KOMPETENZZENTRUM WasserBerlin

REPORT

Cicerostr. 24
D-10709 Berlin
Germany
Tel +49 (0)30 536 53 800
Fax +49 (0)30 536 53 888
www.kompetenz-wasser.de

Project: AQUISAFE 2

WP4: Efficiency of implemented mitigation systems to control diffuse pollution in agricultural landscapes

Pierre-Andre Jacinthe, Amy N. Smith
Department of Earth Sciences, IUPUI

IUPUI INDIANA UNIVERSITY PURDUE UNIVERSITY INDIANAPOLIS

and

Lenore P. Tedesco
Wetlands Institute

Wetlands INSTITUTE

for
KompetenzZentrum Wasser Berlin gGmbH

Preparation of this report was financed through funds provided by Veolia

VEOLIA WATER

2014

Present report was developed in compliance with the requirements of the quality management system DIN EN ISO 9001:2008

Important Legal Notice

Disclaimer: The information in this publication was considered technically sound by the consensus of persons engaged in the development and approval of the document at the time it was developed. KWB disclaims liability to the full extent for any personal injury, property, or other damages of any nature whatsoever, whether special, indirect, consequential, or compensatory, directly or indirectly resulting from the publication, use of application, or reliance on this document. KWB disclaims and makes no guaranty or warranty, expressed or implied, as to the accuracy or completeness of any information published herein. It is expressly pointed out that the information and results given in this publication may be out of date due to subsequent modifications. In addition, KWB disclaims and makes no warranty that the information in this document will fulfil any of your particular purposes or needs. The disclaimer on hand neither seeks to restrict nor to exclude KWB's liability against all relevant national statutory provisions.

Wichtiger rechtlicher Hinweis

Haftungsausschluss: Die in dieser Publikation bereitgestellte Information wurde zum Zeitpunkt der Erstellung im Konsens mit den bei Entwicklung und Anfertigung des Dokumentes beteiligten Personen als technisch einwandfrei befunden. KWB schließt vollumfänglich die Haftung für jegliche Personen-, Sach- oder sonstige Schäden aus, ungeachtet ob diese speziell, indirekt, nachfolgend oder kompensatorisch, mittelbar oder unmittelbar sind oder direkt oder indirekt von dieser Publikation, einer Anwendung oder dem Vertrauen in dieses Dokument herrühren. KWB übernimmt keine Garantie und macht keine Zusicherungen ausdrücklicher oder stillschweigender Art bezüglich der Richtigkeit oder Vollständigkeit jeglicher Information hierin. Es wird ausdrücklich darauf hingewiesen, dass die in der Publikation gegebenen Informationen und Ergebnisse aufgrund nachfolgender Änderungen nicht mehr aktuell sein können. Weiterhin lehnt KWB die Haftung ab und übernimmt keine Garantie, dass die in diesem Dokument enthaltenen Informationen der Erfüllung Ihrer besonderen Zwecke oder Ansprüche dienlich sind. Mit der vorliegenden Haftungsausschlussklausel wird weder bezweckt, die Haftung der KWB entgegen den einschlägigen nationalen Rechtsvorschriften einzuschränken noch sie in Fällen auszuschließen, in denen ein Ausschluss nach diesen Rechtsvorschriften nicht möglich ist.

Colofon

Present report was developed in compliance with the requirements of the quality management system DIN EN ISO 9001:2008

Title
Efficiency of implemented mitigation systems to control diffuse pollution in agricultural landscapes

Authors
Pierre-Andre Jacinthe, Amy N. Smith, Lenore P. Tedesco

Quality Assurance
Pascale Rouault (KWB), Daniel Wicke (KWB)

Publication / Dissemination approved by technical committee members

Christelle Pagotto (Veolia Water, Technical Direction, Paris, France)
Boris David (Veolia Water, Technical Direction, Paris, France)
Magali Dechesne (VERI, Paris, France)
Emmanuel Soyeux (VERI, Paris, France)
Guy Randon (Veolia Eau Région Ouest, Rennes, France)
Lenore Tedesco (Wetlands Institute, USA)
Norbert Litz (Umweltbundesamt, Berlin, Germany)
Bernard Sautjeau (Setude, St. Malo, France)

