Adoption & Implementation of Drainage Water Management
Technical Brief

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This technical brief examines producer and environmental benefits of drainage water management (DWM). The anticipated audience for this brief likely will be potential technology investors and policy makers interested in advancing DWM practices to improve water quality.

The Need for Drainage Water Management

Build-up of the current agricultural drainage network began during the 1870s as part of a national land reclamation policy. Since then, drainage has been both criticized and praised. Drainage systems lowered water tables, resulting in improved field trafficability, decreased risk, and increased yields (Zucker and Brown, 1998). Overall, agricultural drainage enabled previously marginal land to become highly productive and profitable farmland (USDA, 1987, Strock *et al*., 2010). A highly successful agricultural system was established in the central and eastern United States, but intense drainage also contributed to negative environmental impacts, including substantial losses of wetlands and wildlife habitat (USDA, 1987).

Drainage systems consist of both surface and subsurface components; however, this document focuses on subsurface drainage systems and the associated environmental costs and benefits. Subsurface drainage shifts the pathway of water leaving farm fields by redirecting flows to subsurface tile lines, thereby reducing surface runoff. These reductions in surface runoff result in decreased export of sediment and sediment-bound chemicals such as phosphorus and pesticides (Skaggs and Youssef, 2008). However, the increase in high-intensity subsurface drainage also correlated with an increase in nitrogen export from farm fields (Randall and Mulla, 2001; McIsaac and Hu, 2004; Blann *et al*., 2009; Goswami *et al*., 2009).

Subsurface drainage lines act as conduits of nitrate – the mobile form of nitrogen – to surface waters. Under natural conditions, nitrate-laden water would filter through the soil profile and be removed, at least partially, through denitrification. In fields with subsurface drainage, tile lines intercept the water before denitrification can occur. As a result, subsurface drainage effluent typically contains high concentrations of nitrate (Randall and Mulla, 2001; Mitsch *et al*., 2001; Dinnes *et al*., 2002). For example, one study compared two watersheds – one with subsurface drainage and one without drainage. The nitrate losses from the tile-drained watershed were 42.9 kg/hectare, compared to 7.0 kg/hectare from the non-drained watershed (Goswami *et al*., 2009).

Excess nitrate concentrations can pose human health risks and contribute to the impairment of aquatic ecosystems. A review of studies measuring flow-weighted nitrate concentrations in drainage effluent from tile-drained fields found that the majority reported concentrations exceeding the federal drinking water limit of 10 mg/L (Blann *et al*., 2009). However, tile effluent concentrations typically do not reflect stream concentrations. Agricultural nitrogen export also is a major contributor to the degradation of aquatic ecosystems through direct toxicity to organisms, alteration of food webs, and eutrophication (Rabalais, 2002). Although recent research has indicated that phosphorus also plays a role in Gulf of Mexico eutrophication, nitrogen plays a primary role in causing algal blooms in the Gulf and waters along the U.S. Atlantic coast (EPA, 2007). Agricultural activities are considered the dominant source of nitrogen loading to these regions (Downing *et al*., 1999; Rabalais *et al*., 2002; EPA 2007).

Reducing the environmental impacts from agricultural nitrate exports can be achieved through the implementation of a variety of best management practices. In particular, subsurface drainage systems can be re-designed to enable a producer to control the flow of subsurface effluent (Evans *et al*., 1995). Reducing the volume of effluent has been shown to effectively reduce the amount of nitrate leaving the field (Skaggs and Youssef, 2008). Drainage rates and volumes can be controlled with a device that enables the drainage outlet level to be raised and lowered as needed. Raising the outlet level reduces the hydraulic gradient to the drain and elevates the water table (Zucker and Brown, 1998; Frankenberger *et al*., 2006; Skaggs and Youssef, 2008).

Controlled drainage devices can be adjusted based on the season and drainage needs. Outlet levels can be lowered prior to planting to allow the water table to drop and the fields to become sufficiently dry for equipment access. After planting, outlets can be raised to improve water availability to young plants. Then, subject to producer time constraints, the level of the outlet can be adjusted throughout the growing season in response to weather conditions. After harvest, the outlet level is raised to minimize drainage during the non-cropping season. Depending on the type of drainage device installed, a producer potentially can use a computer to raise and lower the outlet level (ADMC, 2011).

