

Best Management Practices for Agricultural Ditch Management in the Phase 6 Chesapeake Bay Watershed Model



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Prepared for

Chesapeake Bay Program
410 Severn Avenue
Annapolis, MD 21403

Prepared by

Agricultural Ditch Management BMP Expert Panel:

Ray Bryant, PhD, Panel Chair, USDA Agricultural Research Service
Ann Baldwin, PE, USDA Natural Resources Conservation Service
Brooks Cahall, PE, Delaware Department of Natural Resources and Environmental Control
Laura Christianson, PhD PE, University of Illinois
Dan Jaynes, PhD, USDA Agricultural Research Service
Chad Penn, PhD, USDA Agricultural Research Service
Stuart Schwartz, PhD, University of Maryland Baltimore County

With:

Clint Gill, Delaware Department of Agriculture
Jeremy Hanson, Virginia Tech
Loretta Collins, University of Maryland
Mark Dubin, University of Maryland
Brian Benham, PhD, Virginia Tech
Allie Wagner, Chesapeake Research Consortium
Lindsey Gordon, Chesapeake Research Consortium

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Executive Summary

The Agricultural Ditch Management BMP Expert Panel convened in 2016 and deliberated to develop the recommendations described in this report in response to the Charge provided to the panel by the Agriculture Workgroup (Appendix D: Panel Charge and Scope of Work). The panel was instructed to review the available science on the nutrient and sediment removal efficiencies associated with agricultural ditch BMPs of particular concern to the Delmarva Peninsula. Specifically, the panel reviewed and assessed:

1. Blind inlets
2. Denitrifying bioreactors
3. Water Control Structures
4. Phosphorus removal systems
5. Saturated buffers
6. Gypsum Curtains
7. Two-stage ditches
8. Denitrifying curtains
9. Ditch dipouts (dredging)

The panel devoted one chapter of this report to each of the first five practices. In each of these chapters: terms were defined; relevant NRCS practice codes were identified; the available scientific literature from inside and outside the watershed was reviewed; reduction efficiency values for total , total phosphorus and sediment were recommended; ancillary benefits and hazards were discussed; and future research needs were identified (Table 1). The panel determined that there was insufficient research at this time to support reduction efficiency values for two-stage ditches, denitrifying curtains, gypsum curtains and ditch dipouts (see Future Research and Management Needs section).

Table 1 - Summary of recommended efficiency values for agriculture ditch BMPs

BMP	NRCS P Code	Reduction efficiency	Application	Credit duration
Blind inlets	620, 606	0% TN, 40% TP, 60% Sed.	Drained area (ac.)	5 Yr
Blind inlets w/ P-sorbing materials		0% TN, 50% TP, 60% Sed.	Drained area (ac.)	5 Yr
Denitrifying bioreactors*	605	20% TN, 0% TP, 0% Sed.	Drained area (ac.)	10 Yr
Water Control Structures	587	0% TN, 0% TP, 0% Sed.	Drained area (ac.)	N/A
Drainage Water Management	554	30% TN, 0% TP, 0% Sed.	Effective drainage control area (ac.)	Annual
P removal systems	782	0% TN, 50% TP, 60% Sed.	Drained area (ac.)	4 yr
Saturated buffers	604	20% TN, 0% TP, 0% Sed.	Drained area (ac.)	10 Yr

*In response to CBP partnership feedback the panel also accepted the inclusion of directly-measured reductions of nitrogen loads from bioreactors that treat springs or seeps; directly-measured systems will be annual BMPs. This practice is described in Appendix E.

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Acronyms

AgWG	Agriculture Workgroup
BMP	Best Management Practice
CAST	Chesapeake Assessment Scenario Tool
CBP	Chesapeake Bay Program
CBW	Chesapeake Bay Watershed
CBWM	Chesapeake Bay Watershed Model
CD	Controlled Drainage
DP	Dissolved Phosphorus
DWM	Drainage Water Management
EP	Expert Panel
EPA	Environmental Protection Agency
ET	Evapotranspiration
FD	Free Draining
GIT	Goal Implementation Team (CBP organizational construct)
HUC	Hydrologic Unit Code
N	Nitrogen
NRCS	Natural Resource Conservation Service (Division of USDA)
P	Phosphorus
PP	Particulate Phosphorus
RDM	Roadside Ditch Management
TN	Total Nitrogen
TP	Total Phosphorus
TSS	Total Suspended Solids
USGS	United States Geological Survey
USDA	United States Department of Agriculture
USDA-NRCS	United States Department of Agriculture-Natural Resource Conservation Service
USWG	Urban Stormwater Workgroup
WCS	Water Control Structure
WIP	Watershed Implementation Plan
WQGIT	Water Quality Goal Implementation Team
WTWG	Watershed Technical Workgroup

Background: Charge and Membership of the Expert Panel

Existing and soon-to-be-approved USDA-NRCS conservation practices related to agricultural ditches are not credited in the Chesapeake Bay Watershed Model (CBWM) for reporting state progress towards nutrient and sediment reduction goals. Currently, water control structures (WCS) is a Chesapeake Bay Program (CBP)-approved best management practice (BMP). Denitrifying ditch bioreactors, saturated buffers, and sorbing materials in ag ditches are recognized interim practices available for state-use in planning scenarios. Agricultural BMPs installed in ditch systems represent a potentially significant source of nutrient loss reduction credit in the Chesapeake Bay, particularly on the Delmarva Peninsula located in the Coastal Plain region of the Chesapeake Bay Watershed (CBW). Seventy percent of Delaware's tax ditches are in the CBW. In Maryland, 821 miles of ditches drain approximately 183,000 acres of land, most of which is located within the CBW.

Table 2 - Panel members and support

Panel member	Affiliation
Ray Bryant, PhD, Panel Chair	USDA Agricultural Research Service
Ann Baldwin, PE	USDA Natural Resources Conservation Service
Brooks Cahall	Delaware Department of Natural Resources and Environmental Control
Laura Christianson, PhD PE	University of Illinois
Dan Jaynes, PhD	USDA Agricultural Research Service
Chad Penn, PhD	USDA Agricultural Research Service
Stuart Schwartz, PhD	University of Maryland – Baltimore County
Support to the panel provided by: Clint Gill (DE Dept. of Agriculture); Loretta Collins (U. of Maryland); Jeremy Hanson (Virginia Tech); Mark Dubin (U. of Maryland); Brian Benham (Virginia Tech); Allie Wagner and Lindsey Gordon (Chesapeake Research Consortium); Andy Ward, PhD, Ohio State University, retired	

The panel was charged by the Agriculture Workgroup (AgWG) to review the available science on the nutrient and sediment removal efficiencies associated with agricultural ditch BMPs of particular relevance to the Delmarva Peninsula but applicable across the CBW, where appropriate.

The Panel was requested to:

- Define which specific BMPs have sufficient research to warrant inclusion in the CBWM.
- Define the conditions in which a reporting agency can receive a nutrient and/or sediment reduction credit for a BMP.
- Define the units to report practices to the CBWM.
- Recommend procedures for reporting, tracking and verification of the BMPs.
- Analyze any potential unintended consequences associated with the BMPs.

BMPs to Review:

The list of BMPs tasked to this panel for consideration of nutrient and sediment reductions in the CBWM are included with brief descriptions below (Table 3). As the panel reviewed available information they refined the list of practices that could reasonably be given an effectiveness estimate for nutrients or sediment at this time. The bulk of this report describes each respective practice or group of practices and the panel's recommended effectiveness estimates for practices with sufficient available information.

Table 3 - Overview of practices considered by the expert panel

Practice:	NRCS Code:	NRCS Definition:	Applicable NRCS Purposes:
Underground Outlet (Blind Inlets)	NRCS Code 620	A conduit or system of conduits installed beneath the surface of the ground to convey surface water to a suitable outlet. Blind inlets allow entry of surface water from small ponded areas into the drain without an open riser.	To carry water to a suitable outlet from terraces, water and sediment control basins, diversions, waterways, surface drains, other similar practices or flow concentrations without causing damage by erosion or flooding. Appropriately designed blind inlets keep sediment out of the underground conduit.
Subsurface Drain (Assessed as a component of Blind Inlets)	NRCS Code 606	A conduit installed beneath the ground surface to collect and/or convey excess water.	Remove or distribute excessive soil water. Erosion and nutrient loss control.
Structure for Water Control	NRCS Code 587	NRCS Definition: A structure in a water management system that conveys water, controls the direction or rate of flow, maintains a desired water surface elevation or measures water.	Control the elevation of water in drainage ditches. Provide silt management in ditches.
Drainage Water Management (Assessed as a component of Structure for Water Control)	NRCS Code 554	NRCS Definition: The process of managing water discharges from surface and/or subsurface agricultural drainage systems.	Reduce nutrient loading from drainage systems into downstream receiving waters. Reduce oxidation of organic matter in soils. Reduce wind erosion or particulate matter emissions.
Denitrifying Bioreactor (The current NRCS standard applies only to subsurface flow, the panel will be examining the same technology applied to open agricultural ditches.)	NRCS Code 605	A structure that uses a carbon source to reduce the concentration of nitrate nitrogen in subsurface agricultural drainage flow via enhanced denitrification.	Improve water quality by reducing the nitrate nitrogen content of subsurface agricultural drainage flow.
Phosphorus Removal System	NRCS Code 782	A system designed to remove dissolved phosphorus (P) from surface runoff, subsurface flow, or groundwater. The system should generally consist of a filter media with a high affinity for dissolved phosphate P, a containment structure that allows flow through the media and retains the media so that it does not move	This standard establishes the minimum requirements to design, operate, and maintain a flow-through P removal system. The system is intended to improve water quality by reducing dissolved phosphorus loading to surface water through the sorption of phosphate P from drainage and runoff water.

		downstream, and a means to remove and replace the filter media.	
Gypsum Curtain (Assessed As a component of Phosphorus Removal System)	NRCS Standard in Development	An underground vertical wall of gypsum installed running parallel to an agricultural ditch, designed to intercept groundwater flowing to the ditch. The gypsum in this system removes dissolved phosphorus from the groundwater.	This practice is a specially designed type of phosphorus removal system.
Saturated Buffer	NRCS Code 604	A subsurface, perforated distribution pipe used to divert and spread drainage system discharge to a vegetated area to increase soil saturation.	To reduce nitrate loading from subsurface drain outlets. To enhance or restore saturated soil conditions in riverine, lacustrine fringe, slope, or depression hydrogeomorphic landscape classes.
Open Channel (Two-Stage Ditch)	NRCS Code 582 (Indiana NRCS FOTG)	Constructing or improving a channel, either natural or artificial, in which water flows with a free surface.	To provide discharge capacity required for flood prevention, drainage, other authorized water management purposes, or any combination of these purposes.
Channel Bed Stabilization (Assessed As a component of Two-Stage Ditch)	NRCS Code 584	Measure(s) used to stabilize the bed or bottom of a channel.	Modify sediment transport or deposition. Manage surface water and groundwater levels. One purpose of a two-stage ditch is to stabilize the channel.
Denitrifying Curtains		An underground vertical wall of sawdust enriched soil installed running parallel to an agricultural ditch, designed to intercept groundwater flowing to the ditch. The sawdust in this system is a carbon source for microbial denitrification that removes nitrates from groundwater.	This practice is a specially designed type of Denitrifying Bioreactor.
Ditch Dipouts		The removal of accumulated sediments in the ditch bottom. The panel will look into new technologies for performing the dipouts and possible mitigation of effects of reapplication of sediment spoil.	

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Background: Agricultural ditches and ditch management in the Chesapeake Bay Watershed

Poorly drained soils with seasonally high water tables can support high levels of agricultural productivity with subsurface drainage. The widespread implementation of ditch and tile drained systems can also significantly alter the regional water balance and accelerate the flux of drain water and dissolved nutrients to receiving waters, affecting both crop yields and receiving water quality (Zhang and Schilling 2006, Schilling et al. 2019, Gilliam and Skaggs 1986, Skaggs et al. 2012).

There are extensive networks of ditches throughout the Chesapeake Bay Watershed, and installation of tile drainage systems have increased in recent years. However, many ditch systems consist of roadside ditches designed to capture and transport runoff from roadways or other developed areas. This report focuses on practices associated with agricultural ditch networks, which help manage drainage from cropland or adjacent agricultural land uses. Ditches can be adjacent to both roads and cropland, so roadside ditches and agricultural ditches are not mutually exclusive networks. One common difference is that the management of roadside ditches is often overseen by state or local transportation or highway agencies, whereas agricultural ditches are managed by Public Drainage Associations in Maryland and Tax Ditch Associations in Delaware. Many additional ditches that are privately-owned and managed feed into the public ditch systems.

It is not within the scope of this report to comprehensively categorize any given ditch as either a roadside ditch, an agricultural ditch, or both. For the purposes of this panel report, it helps to acknowledge that overlap between roadside and agricultural ditches does occur, but that this expert panel report is focused on management practices associated with agricultural ditches and drainage systems. Therefore, such practices are only applicable to agricultural load sources in the Phase 6 CBWM as described in the next section.

Note on roadside ditches and efforts by the Urban Stormwater Workgroup (USWG): In 2016, the (USWG) convened a Roadside Ditch Management (RDM) Team to consider the challenges and opportunities associated with roadside ditch management in response to recommendations from a 2014 STAC workshop (Schneider and Boomer 2016). The RDM team delivered its recommendations to the USWG and AgWG in summer of 2017. The USWG requested the CBP Goal Implementation Team (GIT) FY2018 funding to develop further guidance for enhanced treatment by roadside ditches. Following completion of that CBP GIT-funded project, there are ongoing efforts by the USWG to determine next steps for RDM practices identified in the RDM memo (Roadside Ditch Management Team 2017). Future decisions by the USWG could make RDM practices applicable to developed load sources in the CBWM, but are not discussed in this report.

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How agricultural ditches relate to load sources in the Phase 6 Watershed Model

The CBP uses the CBWM to understand and simulate changes in loads of nitrogen (N), phosphorus (P) and sediment to the tidal portion of the Chesapeake Bay due to management actions implemented in the 64,000 square mile watershed. The Bay states used the latest iteration (Phase 6) of the CBWM to develop their Phase III Watershed Implementation Plans (WIPs). The Chesapeake Assessment Scenario Tool (CAST; <http://cast.chesapeakebay.net/>) allows anyone to generate scenarios or access documentation, source data and reports associated with the Phase 6 CBWM. This section summarizes applicable aspects of the CBWM for the purposes of this report. Readers interested in comprehensive information should refer to CAST and the model documentation (<http://cast.chesapeakebay.net/Documentation/ModelDocumentation>).

The basic structure of the CBWM is illustrated in Figure 1. For this report, it helps to understand that agricultural ditch or drainage BMPs will be simulated as shown for BMPs generally in Figure 1. As suggested, like other BMPs they reduce loads from sources before those loads undergo other simulated attenuation or transport through the landscape and waterways represented by land-to-water factors and stream/river delivery, respectively. In this case, it is agricultural sources (e.g., cropland, pasture and hay) whose loads will be reduced by simulation of these BMPs.

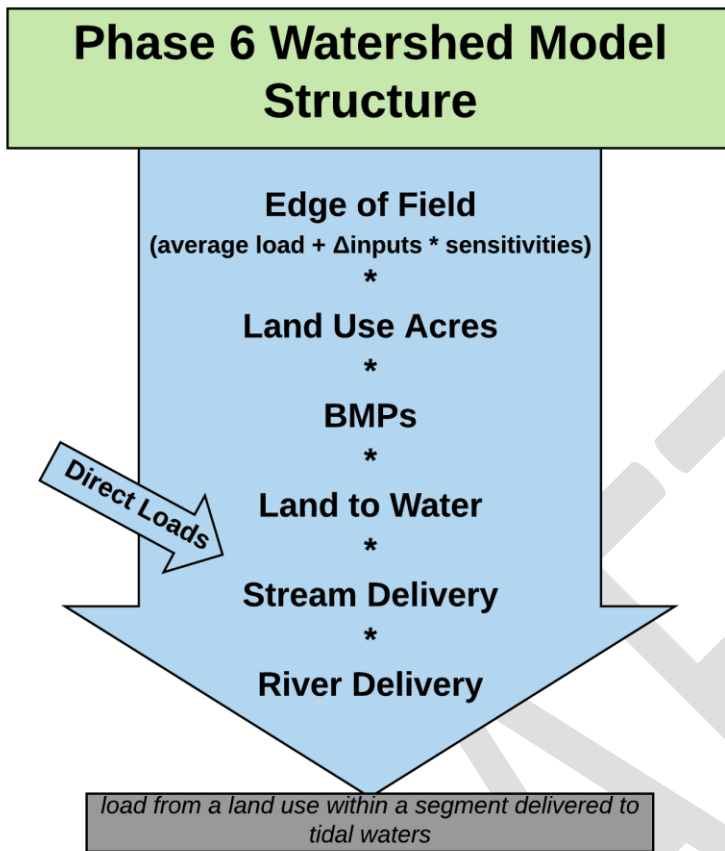


Figure 1 - General structure of the Phase 6 Watershed Model (adapted from Watershed Model documentation, chapter 1)

Agriculture sector loads are represented by various load sources, summarized in Table 4. Ditch and drainage system BMPs discussed in this report apply to the “AG” group of load sources in the CBWM or CAST unless otherwise stated.

Table 4 - Agriculture sector load sources in the Phase 6 Watershed Model. Source: CAST source data, load source definitions

Load Source	Load Source Minor	Description
Non-Permitted Feeding Space	Feeding Space	Non-permitted animal feeding areas including the barn and animal-intensive heavy use areas.
Leguminous Hay	Hay	Hay crops that include species that fix nitrogen such as alfalfa, vetch and clover.
Other Hay	Hay	Hay crops that exclude species that fix nitrogen such as alfalfa and clover. Includes haylage, grass seed, and failed crops.

Ag Open Space	Other Ag	Unmanaged agricultural land that receives no manure, biosolids, fertilizer or other nutrient applications.
Pasture	Pasture	Land used for pasture or grazing animals. Fertilizer and manure may be applied in addition to directly excreted manure.
Riparian Pasture Deposition	Riparian Pasture	The load that is delivered to the stream from direct excretion of animals. This does not encompass a land area.
Double Cropped Land	Row Crops	Double-cropped land represents areas that have two crops grown on the same acre between January and December. Crops eligible for double-cropping vary by state and may include alfalfa, barley, rye, small grain hay, sorghum for silage, soybeans, triticale, wheat, corn for silage or greenchop, and other haylage, grass silage, and greenchop. No other land use includes double cropping.
Full Season Soybeans	Row Crops	Soybeans that are not double-cropped
Grain with Manure	Row Crops	Includes the crops corn and sorghum for grain that is not double-cropped and receives inorganic fertilizer and manure where available
Grain without Manure	Row Crops	Includes the crops corn and sorghum for grain that is not double-cropped and receives only inorganic fertilizer
Other Agronomic Crops	Row Crops	Includes summer fallow, idle cropland, sod, tobacco, cotton, sweet corn, peanuts and dry edible beans
Silage with Manure	Row Crops	Includes the crops corn and sorghum for silage or greenchop that is not double-cropped and receives fertilizer and manure where available
Silage without Manure	Row Crops	Includes the crops corn and sorghum for silage or greenchop that is not double-cropped and receives only inorganic fertilizer
Small Grains and Grains	Row Crops	Includes canola, oats, rye, wheat, barley, buckwheat, emmer and spelt, and triticale that is not double-cropped
Specialty Crop High	Row Crops	Includes bedding/garden plants, cut florist greens, potted plants, mushrooms, other nursery and greenhouse crops, greenhouse vegetables, fruits and vegetables grown outside that are not included in Specialty Crop Low
Specialty Crop Low	Row Crops	Includes aquatic plants, orchards, Christmas trees, asparagus, nursery stock, short-rotation woody crops, sunflower seed, berries, peas, lima and snap beans

Permitted Feeding Space	Feeding Space	Permitted concentrated animal feeding areas including the barn and animal-intensive heavy use areas.
<p>Notes:</p> <p>“Load Source” includes land uses, which encompass a land area in the model, and other sources that do not encompass a land area. For example, Pasture is a land use with defined land area, whereas Riparian Pasture Deposition does not have a land area. Pasture and Riparian Pasture are both load sources in the Phase 6 Watershed Model.</p> <p>“Load Source Minor” is a basic grouping for the various load sources within a sector.</p>		

Other sectors – i.e., developed, natural, septic and wastewater – are not discussed for the purposes of this report, but they are briefly summarized in Table 5 for the reader.

Table 5 - Overview of non-agriculture sector load sources categories in the Phase 6 Watershed Model

Other sources		
Sector	Load sources	Summary
Developed	Impervious Developed; Pervious Developed; construction	Developed land uses are divided among combined sewer system areas (CSS), regulated MS4 areas (MS4), and non-regulated areas. Developed land uses include active construction areas; impervious areas like roads, buildings/other, tree canopy, and turfgrass.
Natural	Forest; Wetlands; Open Space; Shoreline; Stream Bed and Bank	
Septic	Septic and Rapid Infiltration Basins	There is no land area associated with septic systems, but their contributed loads are simulated as a load source.
Wastewater	CSO; Industrial WWTP; Municipal WWTP	Includes the loads from NPDES permitted facilities and regulated CSO areas.

Agricultural ditch and drainage systems are common in places like the Delmarva Peninsula on the Coastal Plain of the CBW, where soils and topography do not always effectively drain precipitation from fields or other areas. Large networks of these ditches are publicly managed through public drainage associations (PDAs), public watershed associations (PWAs), tax ditch organizations or other entities based on federal, state or local laws and regulations. In Maryland’s portion of the Eastern Shore there are over 821 miles of publicly managed ditches that drain over 183,000 acres of cropland, forest, roadways and residential or commercial developments (MDA, 2019). Delaware has another 2,000 miles of public tax ditches statewide

that drain agricultural and developed lands (DNREC, 2019). It is generally understood that hundreds miles of privately managed ditches exist throughout the Delmarva Peninsula, but data and estimates are not available. These ditch networks are a significant factor in overall water drainage and thus potential transport for nutrients and sediment. However, ditches are not explicitly simulated within the model as a unique load source or land use. There are ongoing efforts to map ditch networks using high-resolution land use for the Chesapeake Bay Watershed, but that aspect of the work by the Chesapeake Conservancy will continue past 2020 and primarily inform future iterations of the model (Claggett, pers. comm., 2019; Chesapeake Conservancy, 2019). In the current model, the contribution of ditches are implicitly captured through the process whereby the watershed model is calibrated to measured river loads. Therefore, any ditch or drainage BMPs described in this report are applicable to non-animal agriculture load sources (“AG” load source group) in the CBWM unless otherwise specified.

