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Long-Term Observations of Nitrogen and Phosphorus Export in Paired-Agricultural Watersheds under Controlled and Conventional Tile Drainage

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Abstract

Controlled tile drainage (CTD) regulates water and nutrient export from tile drainage systems. Observations of the effects of CTD imposed en masse at watershed scales are needed to determine the effect on downstream receptors. A paired-watershed approach was used to evaluate the effect of field-to-field CTD at the watershed scale on fluxes and flow-weighted mean concentrations (FWMCs) of N and P during multiple growing seasons. One watershed (467-ha catchment area) was under CTD management (treatment [CTD] watershed); the other (250-ha catchment area) had freely draining or uncontrolled tile drainage (UCTD) (reference [UCTD] watershed). The paired agricultural watersheds are located in eastern Ontario, Canada. Analysis of covariance and paired *t* tests were used to assess daily fluxes and FWMCs during a calibration period when CTD intervention on the treatment watershed was minimal (2005–2006, when only 4–10% of the tile-drained area was under CTD) and a treatment period when the treatment (CTD) watershed had prolific CTD intervention (2007–2011 when 82% of tile drained fields were controlled, occupying >70% of catchment area). Significant linear regression slope changes assessed using ANCOVA ($p \leq 0.1$) for daily fluxes from upstream and downstream monitoring sites pooled by calibration and treatment period were −0.06 and −0.20 (stream water) (negative values represent flux declines in CTD watershed), −0.59 and −0.77 (NH_4^+–N), −0.14 and −0.15 (NO_3^-–N), −1.77 and −2.10 (dissolved reactive P), and −0.28 and 0.45 (total P). Total P results for one site comparison contrasted with other findings likely due to unknown in-stream processes affecting total P loading, not efficacy of CTD. The FWMC results were mixed and inconclusive but suggest physical abatement by CTD is the means by which nutrient fluxes are predominantly reduced at these scales. Overall, our study results indicate that CTD is an effective practice for reducing watershed scale fluxes of stream water, N, and P during the growing season.

Core Ideas

- Paired watershed study evaluates conventional and control drainage impacts at watershed scale.
- Controlled tile drainage imposed en masse at watershed scales reduces stream, nitrate, ammonium, and dissolved reactive P fluxes during the growing season.
- Total P fluxes in stream were shown to reduce and increase depending on location in watershed.

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Nutrients derived from agricultural activities can affect the environment well downstream of specific pollution sources within a watershed. For example, nitrogen (N) and phosphorus (P) from agricultural sources have contributed to water quality degradation in the Gulf of St. Lawrence, the Great Lakes (Fausey et al., 1995; Neilsen et al., 1980; State of the St. Lawrence Monitoring Committee, 2008; Bridgeman et al., 2013), and other surface waters throughout North America (Magnien et al., 1995; Skaggs et al., 1994; Schindler et al., 2012). In fact, one of the world's largest hypoxia dead zones exists in the Gulf of Mexico largely as a result of excessive nutrient inputs from agricultural activity in the Mississippi River basin (Rabalais et al., 2002).

Artificial subsurface (tile) drainage is used to improve field drainage for crop production. Tile drainage is critical in many crop production landscapes, and its importance is reflected by its ubiquity. For instance, in Ontario, Canada, we estimated that over 1.6 million ha of agricultural land is artificially (tile) drained. In the midwestern United States, it is estimated that 17.4 million ha of land is artificially drained (Jaynes and Isenhart, 2014). Tile drains can be efficient pathways by which contaminants from fields can enter the broader surface water environment (Gilliam et al., 1979; Kladivko et al., 1991; Drury et al., 1996; Gentry et al., 1998; Geohring et al., 1999; Jaynes et al., 1999; Lapen et al., 2008; Frey et al., 2012). Tile drainage in the midwestern United States is considered one of the largest contributors of N to the Gulf of Mexico (David et al., 2010), and as such, tile drainage water management is considered one of the most promising beneficial management practices (BMPs) to reduce nutrient loads into surface water (Evans et al., 1995; Wesström and Messing, 2007; Skaggs et al., 2012).

Controlled tile drainage (CTD) is one drainage water management practice that could significantly reduce water quality impairment at field and watershed scales (Evans et al.,

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Abbreviations: AVS, area velocity sensor; BMP, beneficial management practice; CTD, controlled tile drainage; DRP, dissolved reactive phosphorus; FWMC, flow-weighted mean concentration; UCTD, uncontrolled tile drainage.

1992; Lalonde et al., 1996; Jaynes et al., 2010; Skaggs et al., 2012; Ghane et al., 2012; Cooke and Verma, 2012; Sunohara et al., 2014; Wilkes et al., 2014). Controlled tile drainage restricts the amount of tile drainage water that can leave a drain outlet (Gilliam et al., 1979; Skaggs et al., 2010). This practice can reduce water and nutrient losses from fields and can elevate water tables, thereby allowing crops to more readily access water and nutrients during critical crop growth stages.

Meeting larger-scale water quality and quantity targets and identification of potential mitigating factors (e.g., BMPs) can be achieved by using information gleaned from watershed-scale investigations that integrate multiple landscape factors not captured at the field and plot scales (Easton et al., 2008; King et al., 2008; Doody et al., 2012). These factors include, but are not limited to, the influence of variable and mixed land use activities (El-Khoury et al., 2015); multiple inputs of different fecal material (Wilkes et al., 2014); in-stream contribution and abatement processes (Ranalli and Macalady, 2010; Sunohara et al., 2012); and larger-scale, groundwater-surface water interactions (Heathwaite et al., 2000). Studies have documented the potential BMP impact on surface water quality for a vast range of watersheds under a variety of point and nonpoint source pollution pressures (Mulla et al., 2008). Paired watershed designs have been used to investigate the impacts of N management (Koerkle et al., 1997), P management (Bishop et al., 2005), tillage (Clausen et al., 1996), forest management (Wynn et al., 2000), prairie restoration (Schilling, 2002), agroforestry practices (Udawatta et al., 2002), and riparian restoration (Meals, 2001). There are perhaps more studies that focus on model outputs that predict the sensitivity of BMPs on watershed water quality (e.g., Ma et al., 2007; Ale et al., 2012; Rao et al., 2009; Liu et al., 2011; Salazar et al., 2013). Ideally, these kinds of modeling studies are informed by longer-term watershed observational data on BMP performance to help assess how realistic modeling outputs are under a true landscape situation.

It was the purpose of this research to determine if controlled tile drainage imposed en masse on a field-to-field basis in an experimental paired watershed setting in eastern Ontario, Canada, influences stream water, N, and P mass fluxes for time periods from planting to harvest (a time period during the year when producers in the study area who use CTD desire to exclusively impose the practice). To date, there is little research at the watershed scale on this topic because CTD is neither ubiquitous nor densely used in many tile-drained regions in the world and because such interventions from an experimental standpoint can be logistically complex and expensive. However, given that CTD is expected to be an important BMP in meeting watershed water quality targets in the future (ADMSTF, 2012), such studies are important for understanding nutrient loading processes, management constraints, uncertainty, and how models might be used or modified to accurately reflect the influence of CTD at such scales.

This study compared stream water, N, and P fluxes as well as flow-weighted mean concentrations over the growing season from two (paired) experimental watersheds. The treatment (CTD) watershed was dominated by field-to-field CTD during a treatment period (and little CTD impact during a calibration period), whereas the reference watershed was under free or uncontrolled tile drainage (UCTD) throughout the study. We expected to find significant reductions in the export of water, N, and P for the treatment (CTD) watershed relative to the reference (UCTD) watershed during the treatment period of study.

Materials and Methods

Study Area and General Watershed Characteristics

This multiple-year study was conducted from 2005 to 2011 (7 yr) during the cropping season (defined here as May–Oct.). The experimental site is located in the South Nation River basin in eastern Ontario, Canada (Fig. 1). Thirty-year normal annual precipitation (1981–2010) in the area is 981 mm, and average daily air temperature is 6.5°C (Environment Canada, 2014). Average 30-yr precipitation and mean 30-yr air temperatures during the growing season (May–Oct.) are 536 mm and 16°C, respectively (Environment Canada, 2014) (Supplemental Tables S1 and S2).

Two adjacent watersheds of comparable land use, physiography, and tile drainage configurations were selected for this study (Fig. 1). The reference watershed was placed under UCTD, and the treatment watershed was placed under CTD. Total surface catchment areas upstream of the downstream monitoring sites CTD_D and $UCTD_D$ are 467 and 250 ha, respectively. Both streams (drainage ditches) were dredged decades ago to collect subsurface drainage from adjacent fields. Tile-drained fields currently occupy 421 ha (89%) and 220 ha (85%) of the treatment (CTD) and reference (UCTD) watersheds, respectively. Tile systems were gradually installed over the last 40+ years, and tile drainage is the dominant flow process that brings field water to these surface water systems. Surface runoff is minimal and localized in these watersheds as a result of flat topography (slopes are generally below 1%) and ditch berms. The dominant soil type is the Bainsville silt loam (Canadian Soil Classification: Orthic Humic Gleysol; U.S. Soil Taxonomy equivalents: Aquolls, Humaquepts) (Wicklund and Richards, 1962). See Sunohara et al. (2014) for more detail on specific soil properties and groundwater flow dynamics.

Controlled Tile Drainage

The existing tile drainage systems are dominated by parallel lateral subsurface drains (perforated corrugated plastic pipe, 102 mm i.d.) spaced 15 to 17 m apart and installed perpendicular to drainage ditch fronts at an average depth of ~1 m (near outlet end). Tile grade is approximately 0.1%. The lateral drains are connected to a collector or header drain (perforated corrugated plastic, 152 mm i.d.) generally located parallel to adjacent drainage ditches into which the header drains. Starting in 2005, in-line water level control structures (Agri Drain Corp.) were retrofitted on existing header drain outlets (Fig. 2). These control structures provide an adjustable-height weirboard (stoplog) system that restricts drainage when the water level in the control structure drops below the stoplog elevation but permits drainage when water levels are above stoplog elevation. Water overflow depth (e.g., Fig. 2) in the structures for CTD fields was set at a depth of ~0.6 m below the mean soil surface around time of planting and left controlled until about harvest (Cicek et al., 2010; Sunohara et al., 2014). This depth has been shown to augment root–water table interaction (American Society of

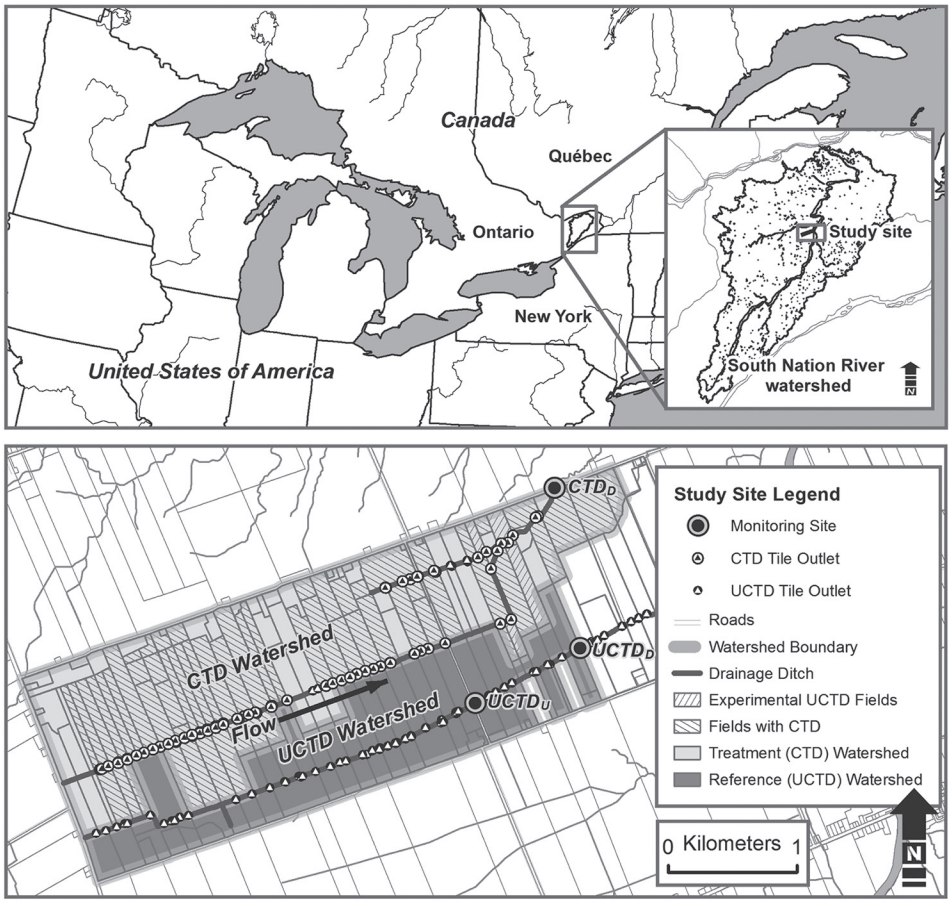

Fig. 1. Map of experimental paired watersheds and location within South Nation River basin in eastern Ontario, Canada. The spatial extent of controlled tile drainage (CTD) fields in the treatment (CTD) watershed represents years 2007 to 2009. UCTD, uncontrolled tile drainage.