Potential for Drainage Water Management Adoption

Optimal sites for drainage water management (DWM) are flat, uniform fields. Slopes should be one-percent or less, although steeper slopes can be accommodated by installing additional drainage control structures (Dinnes, *et al*.; 2002; Frankenberger, 2006). High-priority fields for DWM include regions with high in-stream nitrate concentrations and catchment areas that have been identified as substantial contributors to nutrient loading (see Figure 1).

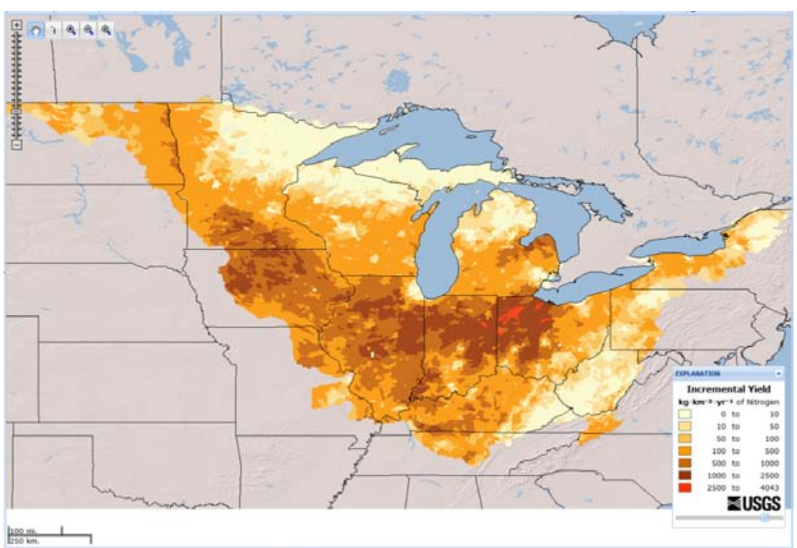

Figure 1. SPARROW model output depicting the incremental yield of nitrogen (kg/km yr) from agricultural practices. Nitrogen hotspots, as indicated in the figure, can be targeted for DWM (See http://cida.usgs.gov/sparrow/).

These nitrogen hot spots typically coincide with high tiling densities (see Figure 2). However, the exact extent and location of tile drains is unclear, as discussed later.

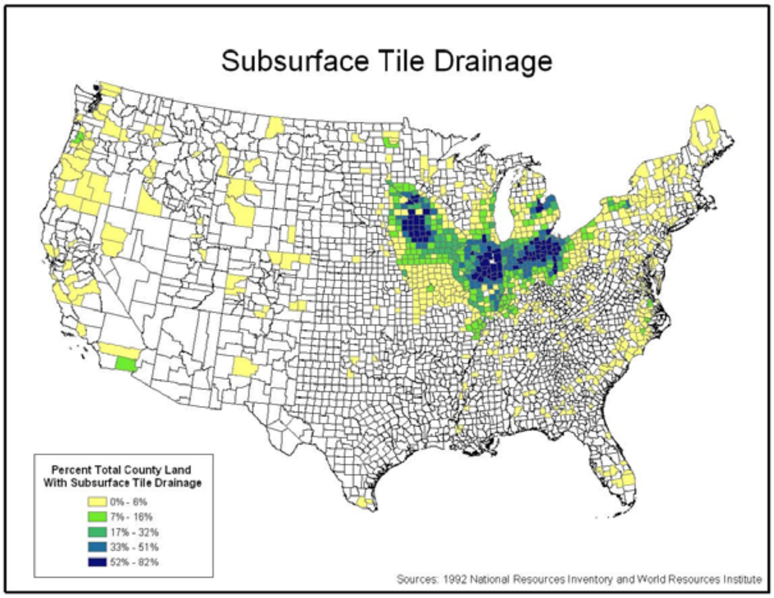

Figure 2. The extent and location of subsurface drainage, as estimated by Sugg, 2007.

