Reduced Till (345)
- Residue and Tillage Management, Ridge Till (346)
- Riparian Forest Buffer (391)
- Riparian Herbaceous Buffer (390)
- Saturated Buffer (604)
- Underground Outlet (620)

Conclusions

CTD has been shown to be highly effective in reducing drainage water discharge from drained cropland and thereby reduce fluxes of N, P, and other pollutants from the drained field (King et al., 2015; Kleinman et al., 2015; Ross et al., 2016; Sunohara et al., 2015). In a review of studies conducted across the U.S. Midwest and Southeast, Helmers et al. (2022) found that changes in drain flow and NO_3-N load from subsurface drainage systems under CTD were similar across a large geographical region on an annual basis despite local differences in annual precipitation, soil types, drainage design, and site-specific agronomic management. Although the dominant consensus of the literature, these findings are not universal; some studies have reported little significant effect of CTD on nutrient export from cropland (Frey et al., 2013; Rozemeijer et al., 2016; Valero et al., 2007). Additionally, it is uncertain how CTD effectiveness in Washington may be affected by differences in climate with places where research on CTD has occurred.

Most of the documented nutrient reductions from CTD are attributed to reduced volume of drainage water exported from the system, but nitrate reductions from drainage management systems can also result from denitrification within the elevated water table in the soil profile, and deep seepage (Frankenberger et al., 2006). Reported reductions are most consistent for N and less consistent for P. Some contradictory results for P load changes under CTD may result from desorption of P from highly P-saturated soils under anaerobic conditions in an elevated water table or from increased transport of particulates by surface runoff.

Research has sometimes documented a corresponding increase in surface runoff under CTD, potentially leading to an offset of reductions in drain flow with higher losses in surface runoff, resulting in a net increase in P export, mainly driven by particulate transport in surface runoff. For this reason and because desorption of soil-bound P can be enhanced by the same conditions that promote N loss through denitrification, some researchers have cautioned about water quality tradeoffs between control of N and P under CTD (King et al., 2015; Mitchell et al., 2023).

Alternatives to Agricultural Subsurface Drainage

Introduction

Subsurface drainage has been described as “the most extensive soil and water management activity in agriculture” (Pavelis, 1987). Drainage is considered to be an essential water management practice on many agricultural fields throughout North America, allowing increased crop production and timely equipment access in fields otherwise too wet to effectively farm. Improved drainage enhances crop growth and yield and improves farm operations by increasing the number of days available for on-farm activities (Christianson & Harmel, 2015). The World Resources Institute estimated 39.3 million tile-drained acres occur in eight U.S. corn-belt states, ranging from 3% (Missouri) to 48% (Illinois) of cropland acres underlain by subsurface drainage (Sugg, 2007).

While promoting improved agricultural production, drainage has significant effects on hydrology and pollutant export from cropland. In general, subsurface drainage tends to decrease surface runoff because it enhances infiltration and soil moisture storage, while increasing overall water yield by promoting subsurface flow. King, Williams, Macrae, et al. (2015) reported that tile drainage increases total water yield between 10% and 25% as it increases the proportion of annual precipitation that reaches surface waters via subsurface flow relative to the amount that is stored, evaporated, or transpired. In several studies the authors reviewed, it was suggested that tile drain flow can constitute the majority of stream flow in agricultural watersheds across the Midwestern United States and Canada during some seasons. Schilling et al. (2019) estimated that tile drainage provided roughly 46%–54% of annual discharge to an Iowa river, and during the March to June period, accounted for a majority of flow in the river.

Subsurface drainage tends to substantially increase losses of nitrate N (NO_3-N) and other soluble contaminants that leach through the soil profile. High nitrogen (N) losses (especially NO_3-N) in tile drainage have been extensively documented (David et al., 1997; Keeney & DeLuca, 1993; Lowrance, 1992; Tomer et al., 2003). Significant export of phosphorus (P) in dissolved and/or particulate form can occur via subsurface drainage and that export can be of equal or greater importance compared to loads in surface runoff (Blann et al., 2009; Kleinman et al., 2015). In the mid-Atlantic region, it has been found that P losses in subsurface runoff can be an important component of the total P export from some agricultural watersheds and should be considered in management strategies to minimize nonpoint source pollution of surface waters (Sims et al., 1998). King, Williams and Fausey (2015) showed that tile drainage accounted for 47% of the discharge, 48% of the dissolved P, and 40% of the TP exported from an Ohio watershed.

Given these documented impacts on water quality, it is worthwhile to consider what, if any, alternatives to subsurface drainage might exist.

Alternatives to Subsurface Drainage

Are there practical alternatives to conventional subsurface drainage on agricultural land that continue to support crop production but mitigate the negative effects of conventional drainage on pollutant transport? Based on a global literature review and historical analysis, Shedekar et al. (2020) argued that subsurface drainage will always be part of the solution to the world's food and water security needs and implied that outside of improvements to drainage efficiency and mitigation/treatment of pollutant loads, no real alternatives to agricultural drainage exist. This argument is supported by the fact that an extensive search of the literature revealed few documented alternatives to subsurface drainage that are likely to be practical in Washington. For this analysis, the concept of alternatives has been expanded to include several approaches to modify or manage agricultural drainage:

- True alternatives to subsurface drainage
- Design and management of tile drainage systems
- Land management
- Treatment of tile discharge

Note that two important options for improving the design and management of tile drainage—controlled tile drainage and blind inlets—are evaluated elsewhere in the chapter on field drainage BMPs.

True Alternatives to Subsurface Drainage

In the San Joaquin Valley (CA), Belitz and Phillips (1995) used a simulation model to evaluate the response of the groundwater table to alternative management of recharge and discharge in the regional groundwater system. The authors determined that land retirement was a viable strategy for reducing drain flow and elevated salt concentration but was not effective in sustaining agricultural productivity at the regional scale. The model indicated that regional-scale changes in the water budget (e.g., reducing recharge and/or increasing groundwater pumping) are effective in reducing soil evaporation and drain flow. It should be noted that this analysis is highly site-specific to the San Joaquin Basin and results may not be easily transferrable.

An approach called bio-drainage, where the removal of groundwater via consumptive use by plants through evapotranspiration, has been proposed as an alternative to land drainage in some parts of the world. In India, Chauhan (2000) reviewed the literature on the use of bio-drainage to reclaim agricultural land with high soil salinity and found little evidence for the successful use of bio-drainage. If there is frequent groundwater recharge, it does not appear feasible to lower the water table by planting tree crops and tree crops have not been shown to be successful in removing salts from a saline high water table soil profile. Also in India, Dash et al. (2005) reported that vegetation has been used in many parts of the world to lower water tables either by reducing recharge or by direct extraction of water from the saturated zone. However, the authors did not report quantitative data to document effectiveness of bio-drainage. Finally, Singh and Lal (2018) reported on case studies of successful bio-drainage in India and Australia, including details of implementation and effects. The authors reported that trees such as *Eucalyptus*, *Populus*, *Casuarina*, *Dalbergia*, *Syzigium*, *Acacia*, *Prosopis*, and *Leucaena* were found to be effective in lowering shallow groundwater tables and reversing salinity trends. In practice, trees such as *Eucalyptus* and *Populus* may be integrated not only into an agroforestry cropping system, but also along farm roads and tracks, field borders, and perimeters of ponds.

One currently available best management practice (BMP) that can provide some of the effects of bio-drainage is alley cropping (NRCS Code 311), an agroforestry practice in which crops or forages are produced in alleys between rows of trees or shrubs that supply additional products. Alley cropping systems can significantly reduce NO_3-N and soluble P leaching through nutrient and water uptake (Andrianarisoa et al., 2016; Gikas et al., 2016; Salceda-Gonzalez, 2023), by drying of deeper soil layers by trees and shrubs through evapotranspiration (Allen et al., 2004; O'Connor et al., 2023) and other processes. However, these effects may be limited to the extent to which water uptake by trees and shrubs coincide with periods of high N and P leaching. Refer to chapter 2 for a complete discussion of alley cropping.

Erickson (1988) reported on the ancient practice of raised field agriculture in the Peruvian Andes. In this system, raised fields are constructed by excavating parallel canals and piling the earth between them to form long, low mounds with flat or convex surfaces. These raised platforms increase soil fertility, improve drainage in low-lying areas, and improve local micro-environments, primarily by decreasing frost risk. The canals between raised fields provide vital moisture during dry periods. The author proposed that this ancient technology could be used today but provided no quantitative data to support the idea.

Kramer et al. (2019) published a design and engineering analysis of converting drainage of a Minnesota row crop field to a system that included a two-stage channel, a rock trench, a rock inlet, and two linear treatment systems. While extensive discussion of design and sizing issues was provided, no pollutant removal data were reported.

Finally, from a synthesis of the literature, Parsinejad and Akram (2018) identified several directions for improving draining efficiency and addressing water quality impacts:

- Integrated on-farm drainage management: a system for using saline drainage water for sequential irrigation, resulting in a reduction of drainage water volume and a safe alternative for disposal of drainage water containing salts and nutrients.
- Controlled drainage: raising base-level of drainage outlet yields decrease in overall irrigation requirements, annual drainage volume, and salt, nutrient, and chemical loads.
- Bio- and dry drainage: excess water is removed by consumptive use of vegetation, lowering the water table.
- Treatment systems: practices that slow down detain drainage water in order to trap particulate pollutants and facilitate biochemical removal of dissolved pollutants, e.g., constructed wetlands and bioreactors.

- Optimization of drainage systems: maximizing net benefits of drainage systems by redesigning the drain depth and spacing.

Some of these options are discussed below and others have been addressed elsewhere.

Design and Management of Tile Drainage Systems

The design of subsurface drainage systems, particularly the depth and spacing of drainage lines, exerts a strong influence on the performance of drainage systems. In Ohio, Hoover and Schwab (1969) evaluated the effects of depth, spacing, and cropping practices on drain discharge. Narrow (~10 m) drain spacing resulted in higher discharge than wider (~20 m) tile line spacing; depth of tile did not significantly affect tile discharge. Long-season crops (e.g., perennial grass) resulted in lower tile discharge than crops that grow actively during a small part of the year (e.g., oats). No sediment or nutrient data were reported.

In a literature review, Fausey et al. (1995) cited research results indicating that water from shallow drains (0.4 m,1.3 ft. depth) contained less sediment and nutrients than water from deeper (0.9 m, 3.0 ft. depth) drains and that export of water, nutrients, and pesticides was greater with narrow drain spacing (6 m, 19.7 ft.) compared to wider (12 m, 39.4 ft. and 24 m, 78.7 ft.) drain spacing. The authors recommended that lowering drainage intensity (wider spacing and shallower depth) would reduce nutrient loads and improve drainage water quality.

In a long-term study of spacing impacts on corn yield and drainage water nitrate load, Hoffmann et al. (2004) found that NO_3-N loads were inversely related to tile spacing: six-year mean nitrate loads were 29.9 kg/ha, 26.6 kg/ha, and 22.6 kg/ha (26.9 lb./ac, 23.9 lb./ac and 20.3 lb./ac) for 10 m, 20 m, and 30 m (32.8 ft., 65.6 ft. and 98.4 ft) tile spacing, respectively. The authors noted that while drainage intensity did influence water quality, the differences were minor compared to controllable (e.g., N fertilizer rate) and uncontrollable environmental factors that influence crop productivity and N use efficiency.

Cordeiro and Ranjan (2015) used a calibrated/validated application of DRAINMOD to evaluate design criteria for tile drainage in Manitoba, Canada prairie cropland. Results indicated that the design criteria generally adopted in the province need not be universally used across different soil types such as 0.9 m (29.5 ft.) depth and 15 m (49.2 ft.) spacing. For the sandy loam soils, a maximum drain spacing of 40 m (131.2 ft.) could be used without significant yield losses. The design criteria of 15 m (49.2 ft.) drain spacing currently practiced can cause over-drainage, which could result in yield reduction due to crop water stress during subsequent periods of high water demand.

In Minnesota, Luo et al. (2010) used a model calibrated with plot data to evaluate the impact of alternative drainage practices (0.9 m, 3.0 ft. vs. 1.2 m,3.9ft depth; 9 m,29.5 ft to 24 m,78.7 ft. spacing) on hydrology and water quality. The simulation study indicated that both shallow drainage and controlled drainage may reduce annual drainage discharge and NO_3-N losses by 20%–30%, while impacting crop yields only slightly, from -3% (yield decrease) to 2%, depending on lateral drain spacing. The authors concluded that the practice of increasing drainage intensity (decreasing drain spacing) beyond recommended values appears to not significantly affect crop yield but may substantially increase drainage discharge and NO_3-N losses to surface waters.

Schilling et al. (2012) modeled groundwater travel times and baseflow discharge in a large Iowa watershed where tile drainage networks occupy many first-order drainageways. Varying tile drainage density from 0/m to 0.0038/m, while maintaining a constant tile depth at 1.2 m (3.9 ft.), resulted in the mean groundwater travel time decreasing exponentially from 40 years to 19 years and increased the tile contribution to baseflow from 0% to as much as 37%. Varying tile depths from 0.3 m to 2.7 m (1.0 ft. to 8.9 ft.), while maintaining a constant tile drainage density of 0.0038/m, caused mean groundwater travel times to decrease linearly from 22 years to 18 years and increased the tile contribution to baseflow from 30% to 54% in a near-linear manner. Because NO_3-N concentrations are higher in baseflow than in total streamflow, modified tile drainage practices that result in greater baseflow contribution to streamflow may result in higher NO_3-N concentrations delivered to streams.

Kim et al. (2022) employed field site monitoring, RZWQM2 modeling, and multi-criteria decision analysis to evaluate optimum tile spacing and depth scenarios on Illinois cropland. NO_3-N losses in runoff, seepage, and lateral flow increased as either tile depth or spacing increased. Meanwhile, NO_3-N losses from tile drain flow decreased. The optimum scenario with 0.6 m depth and 36.5 m (2.0 ft. and 119.8 ft.) spacing resulted in the highest decreases (relative to the baseline scenario) in total nitrate load of 11%–14%. The shallower tile depth and wider tile spacing were optimal choices compared to the baseline scenario for reducing N losses. Deeper tiles and narrower tile spacing resulted in decreases in flow and NO_3-N loss from runoff, seepage, and lateral flow, and increases in tile flow, total nitrate loss, and crop yields. The authors suggest that the advanced drainage scheme can simultaneously enhance crop production and environmental sustainability through advances in tile drainage water treatment.

