Drainage water management effects on tile discharge and water quality

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A B S T R A C T

Drainage water management (DWM) has received considerable attention as a potential best management practice (BMP) for improving water quality in tile drained landscapes. The objective of this study was to evaluate the effects of DWM on subsurface drain discharge as well as on nitrogen (N) and phosphorus (P) loads in drainage water. Tile discharge and nutrient concentrations were measured from two adjacent tile drainage outlets in an Ohio, USA headwater watershed for 7 years (2006–2012). A control structure was installed in 2009 to allow DWM at one of the outlets from 2009 to 2012. A before–after control–impact (BACI) study design was used to assess the impact of DWM on tile discharge and nutrient loads. Results showed that DWM significantly decreased annual tile discharge between 11 and 178 mm, which was equivalent to an 8 to 34% reduction in flow. DWM significantly decreased annual NO3−N loads by −1.3 to 26.8 kg ha−1 (−8 to 44%) and annual dissolved P loads by 0.04 to 0.51 kg ha−1 (40 to 68%). Nutrient concentrations were not significantly affected by DWM indicating that decreases in nutrient loads were primarily due to reductions in tile discharge rather than changes in concentration. Results from the current study support the use of DWM as a BMP to decrease N and P loads in subsurface drain discharge throughout the U.S. Midwest. Future research should focus on quantifying the effect of DWM on nutrient transport in other flow paths (e.g., lateral seepage, surface runoff) to further evaluate its use as a BMP in tile drained landscapes.

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1. Introduction

Subsurface (tile) drainage is used extensively throughout the U.S. Midwest, Canada, and northern Europe to lower the water table and drain soils that are seasonally or perennially wet (Pavelis, 1987). It is often considered “the most extensive soil and water management activity in agriculture” (Pavelis, 1987) and is required for economical agricultural production in many areas of the U.S. Midwest (Zucker and Brown, 1998). Subsurface drainage can also negatively affect surface water quality (Skaggs et al., 1994) as elevated nutrient concentrations in drainage waters are common (e.g., Gilliam et al., 1999). Recently, considerable attention has been given to the promotion of drainage water management (DWM) or controlled drainage as a potential best management practice (BMP) to reduce nutrient loads in drainage water while maintaining profitable crop production (Nistor and Lowenberg-DeBoer, 2007; Skaggs et al., 2012). There are, however, a limited number of studies that have documented the effectiveness of DWM in mitigating nitrogen (N) and phosphorus (P) loads (Skaggs et al., 2012). An improved understanding of the effect of DWM on N and P dynamics in subsurface drainage systems is therefore critical to determining its value as a BMP.

Drainage water management is the practice of seasonally adjusting the outlet elevation of the drainage system through installation of a control structure, which is typically comprised of stackable boards or stop logs. The outlet elevation can be set at any level between the ground surface and the drainage depth. Excess and deficit soil–water conditions in the soil profile can be managed with the use of DWM to provide adequate drainage during critical planting and harvesting operations (Drury et al., 1999) and prevent excessive drainage of the crop root zone after planting or during winter periods when drainage is not required (Cooper et al., 1991). Strock et al. (2010) suggested that DWM will be most effective as a BMP on relatively flat lands where one control structure...
will affect the outlet water level and drainage rates from a large area. Approximately 10 million ha of cropland in the U.S. Midwest (soils normally drained with slopes <0.5%) are suitable for DWM (Janes et al., 2010).

Reductions in subsurface flow volume of 20–95% have been reported with DWM compared to conventional or free drainage (FD) systems (Skaggs et al., 2012). Similar decreases in N loading often accompany reductions in flow volume. Indeed, Cooke and Verma (2012) found that subsurface flow volume was decreased by 38–89% with DWM and resulted in N loads that were reduced by 51–79%. Skaggs et al. (2012) concluded that DWM had the potential to decrease N loading by 18–79%. Phosphorus losses via subsurface drainage have received considerably less attention compared to N losses, but a growing body of literature has highlighted the significance of P transport in agricultural subsurface drainage (King et al., 2014a). Drainage water management may therefore be an effective practice for mitigating subsurface P loads. For example, total P (TP) loads in subsurface drain discharge from two field plots with DWM in Sweden were 60% less than a comparable field plot with FD (Wesström and Messing, 2007). Similarly, Evans et al. (1995) found that TP loads in drain water were 30% less after implementation of DWM.