DWM was recently introduced to the Midwest. It primarily is being implemented at research and demonstration sites (Cooke *et al*., 2005). Given how new the technology is to the region, current implementation rates by land managers are unclear. To promote adoption, the United States Department of Agriculture (USDA), Natural Resources Conservation Service (NRCS) works with landowners and producers encouraging DWM implementation as a water quality BMP (NRCS CPS 554). States can develop their own variations of this standard. Currently, eight of the twelve states in Mississippi River Basin Initiative (MRBI) allow for DWM as a BMP. Of those eight states, payment rates vary from $4.80/acre to $45/acre (NRCS, 2011). To further promote DWM establishment, the USDA and other entities formed the Agricultural Drainage Management Systems Task Force in 2003. A separate group of industry representatives also formed the Agricultural Drainage Management Coalition (ADMC). Recently, this coalition was awarded a USDA-NRCS Conservation Innovation Grant to study the water quality benefits of DWM in five Midwestern states.

Assessing the total potential environmental benefit for DWM requires knowledge of the extent of subsurface drainage and the location of tile lines. However, the existing data sets contain substantial gaps that researchers are attempting to fill (Sugg, 2007). A USDA report (1987) estimated that, as of 1985, artificial drainage was present on approximately 110 million acres throughout the U.S. However, the proportion of drainage that is subsurface varies from state to state. In upper Illinois and Iowa, the report estimated that 85-percent of farm-field drainage consists of subsurface lines (USDA, 1987). To fill the data gap, researchers have used soil and land-use models to estimate the extent of subsurface drainage (Sugg, 2007; Jaynes *et al*., 2010). Sugg (2007) combined soil and land-use datasets and estimated that subsurface drainage existed on approximately 38.7 million acres in Corn Belt and Lake states (see Figure 2). However, not all crop land is sufficiently flat or uniform for DWM. Limiting the analysis to fields with a slope of 0.5-percent or less, Jaynes *et al*. (2010) estimated that 10 million hectares of cropland (4.8 million hectares of corn land) in the Midwest would be suitable for DWM.

Investment costs and uncertainty about water quality benefits are the primary concerns hindering DWM adoption. Site-specific conditions and advanced technology options can make DWM a costly option. A time opportunity cost also is associated with managing the drainage system. Expecting a producer to bear the costs of DWM implementation is complicated by the fact that many of the benefits accrue off-site (Strock *et al*., 2010). In many localities, nitrate concentrations in water are not considered a major problem (Nistor and Lowenberg-DeBoer, 2006). Currently, producers are not required to reduce nutrient exports from farm fields. Therefore, DWM adoption will be largely voluntary and will depend on the potential for private benefits (Nistor and Lowenberg-DeBoer, 2006). Cost-share payments or other funding mechanisms could improve producer incentives and the potential for DWM adoption.

Problems Addressed by DWM

DWM implementation has been shown to substantially reduce nitrate losses from farm fields, thereby contributing to water quality improvements. Nitrate loss reductions are achieved by reducing the drainage volume from tile drain outlets. Drainage water management typically does not reduce the concentration of nitrate in the effluent. Most of the nitrate reductions from DWM systems occur when drain flow is reduced during the non-cropping season. In humid temperate regions, approximately 88 to 95-percent of nitrate loss through drainage occurs during the fallow period and concentrations often exceed the drinking water limit (Drury, *et al*., 2009). Following harvest, the drainage outlet should be adjusted to bring the water table near the surface. The actual distance from the surface will vary based on individual BMP specifications set by the NRCS and state agencies. Additional environmental benefits could be achieved by allowing the water table to rise above the soil surface during the fallow period. Ponding on farm fields would provide wildlife benefits by creating seasonal pot-hole wetlands for migratory birds.

Preliminary Quantification of Potential Water Quality Benefits

Nitrate loss reductions associated with DWM vary widely depending on site-specific conditions. According to one study, reductions can range from 17 to 94-percent (Skaggs and Youssef, 2008). Quantifying the reductions is complicated by a lack of complete understanding about the fate of nitrate remaining in the field. Possible pathways include uptake by crops, increased lateral and deep seepage, and deep percolation and subsequent denitrification (Strock *et al*., 2010). Studies have shown sufficient organic carbon deeper in the soil profile to support denitrification. There also is evidence of denitrified zones in soils that are classified as poorly or very poorly drained. Both of these situations indicate the nitrate is converted to other forms that have less of an impact on water quality. This does not appear to be the case in situations where soil is classified as moderately well-drained (Skaggs and Youssef, 2008). However, subsurface drainage systems are most prevalent in poorly or very poorly drained soils.