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Blind inlets

Terms and definitions

Phosphorus sorption material (PSM) – solid media that has an affinity for DP. Used as a filter material in P removal structures and potentially, blind inlets. PSMs are often industrial by-products rich in iron, aluminum and/or calcium and magnesium.

Subsurface (Tile) Drain - A conduit installed beneath the ground surface for collecting and/or conveying excess water.

Surface (Ditch) Drain - A graded channel on the field surface for collecting and/or conveying excess water.

Tile riser – a perforated pipe extending vertically out of the ground that is connected to a subsurface tile drain pipe. These are used to drain depressions in poorly drained locations within a field.

Tile Well – an open ended pipe extending vertically out of the ground that is connected to a subsurface tile drain. These are used to drain depressions in poorly drained locations within a field.

Open Inlet – tile risers and tile wells are two types of open inlets that allow unfiltered surface water to directly enter a subsurface tile drain. These are used to drain depressions in poorly drained locations within a field.

Blind Inlet - a type of French drain attached to a subsurface tile drain, where perforated pipe is placed at the bottom of an excavated hole that is then backfilled with pervious material (gravel and sand). The uppermost gravel or sand layer is covered with soil. The blind inlet acts to filter drainage water prior to it entering the subsurface tile drain. Blind inlets are used to drain depressions in poorly drained locations within a field.

Gravel Inlet- same in design as the blind inlet- except that the uppermost gravel or sand layer is not covered by soil at the time of construction. In time, soil that washes from the surrounding depression covers the uppermost sand or gravel layer, effectively converting it to a blind inlet. The terms blind inlet and gravel inlet are often used interchangeably. The panel makes no distinction between blind inlets and gravel inlets in terms of sediment and nutrient reduction.

Particulate phosphorus (PP) – P that is bound to the surface of transported sediment

Dissolved phosphorus (DP) - P that is dissolved in solution

Total phosphorus (TP) - Sum of particulate and dissolved P

Specific practices/approaches/NRCS CP codes included and excluded under this practice/category

Blind inlets are structures that drain depressions in fields that are otherwise poorly drained. Blind inlets are intended to reduce the time of ponding that often occurs in such depressions, thereby reducing crop damage and allowing field operations. The blind inlet is a type of French drain, where perforated pipe is placed at the bottom of an excavated hole that is then backfilled with pervious material (gravel and sand). A gradation of particle size is typically used,

with coarser materials placed at the bottom and finer materials near the surface. The uppermost gravel or sand layer is covered with soil. The enveloped perforated pipe is connected to a tile drain that usually drains into a ditch. Surface water that reaches the depression must flow through the soil and pervious material before discharge through the enveloped tile drain pipe, resulting in the accumulation of sediment at the surface. A geo-textile filter fabric is often used to separate the coarse gravel from finer materials placed on top of it. Details of blind inlet construction are found in Smith and Livingston (2013). The terms, “gravel inlet” and “blind inlet” are often used interchangeably, although they are technically different in that, at the time of construction of a gravel inlet, the uppermost gravel or sand layer is left exposed at the surface. However, soil that washes from the surrounding depression eventually covers the uppermost sand or gravel layer, effectively converting a gravel inlet to a blind inlet. The panel makes no distinction between blind inlets and gravel inlets in terms of sediment and nutrient reduction. Figure 2 illustrates a traditional blind inlet.

For clarity, the panel henceforth uses the term “blind inlet” to refer to both construction designs. From the perspective of BMPs and watershed modelling of water quality, it is important to note that addition of a blind inlet to a field that did not previously have a tile riser already in place will actually decrease water quality, because a blind inlet will serve as a direct conduit for dissolved nutrients in surface water to a ditch via a tile drain. ***Therefore, from the perspective of water quality, the benefit from installation of a blind inlet is only realized when the blind inlet replaces an existing open inlet.***

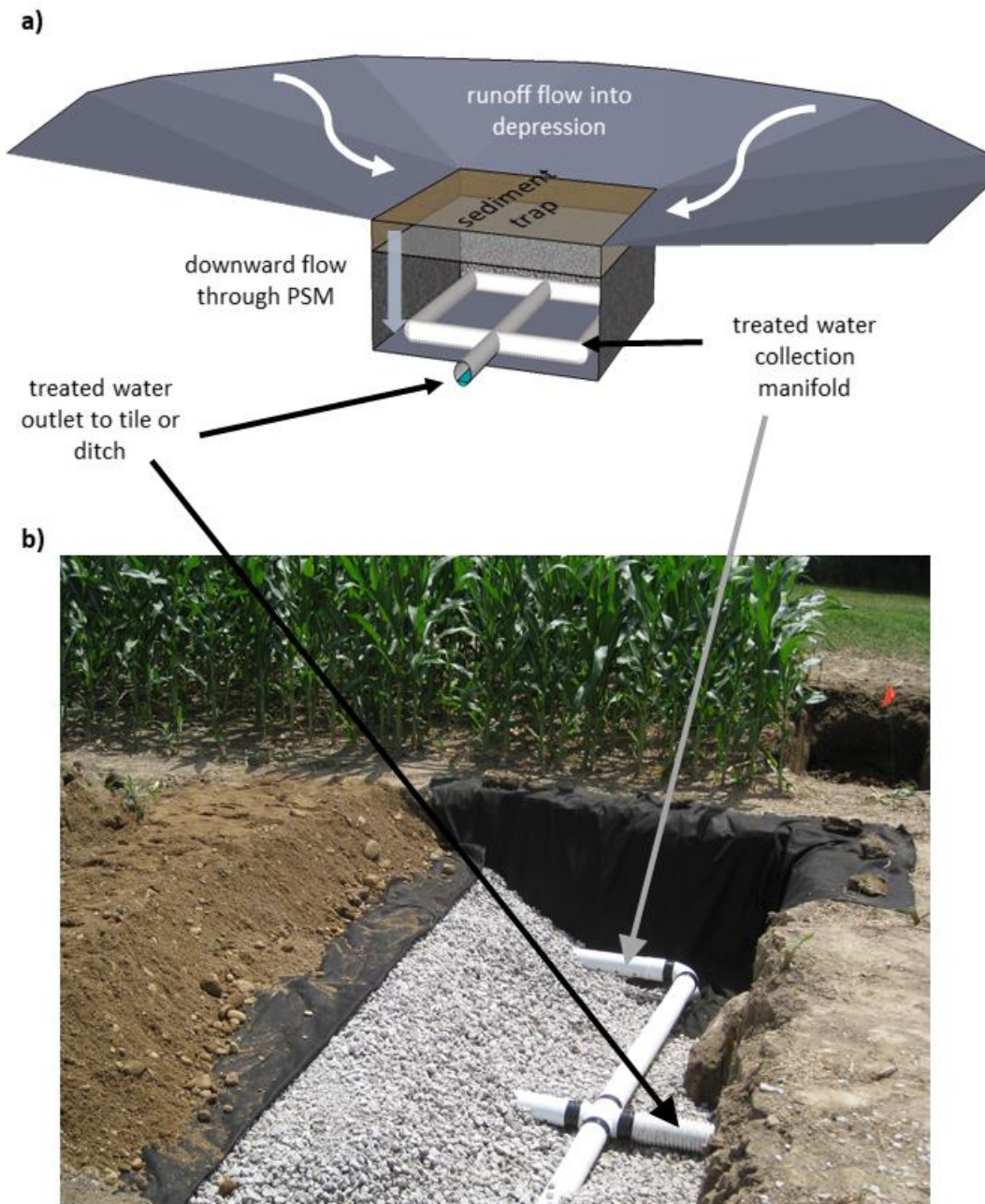


Figure 2. (a) Diagram of a typical blind inlet and (b) cutaway of a blind inlet in the field. Upper image from Penn and Bowen, 2017, lower image is courtesy of USDA-ARS.

It is important to keep in mind that the blind inlet serves the same purpose as an open inlet (tile riser or tile well; Figure 3), but with the added intention of trapping more sediment. It also

allows field operations to proceed uninterrupted across the area, as there is no structure above the soil surface.



Figure 3. Surface tile risers for draining surface water from poorly-drained depressions (Left image from USDA-ARS magazine; right image from Scarve.net)

Although some ponding will still occur for both open inlets and blind inlets, resulting in sedimentation in the surrounding area, blind inlets trap more sediment, and therefore PP, compared to an open inlet, which allows suspended sediment to enter the subsurface drain. The porous gravel/sand and soil cover in a blind inlet allows it to behave as a particle collector/filter by increased impediment of water, in comparison to an open inlet. For this reason, the blind inlet is referenced under NRCS Conservation Practice Standard code 620 (Underground Outlets) as an alternative to open inlets.

From the perspective of BMPs and watershed modelling of water quality, it is important to note that addition of a blind inlet to a field that did not previously have a tile riser already in place will actually decrease water quality, because a blind inlet will serve as a direct conduit for dissolved nutrients in surface water to a ditch via a tile drain. **Therefore, the benefit from installation of a blind inlet is only realized when the blind inlet replaces an existing open inlet.**

When utilizing sand and gravel as the filtration media, blind inlets are expected to remove little to no DP, as shown by Feyereisen et al. (2015) below.

Review of science and literature

Outside of Watershed, Peer Reviewed

Initial evaluation of the blind inlets showed that they could effectively reduce sediment and TP losses. In Minnesota (MN), Feyereisen et al. (2015) monitored drainage from a 65-ha field containing 24 open inlets for three years before converting them to blind inlets and continued

monitoring. Median total suspended solids (TSS) concentrations were significantly reduced from 97 to 8.3 mg L⁻¹, and median DP from 0.099 to 0.064 mg L⁻¹ after converting from open inlets to blind inlets. Although statistical comparison of TSS loads was not possible due to equipment malfunction, values indicated that changing from open to blind inlets reduced TSS loads. However, DP loads did not significantly decrease with the change to blind inlets. The authors attributed this lack of reduction to use of a coarse media with poor P sorption capacity.

Performance of blind inlets in Indiana were reported in Smith and Livingston (2013), Smith et al. (2015), and Feyereisen et al. (2015). The blind inlets were constructed with limestone gravel and coarse limestone sand. Depending on the hardness of the limestone and how easily it dissolves, the limestone may have provided a source of calcium that could result in precipitation of DP. For discharge events from paired closed depressions (~ 4ha) at the field scale, Smith and Livingston (2013) reported decreases in total loading for sediment (8.8 to 79.4%), DP (65.1 and 71.9%), total Kjeldahl P (50.1 and 78%), and total Kjeldahl nitrogen (TKN; 55.3 and 63.9%), for 2009 and 2010, respectively, for blind inlets compared to open inlets. Feyereisen et al. (2015) also reported results for the Indiana sites during the growing season for years 2006 to 2013; sediment loads, DP loads, and TP loads decreased from the blind inlet relative to the open inlet by 59, 60, and 57%. The authors concluded that the Indiana blind inlets will be effective beyond a 10-yr service life.

For the blind inlets in Indiana, catchment scale monitoring (“AME,” the control catchment drained with tile risers and “BME,” the treatment catchment drained with blind inlets) also revealed decreases in the loading for the same parameters, although direct comparisons were not possible. The pre-treatment (i.e. before blind inlet) increased in nutrient and sediment loading after installation of the blind inlet (i.e. BME), but increased less compared to the paired catchment in which the open inlet was not changed (i.e. AME).

While the data from the MN studies provided valid information, the previously reported results on the Indiana blind inlets are now considered invalid. In an analysis of all the data from 2005 to 2018, Williams et al. (2019) determined that the paired closed depressions in Indiana that were used to assess tile risers vs. blind inlets were hydrologically different from each other. The authors then went on to conduct a valid statistical analysis of the long term experiment using before-after control-impact (BACI) technique, and found that there was no difference in flow-weighted mean concentrations of DP, TP, and nitrate, between the tile riser and blind inlet. The BME and AME sites in Indiana provide little information (Smith and Livingston, 2013), because the paired control site was eliminated only one year after the blind inlet was installed.

Regarding DP, there is no valid evidence, nor is it expected, that sand and gravel would remove DP. Wang et al. (2014) showed -4 to 3.1% removal of DP with limestone gravel used in runoff interception trenches. However, replacement of sand and gravel with PSMs such as steel slag (Penn et al. 2012; Penn et al., 2016) would result in some DP removal, depending on the mass of the material used and the P loading for the site.

Current Research: Both peer and Non-peer reviewed

The USDA National Soil Erosion Research Laboratory (NSERL) has constructed a new blind inlet in 2015 that allows for monitoring inflow to the blind inlet, and comparison to the outflow treated water. The data have yet to be processed. Still, that experiment will not provide any information about how the blind inlet performs relative to the standard practice of a tile riser.

Again, the blind inlet is only able to provide a water quality benefit when it is used to replace a currently existing tile riser.

The NSERL recently conducted an experiment where the 11-year old Indiana blind inlet described in Feyereisen et al. (2015) (labelled as “ADE” in that paper and within the current document) was dissected in an attempt to quantify how much PP was trapped. Because the ADE site was not in blind inlet mode at all times over the 11-year evaluation period, it is impossible to know exactly how much of the sediment deposition can be attributed to the blind inlet vs. the tile riser. However, between January 2006 and October 2017, ADE was in blind inlet mode for 76% of the 549 precipitation events > 6mm, and for 67% of the 150 flow events. In addition, in a hydrologic assessment of ADE and another blind inlet located in the same field (ADW), Williams et al. (2018) determined that when ADE was in tile riser mode, the time to peak flow rate and duration of flow was significantly less while peak flow rate, average flow rate, and cumulative flow volume were all significantly greater. All of this suggests that although it cannot be assumed that all deposition at ADE can be attributed to the blind inlet, the majority of deposition probably occurred while in blind inlet-mode. Based on a statistical comparison between the original sand media used to construct the blind inlet and the different blind inlet layers, it was determined that the blind inlet captured 1435 kg of sediment and 1 kg of sediment-bound P (Penn et al., 2019; in review). Analysis of the deposition layer clearly showed that clay, silt, TP and Mehlich-3 extractable P had accumulated at the surface of the blind inlet, which was highly enriched compared to the soils of the contributing area.

Sediment and particulate P (PP) removal

Theoretically, one can calculate the single collector efficiency of a blind inlet through the following (Ryan and Elimelech, 1996):

$$\text{sediment reduction} = 1 - \left(e^{-\frac{3(1-\text{porosity})\eta D}{4r}} \right) \quad (1)$$

Where sediment reduction is expressed as a proportion in decimal form, “porosity” is the porosity of the filter media (decimal form), η is the single collector removal efficiency, D is the thickness of the filter media bed (unit length), and r is the average radius of the filter media (unit length). The value for η is calculated as:

$$\eta = \frac{I}{UC_oA} \quad (2)$$

Where C_o is the sediment inflow concentration (mass per unit volume), U is the fluid approach (superficial) velocity (length per unit time), I is the sediment deposition rate that flows onto the

blind inlet (mass per unit time), and A is the surface area of the blind inlet. The U value is determined by:

$$U = \frac{Q}{A * porosity} \quad (3)$$

Where Q is inflow rate (volume per unit time). In practice, after the value for single collector removal efficiency (η) is determined, that value is inserted into equation 1 to calculate the sediment reduction value for the desired time period. Application of the target peak flow rate for the blind inlet to Q will result in the estimation of a worst-case scenario regarding sediment trapping. If one has knowledge of the particulate P concentration for the site, then particulate P reduction could additionally be estimated (see equations in Penn and Bowen, 2017, chapter 6).

The panel presents an attempt to simulate sediment and PP removal by a typical gravel/blind inlet of the Midwest. In this case, we are assuming a worst-case scenario in which a gravel inlet is initially constructed with steel slag (a P sorption material), and no sediment has yet been deposited on it. With time, as sediment accumulates, the gravel inlet is converted into a blind inlet. Again, this calculation is for the initial gravel only. Assumptions: average particle size of filter media = 50 mm, bulk density = 1.6 g/mL, hydraulic conductivity = 0.4 cm/s, porosity = 0.4, Inflow DP = 0.2 mg/L, TP = 1 mg/L, and sediment concentration = 195 mg/L (from Feyereisen et al., 2015). Using the relationship between average flow rate and total suspended solids from Feyereisen et al. (2015), this sediment concentration would result from about 16 L/s flow rate. From these values, a deposition rate of 185 g/min is estimated to flow onto the blind inlet. Using a design curve representative of a PSM with a low affinity for DP (Penn and Bowen, 2017), the overall TP removal is estimated to be 78% with 87% of the PP removed. Percent sediment removal is equal to % PP removal. Of this TP load, 25% removal is due to DP during the first year, and 53% is due to sediment removal. After 3 years, the cumulative DP removal will have decreased to 18%, decreasing the cumulative TP removal to 73%. After sediment deposition covers the gravel and changes the average particle diameter to 4 mm, the sediment removal is predicted to be 100%. Changing the inflow sediment deposition rate to lower values will greatly decrease the efficiency. For example, decreasing the sediment deposition rate to only 5 g/min will reduce the PP removal to 13% for the initial slag gravel with no overlying soil. On the other hand, decreasing the sediment concentration will increase removal.

Recommended effectiveness estimates, default values

Although reliable data on reductions in total sediment loads and TP loads resulting from replacing a tile riser with a blind inlet are extremely limited at this time, the fundamental principle of filtration afforded by the blind inlet supports a decision to assign a conservative efficiency value for sediment and TP. The panel recommends assigning a 60% reduction efficiency for total sediment load and 40% for TP load over a period of five years (assuming no DP removal and 0.2 mg/L DP concentrations). This decision is based on indications that the true reductions may be much higher. The panel recommends no reduction in N load from the drainage area resulting from installation of a blind inlet. Blind inlets constructed with P sorption materials should receive the same credit for sediment removal (60%), but 50% TP load reduction due to the additional removal of DP.

The area that drains to the blind inlet is the preferred metric for tracking and reporting eligible blind inlets for simulation in the Watershed Model. If the drained area is unknown the panel recommends a default conversion of 1 acre per blind inlet. This conversion rate is conservative based on the panel's experience and the variation and uncertainty associated with blind inlets' drainage areas. Available information does not allow for a conversion to drained area from other metrics that may be associated with blind inlets, such as linear feet.

Ancillary benefits, potential hazards and unintended consequences

Replacement of open inlets with blind inlets may increase ponding due to decreased flow rates. This could be a problem for some sites and crops. Based on the work of Williams et al. (2018), the blind inlet infiltration rate decreased on a tile-drained field, at a rate described by the following: Infiltration rate (cm/h) = $-1.39 * \text{years since installation} + 21.98$.

From the perspective of BMPs and watershed modelling of water quality, it is important to note that addition of a blind inlet to a field will actually decrease water quality; this is because a blind inlet, although able to filter out some sediment, will serve as a direct conduit for surface water to tile drain-ditch. ***Therefore, the benefit from installation of a blind inlet is only realized when the blind inlet replaces an existing tile riser.***

Management requirements and visual indicators of effectiveness

If properly designed and installed, blind inlets require little or no maintenance. Normal field operations are allowed over the top of the blind inlet, but care should be taken to prevent rutting or compaction, as the area around the blind inlet will be the wettest part of the field. Visual signs of rutting or compaction may indicate that the practice has been compromised. Although the outlet of the tile draining the blind inlet cannot always be observed, if the outflow can be observed, it should be relatively clean and free of sediment. Visual signs of entrained sediment in the outflow signifies the presence of preferential bypass flow that is allowing sediment to enter the tile drain.

Future research needs

Better field assessments of the blind inlet are needed in order to better quantify their effectiveness. **Sites should continue to be installed and monitored within the watershed which would require monitoring open inlets prior to replacing them with blind inlets and continued monitoring to assess sediment and nutrient reductions.**

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DRAFT

Denitrifying Bioreactors

Terms and definitions

Carbon Source – Medium (usually wood chips) in the media chamber which provides carbon (electron donor) for heterotrophic denitrification.

Hydraulic Retention Time (HRT) - The length of time it takes water to pass through the media chamber. A design HRT is technically defined as the media chamber volume times the media porosity divided by the bioreactor flow rate under expected highest flow conditions.

Media Chamber – The lined trench containing the carbon source through which water flows to be treated by denitrification.

Subsurface (Tile) Drain - A conduit installed beneath the ground surface for collecting and/or conveying excess water.

Surface (Ditch) Drain - A graded channel on the field surface for collecting and/or conveying excess water.

Water Control Structure (WCS)- A structure in a water management system that conveys water, maintains a desired water surface elevation, and controls the direction or rate of flow. For research purposes, it may also be used to measure rate of water flow.

Specific practices/approaches/NRCS CP codes included and excluded under this practice/category

Denitrifying bioreactors are structures that allow agricultural drainage water (ditch or tile-drained) to enter a carbon medium, achieve denitrifying status, and then reduce dissolved nitrate in this water to atmospheric nitrogen (N_2 gas). Factors affecting the performance of a bioreactor include hydraulic retention time (HRT), water temperature, and microbiology (Christianson et al., 2012). While there exist different designs of bioreactors within the CBW, only the denitrifying media chamber design are considered here. In this design, the discharge is channeled through a lined trench (media chamber, see Figure 4) filled with material (usually wood chips) that provides a carbon source for a microbial population (Addy et al., 2016). A bypass to convey excess flow is also an essential component in the design of this type of bioreactor so as not to reduce in-field drainage capacity and to ensure that flow through the media chamber not be less than the design HRT. Application of this technology for spring and seep discharges is described in Appendix E along with recommendations for receiving nutrient reduction credit, per request of the CBP partnership. Addressing discharges from spring and seeps is outside the charge assigned to this EP upon its formation.

This practice will be applied in crop fields where surface and subsurface drainage systems have been installed to remove excess water. A WCS installed at the inlet diverts water from the drainage system into the media chamber filled with a carbon source (See Figure 5). This WCS also serves to direct flow that exceeds bioreactor capacity (design HRT) to a bypass outlet. A WCS installed at the outlet of the media chamber controls the water level in the chamber and the HRT to achieve optimum treatment to remove nitrate. This WCS then discharges treated water to a surface water outlet. See illustrations below. **Note that WCSs used to direct flow**

through a bioreactor are not stand-alone BMPs. The WCS serves as a component of a denitrifying bioreactor and is not the component that reduces nutrient loads.