The objective of this study was to evaluate the effects of DWM on subsurface drain discharge, nutrient concentrations, and nutrient loads in subsurface drainage water. Two adjacent tile drain outlets in an Ohio, USA headwater watershed were monitored for 7 years using a before–after control–impact (BACI) design in order to assess the impact of DWM over a range of crop, weather, and environmental conditions. Specific objectives were to (1) quantify subsurface discharge and N and P dynamics from a subsurface drainage system before and after implementation of DWM; and (2) assess the potential for DWM to mitigate N and P loss by comparing results from a subsurface drainage system with DWM to a system with FD.

2. Materials and methods

2.1. Site description

The Upper Big Walnut Creek (UBWC) watershed (492 km²) is an 11-digit watershed (HUC 05060001-130) located in central Ohio, USA (Fig. 1). The experimental site for the current study is located within a 389 ha subwatershed of the UBWC identified as subwatershed B (Fig. 1). Crop production agriculture (73%) comprises the largest land use classification within the watershed, with the remainder of the watershed consisting of woodland (6%) and urban/farmstead (21%) land uses. The cropland is primarily in a corn–soybean rotation using rotational tillage. The soils within subwatershed B are a somewhat poorly drained Bennington silt loam (52.9%) and a very poorly drained Pawamo clay loam (46.2%). An estimated 80% of subwatershed B is systematically tile drained with laterals on 15 m spacing and placed approximately at a depth of 1 m. Previous research in the watershed has shown that tile discharge comprises 47% of streamflow at the watershed outlet annually (King et al., 2014b) and is a major contributor of N and P to stream water (King et al., 2014c).

Two adjacent tile drain outlets on the south side of the drainage ditch in subwatershed B were used to investigate the impacts of DWM on subsurface drain discharge and water quality over a 7-year period (2006–2012) (Fig. 1). These tiles drain water from adjacent parts of a large field with uniform management and history. Specific site characteristics are shown in Fig. 1. Precipitation was measured on site using a standard rain gauge and an Isco 674 tipping bucket rain gauge (Teledyne Isco; Lincoln, NE).

A control structure (Agri Drain Co.; Adair, IA) was installed at B4 in 2009 to manage the elevation of the drain outlet. The goal was to restrict the drainage for as much time as possible, while maintaining adequate drainage for spring and fall field operations and root zone aeration. Thus, the outlet at B4 was lowered a few
Fig. 2. Daily tile discharge from B2 and B4. From 2006 to 2008, both sites were free-draining. From 2009 to 2012, B2 remained free-draining, while drainage water management was implemented at B4. The highlighted bars indicate times when the outlet height was raised at B4.

weeks before the beginning of spring or fall fieldwork and raised to 45 cm below the soil surface shortly after the spring or fall fieldwork was completed. Fig. 2 shows the dates when the outlet elevation was adjusted. The other site, B2, was free-draining for the entire study period.

2.2. Flow measurement and sampling

Discharge from the tile drains at B2 and B4 was quantified using compound weir inserts (Thel-Mar, LLC; Brevard, NC) that were installed at the tile outlet. The existing 20-cm diameter tile drain outlet pipes were cut and fitted with 30-cm diameter pipes that could accommodate the compound weir. Each tile outlet was equipped with an Isco 4230 Bubbler Flow Meter, which was programmed to record flow depth every 10 min. An Isco 2150 Area Velocity Sensor was also installed in each tile outlet to aid in the development of a rating curve during submerged conditions. Discharge rates were calculated and aggregated to daily volumes for both tile drains using the 10-min measured stage in conjunction with the standard rating curve for the compound weir insert or the area velocity data collected from the tile outlet.