While not functionally distinct from tile drains, mole drains are an inexpensive means to drain excess water from the soil profile and offer modified subsurface drainage performance that may have some benefits for water quality. Mole drains are shallow unlined tunnels in clay subsoil formed by a mole plow composed of a shank (leg) attached to a cylindrical torpedo foot followed by a cylindrical expander (Ghane, 2022). The expander strengthens the channel by compacting the wall. The process of pulling the mole plow results in soil cracks which increase infiltration and water movement by providing a water pathway to reach the mole channel, thereby improving soil aeration. The effectiveness of mole drains depends on the number of soil cracks that facilitate water movement toward the mole channel. Mole drains are cost less to install than tile drains and therefore may be appropriate to use for fields used for low-value crops. Mole drains can be combined with tile drainage to improve drainage performance in heavy clay soil. The lifetime of a mole drain depends on soil stability and soil moisture at mole depth during installation. If properly installed in suitable soil and moisture condition, mole drains can last beyond 5 years, but mole drains usually require restoration every 3 to 4 years as their agronomic effectiveness declines.

In Ireland, Tuohy et al. (2018) used finite-element modeling of a grazed permanent grassland field to compare the performance of a conventional tile system against one incorporating intersecting mole drains. The tile-only system failed to control the water table with consequent loss in crop production and trafficability. The poor performance of the tile-only system showed the limitations of such systems in fine-textured, impermeable soils. The superior performance of combined tile/mole drainage systems reflected the variations in saturated hydraulic conductivity (K_s) of the material above the mole drain channel. The greater the improvement in soil K_s brought about during mole channel installation, the better the long-term system capacity and performance will be. As the cost of mole drainage is low compared to tile drainage, authors recommended that mole drain installation be repeated regularly (i.e., 2–5 year intervals) to maintain channel integrity and optimize system performance.

Okuda et al. (2020) compared the effectiveness of conventional surface + subsurface drainage against subsurface drainage with additional mole drains connected to the main tile lines on addressing salinity problems on cotton cropland in Uzbekistan. The authors reported that the tested system with added mole drains enhanced salt removal from fields. The remaining infiltrated water in the field decreased by ~26% and the removed net mass of salt was 14 mg/ha (2.0×10^{-4} oz/ac). Salt remaining in the field after leaching was 28% less in the test field compared to the control. The direction of salt movement off the drained field changed from the deeper zone or surrounding field to the open drainage.

Regarding mole drains, it should be noted that soil cracking that occurs during installation and is a key feature of their drainage effectiveness, may promote pollutant delivery to subsurface drainage. Preferential flow through soil cracks can lead to rapid transport of sediment, particulate P, and surface-applied nutrients and pathogens from animal waste application to the subsurface drainage system, bypassing the filtering and P buffering capacity of the soil matrix (Christianson et al., 2016; Geohring et al., 2001; Jamieson et al., 2002; King, Williams, & Fausey, 2015; Vidon & Cuadra, 2011).

Finally, two major BMPs for drainage management are available and are discussed elsewhere in this chapter. Controlled tile drainage (i.e., Drainage Water Management NRCS 554) involves the control of subsurface drainage systems with adjustable structures to establish a desired water table level. The water table is typically maintained at a higher level than free drainage would allow during critical periods of the growing season, especially during rainfall events. This reduces the rate at which water, and consequently nutrients, pathogens, or agricultural chemicals, are leached from the soil into the groundwater or nearby water bodies. By regulating the water table, controlled drainage can minimize the amount of drainage water that leaves the field, reducing pollutant losses. See the Controlled Tile Drainage for a complete discussion of the effectiveness of controlled tile drainage.

A second major BMP for reducing the impacts of conventional subsurface drainage involves the use of blind inlets. A blind inlet, also known as a gravel inlet or French drain, is a structure placed in the lowest point of a farmed depression to drain water that collects in the depression. Tile riser, vertical perforated pipe connected to a tile drain network, have commonly been used to drain these depressions. However, tile risers provide a direct conduit to the tile drainage system and can convey surface runoff, sediment, and both particulate and dissolved pollutants directly into field drainage without treatment. Blind inlets are an alternative conservation practice that can drain depressional areas while providing filtration and sometimes treatment that can remove fine sediments and soluble pollutants. See the section below for a complete discussion of the effectiveness of blind inlets.

Land Management

Some limited research has suggested that alternative land/crop management can reduce the water quality impacts of subsurface drainage. In Ohio, Hoover and Schwab (1969) reported that long-season crops (e.g., perennial grasses) resulted in lower tile discharge than crops that grow actively during a small part of the year (e.g., oats). In Minnesota, Oquist et al. (2007) compared N and P losses under conventional practices (corn-soybean rotation, recommended fertilizer and pesticide rates) with losses under alternative practices (organic management, reduced fertilizer/pesticide inputs, increased species biodiversity). Results indicated that the alternative farming practices reduced subsurface drainage discharge by 41% compared with conventional practices. Flow-weighted mean NO_3-N concentrations during tile flow were 8.2 mg/L (ppm) and 17.2 mg/L (ppm) under alternative and conventional farming practices, respectively. Alternative farming practices reduced NO_3-N concentrations in tile flow by approximately 50% and cut NO_3-N losses by 59%–62% compared with conventional practices. Even under conventional agronomic practices, implementing cropping practices like cover cropping (NRCS 340), conservation crop rotation (NRCS 328), nutrient management (NRCS 590), and pest management (NRCS 595) can reduce the amount of nutrients and pesticides available to be leached into subsurface drainage and exported from cropland.

Treatment of Tile Discharge

Probably the simplest approach to accomplish the capture and treatment of tile discharge involves drainage water recycling (also termed tailwater recovery in irrigation systems). Irrigation and drainage tailwater recovery (NRCS practice code 447) is defined as a system to collect, store, and convey irrigation tailwater, rainfall runoff, field drain water, or a combination for reuse in water distribution to the crop (NRCS, 2023). In addition to recycling water and nutrients for crop production, this practice can reduce loads of nutrients and other pollutants from entering downstream waterways. In a review and analysis that included data from Ohio and Missouri crop fields, Frankenberger et al. (2017) reported significant increases in overall crop yields using drainage water recycling and yield increases of up to 29% in dry years. The authors estimated that a drainage water recycling system similar to those studied could reduce downstream nutrient loads by >800 lb. of NO_3-N and 27 lb. of P per year. From monitored plots in Indiana and Iowa, Reinhart et al. (2019) reported that the Indiana reservoir would capture 37% of annual tile drain flow on average, resulting in average annual load reductions of 11 kg/ha/yr, 9.9 lb./ac/yr (37%) for nitrate- N and 0.05 kg/ha/yr, 0.045 lb./ac/yr (39%) for soluble reactive P. The larger Iowa reservoir would capture 23% of annual tile drain flow on average, with average annual load reductions of 9 kg/ha/yr, 8.1 lb./ac/yr (24%) for nitrate-N and 0.02 kg/ha/yr, 0.018 lb./ac/yr (21%) for soluble reactive P.

Because tile discharge represents a concentrated flow of water and pollutants emanating from a pipe, opportunities may exist to treat the effluent before the pollutants are delivered to surface waters. Mendes (2020) published a literature review evaluating factors influencing the performance of constructed and restored wetlands, vegetated buffer strips, and filter materials for P removal from tile discharge. Though lacking in systematic comparison of effectiveness among the various technologies, the review provides analysis of the factors affecting performance, application, and limitations of the edge-of-field technologies.

In Iowa, Jaynes and Isenhart (2014) reported on field-scale testing of the routing of tile drainage through a saturated riparian buffer for nitrate removal. Drainage from a field in corn-soybean rotation was introduced ~0.8 m (2.5 ft.) below the ground surface for 335 m (1099 ft) along the edge of a 20 m (65.6ft) wide grass/forested riparian buffer. Over 2 years, approximately 55% of total tile discharge from the field redirected as subsurface flow through the buffer. The saturated buffer removed 100% of the 228 kg (506 lb.) of NO_3-N introduced from redirected tile drain. The authors attributed the removal to plant uptake, microbial immobilization, and/or denitrification. The effectiveness of saturated buffers is evaluated in detail in an above section.

Denitrifying bioreactors are flow-through filter beds containing carbon-based substrates which can be an effective edge-of-field technology for N removal from tile drainage. Bioreactors remove nitrate by providing a carbon supply and other conditions favorable for microbial denitrification, conversion of NO_3-N into gaseous N. Bioreactors are typically located at the edge of a field at the discharge point from main tile lines. In a literature review, Schipper et al. (2010) summarized different designs for bioreactors and their effectiveness and limitations. The authors reported NO_3-N removal rates of 0.01 g N/m^3/day to 22 g N/m^3/day and that N removal in bioreactors has been supported for up to 15 years without further maintenance. In Quebec, Canada, Husk et al. (2017) reported that under a three-year corn-soybean-wheat rotation, about 55% of the total annual subsurface drainage water passed through bioreactors, which significantly lowered the total N (72%) and nitrate-N (99%) concentrations in the subsurface drainage outflows to drainage ditches. Loading of NO_3-N from the test fields to surface drainage ditches was reduced by 99%, equivalent to about 11 kg NO_3-N/ha/yr (24.4 lb./ac/yr). Some literature reports suggest that bioreactors may also be effective in removing pathogens and pesticides from agricultural drainage.

Several researchers have assessed specific materials of potential use in P sorption/treatment of tile discharge. In laboratory studies, King et al. (2010) observed over 50% dissolved P load reductions in simulated tile drain flow filtered through activated carbon, zeolite, and activated alumina filters. Oliver et al. (2011) reported that water treatment residuals have the capacity to fix large amounts of P under both aerobic and anaerobic conditions and could be used in the field for P management with low risk of release of sorbed P even under anaerobic conditions. In New Zealand, McDowell et al. (2008) recommended using steel slag as a backfill in drains on a dairy farm, reporting reductions of about 70% in P load with subsurface drainage flow through slag fill. Based on the P sorption capacity of the slag, the authors calculated that it would take about 72 years for the slag to become saturated with P.

Considerable research has been reported on the use of several amendments in filtration through blind inlets for removal of nutrients—mainly P—from water entering tile drain systems. As noted in a section below, blind inlets amended with various filter media can be highly effective in removing sediment, P, N, and herbicides from surface runoff routed through tile drainage systems:

- Sediment (TSS) removal by blind inlets has been reported in the 45%–90% range.
- P removal of >80% has been observed from inlets amended with steel slag, wood chips, or biochar.
- N removal efficiencies of 25%–60% have been reported for total N and NH_4-N, but blind inlets may increase NO_3-N loads if nitrification is promoted in the filter media.
- Significant reductions of atrazine, metolachlor, glyphosate, and other herbicides may be achieved through both gravel and amended blind inlets.

Controlled tile drainage (i.e., drainage water management) has been shown to be highly effective in reducing drainage water discharge from drained cropland and thereby reducing fluxes of N, P and other pollutants from the drained field (King, Williams, & Fausey, 2015; Kleinman et al., 2015; Ross et al., 2016; Sunohara et al., 2015). Most of the documented nutrient reductions from drainage water management are attributed to reduced volume of drainage water exported from the system, but nitrate reductions from drainage management systems can also result from denitrification within the elevated water table in the soil profile, and deep seepage (Frankenberger et al., 2006). See sections above for a complete discussion of the effectiveness of controlled tile drainage.

Conclusions

- While promoting improved agricultural production, subsurface drainage tends to decrease surface runoff because it enhances infiltration and soil moisture storage, while increasing overall water yield by promoting subsurface flow. Subsurface drainage also tends to substantially increase losses of soluble N and P contaminants that leach through the soil profile.
- Practical alternatives to subsurface agricultural drainage do not appear to exist currently. True alternatives like bio-drainage have shown little success or practicality for large-scale agriculture.
- Modification of conventional subsurface drainage systems can be effective in reducing tile discharge volume and reducing pollutant loads from drainage.
    - Reducing drainage density - shallower drains with wider spacing can reduce tile discharge and reduce nutrient loads, without impairing crop yields. Deeper tile lines and narrow spacing, in contrast, generally promote increases in tile flow and N loss.

    - Mole drains - because of their shallow depth and soil cracking during installation, mole drains can improve performance of field drainage, either alone or in combination with conventional tile lines. However, the soil cracking associated with mole drains may promote direct delivery of water and pollutants into subsurface drainage by bypassing soil filtration.
- Two major drainage BMPs, controlled tile drainage (drainage water management) and blind inlets, can significantly mitigate the water quality impacts of subsurface drainage. The effectiveness of these BMPs has been discussed in detail elsewhere in this chapter.
- Land management practices, including crop selection and application of cropping BMPs like cover crops, conservation crop rotation, nutrient management, and pest management may help mitigate subsurface impacts on water quality by reducing the amount of nutrients and pesticides available for leaching.
- Because tile discharge represents a concentrated flow of water and pollutants emanating from a pipe, opportunities may exist to treat the drainage effluent before the pollutants are delivered to surface waters. Some options include:
    - Drainage discharge can be routed through structures/BMPs such as drainage water recycling, constructed wetlands, bioreactors, or saturated riparian buffers for N removal.
    - Filter materials such as steel slag or water treatment residuals can be used to sorb dissolved P from tile drainage, possibly in concert with blind inlets.

Saturated Riparian Buffers

Introduction

A saturated riparian buffer (SRB) (NRCS Practice Code 604) is a vegetated buffer in which a water control structure is used to direct water from a drainage system into a subsurface, perforated pipe that distributes water into the soils beneath the buffer (NRCS, 2020). The water control structure is configured to allow drainage water to bypass the SRB and discharge through a conventional tile outlet when the flow rate exceeds the hydraulic capacity of the SRB, allowing the drainage system to avoid soil saturation in a portion of the cropland should water back-up during wet conditions (Jaynes et al., 2018). SRBs are a recently developed practice (first evaluated by Jaynes and Isenhart (2014)). Most research on SRB performance has come from the Midwest, but the practice is applicable to other geographic regions.

Saturated soils have been shown to remove a significant amount of nitrate (NO_3^-) from shallow subsurface water through denitrification, which tends to be high in riparian soils (Anderson et al., 2014b; Billy et al., 2010; Bragan et al., 1997). In New York, Anderson et al. (2014b) measured *in-situ* groundwater denitrification rates on farmland used for crop production (corn and alfalfa) and pasture. Total denitrification from shallow groundwater in the riparian zone was equivalent to 32% of manure N spread on the adjacent upland field. The contribution of riparian soils to the overall mass of NO_3-N removed through denitrification at the farm scale was less than 10%, which is largely attributable to the small percentage (estimated to be <1%) of the landscape acreage occupied by riparian areas (Anderson et al., 2014a).