The bioreactor must be designed to meet NRCS Conservation Practice Standard 605, Denitrifying Bioreactor (NRCS CPS 605, 2015). The specified target is to treat one of the following: (1) 60% of the long-term average annual flow from the drainage system; (2) peak flow from a 10-year, 24-hour drain flow event, or (3) at least 15% of the peak flow from the drainage system. The bioreactor must be designed to achieve at least a 30% annual reduction in the nitrate-nitrogen load of the water flowing through the media chamber (NRCS CPS 605). The presence of a by-pass means some drainage flow (not more than 40% of the long-term average annual flow) will pass by the media chamber untreated. Based on these requirements, the minimum total annual nitrate-N load reduction required at the edge of the field is 18% (30% of 60%).

An operation and maintenance plan requires that water elevations (regulated by the water control structures) must be established and maintained during various seasons to achieve the desired performance. The carbon media generally has an expected useful life of 10 years. To extend the lifespan of the bioreactor, provisions should be made to replace the media. An outlet that will allow the media chamber to be drained during periods of no-flow or for maintenance should be provided.



Figure 4. Source: USDA NRCS 2011, Flickr. Photographer: Jason Johnson. Installation of denitrifying bioreactor

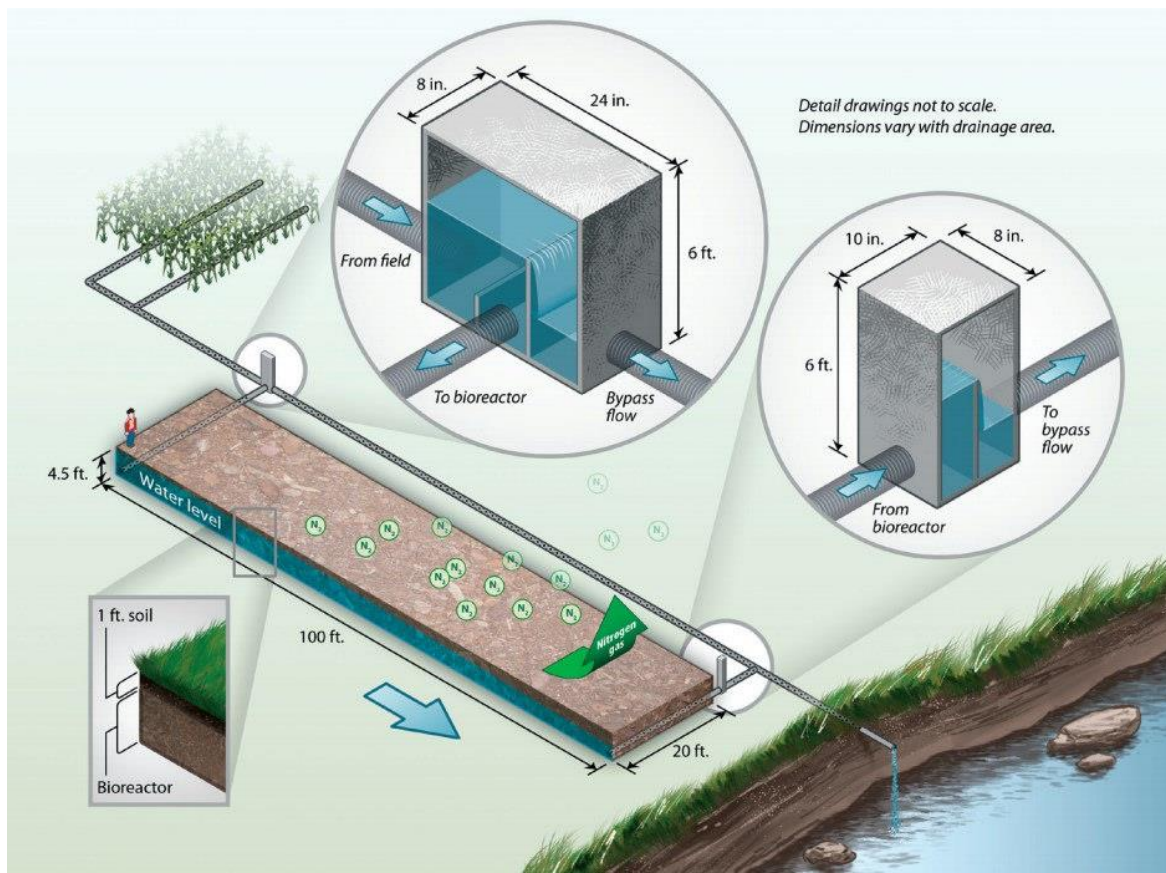


Figure 5. Source: Christianson and Helmers, 2011 (Iowa State University Extension and Outreach, PMR 1008). Illustration of bioreactor.

Review of science and literature

In Watershed, Peer Reviewed:

Rosen and Christianson (2017) examined three tile-drainage bioreactors in the Maryland region of the Delmarva Peninsula. Nitrate removal was achieved in all three sites during all monitoring periods with removal efficiencies of 9-62% (Table 6). Across sites and monitoring periods the removal efficiency was 24%, with highly variable flow and bioreactor performance across the dataset.

In the bioreactor near Ridgely MD, the study found that of the drainage water treated by the media chamber, greater than 96% of the nitrate load was removed. When bypass water was considered, the load reduction fell to 9-16% with only 13-21% of the flow being treated by the media chamber. The flow-weighted concentrations of nitrate into the media chamber ranged from 4.65-7.64 mg NO₃/L and fell to 0.06-0.30 mg NO₃/L leaving the media chamber. The study concluded that this site operated under N-limited conditions and efficiency could have been improved with a higher flow capacity. The study did note that this capacity could be hard to achieve due to a relatively flat hydraulic gradient, a condition not uncommon on the Delmarva.

The Queen Anne Farm bioreactor had a similar load reduction in water treated by the media chamber (>95%), however less water bypass led to a higher load reduction (47-62%) when bypass was included. Treated flow amounted to 50-59% of the total drainage and the retention

time was similar to the Ridgely Farm. The flow-weighted concentrations of nitrate into the media chamber ranged from 8.60-9.22 mg NO₃/L and fell to 0.08-0.23 mg NO₃/L leaving the chamber. The study concluded that this site also operated in an N-limited condition with a nearly ideal flow percentage treated and nitrate reducing conditions.

The Vorhees Farm bioreactor treated nearly all of the drainage flow (98%) but removed only 10% of the nitrate load. The HRT in this reactor was much shorter, 42-56 hours. The flow-weighted concentrations of nitrate into the reactor was 13.46 mg NO₃/L and fell to 11.57 mg NO₃/L leaving the reactor. The study concluded that this site was not N-limited and while treating nearly all of the water from the tile drainage system, could have benefitted from a longer retention time.

Table 6 - Summary of flow treated and nitrate removal within the bioreactor and considering bypass flow ("Total") for three bioreactors in Maryland. Source: Rosen and Christianson (2017)

	Flow		Bioreactor				Total (Including Bypass Flow)				
	Total Volume from Field	Percent Treated in Bioreactor	Flow-Weighted Concentration: IN	Flow-Weighted Concentration: OUT	Nitrate Load: IN	Nitrate Load: OUT	Nitrate Removal Efficiency	Nitrate Removal Rate †	Nitrate Load: IN	Nitrate Load: OUT	Nitrate Removal Efficiency
	m ³	%	mg NO ₃ -N/L		kg N		%	g N Removed per m ³ Bioreactor per d	kg N		%
Ridgely Farm											
8 August 2014–6 August 2015	37,000	13%	4.65	0.06	23	0.3	99%	0.40	251	229	9.0%
6 August 2015–4 May 2016 ‡	5860	21%	7.64	0.30	9.6	0.4	96%	0.21	58	48	16%
Queen Anne Farm											
8 August 2014–6 August 2015	24,400	59%	9.22	0.08	134	1.1	99%	5.36	214	81	62%
6 August 2015–13 April 2015 ‡	24,800	50%	8.60	0.23	106	2.9	97%	5.12	219.0	115.92	47%
Voorhees Farm											
19 December 2104–20 July 2015 ‡	49,700	98%	13.46	11.57	677	607	10%	1.53	688	618	10%

† Removal rate based only on dates when flow was occurring and calculated using the entire bioreactor volume (L × W × D).

‡ Not annual periods due to the grant timeline.

Based on N load weighting across all sites and monitoring periods, the total nitrate removal efficiency of these bioreactors was 24% (summation of Table 6's "Total Nitrate Load IN" minus the sum of "Total Nitrate Load OUT", the quantity of which was divided by the sum of "Total Nitrate Load IN": (1430-1092)/1430 = 24%).

Outside of Watershed, Peer Reviewed:

Christianson et al. (2012) provided a literature review of bioreactors specifically for subsurface agricultural drainage. Most of these studies took place in the Midwest and none were in the watershed. The studies include a wide range of influent concentrations, HRTs and removal efficiencies (See Table 7).

Table 7. Christianson et al. (2012). Review of denitrification treatment for agricultural drainage.

Source	Site	Volume (m ³)	Influent NO ₃ -N Concentration	Retention Time	Percent Reduction (concentration or load noted)	Nitrate-N Removal Rate	Notes
Field-Scale Drainage Treatment Studies							
Blowes et al., 1994	Ontario, Canada	0.2 (barrels)	2 to 6 mg/L	1-6 d	Nearly 100% concentration	NA	Partially buried in a stream bank
Wildman, 2001	South of Chatsworth, Ill. (#1)	27.2	approx. 4 to 16 mg/L	NA	Nearly 100% concentration	NA	4.0-ha treatment area
Wildman, 2001	South of Chatsworth, Ill. (#2)	27.2	approx. 1 to 18 mg/L	NA	Nearly 100% concentration	NA	5.3-ha treatment area
van Driel et al., 2006a	Ontario, Canada; lateral flow	17.2	11.8 mg/L (mean)	9 h (during tracer test)	33% concentration	12 kg N/yr; 2.5 g N/m ³ /d	Fine and coarse wood media
Jaynes et al., 2008	Central Iowa	38.9	19.1 to 25.3 mg/L (control plot)	NA	40%-65% load	0.62 g N/m ³ /d	Flow-through woodchip walls on sides of tile pipe
Moorman et al., 2010	Central Iowa	38.9	20 to 25 mg/L	24 h required to reduce influent to ≤10mg/L	NA	23.6 mg N/kg wood/d	Retention time conclusion based on field data
Chun et al., 2010	Decatur, Ill. (west)	55.8	269.9 g NO ₃ -N slug test	4.4 h	47% load	NA	2.0-ha treatment area
Verma et al., 2010	Decatur, Ill. (west)	55.8	Approx. 5 to >20 mg/L	NA	81%-98% load	NA	2.0-ha treatment area
Woli et al., 2010	East-Central Illinois (De Land, Ill.)	76.9	2.8 to 18.9 mg/L	26 min to 2.8 h	23%-50% load	6.4 g N/m ³ /d	14-ha treatment area
Verma et al., 2010	East-Central Illinois (De Land, Ill.)	76.9	Approx. 3 to 16 mg/L	NA	42% - 48% load	NA	14-ha treatment area
Verma et al., 2010	Decatur, Ill. (east)	NA	Approx. 4 to 15 mg/L	NA	54% load	NA	6.5-ha treatment area
Ranaivoson et al., 2010	Claremont, Minn.	NA	Approx. 11 to 28 mg/L	32 h for 50% conc. reduction	18%-47% load	NA	10.5-ha treatment area
Ranaivoson et al., 2010	Dundas, Minn.	NA	Approx. 7 to 14 mg/L	NA	35%-45% load	NA	
Christianson et al., 2012a	Central Iowa; Pekin	18	1.2 to 8.5 mg/L ^[a]	NA	22%-74% load	0.38-3.78 g N/m ³ /d	1.2-ha treatment area
Christianson et al., 2012a	Northeast Iowa, NERF	128	9.9 to 13.2 mg/L ^[a]	NA	12%-14% load	0.86-1.56 g N/m ³ /d	Trapezoidal cross section; 6.9-ha treatment area
Christianson et al., 2012a	Central Iowa, Greene Co.	127	7.7 to 15.2 mg/L ^[a]	NA	27%-33% load	0.41-7.76 g N/m ³ /d	19-ha treatment area
Christianson et al., 2012a	Central Iowa, Hamilton Co.	102	7.7 to 9.6 mg/L ^[a]	NA	49%-57% load	0.42-5.02 g N/m ³ /d	20.2-ha treatment area

Addy et al. (2016) performed a meta-analysis of 26 peer reviewed articles which included 27 bioreactors ("bed units"). This analysis included removal of nitrate only in the portion of water passing through the bed, the bypass water was not factored in. This study found that there was no significant difference in removal across wood sources (2.6-3.7 g N m⁻³ d⁻¹), however softwood may decay and deplete its carbon supply more quickly due to its low density. Influent N concentrations also had significantly different removal rates. Beds with concentrations higher than 30 mg N L⁻¹ removed 9.3 g N m⁻³ d⁻¹, 10-30 mg N L⁻¹ removed 4.9 g N m⁻³ d⁻¹, and less than 10 mg N L⁻¹ removed 2.4 g N m⁻³ d⁻¹. HRTs greater than 6 hours were significantly more effective at nitrate removal (4.4-6.7 g N m⁻³ d⁻¹) than less than 6 hours (0.7 g N m⁻³ d⁻¹). In addition, beds less than 13 months old had a significantly higher removal rate (9.1 g N m⁻³ d⁻¹) than beds from 13 to 24 months old and beds greater than 24 months old (2.8-2.6 g N m⁻³ d⁻¹), concurring with previous studies that rates after the first year are closer to the long-term removal rates. Bed N limitation also had a significant difference with the non-nitrate limited beds removing 6.7 g N

$\text{m}^{-3} \text{d}^{-1}$ and the nitrate-limited beds removing $3.2 \text{ g N m}^{-3} \text{d}^{-1}$. Finally, the bed temperature fit the general trend of increasing biological activity for increasing temperature; beds with temperatures less than 6°C removing $2.1 \text{ g N m}^{-3} \text{d}^{-1}$, beds with temperatures between 6 and 16.9°C removing $5.7 \text{ g N m}^{-3} \text{d}^{-1}$, and beds with temperatures greater than 16.9°C removing $86 \text{ g N m}^{-3} \text{d}^{-1}$. The mean removal of all bed-style bioreactors was $4.7 \text{ g N m}^{-3} \text{d}^{-1}$.

Jaynes et al. (2016) looked at simulating a bioreactor using a dual porosity model. In the process of fitting the data to the model they studied two years of a bioreactor's performance in central Iowa. The flow from two tile-drained plots totaling 0.26 ha was diverted into the bioreactor and in 2013, 370.9 m^3 was treated by the bioreactor. For this period, the inflow nitrate concentrations were between 11.5 and 15.8 mg N L^{-1} , while the outflow concentrations ranged from 0 to 12.2 mg N L^{-1} . The bioreactor removed 2.23 kg of nitrate which corresponded to 38% of the total entering the bioreactor. In 2014, 690 m^3 of drainage was diverted to the bioreactor. For this period, the inflow nitrate concentrations were between 4 and 16 mg N L^{-1} , while the outflow concentrations ranged from 0 to 12 mg N L^{-1} . The bioreactor removed 3.73 kg of nitrate which corresponded to 49% of the total entering the bioreactor.

Recommended effectiveness estimates, default values

Based on the requirements of CPS 605 for load reduction and the results of the studies cited, the panel recommends that proper installation and maintenance of this practice will achieve a 20% TN load reduction for the drainage system on which it is installed. It is understood, based on experience, that the load reductions will fluctuate from year to year, but over the expected 10 year life of the practice (unless renovated by replacing the carbon source) the average annual TN load reduction will be 20% or greater. Applying the practice to locations where nitrate loads per acre are greater will result in greater overall reductions in TN loading to the Chesapeake Bay.

Current science is not sufficient to support either a TP or a sediment load reduction credit for denitrifying bioreactors (see "Future research and management needs" section).

The drained area that flows into the bioreactor is the preferred metric for tracking and reporting the DNBR for simulation in the Watershed Model. If the drained area is unknown, the panel recommends a default conversion rate that assumes 5 drained acres per denitrifying bioreactor. This conversion rate is conservative, but reasonable, according to the panel's experience and based on lower-end values reported in the literature.

Ancillary benefits and potential hazards or unintended consequences

There may be a negative impact on crop performance if the water table upstream of the bioreactor is elevated to a level that limits aeration in the rooting zone for a prolonged period. However, producers on the Delmarva Peninsula generally seek to benefit from sub-irrigation by restricting drainage during the growing season.

Ponding on the surface of the bioreactor during storm events could negatively affect performance. Typically, the surface of the bioreactor is mounded to prevent ponding even after the woodchips settle.

Be aware of the effects on downstream flows or aquifers that would affect other water uses or users. For example, the initial flow from the bioreactor at start up may contain undesired

contaminants such as dissolved nutrients, organics, and tannins (CPS 605) and result in higher levels of N and P output due to the initial flush of organic material until the bed material is stabilized. The recommendations in this section assume that the denitrifying media is woodchips, which is the most common media reported in the literature. Some mixed media or amendments (e.g., biochar) could result in increased export of phosphate from the bioreactor. One recent laboratory study (Coleman et al., 2019) found that amending woodchip bioreactor columns with pine-feedstock biochar posed a substantial increase to phosphate leaching risk. However, those authors cite other research at field scales that suggest the negative impact of phosphate export may abate as the media ages over several years. Current design standards include woodchips, but not as the only option for media, so the panel acknowledges the greater uncertainty for some media amendments compared to woodchips, and recommends caution and diligence by the practitioner if considering amendments for the bioreactor chamber.

Management requirements and visual indicators of effectiveness

As mentioned previously, an operation and maintenance plan requires that water elevations (regulated by the water control structures) must be established and maintained during various seasons to achieve the desired performance. If the operation and management plan is being followed, visual inspection should reveal that flash boards in the water control structure are positioned to divert a portion of the flow volume through the media chamber. The water control structure should show no visual signs of damage that would prevent insertion or removal of flash boards. The media chamber has a mounded design that prevents water from the surrounding area from ponding on its surface. If ponded water is observed on the surface of the media chamber, it is a visual sign that the carbon source has settled or collapsed and therefore may have severely reduced flow through rate.

Future research needs

Whereas data on bioreactor performance is limited in the CBWM, the panel recommends that flow and nitrate concentrations continue to be monitored on bioreactor installations throughout the watershed in order to better document their overall effectiveness in this region.

Future research should account for different types of bioreactors such as in-ditch bioreactors and sawdust walls.

Denitrifying woodchip bioreactors are capable of removing TSS from both relatively high-solids aquaculture wastewater (Christianson et al., 2016) and agricultural wash-water (Choudhury et al., 2016). However, there is not sufficient evidence in the literature to credit a sediment load reduction for denitrifying bioreactor's treating tile or ditch drainage water. Additional research is also required to assess denitrifying bioreactor's impact on TP. The woodchips themselves can serve as a source of leached P, but under certain conditions, removal of both TP and DP by this technology has been documented (e.g., Goodwin, 2012; Zoski et al., 2013; Choudhury et al., 2016; Sharrer et al., 2016).

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Water Control Structures and Drainage Water Management

Terms and definitions

Surface (Ditch) Drain - A graded channel on the field surface for collecting and/or conveying excess water.

Subsurface (Tile) Drain - A conduit installed beneath the ground surface for collecting and/or conveying excess water.

Water Control Structure (WCS)- A structure in a water management system that conveys water, maintains a desired water surface elevation, and controls the direction or rate of flow. For research purposes, it may also be designed to measure rate of water flow.

Drainage Water Management (DWM) – Generally, the process of managing water discharges from surface and/or subsurface agricultural drainage systems. This section discusses the use of DWM for “controlled drainage” (CD) of agricultural fields to raise and lower the water levels within the soil profile throughout the year following an operation and maintenance plan. The terms DWM and CD are used synonymously throughout this section.

Specific practices/approaches/NRCS Conservation Practice codes included and excluded under this practice/category

This practice refers to NRCS practice 554, Drain Water Management (DWM), and provides BMP credit when DWM is used for “Controlled Drainage” (CD) of tile drained agricultural fields – i.e. to seasonally alter the water table elevation over the field, raising the water table to retain drainwater during the dormant season, and lowering the water table for trafficability and field operations. The terms DWM and CD are used interchangeably throughout this section. **This practice does not consider or provide BMP credit for other uses of DWM with WCSs, such as controlling the water table elevation to maintain constructed wetlands or flow regulation for bioreactors.**

Water control structures are a component in a water management system that conveys water, controls the direction or rate of flow, maintains a desired water surface elevation, or measures water. These structures may be installed for a wide variety of conservation purposes. The structure may be part of a wildlife project that requires modification of the water flow with chutes or cold-water releases. Examples of other uses of this practice include: sluices to provide silt management, screens to keep trash, debris, or weed seeds out of pipelines, tide gates to prevent backflow into a channel, and flow regulation components of bioreactors (flashboard risers). Not all of these uses result in nutrient load reductions.

To receive credit for nutrient reduction in the CBWM, WCSs must be a component of a DWM system designed and operated for the primary purpose of reducing nutrient loading from drainage systems into downstream receiving waters by restricting subsurface drainage from leaving the field. Water control structures that are components of other CBP BMPs, such as wetland restoration or denitrifying bioreactors, are not eligible for standalone credit under the drainage water management BMP defined in this section. The practice is reported in units

of acres, indicating the actual field acreage affected by the active management and seasonal adjustment of the water table elevation for the field.

The operation and management of the water control structure must meet the criteria and follow the operation and maintenance guidelines described in NRCS Conservation Practice Standard (Code 554), *Drainage Water Management*. Drainage water management is defined here as the process of managing water discharges from surface and/or subsurface agricultural drainage systems and does not apply to the management of irrigation water supplied through a subsurface drainage system.

Raising the outlet elevation of the flowing drain shall result in an elevated free water surface within the soil profile, which restricts drainage water and dissolved N and P from leaving the managed area and potentially promotes denitrification by generating reducing conditions in the upper part of the soil profile.

During non-cropped periods, the system shall be in managed drainage mode (elevated water table) within 30 days after the season's final field operation, until at least 30 days before commencement of the next season's field operations, except during system maintenance periods or to provide trafficability when field operations are necessary.

The drain outlet shall be raised prior to and during liquid manure applications to prevent direct leakage of manure into drainage pipes through soil macro pores (cracks, worm holes, root channels). Manure applications shall be in accordance with applicable state nutrient management guidelines or requirements or NRCS Conservation Practice Standards, Nutrient Management (590) and Waste Utilization (633).

An operation and maintenance plan shall be provided that identifies the intended purpose of the practice, practice life safety requirements, and water table elevations and periods of operation necessary to meet the intended purpose. If in-field water table observation points are not used, the relationship of the control elevation settings relative to critical field water table depths shall be provided in the operation plan. The operation and maintenance plan shall include instructions for operation and maintenance of critical components of the drainage management system, including instructions necessary to maintain flow velocities within allowable limits when lowering water tables.