Water samples were collected and analyzed from both tile outlets. An Isco 6712 Portable Sampler was installed at each tile outlet to collect water samples every 6 h, and four aliquots were placed in each bottle to comprise a daily sample. Upon return to laboratory, daily samples were composited into a weekly sample and were then analyzed according to U.S. EPA guidelines for N and P (U.S. EPA, 1983). Briefly, samples were filtered (0.45 μm) and analyzed for concentrations of nitrate–N (NO$_3$–N) and dissolved reactive P (DRP) with a Lachat Instruments QuikChem 8000 FIA Automated Ion Analyzer (Lachat Instruments Inc.; Loveland, CO). Total N (TN) and total P (TP) concentrations in water samples were determined on unfiltered samples following alkaline persulfate oxidation (Koroleff, 1983). Concentrations of NO$_3$–N and DRP comprised greater than 90% of TN and TP, respectively, on all sampling dates; thus, the main focus of this paper is on the dissolved analytes, NO$_3$–N and DRP. All water samples were refrigerated and analyzed within 28 days following collection.

2.3. Statistics and data analysis

To determine nutrient loads from the tile outlets, the weekly analyte concentration was multiplied by the measured water volume for that respective sample. The volume of water associated with any one sample was determined using the midpoint approach. That is, the temporal midpoint between each sample was determined and the volume of water calculated for the midpoint to midpoint time duration. The analyte concentration was assumed to be representative over its sampling interval.

Tile discharge and nutrient data (concentrations and loads) were analyzed using a before–after control–impact (BACI) study design to determine whether applying the DWM treatment at B4 significantly affected tile flow and water quality (Smith, 2002). Data were classified as being from either before (2006–2008) or after (2009–2012) the implementation of DWM at B4. Site B2 was free-draining throughout the entire study period and served as the control, whereas B4 was considered the impact site due to the installation of the control structure and subsequent treatment (i.e., management of the drainage outlet elevation). The strength of the BACI design lies in the assumption that any changes over time (e.g., weather, crop, management) in the impact site, unrelated to the treatment, are controlled for by these same changes over time in the control site. Thus, a significant time (before–after) × treatment (control–impact) interaction indicates that the experimental treatment truly has an effect on the impact site (Smith, 2002). BACI analysis was completed using a repeated measures generalized linear mixed model in R (R Development Core Team, 2011). The effect of DWM on tile discharge and nutrient concentrations and loads
Table 1
Annual rainfall, crop rotation, tile flow, and tile discharge to precipitation ratio (% of annual precipitation that left the site as tile flow) for B2 and B4. From 2006 to 2008, both sites were free-draining. From 2009 to 2012, B2 remained free-draining, while drainage water management was implemented at B4.

<table>
<thead>
<tr>
<th>Year</th>
<th>Rainfall (mm)</th>
<th>Crop rotation</th>
<th>B2 Annual tile flow (mm)</th>
<th>Q/P (%)</th>
<th>B4 Annual tile flow (mm)</th>
<th>Q/P (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>2006</td>
<td>1064</td>
<td>Corn</td>
<td>239</td>
<td>22</td>
<td>245</td>
<td>23</td>
</tr>
<tr>
<td>2007</td>
<td>1095</td>
<td>Soybean</td>
<td>339</td>
<td>31</td>
<td>335</td>
<td>31</td>
</tr>
<tr>
<td>2008</td>
<td>1006</td>
<td>Soybean</td>
<td>222</td>
<td>22</td>
<td>312</td>
<td>31</td>
</tr>
<tr>
<td>2009</td>
<td>938</td>
<td>Corn</td>
<td>154</td>
<td>16</td>
<td>139</td>
<td>15</td>
</tr>
<tr>
<td>2010</td>
<td>773</td>
<td>Soybean</td>
<td>106</td>
<td>14</td>
<td>135</td>
<td>17</td>
</tr>
<tr>
<td>2011</td>
<td>1239</td>
<td>Corn</td>
<td>463</td>
<td>37</td>
<td>351</td>
<td>28</td>
</tr>
<tr>
<td>2012</td>
<td>793</td>
<td>Soybean</td>
<td>84</td>
<td>11</td>
<td>97</td>
<td>12</td>
</tr>
<tr>
<td>Avg.</td>
<td>986</td>
<td></td>
<td>230</td>
<td>22</td>
<td>231</td>
<td>22</td>
</tr>
</tbody>
</table>