Agricultural and Biological Engineers, 1990) and in this case was also deemed to help minimize surface water ponding and overland flow potential. For fields within the reference (UCTD) watershed, no flow control was imposed, and tile drainage was therefore left to flow freely throughout the study. The researchers of this study managed all the drainage systems (controlled and uncontrolled) in the watersheds to ensure they were working in coherence with study objectives.

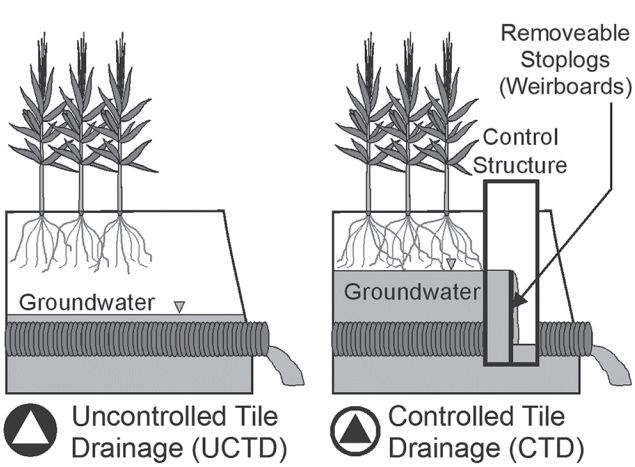

Fig. 2. Schematic of uncontrolled tile drainage (UCTD) and controlled tile drainage (CTD) (during a water overflow situation) with an in-line water level control structure. When water levels are below the elevation of the top stoplog, no tile flow occurs.

To comply with the requirements of the producers and land owners, tile drainage control was used only during the agronomic or growing season (roughly planting to harvest time; approximately May–Oct.), and not during the nongrowing season. Some issues and perceptions that were formative for this decision included: (i) the potential for deleterious effects of impeded drainage over the off season on forage or winter wheat production (Hayhoe et al., 2003; Bélanger et al., 2006); (ii) the potential to increase surficial flow processes during winter months (and mobilization of P via high relative saturations of surface soils), which can greatly increase P and sediment loads to streams (Ball Coelho et al., 2012); (iii) the potential to promote ice sheeting and shallower and longer-lasting frost, thereby reducing trafficability and soil warming time periods in spring (Ontario Ministry of Food and Rural Affairs, 1991; Geohring and Steenhuis, 1987); (iv) the potential for control structures to ice up internally, which could result in flow impedance and/or difficulties making manual adjustments of the flow control structure (Frankenberger et al., 2006); and (v) producer requirements for free drainage during time periods in the spring and fall for field trafficking purposes. From an experimental perspective, there were also significant logistical difficulties in accurately monitoring flow and drainage processes in winter due to ice formation in the drainage ditches examined (Wilkes et al., 2011).

As a result of increased producer and land owner adoption of controlled tile drainage over time, the percentage of tile-drained

area under controlled tile drainage in the treatment (CTD) watershed increased abruptly from 4% in 2006 to 82% in the spring of 2007. The impetus to adopt was multifactorial (Dring et al., 2015). Even though the research program offered to cover the costs for the purchase and installation of control structures to land owners, a major incentive was documented yield boosts in 2005–2006 that was communicated to other producers and land owners in the watershed. This impetus provided the basis for the study to compare calibration period (pre-CTD intervention) water quality and quantity results with treatment (CTD intervention) period results.

Land Management Practices

The land in both watersheds is used for agriculture, with comparable land use attributes consisting of tile-drained fields (~10-ha size with existing drainage systems dictating that control structures manage ~5 ha) where crops of corn, soybean, and forage are grown in rotation (Supplemental Fig. S1). Most farms are associated with total confinement dairy systems of size generally around 100 to 200 ha. Cash cropping farms are of smaller size. As of 2012, field crops and forage occupied around 92 and 95% of the CTD and UCTD watersheds, respectively. Tillage practices typically consist of fall moldboard plowing and spring cultivation using chisel-style implements. This form of tillage management is conventional for row crop agriculture in this region of Canada (Ontario Ministry of Food and Rural Affairs, 2011). Generally, crops are planted in early to late May and harvested between late September and early November. Mineral fertilizer use in the watershed generally consisted of broadcast application of granular urea before planting and a granular starter application. Nominal fertilizer rates were ~170 kg N ha^{-1} and ~50 kg P ha^{-1} for corn and ~10 kg N ha^{-1} and ~60 kg P ha^{-1} for soybean (Sunohara et al., 2014; Ontario Ministry of Food and Rural Affairs, 2009). Liquid dairy manure applications (often at rates of ~70,000 L ha^{-1}) occur in the spring and fall on many, but not all, corn, soybean, and newly seeded forage fields over both watersheds (Supplemental Table S3 shows a proportion of each watershed where manure applications occurred at least once during study period). In some cases, forage fields receive an application of manure after first cut. There were no documented land applications of manure in winter. Milk houses and barns associated with dairy operations are located along the roads that bound the two watersheds to the north and south, so direct farm yard runoff and drainage into the ditches from these dairy operations does not occur (Schmidt et al., 2013; Sunohara et al., 2014). A small hobby farm adjacent (southwest) to CTD_D (Fig. 1) consisted of penned animals (goats, donkeys, and horses) of low and variable seasonal intensity (Wilkes et al., 2013).

Paired Watershed Design

Paired watershed approaches are well documented (e.g., Clausen and Spooner, 1993; Bishop et al., 2005; Frankenberger et al., 2005). Three monitoring sites were used to quantify stream (ditch) discharge and to sample and measure water physical and chemical properties. These sites were located at a downstream location of the treatment (CTD) watershed (CTD_D) and at upstream ($UCTD_U$) and downstream ($UCTD_D$) locations within the reference (UCTD) watershed (Fig. 1). Comparisons were made among treatment site CTD_D and reference sites $UCTD_D$ and $UCTD_U$ (the latter as a form of replication).

We also examined periods of pre-CTD intervention outputs (which we refer to as a *calibration period*) in relation to CTD treatment period outputs to help assess CTD intervention effects on water quality and quantity. The calibration period was during the years 2005 and 2006 when only a few fields were actively under CTD (~4–10% of the tile-drained area) in the treatment (CTD) watershed. As of 2007, ~82% of the tile-drained area in the treatment (CTD) watershed was actively under CTD; therefore, the years 2007 to 2011 served as a treatment (CTD intervention) period.

Stream Monitoring

At each monitoring site, continuous stage and velocity measurements at a 15-min temporal resolution were collected using ISCO 4150 area velocity flow modules (Teledyne Isco, Inc.) and low-profile area velocity sensors (AVSs) installed directly in circular road culverts. An open channel area × velocity calculation was used to determine stream discharge (Grant and Dawson, 2001). Discharge estimated with the AVSs was compared with discharge calculated from weekly stage and Flow Probe (Global Water Instrumentation) velocity measurements. Linear regressions of discharge were determined with the AVS and the flow probe resulted in a slopes of ~1.0 and intercepts of ±0.001 m^3 s^{-1} (r^2 ~ 0.99) (Sunohara et al., 2012). Stage–discharge rating curves developed from the AVS data were used to predict discharge during periods when velocity data were unavailable.

Water samples were collected on a routine basis and during storm flow events from typically ice-free conditions from May to November of each year. Water sampling was performed using an ISCO 6712 portable sampler (Teledyne Isco, Inc.). Hardware cloth–protected vinyl sampler suction lines with polypropylene strainers were affixed to rebar posts in the existing culverts at a minimum depth from bottom of 0.3 m to prevent collection of bottom sediment and stream detritus. The strainers were routinely cleaned. A composite water sample (4 × 250-mL samples per 1-L bottle sampled every 2 h) was collected twice a week. Storm events were sampled more intensively. Samplers were initiated when a prescribed amount of rainfall (5–15 mm, depending on season and antecedent stream and soil conditions) was recorded via an ISCO 674 rain gauge set in an open area adjacent to the stream. Because nutrient loads can be substantial during initial stages of a flow event (Ball Coelho et al., 2010), samples were collected with decreasing frequency over time (i.e., 4 × 250 mL every 15 min for the first four 1-L bottles, 30 min for the next four bottles, 1 h for the next four bottles, 3 h for the next four bottles, and 6 h thereafter to fill the remaining 1-L bottles held in each sampler). Water samples were taken to the laboratory in coolers packed with ice within 24 h of sampling. If necessary, samples were frozen at −20°C until analysis. Thirty-minute meteorological data (e.g., precipitation and air temperature) were recorded at a HOBO weather station (Onset Computer Corp.) centrally located within the study watersheds.

Stream water was monitored continuously at each site using YSI 6600 multi-parameter sondes (YSI Inc.). Sondes were set with sensors measuring (range) turbidity (0–1000 NTU), specific conductivity (0–100 dS m^{-1}), oxidation–reduction

potential (−999 to 999 mV), pH (0–14 units), dissolved oxygen (0–50 mg L^{-1}), and temperature (−5 to 45°C). The sondes were set to log readings at a 15-min temporal resolution and were calibrated on a biweekly basis throughout the monitoring season. Sondes data were classified on the basis of percentiles of stream discharge measurements obtained for each season, where "no flow" was defined as discharge <0.002 m^3 s^{-1}, "low flow" was defined as discharge ≥0.002 m^3 s^{-1} and less than the 25th percentile (0.006 and 0.005 m^3 s^{-1} for CTD_D and $UCTD_D$, respectively), "medium flow" was defined as discharge greater than or equal to the 25th and less than the 75th percentiles (0.019 and 0.015 m^3 s^{-1} for CTD_D and $UCTD_D$, respectively), and "high flow" was defined as discharge greater than or equal to the 75th percentile (Sunohara et al., 2012).