Drainage water management and WCSs work in concert when utilized in an agricultural drainage ditch environment. The presence of a WCS is no guarantee that DWM is occurring. Drainage water management requires active management of the drainage outlet height, dropping the water table for times when crop management activities occur, and raising the water table to appropriate levels at other times.

Review of science and literature

Basic Operation Controlled Drainage (CD), also referred to as Drainage Water Management (NRCS 2012), refers to the use of a drain control structure to vary the controlling elevation at the outlet of an agricultural drainage system. The water table elevation over the drained field responds to changes in the controlling elevation at the drain control structure, with a response that varies with the spacing, depth, and orientation of drain lines. The field area influenced by the control structure also varies with both the field slope, soil texture and profile, and the intensity and capacity of the drain system.

Conceptually, the drain control structure is used to lower the water table and dry the soil to allow field operations for planting and harvest. During the growing season, the water table may

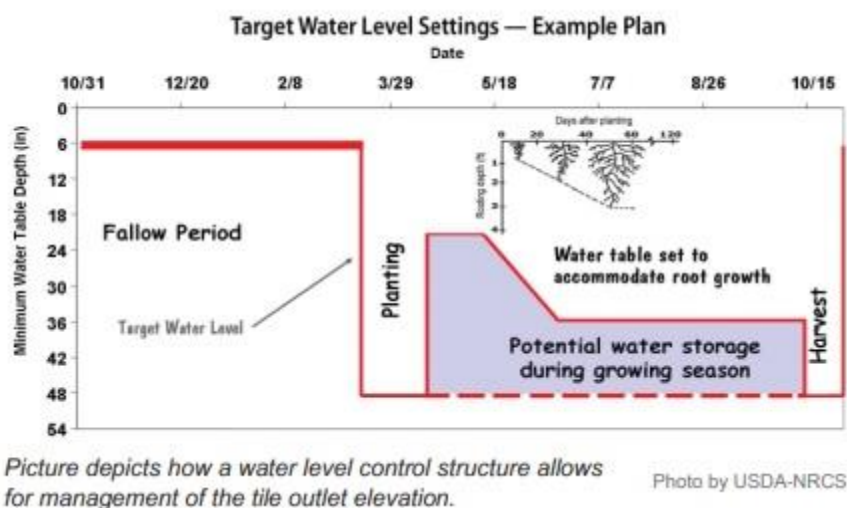


Figure 6 Drain Water Management

also be managed to maintain soil moisture in the root zone, slowly lowering the water table as roots grow deeper in order to maintain plant available water without saturating the root zone (Figure 6). Following harvest, the drain elevation is raised to capture and retain drainage water during the fallow period, until the next annual planting cycle.

Drain Design and Tradeoffs

Drain design for passive or free draining (FD) drainage systems are developed to provide cost-effective drainage for fields with poorly draining soils. Refined design guidance identifies drain design parameters intended to ‘optimize’ profit (yield x price – amortized drain costs) for field-specific (Skaggs and Chescheir 1999) or regional conditions (Skaggs et al. 2006, Skaggs 2007). By the 1970’s, the agricultural success of drain systems was tempered by recognition that increased drainage for improved agronomic productivity also impaired surface water quality due to the increased nutrient loads in drain water discharges to receiving waters (Bolton et al. 1970, Aldrich 1972, Green 1973, Baker et al. 1975, Gambrell et al. 1975a). The need to balance agronomic and environmental objectives in CD is now widely recognized, motivating the design

and operation of DWM systems for multiple objectives (Gilliam and Skaggs 1986, Gilliam et al. 1979, Skaggs and Gilliam 1981, Breve et al. 1998, Skaggs et al. 1994).

Optimizing Free Draining Systems

For a uniform Typic Umbraqualt soil in North Carolina (NC), Skaggs and Gilliam (1981) used model-based simulation to identify a range of different drainage intensities that all satisfied agronomic drainage objectives for the simulated field. The environmental impacts of these different agronomically acceptable drain designs spanned a *3-fold difference* in N loss and drain water discharge. Using DRAINMOD simulations, Skaggs et al. (1995) similarly found significant (47%) N reductions could be realized while meeting agricultural production objectives on a Portsmouth sandy loam, by increasing FD drain spacing from 20 m to 40 m. For the same soil, simulated drain spacing of 30m with CD realized even higher drain water nitrogen reductions (52%), while still satisfying agronomic goals. These results suggest the significant opportunities to reduce environmental impacts in the design of FD drain systems, by exploiting the equifinality of drain design for agronomic objectives. The sensitivity and extent to which optimized FD designs may enhance or constrain the marginal effectiveness of a drain control structure that is *retrofit* to an existing subsurface drain system, is less clear.

Benefits & Confounding Effects of Controlled Drainage

Field Access – CD allows the water table to be reliably lowered in poorly drained fields to assure trafficability of otherwise poorly drained fields, to enable timely planting and harvest.

Crop Yield – Improved yields may result from maintaining marginally higher root zone soil moisture during dry growing seasons, and draining excessive soil water to avoid an extended saturated (anaerobic) root zone during wet growing seasons. These marginal benefits may be less significant in extreme wet or dry years. Retaining nutrient enriched soil water may also enhance yields and reduce needed nutrient inputs.

Drainage reduction – The principal water quality benefit from CD is the reduction in drain water flows and their associated nitrate loads compared to free draining systems. Drain water reduction results from (a) vertical seepage of soil water retained during the fallow period, and (b) enhanced evapotranspiration (ET; ~<10%) of retained soil water during the growing season (Skaggs et al. 2010). Reductions in observed drain flows may also result from increased surface runoff from fields with a high water table, as well as lateral seepage of retained water that may drain to adjoining fields or receiving waters as shallow return flow. Observed reductions in the N load from drain water may overestimate the water quality benefit of CD if alternate flow paths for drain water return-flow are significant.

Nitrogen Reduction & Denitrification– Nitrogen loads to receiving waters are most commonly reduced by the reduction of drain water discharge with CD. Nitrogen discharges may also be reduced by crop uptake during the growing season (Poole et al. 2018).

Some studies have observed (El-Sadek et al. 2002) or simulated (Youssef et al. 2018) significant reductions in nitrate concentrations in drain water, consistent with denitrification. Maintaining a seasonally high water table can induce anoxic conditions in organic soils, providing favorable conditions for denitrification. Reduced nitrate concentrations, and reduced nitrate:chloride ratios have been observed in drain water, contributing to reduced nitrate discharges due to denitrification. CD fields on which deeper groundwater (below the drain depth) has been monitored, frequently show anoxic conditions, low N concentrations, and redox conditions consistent with denitrification (Gilliam et al. 1979, Burchell et al. 2005, Gambrell et al. 1975b). The contribution of denitrification to reducing nitrate loading by CD, depends on the reliable presence of organic rich reducing zones along the shallow flow paths *that contribute to drain flow*. Given the variability in soils, field topography, and drain system configurations, the mere addition of a WCS at the outlet of a tile drain system cannot automatically be assumed to significantly reduce N loads due to denitrification.

Design Considerations – CD is best suited for flat fields with poorly drained soils. As field slopes increase the field area influenced by the drain control structure decreases. The spacing, depth, and spatial configuration of drain lines (Figure 7) significantly affects the field-scale water table response and field area affected by drain control adjustments. New drain systems designed specifically for CD, layout the tile lines along the field contours, with breaks at field boundaries. The recommended change in field elevation managed by a single drain control structure is limited to about 35-45 cm (Cooke et al. (2006)). This criterion suggests a rough estimate bounding the change in the area influenced by CD, with field slope. For tile drains laid out on the contour, a drain control structure could be estimated to influence an upslope field width of 90 m for fields with a slope of 0.5% ($0.45 \text{ m} \div 0.05$). The estimated field width influenced by a control structure would be reduced to only 22.5 m on a 2% slope. Older drain systems, installed to achieve drainage and crop goals, are more commonly laid out to minimize the total length (and costs) of tile drainage, while conforming to locally recommended drain spacing and depth (Skaggs 2007). The effect of drain system layout (as in Figure 7, taken from Cooke et al. (2006)) and design on the performance of a drainage system that has been *retrofit* for CD, has not been thoroughly examined.

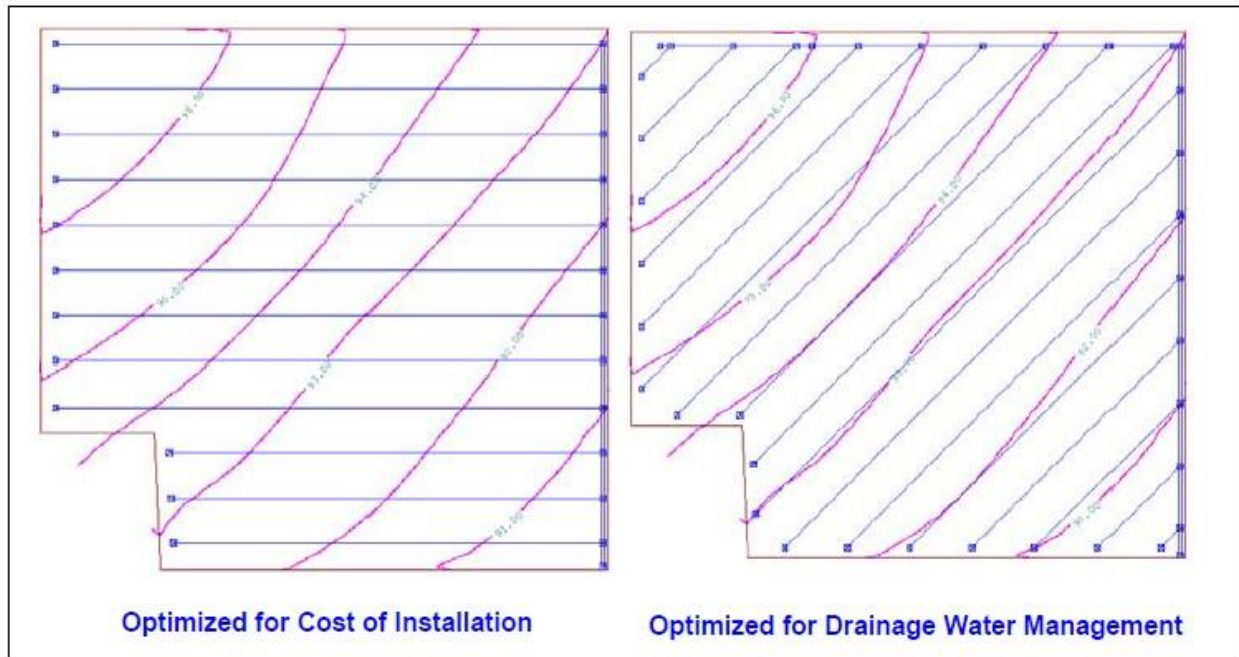


Figure 7 Drain Layout to optimize cost vs. controlled drainage. from Cooke et al (2008)

Meteorology

Meteorological variability strongly influences CD effectiveness. In dry years, there may not be sufficient precipitation to significantly raise the water table, limiting the reduction of drain water discharge compared to free draining fields. Similarly, unusually wet years may also limit the effectiveness of CD, resulting in sustained high-water tables and increased surface runoff regardless of drain settings at the control structure. The greatest N reductions may be observed in the year following an extreme drought year. Low plant uptake due to drought stress leaves significant residual soluble N available for mobilization the following year. Skaggs et al. (1995) found that careful drain water management in the year following a severe drought could reduce the annual N load by 70% compared to FD systems.

Expected Nutrient Reductions

Recent reviews of CD effects (Skaggs et al. 2010, Ross et al. 2016, Gramlich et al. 2018) continue to report variable though relatively consistent results, that broadly reinforce the field-based studies in Skaggs's (2012) synthesis. The tabular summary of the field studies considered by Skaggs (2012) is reproduced in the Table 8.

Table 8 - Field study results. Reproduced from Skaggs (2012), Table 1.

Reference	Location	Soil	Years observed	Area (ha)	Drain spacing (m)	Drain depth (m)	Control depth* (m)	Percent drainage	Reduction nitrogen loss
Gilliam et al. 1979	North Carolina	Portsmouth sandy loam	3	5 to 16	30 and 80	1.2	0.3 to 0.5	50	50
	North Carolina	Goldsboro sandy loam	3	3	30	1	0.3	85	85
Evans et al. 1989	North Carolina	Ballanhack sandy loam	2	4	18	1	0.6	56	56
	North Carolina	Wasda muck	2	4	100	1.2	0.6	51	56
	North Carolina	Wasda muck	2	4	18	1	0.6	17	18
Lalonde et al. 1996	Ontario	Bainesville silty loam	2	0.63	18.3	1	0.75	49	69
							0.5	80	82
Breve et al. 1997†	North Carolina	Portsmouth	1.2	1.8	22	1.2	0.4 to 0.5	16	20
Tan et al. 1998	Ontario	Brookston clay loam	2	2.2	9.3	0.65	0.3	20	19
Gaynor et al. 2002‡	Ontario	Brookston clay loam	2	0.1	7.5	0.6	0.3	16	
Drury et al. 2009§	Ontario	Brookston clay loam	4	0.1	7.5	0.6	0.3	29	31 to 44
Wesstrom and Messing 2007	Sweden	Loamy sand	4	0.2	10	1	0.2 to 0.4	80	80
Fausey 2005	Ohio	Hoyville silty clay	5	0.07	6	0.8	0.3	41	46
Jaynes 2012	Iowa	Kossuth/Ottosen	4	0.46	36	1.2	0.6	18	21
Helmers et al. 2012	Iowa	Taintor/Kalona	4	1.2 to 2.4	18	1.2	0.3	37	36
Adeuya et al. 2012	Indiana	Rensselaer	2	3	21	1	0.15 to 0.6	19	23
	Indiana	Rensselaer	2	6 to 9	43				18
Cooke and Verma 2012	Illinois	Drummer	2	15	30	1.15	0.15	44	51
		Drummer/Dana	1 to 2#	8.1	15	1.15	0.15	44	52
		Orion Haymond	1 to 2#	5.7	18 to 21	1.15	0.15	89	79
		Patton/Montgomery	1 to 2#	16.2	12	0.85	0.15	38	73

* Control typically removed during seedbed preparation, planting, and harvesting periods.
† Controlled drainage (CD) during the growing season only. CD reduced subsurface drainage volume by 16%; Nitrogen loss from subsurface drain + runoff by 20%.
‡ CD reduced subsurface drainage by 35%, increased surface runoff by 28%, and reduced total outflow by 16%. Nitrogen results were not reported and effects on pesticide loss were reported.
§ CD reduced subsurface drainage by 29%, increased surface runoff by 38%, and reduced total outflow by 11%.
|| CD reduced nitrogen loss by 44% for recommended nitrogen application rates and by 31% for elevated nitrogen rates.
Drainage volume measured for two years and nitrogen losses measured for one year for these locations.

Generalizing the observed findings without attempting to separate site-specific influences of soil, topography, climate, and drain system layout, led to some consistent basic common findings:

- CD commonly reduces annual drain water discharge and nitrate loads to receiving waters.
- Most commonly, the nutrient load reduction is comparable to the reduction in drain water discharge, suggesting a limited influence of denitrification.
- Drain water reduction from CD results from vertical seepage and, to a more limited extent, ET (<10%).

- Drain water reductions may also result from increased surface runoff and lateral seepage that can discharge to receiving waters beyond the drain outlet.
- The need for improved, more reliable, field-scale understanding of all of the flow paths responsible for observed drain water reductions was a common theme identified in these CD review and synthesis papers.
- In a recent review, Ross et al. (2016) observed that TP and DP loads were also reduced by CD, but the review recommended that future research focus on P reductions as there is a paucity of research on the topic.

Drain water reductions as high as 80% have been observed. However, the attribution of these reductions to vertical seepage (i.e. to deep groundwater) alone (implying a net reduction to the surface water system), should be viewed with caution. The presence of a drainage system signals the limited drainage provided by the soil profile. All else being equal, high rates of drain water reduction from these soils may be better explained by alternate flow paths that are more likely to return drain water to the surface water system, albeit, not through the drain outlet. Similarly, reduced nitrate concentrations and reduced nitrate:chloride ratios consistent with significant denitrification in the soil profile above the drain invert, have also been observed. Caution should be exercised in ‘projecting’ denitrification to significantly reduce N loads in any drain system adopting CD.

Reported annual drain water and nutrient load reductions vary significantly with soils, slope, climate, and drain system design. Over a wide range of conditions, Gramlich et al. (2018) reported the 30% N reduction credit currently associated with CD in the CBW, still represents a planning-level “consensus” estimate that is not considered unreasonable (Poole et al. 2018, Youssef et al. 2018, Evans et al. 1995).

Considering a nutrient reduction credit for CD in the CBW, the distinction between new systems designed for CD, versus the retrofit of older existing drain systems should be a significant consideration. The effectiveness of CD when retrofit to older drain systems that were laid out for free drainage or designed only for agronomic objectives has not been systematically compared, and does not appear to be well understood. Tile drain systems are most common on the Delmarva Peninsula. Less information is available about subsurface drainage in the Piedmont. Anecdotal information suggests Piedmont drainage systems in the Chesapeake Bay watershed may be more limited, and targeted to poorly draining problem areas – limiting the area affected (and therefore the load reduced) from CD.

Beyond mean reductions reported from literature meta-analysis, a nutrient reduction credit for controlled drainage systems in the CBW could consider regional and site specific criteria. For example, Skaggs has suggested that CD system design and performance may be at least partially normalized to standard system indices or parameters that embody both depth and spacing of drain lines as well as the transmissivity and depth to confining layer of the soil profile (Skaggs 2007). He has suggested future drain management studies consistently report the drainage coefficient (DC) characterizing the drain system, the drainage intensity (DI) characterizing a standard drain rate of the drain system in the soil column, and a steady state

subsurface drainage rate that Skaggs refers to as the Kerkham coefficient for the drained soil profile soil (Skaggs 2017). Comparing CD effects from widely differing soils and climate forcing in NC and Iowa (IA), Skaggs et al. (2005) reported similar trends for drainage and N losses with drainage density. Their results suggest how drain-system performance might be normalized across different site-specific conditions.

A related approach to generalize transferable performance expectations was introduced by Negm et al. (2016). They performed extensive multiyear simulations of CD and FD systems over a wide range of drain design and soil parameters for agricultural soils of eastern NC. The large database of consistent model-simulated results was used to develop multivariate regression equations relating site-specific characteristics and drain system parameters to the simulated performance of CD systems for drained agricultural soils of Eastern NC. A similar approach could be taken (and has been reported to be underway) for other regions as well.

In the future, a CD credit for the CBW could consider the drain layout and field topography to evaluate the likely field area actually affected by CD.

Information Gaps and Uncertainty

- Magnitude of other hydrologic fluxes: Most studies have not fully resolved the relative magnitudes of ET, vertical vs. lateral seepage, and surface runoff. Reported drain water reductions may therefore significantly overestimate the net reduction to receiving waters.
- Field Area Influenced by CD- Controlled Drainage is, perhaps, most frequently applied to uniform fields with slopes of 0.5%, with little topographic variation. The large body of studies from the Coastal Plain of NC were implicitly relatively homogeneous, making results relatively consistent and transferable to other drained coastal plain soils in NC. The area affected by CD when projecting a CD credit for very different landscapes and soils introduces significantly greater uncertainty. For example, considering CD on the different topography and soil structure of the Piedmont, raises a significant question about the area or percent of the drained field that would actually be affected by retrofitting a drain control structure to an existing drain system. Considering a credit across the Chesapeake Bay's agricultural watersheds, the field area actually affected by CD seems to be a very significant source of uncertainty. For example, the study by Lavaire et al. (2017) reported results from a 34 ha field with two drain systems. When CD was applied to only one field, they found the drain structure only affected 2 ha, and the elevated water table on the CD system was draining to the adjoining FD field and actually increased the FD drain discharge. When both drain systems were operated with CD the field area actually affected increased to 6 ha (~18% of total field area). Net reduction in drain water and nitrate discharge was only 10% with no evidence of denitrification during the 3-year study.

- *Denitrification potential*- The potential for denitrification to amplify nutrient load reductions from CD is an appealing possibility. The occurrence of denitrification zones along the shallow soil flow paths contributing to drain water is difficult to predict, and depends on climate as well. Youssef et al. (2018) reported a model-based north-south gradient in denitrification across the upper Midwest, attributed to temperature and rainfall. In contrast, most of the studies reported in Table 8 show little difference between nitrate reduction and drain water reduction, suggesting minimal denitrification. Although CD can create anoxic conditions in saturated soils, the seasonal operation of CD systems typically produces the largest volume of potentially anoxic organic soil water during the coldest part of the year, when soil bacterial processes are least active. For comparison, Raciti et al. (2011) found extraordinarily high rates of denitrification in the soil column of urban lawns during the very short (~2-week) window of optimal moist warm conditions in spring. The extent to which denitrifying conditions may be reliably produced at soil depths shallow enough to influence drain water, cannot be reliably anticipated from the presence of CD alone.
- *Transferability or scaling of observed CD effects*- The use of consistent scaling parameters that capture the interaction of the drain design and the soil profile is an intriguing path to help anticipate systems for which CD would be most beneficial. Detailed data sets – such as that developed by Negm et al. (2016), suggest a valuable feasible path to generalize site-specific attributes to CD performance. At this time, the utility of such an approach has not been demonstrated, but represents a promising path forward.

Recommended effectiveness estimates, default values

The panel recommends that WCSs that are used as a component of a DWM system designed and operated for the primary purpose of reducing nutrient loading from drainage systems into downstream receiving waters by restricting subsurface drainage from leaving the field will achieve a 30% TN load reduction for the acreage affected by the water control structure.

The vast majority of studies evaluating the water quality effect of CD have compared N loading (and drain water discharge) at the outlet of the tile drain system. The fewer studies that have closed the field water balance by also considering lateral seepage and increased overland flow have shown that observed reductions at the drain outlet may include increased N loading through other flow pathways. For this reason the panel concluded 30% TN reduction represented a realistic and credible reduction in the net nutrient load to receiving waters, notwithstanding higher nominal drain water reductions reported in the literature.

The panel similarly recognized that much of the literature on the water quality benefits of CD was performed on gently sloped (e.g. < 0.5%) tile drained agricultural fields in Coastal Plain regions. The net nutrient reduction from CD could be expected to vary when implemented in regions with steeper more varied topography and different soils, and in different physiographic settings. For this reason, the panel recommended the 30% N reduction should only apply to the

field area in which active management of the drain control structure can be expected to significantly influence or control the water table elevation throughout the year.