The primary purposes of a SRB are to (NRCS, 2020):

- Reduce NO_3^- loads from subsurface drain outlets through vegetation uptake and denitrification.
- Enhance or restore saturated conditions in riverine, lacustrine fringe, slope, or depression wetland soils.

Relevance to Washington

SRBs can be implemented on tile-drained cropland in Washington where tile drain outlets discharge directly into ditches or streams.

Although the NRCS practice standard specifies that SRBs only apply to subsurface drainage systems, it is conceivable that the practice could be applied to a surface drainage system (e.g., drainage ditches). For example, it may be feasible to install a water control structure (with inlets capable of filtering out sediment and debris) on a drainage ditch in order to convert the surface drainage into subsurface drainage, which can then be distributed within a vegetated buffer. However, because this application of SRBs has not been implemented and tested, its potential feasibility and effectiveness are unknown.

When and Where Saturated Riparian Buffers are Most Appropriate

According to the NRCS Conservation Practice Standard (NRCS, 2020), there are several site conditions that determine the applicability of the SRB practice:

- SRBs only apply to lands with a subsurface drainage system that can be adapted to discharge into the soils of a riparian buffer.
- The soils and topography of the buffer need to be able to maintain a raised water table without adverse effects to crops, channel banks, shorelines, or adjacent land.
- SRBs generally do not apply to field drainage with underground outlet systems. If the system includes surface inlets, an SRB applies only if the inlets are adequately protected to prevent entry of soil and debris capable of plugging the distribution pipes.
- SRBs are not appropriate for discharging septic system effluent or animal waste.

The NRCS Conservation Practice Standard 604 (NRCS, 2020) identifies additional criteria for the vegetated buffer area that influence SRB applicability:

- Soil conditions, such as a restrictive layer, must be present to create saturated conditions when water is discharged from a subsurface drainage system into a riparian buffer.
- There should be no soil voids or layers of high hydraulic conductivity soil that could provide preferential flow paths.
- A minimum of 0.75% organic carbon (1.2 percent organic matter) should be present in the top 0.8 m (2.5 ft.) of soil.

- There should be no abandoned drainpipes in the buffer area that could continue to drain the buffer.
- The vegetated buffer zone should be at least 9.1 m (30 ft) wide.

SRBs are most appropriate for croplands where tile drainage export of NO_3^- to surface waters is a significant water quality concern. However, as noted above the suitability of an SRB to a specific site is determined by several topographic, soil, and hydrologic characteristics. For example, surface soil layers should have a moderate hydraulic conductivity. Denitrification will be limited by short water retention times where soil layers have a high hydraulic conductivity. Where soils have a very low hydraulic conductivity, denitrification will be limited by a lack of infiltrating drainage water; too much of the drainage water will bypass the SRB. Other characteristics affecting SRB suitability are summarized in the Design and Management Considerations section.

To provide perspective on the watershed-scale applicability of SRBs, Tomer et al. (2018) performed a landscape analysis of the suitability of SRBs in three midwestern HUC-12 watersheds based on proposed criteria for suitable soil and topographic characteristics. These criteria were:

- A soil organic matter content greater than 1% between depths of 0.75 m and 1.2 m (2.5 ft. and 3.9 ft.).
- Relatively fine-textured subsoils,<50% sand between depths off 0.75 m and 1.2 m (2.5 ft. and 3.9 ft.).
- A water table surface less than 1 m (3.3 ft.) below the soil surface from April through June.

The researchers concluded that 30%–60% of streambanks in the watersheds were suitable for SRB installation and that the topographic criteria were more restrictive than soils criteria, particularly in areas with the lowest topographic relief.

In comparison to conventional riparian buffers, SRBs may not be as effective at removing pollutants from surface runoff during periods when tile drainage flow rates are relatively high. This is because pollutant removal from surface runoff is partially driven by runoff infiltration and the saturated soils of SRBs will have a reduced infiltration capacity. Where concurrent removal of pollutants from surface and subsurface drainage is needed, other BMPs such as a constructed wetland, denitrifying bioreactor along with a riparian buffer, or an integrated buffer zone (IBZ) may be more appropriate. For example, in an IBZ, tile drain discharge and surface runoff flow into a pond, allowing particles to settle, and then the water percolates through a filter bed consisting of a shallow inundated vegetated zone before entering a stream (Carstensen et al., 2021).

Water Quality Concerns Addressed by Saturated Riparian Buffers

SRBs are primarily used to reduce NO_3^- delivery to surface waters. However, they may have potential to address dissolved P if buffer soils are not high in soil test P or if their design can be modified to incorporate materials with a high ion-exchange potential (Singh et al., 2020). For example, trenches backfilled with materials having a high ion-exchange capacity (e.g., steel slag or water treatment residuals with a high iron content) can be incorporated into the design to remove P. It is likely that saturated buffers also remove pesticides from subsurface flow since conventional riparian buffers have been demonstrated to do so (Aguiar et al., 2015).

Effectiveness

The SRB practice was first developed in 2010, with first research data from Iowa published by Jaynes and Isenhart (2014). The removal of NO_3^- by SRBs is an active area of research for this relatively new practice. In contrast, there has been very little examination of SRB performance at phosphorus (P) removal, and no studies were located that address removal of pathogens or pesticides.

Ongoing soil saturation in an SRB facilitates the development of anoxic conditions under which denitrification of NO_3^- occurs through microbial processes, mediated by the availability of organic carbon (Carstensen et al., 2020; Fellows et al., 2011). Additional constraints on denitrification rates include nitrate loading, water temperature, and hydraulic residence time (Carstensen et al., 2020; Groh et al., 2019a). N removal in SRBs also occurs through microbial immobilization and uptake by vegetation within the buffer (Carstensen et al., 2020; Jaynes et al., 2018).

Table 4 below displays N reductions reported for studies of SRB effectiveness. P reductions are not included in the table because the single study that addressed P removal (Chandrasoma et al., 2022) found no significant reduction. Summaries of effectiveness studies for individual pollutants follow the table. In evaluating SRB effectiveness, it is important to distinguish between pollutant removal from the drainage water actually entering an SRB and the overall pollutant removal for the entire volume of drainage water (i.e., water entering the SRB plus water bypassing the SRB (Jaynes et al., 2018)). For example, an SRB may have an NO_3-N removal efficiency of 90% but may only have the capacity to treat 20% of the total drainage water volume from a single tile drain outlet, resulting in an overall removal efficiency of 18%.

Table 4. Effectiveness of saturated buffers at reducing pollutant loads in tile drainage.

Location	Study Type	Amount of tile drain flow treated (%)	Nitrate reduction (%)	Source
Iowa	Observational, field scale	55	55[1]	Jaynes and Isenhart (2014)
Iowa	Observational, field scale	21–94	37 (avg.) 8–84[1]	Jaynes and Isenhart (2019a)
Iowa	Observational, field scale		27–96[2]	Groh et al. (2019b)
Iowa	Observational, field scale		≥90[3]	Streeter and Schilling (2021)
Illinois	Observational, field scale	22–80	40–60[1]	Chandrasoma et al. (2022)
South Dakota	Observational, field scale	83–100	67–96[1]	Sharma (2018)

Notes:

[1] The value represents the NO3-N load reduction for the total tile drain discharge (i.e., SRB treated flow + flow bypassing SRBs) rather than only the proportion of the discharge treated by the SRBs.

2 The value represents the NO3-N load reduction for only the proportion of tile outlet discharge infiltrating the SRB.

3 The value represents the observed reduction in NO3-N concentrations; loads could not be calculated due to a lack of flow monitoring.

"-" indicates no data reported

Nitrogen

Most of the effectiveness research on SRBs has been performed in Iowa, where the first published study on SRB effectiveness by Jaynes and Isenhart (2014) was conducted. The authors evaluated the performance of a saturated buffer (20m wide x 336m long) at removing NO_3-N from tile drained cropland used for corn-soybean rotations over a two-year period. Overall, approximately 55% of tile drain flow was redirected into the SRB, which removed 100% of the NO_3-N load, equating to a 55% reduction for the total tile drain discharge.

Jaynes and Isenhart (2019a) expanded their research on SRB effectiveness at removing NO_3^- at six sites installed across Iowa, resulting in a total of 17 site-years of data. NO_3-N removal rates ranged from 8%–84% of the total nitrate load (treated + bypassed). Average annual NO_3^- load removal ranged from 13 kg N to 179 kg N (28.9 lb. to 397.4 lb.) for drainage areas ranging from 3.4 ha to 40.5 ha. The amount of untreated tile drainage (i.e., exceeding the SRB capacity) averaged 6% to 79 % across the six SRBs.

Groh et al. (2019b) evaluated denitrification potential in three SRBs adjacent to tile drained cropland used for corn-soybean rotations in Iowa. The age of the riparian buffers were 21, 16, and less than one year old, with conversion to an SRB through installation of a water control structure and water distribution lines several years prior to this study. The SRBs removed 27%

to 96% of NO_3-N loads from tile drain water, with denitrification accounting for an estimated 33% to more than 100% of the total NO_3-N tile drainage load removed in the buffer; denitrification accounting for more than 100% of the tile drainage loading indicates the presence of additional sources of NO_3^-, such as from surface runoff or plant litter.

Streeter and Schilling (2021) characterized reductions in NO_3-N concentrations for two different saturated buffers at a farm in eastern Iowa. In 2003, riparian buffer strips approximately 20 m (65.6 ft.) wide were installed along the stream. The buffer vegetation consisted mostly of cool-season grasses, harvested for hay. Saturated buffers were installed in 2017 on both sides of the creek, one with a 162 m (531.5 ft.) length and a 10 ha (24.7 ac) drainage area, the other with an 87 m (285.4 ft.) length and a drainage area of roughly 6 ha (14.8 ac). The authors noted that dissolved oxygen levels and redox conditions in the SRB appeared to be suitable, but not ideal for denitrification to occur. Due to the type of water control structure installed for the SRB, tile drainage flow into the buffer could not be monitored; this limited NO_3-N reduction estimates to concentrations rather than loads. The two saturated buffers reduced average NO_3-N concentrations in tile drainage water from input concentrations of approximately 15 mg/L (ppm) to less than 1.5 mg/L (ppm) at the streamside well locations (roughly ≥90% reduction). However, note that because drainage flow could not be measured, the NO_3-N reduction relative to the total tile outlet load could not be directly estimated. They did, however, develop approximate NO_3-N load reduction estimates of 12.3 kg N/ha to 13.5 kg N/ha (27.3 lb./ac to 30.0 lb./ac)) based on the watershed areas contributing to the SRBs and assumptions derived from similar studies of SRBs in Iowa.

In Illinois, Chandrasoma et al. (2022) examined the performance of three SRBs at removal of NO_3^- and dissolved reactive P (DRP) over a cumulative of 10 site-years. The SRBs removed an estimated 54% to 97% of NO_3-N diverted into the buffers, resulting in an average removal of 49% of the overall NO_3-N load leaving the fields (i.e., accounting for loads in treated + bypass water). NO_3-N reductions normalized to field drainage areas ranged from 3.5 kg NO_3–N/ha/yr to 25.2 kg NO_3–N/ha/yr (7.77 lb. to 56.0 lb.) removed annually. DRP concentrations at all sampling locations were low (i.e., median concentrations were at or below the analytical detection limit of 0.01 mg/L, ppm) and were not reduced by the SRBs. Overall NO_3-N removal rates were correlated with the proportion of tile outlet discharge that was diverted into the buffer, ranging from 22%–80%. The authors highlighted an observed inconsistency in flow rates that initiated bypass flows in this study because it directly affects the overall effectiveness of nitrate load removal. They recommended that future research compare the current method of designing SRBs (based on estimated peak tile flow rates) to simpler methods that are used for similar BMPs, for example the design method for constructed wetlands scales the wetland pool to a percentage (0.5% to 2%) of the contributing area.

In South Dakota, Sharma (2018) evaluated SRB effectiveness at two sites with tile drained cropland. At the first site the tile system drained 6 ha, and the SRB was 61 m (200 ft.) long. The SRB at this site had an overall NO_3-N load reduction of 96% for a single year, during which time 100% of drainage water was treated by the buffer. At the second site, the drainage area was 35 ha, and the buffer was 101 m (331 ft.) long. The NO_3-N load reduction over the course of two years at this site was 67%; in the first year 97% of the tile outlet flow was diverted into the buffer, while in the second year 83% was diverted. The reduced effectiveness during year two at the second site was attributed to an increase in drainage volumes relative to year one, resulting in lower NO_3-N uptake by plants and insufficient water residence time for denitrification to occur. Comparing the two sites, the author noted that greater water retention time at the first site resulted in consistent nitrate reduction throughout the cropping season, while variable soil wetting and drying at the second site inhibited denitrification.

Several researchers have evaluated important factors that can influence the effectiveness of SRBs on N removal. Carstensen et al. (2020) attempted to perform a meta-analysis on the effectiveness of SRBs at reducing NO_3-N and total P but found too few studies available for this relatively new practice. In their review of the limited available literature, the authors noted that the effectiveness of SRBs may be limited by either site conditions (e.g., the presence of soils with high permeability) or design factors (e.g., an SRB designed to treat only a small fraction of the drainage discharge). They also noted that greater effectiveness has been found at sites with established perennial vegetation, which may be due to higher carbon inputs to the soil and/or to enhanced immobilization of N by microbial communities.

The net effectiveness of an SRB for N removal largely depends on its hydraulic capacity relative to the discharge from the drainage system. An undersized SRB will treat only a portion of the drainage system discharge when drainage volumes are high. Treatment area can be increased by increasing the length or number of the distribution pipes (Carstensen et al., 2020; Jaynes & Isenhart, 2019b). Where most loading of NO_3-N occurs between autumn and early spring, denitrification rates will be depressed due to low water temperatures (Carstensen et al., 2020). Additionally, Groh et al. (2019b) found evidence that supplies of either NO_3-N or carbon could limit denitrification potential in the SRBs and that NO_3-N removal increases with buffer age.

Denitrification rates tend to be positively correlated with organic carbon levels in the subsoil (Addy et al., 1999). Because organic carbon levels tend to decrease with increasing soil depth, greater denitrification rates tend to occur where the water table level submerges soil layers with greater levels of organic carbon. At the farm scale in New York, Anderson et al. (2014a) estimated that nearly half of the total denitrification flux occurred where the water table was within a few meters of the soil surface, based on whole-farm N balance and the results of a topographic index-denitrification model. Bragan et al. (1997) measured denitrification rates along a riparian transect spanning a field, a grass buffer, and riparian forest. Denitrification rates near the water table surface below the field and at 2 m (6.6 ft.) depth throughout the transect were less than 25% of the highest rates, which were observed near the surface of the water table within the grass buffer and riparian forest (2.8 mg N/L per day and 3.3 mg N/L per day, respectively). These results likely correspond to spatial variation in pools of soil organic carbon.