Nitrate reduction benefits from DWM can be limited by certain non-controllable factors such as site characteristics, temperature, and precipitation (Randall and Mulla, 2001). DWM nitrate loss reduction potential tends to be greatest in warmer climates (Thorp *et al*., 2008). For example, a model simulation estimated annual maximum loss reductions of 49.6 kg/hectare in Memphis, Tennessee, and annual minimum loss reductions of 7.1 kg/hectare in Fargo, North Dakota (Thorp *et al*., 2008). Loss reductions also depend on hydrology and the nitrogen dynamics in the soil profile (Skaggs and Youssef, 2008). Increased surface runoff associated with DWM also could potentially offset nitrate loss reductions if additional overland flow increases loading of sediment, pesticides, and phosphorus (Strock *et al*., 2010). However, one study that observed increased surface runoff found this runoff only resulted in a minor surface loss of nitrate (Drury *et al*., 2001). DWM can be utilized to reduce soil erosion if the system is managed to keep the water table below the surface and prevent ponding. However, this reduces the potential habitat value of surface waters called for by some NRCS 554 standards.

Estimating the full nutrient reduction potential of DWM has required use of models to fill data gaps presented by experimental studies. Field research assessing DWM benefits typically was performed in short time periods at a limited number of sites (Thorp *et al*., 2008). Differences in site-specific characteristics reduce the ability for these results to be extrapolated to other parts of the United States. To address the need for additional data, various researchers have used models to estimate long-term effects of DWM on water quality. Thorp *et al*. (2008) used a hybrid agricultural systems and crop growth model to simulate drain flow from conventional drainage and DWM in the Midwest. The model results estimated a regional percent reduction in flow of 53-percent and a regional reduction in nitrogen loss of 51-percent (Thorp *et al*., 2008). Jaynes *et al*. (2010) used Thorp's simulation results to estimate the nitrate reduction potential specifically on corn land suitable for DWM. According to Jaynes *et al*. (2010), 4.8 million hectares of cornlands are suitable for DWM, if only fields with slopes less than or equal to 0.5-percent are included. Implementing the practice on all suitable corn land in the Midwest would result in nitrate loss reductions of about 83 million kg/yr (Jaynes *et al*., 2010). In the Upper Mississippi River and Ohio/Tennessee River watersheds alone, DWM could be implemented on 2.9 million hectares and reduce nitrate losses by 52 million kg/yr (Jaynes *et al*., 2010)[1].

The nitrate loading reduction estimates produced by Jaynes *et al*. (2010) were compared to the overall nitrogen loading to the Gulf of Mexico. Preliminary calculations estimated the potential contribution of DWM to reducing nutrient loading to the Gulf. All percent reductions are based on 2001 to 2005 loading estimates (EPA, 2007). During these years, an average of 813,000 metric tonnes of nitrate-N (813 million kg) per year were reportedly transported to the Gulf. Based on this loading estimate and the DWM nitrate reduction estimate of 52 million kg/year, implementing DWM on all suitable lands in the Upper Mississippi and Tennessee/Ohio watersheds could reduce overall nitrate loading to the Gulf by 6.4%.

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[1]In Jaynes *et al.,* (2010), it was unclear if the mass reductions were reported in terms of nitrate or nitrate-N. Correspondence with Dr. Dan Jaynes clarified that the reductions were nitrate as nitrogen.

Preliminary Calculations of Nutrient Reduction Costs

Costs of implementing DWM vary based on site characteristics, drainage system design, and the type of control structure installed. One estimate calculated a total cost range of $300 for a homemade weir, to more than $3,000 for a large prefabricated structure (Evans *et al*., 1996). Another study estimated costs could range from $161/hectare for a new installation on a 6-inch main to $217/hectare for a retrofit on a 12-inch main (ADMC, 2011). Annualizing these costs based on a 15-year lifetime and a 8-hectare treatment area, estimated costs ranged from $6.73/year on a 6-inch main and $9.08/year on a 12-inch main (ADMC, 2011). Cooke *et al*. (2005) estimated $50/hectare to $100/hectare for a retrofit installation and $220/hectare for a new system in complex topography. Assuming a 30-percent nitrogen load reduction, the costs for a retrofit would be $1.45/kg to $2.05/kg and the costs for a new installation would be $6.30/kg to $9.20/kg (Cooke *et al*., 2005). Jaynes *et al*. (2010) estimated a cost of $2.71/kg when the costs were applied over a 20-year lifetime at a 4% interest rate, and found this price to be cost-competitive with other nitrogen removal practices. For example, constructed wetlands cost $3.26/kg, fall cover crops cost $11.06/kg, and bioreactors cost $2.39/kg to $15.17/kg (Jaynes *et al*., 2010). Advances in technology are likely to reduce the cost of DWM implementation.