N and P Measurements

For water samples, the NO_2^-–N + NO_3^-–N (referred to as NO_3^-–N throughout the text because NO_2^-–N levels are typically negligible) and NH_3–N + NH_4^+–N (referred to as NH_4^+–N throughout text because NH_3–N levels are also typically negligible) were analyzed colorimetrically with a TrAAcs 800 autoanalyzer (Bran+Luebbe) using the cadmium reduction method for NO_2^-–N + NO_3^-–N (standard method 4500−NO_3^-I) and the phenate method for NH_4^+–N (standard method 4500–NH_3H) (American Public Health Association, 2005). Standard samples with known concentrations were analyzed every 20 samples during environmental sample analysis sequences. Dissolved reactive P (DRP) was analyzed with a QuikChem FIA+8000 series autoanalyzer (Lachat Instruments) using the ascorbic acid method (standard method 4500−P G), and total P was analyzed with the Smart Spectro spectrophotometer (LaMotte Co.) using the same method preceded by a persulfate digestion. Blank and standard samples with known concentrations were routinely analyzed during environmental sample analysis sequences. More details on analysis methods can be found in American Public Health Association (2005). Water samples were not analyzed for DRP in 2005.

Statistical Analyses

Paired watershed statistical analyses (USEPA, 1993; USDA-NRCS, 2003) were used to evaluate the overall effect of CTD on mass fluxes and flow-weighted mean concentrations (FWMCs) during the growing season. Typical water quality data transformation functions (Ponce, 1980) did not improve the normality or linearity of the one-to-one relationships of the data, so untransformed data were used in the analysis (e.g., Adeuya et al., 2012). All statistical analyses were performed using Statistica 10 (StatSoft Inc.).

Mass fluxes were calculated as: volume (m^3 of water) or mass (kg of N or P)/watershed contributing area (ha)/time (daily or entire growing season). Flow-weighted mean concentrations were calculated as: mass load (kg of N or P)/total stream discharge (m^3). Due to the temporal density of the sampling, we choose to linearly interpolate concentrations between measured data points to develop daily datasets (USDA-NRCS, 2003; Kladivko et al., 2004; Strock et al., 2004; Gentry et al., 2007). The data were also pooled on a temporal basis by calibration period (2005–2006) and CTD treatment period (2007–2011). Yearly growing season data for the treatment period were also compared with calibration period data. Mean differences in paired daily and seasonal watershed fluxes and FWMCs were examined using paired *t* tests (two-tailed). Percentage change in mean fluxes and FWMCs between watersheds was calculated as: Change(%) = [(CTD-UCTD)/UCTD] × 100. Percentage point difference was calculated as: $\text{Change}(\%)_{\text{calibration period}} - \text{Change}(\%)_{\text{treatment period}}$. Due to the temporal nature of the data, statistical analyses were performed on the interpolated datasets (Kladivko et al., 2004; Strock et al., 2004). We highlighted significance at the 0.1 level.

Analysis of covariance (ANCOVA) was used to determine the impact of controlled drainage on water quantity and quality parameters by examining changes in one-to-one relationships between treatment (CTD) and reference (UCTD) watershed outputs for calibration and treatment periods. Specifically, linear regressions between the reference (UCTD) and treatment (CTD) watersheds were calculated for parameters of interest during both the calibration and treatment periods, and then the level of significance of slope and intercept differences between those regressions determined with ANCOVA (USEPA, 1993; USDA-NRCS, 2003; Bishop et al., 2005).

Results

Climate, Hydrograph, and Multiple-Parameter Sonde Trends

Figure 3 provides daily rainfall data at the study site for 2005 to 2011. Monthly and long-term (1981–2010) average precipitation and temperature at a nearby monitoring station are shown in Supplemental Tables S1 and S2. Mean monthly temperatures were comparable to the monthly normals in all 7 yr of study (Supplemental Table S3). Total seasonal (May–Oct.) rainfall was greater than the 30-yr normals for all years except 2007 and 2011.

During the calibration period, discharge peaks were somewhat greater in the treatment (CTD) watershed relative to the reference (UCTD) watershed. During the treatment period, the stream hydrographs for the reference (UCTD) watershed showed somewhat greater discharge lags on the recession limbs and overall higher peak flows relative to the treatment (CTD) watershed.

For the CTD_D and $UCTD_D$ sites, multi-parameter sondes indicated that there were relatively lower oxygen concentrations and higher turbidity under lower-flow conditions in relation to higher-flow conditions (Supplemental Table S4). Specific conductivity and pH, however, were more consistent among flow classes and years.

Stream Water Mass Fluxes

During the calibration period, mean growing season stream water mass fluxes for the CTD_D, $UCTD_D$, and $UCTD_U$ sites were $8.80 \times 10^2 \pm$ SD 1.24×10^2, $9.25 \times 10^2 \pm 3.42 \times 10^2$, and $9.16 \times 10^2 \pm 3.61 \times 10^2$ m^3 ha^{-1} $season^{-1}$, respectively (Table 1; Fig. 4). During the treatment period, when CTD intervention in the treatment (CTD) watershed was more prolific, mean growing season stream water mass fluxes were $4.02 \times 10^2 \pm 2.98 \times 10^2$, $5.04 \times 10^2 \pm 3.07 \times 10^2$, and $5.14 \times 10^2 \pm 3.02 \times 10^2$ m^3 ha^{-1}

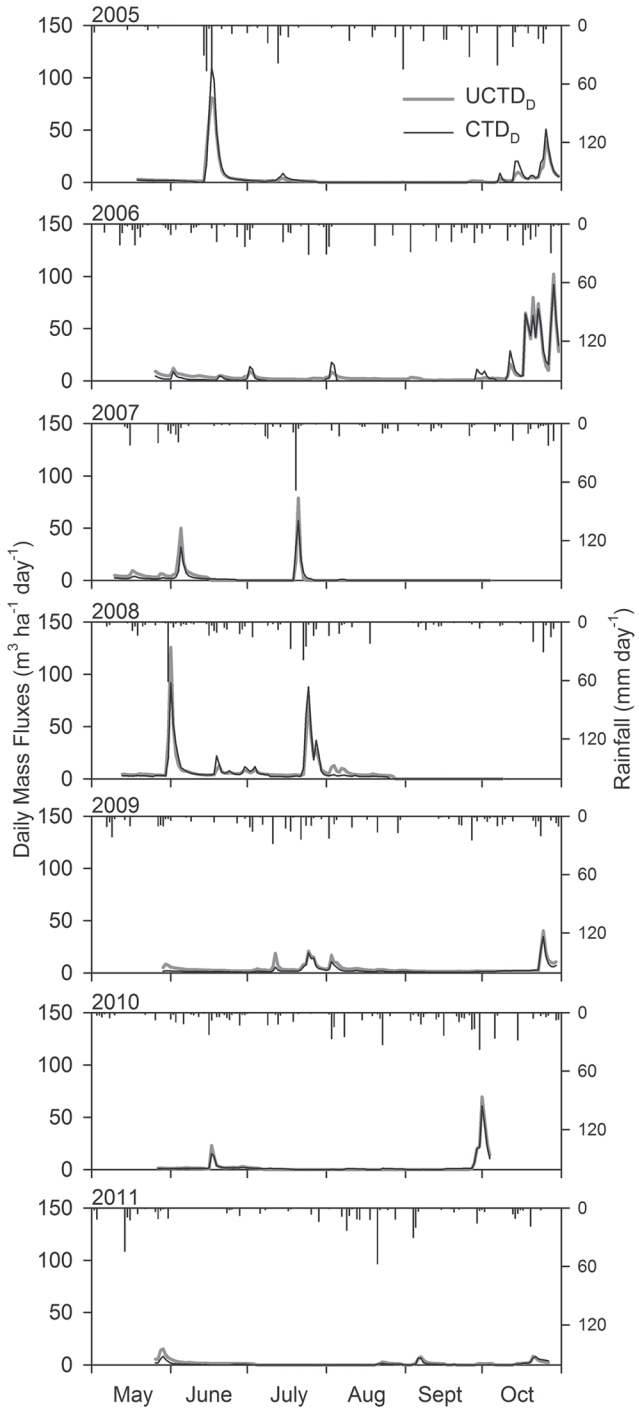

Fig. 3. Growing season daily mass fluxes for downstream controlled tile drainage (CTD_D) and downstream uncontrolled tile drainage ($UCTD_D$) sites for years 2005 to 2006 (calibration period) and 2007 to 2011 (treatment period). Daily rainfall (mm) is superimposed at top of each graph.

season^{-1} for the CTD_D, $UCTD_D$, and $UCTD_U$ sites, respectively. There were no significant differences in growing season stream water mass fluxes among the treatment (CTD) and reference (UCTD) watersheds during the calibration period, but there were significant differences during the treatment period in favor of lower fluxes in the treatment (CTD) watershed (Table 1).

No significant differences in pooled growing season daily stream water mass fluxes were observed during the calibration period (Table 2). Pooled by treatment period, however, there were significantly ($p \leq 0.1$) lower growing season daily stream water mass fluxes for the treatment (CTD) watershed, which supports a CTD treatment effect (Table 2; Fig. 4). These findings are reinforced by the ANCOVA analyses where significant regression slope changes, in favor of CTD intervention effects, were documented (Fig. 5; Supplemental Table S11). The slope changes (treatment − calibration period) were −0.06 and −0.20, for the two monitoring site comparisons (Fig. 5; Supplemental Table S11). On a growing season-to-growing season and site-to-site basis for the calibration period, two of four *t* test comparisons indicated significantly lower daily stream water fluxes for the treatment (CTD) watershed, and the other two comparisons were in the reverse direction (Table 2). However, for the treatment period, all growing season-to-growing season and site-to-site comparisons displayed lower daily stream water mass fluxes for the treatment (CTD) watershed (6 of 10 comparisons with $p \leq 0.1$). The combined results of the total growing season fluxes, daily mass fluxes, and ANCOVAs indicate an overall CTD effect (flow reductions) on stream water fluxes.

NH_4^+–N and NO_3^-–N Mass Fluxes and Flow-Weighted Mean Concentrations

No significant differences were found in growing season NH_4^+–N mass fluxes and FWMCs among the two watersheds for the calibration and treatment periods of study (Table 1; Supplemental Table S6). However, for pooled calibration and treatment period daily NH_4^+–N fluxes, there were higher fluxes for the treatment (CTD) watershed during the calibration period (although not significant at the 0.1 probability level) and significantly lower daily fluxes in the treatment (CTD) watershed during the treatment period (Table 3). During the treatment period, pooled growing season daily NH_4^+–N FWMCs were also significantly lower in the treatment (CTD) watershed in relation to the reference (UCTD) watershed for CTD_D vs. $UCTD_D$; other pooled comparisons were not significant (Supplemental Table S7). The CTD intervention effect on reductions in NH_4^+–N fluxes is supported by significant ANCOVA results (Fig. 5; Supplemental Table S11) where slope changes (treatment − calibration period) were −0.59 and −0.77 for the two site comparisons (Fig. 5; Supplemental Table S11).

For the calibration period, there were no significant differences in growing season NO_3^-–N mass fluxes among the two watersheds; however, for the treatment period, there were significantly lower growing season mass fluxes in the treatment (CTD) watershed (Table 1). There were no significant differences in calibration and treatment period growing season FWMCs among the watersheds (Supplemental Table S6).

For pooled daily NO_3^-–N fluxes, there were no significant differences in fluxes among the two watersheds during the calibration period, but there were statistically lower pooled daily fluxes in the treatment (CTD) watershed relative to the reference (UCTD) watershed during the treatment period (Table 4). For pooled daily NO_3^-–N FWMCs for the calibration period, both site comparisons were significant in favor of higher concentrations in the treatment (CTD) watershed; however, pooled daily NO_3^-–N FWMCs were insignificant during the treatment period (Supplemental Table S8). The ANCOVA results were significant in supporting a reduction of daily

Table 1. Mean growing season (yearly study period) mass fluxes and paired *t* test results from downstream controlled tile drainage, downstream uncontrolled tile drainage, and upstream uncontrolled tile drainage sites for the calibration (2005–2006) and treatment periods (2007–2011).