Deliverable number
D 3.1

Abstract

The export of agricultural contaminants from agricultural landscapes of the US Midwest has contributed to the impairment of surface waters throughout the Mississippi River Basin and has been linked to various human health concerns. Natural treatment systems (wetlands, bioswales, bioreactors) can capture agricultural runoff and significantly reduce nutrient loading to downstream waters but there is a paucity of data on the effectiveness of these treatment systems to attenuate the suite of pollutants (nutrients and synthetic organics) typically found in agricultural runoff. This understanding is important given that the degradation of different pollutants involves metabolic pathways that often require different redox environments. As part of the Aquisafe-2 project, a bioretention swale comprising two treatment cells (a subsurface cell in series with a surface cell) was monitored, and its performance evaluated over a three-year period (2011 - 2013). Results showed that the bioswale was moderately efficient with regard to nitrate (NO_3^-; retention range: 16-58 %). N removal averaging 30 % was measured during a series of wetting events during which the bioswale operated at an estimated average hydraulic retention time (HRT) of 0.97 day. Spatial analysis of the data showed that almost all the NO_3^- removal occurred in the subsurface cell; however, N removal was also measured in the surface cell under low flow conditions (estimated HRT: 2.5 days). The highest rates of N removal (~ 58 %) were measured when the bioswale stayed wet for several days probably due to the development of a more optimum environment for denitrifying microbes. Nitrate removal capacity was limited by NO_3^- availability, short retention times during high flows, and the frequent fluctuation between oxic and anoxic conditions, but not by water temperature (8.3-16.6 ^oC) and dissolved organic carbon (DOC; 1.9 - 29.2 mg C L^{-1}). The bioswale performance with regard to soluble reactive phosphorus (SRP) and atrazine was more variable, with net retention during some periods and net release at other times. The bioswale was a net source of P during most sampling periods with an average SRP release corresponding to 13 % of input, probably due to desorption of water soluble P from the topsoil applied during construction. This interpretation is supported by the progressive decline in P release observed between the first and third year of monitoring. The subsurface and the surface cells contributed almost equally to the fate of P in the bioswale. Likewise, the bioswale was at times a small/moderate sink (13-31 % retention) for atrazine, and a net source (-38 % to -15 %) during periods when the bioswale received overland runoff from the adjacent crop field which bypassed the subsurface cell. Results suggested that competition between atrazine and DOC for sorption sites is a possible mechanism affecting atrazine removal efficiency. Additional work is needed to compare the efficiency of the subsurface and surface cells with regard to atrazine, and elucidate the biogeochemical factors controlling its fate in the bioswale.

Acknowledgements

Thanks to IUPUI Center for Earth and Environmental Science staff and IUPUI students for their support on this project: Pamela Martin, Robert Hall, Robert Barr, Michael Stouder, Nicholas Clercin, Samantha Simpson, Ty Winslow, Lori Bebinger, Brittani Burton, Makenna Wesner, Lauren Thomas, Samapriya Roy, Deborah Piemonti, Elizabeth Johnson, Austin Taylor, Shiva Ladan, Prajaya Prajapati, Amanda Brumbaugh, and Sean Stultz.

Table of Contents

List of Figures

List of Tables

Chapter 1 Introduction

The export of nutrients (mainly nitrogen and phosphorus) from tile-drained agricultural fields of the US Midwest has contributed to the impairment of surface waters throughout the Mississippi Basin and eutrophication of coastal areas. Continued input of nutrients to surface waters generally results in eutrophication, and a host of water quality issues. The US Environmental Protection Agency (EPA) has established a maximum concentration limit of 10 mg N L^{-1} for nitrate in drinking water. This limit is often exceeded in rivers and streams, especially after rainfall events following fertilizer application to crop fields. Under optimum light and temperature conditions, nutrient-enriched aquatic ecosystems often experience algal blooms. Decomposition of algal biomass by resident microbes leads to dissolved oxygen (DO) depletion, ultimately resulting in hypoxia and enhanced potential for fish kills. The so-called dead zone (dissolved oxygen < 2 mg L^{-1}) in the Gulf of Mexico is one of the best known hypoxic zones in the world. Its size has doubled during the last two decades (~$8x10^3$ km^2 prior to 1993 to nearly $20x10^3$ km^2 in 2006; Joyce, 2000). With 58 % of the Mississippi River basin under various agricultural land-uses (NRC, 2008), it has been argued that intensive agriculture in the Corn Belt States (including Indiana) poses the greatest threat to water quality and ecosystem health in the Mississippi River delta (Turner and Rabalais, 2003).