1 Q/P = discharge to precipitation ratio.

was determined for the entire study period as well as for individual years following implementation of DWM. A probability level of 0.05 was used to evaluate statistical significance.

Annual percent reductions in tile discharge and nutrient loads were calculated as outlined by Clausen and Spooner (1993). Briefly, using data from 2006 to 2008, the pretreatment flow and water quality relationships between B2 and B4 were established by linear regression (see footnote in Table 3). These equations were subsequently used to predict 2009–2012 flow and water quality from B4 based on measured values at B2. The change in both tile discharge and nutrient loads was determined by summing the difference in the observed value from B4 and the value predicted for B4 without DWM during 2009–2012.

3. Results

3.1. Tile flow

Annual tile flow at B2 and B4 tended to follow annual trends in precipitation (i.e., more precipitation, more tile flow) (Table 1). From 2006 through 2012, average annual precipitation was 986 mm, which is consistent with the long-term (30 year) average precipitation (985 mm) in the watershed. On average, 22% of annual precipitation was discharged as tile flow from B2 and B4 (Table 1). Annual tile discharge at B2 ranged from 84 to 463 mm, whereas annual tile discharge at B4 ranged from 90 to 351 mm.

Daily tile flow was similar in magnitude and duration between B2 and B4, which was expected given their similar area and management (Fig. 2; Table 2). Mean daily tile flow at B2 was 0.6 mm and ranged from 0.0 to 23.4 mm, while mean daily tile flow at B4 was 0.5 mm and ranged from 0.0 to 21.0 mm (Fig. 2). Tile flow at both sites was generally greatest during the winter and spring compared to the summer and fall, which was likely due to increased evapotranspiration during the summer and fall. Mean daily tile flow varied between 2006–2008 and 2009–2012 (p < 0.001) with greater mean daily flow rates observed from 2006 to 2008 (Fig. 2). A significant time (before–after) × treatment (control–impact) interaction was also found (p < 0.001), indicating that DWM significantly impacted daily tile flow at B4 (Table 2).

The relationship between mean daily tile flow at B2 and B4 from 2006 to 2008 period provided an estimate of mean daily tile discharge from B4 for the 2009–2012 period as if DWM was not implemented (Table 3). Comparison of observed and predicted values for B4 suggests that DWM decreased annual tile flow between 11 and 178 mm, with an average reduction of 69 mm. This is equivalent to a 7.5 to 33.6% decrease in annual tile flow. Results suggest that DWM had the largest impact on tile flow at B4 during years when the precipitation amount was average (2009) or above.

Table 2
Repeated measures analysis of tile flow and nutrient data (loads and concentrations) from B2 and B4 using generalized linear model techniques. Bolded p-values are significant at p < 0.05. A significant time (before–after) × treatment (control–impact) interaction indicates that drainage water management had an effect on tile flow, nutrient loads, or nutrient concentrations at B4.