Parameter	Period†	Site pair‡	CTD§ (mean ± SD)	UCTD¶ (mean ± SD)	CTD – UCTD (difference ± SD)	*n*	*P* value	Change
			$m^3\ ha^{-1}$ study $period^{-1}$					%
Stream water	CAL	CTD_D vs. $UCTD_D$	$8.80 \times 10^2 \pm 1.24 \times 10^2$	$9.25 \times 10^2 \pm 3.42 \times 10^2$	$-4.48 \times 10^1 \pm 2.18 \times 10^2$	2	0.820	−5
		CTD_D vs. $UCTD_U$	$8.80 \times 10^2 \pm 1.24 \times 10^2$	$9.16 \times 10^2 \pm 3.61 \times 10^2$	$-3.59 \times 10^1 \pm 2.37 \times 10^2$	2	0.865	−4
	TRT	CTD_D vs. $UCTD_D$	$4.02 \times 10^2 \pm 2.98 \times 10^2$	$5.04 \times 10^2 \pm 3.07 \times 10^2$	$-1.01 \times 10^2 \pm 7.90 \times 10^1$	5	**0.046#**	−20
		CTD_D vs. $UCTD_U$	$4.02 \times 10^2 \pm 2.98 \times 10^2$	$5.14 \times 10^2 \pm 3.02 \times 10^2$	$-1.11 \times 10^2 \pm 7.76 \times 10^1$	5	**0.033**	−22
			kg ha^{-1} study $period^{-1}$					
NH_4^+–N	CAL	CTD_D vs. $UCTD_D$	$3.43 \times 10^{-2} \pm 2.30 \times 10^{-2}$	$2.94 \times 10^{-2} \pm 2.77 \times 10^{-2}$	$4.93 \times 10^{-3} \pm 4.67 \times 10^{-3}$	2	0.376	17
		CTD_D vs. $UCTD_U$	$3.43 \times 10^{-2} \pm 2.30 \times 10^{-2}$	$3.19 \times 10^{-2} \pm 9.19 \times 10^{-3}$	$2.40 \times 10^{-3} \pm 1.38 \times 10^{-2}$	2	0.847	8
	TRT	CTD_D vs. $UCTD_D$	$4.50 \times 10^{-2} \pm 3.56 \times 10^{-2}$	$9.56 \times 10^{-2} \pm 1.12 \times 10^{-1}$	$-5.06 \times 10^{-2} \pm 8.13 \times 10^{-2}$	5	0.236	−53
		CTD_D vs. $UCTD_U$	$4.50 \times 10^{-2} \pm 3.56 \times 10^{-2}$	$7.58 \times 10^{-2} \pm 7.41 \times 10^{-2}$	$-3.08 \times 10^{-2} \pm 4.94 \times 10^{-2}$	5	0.236	−41
NO_3^-–N	CAL	CTD_D vs. $UCTD_D$	$3.30 \pm 5.87 \times 10^{-1}$	$3.38 \pm 4.62 \times 10^{-1}$	$-8.28 \times 10^{-2} \pm 1.05$	2	0.929	−2
		CTD_D vs. $UCTD_U$	$3.30 \pm 5.87 \times 10^{-1}$	$3.51 \pm 3.43 \times 10^{-1}$	$-2.06 \times 10^{-1} \pm 9.30 \times 10^{-1}$	2	0.807	−6
	TRT	CTD_D vs. $UCTD_D$	1.29 ± 1.07	1.73 ± 1.36	$-4.32 \times 10^{-1} \pm 3.54 \times 10^{-1}$	5	**0.053**	−25
		CTD_D vs. $UCTD_U$	1.29 ± 1.07	1.96 ± 1.58	$-6.64 \times 10^{-1} \pm 5.78 \times 10^{-1}$	5	**0.062**	−34
DRP	CAL	CTD_D vs. $UCTD_D$	1.01×10^{-1}	3.28×10^{-2}	6.78×10^{-2}	1	NA††	206
		CTD_D vs. $UCTD_U$	1.01×10^{-1}	9.94×10^{-3}	9.07×10^{-2}	1	NA††	912
	TRT	CTD_D vs. $UCTD_D$	$1.55 \times 10^{-2} \pm 1.08 \times 10^{-2}$	$9.63 \times 10^{-3} \pm 3.93 \times 10^{-3}$	$5.85 \times 10^{-3} \pm 7.34 \times 10^{-3}$	5	0.150	61
		CTD_D vs. $UCTD_U$	$1.55 \times 10^{-2} \pm 1.08 \times 10^{-2}$	$1.31 \times 10^{-2} \pm 4.32 \times 10^{-3}$	$2.33 \times 10^{-3} \pm 7.03 \times 10^{-3}$	5	0.499	18
Total P	CAL	CTD_D vs. $UCTD_D$	$1.01 \times 10^{-1} \pm 1.98 \times 10^{-2}$	$6.55 \times 10^{-2} \pm 1.05 \times 10^{-3}$	$3.54 \times 10^{-2} \pm 2.09 \times 10^{-2}$	2	0.251	54
		CTD_D vs. $UCTD_U$	$1.01 \times 10^{-1} \pm 1.98 \times 10^{-2}$	$1.18 \times 10^{-1} \pm 5.69 \times 10^{-3}$	$-1.66 \times 10^{-2} \pm 1.41 \times 10^{-2}$	2	0.344	−14
	TRT	CTD_D vs. $UCTD_D$	$5.74 \times 10^{-2} \pm 4.13 \times 10^{-2}$	$7.02 \times 10^{-2} \pm 5.61 \times 10^{-2}$	$-1.28 \times 10^{-2} \pm 5.24 \times 10^{-2}$	5	0.615	−18
		CTD_D vs. $UCTD_U$	$5.74 \times 10^{-2} \pm 4.13 \times 10^{-2}$	$5.17 \times 10^{-2} \pm 3.32 \times 10^{-2}$	$5.76 \times 10^{-3} \pm 2.76 \times 10^{-2}$	5	0.665	11

† CAL, calibration; TRT, treatment.

‡ CTD_D, downstream controlled tile drainage; $UCTD_D$, downstream uncontrolled tile drainage; $UCTD_U$, upstream uncontrolled tile drainage.

§ Controlled tile drainage.

¶ Uncontrolled tile drainage.

Bold *P* values are significant at the ≤0.1 probability level.

†† Not applicable due to $n = 1$.

NO_3^-–N fluxes by CTD (Fig. 5; Supplemental Table S11). Significant slope changes in the direction of reduced fluxes for the treatment (CTD) watershed are −0.14 and −0.15, for the two site comparisons (Fig. 5).

Dissolved Reactive P and Total P Mass Fluxes and Flow-Weighted Mean Concentrations

Growing season DRP fluxes were not significantly different for either calibration or treatment period of study (Table 1). However, there were significantly higher growing season DRP FWMCs for the treatment (CTD) watershed during the treatment period (Supplemental Table S6).

Significantly higher pooled daily DRP fluxes were observed in the treatment (CTD) watershed during the calibration period (Table 5). However, for the treatment period, only one site comparison for the pooled data was significant, and it was in favor of higher daily DRP fluxes for the treatment (CTD) watershed; the other site comparison was insignificant (Table 5). Yet, the ANCOVA results suggested flux reductions from the treatment (CTD) watershed during the treatment period (Fig. 5; Supplemental Table S11). For daily FWMCs, ANCOVA results were mixed, depending on site comparisons and year (Supplemental Table S12).

Growing season total P fluxes from the two watersheds were insignificantly different for both calibration and treatment periods of study (Table 1), but there were significantly higher growing season FWMCs in the treatment (CTD) watershed for both monitoring site comparisons during the calibration period (Supplemental Table S6).

Pooled daily total P fluxes at CTD_D were significantly higher than those at $UCTD_D$ during the calibration period (Table 6). During the treatment period, pooled daily total P fluxes were significantly lower at CTD_D in relation to $UCTD_D$ but were significantly higher ($p \leq 0.09$) at CTD_D in relation to $UCTD_U$. For pooled daily total P FWMCs, significantly higher concentrations were observed in the treatment (CTD) watershed for the calibration (for both monitoring site comparisons) and treatment periods (for the upstream comparisons only) (Supplemental Table S10).

Significant ANCOVA results (Fig. 5; Supplemental Table S11) indicate daily total P flux reductions in the treatment (CTD) watershed for the CTD_D vs. $UCTD_D$ comparison and significant increases in the total P fluxes in the treatment (CTD) watershed for the CTD_D vs. $UCTD_U$ comparison. Slope changes (treatment − calibration period) for the former and latter comparisons were −0.28 and 0.45, respectively. For daily total P FWMCs, there was one significant ANCOVA slope result (for the CTDD vs. UCTDD comparison), where the slope shift was −1.02, which is indicative of a reduction of total P FWMC in the treatment (CTD) watershed (Supplemental Table S12).

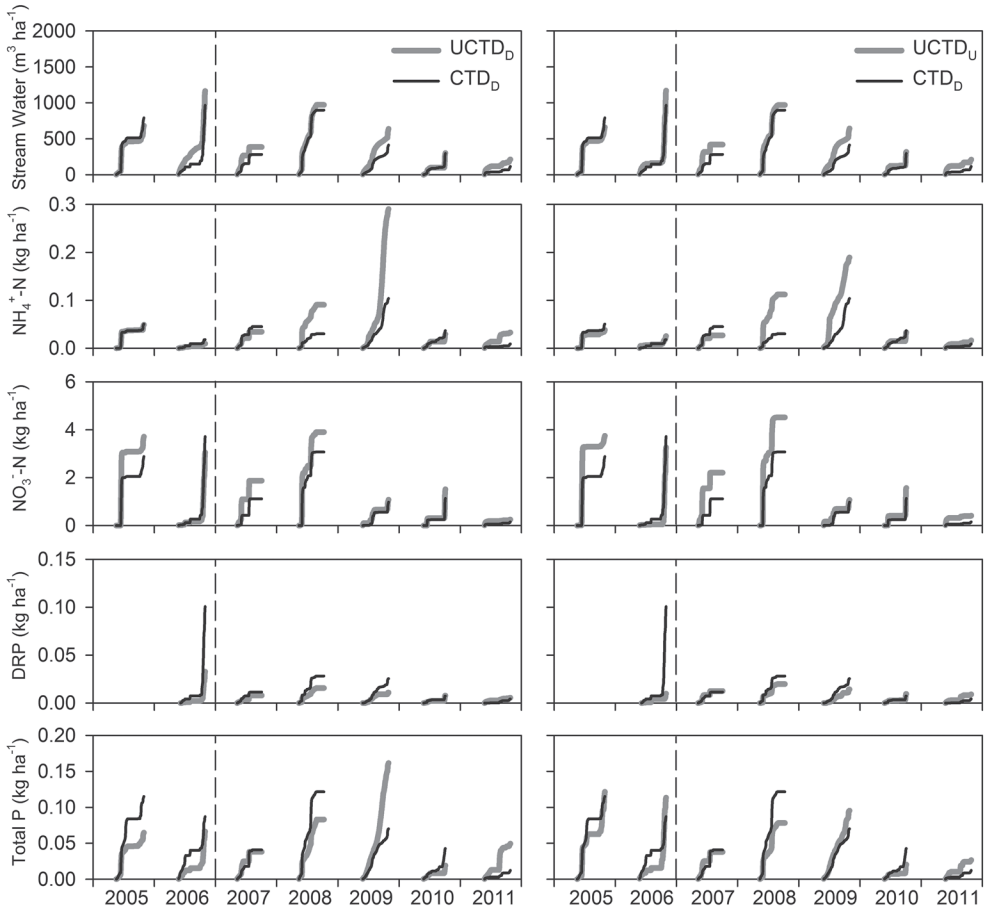

Fig. 4. Cumulative growing season daily mass fluxes for calibration (2005 and 2006) and treatment period years (2007–2011) for downstream controlled tile drainage (CTD_D), upstream uncontrolled tile drainage ($UCTD_U$), and downstream uncontrolled tile drainage ($UCTD_D$) monitoring sites. DRP, dissolved reactive P.