Some research has examined the influence of SRB width on NO_3-N removal. McEachran et al. (2020) investigated six existing SRBs in central Iowa receiving tile drainage from fields in a corn and soybean rotation (the same sites as in Groh et al. (2019a); Groh et al. (2019b); Jaynes and Isenhart (2019a)), to evaluate their current width relative to their optimal width for NO_3-N removal. The SRBs varied in width, distribution pipe length, drainage area, topography, flow rates, and nitrate removal parameters. To determine optimal SRB width, the authors estimated the distance at which nitrate concentrations dropped to zero in an SRB and then estimated the width that could accommodate more flow without a net decrease in nitrate load removal. For all six SRBs, the calculated optimal width was narrower than the current width; optimal widths ranged from about 4 m to 18 m (13.1 ft. to 59.1 ft.) (± 57% percent). At four of the sites, narrower buffer widths were projected to accommodate more tile drain flow and potentially increase NO_3-N removal by roughly 50%–120%; two of the sites had an optimal width under 9.1 m (29.9 ft.), which is less than the current NRCS specification for practice 604. The authors assert that a tradeoff exists in choosing the SRB width. Water residence time increases with increasing SRB width, allowing more time for nitrate removal. However, a wider SRB has a lower hydraulic gradient, which reduces the proportion of tile outlet discharge that is diverted into the SRB. The study findings suggest that:

- Wider SRBs do not necessarily result in greater nitrate load removal.
- Increasing the width of an SRB decreases the hydraulic gradient and therefore decreases the flow rate through the SRB; this potentially increases the proportion of the nitrate load that will bypass an SRB without treatment.
- The optimal SRB width maximizes nitrate removal by balancing flow through the SRB with water residence time.
- In some cases, the optimal SRB width is narrower than the minimum width in NRCS design specifications.
- The buffer width does not need to be reduced to optimize nitrate removal. Instead, the placement of the water distribution pipes can be adjusted to alter the saturation width within the overall riparian buffer.

However, there appear to be weaknesses in the methodology used by McEachran et al. (2020) to evaluate optimal SRB width. The methods assume that the NO_3-N removal rate is constant throughout the SRB width, although in reality the rate may vary differ due to spatial variability in factors such as organic carbon supply. Furthermore, denitrification rates are likely proportional to NO_3-N concentrations, so as the concentration goes down, denitrification rates might also decline. Lastly, all NO_3-N removal was attributed to denitrification instead of accounting for potential variability in NO_3-N uptake by vegetation and microbial immobilization of N as SRB width is altered. Nevertheless, the study does provide a conceptual foundation for potential optimization of SRB performance by tailoring SRB design to site conditions.

In a follow-up study on the same sites, McEachran et al. (2023) modeled groundwater flow through SRBs to evaluate how spatial variation in flow influences NO_3-N removal. The model indicated that flow path of water toward the stream depended on where it exited the distribution pipe. The results also indicated that when nitrate is not limiting, the potential for nitrate removal depended on the length of the path (and thus travel time) as well as depth. The authors concluded that the potential for N removal is greatest for flow discharged from the top perforations in the distribution pipes and also that SRB performance can be improved by directing more flow towards the ends of the distribution pipes. However, the authors acknowledged that model results should be interpreted with caution because the model neglects heterogeneity in the soil properties, temporal variations in flow, effects of hydrodynamic dispersion and biogeochemical reactions, and plant uptake.

Phosphorus

Utt et al. (2015) evaluated the potential removal of total dissolved P (TDP) in tile drainage by SRBs in Illinois, Indiana, Iowa, and Minnesota. TDP samples were collected at multiple points along transects spanning the width of SRBs. Based on a lack of consistent spatial trends in TDP concentrations along the transects, the authors concluded that saturated buffers cannot be assumed to remove dissolved P. However, aspects of the study methodology are problematic. First, there was no characterization of TDP levels in the riparian buffer prior to conversion to an SRB and it is not uncommon for both crop fields and riparian buffers to contain accumulations of P from historical capture of runoff P by the buffer (Singh et al., 2020). This means that observed TDP concentrations in the SRBs may have been partially attributable to P loading that occurred prior to the diversion of tile drainage into the buffers. Second, the authors assumed that a decrease in TDP concentrations from a tile outlet across a buffer were attributable to retention of the TDP by the SRB soils. However, in addition to adsorption of P to soils, changes in TDP concentrations may occur due to uptake by vegetation and also by dilution or concentration (e.g., if riparian vegetation takes up water but not P). Furthermore, the findings of McEachran et al. (2023) indicate that drainage water flow paths within an SRB may be three-dimensional. If the water flow along the transect is not one-dimensional, then successive points along the transect may not represent successive changes to a volume of water as it flows through the SRB. Based on these methodology flaws, it is arguable that the observed TDP concentrations provide real insight into the question of SRB effectiveness for dissolved P removal.

Singh et al. (2020) noted the potential for dissolved reactive P to adsorb to iron and aluminum exchange sites within the soils of saturated buffers. The authors advocated for research to explore the use of steel slag or other industrial by-products with a high ion exchange capacity to backfill portions of saturated buffers or the tile drainage systems to enhance P removal. This recommendation was based on a cited study showing that by-products from steel and energy industries can remove P from tile drains. Note that the use of P adsorbents in blind inlets to reduce P loads in tile drainage has been discussed elsewhere in this chapter.

Other Pollutants

No studies were found that examined the effectiveness of SRBs for removing other pollutants occurring in tile drainage. However, it seems plausible that reductions in pathogens and pesticides may occur as tile drainage is filtered through riparian soils, provided preferential flow paths are not extensive. Due to the potential for clogging of distribution pipes, SRBs are not appropriate for removing sediment from tile drainage. Note that the NRCS 604 practice standard states that SRBs are only applicable to tile drain systems having surface inlets if the surface inlets are designed to prevent soil and debris from entering the tile lines (NRCS, 2020).

Design and Management Considerations

SRBs should meet Ecology's established guidelines for riparian buffers outline in Chapter 12 of the Clean Water Guidance for Agriculture, however, there are additional considerations specific to SRBs. The following are key considerations for SRBs, as presented by Jaynes et al. (2018):

- An existing or planned buffer needs to have a tile outlet crossing it; alternatively, a perforated tile main running parallel to the stream may also be suitable.
- The drainage area for the tile outlet must provide sufficient flow and nitrate loads to treat within the saturated buffer.
- The buffer should be at least 9.1 m (30 ft.) wide and planted to perennial vegetation.
    - Woody vegetation should be kept as far from the distribution pipe as possible to prevent possible plugging of the distribution pipes.
- Soils within the buffer need to be suitable, ideally a loam or clay loam soil.
    - Soils with a gravel or sand layer within the top 4 feet of the surface are not suitable due to their high permeability.
    - Soils with a high clay content may not be permeable enough.
    - Soils must show evidence of a seasonally or continuously high water table, which typically indicates that the drainage water will flow laterally toward a surface water body as shallow groundwater.
    - Soils needs to contain at least 1.2% organic matter to a depth of 0.8 m (2.5 ft.) to help support denitrification.
- Adjacent cropland should have an elevation a few feet higher than the buffer to allow the SRB to have a water table elevation near the soil surface without inhibiting field drainage. According to Carstensen et al. (2020), the soil slope perpendicular to the waterbody being buffered should be roughly 2%–8%. If the elevation difference is insufficient, the water level within the saturated buffer must be adjusted seasonally to avoid soil saturation in a portion of the adjacent field.
- Stream banks should be less than 8 ft. (2.4 m) high to prevent bank instability and sloughing. Channels incised more than 8 ft. (2.4 m) should not be used for a saturated buffer unless a slope stability analysis shows a low risk of saturated streambank failure.
- The length of an SRB should be as long as practicable because its length influences the amount of water that can discharge into the buffer, and therefore influences its nitrate removal capacity. Bypassed flow is a function of the length of the distribution pipe and hydraulic conductivity of the SRB soils.
    - The length of the saturated buffer is typically constrained by the length of the buffer, property boundaries, or physical barriers such as ditches, gullies, or roads that limit distribution pipe installation.

- Another potential constraint on distribution pipe length is related to the gradient of the valley floor parallel to the adjacent surface waterbody. When the distribution pipes are level underground, it allows water to be distributed parallel to the surface waterbody in two directions from the water control structure (e.g., for a stream, the water is distributed in both up and downstream directions before entering the buffer soils). For valleys with higher gradients, the distribution pipe needs to be installed deeper in the upstream direction and shallower in the downstream direction to keep the distribution pipes level.

Based on groundwater modeling, McEachran et al. (2023) provided design recommendations for adjusting the flow from the distribution pipes to improve SRB performance. The authors recommended that distribution pipes be modified to generate increasing discharge with increasing distance from the water control structure. Suggested methods for modifying the spatial pattern of discharge included installing the distribution pipe with a minor grade away from the structure, including fewer or smaller openings in the distribution pipe near the control structure, or incorporating a geotextile or fill around the distribution pipe.

Additionally, NRCS (2020) has identified numerous design and management considerations for SRBs related to hydrologic, hydraulic, soil, and vegetation factors. For example, one key design criteria is that the SRB hydraulic capacity should be at least 5% of the maximum tile outlet flow rate or as much as practical based on the available length of the vegetated buffer. Readers are referred to the NRCS Conservation Practice Standard (604) for detailed information on design and management considerations.

Potential Challenges

Research has identified several potential challenges and risks associated with the site selection, design, and management considerations for implementing SRBs.

One challenge is the risk of reduced streambank stability following saturated buffer installation. In Iowa, Dickey et al. (2021) examined the association between saturated buffers and streambank stability through numerical slope stability modeling, validated with data from field sites. The study was primarily focused on evaluating whether bank height and SRB width correlate with the probability of streambank failure in reference to NRCS practice standards for saturated buffers (604). The NRCS practice standard (NRCS, 2020) specifies criteria for streambank height (maximum height of 2.4 m (7.9 ft.), unless an evaluation indicates appropriate slope stability) and buffer width (a minimum 9.1 m,29.9 ft. setback from the SRB distribution pipe to the streambank) to minimize the risk of streambank failure. Dickey et al. (2021) found the probability of a previously stable streambank becoming unstable when SRB flow was added to be 3%. Bank height alone was not a good predictor of streambank instability for SRBs. The two conditions found to be associated with a change in bank stability were soils with effective cohesion less than 2 kPa or sites with buffer widths less than 2 m, 6.6 ft. (Dickey et al., 2021). It was noted that these conditions were unlikely to occur with SRB implementation because the NRCS standards restrict the practice from highly conductive soils such as sands or gravels that have little soil cohesion (e.g., < 2 kPa) and buffer widths greater than 2 m (6.6 ft.) are likely needed to achieve sufficient water residence time for denitrification to occur.

A second potential challenge to SRB effectiveness is that it is typical for drainage water volumes to exceed the hydraulic capacity of an SRB, particularly during the spring or other wet seasons. Allowing water to bypass the SRB during periods of high drainage flow is a design feature intended to prevent water from backing up and saturating a portion of the drained field. Consequently, a large proportion of the annual nitrate load may bypass the SRB during these periods. As mentioned previously, the hydraulic capacity of an SRB can be increased by increasing the length of the distribution pipes and by installing additional distribution pipes within the buffer instead of only at the buffer edge. Jaynes and Isenhart (2019b) caution that the latter design upgrade reduces the buffer width through which a portion of the drainage water will travel, and thus may reduce the NO_3^- removal rate for that volume of water.

Another potential risk is that loading of dissolved P will exceed the P retention capacity of an SRB. Under ongoing high P loadings, soil adsorption sites in riparian buffers may become saturated, limiting the P retention capacity and decreasing the SRBs ability to retain P over time. Buffer strips also tend to have high microbial respiration rates which increases P solubilization, particularly in the surface soils (top 5 cm), resulting in higher concentrations of bioavailable (i.e., dissolved reactive) P (Cole et al., 2020). In other words, soils with high soil P saturation tend to display high dissolved P losses (Sharpley et al., 1993). Buffer vegetation may play an important role in regulating the pool of P in the soils of SRBs, but research on this topic is lacking. In these regards, one unknown, but important factor is how readily SRBs may become saturated with P associated with tile drainage and how P saturation may affect dissolved P loading to surface waters.

Furthermore, there is a risk of increased greenhouse gas emissions resulting from SRB implementation. When NO_3^- delivery approaches the maximum amount that microbial communities can denitrify under existing conditions, release of the potent greenhouse gas N_2O can be heightened (Cole et al., 2020). A study by Wang et al. (2019) suggested that N_2O emissions from riparian buffers can be reduced by manipulating water tables in a way that shifts NO_3^- removal from denitrification to the anammox pathway (anaerobic ammonium oxidation). However, the means by which this could be achieved is unclear. Furthermore, some research casts doubt on the importance of N_2O emissions from SRBs. For example, in Iowa, Davis et al. (2019) found that total N_2O emissions from SRBs were similar to those from traditional buffers and less than those from fertilized corn–soybean cropland.

Finally, it should be noted that saturated soils in an SRB may have a reduced capacity for infiltrating surface runoff, thereby decreasing their effectiveness at reducing transport and delivery of pollutants in surface runoff. Physical sediment trapping of sediment is likely to occur on saturated soils due to flow resistance by vegetation. Dissolved pollutants such as pesticides and NO_3-N rely upon water infiltration to reduce delivery to surface waters. Because of this, SRBs may be more appropriate for fields where pollutant transport in surface runoff is negligible or other BMPs effectively prevent pollutant transport in surface runoff. However, this is an area of uncertainty as no studies have compared the performance of SRBs and conventional riparian buffers at pollutant removal from surface runoff.

NRCS Practices

The NRCS practices[6] commonly associated with SRBs include:

- Saturated Buffer (604)
- Structure for Water Control (587)
- Subsurface Drain (606)

Numerous additional conservation practices can be associated with SRBs, including the following common NRCS practices[1]:

- Blind Inlet (Underground Outlet (620))
- Cover Crop (340)
- Critical Area Planting (342)
- Drainage Water Management (554)
- Nutrient Management (590)
- Residue and Tillage Management, No Till (329)
- Residue and Tillage Management, Reduced Till (345)
- Riparian Forest Buffer (391)
- Riparian Herbaceous Cover (390)
- Streambank and Shoreline Protection (Code 580)

Conclusions

- Where site conditions are amenable, SRBs represent a relatively simple upgrade to tile drain systems that can be implemented in established or new riparian buffers.
- SRBs remove NO_3^- through denitrification, microbial immobilization, and plant uptake.
- SRBs can effectively remove NO_3^- from drainage water, with up to 100% removal observed for water infiltrating an SRB.
    - Net NO_3^- removal effectiveness strongly depends upon the proportion of total tile outlet discharge that the SRB can infiltrate.
    - When accounting for total tile outlet discharge (i.e., flow that is infiltrated and flow that bypasses the SRB), research from the Midwest indicates that SRBs can reduce the NO_3-N load by 50%–80% where site conditions are favorable.