It is understood, based on experience, that the TN load reductions will fluctuate from year to year, but over the life of the practice the average annual TN load reduction is expected to be 30% or greater. The TN removal efficiency of 30% represents a conservative estimate of the annual N load reduction consistently observed in drain water *from the affected field area*. Whereas the practice can effectively reduce TN loads, applying the practice to locations where N loads per acre are greater will result in greater overall reductions in TN loading to the Chesapeake Bay. The panel recommends no credit for sediment reduction and no credit for P reduction pending further research.

Ancillary benefits and potential hazards or unintended consequences

The panel expresses its concern that many WCSs that were installed for the purpose of DWM are not properly managed. In fact, many land managers keep the structures open in the winter months in order to keep the field readily accessible and close the structures during the summer to preserve soil moisture. Verification of the proper management of water control structures may need to be a requirement for crediting water quality improvements accruing from this practice.

The panel also notes that Water Control Structure (Code 587) is an established conservation practice that can be installed for a wide variety of purposes and is not necessarily installed as part of a DWM system. The CBP established its current “water control structure” BMP in the Phase 4.3 Watershed Model for use in the jurisdictions’ tributary strategies (see Appendix C). While the three studies referenced at that time (Evans et al. 1989; Evans et al. 1996; Osmond et al. 2002) included WCSs, the structures were part of DWM system for CD, consistent with the DWM practice described and recommended in this section. The water quality benefits are associated with the management of the water levels and drainage system throughout the year, and not the presence of the WCS itself.

Management requirements and visual indicators of effectiveness

To receive credit for the Drainage Water Management practice, the land manager must operate the CD in accordance with a drainage water management plan designed to maintain a shallow water table that promotes denitrification during periods when field access by machinery is not required (usually winter months). Visual inspection of the position of the flash boards during these periods should confirm whether the drainage water management plan is being followed. Since the structural life of this practice is for as long as the WCS is in good repair and properly managed, inspection should also include visual signs of physical damage to the WCS that would prevent insertion and/or removal of the flash boards. For modeling purposes, the BMP should be considered annual with a credit duration of 1-year.

Future research needs

There is a paucity of research on the effectiveness of DWM for reducing P losses, and this needs further investigation.

The water quality benefits of retrofitting tile drain systems with CD will vary with the design and layout of the existing drain system. Negm et al. (2016) used extensive DRAINMOD simulation to estimate the effect of different intensities and capacity of drain systems in the NC coastal plain, and used these simulated results to develop regression equations for design. A similar series of model-based analysis performed for the soils and topography of the Piedmont and the Delmarva Peninsula, could provide more reliable, consistent, site-specific design and crediting information.

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DRAFT

Phosphorus removal systems

Terms and definitions

Phosphorus sorption material (PSM) – solid media that has an affinity for dissolved P. Used as a filter material in P removal structures and potentially, blind inlets. PSMs are often industrial by-products rich in iron, aluminum and/or calcium and magnesium.

Subsurface (Tile) Drainage - A conduit installed beneath the ground surface for collecting and/or conveying excess water.

Surface (Ditch) Drainage - A graded channel on the field surface for collecting and/or conveying excess water.

Particulate phosphorus (PP) – P that is bound to the surface of transported sediment

Dissolved phosphorus (DP)- P that is dissolved in solution

Total phosphorus (TP) – Sum of particulate and dissolved P

P removal design curve – mathematical relationship that describes discrete DP removal as a function of P loading to PSM

Specific practices/approaches/NRCS CP codes included and excluded under this practice/category

A P removal structure (NRCS Code 782) is essentially a landscape-scale filter for trapping DP in drainage water (See Figure 8). The structures can take on many styles and forms, but each possesses the following core components (Penn et al., 2018):

1. It contains a sufficient mass of an unconsolidated P sorption material (PSMs). PSMs are usually industrial by-products or manufactured materials—although some occur naturally—characterized by a capacity to strongly sorb P.
2. The PSM is contained and placed in a hydrologically active area that receives and/or exhibits dissolved P concentrations greater than 0.2 mg L⁻¹.
3. High DP water is able to flow *through* the contained PSM at a suitable flow rate.
4. The PSM can be removed and replaced after it is no longer effective at removing P at the minimum desired rate.

The P removal structure can be utilized for treating any dissolved P source: urban, agricultural, golf course, horticultural, and wastewater. In fact, most of the early work conducted on P removal structures was in the context of municipal, domestic, and agricultural wastewater; the structures were often used in conjunction with treatment wetlands (See Table 9). Different styles of P removal structures comply with these four characteristics, including surface runoff confined bed filters (Penn et al., 2012; Penn et al., 2014), PSM beds for wastewater (Shilton et al., 2006; Dobbie et al., 2009), subsurface beds for wetlands (Ballantine et al., 2010; Arias et al., 2003; Weber et al., 2007), subsurface tile drain filters (Penn et al., 2018), enveloped tile drains (Groenenberg et al., 2013; McDowell et al., 2008), drainage ditch filters (Penn et al., 2016; Klimeski et al., 2015; Kirkkala et al., 2012), modular perforated boxes (Penn et al., 2016), bio-

retention cells (Chavev et al., 2015; Liu and Davis, 2014), and blind inlets (Feyereisen et al., 2015).

Regardless of the source of DP—e.g., municipal, residential, agricultural, etc.—and the style of structure, the heart of the P removal structure is the PSM contained within it. Several studies have highlighted and reviewed many different PSMs (Lyngsie et al., 2014; Karczmarczyk and Bus, 2014; Eveborn et al., 2009; Johansson, 1999; Vohla et al., 2011; Hedstrom, 2006; Klimeski et al., 2012; Westholm, 2006). In general, PSMs can be reduced to two main categories based on P sorption mechanism: iron (Fe)/aluminum (Al) based PSMs that remove P by ligand exchange reactions, and calcium (Ca)/magnesium (Mg) based PSMs that work by precipitating Ca and Mg phosphate minerals (See Figure 9). Some PSMs are able to remove P by both mechanisms. A detailed discussion of different PSMs and their P sorption mechanisms can be found in Stoner et al. (2012) and Penn and Bowen (2018).

Because of the inherent variability of P removal structures, their efficacy can substantially vary. Different criteria have been used to characterize performance and estimate that efficacy, which frequently impedes the direct comparison among P removal structures. With so many influential factors and ways to report P retention, the evaluation of P removal structures is often interpreted in isolated scenarios, and their results are only valid under certain conditions.

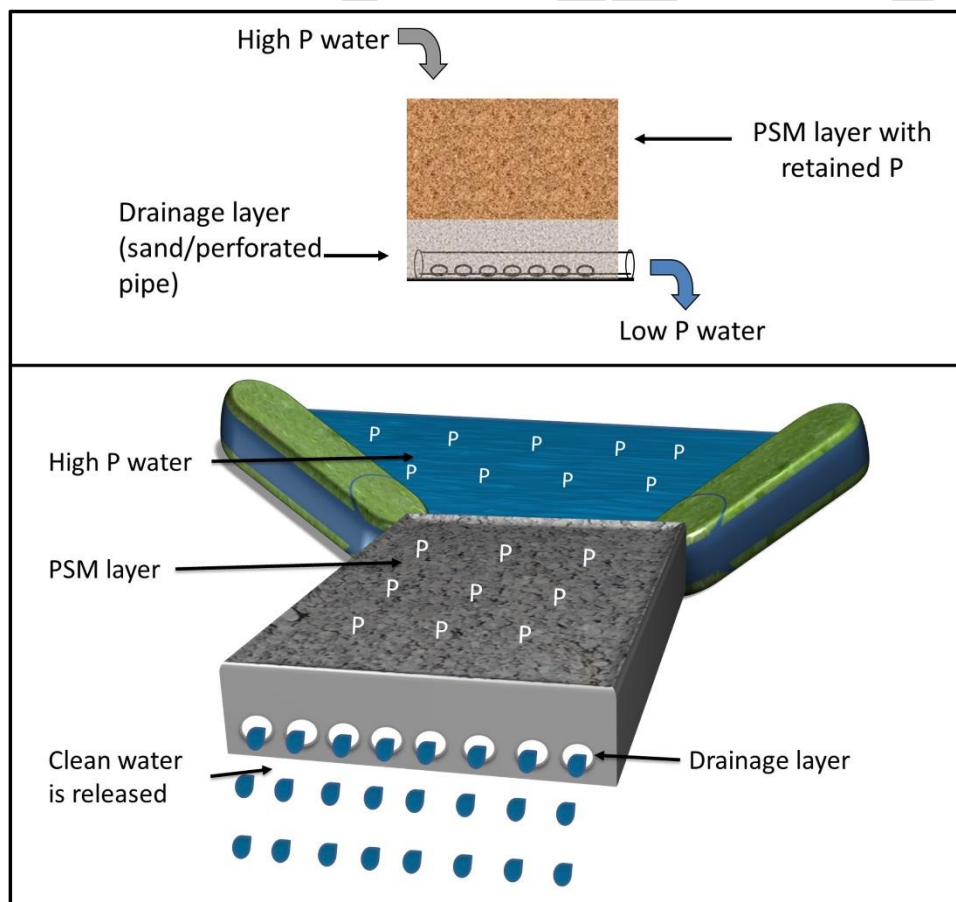


Figure 8. Diagram illustrating the basic concept of P removal through a P removal structure. From Penn and Bowen, 2017.

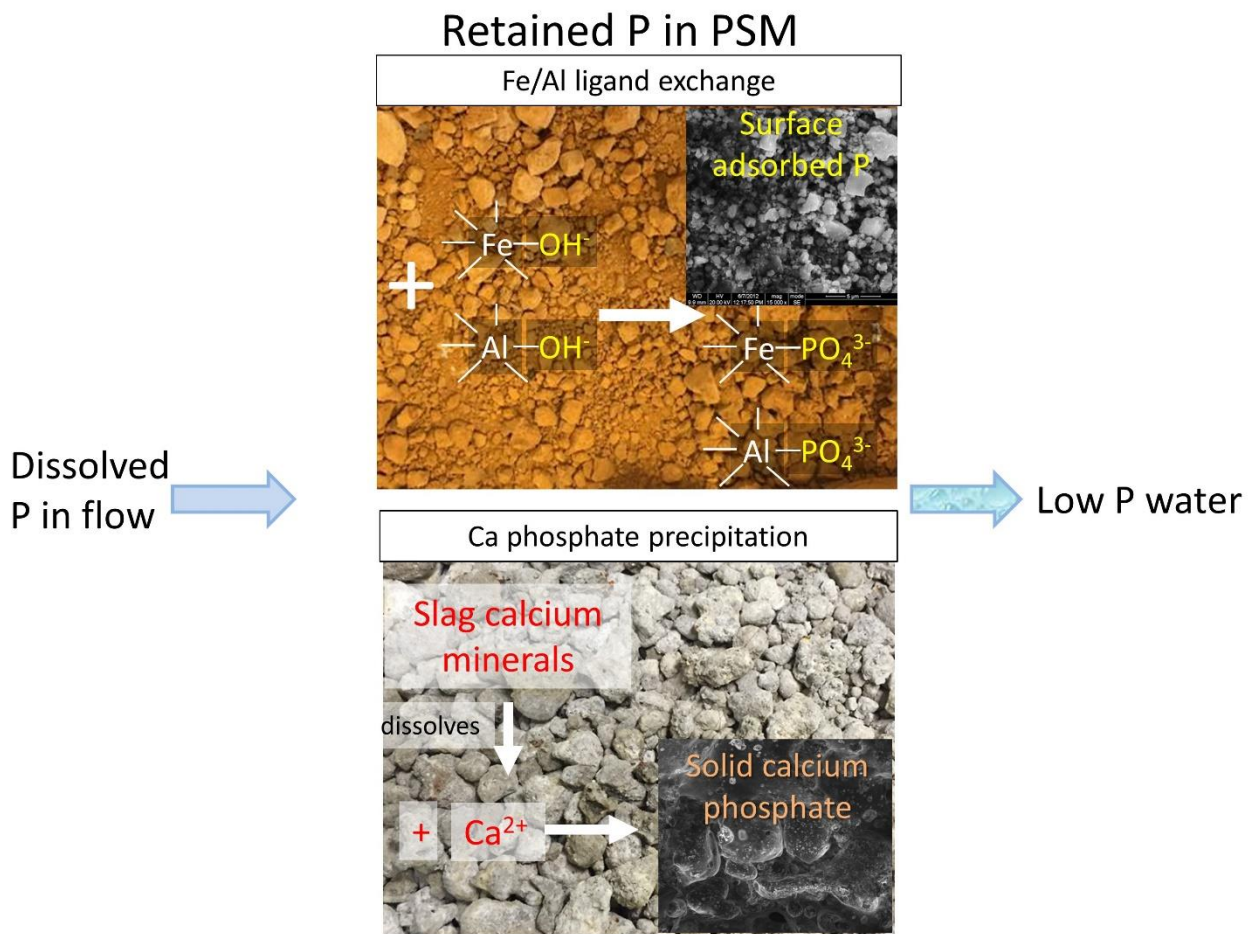


Figure 9. Illustration of P sorption materials (PSMs) utilized in P removal structures for trapping P. From Penn and Bowen, 2017.

Phosphorus Sorption Materials (PSMs):

Many PSMs are by-products from different industries, and therefore can be obtained for low cost. Some PSMs are manufactured. However, all PSMs must first be screened for safety before use in a P removal structure (See Figure 10).



Figure 10. Examples of several P sorption materials (PSMs)

Types of P Removal Structures

P removal structures can appear in many different forms. They can be located on the surface, subsurface, in ditches, tile drains, drainage swales, drop inlets, blind/surface inlets, etc. Any unit that possesses the four basic components listed above is essentially a P removal structure (See Figure 11).

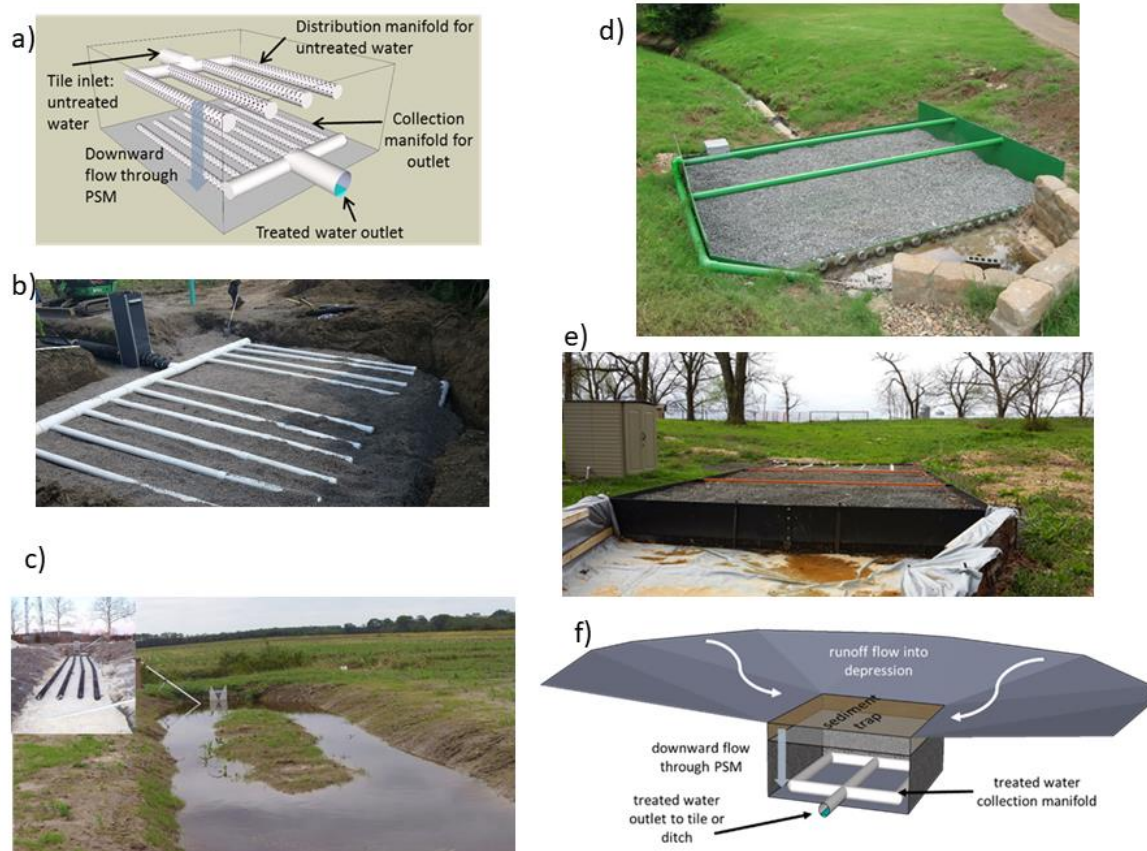


Figure 11. Example P removal structures: a) subsurface tile drain filter diagram, b) subsurface tile drain filter during construction, c) ditch filter, d) surface runoff filter, e) surface runoff filter, and f) blind inlet.

In order to qualify as a potential site for construction of a P removal structure, a site must possess:

1. Flow convergence to a point where water can be directed into a structure, or the ability to manipulate the landscape
2. At least 0.2 ppm DP in water
3. Hydraulic head required to “push” water through structure: function of elevation change or drainage ditch depth
4. Sufficient space to accommodate a PSM

Design of a P removal structure

Techniques for designing and evaluating a P removal structure are presented in detail in Penn and Bowen (2018). Design and evaluation of P removal structures should be conducted from the perspective of DP loads delivered to the water body, rather than edge-of-field P concentration only. Briefly, DP concentrations in water bodies are dynamic because of in-stream and in-water body processes. Hence, in regard to concentrations, P within the water body of interest constitutes the end of most importance, which is a function of P load delivered.

In essence, design inputs can be reduced to three categories (See Figure 12): (i) site hydrology and water quality characteristics; (ii) target P removal and lifetime; and (iii) PSM characteristics. The ability of PSMs to sorb P strongly dictates the size of the P removal structure (i.e., mass and volume of PSM). The core of the design process is sizing the structure as a function of the PSM's "P removal design curve", which considers site inputs and target P removal. The P removal curve is simply a mathematical description of P removal under flowing conditions for a given P inflow concentration and retention time (RT), expressed as a function of P loading (i.e., P added per unit mass of PSM). Similarly, the P removal curve can also be used to predict the performance of an existing P removal structure. Note that batch isotherms are not acceptable for designing P removal structures or predicting their performance. This is because batch isotherms: (i) utilize excessive P concentrations; (ii) do not allow for continuous addition of reactants and removal of reaction products; and (iii) normally have long retention times compared to field-scale P removal structures for non-point drainage (Penn and Bowen, 2018; Klimeski et al., 2012; Stoner et al., 2012; Penn and McGrath, 2011).

Several inputs and target goals are required for designing a site specific structure. The Phrog (phosphorus removal online guidance) software, available to the USDA-NRCS, can be used to quickly design a structure. Briefly, the required inputs include:

- Annual flow volume
- Dissolved P concentration
- Ditch size dimensions or tile drain diameter
- Maximum area or length willing to utilize for structure
- Diameter of pipe to be used in structure
- Chemical and physical characterization of PSM
- Desired retention time
- Desired cumulative P removal and lifetime
- Optional: TP concentration and sediment loading
- For ditches: maximum %loss of ditch flow capacity

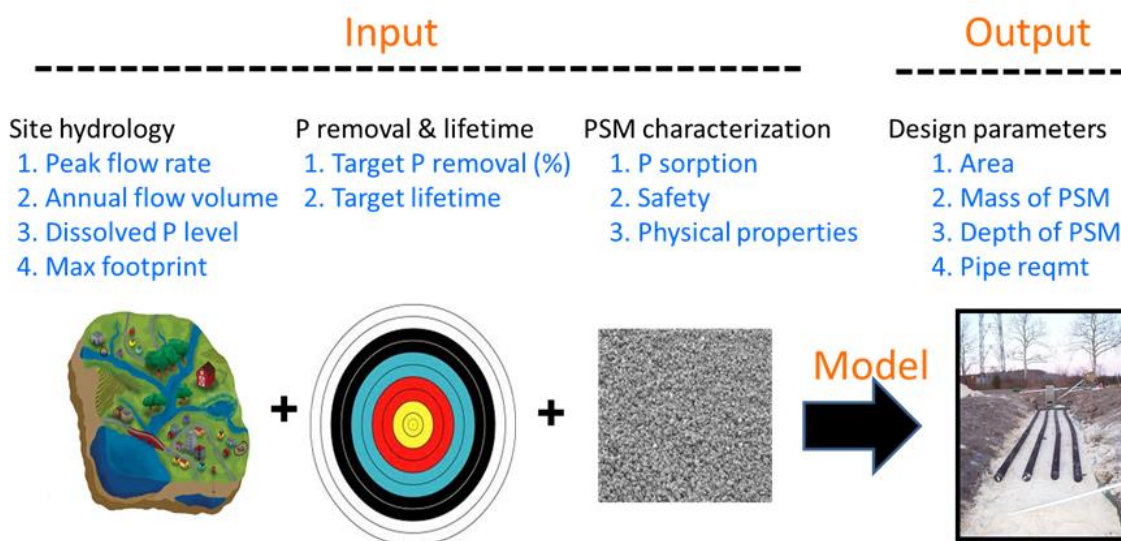


Figure 12. Simplified inputs for design of a P removal structure. Details are found in Penn and Bowen (2017).

In addition to its fundamental role to design P removal structures, the P removal curve is also a necessary tool for evaluating PSMs and P removal structures. Not only will the P removal curve vary with PSM, but it will also shift as a function of inflow dissolved P concentration and RT (Stoner et al., 2012; Lyngsie et al., 2015). Greater efficiency in P removal with increasing inflow P concentration is typical (See Figure 13). In fact, this variation in efficiency is a function of the thermodynamics of the reactions; higher P concentrations translate to higher concentrations of reactants and greater chemical potential for ligand exchange and precipitation reactions to occur. For this reason, P removal is generally more efficient on a mass basis for P sources such as wastewater than for non-point drainage water, which is much more dilute in P. Inflow DP concentration is therefore an important factor in comparing different PSMs and performance of P removal structures. However, its influence over P removal will vary among PSMs.