Table 2. Growing season mean daily stream water mass fluxes and paired *t* test results from downstream controlled tile drainage, downstream uncontrolled tile drainage, and upstream uncontrolled tile drainage sites for calibration (2005–2006) and treatment periods (2007–2011).

Year	Site pair†	CTD‡ (mean ± SD)	UCTD§ (mean ± SD)	CTD – UCTD (difference ± SD)	*n*	*P* value	Change
		m^3 ha^{-1} d^{-1}					%
2005	CTD_D vs. $UCTD_D$	4.77 ± 14.07	4.11 ± 11.39	0.66 ± 3.53	166	**0.017¶**	16
	CTD_D vs. $UCTD_U$	4.77 ± 14.07	3.98 ± 11.46	0.79 ± 4.2	166	**0.016**	20
2006	CTD_D vs. $UCTD_D$	6.09 ± 15.27	7.34 ± 15.29	−1.25 ± 3.14	159	**0.001**	−17
	CTD_D vs. $UCTD_U$	6.09 ± 15.27	7.37 ± 21.62	−1.28 ± 7.57	159	**0.035**	−17
2005–2006 (pooled)	CTD_D vs. $UCTD_D$	5.42 ± 14.66	5.69 ± 13.52	−0.28 ± 3.47	325	0.153	−5
	CTD_D vs. $UCTD_U$	5.42 ± 14.66	5.64 ± 17.25	−0.22 ± 6.17	325	0.519	−4
2007	CTD_D vs. $UCTD_D$	1.91 ± 6.32	2.62 ± 8.38	−0.7 ± 3.21	148	**0.008**	−27
	CTD_D vs. $UCTD_U$	1.91 ± 6.32	2.84 ± 8.9	−0.92 ± 4.4	148	**0.012**	−33
2008	CTD_D vs. $UCTD_D$	6 ± 13.25	6.48 ± 13.58	−0.48 ± 4.63	150	0.207	−7
	CTD_D vs. $UCTD_U$	6 ± 13.25	6.48 ± 16.95	−0.47 ± 5.77	150	0.318	−7
2009	CTD_D vs. $UCTD_D$	2.67 ± 4.09	4.13 ± 4.79	−1.46 ± 1.6	155	**≤0.001**	−35
	CTD_D vs. $UCTD_U$	2.67 ± 4.09	4.16 ± 4.99	−1.49 ± 1.87	155	**≤0.001**	−36
2010	CTD_D vs. $UCTD_D$	2.27 ± 7.21	2.34 ± 8.26	−0.07 ± 1.33	132	0.545	−3
	CTD_D vs. $UCTD_U$	2.27 ± 7.21	2.47 ± 8.66	−0.2 ± 2.18	132	0.294	−8
2011	CTD_D vs. $UCTD_D$	0.74 ± 1.61	1.36 ± 2.3	−0.62 ± 1.25	154	**≤0.001**	−45
	CTD_D vs. $UCTD_U$	0.74 ± 1.61	1.34 ± 2.68	−0.6 ± 1.59	154	**≤0.001**	−45
2007–2011 (pooled)	CTD_D vs. $UCTD_D$	2.72 ± 7.74	3.41 ± 8.51	−0.69 ± 2.78	739	**≤0.001**	−20
	CTD_D vs. $UCTD_U$	2.72 ± 7.74	3.47 ± 9.84	−0.75 ± 3.59	739	**≤0.001**	−22

† CTD_D, downstream controlled tile drainage; $UCTD_D$, downstream uncontrolled tile drainage; $UCTD_U$, upstream uncontrolled tile drainage.

‡ Controlled tile drainage.

§ Uncontrolled tile drainage.

¶ Bold *P* values are significant at the ≤0.1 probability level.

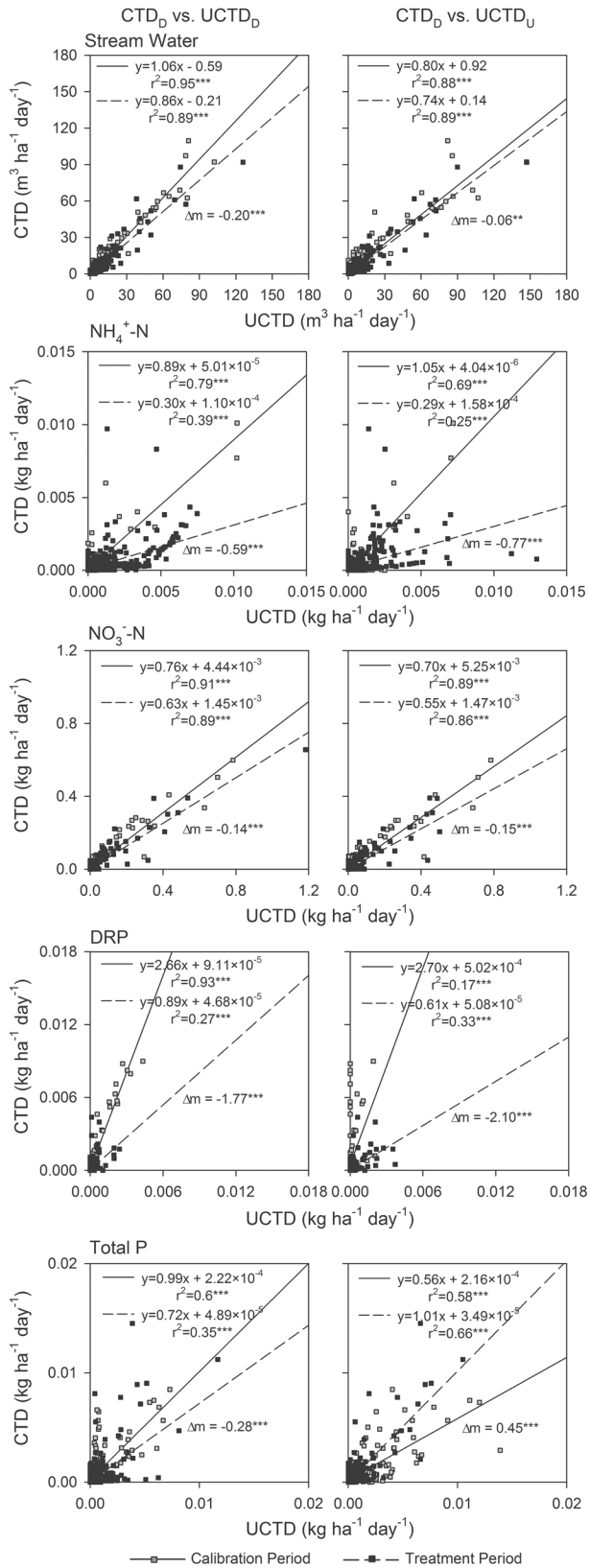

Fig. 5. Scatter plots showing comparisons of daily fluxes of stream water, NH_4^+–N, NO_3^-–N, dissolved reactive P (DRP), and total P for treatment (CTD) vs. reference (UCTD) watersheds for calibration (2005–2006) and treatment (2007–2011) periods. Linear regression models for each period and the significance (* $P < 0.001$) of r^2 and slope differences (Δm) are included. y = CTD fluxes; x = UCTD fluxes.**

Discussion

The experimental dataset described herein provides a means to examine how CTD, when imposed en masse on a field-to-field basis over a watershed during the growing season, might express itself with respect to reducing mass fluxes and concentrations of nutrients in an open system subjected to a variety of physical, chemical, and microbiological in-field and in-stream processes. Producers who have the capacity to use CTD in climatic regions such as where this study was conducted may choose to use CTD only during the production or growing season (Dring et al., 2015), in contrast to many other more temperate regions of North America where CTD is often used during the "fallow" or nonproduction season (late fall to early spring) (Adeuya et al., 2012; Cooke and Verma, 2012; Ghane et al., 2012; Jaynes, 2012). Hence, this study only presents watershed results for a portion of the year for reasons previously mentioned. However, the growing season is a time of year when water and nutrient conservation imposed by CTD could benefit crop production (Cicek et al., 2010; Poole et al., 2011; Delbecq et al., 2012; Ghane et al., 2012; Sunohara et al., 2014).

In this study, tile drainage represents a majority of stream water discharge from the watersheds. The surface water systems (ditches) were artificially produced to collect subsurface drainage from adjacent fields. For the treatment (CTD) watershed, over 75 water flow control structures were installed to manage drainage from ~82% of the tile-drained area in the watershed, representing 71% of the total watershed catchment area. Therefore, we expected to see lower stream water fluxes in the treatment (CTD) watershed.

Adoption of CTD by producers in the experimental watersheds increased abruptly in 2007 largely as a result of positive field-scale benefits observed by adopters, the communication of positive results to nonadopters regarding crop yield boosts (Sunohara et al., 2014), and visual observations made by producers of so-called "taller" and "more healthy looking crops" (Cicek et al., 2010). Environmental benefits were deemed by many producers in general terms to be of secondary consideration relative to on-farm production benefits because there was on average a 3 to 5% yield boost over the years of study at the site (Sunohara et al., 2014; Dring et al., 2015).

The ANCOVA results for daily fluxes described herein demonstrate that CTD imposed en masse at watershed scales can significantly and consistently reduce fluxes of stream water, NO_3^-–N, NH_4^+–N, DRP, and, for one site comparison, total P (90% of site comparisons show daily flux reductions associated with CTD). The ANCOVA results are supported by negative percentage point differences in percentage change from the calibration to the treatment period in mean daily stream water (−17), NH_4^+–N (−60), NO_3^-–N (−26), DRP (−519), and total P (−24) fluxes. The impact of controlled tile drainage on FWMCs was more variable, and many results were insignificant in relation to fluxes. The ANCOVA slope changes among daily FWMC site-to-site comparisons were 0.15 and 0.32 (NH_4^+–N) ($p > 0.1$), −0.02 and −0.03 (NO_3^-–N) ($p > 0.1$), −1.17 and 0.31 (DRP) ($p \leq 0.01$), and −1.02 ($p \leq 0.001$) and −0.07 (total P) [negative values represent reductions in treatment (CTD) watershed FWMCs]. Often, CTD-mitigating effects on nutrient fluxes are equated with the capacity for CTD to physically govern flow

Table 3. Growing season mean daily NH_4^+–N mass fluxes and paired *t* test results from downstream controlled tile drainage, downstream uncontrolled tile drainage, and upstream uncontrolled tile drainage sites for calibration (2005–2006) and treatment periods (2007–2011).