In addition to sediments and plant nutrients, the presence of herbicides in agricultural runoff has further contributed to water quality deterioration. Atrazine (2-chloro-4-ethylamino-6-isopropyl amino-1,3,5-triazine) is a widely-used herbicide in the region's agriculture, but could have negative effects on aquatic life (limiting aquatic plants and endocrine disruptors in amphibians and fish) and human health. A maximum contaminant level (MCL) of 3 µg L^{-1} on an annual average basis has been established by the USEPA for atrazine in drinking water. Despite its relatively short half-life (60-100 days depending on soil aeration and temperature; USDA, 2001; Chirnside et al., 2009), atrazine and its decomposition by-products can persist in the environment. For municipalities like Indianapolis that rely on surface water as their primary source of drinking water, the presence of agricultural contaminants increase drinking water production costs due to the extensive treatment that is required. Thus, there is a need to protect water resources from contamination in order to reduce both the cost of drinking water treatment and the risk to public health.

The interception of agricultural runoff and tile-drainage discharge in vegetated buffers and/or wetlands (natural or constructed) strategically located downslope of cultivated fields could help reduce nutrient export and mitigate these water quality challenges. AQUISAFE-1 project concluded that nutrient enrichment remains the most pervasive issue facing water quality managers and proposed the installation of near-natural mitigation zones to attenuate agricultural diffuse pollution through the capture of subsurface drainage and treatment. These mitigation zones can supplement nutrient management practices implemented by farmers. The AQUISAFE-2 project was launched with a primary goal of assessing the merit of that proposition through a series of technical-scale and field-scale evaluations.

Several studies have examined the effectiveness of constructed wetland systems and their capacity to retain nutrients in agricultural runoff (Hunt et al., 1995; Carpenter et al., 1998; Casey and Klaine, 2001; Kovacic et al., 2006; Reinhardt et al., 2006; Jacinthe et al., 2009). These semi-natural systems could provide an effective approach for the retention of agricultural pollutants, and previous studies have evaluated their effectiveness for various point source pollutants

including heavy metals from coal-ash processing plants (Ye et al., 2001) and estrogens from animal feeding facilities (Shappell et al., 2007). However, the control of diffuse pollutants could be more challenging due to the multiplicity of pollutant sources (overland, atmospheric and subsurface) and seasonal variability of inputs (Carpenter et al., 1998). Nonetheless, several successful case studies have been reported, particularly with regard to the control of nitrogen (N) and phosphorus (P) export in agricultural landscapes. Hunt et al. (1995) reviewed the nutrient retention effectiveness of several treatment wetlands that capture discharge from swine and dairy farm facilities, and reported TP and NH_4-N (ammonium-N) reductions between 59-80 % and 54-94 %, respectively. The removal of nitrate (NO_3^-) and phosphate (PO_4^{-3}) averaged 80 and 74%, respectively in a South Carolina riparian wetland receiving runoff from fertilized golf courses (Casey and Klaine, 2001). This limited review of the available literature indicated that constructed wetlands can effectively remove large amounts of N and P, but rates can be highly variable probably due to variable flow conditions and other environmental factors. Variation in the chemical composition of agricultural discharge could also contribute to the variability of system effectiveness. Further, the wetland systems investigated in past studies were generally designed for optimum attenuation of one pollutant, but not for the various classes of contaminants (eg. nutrients, organics) that are often present in agricultural runoff. Biological transformation of different pollutants may require widely different redox conditions, for example. Past studies have also suggested that microbial degradation of NO_3^- and atrazine is optimized under different conditions. While aerobic conditions generally favor atrazine degradation, NO_3^- degradation takes place in anaerobic conditions (Mudhoo and Garg, 2011).