Retention time (RT) can also shift the P removal curve for a given PSM, and the magnitude of that shift will likewise vary among PSMs. For PSMs that are sensitive to RT, an increase in RT will generally increase P removal. However, Fe/Al-based PSMs are less sensitive to RT compared to Ca/Mg-based PSMs. This difference is due to the fact that Ca and Mg phosphate precipitation is usually slower than ligand exchange of P onto Fe and Al oxides/hydroxides (Stoner et al., 2012; Lyngsie et al., 2015; Klimeski et al., 2014). An important exception is Ca/Mg-based PSMs that: (i) possess a high pH (above 8); (ii) are well buffered with regard to pH; and (iii) produce readily soluble Ca or Mg. Such Ca/Mg PSMs are less sensitive to RT since they are able to precipitate P quickly. Stoner et al. (2012) showed that PSMs such as fly-ash were generally insensitive to RT compared to gypsum, which is an excellent soluble Ca source but poorly pH buffered.

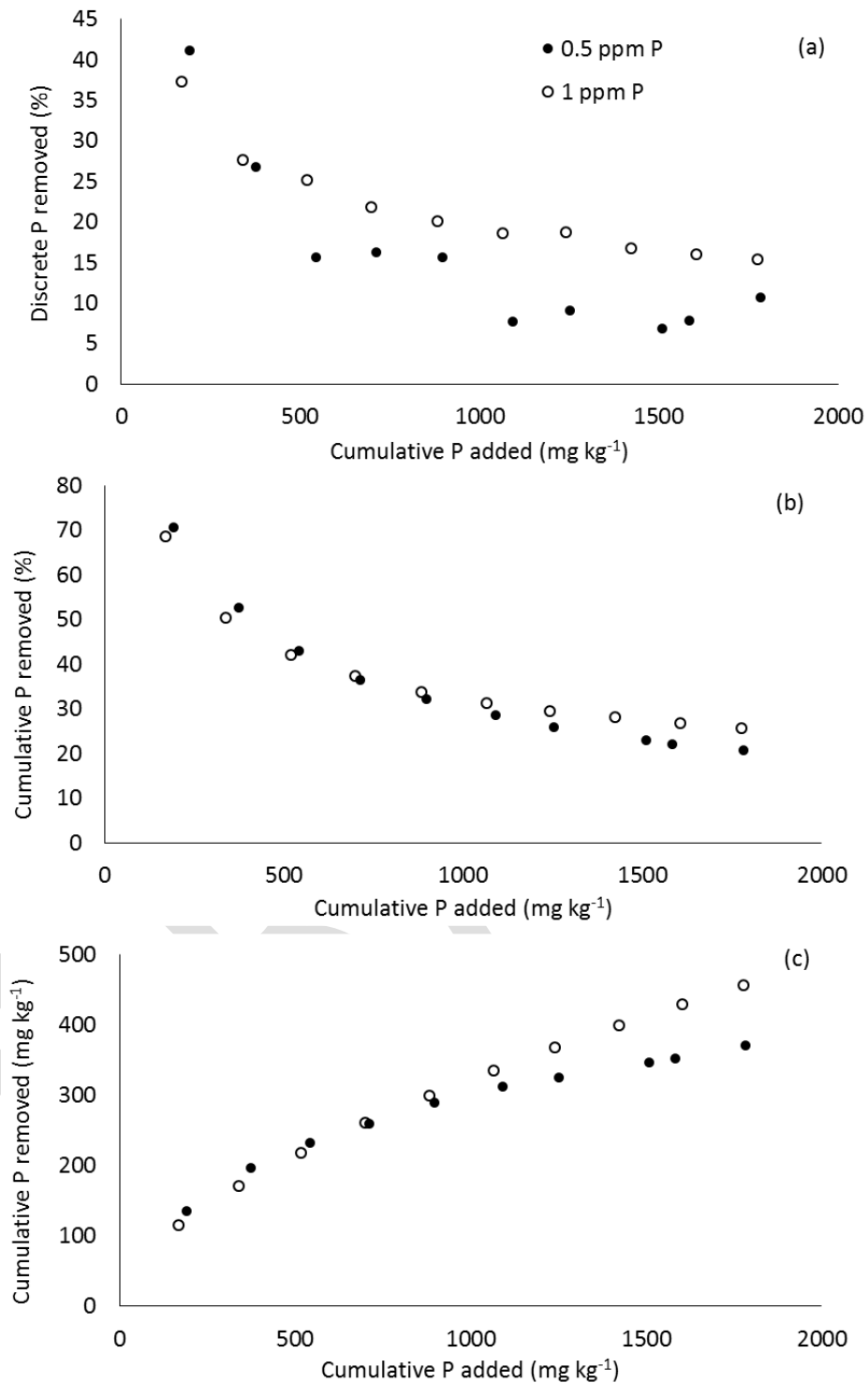


Figure 13. Examples of: (a) discrete P removal curve; (b) cumulative P removal curve expressed as a percentage; and (c) cumulative P removal curve expressed as mg P removed kg⁻¹ P sorption material (PSM). Flow-through experiment conducted at a retention time of 16 s with inflow dissolved P concentrations of 0.5 and 1 mg L⁻¹. The PSM in this example is a Fe-rich mine drainage residual (MDR).

Review of science and literature

Outside of Watershed, Peer Reviewed:

Penn et al. (2017) summarized P removal performance and conditions for several different P removal structures (See Table 9), organized primarily based on the inflow water type: wastewater vs. non-point drainage. This separation often reflects the degree of inflow dissolved P concentrations, which can have a dramatic impact on P removal. Another useful consequence of separating wastewater is that it also tends to reflect the RT, as the field and pilot scale structures for wastewater typically utilize longer retention times when compared to non-point drainage P removal structures. The wastewater treatment structures often had RTs in the range of hours to days, while non-point drainage water mostly operated with RTs of seconds to hours. The secondary variable for table organization was PSM type, which obviously has a major impact on P removal performance. Shale, sand, and soil were all placed in the same category due to having the least expected P sorption affinity and capacity. Calcium-based PSMs were placed together due to a consistency in P removal mechanism: Ca-phosphate precipitation. Similarly, Fe- and Al-based PSMs were grouped together. One exception in the group of Ca-based PSMs was steel slag. Steel slag is generally a Ca-based material (evidenced by high soluble Ca content, high pH, and pH buffer capacity), but due to the large body of work conducted on this material in P removal structures, it was given its own category. Most of the steel slag structures reviewed in this paper are electric arc furnace slag and blast furnace slag, and only a few studies utilized melter slag. The last variable for organization of Table 9, after PSM type, was simply chronology.

Mass of PSMs was indicated in Table 9 in order to convey the relative size of the P removal structures. In addition, particle size was included for two reasons. First, particle size indirectly provides information about the potential for that media to conduct water through it, which is necessary for any P removal structure. Second, particle size is inversely related to surface area, and therefore reactivity.

A brief examination of Table 9 will quickly reveal the diversity in cumulative percent P removal. However, as previously discussed, this value is useless without the associated input P loading. For example, in the artificial wetland constructed with Norlite (Hill et al., 2000), the cumulative percent P removal might appear superior to that of Pant et al. (2001) with 34% versus 12% removal, respectively. However, notice that the input P loading to the Norlite with 34% removal was only 150 mg kg⁻¹ compared to 786 mg kg⁻¹ for the shale gravel used in the Pant et al. (2001) study. Keep in mind that for a given material, the cumulative percent P removed will decrease with increased loading (Figure 13b). The PSM separation within the categories of wastewater and non-point drainage water serves to organize the data in a fashion that allows for a crude comparison between P removal structures and PSMs, while indirectly taking into account the inflow P concentration and RT.

Maintenance

Regarding maintenance, the most important considerations are potential clogging and the regular replacement of the PSMs after they have reached their useful lifetime. First, clogging can be avoided by use of more conventional BMPs that prevent erosion and sediment transport. For

example, buffer strips can filter runoff before entering a P removal structure. Materials that consist of small particle size are more likely to experience a reduction in hydraulic conductivity with excessive sediment loading compared to coarse-textured PSMs. Since proper design of a P removal structure takes into account lifetime of the PSM, the cost of PSM cleanout and replacement can also be determined. Simply put, structures designed and constructed for shorter lifetimes—i.e., reduced mass of PSM, smaller structure—will require more frequent replacement of PSMs, although the cost per cleanout will be potentially less compared to structures designed for longer lifetimes, given the same PSM and site.

How to assign a nutrient removal value to P removal structures

It is impossible to assign a blanket value of P removal efficiency to all P removal structures. Each structure will perform as a function of how it was designed and constructed for the site conditions, as illustrated in the reviewed literature of Table 9. That said, the overall average performance of P removal structures is around 40% DP removal, and most structures are specifically designed for a cumulative 40% DP removal. However, that provides no information on lifetime.

Most P removal structures currently being designed in the US are for achieving 40% cumulative *dissolved P* removal over the desired lifetime. Choice of this design target is somewhat meaningless, since the lifetime also dictates actual mass of P removed. For example, a recently designed structure located in Dekalb County, Indiana (IN), was designed for achieving a 30-year lifetime at 40% cumulative removal. This means that 40% all DP mass that enters the structure over 30 years will be trapped by the filter. However, if one considers only the first 10 years, the structure will remove around 60% of the cumulative input DP. Thus, while the choice of 40% cumulative removal is wise from the perspective of PSM efficiency, this value is meaningless without an associated lifetime, and the lifetime of the structure is one of the inputs considered during design. Although it depends on the effectiveness and cost of the PSM, most structures are designed for a 40% cumulative removal for a lifetime of 1 to 5 years. Using more potent PSMs, structures can be designed for a much longer time period. These structures are also mostly constructed to treat water from 10 to 30 acres, although larger structures can be constructed if feasible.

Keep in mind that P removal structures are intended to remove DP, and they are designed for that target. However, similar to blind inlets, the P removal structures will also act as a particle filter for sediment, as described using equations for single collector efficiency (see Penn and Bowen, 2017 Chapter 6). The Phrog software does take into account PP removal, and therefore TP removal when adding DP. Particulate P removal will vary dramatically as a function of the mean particle size of the PSM and the sediment deposition rate onto the P removal structure. For example, a sieved slag with mean particle size of 18 mm with 10 g sediment deposited per minute and 100 mg sediment/L will result in PP removal of 37%. Increasing the deposition rate to 100 g/minute increases the PP removal to 99%. For finer material such as gypsum, both sediment deposition scenarios results in 100% PP removal.

Table 9. Summary of design, conditions, and phosphorus (P) removal performance for various wastewater and non-point drainage P removal structures. Data are primarily organized based on inflow water type (wastewater vs. non-point drainage), secondary by type of P sorption material (PSM), and tertiary by chronology. DP: dissolved P. EAF: electric arc furnace. BOF: blast oxygen furnace. MDR: mine drainage residual. WTR: water treatment residual. CKD: cement kiln dust. From Penn et al., 2017.

Wastewater: Sand, Shale, or Soil									
Study	Notes	PSM	Mass	Particle Size	Retention Time	Influent DP Concentration	Cumulative P Added	Cumulative P Removed	Cumulative P Removed
			(Kg)	(mm)		(mg L ⁻¹)	(mg kg ⁻¹)	(mg kg ⁻¹)	(%)
Hill et al. [38]	Artificial wetland for dairy barnyard runoff	Soil (fine loamy, mixed, mesic Glossic Hapludalf)	53,625	NA	8 d	14.2	92	52.5	52.7
		Norlite, crushed and fired shale	33,000	NA	7.5 d	14.2	150	74	34
Pant et al. [39]	Constructed wetland with subsurface flow for wastewater	Queenston shale gravel	47,700	2–64	4.2–8.4 d	5.8–11.7	786	97	12
			47,700	2–64	4.2–8.4 d	3.3–6.7	720	92	13
		Fonthill sand	1560	0.0625–2	3–6 d	2.8–5.5	1042	307	29
			1560	0.0625–2	3–6 d	0.9–1.8	693	3	0.4
Forbes et al. [40]	Pilot scale wetlands for wastewater	Expanded shale	1400	0.72	17.3 h	0.36–2.25	648	449	69.3
		Masonry sand	3220	0.11	10.6 h	0.36–2.25	247	53.5	22.7
Kholoma et al. [41]	P filter for wastewater	Sand	245	0.33–25	130–180 m	6.4	23.7	4.5	19
		Gas concrete (Sorbulite) and charcoal	140 (Sorbulite) 97 (charcoal)	0.5–20	120–150 m	6.4	60	24	40
		Sand and charcoal	245 (Sorbulite) 97 (charcoal)	0.33–25	120–150 m	6.4	39	10	26
Wastewater: Ca-Rich Materials									
Study	Notes	PSM	Mass	Particle Size	Retention Time	Influent DP Concentration	Cumulative P Added	Cumulative P Removed	Cumulative P Removed
			(Kg)	(mm)		(mg L ⁻¹)	(mg kg ⁻¹)	(mg kg ⁻¹)	(%)

Szögi et al. [42]	Bed filter for swine effluent	Marl	1237	4.7–19	15.8 h	82	71	36	37–52
Gray et al. [43]	Artificial wetland (pilot scale) for treating wastewater	Marl	21	NA	5 d	7	48	47.6	99
Hill et al. [38]	Artificial wetland for dairy barnyard runoff	Crushed limestone	70,125	6–25	7.8 d	14.2	70	14.4	4.3
		Wollastonite and limestone	39,188	NA	7 d	14.2	126	40	9.5
Comeau et al. [44]	Pilot plant constructed wetland for trout farm effluent	Limestone: bed 1	148,500	2.5–5	31.2 h	0.03–0.61	4.61	3.99	87
		Limestone: bed 2	164,700	0–2.5	28.8 h	0.02–0.08	0.54	0.2	37.5
Pant et al. [39]	Constructed wetland with subsurface flow for wastewater	Lockport dolomite	45,300	0.0625–2	4–8 d	1.6–3.2	218	35	16
Arias et al. [17]	Constructed wetland for wastewater	Calcite	189	<2	28–99 m	7.3	13,904	3174	23
Vohla et al. [45]	Constructed subsurface wetland for wastewater	Calcareous sediment from oil-shale ash plateau	1400	0.002–0.125	48 h	6.94	11,743	656	5.6
Søvik and Klove [46]	Meso-scale filter for wastewater from single household	Shell sand (“Korall sand”)	666	>1 (pre-filter) <1 (main filter)	4.4–10.5 d	7.8	335	285	85
Ádám et al. [47]	Meso-scale wastewater treatment	Filtralite-P	359	0–4	4.3 d	6	526	521	99
	Large-scale wastewater treatment		99,000	0–4	17.7 d	2.9	54	52	97
Karcmarczyk and Renman [48]	Subsurface constructed wetland for wastewater	Sand, Ca addition, scrap iron, bentonite, bark, straw	NA	0.05–2	8.6 d	8	NA	373	24–96
Shilton et al. [49]	Column field test for wastewater	Tararua limestone	24	NA	12 h	10	1344	968	72
Wastewater: Steel Slag									

Study	Notes	PSM	Mass	Particle Size	Retention Time	Influent DP Concentration	Cumulative P Added	Cumulative P Removed	Cumulative P Removed
			(Kg)	(mm)		(mg L ⁻¹)	(mg kg ⁻¹)	(mg kg ⁻¹)	(%)
Shilton et al. [49]	Column field test for wastewater	Iron slag	24	NA	12 h	10	1168	210	18
Shilton et al. [14]	Confined bed for wastewater treatment	Steel slag	17,773,695	10–20	72 h	8.4 (total P)	3400	1200	35
Korkusuz, et al. [50]	Vertical subsurface flow wetland for wastewater	Blast furnace slag	9389	<3	2.9 d	4.6	493	248	50
Weber et al. [18]	P filter for wastewater connected to artificial wetland	Steel slag	113	5–14	12–24 h	29	2170	1700	75
	Stand-alone P filter for wastewater		113	5–14	12 h	29	1900	1200	72
Bird and Drizo [51]	Constructed wetlands for milk parlor effluent.	EAF steel slag; after two feeding cycles	829	5–20	18 h	42.5	2100	1464	70
Renman and Renman [52]	Wastewater treatment	Polonite (Ca silicate)	560	2–5.6	1–72 h	4.9	613	545	89
Barca et al. [53]	Subsurface flow filter to treat wastewater effluent from constructed wetland	EAF steel slag	10,800	20–40	17.5–23.8 h: then 48 h after 9 w	7.8	925	320	37
		BOF steel slag	9600	20–40	19 h–25.7 h: then 48 h after 9 w	7.8	1040	610	62
Wastewater: Mine Drainage Residuals (MDR) and Fe-Rich Materials									
Study	Notes	PSM	Mass	Particle Size	Retention Time	Influent DP Concentration	Cumulative P Added	Cumulative P Removed	Cumulative P Removed
			(Kg)	(mm)		(mg L ⁻¹)	(mg kg ⁻¹)	(mg kg ⁻¹)	(%)
Wood and McAtamney [54]	Pilot-scale constructed wetland for landfill leachate	Laterite	3000	2–3.5	8 d	1.46	2.45	2.28	93
Dobbie et al. [15]	Wastewater treatment plant	MDR (granular)	Initially 2100, then 1075 after	0.002–5	26 m (theoretical)	4	57,566	21,900	38

		substitution with gravel		12 m (measured)					
		MDR (granular)	505	6.4–9.5	16 m	3–5	28,374	5970	21
Sibrell and Kehler [55]	Pilot scale P filter for trout farm effluent	Toby creek MDR: 12 h resting period	11.2	0.85–6.3	1.95 m	0.0315	3303	1585	48
		Blue valley MDR: 12 resting period	11.2	0.85–4	1.93 m	0.03–0.26	3188	1689–1976	53–62
		GFH (manufactured Fe oxide): 12 h resting period	11.2	0.21–2	1.93 m	0.03–0.26	3188	1881–2040	59–64
		Blue valley MDR: regenerated after sorption cycle	11.2	0.43–2	1.93 m	0.12	3283	1871	57
		GFH (manufactured Fe oxide): regenerated after sorption cycle	12.2	0.21–2	1.93 m	0.12	3283	1684	52
<u>Non-Point Drainage: Non-Steel Slag Materials</u>									
Study	Notes	PSM	Mass	Particle Size	Retention Time	Influent DP Concentration	Cumulative P Added	Cumulative P Removed	Cumulative P Removed
			(Kg)	(mm)		(mg L ⁻¹)	(mg kg ⁻¹)	(mg kg ⁻¹)	(%)
Penn et al. [56]	Confined ditch filter for agricultural runoff	Mine drainage residuals	200	0.35 (mean)	0.7 m	6–16	2727	2700	99
Faucette et al. [57]	Runoff socks for treating synthetic runoff	Compost	7.2	0–25	0.87 s	0.86	33	3.15	9.5
		Compost and “natural sorbent”	7.2 compost, 0.165 “natural sorbent”	0–25	0.87 s	0.86	33	11.5	35
Bryant et al. [58]	Drainage ditch filter for agricultural runoff	Flue gas desulfurization gypsum	110,000	0.045	31 h	1.21	66	23	35
Kirkkala et al. [23]	Filters for treating agriculture runoff	Spent lime and burnt lime	2022	<3	20 h	2.6	4888	3031	62

		Burnt lime	58,500	<3	25 h	0.01	6	3.1	52
		Burnt lime	43,875	<3	85 h	0.003	7	3.22	46
		Mixed lime	43,875	<3	71 h	0.009	7	3.22	46
Groenenberg et al. [19]	Enveloped tile drain in agricultural field	Fe-coated sand	9240	NA	NA	1.4–3.1	42–93	38–89	90–95
Liu and Davis [24]	Bio-retention cell that collects runoff from parking lot	Soil +5% WTRs	7059	NA	NA	0.07	61	30.5	60
Klimeski et al. [22]	Ditch filter for agricultural runoff	Ca-Fe oxide granules (Sachtofer)	7000	3–15	10–3000 m	0.05–0.25	220	60	27
Penn et al. [21]	Ditch filter for agricultural runoff	Flue gas desulfurization gypsum	58,297	0.04	1–3 h	0.5	66	18	27
			46,054			1.6	19	7	37
			48,969			1.3	148	22	15
Non-Point Drainage: Steel Slag									
Study	Notes	PSM	Mass	Particle Size	Retention Time	Influent DP Concentration	Cumulative P Added	Cumulative P Removed	Cumulative P Removed
			(Kg)	(mm)		(mg L ⁻¹)	(mg kg ⁻¹)	(mg kg ⁻¹)	(%)
McDowell et al. [59]	Filter “socks” placed in a stream bed	Steel slag	1916	2–5	1.34 m	0.024	3311	1456	44
McDowell et al. [20]	Enveloped tile drain and filter socks in agricultural field	Melter slag (no socks)	72,000	NA	NA	0.33	60	36	60
		Melter slag, with 10 kg socks per drain	72,120	NA	NA	0.33	60	41	69
Agrawal et al. [60]	Filter cartridges for subsurface drains on golf course	Activated carbon, cement kiln dust (CKD) with 95% sand, steel slag, and zeolites	14.7 slag, 7.8 zeolite, 5 activated carbon, and 16.8 CKD/sand mixture	NA	Median 3.4 m (day 1) and 2.7 m (day 2)	0–1	69	–101	–150
Penn and McGrath [37]	Confined bed filter for treating pond water	EAF slag	454	6.3–14	10 m	0.38	172	59	34
		Treated EAF slag	454	6.3–14	10 m	0.34	149	54	36
		EAF slag	285	6.3–14	7 m	0.26–0.62	376	83	22

Penn and Bowen [11]	Confined bed filter for treating pond water	Treated EAF slag: first coating	285	6.3–14	7 m	0.16–0.62	233	82	35
		Treated EAF slag: second coating	285	6.3–14	7 m	0.18–0.41	285	80	28
Penn et al. [12]	Confined surface bed for golf course runoff	EAF slag	2721	6.3–14	9 m	0.3–1.6	103	26	25
Penn et al. [13]	Confined surface bed for golf course runoff	EAF slag	2721	0.5–14	10 m	0.5	160	53	33
Wang et al. [61]	Runoff interception trenches	EAF slag	6048	6.3–14	1 m	4.3	44	8	18
Penn et al. [13]	Confined surface bed for poultry farm runoff	Treated EAF slag	36,000	6.3–14	16.8 m	0.5–15	560	116	21
Penn et al. [21]	Modular boxes for treating pond water from poultry farm runoff	EAF slag	15,000	6.3–14	NA	1.04	37	10	27
	Ditch filter for agricultural runoff	EAF slag	79,495	6.3–14	20 m	0.6	43	11	26
			62,801			1.5	73	8	11
			66,776			0.9	107	26	24

Inside of Watershed, Peer Reviewed:

Specific to the CBW, Table 10 shows the result of four years of monitoring several ditch P removal structures and a storm water basin filter. Using the locally available FGD gypsum and slag, the P removal varied from 11 to 36% cumulative removal. In order to maintain a minimum 40% cumulative DP removal goal, these structures should have had the PSMs replaced after 2-3 years. Note however, that these structures could have easily removed cumulative 40% DP over four years if the structures had been constructed with a larger mass of PSM.