A simple analysis was conducted to estimate the cost of DWM under various scenarios and assumptions. Provisional implementation costs were calculated based on the assumptions used by Jaynes *et al.* (2010), with a few modifications. Jaynes *et al.* determined that 2.9 million hectares of cornland in the Upper Mississippi and Tennessee/Ohio watersheds were suitable for DWM. Within these areas, 20-percent of DWM implementation would be retrofits and 80-percent would be new installations. A retrofit was assumed to drain 4.8 hectares while a new installation would drain 8 hectares. Both the new and retrofit practices had a unit cost of $1,100, and new installations included an additional cost of $80.36/hectare[2]. Applying these assumptions, a basic analysis indicates the following costs associated with DWM implementation:

- The total cost of implementing DWM on all suitable cornland in the Upper Mississippi and Tennessee/Ohio watersheds would be $638 million
- The cost of retrofit installations would be $229/hectare
- The cost of new installations would be $218/hectare
- The cost of nitrate reductions achieved by implementing DWM on all suitable cornland in the Upper Mississippi and Tennessee Ohio watersheds would be $12.28/kg nitrate

DWM implementation costs potentially could be offset by a yield increase, depending on the specific application of controlled management. Yield increases associated with DWM ranged

[2] It is unclear how Jaynes *et al.* (2010) derived these annualized costs for nitrate reductions associated with DWM. As such, some of the numbers included here differ from those reported by Jaynes *et al.* (2010). The cost analysis could be adjusted to include data that might better represent the current status of DWM technologies.

from less than 5-percent (Cooke *et al*., 2005) to 9-percent with average rainfall and 58-percent with below-average rainfall (Zucker and Brown, 1998). A yield increase of 1.68 bushels/acre for a 6-inch main and 2.27 bushels/acre for a 12-inch main would offset the control structure expense, assuming $4/bushel corn (ADMC, 2011)[3].

Implementation costs also potentially could be covered through a water quality trading (WQT) program. With WQT, municipal wastewater treatment plants (WWTPs) could meet their own regulatory compliance goals for nitrogen by purchasing nitrogen reduction credits from producers implementing DWM. A recent WQT Feasibility Study of costs for WWTPs in the Wabash River basin of Indiana cited new nitrogen controls ranging from $2.20 to 8.53/kg for large plants (discharging more than 5 million gallons per day--MGD) to $3.02 to 146.48/kg for medium plants (0.3 to 5MGD) to $45.26 to 174.91/kg for smaller plants (less than 0.3 MGD) (CTIC, 2011). WQT programs account for uncertainties in treatment efficiencies by applying a trade ratio. Even with the application of a typical 2:1 trade ratio and incorporating transaction costs, DWM would still be highly cost-effective compared to achieving nitrate reductions through WWTP upgrades.

Conclusions

It has been documented that DWM practices substantially reduce nitrate losses to surface waters. These reductions primarily are due to reductions in the volume of tile drain effluent. The magnitude of potential nitrate reductions makes DWM a promising option for addressing the environmental impacts of nitrogen loading, including hypoxia in the Gulf of Mexico. However, it should be noted that certain site-specific characteristics might limit the nitrate reduction potential in some areas. There is substantial regional variability in overall nitrate loss reductions achieved with DWM. In addition, data gaps prevent a clear picture of the potential for DWM, but ongoing research is attempting to fill these gaps. Cost estimations also would benefit from more updated information that better represents the current status of drainage management technologies. For regions that could benefit from DWM, cost-share programs or other financial options, such as market-based incentives, could increase adoption rates. Specific applications of DWM also could result in yield increases, making the technology even more attractive to the producer.

[3] These estimates are based on annual per acre installation costs of $6.73 for a 6-inch main and $9.08 for a 12-inch main. The costs were calculated by annualizing total costs over 15 years at 6% interest.

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