Year	Site pair†	CTD‡ (mean ± SD)	UCTD§ (mean ± SD)	CTD – UCTD (difference ± SD)	*n*	*P* value	Change
		kg ha^{-1} d^{-1}					%
2005	CTD_D vs. $UCTD_D$	$3.05 \times 10^{-4} \pm 1.20 \times 10^{-3}$	$2.95 \times 10^{-4} \pm 1.23 \times 10^{-3}$	$9.77 \times 10^{-6} \pm 4.76 \times 10^{-4}$	166	0.792	3
	CTD_D vs. $UCTD_U$	$3.05 \times 10^{-4} \pm 1.20 \times 10^{-3}$	$2.32 \times 10^{-4} \pm 9.08 \times 10^{-4}$	$7.34 \times 10^{-5} \pm 5.16 \times 10^{-4}$	166	**0.069¶**	32
2006	CTD_D vs. $UCTD_D$	$1.14 \times 10^{-4} \pm 3.47 \times 10^{-4}$	$6.18 \times 10^{-5} \pm 1.60 \times 10^{-4}$	$5.18 \times 10^{-5} \pm 3.45 \times 10^{-4}$	159	**0.061**	84
	CTD_D vs. $UCTD_U$	$1.14 \times 10^{-4} \pm 3.47 \times 10^{-4}$	$1.60 \times 10^{-4} \pm 3.91 \times 10^{-4}$	$-4.64 \times 10^{-5} \pm 4.75 \times 10^{-4}$	159	0.219	−29
2005–2006 (pooled)	CTD_D vs. $UCTD_D$	$2.11 \times 10^{-4} \pm 8.94 \times 10^{-4}$	$1.81 \times 10^{-4} \pm 8.95 \times 10^{-4}$	$3.03 \times 10^{-5} \pm 4.17 \times 10^{-4}$	325	0.191	17
	CTD_D vs. $UCTD_U$	$2.11 \times 10^{-4} \pm 8.94 \times 10^{-4}$	$1.97 \times 10^{-4} \pm 7.04 \times 10^{-4}$	$1.48 \times 10^{-5} \pm 4.99 \times 10^{-4}$	325	0.594	8
2007	CTD_D vs. $UCTD_D$	$3.05 \times 10^{-4} \pm 1.13 \times 10^{-3}$	$2.32 \times 10^{-4} \pm 7.19 \times 10^{-4}$	$7.25 \times 10^{-5} \pm 8.49 \times 10^{-4}$	148	0.301	31
	CTD_D vs. $UCTD_U$	$3.05 \times 10^{-4} \pm 1.13 \times 10^{-3}$	$1.83 \times 10^{-4} \pm 5.41 \times 10^{-4}$	$1.22 \times 10^{-4} \pm 8.85 \times 10^{-4}$	148	**0.096**	67
2008	CTD_D vs. $UCTD_D$	$2.01 \times 10^{-4} \pm 4.76 \times 10^{-4}$	$6.06 \times 10^{-4} \pm 2.00 \times 10^{-3}$	$-4.05 \times 10^{-4} \pm 1.67 \times 10^{-3}$	150	**0.003**	−67
	CTD_D vs. $UCTD_U$	$2.01 \times 10^{-4} \pm 4.76 \times 10^{-4}$	$7.49 \times 10^{-4} \pm 2.05 \times 10^{-3}$	$-5.48 \times 10^{-4} \pm 1.71 \times 10^{-3}$	150	**≤0.001**	−73
2009	CTD_D vs. $UCTD_D$	$6.70 \times 10^{-4} \pm 7.35 \times 10^{-4}$	$1.87 \times 10^{-3} \pm 1.89 \times 10^{-3}$	$-1.20 \times 10^{-3} \pm 1.38 \times 10^{-3}$	155	**≤0.001**	−64
	CTD_D vs. $UCTD_U$	$6.70 \times 10^{-4} \pm 7.35 \times 10^{-4}$	$1.22 \times 10^{-3} \pm 1.32 \times 10^{-3}$	$-5.53 \times 10^{-4} \pm 1.26 \times 10^{-3}$	155	**≤0.001**	−45
2010	CTD_D vs. $UCTD_D$	$2.83 \times 10^{-4} \pm 5.56 \times 10^{-4}$	$2.26 \times 10^{-4} \pm 6.15 \times 10^{-4}$	$5.69 \times 10^{-5} \pm 3.30 \times 10^{-4}$	132	**0.050**	25
	CTD_D vs. $UCTD_U$	$2.83 \times 10^{-4} \pm 5.56 \times 10^{-4}$	$2.54 \times 10^{-4} \pm 8.32 \times 10^{-4}$	$2.91 \times 10^{-5} \pm 3.97 \times 10^{-4}$	132	0.402	11
2011	CTD_D vs. $UCTD_D$	$5.55 \times 10^{-5} \pm 1.27 \times 10^{-4}$	$2.11 \times 10^{-4} \pm 4.48 \times 10^{-4}$	$-1.56 \times 10^{-4} \pm 4.30 \times 10^{-4}$	154	**≤0.001**	−74
	CTD_D vs. $UCTD_U$	$5.55 \times 10^{-5} \pm 1.27 \times 10^{-4}$	$1.06 \times 10^{-4} \pm 1.81 \times 10^{-4}$	$-5.04 \times 10^{-5} \pm 1.17 \times 10^{-4}$	154	**≤0.001**	−48
2007–2011 (pooled)	CTD_D vs. $UCTD_D$	$3.05 \times 10^{-4} \pm 7.17 \times 10^{-4}$	$6.47 \times 10^{-4} \pm 1.48 \times 10^{-3}$	$-3.42 \times 10^{-4} \pm 1.18 \times 10^{-3}$	739	**≤0.001**	−53
	CTD_D vs. $UCTD_U$	$3.05 \times 10^{-4} \pm 7.17 \times 10^{-4}$	$5.13 \times 10^{-4} \pm 1.26 \times 10^{-3}$	$-2.08 \times 10^{-4} \pm 1.09 \times 10^{-3}$	739	**≤0.001**	−41

† CTD_D, downstream controlled tile drainage; $UCTD_D$, downstream uncontrolled tile drainage; $UCTD_U$, upstream uncontrolled tile drainage.

‡ Controlled tile drainage.

§ Uncontrolled tile drainage.

¶ Bold *P* values are significant at the ≤0.1 probability level.

Table 4. Growing season mean daily NO_3^-–N mass fluxes and paired *t* test results from downstream controlled tile drainage, downstream uncontrolled tile drainage, and upstream uncontrolled tile drainage sites for calibration (2005–2006) and treatment periods (2007–2011).

Year	Site pair†	CTD‡ (mean ± SD)	UCTD§ (mean ± S)	CTD – UCTD (difference ± SD)	*n*	*P* value	Change
		kg ha^{-1} d^{-1}					%
2005	CTD_D vs. $UCTD_D$	$1.74 \times 10^{-2} \pm 6.98 \times 10^{-2}$	$2.24 \times 10^{-2} \pm 1.02 \times 10^{-1}$	$-4.97 \times 10^{-3} \pm 3.72 \times 10^{-2}$	166	**0.088¶**	−22
	CTD_D vs. $UCTD_U$	$1.74 \times 10^{-2} \pm 6.98 \times 10^{-2}$	$2.26 \times 10^{-2} \pm 1.06 \times 10^{-1}$	$-5.20 \times 10^{-3} \pm 4.59 \times 10^{-2}$	166	0.146	−23
2006	CTD_D vs. $UCTD_D$	$2.34 \times 10^{-2} \pm 6.57 \times 10^{-2}$	$1.92 \times 10^{-2} \pm 6.28 \times 10^{-2}$	$4.14 \times 10^{-3} \pm 1.46 \times 10^{-2}$	159	**≤0.001**	22
	CTD_D vs. $UCTD_U$	$2.34 \times 10^{-2} \pm 6.57 \times 10^{-2}$	$2.05 \times 10^{-2} \pm 7.36 \times 10^{-2}$	$2.84 \times 10^{-3} \pm 2.01 \times 10^{-2}$	159	**0.077**	14
2005–2006 (pooled)	CTD_D vs. $UCTD_D$	$2.03 \times 10^{-2} \pm 6.78 \times 10^{-2}$	$2.08 \times 10^{-2} \pm 8.47 \times 10^{-2}$	$-5.10 \times 10^{-4} \pm 2.88 \times 10^{-2}$	325	0.750	−2
	CTD_D vs. $UCTD_U$	$2.03 \times 10^{-2} \pm 6.78 \times 10^{-2}$	$2.16 \times 10^{-2} \pm 9.14 \times 10^{-2}$	$-1.27 \times 10^{-3} \pm 3.58 \times 10^{-2}$	325	0.524	−6
2007	CTD_D vs. $UCTD_D$	$7.54 \times 10^{-3} \pm 3.96 \times 10^{-2}$	$1.27 \times 10^{-2} \pm 5.98 \times 10^{-2}$	$-5.13 \times 10^{-3} \pm 2.64 \times 10^{-2}$	148	**0.019**	−41
	CTD_D vs. $UCTD_U$	$7.54 \times 10^{-3} \pm 3.96 \times 10^{-2}$	$1.49 \times 10^{-2} \pm 6.18 \times 10^{-2}$	$-7.38 \times 10^{-3} \pm 3.40 \times 10^{-2}$	148	**0.009**	−49
2008	CTD_D vs. $UCTD_D$	$2.05 \times 10^{-2} \pm 7.24 \times 10^{-2}$	$2.60 \times 10^{-2} \pm 1.12 \times 10^{-1}$	$-5.53 \times 10^{-3} \pm 5.12 \times 10^{-2}$	150	0.188	−21
	CTD_D vs. $UCTD_U$	$2.05 \times 10^{-2} \pm 7.24 \times 10^{-2}$	$3.01 \times 10^{-2} \pm 1.30 \times 10^{-1}$	$-9.61 \times 10^{-3} \pm 6.67 \times 10^{-2}$	150	**0.080**	−32
2009	CTD_D vs. $UCTD_D$	$6.33 \times 10^{-3} \pm 1.89 \times 10^{-2}$	$6.97 \times 10^{-3} \pm 1.95 \times 10^{-2}$	$-6.37 \times 10^{-4} \pm 8.16 \times 10^{-3}$	155	0.333	−9
	CTD_D vs. $UCTD_U$	$6.33 \times 10^{-3} \pm 1.89 \times 10^{-2}$	$6.95 \times 10^{-3} \pm 1.90 \times 10^{-2}$	$-6.17 \times 10^{-4} \pm 1.03 \times 10^{-2}$	155	0.457	−9
2010	CTD_D vs. $UCTD_D$	$8.64 \times 10^{-3} \pm 3.79 \times 10^{-2}$	$1.16 \times 10^{-2} \pm 5.59 \times 10^{-2}$	$-2.95 \times 10^{-3} \pm 1.94 \times 10^{-2}$	132	**0.082**	−25
	CTD_D vs. $UCTD_U$	$8.64 \times 10^{-3} \pm 3.79 \times 10^{-2}$	$1.21 \times 10^{-2} \pm 5.69 \times 10^{-2}$	$-3.42 \times 10^{-3} \pm 2.14 \times 10^{-2}$	132	**0.068**	−28
2011	CTD_D vs. $UCTD_D$	$1.01 \times 10^{-3} \pm 2.49 \times 10^{-3}$	$1.54 \times 10^{-3} \pm 4.40 \times 10^{-3}$	$-5.29 \times 10^{-4} \pm 3.05 \times 10^{-3}$	154	**0.033**	−34
	CTD_D vs. $UCTD_U$	$1.01 \times 10^{-3} \pm 2.49 \times 10^{-3}$	$2.57 \times 10^{-3} \pm 8.21 \times 10^{-3}$	$-1.56 \times 10^{-3} \pm 6.93 \times 10^{-3}$	154	**0.006**	−61
2007–2011 (pooled)	CTD_D vs. $UCTD_D$	$8.75 \times 10^{-3} \pm 4.18 \times 10^{-2}$	$1.17 \times 10^{-2} \pm 6.29 \times 10^{-2}$	$-2.92 \times 10^{-3} \pm 2.75 \times 10^{-2}$	739	**0.004**	−25
	CTD_D vs. $UCTD_U$	$8.75 \times 10^{-3} \pm 4.18 \times 10^{-2}$	$1.32 \times 10^{-2} \pm 7.03 \times 10^{-2}$	$-4.49 \times 10^{-3} \pm 3.54 \times 10^{-2}$	739	**0.001**	−34

† CTD_D, downstream controlled tile drainage; $UCTD_D$, downstream uncontrolled tile drainage; $UCTD_U$, upstream uncontrolled tile drainage.