Designing structures to achieve a minimum peak flow rate is critical to performance. If untreated water is unable to pass through the PSM, then the water cannot be treated. Overflow or bypass water that simply flows along the surface of the PSMs will remove little to no DP. For this reason, it is recommended that all P removal structures be designed to treat as much of the flow as possible, for the largest events. This will vary for each site, and it may not be feasible to construct a structure that is able to handle an extremely large event. In essence, it is recommended that all structures be designed to handle as much of the highest flow rate as economically and feasibly possible. For example, while it may be feasible to design a structure to handle the flow rate for a 10-yr, 24-hr storm for

a particular site, it might only be feasible to construct a structure for another site that can handle the flow rate for a 2-yr, 24-hr storm. These are important decisions to be made by the designer. For ditch structures, Manning's equation can be used in combination with a chosen design depth, i.e. choose the depth of water in the ditch that the structure will be able to handle, and this corresponding flow rate becomes the target design minimum flow rate. For buried P removal structures that treat tile drains, the worst-case scenario (i.e. highest flow rate) can be determined simply based on the pipe diameter and slope; this value can then be used as the target minimum flow rate during a design.

For the Maryland ditch structures described in Table 10, their flow rate varied from around 200 to 500 gallons per minute (gpm). Again, this varies by design and conditions. A poultry farm stormwater runoff filter constructed in Eastern Oklahoma (steel slag) was able to handle nearly 1000 gpm. Achieving a minimum desired peak flow rate and a minimum retention time can be very difficult to achieve for some PSMs, conditions, and P removal goals. The Phrog software contains several algorithms that achieve this balance in design, since flow rate and retention time are inversely related to each other (details found in Penn and Bowen, 2017, Chapter 6). Some combinations of retention time and target minimum peak flow rate are impossible to attain.

Table 10. Summary of dissolved phosphorus (P) removal by six ditch filter structures and one storm water basin filter (Centreville; after four years. All structures were located in Maryland, USA, and contained either flue gas desulfurization (FGD) gypsum or electric arc furnace steel slag as the P sorption material (PSM). From Penn et al., 2016.

Site	PSM	Cumulative inflow P load	Flow- weighted inflow P concentrati on	PSM mass	Average P removal per event	Predicted P load removed	Measured P load removed
		kg	mg L ⁻¹	Mg	g	kg	kg
Barclay	FGD gypsum	3.8	0.48	58	75.9	0.77	1.06
Marion	FGD gypsum	0.86	1.58	46	16.4	0.62	0.31
Westover	FGD gypsum	7.2	1.3	49	132.4	1.09	1.06
Barclay	Slag	3.4	0.57	80	25.5	1.13	0.84
Marion	Slag	4.6	1.49	62	29.8	1.19	0.51

Westover	Slag	7.1	0.88	67	133.2	0.91	1.73
Centreville	Slag	0.53	1.04	15	12.9	0.32	0.14

DRAFT

Recommended effectiveness estimates, default values

The panel recommends that P removal structures in the CBW be designed with a goal to achieve a minimum reduction in DP of 40% over a minimum lifetime of 4 years. The design must also identify the expected life of the structure (period of time over which it is designed to remain effective) and the target minimum peak flow rate. However, the panel recognizes that site conditions may pose constraints that may not allow a structure to meet this goal while other structures may far exceed a 40% DP reduction. The structures can also remove sediment, and therefore PP, which contributes to additional TP removal. This sediment removal will vary dramatically with the PSM employed. For example, gypsum will retain much more sediment than a sieve steel slag. The Phrog software previously described will predict how much PP and sediment is removed for any given P removal structure. The panel recommends a credit for 60% sediment removal. However, the P removal structures are not typically constructed in locations where appreciable sediment is transported, since excessive sediment could reduce the lifetime of the structure, as well as the fact that there are other less expensive BMPs for treating PP and sediment. Although P removal structures can vary greatly with respect to their effectiveness in removing TP, the panel recommends that the cumulative TP load reduction for the drainage area on which it is installed be reduced by 50% over the design life of the structure. This is based on the assumption that most structures will be designed to remove 40% of DP, and the structure will be able to remove a conservative estimate of 10% PP as well. Individual P removal structures may be designed for variable target lifespans, based on the set of inputs used to calculate the design outputs (e.g., mass and depth of PSMs, area, and pipe requirements). However, for modeling, tracking and verification purposes the panel recommends a credit duration of four years for the P-removal structure BMP. The same credit (50% TP reduction) should be assigned for renovation of a structure to extend its effectiveness and should be based on the new design life of the renovated structure as that may change depending on what PSM is chosen to replace the original material. Applying the practice to locations where P loads per acre are greater will result in greater overall reductions in P loading to the Chesapeake Bay. The panel recommends no reduction in TN load from the drainage area resulting from installation of a P removal structure.

Drainage area is the ideal metric to track and report the P removal structure for simulation in the Watershed Model. However, if the drainage area is unknown, the panel recommends a default conversion rate that assumes 5 acres of drainage area per P removal system. This conversion rate is conservative, but reasonable, based on the available literature and the panel's experience.

Ancillary benefits and potential hazards or unintended consequences

Some PSMs contain heavy metals or other constituents that may be of concern if released into the environment. A total elemental analysis of potential PSMs should be conducted to identify potential hazards. However, presence of such constituents does not necessarily pose a hazard. In oxide form, a metal may be insoluble, but care should be taken to ensure that the P removal structure is not subjected to reducing conditions that may result in making the metal soluble. Water extraction tests can also provide information about the potential hazard for a PSM, when examined in the context of certain threshold values.

Management requirements and visual indicators of effectiveness

Regarding maintenance, the most important considerations are potential clogging and the regular replacement of the PSMs after they have reached their useful lifetime. Reduced outflow during a flow event is a visual sign of clogging and reduced effectiveness. Clogging can be avoided by use of more conventional BMPs that prevent erosion and sediment transport. For example, buffer strips can filter runoff before entering a P removal structure. Materials that consist of small particle size are more likely to experience a reduction in hydraulic conductivity with excessive sediment loading compared to coarse-textured PSMs. Since proper design of a P removal structure takes into account lifetime of the PSM, the cost of PSM cleanout and replacement can also be determined. Simply put, structures designed and constructed for shorter lifetimes—i.e., reduced mass of PSM, smaller structure—will require more frequent replacement of PSMs, although the cost per cleanout will be potentially less compared to structures designed for longer lifetimes, given the same PSM and site.

Future research needs

Whereas data on P removal structure performance is limited in the CBW, the panel recommends that flow and P concentrations continue to be monitored on P removal structures throughout the watershed in order to better document their overall effectiveness in this region. Data on sediment entrapment within the P removal structure and particulate bound P in that sediment is also needed.

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Saturated buffers

Terms and definitions

Riparian Buffer – A vegetated area (a "buffer strip") near a stream, consisting of perennial vegetation, which helps protect a stream from the impact of adjacent land uses. It plays a key role in increasing water quality in associated streams, rivers, and lakes, thus providing environmental benefits.

Subsurface (Tile) Drain - A conduit installed beneath the ground surface for collecting and/or conveying excess water.

Tile Outlet – The outlet pipe that conducts water in tile drains to a stream or ditch.

Water Control Structure (WCS)- A structure in a water management system that conveys water, maintains a desired water surface elevation, and controls the direction or rate of flow. For research purposes, it may also be designed to measure rate of water flow.

Specific practices/approaches/NRCS CP codes included and excluded under this practice/category

A saturated buffer is an edge-of-field practice that removes nitrate from tile drainage water before it enters ditches, stream, and other surface waters. When properly sited and installed (See Figure 14), a saturated buffer will remove nitrate whenever the tile is flowing and requires



Figure 14. Installation of a saturated buffer. Source: USDA ARS 2015. Photographer: Dan Jaynes.

limited annual maintenance to insure effective operation. Nitrate removal is primarily through denitrification (reduction of nitrate to N_2O or N_2 gas), although immobilization within the buffer vegetation may also remove nitrate. Factors affecting the performance of a saturated buffer are its length and width, buffer soil properties, and drainage area served by the saturated buffer. The basic components of a

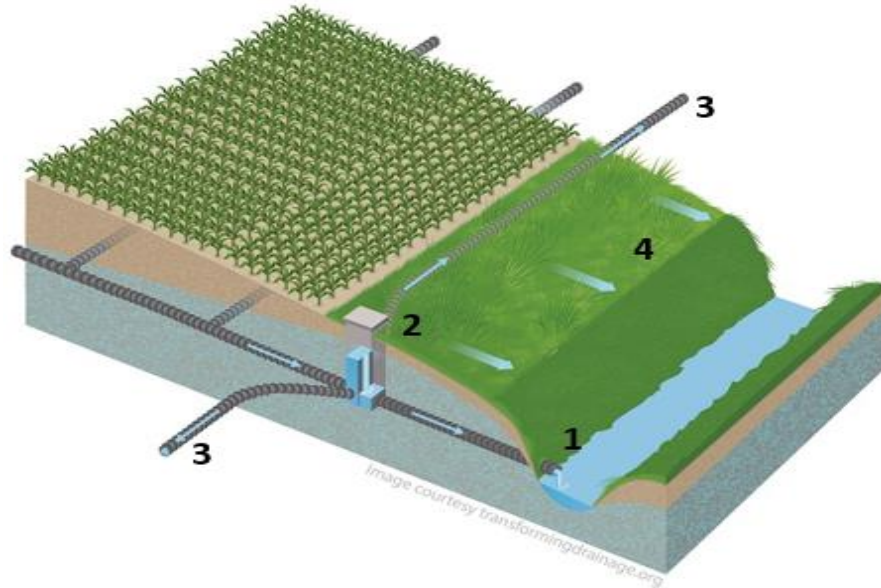


Figure 15. Basic components of a saturated buffer: 1. tile outlet, 2. water control structure, 3. perforated distribution tile or pipe, and 4. riparian buffer with established perennial vegetation. Adapted from <https://transformingdrainage.org/>

saturated buffer are the tile outlet, a WCS, a perforated distribution tile or pipe, and a riparian buffer with established perennial vegetation (See Figure 15). The WCS is installed within the buffer on the tile outlet. Perforated distribution pipes are connected to the WCS and installed within the riparian buffer roughly parallel to the stream at a shallow depth below the ground surface.

This practice will be applied in crop fields where subsurface drainage systems have been installed to remove excess water. When operating, a saturated buffer directs a portion of the subsurface tile drainage into the riparian buffer rather than discharging directly to surface water. The diverted water fills a distribution pipe and slowly seeps out into the soil following the natural gradient towards the stream. While moving to the stream, the nitrate contained within the water is removed by denitrification – a soil microbial process that converts nitrate to harmless N_2 gas – or is immobilized in microbial biomass or taken up by the vegetation within the buffer and incorporated into plant material. Denitrification is driven by existing soil carbon and new carbon from the turnover of roots and root exudates of the perennial vegetation.

The saturated buffer must be designed to meet NRCS Conservation Practice Standard 604, Saturated Buffers (NRCS CPS 604, May 2016R). The specified target is to treat either: (1) 5% or more of the drainage system capacity or (2) as much as practical based on the available length of the vegetated buffer. Specific state version of the practice may differ.

Review of science and literature

In Watershed, Peer Reviewed:

The panel is unaware of data from any saturated buffer within the CBW.

Outside of Watershed, Peer Reviewed:

Most studies of saturated buffers have taken place in the Midwest. A useful summary of results are provided in Table 11:

Table 11. Review of nitrate removal results for Saturated Buffers. First six sites are from Jaynes and Isenhardt, 2018. Last four sites are from Utt et al., 2015.

Location	Drainage Area, ha	Saturated Buffer Length, m	Saturated Buffer Width, m	% of Tile Flow Diverted to Saturated Buffer	Nitrate Removed, kg-N	% of Total NO ₃ Load Removed	NO ₃ Removal Rate, g-N m ⁻¹ d ⁻¹
Hamilton Co., IA	10.1/5.91*	305	21	42%	97	39%	1.5
Hamilton Co., IA	5	308	24	94%	52	84%	1.3
Tama Co., IA	7	115	4	51%	24	48%	2.0
Story Co., IA	22	124	14	26%	55	25%	1.7
Hamilton Co., IA	40	168	22	21%	118	8%	2.6
Boone Co., IA	3	266	19	49%	22	17%	0.4
Benton Co., IA	60	366	135	30%	408	29%	6.6
Edgar Co., IL	15	178	75	32%	68	29%	3.3
Rock Island CO., IL	60	219	120	26%	161	11%	3.0
Dodge Co., MN	20	280	80	22%	26	16%	4.2
avg.	25	233	51	39%	103	30%	2.7

*Additional tile installation in 2013 changed the area drained to the outlet. Adjoining fields were predominantly planted to corn and soybean. The Utt et al., 2015 data includes only sites that met the CPS 604 standard and had at least one year of data.

Jaynes and Isenhardt (2014) published the first report on the performance of a saturated buffer. The 335 m long saturated buffer was located in Iowa on a tile outlet draining 10.1 ha in a corn – soybean rotation. Average nitrate concentration in the water draining from outlet averaged 12.9 mg N L⁻¹. They found that 55% of the water from the tile outlet could be redirected into the buffer as shallow ground water over a two-year period. Additionally, all the nitrate contained in the diverted flow, a total of 228 kg-N, was removed within the buffer.

This study was followed by Jaynes and Isenhardt (2018) documenting the performance of the same saturated buffer for an additional 5 years as well as five other saturated buffers located in Iowa. The saturated buffers had been in place from 2 to 7 years. Two of the buffers were installed on riparian soils transitioning from row-cropped ground to perennial buffers planted with a pollinator mix, while the other three saturated buffers were installed on existing grass-covered buffers. Tile outlets drained areas ranging from 5 to 60 ha that were planted in a corn-soybean rotation and had average annual nitrate concentrations ranging from 4.2 to 24.8 mg N L⁻¹.

Another study by Utt et al. (2015) looked at 15 saturated buffers over two years. These saturated buffers were installed across Indiana (IN), Illinois (IL), Iowa (IA), and Minnesota (MN) and consisted of a range of riparian soils and landscape characteristics. Tile outlets intercepted by the saturated buffers drained areas from 3 to 60 ha used for row crop production. Some of the saturated buffers were deliberately located on sites that did not meet CPS 604 guidelines.

This project had difficulties at some of the locations measuring water flow from the fields and into the buffers and thus performance of these locations could not be determined.

Performance for the saturated buffers reported by Jaynes and Isenhardt (2019) and the sites meeting the CPS 604 standard and with flow data from Utt et al. (2015) are shown in Table 11. The saturated buffers varied in length from 115 to 366 m and in width from 4 to 135 m and thus some did not meet the minimum width of 9.1 m specified by CPS 604. Nevertheless, the saturated buffers diverted an average of 39% (21 - 92) of the tile flow to the riparian buffer where an average of 30% (8 - 84) of the nitrate was removed. These removal rates were the equivalent of 103 kg-N (22-408) removed each year – nitrate that would otherwise have drained directly into the adjacent streams. Nitrate removal rates averaged 2.7 g N m⁻¹ d⁻¹ (0.4 - 6.6).

Not Peer Reviewed, Outside Watershed

Utt et al. (2015) installed and monitored 15 saturated buffers in IA, IL, IN, and MN. This study found DP was removed in only one of the sites, and therefore concluded that there was little evidence of DP removal in the saturated buffer. While detailed information about the nitrate removal performance is provided below (Table 12), in summary the saturated buffers had mixed results in nitrate removal. The authors speculate that the failure of some of the sites can be attributed to a coarse soil layer preventing an elevated water table, inadequate carbon levels in the water table, improper design or installation and high water levels that prevented water from moving through the buffer. These are factors that should be considered when siting and designing a system.

Table 12. Source: Utt et al. (2015). Matrix showing results and suitability of each site for nitrate removal. A “+” means the site meets criteria, a “-” means it does not and a 0 means it is intermediate. Missing data are indicated by n.d.

Site	2014 lbs Nitrate removed	2014 %NO3 removed	2015 lbs Nitrate removed	2015 %NO3 removed	2014 %flow diverted	2015 %flow diverted	promising [NO3] trend	Need to adjust boards	soil carbon >2% @ 2.5 ft	High water table	Saturated buffer performance			comment
											performing	promising	not performing	
IA-1	94	64	107	77	64	91	+		+	+	+			
IA-2	0	0	0	0	0	0	+	-	-	+			+	no flow, coarse soil, low C
IA-3	n.d.	n.d.	408	29		30	+		+	+	+			
IL-1	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	-		+	+			+	no flow to buffer
IL-2	293	15	n.d.	n.d.	64	n.d.	+	-	+	+		+		as controlled drainage
IL-3	3	19	68	28	19	33	0	-	+	+	+			
IL-4	84	83	6.4	4	91	13	+/-		+	+		+		need better flow data
IL-5	13	28	161	11	91	26	+0		+	+	+			
IN-1	0	0	3.5	5	81	6	0	-	-	-			+	low C, coarse soil
IN-2	1.5	85	2.3	3	99	4	+0	-	-	-			+	low C
IN-3			0	0	n.d.	n.d.	-	-	+	+/-			+	flooding, coarse soil layer
MN-1	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	-	+	n.d.			+	flooding
MN-2	0	0	26	16	0	22	0/-		+	+			+	coarse soil layer
MN-3	5	32	0	0	40	0	+		+	-		+		coarse soil layer
MN-4	11	18	3.4	2	58	4	0/-	-	+	-			+	coarse soil layer

Recommended effectiveness estimates, default values

Based on the requirements of CPS 604 for load reduction and the results of the studies cited, the Agricultural Ditch Management Expert Panel recommends that proper installation and

maintenance of this practice will achieve a 20% TN load reduction for the drainage system on which it is installed. The panel notes that saturated buffers require less management throughout the year to function as intended when compared to DWM. Furthermore, saturated buffers are more typically installed for the purpose of water quality, and when the intent is combined with the design standards the panel felt confident that 10 years is a reasonable credit duration for CBP modeling purposes. It is understood, based on experience, that the load reductions will fluctuate from year to year, but over the recommended 10-year credit duration of the practice the average annual TN load reduction will be 20% or greater. When locating the practice where the buffer length does not equal 5% of tile flow as required in CPS 604, the nitrate load reduction will scale in direct proportion to the actual length divided by the 5% requirement length. Given the lack of sufficient data for P and sediment removal by this practice, no removal efficiency is recommended for those pollutants.

The saturated buffer BMP is simulated as a land use change in addition to the reduction from the upland drained area treated by the efficiency value. To simulate both aspects in the Watershed Model, it is assumed that 10 upland acres are treated per 1 acre of saturated buffer; this assumption was previously agreed to by the Agriculture Workgroup in coordination with the panel while establishing the interim BMP for saturated buffers. Therefore, the area of the saturated buffer is the preferred metric to track and report the BMP for simulation in the Watershed Model. If the area of the saturated buffer is unknown, then the linear distance or length of the buffer (feet) can be converted to area assuming a 30-ft buffer width. This assumed width is conservative based on the panel's experience as well as the practice design standards and specifications.

Ancillary benefits and potential hazards or unintended consequences

There may be a negative impact on crop performance if the water table upstream of the saturated buffer is elevated to a level that limits aeration in the rooting zone for a prolonged period or prevents timely field operations. However, if properly sited, the water table in a saturated buffer will not impact cropping activities in the producer's field.

The use of a WCS to direct flow in a saturated buffer does not meet the requirements for receiving additional credit for installing a WCS for DWM as described above in the "Water Control Structures and Drainage Water Management" section of this report.

A saturated buffer may infiltrate less overland flow than a traditional buffer.

Installation of this practice may enhance wildlife and pollinator habitats.

The WCS should be set to keep the water table as high as possible in the buffer without resulting in water on the soil surface.

Saturated buffers do not remove appreciable suspended solids and in fact, should be designed and placed where minimal suspended solids enter the buffer as they may plug the distribution pipes and shorten the effective life span. There is mixed evidence of P removal in saturated buffers. Buffer soil can either adsorb or release P as indicated by measured declining and increasing total DP within the riparian buffers (Utt et al., 2015). No credit for P removal should be taken at this time pending further research.

Management requirements and visual indicators of effectiveness

An operation and maintenance plan requires that water elevations (regulated by the water control structures) must be established and maintained during various seasons to achieve the desired performance. The maintenance plan includes inspection and maintenance requirements of the water control structure(s), distribution pipe(s), and contributing drainage system, especially upstream surface inlets. If the site is designed to be monitored, the plan will include monitoring and reporting requirements designed to demonstrate system performance and provide information to improve the design and management of this practice. At a minimum, water levels (elevations) at the control structure, observation ports, and if used, observation wells will be recorded biweekly when a water table is present and following precipitation events that result in high flows. Invasive trees or shrubs must be periodically removed to reduce distribution line plugging.

Future research needs

Whereas data on saturated buffer performance does not exist for the CBW, the panel recommends that flow and nitrate concentrations be monitored on saturated buffers installed within the watershed in order to better document their effectiveness in this region. Phosphorus removal potential needs further study because some saturated buffers have shown removal of measureable amounts of P for at least a few years, but neither the duration of removal nor the buffer soil characteristics that contribute to P removal are known.

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DRAFT

Future Research and Management Needs

Gypsum Curtains

Gypsum curtains, gypsum-filled trenches adjacent to agricultural drainage ditches, are designed to intercept shallow groundwater flow in very flat landscapes on the lower Eastern Shore of the Delmarva Peninsula. They were developed in response to a study that concluded that 90% of dissolved P that enters drainage ditches comes from shallow groundwater flow; only 10% comes from surface runoff (Kleinman et al., 2007). These permeable reactive barriers intercept ground water that transports DP to drainage ditches and tile drains, and remove P by precipitation with calcium, where it will remain trapped in particulate form until such time that all gypsum is dissolved. Gypsum curtains that were installed in 2009 on the University of Maryland Eastern Shore Research and Teaching Farm have been continuously monitored. In 2011, under the terms of a USDA-NRCS Conservation Innovation Grant, gypsum curtains were installed on all ditches at three farms and on selected tile drains on a fourth farm near Crisfield, MD (Allen and Bryant, 2015). Collected data indicates up to 90 % reduction in P concentrations in water that passes through the curtains, but load calculations are difficult since the rate of groundwater flow can only be estimated. Additionally, recent data suggest that there may be failures in some spots along the curtain, where concentrations are the same on both sides of the curtain. This suggests that animals, such as muskrats, may be burrowing through the curtain and providing a path for bypass flow. This affects the length of effectiveness of the practice and the need for maintenance, but it has not yet been investigated. Due to these uncertainties, the panel recommends further research be conducted on gypsum curtains before they are incorporated into the CBWM as a BMP.