[6] NRCS practice codes are noted in parentheses

    - Favorable conditions for NO_3^- removal include well-established vegetation, appropriate soil texture and topography, and SRB hydraulic capacity of at least 50% of the annual tile outlet discharge.
- Nearly all SRBs studied to date did not have the hydraulic capacity to infiltrate all the drainage water when field drainage was at a maximum during spring.
- There is evidence that SRB effectiveness at NO_3^- load removal increases as buffer age increases but does not necessarily increase as buffer width increases.
- The effectiveness of SRBs at removing P, pesticides, and pathogens is unknown due to a lack of research.
- SRBs may not be as effective at removing certain pollutants from surface runoff in comparison to conventional riparian buffers because saturated soils will have a reduced infiltration capacity.
- The effectiveness of SRBs may be improved through complementary practices that can reduce or regulate tile drain discharge such as alley cropping or controlled tile drainage.

Blind Inlet

Introduction

A blind inlet, also known as a gravel inlet or French drain, is a structure placed in the lowest point of a farmed depression (e.g., a pothole or a terrace pool) to drain water that collects in the depression. Tile risers—vertical perforated pipe connected to a tile drain network—have commonly been used to drain these depressions. However, tile risers provide a direct conduit to the tile drainage system and can convey surface runoff, sediment, and both particulate and dissolved pollutants directly into field drainage without treatment. Blind inlets are an alternative conservation practice that can drain depressional areas while providing filtration that can remove fine sediments and soluble pollutants.

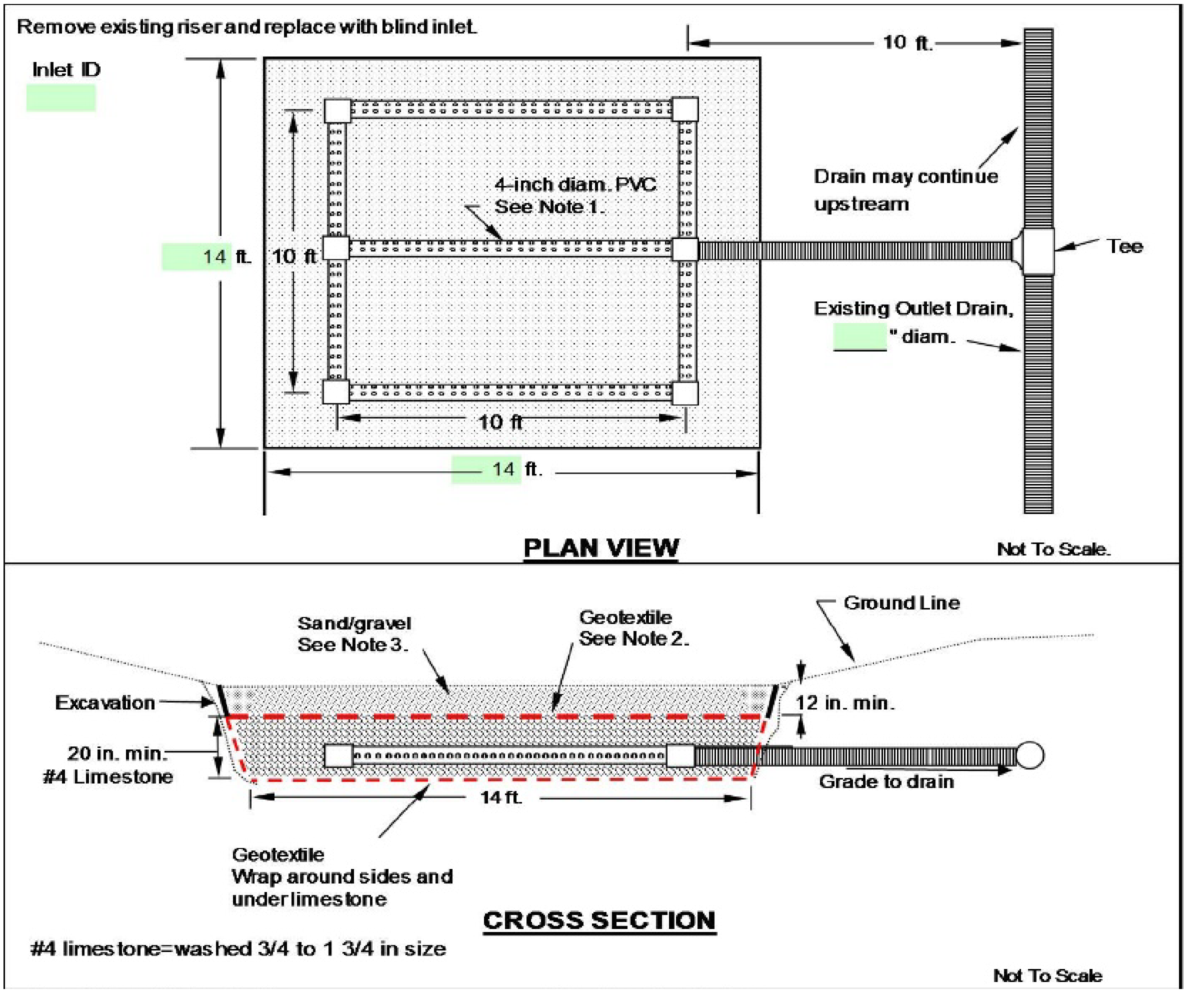

Figure 2. Schematic of blind inlet (Gonzalez et al., 2016a)

A schematic of a typical blind inlet is shown in Figure 2. The surface component of the inlet is typically filled with a combination of fine and coarse materials (e.g., gravel, sand, soil, and sometimes adsorbent materials to remove soluble pollutants) to filter surface runoff before it leaves the field through tile drainage. In midwestern states, the contributing area for a blind inlet typically ranges up to 20 acres (9.1 ha); the surface area of a blind inlet can be as large as 50 x 50 ft (15.2 m x 15.2 m) (Ghane, 2022). Daily farming operations can usually be conducted over the blind inlets, including tillage, planting, and harvesting.

When implemented through USDA-NRCS programs, blind inlets are installed under the conservation practice standard for Underground Outlet (620), which is defined as a conduit or system of conduits installed beneath the ground to convey surface water to a suitable outlet (NRCS, 2023). Note that several different types of inlets (including open risers as well as blind inlets) are acceptable within the 620 standard. However, the NRCS practice standard does caution that because underground outlets may provide a direct conduit to receiving waters for contaminated runoff, they should be installed with accompanying structures or practices as part of a conservation system that addresses issues such as nutrient and pest management, residue management, and runoff filtration.

Relevance to Washington

Blind inlets are widely used in farmed pothole regions in the northern Midwest and are applicable to cropped areas in Washington where potholes may be present or where best management practices (BMPs) such as terraces, diversions, or water and sediment control basins create closed depressional areas. One of the primary benefits to the farmer of using a blind inlet is the ability to drive equipment over the inlet, eliminating the need to drive around a standard tile riser.

When and Where Blind Inlets are Most Appropriate

Blind inlets can be implemented on any cropland where removal of surface water is necessary or an outlet for a terrace or other water-detention structure is needed. Blind inlets are applicable to cropland where natural surface drainage patterns result in isolated depressional areas (i.e., potholes) and where the climate is sufficiently humid to result in reduced trafficability or the loss of crops due to excessive high water table. The cropland must be drained by subsurface systems (tile drains) or ditches so that water collected by the blind inlet can be safely discharged. Blind inlets can also be used in depressional areas created behind other BMPs such as terraces (600) and Water and Sediment Control Basins (638).

Water Quality Concerns Addressed by Blind Inlets

Conventional tile risers are direct conduits for surface runoff to agricultural drainage ditches or streams, representing a risk of excessive loading of sediment and other contaminants to surface water from fields that are often distant from the ditch or stream. Blind inlets can provide removal of sediment and particulate pollutants like phosphorus (P), or agrichemicals adsorbed to fine sediments, and microorganisms through physical filtration through gravel and soil. The addition of specialized adsorbent materials to the inlet system can enhance the removal of soluble P and nitrogen (N) and soluble pesticides.

Effectiveness

Research into the effectiveness of blind inlets has been reported for both standard inlets where sand, gravel, and soil are the primary filtration media and for inlets with various amendments in the filtration media to enhance adsorption of dissolved pollutants. Those results will be reported separately. Note that nearly all of the reported results are from studies conducted by a few researchers on a limited number of sites in Indiana, and that rainfall intensities in the Midwest are typically greater than those of Washington. Reductions reported in discharge volume and sediment and nutrient loads in gravel blind inlets vs. tile risers are summarized in Tables 5 and 6.

Hydrologic Changes

There has been little research reported on the effects of conversion to blind inlets on field hydrology or drainage water quantity. Gonzalez et al. (2016a) evaluated blind inlet performance in Indiana and reported a 45% reduction in surface runoff volume passing through the blind inlet compared to the discharge through a tile riser. In a separate project, Gonzalez et al. (2016b) reported a 62% decrease in tile discharge from a blind inlet vs. a tile riser over six storms in 2010. Also in Indiana, Williams et al. (2020) found that replacing a tile riser with a blind inlet did not increase the frequency of discharge from drained closed depressions but may increase or decrease the duration of flow compared to a tile riser. Installing a blind inlet in a depression with lower subsurface drainage intensity may result in shorter flow duration compared to the tile riser due to the additional subsurface drainage capacity provided by the added tile lines in the blind inlet and potentially drier antecedent conditions compared to a tile riser. Installing a blind inlet in a depression with greater subsurface drainage intensity may result in less cumulative flow compared to a tile riser, which could result in larger nutrient load reductions than a blind inlet installed in a depression with lower subsurface drainage intensity.

Sediment

By filtering surface runoff through gravel/soil media, blind inlets can effectively remove significant quantities of sediment. In Indiana, Smith and Livingston (2013) observed 51%–62% reduction in sediment loads in drainage from fields with multiple blind inlets compared to fields with tile risers. Feyereisen et al. (2015) reported that converting all tile risers to blind inlets within a 4 ha Indiana crop field led to a 59% reduction in total suspended solids (TSS) loads during growing-season storm events. Gonzalez et al. (2016b) reported a total reduction of 79% in sediment transport through a blind inlet compared to a tile riser over six storms in 2010.

In an extensive literature review including recent data from an Indiana field study, Penn, Gonzalez, et al. (2020) reported a 40% decrease in sediment loads in drainage from blind inlets vs. tile risers. In a modeling study based in Indiana using the Soil and Water Assessment Tool (SWAT), Wallace et al. (2017) projected that blind inlets would reduce sediment loss by 42%–69% at the watershed level.

Phosphorus

On field-sized catchments in Indiana, Smith and Livingston (2013) documented reductions of 50%–78% in total P (TP) loads and 65%–72% reductions in soluble reactive P (SRP) loads in storm-events following replacement of all tile risers in the study field with blind inlets. Also in Indiana, Feyereisen et al. (2015) reported that TP and SRP loads were 66% and 50% less, respectively, for blind inlets vs. tile risers. Smith et al. (2015) observed that growing-season SRP loads from surface runoff were reduced by 84% through blind inlets compared to tile risers and TP loads were reduced by 76%. The authors noted that because P loads in overall tile drainage water were not significantly different between fields drained by blind inlets vs. tile risers, P loading to the tile lines from subsurface water exceeded the loading from surface runoff. Gonzalez et al. (2016b) reported a total reduction of 72% in dissolved P transport through a blind inlet compared to a tile riser over six storms in 2010. In a SWAT modeling study based in Indiana, Wallace et al. (2017) reported that the effectiveness of blind inlets in reducing total and soluble P was not significant. In an Indiana field study, Penn, Gonzalez, et al. (2020) observed a 40% decrease in particulate P loads in drainage from blind inlets vs. tile risers. The authors noted that traditional blind inlet sand media are generally unable to remove appreciable amounts of dissolved P compared to alternative media (discussed below).

Nitrogen

Few reports of N reductions through conventional gravel/soil blind inlets were found. Smith and Livingston (2013) reported 28%–59% reductions in ammonium N (NH_4-N) and 55%–64% reductions in total Kjeldahl N (TKN) for blind inlets vs. tile risers. The study found mixed results for nitrate N (NO_3-N) behavior through blind inlets, ranging from a 38% increase to a 24% decrease. Gonzalez et al. (2016b) reported a total reduction of 59% in NH_4-N and a reduction of 25% in NO_3-N transport through a blind inlet compared to a tile riser over six storms in 2010. Wallace et al. (2017) reported that SWAT modeling in Indiana watersheds (in which all tile risers were replaced with blind inlets) indicated a reduction in total N (TN) loss of 35%–55% would occur through blind inlet implementation, but that negligible reductions in soluble N or TN would occur at the watershed level.

Herbicides

Only two reports were found that evaluated herbicide reductions through a soil/gravel blind inlet. Both reported significant reductions in herbicide transport through blind inlets. In Indiana, Gonzalez et al. (2016a) compared blind inlet to tile riser performance for 20 runoff events over six years and reported the sum of herbicide load reductions:

- Atrazine: 69%
- Deethylatrazine: 57%
- 2,4-D: 58%
- Metolachlor: 53%
- Glyphosate: 11%

In a separate publication, Gonzalez et al. (2016b) reported a total reduction of 79% in sediment transport through a blind inlet compared to a tile riser over six storms in 2010, and the following reductions in herbicide loads:

- Atrazine: 82%
- 2,4-D: 58%
- Metolachlor: 80%
- Glyphosate: 72%

Water Quality Benefits of Amended Blind Inlets

Several researchers have evaluated the effects of amending blind inlet filter media with specialized materials capable of adsorbing dissolved pollutants such as soluble P and herbicides. These amendments have included wood chips, iron/steel shavings, steel slag, biochar, and activated charcoal. Results are summarized in Tables 5 and 6.

Phosphorus

In a laboratory experiment, Yang et al. (2021) tested the ability of biochar filter media to capture dissolved P from aqueous solutions. For incoming P concentrations < 1.0 mg/L (ppm), the biochar completely removed dissolved P; for input concentrations of 10 mg/L (ppm), the biochar removed ~76% of incoming P. It is not known if the fine biochar media would be appropriate in a blind inlet system.