‡ Controlled tile drainage.

§ Uncontrolled tile drainage.

¶ Bold *P* values are significant at the ≤0.1 probability level.

Table 5. Growing season mean daily dissolved reactive P mass fluxes and paired *t* test results from downstream controlled tile drainage, downstream uncontrolled tile drainage, and upstream uncontrolled tile drainage sites for calibration (2005–2006) and treatment periods (2007–2011).

Year	Site pair†	CTD‡ (mean ± SD)	UCTD§ (mean ± SD)	CTD – UCTD (difference ± SD)	*n*	*P* value	Change
		kg ha^{-1} d^{-1}					%
2005	CTD_D vs. $UCTD_D$	NM¶	NM				
	CTD_D vs. $UCTD_U$	NM	NM				
2006	CTD_D vs. $UCTD_D$	$6.85 \times 10^{-4} \pm 1.85 \times 10^{-3}$	$2.23 \times 10^{-4} \pm 6.73 \times 10^{-4}$	$4.61 \times 10^{-4} \pm 1.21 \times 10^{-3}$	147	**≤0.001#**	206
	CTD_D vs. $UCTD_U$	$6.85 \times 10^{-4} \pm 1.85 \times 10^{-3}$	$6.76 \times 10^{-5} \pm 2.84 \times 10^{-4}$	$6.17 \times 10^{-4} \pm 1.75 \times 10^{-3}$	147	**≤0.001**	912
2005–2006 (pooled)	CTD_D vs. $UCTD_D$	$6.85 \times 10^{-4} \pm 1.85 \times 10^{-3}$	$2.23 \times 10^{-4} \pm 6.73 \times 10^{-4}$	$4.61 \times 10^{-4} \pm 1.21 \times 10^{-3}$	147	**≤0.001**	206
	CTD_D vs. $UCTD_U$	$6.85 \times 10^{-4} \pm 1.85 \times 10^{-3}$	$6.76 \times 10^{-5} \pm 2.84 \times 10^{-4}$	$6.17 \times 10^{-4} \pm 1.75 \times 10^{-3}$	147	**≤0.001**	912
2007	CTD_D vs. $UCTD_D$	$7.85 \times 10^{-5} \pm 2.04 \times 10^{-4}$	$5.42 \times 10^{-5} \pm 2.34 \times 10^{-4}$	$2.42 \times 10^{-5} \pm 9.63 \times 10^{-5}$	148	**0.003**	45
	CTD_D vs. $UCTD_U$	$7.85 \times 10^{-5} \pm 2.04 \times 10^{-4}$	$8.50 \times 10^{-5} \pm 3.96 \times 10^{-4}$	$-6.50 \times 10^{-6} \pm 2.92 \times 10^{-4}$	148	0.787	−8
2008	CTD_D vs. $UCTD_D$	$1.89 \times 10^{-4} \pm 5.77 \times 10^{-4}$	$1.05 \times 10^{-4} \pm 2.30 \times 10^{-4}$	$8.36 \times 10^{-5} \pm 5.16 \times 10^{-4}$	150	**0.049**	80
	CTD_D vs. $UCTD_U$	$1.89 \times 10^{-4} \pm 5.77 \times 10^{-4}$	$1.32 \times 10^{-4} \pm 4.07 \times 10^{-4}$	$5.64 \times 10^{-5} \pm 4.49 \times 10^{-4}$	150	0.126	43
2009	CTD_D vs. $UCTD_D$	$1.65 \times 10^{-4} \pm 2.58 \times 10^{-4}$	$7.07 \times 10^{-5} \pm 1.11 \times 10^{-4}$	$9.46 \times 10^{-5} \pm 2.17 \times 10^{-4}$	155	**≤0.001**	134
	CTD_D vs. $UCTD_U$	$1.65 \times 10^{-4} \pm 2.58 \times 10^{-4}$	$9.33 \times 10^{-5} \pm 1.77 \times 10^{-4}$	$7.21 \times 10^{-5} \pm 1.95 \times 10^{-4}$	155	**≤0.001**	77
2010	CTD_D vs. $UCTD_D$	$5.61 \times 10^{-5} \pm 2.09 \times 10^{-4}$	$5.97 \times 10^{-5} \pm 2.31 \times 10^{-4}$	$-3.53 \times 10^{-6} \pm 1.97 \times 10^{-4}$	132	0.837	−6
	CTD_D vs. $UCTD_U$	$5.61 \times 10^{-5} \pm 2.09 \times 10^{-4}$	$7.32 \times 10^{-5} \pm 2.86 \times 10^{-4}$	$-1.71 \times 10^{-5} \pm 2.49 \times 10^{-4}$	132	0.431	−23
2011	CTD_D vs. $UCTD_D$	$2.89 \times 10^{-5} \pm 6.65 \times 10^{-5}$	$3.59 \times 10^{-5} \pm 5.24 \times 10^{-5}$	$-7.09 \times 10^{-6} \pm 6.56 \times 10^{-5}$	154	0.182	−20
	CTD_D vs. $UCTD_U$	$2.89 \times 10^{-5} \pm 6.65 \times 10^{-5}$	$5.97 \times 10^{-5} \pm 1.69 \times 10^{-4}$	$-3.08 \times 10^{-5} \pm 1.37 \times 10^{-4}$	154	**0.006**	−52
2007–2011 (pooled)	CTD_D vs. $UCTD_D$	$1.05 \times 10^{-4} \pm 3.20 \times 10^{-4}$	$6.52 \times 10^{-5} \pm 1.86 \times 10^{-4}$	$3.96 \times 10^{-5} \pm 2.74 \times 10^{-4}$	739	**≤0.001**	61
	CTD_D vs. $UCTD_U$	$1.05 \times 10^{-4} \pm 3.20 \times 10^{-4}$	$8.90 \times 10^{-5} \pm 3.04 \times 10^{-4}$	$1.58 \times 10^{-5} \pm 2.87 \times 10^{-4}$	739	0.135	18

† CTD_D, downstream controlled tile drainage; $UCTD_D$, downstream uncontrolled tile drainage; $UCTD_U$, upstream uncontrolled tile drainage.

‡ Controlled tile drainage.

§ Uncontrolled tile drainage.

¶ Not measured.

Bold *P* values are significant at the ≤0.1 probability level.

Table 6. Growing season mean daily total P mass fluxes and paired *t* test results from downstream controlled tile drainage, downstream uncontrolled tile drainage, and upstream uncontrolled tile drainage sites for calibration (2005–2006) and treatment periods (2007–2011).

Year	Site pair†	CTD‡ (mean ± SD)	UCTD§ (mean ± SD)	CTD – UCTD (difference ± SD)	*n*	*P* value	Change
		kg ha^{-1} d^{-1}					%
2005	CTD_D vs. $UCTD_D$	$6.92 \times 10^{-4} \pm 1.47 \times 10^{-3}$	$3.90 \times 10^{-4} \pm 1.02 \times 10^{-3}$	$3.02 \times 10^{-4} \pm 9.81 \times 10^{-4}$	166	**≤0.001¶**	77
	CTD_D vs. $UCTD_U$	$6.92 \times 10^{-4} \pm 1.47 \times 10^{-3}$	$7.32 \times 10^{-4} \pm 1.73 \times 10^{-3}$	$-4.00 \times 10^{-5} \pm 9.96 \times 10^{-4}$	166	0.606	−5
2006	CTD_D vs. $UCTD_D$	$5.47 \times 10^{-4} \pm 1.11 \times 10^{-3}$	$4.17 \times 10^{-4} \pm 1.02 \times 10^{-3}$	$1.30 \times 10^{-4} \pm 6.11 \times 10^{-4}$	159	**0.008**	31
	CTD_D vs. $UCTD_U$	$5.47 \times 10^{-4} \pm 1.11 \times 10^{-3}$	$7.14 \times 10^{-4} \pm 1.83 \times 10^{-3}$	$-1.67 \times 10^{-4} \pm 1.29 \times 10^{-3}$	159	0.104	−23
2005–2006 (pooled)	CTD_D vs. $UCTD_D$	$6.21 \times 10^{-4} \pm 1.30 \times 10^{-3}$	$4.03 \times 10^{-4} \pm 1.02 \times 10^{-3}$	$2.18 \times 10^{-4} \pm 8.24 \times 10^{-4}$	325	**≤0.001**	54
	CTD_D vs. $UCTD_U$	$6.21 \times 10^{-4} \pm 1.30 \times 10^{-3}$	$7.23 \times 10^{-4} \pm 1.78 \times 10^{-3}$	$-1.02 \times 10^{-4} \pm 1.15 \times 10^{-3}$	325	0.110	−14
2007	CTD_D vs. $UCTD_D$	$2.75 \times 10^{-4} \pm 1.27 \times 10^{-3}$	$2.58 \times 10^{-4} \pm 8.54 \times 10^{-4}$	$1.70 \times 10^{-5} \pm 9.78 \times 10^{-4}$	148	0.832	7
	CTD_D vs. $UCTD_U$	$2.75 \times 10^{-4} \pm 1.27 \times 10^{-3}$	$2.55 \times 10^{-4} \pm 9.17 \times 10^{-4}$	$1.94 \times 10^{-5} \pm 7.94 \times 10^{-4}$	148	0.767	8
2008	CTD_D vs. $UCTD_D$	$8.11 \times 10^{-4} \pm 1.79 \times 10^{-3}$	$5.53 \times 10^{-4} \pm 1.17 \times 10^{-3}$	$2.58 \times 10^{-4} \pm 1.07 \times 10^{-3}$	150	**0.004**	47
	CTD_D vs. $UCTD_U$	$8.11 \times 10^{-4} \pm 1.79 \times 10^{-3}$	$5.22 \times 10^{-4} \pm 1.38 \times 10^{-3}$	$2.89 \times 10^{-4} \pm 7.68 \times 10^{-4}$	150	**≤0.001**	55
2009	CTD_D vs. $UCTD_D$	$4.52 \times 10^{-4} \pm 5.14 \times 10^{-4}$	$1.04 \times 10^{-3} \pm 6.63 \times 10^{-4}$	$-5.90 \times 10^{-4} \pm 7.85 \times 10^{-4}$	155	**≤0.001**	−57
	CTD_D vs. $UCTD_U$	$4.52 \times 10^{-4} \pm 5.14 \times 10^{-4}$	$6.15 \times 10^{-4} \pm 5.50 \times 10^{-4}$	$-1.63 \times 10^{-4} \pm 4.72 \times 10^{-4}$	155	**≤0.001**	−27
2010	CTD_D vs. $UCTD_D$	$3.24 \times 10^{-4} \pm 5.09 \times 10^{-4}$	$1.45 \times 10^{-4} \pm 4.21 \times 10^{-4}$	$1.79 \times 10^{-4} \pm 4.43 \times 10^{-4}$	132	**≤0.001**	123
	CTD_D vs. $UCTD_U$	$3.24 \times 10^{-4} \pm 5.09 \times 10^{-4}$	$1.56 \times 10^{-4} \pm 5.23 \times 10^{-4}$	$1.68 \times 10^{-4} \pm 4.82 \times 10^{-4}$	132	**≤0.001**	108
2011	CTD_D vs. $UCTD_D$	$7.86 \times 10^{-5} \pm 1.64 \times 10^{-4}$	$3.19 \times 10^{-4} \pm 7.89 \times 10^{-4}$	$-2.41 \times 10^{-4} \pm 7.43 \times 10^{-4}$	154	**≤0.001**	−75
	CTD_D vs. $UCTD_U$	$7.86 \times 10^{-5} \pm 1.64 \times 10^{-4}$	$1.71 \times 10^{-4} \pm 4.06 \times 10^{-4}$	$-9.29 \times 10^{-5} \pm 3.39 \times 10^{-4}$	154	**0.001**	−54
2007–2011 (pooled)	CTD_D vs. $UCTD_D$	$3.89 \times 10^{-4} \pm 1.06 \times 10^{-3}$	$4.75 \times 10^{-4} \pm 8.82 \times 10^{-4}$	$-8.63 \times 10^{-5} \pm 8.94 \times 10^{-4}$	739	**0.009**	−18
	CTD_D vs. $UCTD_U$	$3.89 \times 10^{-4} \pm 1.06 \times 10^{-3}$	$3.50 \times 10^{-4} \pm 8.56 \times 10^{-4}$	$3.90 \times 10^{-5} \pm 6.20 \times 10^{-4}$	739	**0.088**	11

† CTD_D, downstream controlled tile drainage; $UCTD_D$, downstream uncontrolled tile drainage; $UCTD_U$, upstream uncontrolled tile drainage.