<table>
<thead>
<tr>
<th>Flow</th>
<th>NO₃-N</th>
<th>DRP</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>F-value</td>
<td>p-value</td>
</tr>
<tr>
<td>C vs. I</td>
<td>0.9</td>
<td>0.394</td>
</tr>
<tr>
<td>B vs. A</td>
<td>24.4</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>CI × BA</td>
<td>18.9</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>Effects in 2009</td>
<td>23.8</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>Effect in 2010</td>
<td>2.3</td>
<td>0.126</td>
</tr>
<tr>
<td>Effect in 2011</td>
<td>72.2</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>Effect in 2012</td>
<td>15.1</td>
<td>0.003</td>
</tr>
<tr>
<td>C vs. I</td>
<td>274.1</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>B vs. A</td>
<td>65.5</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>CI × BA</td>
<td>0.9</td>
<td>0.394</td>
</tr>
<tr>
<td>Effects in 2009</td>
<td>0.5</td>
<td>0.469</td>
</tr>
<tr>
<td>Effect in 2010</td>
<td>0.2</td>
<td>0.691</td>
</tr>
<tr>
<td>Effect in 2011</td>
<td>1.8</td>
<td>0.186</td>
</tr>
<tr>
<td>Effect in 2012</td>
<td>0.8</td>
<td>0.273</td>
</tr>
</tbody>
</table>

1 Main effects = control vs. impact (C vs. I) and before vs. after (B vs. A); Interaction effect = control–impact vs. before–after (CI × BA).

1 Effect of the treatment in each year following implementation. Only the interaction effect (CI × BA) is shown.
3.2. NO$_3$–N and DRP concentrations

Nitrate-N and DRP concentrations over the study period are shown in Fig. 3. Nitrate-N concentration at both sites ranged from 0.1 to 60.2 mg L$^{-1}$ throughout the study period and averaged 10.0 and 16.9 mg L$^{-1}$ at B2 and B4, respectively. Mean NO$_3$–N concentration at both sites was typically greater during the winter and spring (14.7 mg L$^{-1}$) compared to the summer and fall (8.1 mg L$^{-1}$). Mean weekly NO$_3$–N concentration in drainage water was significantly greater at B4 than mean weekly NO$_3$–N concentration at B2 (p < 0.001) (Fig. 4). Mean NO$_3$–N concentration was significantly greater from 2009 to 2012 compared to the mean NO$_3$–N concentration from 2006 to 2008 (p < 0.001) (Fig. 4). Mean NO$_3$–N concentration from 2009 to 2012 was 6.6 mg L$^{-1}$ greater than mean NO$_3$–N concentration from 2006 to 2008. The difference in mean NO$_3$–N concentration between the two periods was primarily due to a large increase in NO$_3$–N concentration in 2009 (Fig. 4). The time (before–after) x treatment (control–impact) interaction, however, was not significant (p > 0.05), indicating that the implementation of DWM at B4 did not influence NO$_3$–N concentration (Table 2).

Mean weekly DRP concentration was generally greater at B2 compared to mean DRP concentration at B4 (p > 0.05) (Figs. 3 and 4). Mean weekly DRP concentration was 0.11 and 0.09 mg L$^{-1}$ at B2 and B4, respectively. DRP concentrations ranged from 0.01 to 0.79 mg L$^{-1}$ at B2, whereas DRP concentrations ranged from 0.01 to 1.28 mg L$^{-1}$ at B4. Mean DRP concentration were typically greater in the winter and spring (0.13 mg L$^{-1}$) compared to the summer and fall (0.08 mg L$^{-1}$). There was no significant difference in mean DRP concentration between the 2006–2008 and 2009–2012 periods (p > 0.05) (Table 2). Similar to NO$_3$–N concentration, the time x treatment interaction for mean DRP concentration was not significant (p > 0.05), indicating that DWM did not influence DRP concentration at B4 (Table 2).

3.3. NO$_3$–N and DRP loads

Mean weekly NO$_3$–N load over the study period was significantly greater at B4 (0.54 kg ha$^{-1}$) compared to mean NO$_3$–N weekly load at B2 (0.36 kg ha$^{-1}$) (p < 0.001). Weekly NO$_3$–N loads were between 0.00 and 6.50 kg ha$^{-1}$ for B2, while weekly NO$_3$–N loads were between 0.00 and 5.25 kg ha$^{-1}$ for B4. Mean weekly NO$_3$–N load for each year of the study is shown in Fig. 4. Mean NO$_3$–N load did not significantly differ between the 2006–2008 and 2009–2012 periods (p > 0.05) (Table 2). However, a significant time (before–after) x treatment (control–impact) interaction was observed (p = 0.034) (Fig. 4; Table 2), which indicates that DWM significantly decreased mean weekly DRP load at B4 compared to FD at B2.