In a farmed pothole catchment in Iowa, Buseman (2022) compared changes in P concentrations in a blind inlet with a P sorption medium composed of steel shavings mixed with gravel above the blind inlet relative to a conventional surface inlet. The amended blind inlet removed an average of 83% of the influent dissolved P concentration and 62% of the influent TP concentrations compared to the surface inlet.

In an Indiana field study, Gonzalez et al. (2020b) evaluated P loads before and after conversion of a tile riser to a blind inlet amended with steel slag. The amended inlet removed ≥45% of the incoming dissolved P load and ≥22% of the TP load; the amended inlet was still effective after 3 years of operation. The dissolved P removal efficiency was greater with higher inflow P concentrations.

In Indiana, Penn, Livingston, et al. (2020) tested the P removal efficiency of a subsurface P removal structure (down-gradient from a blind inlet) amended with steel slag. The subsurface structure removed 55% of incoming dissolved P load. The authors noted that dissolved P removal was much greater when the treated water had a pH below 8.5. The authors also reported that the structure under-performed relative to laboratory experiments testing the slag media.

Nitrogen

In Michigan, Ghane (2022) reported data suggesting that a mixture of gravel and wood chips (3:1 ratio) could remove 90% of sediment and 36%–60% of NO_3-N.

In an Indiana field study before and after conversion of a tile riser to a blind inlet amended with steel slag, Gonzalez et al. (2020b) reported that the inlet removed 67% of incoming NH_4-N, but that cumulative NO_3-N content of the outflow was 25% higher than inflow, suggesting that the inlet was a source of NO_3-N, possibly due to nitrification in the filter media.

Herbicides

Gonzalez et al. (2020a) conducted lab studies of the effectiveness of activated charcoal cloth as an amendment to a blind inlet to remove atrazine from water. The researchers reported that activated charcoal-impregnated cloth can sorb 75%–100% of soluble atrazine, depending on initial herbicide concentrations. They suggested that activated charcoal cloth has the potential to be used in pollution control structures such as blind inlets, although no field trials were conducted.

In an Indiana field study before and after conversion of a tile riser to a blind inlet amended with steel slag, Gonzalez et al. (2020b) reported removal of >70% of glyphosate and dicamba from inflow.

Other Environmental Benefits

No other environmental benefits to blind inlets have been identified in the literature. As noted above, the ability to conduct farming operations over a blind inlet can be a significant benefit to the farmer

Table 5. Percent reductions in surface runoff quantity and pollutant load reductions from soil and gravel blind inlets[1].

Location	Conditions	Q (%)	TSS (%)	TN (%)	TKN (%)	NO_3-N (%)	NH_4-N (%)	TP (%)	Diss P (%)	Source
Indiana	Wheat-corn-soybean-oat rotation	-	51–62	-	55–64	(-38)–24	28–59	50–78	65–72	Smith and Livingston (2013)
Indiana	Corn-soybean-grain rotation	-	59	-	-	-	-	66	50	Feyereisen et al. (2015)
Minnesota	Corn-soybean-grain rotation	-	88	-	-	-	-	-	35	Feyereisen et al. (2015)
Indiana	Corn-soybean-wheat rotation	-	-	-	-	-	-	75	84	Smith et al. (2015)
Indiana	Corn-soybean, no-till	45	-	-	-	-	-	-	-	Gonzalez et al. (2016a)
Indiana	Corn-soybean, no-till	-	62	79	-	25	59	-	72	Gonzalez et al. (2016b)
-	SWAT model	-	42–69	35–55	-	ns	-	-	-	Wallace et al. (2017)
Indiana	Corn-soybean-grain, disc plow	-	40	-	-	-	-	40	-	Penn, Gonzalez, et al. (2020)

Notes: [1] Negative values represent increase in load. “-“ indicates no data reported; “ns” represent nonsignificant difference. “-“ indicates no data reported.

Table 6. Percent reductions in surface runoff quantity and pollutant load attributed to amended blind inlets[1].

Location	Conditions	Q (%)	TSS (%)	TN (%)	TKN (%)	NO_3-N (%)	NH_4-N (%)	TP (%)	Diss P (%)	Source
Indiana	Corn-soybean, no-till/steel slag	-	-	-	-	(-25)	67	$\geq$22	$\geq$45	Gonzalez et al. (2020b)
Indiana	Corn-soybean/steel slag structure	-	-	-	-	-	-	-	55	Penn, Livingston, et al. (2020)
===	Laboratory/biochar	-	-	-	-	-	-	-	76–100	Yang et al. (2021)
Iowa	Corn-soybeans, conv. till/steel shavings	-	-	-	-	-	-	62	83	Buseman (2022)
Michigan	Unknown crop/wood chips	-	90	-	-	36–60	-	-	-	Ghane (2022)

Notes: [1] Negative values represent increase in load. “-“ indicates no data reported; “ns” represent nonsignificant difference. “-“ indicates no data reported.

Design and Management Considerations

Because blind inlets are engineered structures, they must be designed to fit each site. Principal considerations include:

- Sizing: based on precipitation patterns and contributing area.
- Aggregate selection: materials, mix of fine and coarse materials to provide filtration, choice of amendments if desired.
- Hydraulics: permeability of filter media, pipe sizing, slope of piping, connection to tile drain system or outlet.

Blind inlets require occasional maintenance to ensure continuing effectiveness. Over time, the filter materials can become clogged with sediment and debris, reducing the inlet's ability to convey water and capture pollutants. Regular inspection and cleaning are necessary to maintain their performance and prolong their service life. Care is required while operating farm equipment near the blind inlet to avoid mixing soil and sediment into the sand filter layer. Operators should minimize driving over the inlet and lift implements while passing over the gravel/sand surface.

Because the blind inlet is designed to filter sediment from the water flowing to it, the surface will eventually become covered with sediment. Weeds and vegetation may also begin to grow on the surface. Both may adversely affect the inlet capacity. Therefore, sediment and vegetation should be periodically removed to a depth that exposes clean gravel/sand at the surface. Any filter materials removed along with the sediment should be replaced to maintain a filter layer of at least one foot thickness.

The life expectancy of blind inlets depends on the amount of sediment generated from field practices, but they can have a useful service life of up to 10 years Ghane (2022). Tillage and soil disturbance in the vicinity of a blind inlet can reduce its lifespan due to soil erosion and sedimentation clogging the filter layer. Williams et al. (2020) noted that blind inlets installed in closed depressions with extensive tillage may become ineffective at draining ponded surface water after 8 to 10 years of service due to impeded infiltration. The authors suggested that blind inlets would maintain higher infiltration rates for longer periods of time in fields with reduced or no-tillage, extending the service life well beyond 10 years. Cover crops, manure or compost, and diverse rotations improve soil health, thereby reducing soil erosion.

Blind inlets can be used in conjunction with many other field BMPs. As noted earlier, blind inlets can serve to drain not only natural closed depressions like potholes but also other areas where surface water can accumulate such as behind terraces, diversions, or water and sediment control structures. There are also several complementary practices that, if applied in conjunction with the blind inlet, can increase water quality benefits, decrease the amount and frequency of maintenance, and lengthen design life. These complementary practices include filter strips (393), cover crops (340), no-till or other tillage management practices (329, 345, 346), and nutrient management (590). Any crop management practice that reduces erosion and soil loss can be useful to minimize the amount of sediment that is delivered to the blind inlet to maintain the inlet's drainage capacity.

NRCS Practices

- Underground Outlet (620)[7]

Commonly Associate Practices

Numerous additional conservation practices can be associated with blind inlets, including the following common NRCS practices[1]:

- Constructed Wetland (656)
- Contour Farming (330)
- Contour Buffer Strips (332)
- Cover Crop (340)
- Denitrifying Bioreactor (605)
- Drainage Water Management (554)
- Filter Strip (393)
- Grassed Waterway (412)
- Nutrient Management (590)
- Residue and Tillage Management, No Till (329)
- Residue and Tillage Management, Reduced Till (345)
- Residue and Tillage Management, Ridge Till (346)
- Riparian Forest Buffer (391)
- Riparian Herbaceous Buffer (390)
- Saturated Buffer (604)
- Subsurface Drain (606)
- Terrace (600)
- Underground Outlet (620)
- Water and Sediment Control Basin (638)

Conclusions

A blind inlet is an engineered structure that is placed in the lowest point of farmed depressions or potholes to minimize the amount of sediment, P, N, and other contaminants that would be transported to receiving ditches or streams through a field's subsurface or surface drainage

[7] NRCS practice codes are noted in parentheses.

system. Blind inlets can replace open tile risers, which provide a direct path for untreated surface runoff to leave the depression. At the land surface, blind inlets include filter media—either gravel/soil mixture or special amendments to improve pollutant removal—that capture both particulate and dissolved pollutants from field runoff before it enters the drainage system.

Blind inlets can be highly effective in removing sediment, P, N, and herbicides from surface runoff.

- Sediment (TSS) removal by blind inlets has been reported in the 45%–90% range.
- P removal efficiencies of 35%–78% have been reported from gravel blind inlets; P removals of >80% have been observed from inlets amended with steel slag, wood chips, or biochar.
- N removal efficiencies of 25%–60% have been reported for TN and NH_4-N, but blind inlets may increase NO_3-N loads if nitrification is promoted in the filter media.
- Significant reductions of atrazine, metolachlor, glyphosate, and other herbicides can be achieved with both gravel and amended blind inlets.

Blind inlets are highly suitable for depressional areas (natural or created by other BMPs) on cropland where subsurface or surface drainage exists. Specifications for blind inlets must be adapted to site-specific climate, soil, and field conditions. Blind inlets typically have a design lifetime of up to 10 years, but maintenance and control of soil erosion from the contributing area are advisable.

Finally, blind inlets can be implemented in concert with many field cropping BMPs because farm operations can be done across the inlet and do not require avoidance of above-ground structures. Blind inlets can complement numerous other water management, erosion control, and crop management BMPs.

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Chapter 8 Appendix Part B: Implementation Considerations Subsurface Drainage Management

Adoption of Subsurface Drainage Management in Washington State

Washington state is a significant producer of a diverse array of agricultural crops. Depending on the location in the state and the specific climatic conditions present, producers may desire to drain excess water from their fields or may want to retain soil water. Draining fields results in a discharge of field runoff that may contain a variety of pollutants, requiring filtration of that field runoff before it enters either ground or surface water. Appropriate management of field drainage and runoff requires active maintenance and management. In many cases, subsurface drainage management provides benefits to the operator, whether it is to drain low-lying areas of fields or to maintain a higher water table to support plant growth.

Requirements for subsurface drainage management systems should be tailored to the climatic conditions and size of the facility. Western and eastern Washington have differing climates, topographies, soil types, and annual precipitation rates, with variations at even the local level, requiring site specific planning for subsurface drainage management.

Climatic Conditions

It is important to consider the climate conditions of individual operations; given the climatic differences between eastern and western Washington, the water volume capacity of subsurface drainage systems will vary. For example, on average, western Washington receives more rainfall, and BMPs will need to accommodate more consistent, larger volumes of flow. BMPs in eastern Washington will need to account for larger volumes of snow melt runoff, pipes freezing, and building any aboveground structural elements to withstand the weight of snow and ice.

Soil Type

Soil types vary across Washington state, which can influence the performance of subsurface drainage management systems. For example, well drained soils or soils with large macropores may infiltrate water rapidly, allowing water, dissolved chemicals, and bacteria to enter subsurface drainage systems, bypassing the filtration properties of the soil. This can reduce the effectiveness of some subsurface drainage management BMPs to reduce pollution. Additionally, the desired drainage level and amount of drainage needed for crop production will vary based on soil types and crops grown, which affects how subsurface BMPs are implemented. Therefore, it is recommended to incorporate site-specific soil conditions when planning and designing subsurface drainage management BMPs.

Challenges associated with subsurface drainage management systems

There are logistical and practical challenges associated with implementing subsurface drainage management systems. These systems often require specialized expertise to ensure proper installation along with active management and careful and accurate recordkeeping to support adaptive management and to ensure both crop needs and water quality outcomes are achieved. This appendix includes information on potential challenges associated with subsurface drainage management, and where applicable, potential solutions or considerations to facilitate addressing challenges.

Cost and Benefits Associated with Subsurface Drainage Management

The costs of implementing subsurface drainage management BMPs vary with the BMP being implemented, but may be high for those requiring drainage control structures, the use of heavy equipment, and contractors or engineers. Implementation costs will vary depending on practice, design, and the availability of materials. Costs can be reduced by producer conducting their own implementation work with their own equipment, when possible, though producers should consider receiving additional support from designers and engineers, to ensure that the practices will meet the needs of their property. Costs may also increase depending on federal, state, and local permitting requirements.

Often there are resources available to support the planning and implementation of BMPs that provide co-benefits of protecting and/or restoring water quality. Local conservation districts, NRCS offices, and/or county programs may be able to provide cost-free technical assistance and even financial support for implementation. For example, NRCS' Conservation Technical Assistance (CTA) can help with resource assessments and monitoring, practice design, and the development of conservation plans, which can be implemented using financial assistance from other local, state, and/or federal funding sources. There are several federal funding sources that may support the implementation of subsurface drainage management, such as NRCS' Environmental Quality Incentives Program (EQIP), that provides both technical and financial assistance to address a variety of natural resource concerns, including improved water quality, conserved ground and surface water, and increased soil health, among others. Additionally, the Conservation Stewardship Program (CSP) provides support for improving operation enhancement and stewardship improvement. See NRCS' information on Washington State Programs and Initiatives[8] to learn more and find a local field office.

Permitting

Permitting requirements should be considered at the early planning stage when designing subsurface drainage management systems, as this may increase the time needed to complete a

[8] https://www.nrcs.usda.gov/state-offices/washington

project, and bring associated costs. Permits may vary at the local level, and County permitting offices should be consulted. Permits and requirements may include (but are not limited to):

- Cultural Resources Permit may be required when disturbing land with historic cultural significance. Complete permit requirements can be found in the Washington Administrative Code 25-48-060[9].
- Hydraulic Project Approval (HPA) is required when planning projects in or near state waters. An HPA from the Washington Department of Fish and Wildlife ensures that construction is done in a manner that protects fish and their aquatic habitats. Visit WDFW's information page[10] on HPAs to learn more about fish-friendly projects and to talk with a habitat biologist for technical assistance and to learn about the planning and approval process.
- Activities along streambanks, in-water, and near wetlands may require additional permits. See Ecology's Clean Water Act: Section 401 Water Quality Certifications webpage[11] to learn more about these permits and the wetlands regulations[12] webpage for information specific to projects with activities in wetland areas.