‡ Controlled tile drainage.

§ Uncontrolled tile drainage.

¶ Bold *P* values are significant at the ≤0.1 probability level.

(Skaggs et al., 2010, 2012). To a large degree, such a contention is supported by the largely mixed and inconclusive FWMC results of this watershed-scale study.

In the case of stream water fluxes, our results are consistent and can be compared with other experimental and modeling studies (Evans et al., 1992; Ale et al., 2012). Considering that the study reported here documented mitigation during the growing season only, our watershed-scale reductions seem to compare well with other studies on CTD NO_3^-–N loadings. Experimental studies at smaller scales or modeling efforts (using CTD variably over the course of the entire year) have shown that controlled drainage reduced N export between ~20 to 80% over free-drainage systems (Lalonde et al., 1996; Tan et al., 1998; Thorp et al., 2008; Drury et al., 2009; Jaynes 2012), and a year-round field-scale study by Adeuya et al. (2012) showed nitrate reductions between 18 and 23% using controlled drainage.

One process that could dampen tile NO_3^-–N flux reductions by CTD as expressed in surface water trends is seepage fluxes of mineral N from CTD fields to stream. Sunohara et al. (2014) found that although CTD significantly reduced tile mineral N fluxes for fields in this watershed (by 59% for corn and 44% for soybean), growing season seepage fluxes from fields to stream for CTD fields were approximately 0.05 kg N m^{-1} seepage front relative to UCTD fields, which had average fluxes of 0.025 kg N m^{-1}. This was a result of higher groundwater concentrations of N and greater flow gradients imposed by water table management. Skaggs et al. (2010) similarly report the importance of increases in lateral and vertical seepage where controlled drainage is used, potentially increasing N fluxes to surface water.

Both streams in this study would also have promoted in-stream denitrification given the lower oxygen concentrations observed (Supplemental Table S4), especially at lower flow conditions. Although it is unclear why, the reference (UCTD) watershed had slightly lower oxygen concentrations (average, 5.02 mg L^{-1}) than the treatment (CTD) system (average, 6.12 mg L^{-1}) as well as somewhat more prolonged oxygen concentrations below 2 mg L^{-1} in relation to the CTD stream (Supplemental Table S5). Theoretically, these low oxygen conditions could affect how CTD N mitigation was translated at the watershed scale of investigation because the reference (UCTD) watershed could have dampened N fluxes via in-stream processes to a greater extent than the treatment (CTD) watershed via these means. However, from a net watershed output perspective, in-stream denitrification (Mulholland et al., 2009) and uptake of nutrients by plants in the channel and riparian area (Hefting et al., 2005) would further reduce net fluxes from both watershed systems, thereby reducing further downstream burdens on water quality.

To date there have been few, if any, experimental efforts that evaluate P fluxes as a result of field-to-field–based CTD intervention at watershed scales as documented here. At the plot scale, Wesström and Messing (2007) found overall lower total P export under controlled drainage compared with free drainage. Feser et al. (2010), however, found in their plot-scale study that, although there were lower P exports under controlled drainage conditions, the flow-weighted mean P concentrations were overall higher. Deal et al. (1986), through simulations, found that controlled drainage could reduce nitrate export by as much as 34% in some soil/management situations, although they predicted small increases in P export as a result of controlled drainage. Overall, our total P and DRP fluxes during the growing season from both tile-drained watersheds were generally low in relation to fluxes in other agriculturally dominated watersheds (e.g., Vanni et al., 2001; Royer et al., 2006; Macrae et al., 2007) despite the fact that dairy cow manure is applied as an organic fertilizer to many fields in both watersheds on a yearly basis in spring and/or fall. The maximum total study season flux of DRP and total P from the reference (UCTD) watershed was ~0.03 (2006) and ~0.16 kg ha^{-1} study $season^{-1}$ (2009), respectively. Reasons for this could be related to seasonality, such as the study being conducted during time periods when manure applications were muted, discharge events were fewer, and antecedent water contents were lower in relation to earlier in spring and later in fall. Usually the largest fluxes of nutrients occur over winter and early spring (Royer et al., 2006), periods not covered during our study for the aforementioned reasons.

Additionally, although tile drains have been shown to contribute variably to DRP export, a very large source of P export is attributed to particulate P from overland flow (Sims et al., 1998; Royer et al., 2006; Gentry et al., 2007; Frey et al., 2015), a transport pathway not observed to have occurred on the flat watershed fields in our study to any spatially extensive degree. Phosphorus mobilization and enhanced P export via surface and subsurface flow pathways are generally well documented (Beauchemin et al., 1998; Sims et al., 1998; Gentry et al., 2007; Valero et al., 2007; Tan and Zhang, 2011; Ball Coelho et al., 2012; Kleinman et al., 2015). Phosphorus has the potential to be mobilized through increased soil water content (Scalenghe et al., 2002), and CTD can increase water tables as well as surface soil water content over fields in this watershed system (Nangia et al., 2013). However, the growing season soil water contents at the surface are usually lower than those over the nongrowing season, which would likely result in lower P mobilization in the near surface during the growing season in relation to the nongrowing season (Ball Coelho et al., 2012).

The degree of soil P release and transport depends in part on soil P saturation (Pote et al., 1996). In this study, fields in both watersheds had a mix of lower to higher degrees of P saturation (1–22%; n = 33 fields) (0–0.3 m depth). The potential for elevated water tables interacting with P-rich surface soils more readily in the CTD fields could help mobilize P into preferential flow paths (Grant et al., 1996), thereby promoting additional inputs to tile (which for the CTD system would contribute during control structure overflow events). In fact, Frey et al. (2013) found, for clay loam field plots after manure applications in the fall, that there were no significant differences in total P and reactive P loads (or any other nutrient) among CTD and UCTD plots (rainfall soon after application elevated water tables in the zone of manure augmenting nutrient fluxes in CTD overflow).

The experimental data herein suggest that, during the growing season, watershed-scale DRP and total P fluxes decreased as a result of CTD intervention, with the exception of one site comparison (CTD_D vs. $UCTD_U$ for daily total P fluxes) (Fig. 5; Table 6). For specific paired fields studied on the treatment (CTD) watershed, CTD was found to reduce total P fluxes from field tile outlets by over 50% during the growing season (Sunohara et al., 2010). These findings,

combined with (i) the significant overall declines in (Fig. 5) stream water, NH_4^+–N, NO_3^-–N, and DRP daily fluxes in the treatment (CTD) watershed (relative to both reference site $UCTD_D$ and $UCTD_U$ trends); (ii) overall treatment period similarities in yearly regression slope changes among daily total P site comparisons where slope changes for CTD_D vs. $UCTD_U$ are always greater than slope changes for CTD_D vs. $UCTD_D$ (Supplemental Table S11); and (iii) lower regression slopes among daily total P for CTD_D vs. $UCTD_U$ during the calibration period in relation to CTD_D vs. $UCTD_D$ indicative of higher fluxes at the $UCTD_U$ site (Fig. 5), suggest that in-stream particulate P processes could have been formative in the anomalous CTD_D vs. $UCTD_U$ daily total P flux observations (in particular during the calibration period). Particulate P historically transports via variable flow pathways from fields to adjacent stream systems where it can accumulate and become available for remobilization and transport (e.g., Svendsen and Kronvang, 1993; Smith et al., 2005; Sharpley et al., 2013). In fact, Frey et al. (2015) documented that when stream sediments are exposed to the atmosphere during seasonally lower water levels (for stream systems in this study and in other stream systems nearby), significant mobilization and transport of P to surface water can occur due to rainfall erosion of those sediments (up to ~1.5 kg total P ha^{-1} yielded during a 15-min rain event). In contrast to the Frey et al. (2015) maximum observed total P yield given above, the maximum growing season total P flux for the $UCTD_D$ site in the reference (UCTD) watershed was 0.16 kg ha^{-1} study $season^{-1}$ (2009). This underscores the potential of in-stream processes in these small, low-gradient streams to mask or dampen environmental BMP effects on water quality. Of course, although there is large potential for particulate P inputs from in-stream sources to effectively conceal edge-of-field BMP effects on surface water quality, there can be equally affecting in-stream elements that reduce particulate P mobilization and export (e.g.,, small naturally produced impoundments and in-stream vegetation augmenting sedimentation) (Bilby, 1981; Reddy et al., 1999; Sunohara et al., 2012).

Conclusions and Recommendations

To our knowledge, this is the first study to date to examine the effects of field-to-field implementation of controlled tile drainage at the watershed scale on total and daily growing season stream water, N and P fluxes, and associated FWMCs. This study evaluated CTD effects at the watershed scale in three important ways: (i) comparisons of mass fluxes/FWMCs derived from a treatment watershed during a calibration period (~4–10% of tile-drained land under CTD in treatment watershed) with mass fluxes/FWMCs during a treatment period [~82% of tile-drained area was actively under CTD in the treatment (CTD) watershed during this 5-yr period]; (ii) comparison of treatment watershed results with a paired watershed completely under uncontrolled tile drainage [reference (UCTD) watershed] for these two periods of study, and (iii) assessment of mass fluxes/FWMCs from multiple stream monitoring sites.

The findings herein indicated that CTD imposed en masse on a field-to-field basis over a watershed during the growing season can significantly reduce growing season fluxes of stream water, NH_4^+–N, NO_3^-–N, DRP, and total P at the watershed scale. However, because watersheds are open systems, vigilance and awareness are crucial because of the the many possible environmental/land use drivers that can potentially influence, dampen, or mask BMP mitigation effects (e.g., in-stream processes). For FWMCs, results were more mixed and inconclusive, suggesting that physical abatement of tile flow and entrained nutrients is likely the overriding mechanism by which N and P fluxes are affected by CTD at this scale for growing season time periods. This finding is supportive of the use of simplified flow restriction approaches for predicting N and P fluxes affected by CTD at larger scales of investigation.

Our study results are promising for (i) supporting and promoting drainage water management initiatives and cost-share programs aimed at meeting N and P water quality targets for watersheds and river basins affected by tile drainage; (ii) refining, building, calibrating, and validating watershed hydrological models used to assess the impacts of drainage water management practices on the N and P cycle; and (iii) underscoring the potential of drainage water management practices to reduce downstream flood risks because CTD was found overwhelmingly in this study to significantly reduce stream discharge at watershed scale.

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