Mean weekly DRP load was not significantly different between B2 (0.01 kg ha$^{-1}$) and B4 (0.01 kg ha$^{-1}$) or between the 2006–2008 and 2009–2012 periods (p > 0.05). Dissolved reactive P load at B2 ranged from 0.00 to 0.13 kg ha$^{-1}$, whereas DRP load at B4 ranged from 0.00 to 0.07 kg ha$^{-1}$. Mean DRP load for each year of the study is shown in Fig. 4. The time (before–after) x treatment (control–impact) interaction was however statistically significant (p = 0.049) indicating that DWM significantly decreased mean weekly DRP load at B4 compared to FD at B2 (Table 2).

Annual NO$_3$–N and DRP loads from B2 to B4 are shown in Fig. 4. During the 2006–2008 period, annual NO$_3$–N load was greater at B4 (23.9 kg ha$^{-1}$) compared to B2 (17.8 kg ha$^{-1}$). Annual DRP load over the same period tended to be greater at B2 (0.31 kg ha$^{-1}$) compared to B4 (0.29 kg ha$^{-1}$). Comparison of observed and predicted values for B4 (Table 3) suggests that DWM decreased annual NO$_3$–N load by 1.3 and 26.8 kg ha$^{-1}$, with an average reduction of 9.4 kg ha$^{-1}$. Therefore, DWM reduced the average annual NO$_3$–N load by 21.3% compared to FD (Table 3). Average annual DRP loads were also decreased with DWM compared to FD by 0.04 to 0.51 kg ha$^{-1}$, with an average reduction of 0.19 kg ha$^{-1}$. This was equivalent to a 40.0 to 68.0% decrease in DRP load. Similar to reductions in tile flow, the impact of DWM on NO$_3$–N load tended to be greater during years with average (2009) or above average (2011) precipitation compared to years with below average precipitation (Table 3). However, the impact of DWM on DRP load did not appear to be related to precipitation amount (Table 3).

4. Discussion

4.1. Impact of drainage water management on tile flow

Results showed that annual tile flow was decreased by 8 to 34% when the tile outlet elevation was managed seasonally with a control structure. This finding was generally less than, but comparable to other values reported in the literature. For example, Adeuya et al. (2012) observed a 15–24% reduction in tile flow from a field in Indiana, USA with DWM compared to a field with FD. In North Carolina, USA, both Gilliam et al. (1979) and Evans et al. (1995) found greater than 50% reductions in tile flow following implementation of DWM. In Ontario, Canada, Tan et al. (1998) measured a 20% decrease in tile flow with DWM. On several occasions, greater than 80% reductions in tile flow have also been recorded on plots with DWM compared with FD plots (Cooke and Verma, 2012; Westrom and Messing, 2007).

The large range (<20–95%) in tile flow reductions with DWM among studies is likely due to several factors including the height to which the outlet is raised. In the current study, the height of the outlet was raised to 45 cm from the soil surface and was only
lowered to accommodate spring and fall field operations. According to a review by Skaggs et al. (2012), the outlet height is typically raised to within 15 to 75 cm of the soil surface with an average height of approximately 30 cm. The closer the outlet height is set to the soil surface, the greater the potential to decrease tile flow; thus, increasing the outlet height in the current study may have yielded larger reductions in tile flow. While management goals must be considered when choosing the outlet height, research has also shown that varying the height of the outlet by season (i.e., growing vs. dormant) can be a viable strategy to decrease tile flow and still maintain profitable crop production (Adeuya et al., 2012; Westrom and Messing, 2007).