Maintenance

Subsurface drainage management BMPs require periodic maintenance activities to ensure they remain operable and effective over time. It is important to account for the time and money of continued maintenance. Maintenance activities for subsurface drainage management BMPs may include:

- Routine checks following winter season and high precipitation events.
- Manual labor, including clearing of inlets, maintenance of vegetation, and tracking of water level elevations.

Producers should consider the maintenance needs and time required to maintain subsurface drainage manage systems in relation to normal operating routines and workloads, and should select a series of BMPs that work best within existing responsibilities. Doing so provides the highest likelihood of maintaining BMPs over time.

Site Specific Requirements

The design and implementation of subsurface drainage management BMPs is extremely site-specific. Site characteristics such as topography and slope, field size, existing riparian vegetation (in the case of saturated riparian buffers), and soil characteristics should be considered when selecting BMPs and planning for proper long-term maintenance. Although there are some

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[9] https://app.leg.wa.gov/WAC/default.aspx?cite=25-48-060
[10] https://wdfw.wa.gov/licenses/environmental/hpa
[11] https://ecology.wa.gov/regulations-permits/permits-certifications/401-water-quality-certification
[12] https://ecology.wa.gov/water-shorelines/wetlands/regulations/state-wetland-regulations

practices that could be developed and constructed by producers, it is recommended to work with engineers and other technical professionals at your local conservation district or NRCS office. This can help to ensure the system will work properly, meet all regulatory requirements, and will maximize its effectiveness, to support both water quality and farm operations. Local offices may also have insights into how to plan drainage management systems to successfully withstand extreme weather events and help to plan for anticipated impacts of climate change in your area. Incorporating these considerations early in planning can limit challenges in the future and reduce the likelihood of needing to correct or re-do BMPs.

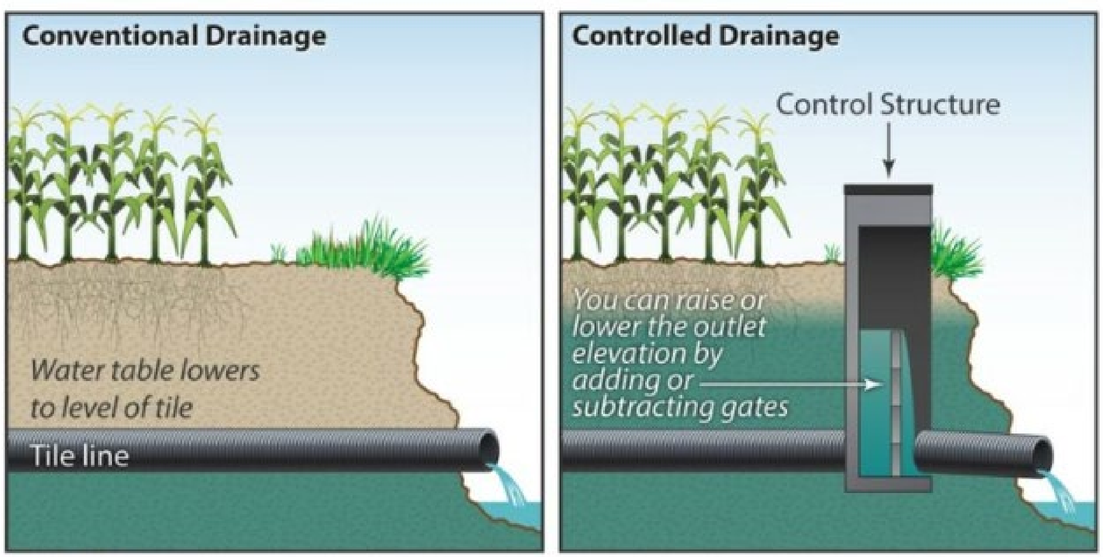
Implementation Considerations and Costs

This section includes considerations for implementation of controlled tile drainage, saturated riparian buffers, and blind inlets. Decisions regarding the design, planning, and implementation of these practices are highly dependent upon site-specific characteristics.

Controlled Tile Drainage

Use Cases

CTD uses a water control structure to raise the depth of the field's drainage outlet, to hold water in the field when drainage is not needed. CTD increases water retention in field soils, and with careful management can improve root health and plant productivity. CTD is suitable for fields that need drainage and is most effective on low slope fields (less than 2%), to allow the least number of control structures to manage the water table of the most amount of acres (Frankenberger et al., 2006).



NRCS/SWCS photo by Lynn Betts

Figure 3: Water control structure used for controlled tile drainage.

Benefits

CTD supports retaining water in fields later in the growing season, which can improve the productivity, health, and vigor of plants. By raising the water table during critical periods of the growing season and following rainfall events, the rate at which water, and therefore nutrients and agricultural chemicals, are leached from the soil is reduced. By reducing the amount of drainage that leaves the field, pollutant losses are reduced. CTD allows producers to adjust the drainage intensity depending on plant and field access needs, and well managed CTD with appropriate rainfall has the potential to improve crop yields beyond the crop response to drainage alone (Frankenberger, 2006). It is during the fallow seasons when drainage management has the greatest impact on reducing drainage volume and N runoff; however, CTD can also limit pollutant disharges to surface waters throughout the growing season, especially when managed in coordination with irrigation, fertlizer, pesticide, and other agrochemical applications.

Challenges

CTD is an active management practice, and the producer must regularly adjust drainage discharges and water levels to balance growth and environmental objectives. The producer should maintain a management calendar that specifies the target water control structure outlet elevation throughout the year to meet the desired goals and outcomes. Specific management activities and schedules will depend on desired outcomes, climate, soil conditions, crop(s) produced, and other factors; this requires careful planning, monitoring, and adaptive management. Consultation with a technical service provider can be valuable during the planning stage, to ensure that site specific conditions are appropriately considered and integrated into implementation plans.

CTD may interact with reduced tillage practices in a negative way, as the increased formation of soil macropores can result in rapidly transmitted water, dissolved chemicals, and bacteria into soil water and drainage flow, bypassing the filtration properties of the soil matrix. This increased transmission of pollutants into groundwater and drainage systems may reduce the effectiveness of CTD in pollutant reduction, as CTD are not able to fully element nutrients, bacteria and dissolved chemicals. Producers should consider the addition of saturated riparian buffers to address these transmitted pollutants and implement practices the reduce the potential for chemical and fertilizer leaching such as nutrient, irrigation and pesticide management.

In addition to reducing water-driven losses of N and P, CTD can also affect crop growth, water availability, nutrient use efficiency, and nutrient removal by crop uptake. Thus, CTD will potentially interact with the NRCS Nutrient Management (590) practice by altering N and P requirements in fertilizers and/or manure to be applied; refer to Chapter 3- Nutrient Management for more information and management considerations. Active management of soil moisture and nutrient levels is recommended for successful implementation of this BMP.

Table 7. Implementation Considerations for Controlled Tile Drainage

<table>
<tr><th>Considerations</th><th>Details</th></tr>
<tr><td>Costs</td><td>

It is assumed that CTD is being implemented on land with a subsurface drainage system already in place that can be adapted to include CTD.
Purchase and installation of the water control structures; the costs below are estimates per water control structure for Practice 587 from NRCS Washington Practice Scenarios for Fiscal Year 2025.

1 foot diameter slide gate made of steel and operated manually: $1,352
4 foot diameter slide gate made of steel and operated manually: $10,754
Automation retrofit to manually operated slide gate: $6,190
Automated DWM control structure on a 12-18 inch diameter pipe: $12,329

Maintenance and operations

Operation costs below are estimates from NRCS Washington Practice Scenarios for Fiscal Year 2025.

Example scenario: Manually operated water control structure (riser boards) located 5 miles from operation HQ. Assumes 6 trips to field, with 0.5 hours to make adjustments per structure (including travel time). Assumes 3 water level control structures for a 75ft field. Cost: $177/structure.
Automated water management with telemetry data systems and cloud data management; affected area for automated control structure is 10-20 acres. Cost: $12.02/acre.

Inexpensive water table observation wells can improve management.

Financial planning

To be economical and practical, each control structure should influence a significant amount of the field.
This practice may be compatible with existing federal and state cost-share programs.

</td></tr>
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Considerations	Details

Operational and Maintenance Requirements	**Installation and operation**
	• Develop and implement a management plan that specifies the management objectives and target water control structure outlet elevation to achieve the desired objectives:
	    o Include outlet elevation adjustments following manure or fertilization application.
	    o Specify other conditions in which adjusting the outlet elevation may be required (for example, significant rainfall events).
	• Each water control structure should be kept in a managed drainage mode except when the water table must be lowered for adverse weather conditions, trafficability for field work, or system maintenance.
	Cold weather considerations
	• To avoid freezing field drainage pipes, lower the outlet elevation at the control structure during winter in cold climates, after drain flow has stopped.
	Monitoring/adaptive management
	• Periodically and/or during different weather conditions monitor and record the water levels in the water control structures and the water table within the control zones- this will inform management adaptations that may be needed to accommodate changing weather conditions. Update management plan as needed.

Technical Requirements	• Prepare detailed plans that include a topographic map with 1ft or less contours, field boundaries, drained area boundaries, and soils map with soil classes. This should inform development of a map of the planned drainage system, including location of water control structure(s) and size and location of all field drain mains and laterals, and the control zone of each control structure. • Material and structural specifications (see NRCS Practice Standard 554 (Drainage Water Management). • Consider soliciting planning assistance from an engineer or other technically trained staff, such as that available from a local conservation district and/or NRCS office.
Lifespan	The lifespan of Controlled Tile Drainage systems depends on the quality of the components, maintenance, and environmental

Considerations	Details
	conditions. Materials used in control structures may include plastics, metal, rubber, and electronic components for automated structures. Anecdotal reports include structures working well after 20-25 years.
Land Area Requirements	**Soil considerations and site selection** • Combining with other field practices, such as cover cropping, may have additional benefits. • Consider soil health- the improved soil health and resulting increase in macropores that result from reduced tillage may increase the transmission of dissolved chemicals and bacteria into soil water, enhancing transmission of pollutants into drainage water. Additional practices, such as a saturated riparian buffer may be needed to further filter water to meet water quality goals. • CTD may increase base flow in streams and ditches. • Consider soil types and avoid installation in trafficked areas on fine-textured soils to minimize soil compaction. • Consider and avoid impacts to any septic fields. **Topographical and location requirements** • Field slope impacts how many drainage control structures are needed to manage the water table. • Consider field topography patterns and the use of “drainage management zones,” each controlled by one drainage control structure. • Locate structures and pumps where they are easy to access and maintain. • Consider location and condition of adjacent fields, and manage water level accordingly, as retained water may escape the controlled tile drain system via neighboring free tile drain systems (Lavaire et al. (2017)).
Other implementation factors	To reduce the amount of nutrients and pesticides available to leach into water, consider the use of other land management practices, such as crop selection and application of cropping BMPs like cover crops, conservation crop rotation, nutrient management, irrigation management and pest management.

Considerations	Details
Resources	[NRCS Practice Scenario cost estimate for Washington, Fiscal Year 2025][13]

Saturated Riparian Buffers

Use Cases

Saturated Riparian Buffers may be used to help prevent excess nutrients and other pollutants from tile drainage systems from leaving crop fields and polluting waterways. SRBs are most appropriate for croplands where tile drainage export of NO_3^- to surface waters is a significant water quality concern. Much of the research evaluating the effectiveness of SRBs at treating field runoff has focused on nitrate, and the ability of SRBs to treat P, pesticides, and pathogens is unknown. SRBs are appropriate for use at operations where buffer and tile outlet are already in place, with the vegetated buffer being at least 30 feet wide and with appropriate soils to effectively treat tile drainage.

To provide perspective on the watershed-scale applicability of SRBs, Tomer et al. (2018) performed a landscape analysis of the suitability of SRBs in three midwestern HUC-12 watersheds based on proposed criteria for suitable soil and topographic characteristics. These criteria were:

- A soil organic matter content greater than 1% between depths of 0.75 m and 1.2 m (2.5 ft. and 3.9 ft.).
- Relatively fine-textured subsoils,<50% sand between depths off 0.75 m and 1.2 m (2.5 ft. and 3.9 ft.).
- A water table surface less than 1 m (3.3 ft.) below the soil surface from April through June.

According to the NRCS Conservation Practice Standard (NRCS, 2020), there are several site conditions that determine the applicability of the SRB practice:

- SRBs only apply to lands with a subsurface drainage system that can be adapted to discharge into the soils of a riparian buffer and where the soils and topography of the buffer are able to maintain a raised water table without adverse effects to crops, channel banks, shorelines, or adjacent land. SRBs are not appropriate for discharging septic system effluent or animal waste.

The NRCS Conservation Practice Standard 604 (NRCS, 2020) identifies additional criteria for the vegetated buffer area that influence SRB applicability:

- Soil conditions, such as a restrictive layer, must be present to create saturated conditions when water is discharged from a subsurface drainage system into a riparian buffer.

[13] https://www.nrcs.usda.gov/sites/default/files/2024-11/fy25-wa-scenarios.pdf

- There should be no soil voids or layers of high hydraulic conductivity soil that could provide preferential flow paths.
- A minimum of 0.75% organic carbon (1.2 percent organic matter) should be present in the top 0.8 m (2.5 ft.) of soil.
- There should be no abandoned drainpipes in the buffer area that could continue to drain the buffer.
- The vegetated buffer zone should be at least 9.1 m (30 ft.) wide.

Benefits

SRBs are an effective option for removing nitrate from field drainage, with up to 100% removal from water infiltrating the SRB. When accounting for total tile outlet discharge, including that which infiltrates soil and bypasses the SRB, SRBs have still been shown to reduce loading by up to 50-80%. SRB effectiveness at treating pollutants may be further improved through pairing with complementary practices, such as controlled tile drainage, discussed in more detail above. SRBs are also important for restoring or enhancing saturated conditions in many riverine, lacustrine fringe, slope, or depression wetland soils, improving habitat as well as water quality.

Challenges

Although effective at removing nitrate from soil runoff, SRBs are not appropriate for all croplands, and several topographic, soil, and hydrologic characteristics must be considered. For example, soil surface layers should have a moderate hydraulic conductivity, as denitrification will be limited by short water retention time in areas where soil layers have high hydraulic conductivity, and too low hydraulic conductivity will limit denitrification as too much of the drainage water will bypass the SRB.