References

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Two-stage ditches

The two-stage ditch represents an attempt to replace the straight trapezoidal ditch common in agricultural drainage with a ditch that is more consistent with natural stream processes. This form has a first stage which is associated with channel forming discharges, as well as a second stage which is the construction of a floodplain still within the ditch that is associated with the size to provide bank stability and a desired conveyance capacity (D'Ambrosio et al., 2015). While these systems are mainly installed with an eye toward reduced ditch maintenance and return to natural conditions, nutrient processing is a part of the potential benefits. NRCS Code 582 (Indiana NRCS FOTG) for Open Channel (Two-Stage Ditch) provides design criteria, but

there is no design criteria for this practice that is specific to the Delmarva Peninsula or elsewhere in the CBW

Davis et al. (2015) studied four agricultural streams in Indiana for 2-6 years using an upstream-downstream comparative sampling strategy. These areas were dominated by row crop production and tile drained and ditched agricultural land. This study found significant TSS and TP reductions in only one of the four streams. TSS in this stream went from 0.006 to 0.005 mg L⁻¹, and the TP from 0.095 to 0.083 mg L⁻¹. When pooled with the other sites, the reductions became insignificant. A significant reduction in SRP and nitrate was also found in only one stream (labeled SHA), however when pooled with the others the differences remained significant. In SHA the reduction in SRP was 0.022 to 0.016 mg L⁻¹ and pooled it reduced from 0.026 to 0.0245, for nitrate the SHA reduction was 0.482 to 0.472 mg L⁻¹ and pooled it was a 2.618 to 2.610 mg-L reduction. The conclusions reached were that a lower bench may have contributed to more sediment deposition on SHA, also that nitrate levels above 1 mg L⁻¹ may be too high for denitrification in this system.

Mahl et al. (2015) used a similar approach from 2009-2010 to study six agricultural streams (1 natural, 5 constructed) at various points in their development (0-10 years). They reached similar conclusions, baseflow conditions reduced SRP 3-53% with no significant reductions in nitrate due to high concentrations. In addition, a reduction in TSS was implied by a 15-82% decrease in turbidity. In addition, they concluded that the site with the lowest bench and increased inundation of the floodplain did decrease surface nitrate by 4%.

The SHA site mentioned above was also studied in Roley et al. (2012) when the two stage restoration was installed using a similar sampling strategy. They found in stream denitrification rates to be similar before and after restoration (3.2 to 20.3 mg N₂O-N·m⁻²·h⁻¹), and lower on the constructed floodplains (0.02 to 6.7 mg N₂O-N·m⁻²·h⁻¹). Using storm flow simulations, they concluded that while the two-stage ditch contributed significantly to nitrate removal during storm events, <10% of nitrate was removed in all storm flow events. This may have been due to high nitrate loads in at the site, the highest percentage of removal came at the lowest loads.

Powell and Bouchard (2010) studied 10 one-stage ditches and 10 two-stage ditches in northwestern Ohio. Using the static core method in the laboratory, they tested the potential denitrification rates of the different ditch types. Their conclusions were that the two different ditch types had similar denitrification rates in the channels themselves, but that the denitrification rates for the benches of the two-stage ditches were higher than the slopes of the one-stage ditches. They conclude that the reason for this is the higher organic matter in the bench sediment, due to increased plant biomass or changes in hydrological processes. Due to the difficulties in measuring in situ denitrification rates, they were unable to quantify this difference.

The panel recognizes the need for design criteria for two-stage ditches specific to soil and landscape conditions on the Delmarva Peninsula and the need for more research on their effectiveness for reducing sediment and nutrient losses.

References

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Denitrifying Curtains

Several categories of enhanced-denitrification practices fall under “*bioreactor*” terminology (Schipper et al. 2010), the most common of which is the bioreactor “*bed*” as discussed in this report. However, in the Chesapeake Bay region, another category of bioreactor, sawdust-amended denitrifying walls, has the potential to greatly reduce N loadings in shallow groundwater. These flow-intercepting walls are installed perpendicular to the direction of groundwater flow, parallel to drainage channels or streams, with lengths ranging from 35 to 55 m (115-180 ft) and widths and depths ranging from 1.5 to 3.0 m (5-10 ft). Denitrifying walls are filled with mixtures of native soil and sawdust that range from 20 to 50% sawdust by volume. The only peer-reviewed study of a sawdust denitrifying wall within the Chesapeake Bay region reported this type of bioreactor was simple to design and construct, inexpensive, and resulted in >90% NO₃-N concentration reductions at the one monitored site (Christianson et al., 2017). However, further research on this potentially very important practice for the region is needed because, as Christianson et al. (2017) noted: “*Documenting NO₃-N removal effectiveness for the sawdust wall was difficult due to the challenge of measuring lateral groundwater flow rates. To give credit for NO₃-N reduction with this practice, a practical solution would be to develop regional estimates of groundwater flow rates and accept those as applicable to certain soil and landscape conditions.*”

References

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Schipper, L.A., W.D. Robertson, A.J. Gold, D.B. Jaynes, and S.C. Cameron. 2010. Denitrifying bioreactors--An approach for reducing nitrate loads to receiving waters. *Ecological Engineering* 36:1532-1543.

Ditch Dipouts (Dredging)

Ditch dipouts (also called dredging or clean-out in scientific literature) refer to the process of removing sediment and vegetation from the bottom of an agricultural drainage ditch. As currently practiced, dipouts occur when conveyance of drainage water is slowed to the point that drainage of adjacent and upstream fields is inadequate. Following a dipout, the subsoil is exposed at the bottom of the ditch, and on the Eastern Shore, the exposed sediments are usually coarse textured and extremely low in organic matter. Subsequently, ditch walls collapse, topsoil tends to fall into the ditch, and eroded sediments from topsoils begin to accumulate in the ditch bottom. Topsoil sediments are typically much finer textured and contain appreciable clay, silt, organic matter and sorbed P. These processes continue until the need for another dipout. During dipout, these finer textured sediments are placed in the field and are spread across the surface to return the shape to its original contour. The argument for the benefit to water quality is that these P-rich sediments are removed from the drainage pathway, and therefore the possibility of P desorbing and being transported to sensitive waterbodies is also removed.

Early research by Sallade and Sims (1997) in Delaware, recognized that ditch sediments can be both a source and a sink for P depending on the P concentration in runoff. They also recognized seasonal changes in typical P concentrations in runoff and differences in sediment characteristics that determine relative P sorption capacity. They concluded that ditch dipouts could be prioritized based on these factors. Ditch sediments prior to dipout and after dipout from a ditch in Maryland were used in a fluvium study to show that the practice can negatively impact the P buffering capacity of ditches draining agricultural soils with a high potential for P runoff (Shigaki et al. 2008). Additionally, sediments exposed after dipout had a lower capacity to remove ammonium, although nitrate was relatively unaffected by sediment type (Shigaki, 2009). Studies in Illinois reached the same conclusion; after dipout, sediments and their associated microbial populations are no longer able to buffer nutrient concentrations in ditch flow as they were prior to dipout (Smith et al., 2006; Smith and Pappas, 2007). Disruption of ditch bottom sediment and vegetation by dipout may also make the ditch system susceptible to greater sediment losses (Needelman et al., 2007). Sharpley et al. (2007) concludes, and the panel agrees, *“more studies are needed to assess whether dipouts can be used as a BMP.”*

References:

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[**Editor's Note:** The section discussing DNBRs for freshwater springs or seeps was revised and moved into Appendix E, at the end of this document.]

Appendix A: Technical Appendix

[Provided as a separate document to simplify comment/review process with CBP and WTWG]

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Appendix B: Conformity of report with BMP Protocol

[Editor's Note: Will be posted as a separate document when available for CBP review]

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Appendix C: Phase 4.3 documentation for Water Control Structure BMP

Editor's Note: This appendix is documentation from when the Water Control Structure BMP was established (circa 2005-2007) for use in the jurisdictions' tributary strategies in the Phase 4.3 Watershed Model. The text was provided by Jeff Sweeney (EPA, CBPO) and is provided here with minimal formatting updates.

Drainage Water Control Structure Best Management Practice Definition, and Nutrient and Sediment Reduction Efficiencies For use in Phase 4.3 of the Chesapeake Bay Program Watershed Model

Recommendations for Formal Approval by the Nutrient Subcommittee's Tributary Strategy and Agricultural Nutrient Reduction Workgroups

This document summarizes the recommended definition and nutrient and sediment reduction efficiencies for the Drainage Water Control Structure Best Management Practice. The Nutrient Subcommittee approved this practice for inclusion in Phase 4.3 of the Chesapeake Bay Program Watershed Model pending consensus on the definition and efficiencies from the Subcommittee's Agricultural Nutrient Reduction Workgroup.

Attached to these recommendations is a full accounting of the Chesapeake Bay Program's discussions on this practice and how these recommendations were developed, including data, literature, data analysis results, and discussions of how various issues were addressed.

Recommended Water Control Structure Best Management Practice Definition

The Drainage Water Control Structure Best Management Practice (BMP) consists of installing and managing boarded gate systems in agricultural land that contains surface drainage ditches. These ditch systems are often necessary in coastal plain regions in order to create agricultural land suitable for cultivation on the very flat topography. The load reduction occurs as the result of two processes: (1) volume flow and (2) nutrient concentration. By design nature, these drainage water control structures reduce the total volume of water flow. Also, the inorganic nitrogen concentrations in the drainage waters are reduced through denitrification and/or recycled for plant growth. As runoff occurs beyond the agronomic growing season, nitrogen continues to be reduced by denitrification. For application of this practice to the Chesapeake Bay region's coastal soils, a nitrogen reduction efficiency of 33% is provided for each managed and drained acre.

Proper installation and management of the boarded gate structures is critical to achieve the stated nitrogen reductions. Installation can be according to NRCS code number 537 *[sic]* and must include an operation and maintenance plan using the following methods: (1) maintain flashboard settings to retain storm runoff water levels within 30 inches of the ground surface along at least 50% of the upstream ditch reach all year; and (2) maintain flashboard settings to retain storm runoff water levels within 12-18 inches of the ground surface in winter if no small grain crop is present.

Recommended Level of Conservatism

Developing a realistic pollution reduction efficiency for a BMP is difficult since research-based BMP reductions are seldom achieved in actual practice and the full range of potential climatologic variation in a field study or model is difficult to capture.

Some of the factors that caused us to take this more conservative approach are listed below. For a full description of analyses see Appendix A.

- Studies in North Carolina found nitrogen loads from agricultural fields with properly managed drainage water control structures to be 45% less than from fields without this practice, on average (Evans et al., 1989; Evans et al., 1996; and Osmond et al., 2002). The adjustment to 33% is based on these differences between North Carolina and the Chesapeake Bay region's coastal plain: lower precipitation levels; a cooler climate causing less denitrification; and tighter nutrient management. Phosphorus reductions from water control structures have been noted in the research; however many Chesapeake Bay region soils with drainage systems have higher than average soil phosphorus levels, so no reduction is currently assigned.

References

Evans, R.O., J.W. Gilliam, R.W. Skaggs. 1989. *Effects of Agriculture Water Table Management on Drainage Water Quality*. The Water Resources Research Institute, Report No. 237.

Evans, R.O., J.W. Gilliam, R.W. Skaggs. 1996. *Controlled Drainage Management Guidelines for Improving Drainage Water Quality*. North Carolina Cooperative Extension Service, Publication Number: AG 443.

Osmond, D.L., J.W. Gilliam, and R.O. Evans. 2002. *Riparian Buffers and Controlled Drainage to Reduce Agriculture and Nonpoint Source Pollution*. North Carolina Agriculture Research Service Technical Bulletin 318, North Carolina State University, Raleigh, NC

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(Appendix A): Water Control Structure Best Management Practice
Supporting Technical Information and Historic Record for Developing
BMP Definition and Nutrient and Sediment Reduction Efficiencies

This appendix represents the comprehensive documentation pertaining to all Nutrient Subcommittee and workgroup discussions of this BMP, all literature evaluated, all data analysis conducted, and all technical reviews conducted in chronological order. This Appendix serves as the historic record of discussions leading up to the final recommended BMP definition and pollution reduction efficiencies.

Description of Analysis to Determine Level of Conservatism for CNT Efficiencies

The Chesapeake Bay Program's Agricultural Nutrient Reduction Workgroup (AgNRWG) discussed the BMP at meetings from December 2004 to March 2006. The Chesapeake Bay Program's Tributary Strategy Workgroup (TSWG) approved the AgNRWG's BMP efficiency recommendations on November 7, 2005, with the understanding that the AgNRWG would finalize the language of the BMP at their upcoming meeting. The Chesapeake Bay Program's Nutrient Subcommittee (NSC) gave final approval to the BMP on December 7, 2005, with the caveat that the AgNRWG will be entrusted to finalize the language defining the BMP.

Excerpts from Meeting Minutes – All Relevant Discussion of the Drainage Water Control Structure BMP:

AgNRWG December 14, 2004

- DE requests a 33 percent reduction in N from Water Control Structure (WCS) BMPs. No credit is yet given by the WSM to these structures. The 33 percent is an annual average.
- WCSs act as drainage ditches during planting and harvesting, removing excess water from fields, and as irrigation reservoirs when they are dammed up during the growing season. It is during the growing season that there is an associated denitrification occurring in the WCS in the slow moving water. DE will assign the 33 percent reduction of N to WCS in their Pollution Control Strategies.
- See handout, Nitrogen Reduction due to Water Control Structures as BMPs. Jeff is to get the three reference documents listed on the handout and email them to the BMP Task Force.
- Jeff stated that the WSM already credits DE by attributing good drainage all the time. The WSM does not account for ditches, which have negative water quality impacts during planting/fertilizing season, etc.

AgNRWG June 9, 2005

- At the December 2004 AgNRWG meeting, DE asked the workgroup to consider accounting for N reductions obtained through water (drainage) control structures ubiquitous on

eastern shore. The minutes detailing the request are available at http://www.chesapeakebay.net/pubs/calendar/ANRWG_12-14-04_Minutes_1_5165.pdf.

- Background
 - Water control structures act as drainage ditches during planting and harvesting by removing excess water from fields, and as irrigation when they are gated during the growing season. During the growing season, denitrification occurs in the saturated conditions. DE has assigned the 33% reduction of N to water control structures in their Pollution Control Strategies.
 - The reduction applies to the area being treated by the water drainage control structure. In DE there is a calculated loss of 21 lbs N/acre.
 - DE is not requesting a reduction in P or sediment.
- At the December meeting, DE supplied the workgroup with the handout, Nitrogen Reduction due to Water Control Structures as BMPs. Before making a decision, the workgroup requested that DE supply the three reference documents listed on the handout. DE complied, and the reference documents were mailed out to the BMP Taskforce in January 2005. They are:
 - Controlled Drainage Management Guidelines for Improving Drainage Water Quality
 - Riparian Buffers and Controlled Drainage to Reduce Agricultural Nonpoint Source Pollution
 - Effects of Agricultural Water Table Management on Drainage Water Quality
- MD also has had a drainage program for past four years. John Rhoderick of MDA stated that they have preliminary monitoring results from past four years showing substantial reductions.
 - Conservation districts cost share the practice in MD.
 - MD farmers are willing to manage the structures in winter for wildlife.
- The NC “Riparian” study indicates on page 27 that, in order to achieve the measured 44% N reductions, proper maintenance and management of the structures is labor intensive and exact.
 - This is why DE has requested a more conservative reduction of 33%.
- Summary
 - 33% is an annual rate, but the reductions occur mainly in summer
 - The reduction is applied to (cropland) acres drained
 - 33% reduction is relative to the load
 - Adjust for maintenance, management and leaching from max of 44%
 - Inland Bays have 51 structures at 31 acres apiece.
- DE will summarize in a couple paragraphs the practice and the decisions of the workgroup.

TSWG November 7, 2005

- Bill Rohrer, Delaware Nutrient Management Commission, on behalf of the Agriculture Nutrient Reduction Workgroup will present the recommended efficiencies for water control structures for the TSWG to review.
- Highlights from presentation:
- Cropland loading rates x acres treated by BMP x efficiency (%) = nitrogen reduction to stream (lb/yr)
- Structure turns drainage on or off resulting in denitrification, no phosphorous reduction.
- Questions and Comments:
- Reduction based on literature and DE scientists' knowledge.
- AgNRWG based on research recommended by bumping down NC numbers and comfortable with 33%.
- Model does not model ditches.
- 33% nitrogen load reduction from agriculture fields with water control structures than fields without this practice.
- Literature does not encourage denitrification.
- DE common rotation – corn, soy, wheat full season at 1/5 rotations/acre/year.
- At the December 2005 AgNRWG meeting the workgroup will work on the language for the BMP so they can support water control structures.
- Stipulations – final definition developed with AgNRWG. Have not addressed groundwater by-pass. Once landuse in Phase V must be addressed.

NSC December 7, 2005

- The TSWG recently approved the Water Control Structure BMP proposal with Nitrogen reduction efficiencies of 33% (no reductions yet applied to P and sediment), with the caveat that there's no drained landuse in current version of the watershed model (4.3). Tom asked for and received approval from the NSC on the Water Control Structure BMP as defined, with same caveats applied to the CNT proposal – the workgroups will be entrusted to finalize the BMP.

AgNRWG January 13, 2006

- See handouts: Bill Rohrer's original write-up with Tom Juengst's tracked changes; water control structures definition incorporated into the CBP BMP template.
- Tom requests the addition of "drainage" to title of BMP to separate it from other water control structures.
- Bill Rohrer's comments on Tom's edits had not been received yet.
- Workgroup comments on the original write-up:
 - Paragraph 2: insert "maintain flashboard settings to retain storm runoff" before "water levels within 30 inches of the ground".
 - Jen mentioned there is more specificity concerning water levels in this draft than the original.
 - Paragraph 3:
 - replace "Delmarva" with "Chesapeake Bay Region".
 - sentence 1: insert "properly managed" before "drainage water control structures".

- Replace “this adjustment” with “the adjustment to 33%”.
- Shift sentence “For application...” to end of first paragraph.
- Put the revised write-up into the CBP’s new BMP template that will be used for all new BMPs. Most of paragraph 3 will fit into the conservatism section of the template. Adam will email the revised template to the Workgroup.
- Jen Campagnini will check into the cost share lifespan of the BMP.

Handout: 33% Nitrogen Reduction Due to Water Control Structure Best Management Practices

Delmarva’s poorly drained lands have been outfitted with an intensive and extensive artificial drainage system. These ditch systems are often necessary in coastal plain regions in order to create agricultural land suitable for cultivation on the very flat topography. Water control structures are also efficient Best Management Practices (BMPs) for water quality, when properly designed and managed. Nitrogen loads from agricultural fields with water control structures have been found to be 45% less than from fields without this practice, on average (Evans et al., 1989; Evans et al., 1996; and Osmond et al., 2002). The load reduction occurs as the result of two processes; (1) volume flow and (2) nutrient concentration. By design nature, water control structures reduce the total volume of water flow. In turn, the inorganic nitrogen concentrations in the drainage waters are often reduced. Since water control structures effects the water table, more water and nitrogen can be taken up by crops and vegetation in or along the ditch. During the non-growing season, water control structures still provide opportunities for nitrogen reduction as denitrification occurs.

In order to estimate nutrient reductions from the BMPs currently in use, a nitrogen reduction efficiency of 33% has been assigned to the nitrogen load of acres drained. Additionally, phosphorus reductions from water control structures are typical. However, many Delmarva soils with drainage systems have higher than average soil phosphorus levels, hence, no reduction is currently assigned.

REFERENCES

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Appendix D: Panel Charge and Scope of Work

[Editor's Note: Provided as a separate document for CBP review purposes. Additional Appendices will be made available as needed, e.g., Appendix E to compile CBP partnership feedback and responses from the Panel Chair and Coordinator and Appendix F to compile all panel minutes]

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Appendix E: Monitored Denitrifying Bioreactors for Springs or Seeps

During the EP's review of practices included in this report, it became aware of an emerging interest in the use of denitrifying bioreactors to reduce N loads from freshwater springs in areas such as the Shenandoah Valley in Virginia. Application of denitrifying bioreactor technology in this setting is outside of this EP's charge, however, there is acknowledgement across the CBP partnership that the BMP review protocol, based on the National Academies of Science, is time and resource intensive and this EP's report may be the best place to house any recommendations for this emerging technology for the time being.

Review of science and literature

Limited information is available in order for this EP to recommend an efficiency value for DNBRs as a water quality BMP when applied to a freshwater spring or seep. During the public feedback period for this report, the EP was presented with a contemporaneously published study (Easton et al. 2019) illustrating the nutrient reduction potential of denitrifying bioreactors applied to springs.

To estimate bioreactor performance effectiveness, Easton et al. used literature values and measurements from a pilot-scale bioreactor installed on a spring in southwest Virginia. They then used data for USGS-identified freshwater springs to estimate potential reductions for freshwater springs for a range of nitrate concentrations and bioreactor treatment volumes. They concluded that implementation of bioreactors on 48 springs with nitrogen concentrations of 3 mg/L or higher and flows of at least 500 m³/d (~132,086 gallons/day) could remove approximately 710 lbs to 1300 lbs N per day. They estimate that the annual unit cost to remove N ranges between \$0.54 and \$7.60/kg per year, depending on the efficiency of the bioreactor and the influent N concentration.

Easton et al. also noted that the mean nitrate removal rate (8.8 g/m³ per day) for their pilot-scale spring bioreactor was greater than the mean removal rate of bioreactors applied to agricultural drainage systems, despite lower influent N concentrations in the spring relative to typical agricultural drainage water.

Recommended effectiveness estimates

Given the results demonstrated by Easton et al. (2019), the panel is comfortable with a directly-measured version of the DNBR practice to reduce loads from the "AG" load source group. The amount of TN removed must be calculated from monitored nitrate concentrations and the total treated flow volume for each eligible bioreactor. This directly-measured option is restricted to DNBRs that treat springs or seeps, and is not applicable to other DNBRs such as those that treat tile-drainage flow.

Future research needs

Most information known to the panel has been anecdotal thus far, and only one peer-reviewed study (Easton et al., 2019) was provided during partnership review of this panel's draft report.

Virginia noted the potential application of this BMP within their draft Phase III WIP and therefore the panel would expect VA DEQ and its academic partners to share new data with the partnership when available in the future.

References

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