Differences in site characteristics (e.g., drainage intensity, soil properties) can also play a role in determining the impact of DWM on tile flow. For example, a study by Sands et al. (2008) found that shallow drain pipe installation and drainage systems designed for lower drainage intensity resulted in less tile flow. The impact of DWM on tile flow in drainage systems with closely spaced tile drains may therefore be more pronounced compared to drainage systems with tile drains that are spaced further apart (Strock et al., 2010). However, no studies have specifically examined the impact of DWM on drainage systems with different tile spacing. Research has also shown soil properties can influence tile hydrology. Poorly drained soils generally have lower peak discharge rates with subsurface drainage compared to sites that depend primarily on surface drainage (Skaggs et al., 1994). In contrast, on more permeable soils, where infiltration, water storage capacity, and lateral seepage are great enough to handle a given precipitation event, subsurface drainage can increase peak discharges (Wiskow and van der Ploeg, 2003). Thus, reductions in tile flow resulting from DWM are likely to vary depending on these site characteristics.

4.2. Impact of drainage water management on nutrient concentrations

Implementation of DWM at B4 did not significantly affect either NO$_3$–N or DRP concentrations compared to FD. The majority of studies investigating the role of DWM on water quality have reported similar findings. For example, Fausey (2005) and Adeuya et al. (2012) found no significant difference in NO$_3$–N concentration between DWM and FD; although NO$_3$–N concentration tended to be lower (<10%) in fields with DWM in both studies. Feser et al. (2010) and Valero et al. (2007) observed no significant difference in DRP concentration between DWM and FD sites, but both studies noted that DRP concentrations tended to increase with DWM compared to FD. Changes in nutrient concentrations are often attributed to higher water tables resulting from DWM, which promote anaerobic soil conditions.

Anaerobic soil conditions are required for denitrification to occur, which results in a decrease in groundwater NO$_3$–N concentration. Denitrification is also influenced by other factors including the amount of decomposable soil organic matter and carbon and temperature. Soil organic matter or carbon is generally greater near the soil surface due to the presence of crop residues and cover crops; therefore, in management scenarios where the outlet height is raised close to the soil surface, larger decreases in NO$_3$–N concentration may occur. However, in the current study as well as many others, the water table is often the closest to the soil surface during the winter and spring. Cold winter temperatures have been shown to limit the amount of denitrification (Stanford et al., 1975). Thus, while the potential for denitrification is high with DWM, other factors may ultimately limit the amount of denitrification that occurs and result in no significant differences in NO$_3$–N concentration between fields with DWM and FD.
In contrast to NO$_3$–N concentration, anaerobic soil conditions can increase the solubility and mobility of P (McDowell et al., 2012). Incubation experiments have shown that as soils become reduced, soil available P increased (Sallade and Sims, 1997; Vadas and Sims, 1997). In acid soils, this soluble P increase has been attributed to the reduction of Fe and Al oxides and the subsequent release of P bound with these compounds (Holford and Patrick, 1979). The potential for P loss based on changes in P solubility and extractability during soil reduction and reoxidation is therefore greater in P saturated top soils than in soils deeper in the profile (Vadas and Sims, 1997). However, several studies have indicated that the bulk soil does not become reduced long enough or to the extent necessary with DWM to cause reducing conditions in the soil that would significantly increase DRP concentrations (Wahba et al., 2001).

4.3. Impact of drainage water management on nutrient loads

Drainage water management decreased NO$_3$–N and DRP loads during the 4 years following implementation compared to FD. Since NO$_3$–N and DRP concentrations were not significantly different between B2 and B4, reductions in nutrient loads at B4 were primarily due to decreases in tile flow. Annual reductions in tile flow with DWM and the resulting reductions in NO$_3$–N load tended to be proportional (i.e., greater reductions in tile flow, the lower the NO$_3$–N load); however, decreases in DRP loads were similar among years regardless of the reduction in tile flow. This suggests that NO$_3$–N loads are more strongly tied to subsurface hydrology compared to DRP loads.