High precipitation or runoff events can exceed the hydraulic capacity of SRBs to infiltrate all drainage; this has been observed occurring during spring field drainage in studied SRBs. Additionally, it’s typical for drainage water volumes to exceed the hydraulic capacity of an SRB, particularly during the spring or other wet seasons. Allowing water to bypass the SRB during periods of high drainage flow is a design feature intended to prevent water from backing up and saturating a portion of the drained field. Consequently, depending on the timing and amount of precipitation, there is a potential for a large proportion of the annual nitrate load to bypass the SRB during these periods.

Another potential risk is that loading of dissolved P will exceed the P retention capacity of an SRB. Under ongoing high P loadings, soil adsorption sites in riparian buffers may become saturated, limiting the P retention capacity and decreasing the SRBs ability to retain P over time. Buffer vegetation may play an important role in regulating the pool of P in the soils of SRBs, but research on this topic is lacking. In these regards, one unknown, but important factor is how readily SRBs may become saturated with P associated with tile drainage and how P saturation may affect dissolved P loading to surface waters.

Finally, it should be noted that saturated soils in an SRB may have a reduced capacity for infiltrating surface runoff, thereby decreasing their effectiveness at reducing transport and delivery of pollutants in surface runoff. Physical sediment trapping of sediment is likely to occur on saturated soils due to flow resistance by vegetation. Dissolved pollutants such as pesticides and NO_3-N rely upon water infiltration to reduce delivery to surface waters. Because of this, SRBs may be more appropriate for fields where pollutant transport in surface runoff is negligible or other BMPs (e.g. filter strips, tillage and residue management, cover crops) effectively prevent pollutant transport in surface runoff. However, this is an area of uncertainty as no studies have compared the performance of SRBs and conventional riparian buffers at pollutant removal from surface runoff.

Table 8. Implementation Considerations for Saturated Riparian Buffers

Considerations	Details
Costs Costs, continued	**Initial investments** • It is assumed that SRB is being implemented on lands with a subsurface drainage system already in place that can be adapted to discharge into the soils of a riparian buffer. For improved control over drainage, a structure for water level control may also be in place (see above practice Controlled Tile Drainage). • Purchase and installation of distribution pipe. The materials and costs will vary depending on the volume of field runoff being treated and the length of the SRB. **Materials** ○ Corrugated single wall pipe: \$0.50 - \$3.00/ft (for typical agricultural sizes). ○ Drainage lateral connection: \$33.88 for 6” drainage lateral to main drain, including excavation to 6’ depth, installation of and material for tee on main line, connect lateral, and backfill. **Construction and Installation** ○ Trenching: \$2.48 - \$3.78 per ft. ○ General Labor: Approximately ~33.98 per hour. ○ Large equipment rental (if required): \$761.41 - \$919.30. ○ Equipment operators (if required): \$51.15 per hour. • Riparian plant purchase and installation, as needed to supplement the vegetation present in the riparian buffer area (see Chapter 12- Riparian Areas and Surface Water Protection for riparian buffer implementation considerations).

Considerations	Details
	• Engineered designs may be needed to ensure proper soil characteristics, drainage areas, and soil slopes to ensure proper soil saturation. Producers may consider contacting the local conservation district and/or NRCS office to enquire about free technical assistance planning. **Maintenance** • Pipe repairs, as needed. • Riparian buffer vegetation maintenance and replacement, as needed, to ensure optimal function of the buffer. **Financial planning** • For cost effectiveness, consider locating the SRB where it will intercept a subsurface drain outlet that drains at least 15 acres. • This practice may be compatible with existing federal farm bill and state cost-share programs.
Operational and Maintenance Requirements	**Installation and operation** • Ensure that the streambanks are stable and have adequate vegetative cover. See resources below for NRCS' Stream Visual Assessment Protocol 2 (SVAP2) for elements to consider when completing a bank condition evaluation. • Place the distribution pipe, at a minimum, 2 feet below the surface. • Prior to installation, consider use of 4-feet soil cores along the length and width of the buffer area, to evaluate the presence of high conductivity soil layers that could provide undesirable preferential flow paths. • Consider field verification to ensure adequate topsoil organic matter content. **Monitoring** • Consider installing observation wells in the buffer midway between the distribution pipe and the receiving

Considerations	Details
	channel to facilitate water table monitoring and water quality sampling. • Monitoring plan should include recording water levels at the control structure; record water levels biweekly when water table is present and following precipitation events that result in high flows, this will inform adaptive management decisions. **Maintenance plan should include:** • Planned water level management and timing. • Inspection plan for water control structures, distribution pipes, and upstream surface inlets. • Recurring maintenance to, as necessary, remove larger trees/shrubs too close to the distribution line, to reduce plugging.
Technical Requirements	**Technical Considerations** • Determine the drainage system capacity: consider using drainage system drainage coefficient with area drained, drainage mainline capacity, or model simulations, such as DRAINMOD. • Use site specific drainage capacity information to compute minimum buffer dimensions needed to meet selected SRB design flow. • Refer to NRCS Practice Standard 606 (Subsurface drain) for additional criteria to consider when determining drain pipe size. • Consider soliciting planning assistance from engineer or other technically trained staff, such as that available from a local conservation district and/or NRCS office.
Lifespan	The lifespan of saturated riparian buffers depends on the quality of the components, maintenance, and environmental conditions.
Land Area Requirements	**Soil considerations and site selection** • Successful implementation of saturated buffers requires consideration of soil conditions- buffer soils and topography must be able to maintain a raised water table without adverse effects. • Proper perennial vegetation must be maintained within the riparian buffer, however, woody vegetation should

Considerations	Details
Land Area Requirements, continued	be avoided close to the distribution pipe, to avoid plugging of the pipe. • Successful function of this practice requires soils with suitable permeability characteristics and appropriate levels of organic matter. • Soil conditions, such as a restrictive layer, must be present to create saturated conditions when water is discharged from a subsurface drainage system into a riparian buffer. • There should be no soil voids or layers of high hydraulic conductivity soil that could provide preferential flow paths. • A minimum of 0.75% organic carbon (1.2 percent organic matter) should be present in the top 2.5 ft. of soil. **Topographical and location requirements** • To function appropriately, a functioning riparian buffer of at least 30 feet must be available to receive discharge from the drainage system (see recommendations of Chapter 12- Riparian Areas and Surface Water Management to determine the appropriate buffer size for your water body). • Additional siting aspects to consider include: ○ Location of animal waste and septic system effluent or animal waste. ○ Location of abandoned drainpipes that would continue to drain the buffer and prevent appropriate filtering of discharged material. ○ Landscape position and drainage patterns. ○ Property boundaries, structural elements, and utility locations. ○ The SRB area must be protected from development activities and potential sources of damage, to ensure long-term function and integrity. ○ Stream bank integrity and length of riparian forest buffer available. Older riparian vegetation within the buffer increases the nitrate removal effectiveness of the SRB.

Considerations	Details
Resources	Stream Visual Assessment Protocol 2

Blind Inlets

Use cases

Blind inlets are structures placed in the lowest point of a farmed depression to drain the water that collects there. Blind inlets are an alternative to tile risers, which provide a direct connection to the tile drainage system and often convey surface runoff, sediment, and associated pollutants directly into field drainage without treatment. Blind inlets provide the same service as tile risers but provide filtration to remove fine sediments and soluble pollutants.

Benefits

In contrast to tile risers, which include a pipe that protrudes above the surface of the ground and poses an obstacle to farm equipment, blind inlets provide the producer the ability to drive equipment over the inlet.

Blind inlets have been observed to remove significantly more sediment, phosphorus, and herbicides from field runoff than tile risers. Although the extent of nitrogen reduction found depends on the form nitrogen is in, nitrogen reductions from blind inlets met or exceeded those found in runoff from tile risers. See Appendix A for more details on pollutant and nutrient reductions.

Challenges

Blind inlets require occasional maintenance to ensure continuing effectiveness. Over time, the filter materials can become clogged with sediment and debris, reducing the inlet's ability to convey water and capture pollutants. Care is required while operating farm equipment near the blind inlet to avoid mixing soil and sediment into the sand filter layer. Operators should minimize driving over the inlet and lift implements while passing over the gravel/sand surface.

Because the blind inlet is designed to filter sediment from the water flowing to it, the surface will eventually become covered with sediment. Weeds and vegetation may also begin to grow on the surface. Both may adversely affect the inlet capacity. Therefore, sediment and vegetation should be periodically removed to a depth that exposes clean gravel/sand at the surface. Any filter materials removed along with the sediment should be replaced to maintain a filter layer of at least one foot thickness. To maintain higher infiltration rates for longer periods of time, producers could consider implementing additional BMPs, such as reduced or no-tillage, cover crops, and diverse rotations to improve soil health, thereby reducing soil erosion; this can extend blind inlet service life.

Additionally, because underground outlets may provide a direct conduit to receiving waters for contaminated runoff, they should be installed with accompanying structures or practices as part of a conservation system that addresses issues such as nutrient and pest management, residue management, and runoff filtration, such as controlled tile drainage and/or saturated riparian buffers.

Table 9. Implementation Considerations for Blind Inlets

Considerations	Details

Costs	**Initial investments**
	• Piping to convey stormwater to controlled tile drain and/or saturated riparian buffer:
	  ○ Corrugated single wall pipe: $0.50 - $3.00/ft (for typical agricultural sizes).
	  ○ Drainage lateral connection: $33.88 for 6" drainage lateral to main drain, including excavation to 6' depth, installation of and material for tee on main line, connect lateral, and backfill.
	• Costs of coarse and fine filtering material will vary depending on several factors, including delivery distance.
	• Large equipment rental and operation costs will vary, estimates include
	  ○ Light-heavy equipment operators: $32.21 to 51.15 per hour
	  ○ Skid steer loaders can range from $200-400 per day to rent
	Maintenance
	• Replacement of filter material to maintain at least 1 foot of material at the surface.
	Financial planning
	• This practice may be compatible with existing federal and state cost-share programs.
	• In July 2025, blind inlet installation with locally sourced materials is estimated to be $1,200 in northeastern Indiana (Ghane, 2025).

Operational and Maintenance Requirements	**Installation and operation** • Consider the soil stability and evaluate the need to wrap nonwoven geotextile fabric around the coarse material to keep sediment from adjacent soil profiles from moving into the blind inlet sediments.

Considerations	Details
	Maintenance • Cleaning to maintain performance and prolong service life. • Periodic removal of sediment and vegetation to a level that exposes clean gravel/sand. • Maintain a filter layer of at least one foot thick.
Technical Requirements	**Materials** • Filtering materials should be a combination of fine and coarse materials and may include gravel, limestone, pea gravel and sand. Consider adding adsorbent materials to remove soluble pollutants. • Consider soliciting planning assistance from engineer or other technically trained staff, such as that available from a local conservation district and/or NRCS office.
Lifespan	Lifespan is dependent upon regular maintenance of the blind inlet and the amount of sediment generated from field practices; may have a service life up to 10 years. Reduced or no-till field will extend the service life beyond 10 years.

Land Area Requirements	• Isolated depressional areas and areas where surface water accumulates.
	• Consider the location and function of controlled tile drainage and saturated buffer system that the blind inlet may connect to.
	• Additional locations aspects to consider include:
	  ○ Location of animal waste and septic system effluent or animal waste.
	  ○ Landscape position and drainage patterns.
	  ○ Property boundaries, structural elements, and utility locations.
	  ○ Common driving pathways.
	  ○ Inundation tolerance of the surrounding vegetation.
	  ○ Current precipitation patterns and anticipated changes due to climate change.

Considerations	Details
Other implementation factors	Tillage and soil disturbance can input sediment into the blind inlet, reducing its effectiveness and lifespan. Consider reduced or no-till field practices to reduce erosion and extend the life of blind inlets.
Resources	[Michigan State University Extension Bulletin E3454: Blind Inlet]

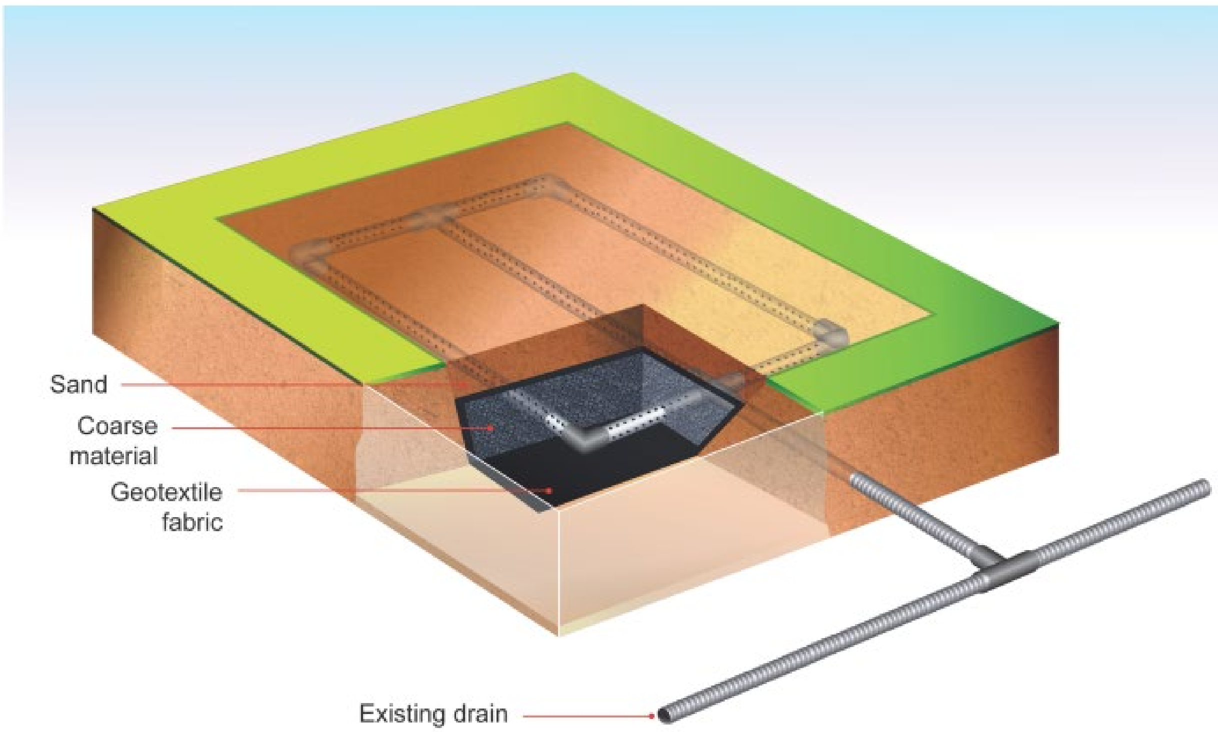

Figure 4. Blind inlet where the nonwoven geotextile fabric is wrapped around the coarse material layer (Ghane, 2025).