Reductions in NO$_3$–N load observed with DWM in the current study were comparable to reductions in NO$_3$–N load reported in the literature. In Illinois, USA, Cooke and Verma (2012) found that DWM resulted in NO$_3$–N loads that were reduced by 51–79%. Similarly, Lalonde et al. (1996) found that DWM decreased NO$_3$–N loads by 50–75% compared to FD in Ontario, Canada. In Ohio, USA, Fausey (2005) showed that DWM could reduce NO$_3$–N loads by 41%. At three sites in Ontario, Canada, Tan et al. (1998), Gaynor et al. (2002), and Drury et al. (2009) observed mean reductions in NO$_3$–N loads of 20, 16, and 29%, respectively. In all of these studies, decreases in
NO$_3$–N load with DWM were accompanied by similar decreases in tile flow.

The majority of water quality research associated with DWM has focused on NO$_3$–N loads; however, several studies have reported substantial reductions in DRP loads. Indeed, Wessström et al. (2001) and Wessström and Messing (2007) observed reductions in DRP load between 58 and 95% on field plots in Sweden with DWM compared to FD. In Minnesota, USA, Fezer et al. (2010) found that DWM decreased DRP loads by 63%. On the western delta of Egypt, Wabha et al. (2001) also found that DWM reduced DRP loads by 30 to 77% compared to FD. Similar to the current study, reductions in DRP loads reported in these studies were not always proportional to reductions in tile flow. Wessström et al. (2001) and Wessström and Messing (2007) were the only two studies that indicated that DRP loads were proportional to decreases in tile flow.

Although results from this study and others suggest that DWM substantially decreases nutrient loads compared to FD, the fate of nutrients that are retained by raising the outlet height is unknown. Increased denitrification rates are often cited as the main mechanism resulting in decreased NO$_3$–N loading from drainage systems with DWM; yet, most research opposes this claim. In addition to changes in denitrification rates, higher water tables during the growing season associated with DWM have been hypothesized to increase evapotranspiration and crop nutrient uptake. A review by Skaggs et al. (2012), however, indicated that DWM does not generally affect crop yields. Raising the outlet height may also result in groundwater and nutrients to be transported via other flow pathways, essentially bypassing the tile outlet. For example, higher water tables may promote increased lateral seepage of groundwater and nutrients into surface streams. Drainage water management may also increase the potential of surface runoff and associated nutrient loadings into surface waters. Giliam and Skaggs (1986) predicted with DRAINMOD that DWM would increase surface runoff compared to FD. Drury et al. (2009) also suggested that while DWM may reduce nutrient loads in tile flow, water quality trade-offs must be considered as nutrient loads in surface runoff increased with DWM.

5. Conclusions

Two adjacent tile drain outlets were monitored for 7 years in order to evaluate the effect of DWM on tile discharge, nutrient concentrations and nutrient loads in drainage water. Drainage water management significantly decreased mean annual tile flow, NO$_3$–N load, and DRP load compared to FD by 23.1, 21.3, and 55.9%, respectively. Nitrate-N and DRP concentrations were not significantly affected by DWM; therefore, reductions in NO$_3$–N and DRP loads were primarily due to decreases in tile flow. In general, the impact of DWM on tile flow and nutrient loads in this study was not as large as reported in other studies. Differences between the current study and previous research on the impact of DWM is likely due to several factors including the height to which the outlet was raised, site drainage intensity and soil properties. These results support the use DWM as a BMP for improving water quality in tile drained landscapes across the U.S. Midwest. Similar to many BMPs, the effectiveness of DWM in mitigating nutrient loads will likely vary by field and a site’s potential for implementation of DWM should be evaluated on a case-by-case basis. While DWM may decrease nutrient loads in tile discharge, it may increase nutrient loads in other flow paths (e.g., lateral seepage, surface runoff). Thus, water quality trade-offs need to be investigated to further understand the effectiveness of DWM.

References