The US Midwest is characterized by a temperate humid climate, flat landscapes, and poorly-drained soils developed in dense glacial till. Because of these hydrogeomorphic settings, agricultural fields are equipped with an extensive network of subsurface tile drains (on average 1.5 m deep) that flow underneath the crop field and discharge into a nearby ditch. Installation of this infrastructure is necessary for the removal of excess water and timely implementation of farming activities in the region's croplands. At the same time, tile drains accelerate the export of pollutants and exacerbate surface water impairment. Most agricultural best management practices, such as riparian buffer strips and grassed waterways and swales which potentially could help reduce the export of agrochemicals to streams, have proven ineffective. When subsurface tile drainage systems (pipes) are present, the water table remains deep and limited interactions occur between nutrient-laden drainage waters and the biologically-active surface soil layer. This hydrological disconnection effectively negates the effectiveness of these management practices in tile-drained landscapes. New emphasis has now been placed on constructed wetlands for agricultural water quality management. Current effort is on modifying the operation of subsurface drains by channeling tile discharge into a constructed wetland where enhanced biotransformation of pollutants can take place. While this approach could be successful, limited data exists with regard to their effectiveness. In previous studies, tile discharge was directed into bioreactors placed either underground (Robertson et al., 2000; van Driel et al., 2006; Jaynes et al., 2008) or within existing stream channels (Robertson and Merkley, 2009). These treatment systems often rely on the use of carbonaceous media such as mulch, sawdust, and compost. Depending on the media used, variable rates of nitrate (NO_3^-) removal have been reported (Robertson et al., 2000; van Driel et al., 2006; Jaynes et al., 2008; Robertson and Merkley, 2009). These studies have not addressed the long-term sustainability of these systems with regard to their C-supplying capacity to denitrifiers.

The design and operations of constructed wetlands near a tile-drained agricultural field could pose unique challenges requiring managers to strike the right balance between farm accessibility and system effectiveness. For example, when tile drain outlets are channeled to constructed wetlands, the flow of water from cropland can be impeded, resulting in delayed planting or stunted growth of already planted crops. Therefore, acceptance of wetlands for the treatment of agricultural runoff by the farming community hinges primarily on the ability of these treatment systems to remain effective without interfering with farming operations. Past studies (Robertson et al., 2000; van Driel et al., 2006; Jaynes et al., 2008; Robertson and Merkley, 2009) have generally shown that hydraulic retention time (HRT) is the factor that most strongly controls the efficiency of bioreactors, with longer HRT being generally correlated with greater nutrient removal efficiency. However, as stated above, long HRT may not be possible in some settings as this could cause wet soils in adjacent agricultural fields, and ultimately lead to farmers' rejection of the technology. It is also not clear, how flow path (subsurface versus surface) could affect the optimum HRT for effective removal of different contaminants that may be present in agricultural runoff.

In this field research and demonstration, the nutrient removal efficiency of a bio-retention swale (hereafter referred to as bioswale) was evaluated. Hydrological and biogeochemical parameters were monitored to identify relevant processes. The bioswale includes both a subsurface flow cell and a surface treatment system. Tile discharge is directed to the subsurface flow cell (anaerobic) and then to the surface flow cell (aerobic) prior to being discharged into a nearby stream. This design was selected in order to investigate the impact of the redox environment in each cell on the fate of different contaminants.

The bioswale is located downslope from an intensively-managed crop field. It receives runoff and drainage water from a subsurface tile (10.2 cm diam.) emerging from underneath the cultivated field. The field is in corn-soybean rotation and under no-till since 2004. During the corn year, fertilizer is applied to the crop field at rates ranging between 178-200 kg ha^{-1} for N and 110 kg ha^{-1} for P. No N fertilizer is applied when soybean is grown. During the 2011 and 2013 growing seasons, the adjacent field was planted to corn (*Zea mays*). Soybean (*Glycine max.*) was grown in 2012.

1.1 Objective and significance

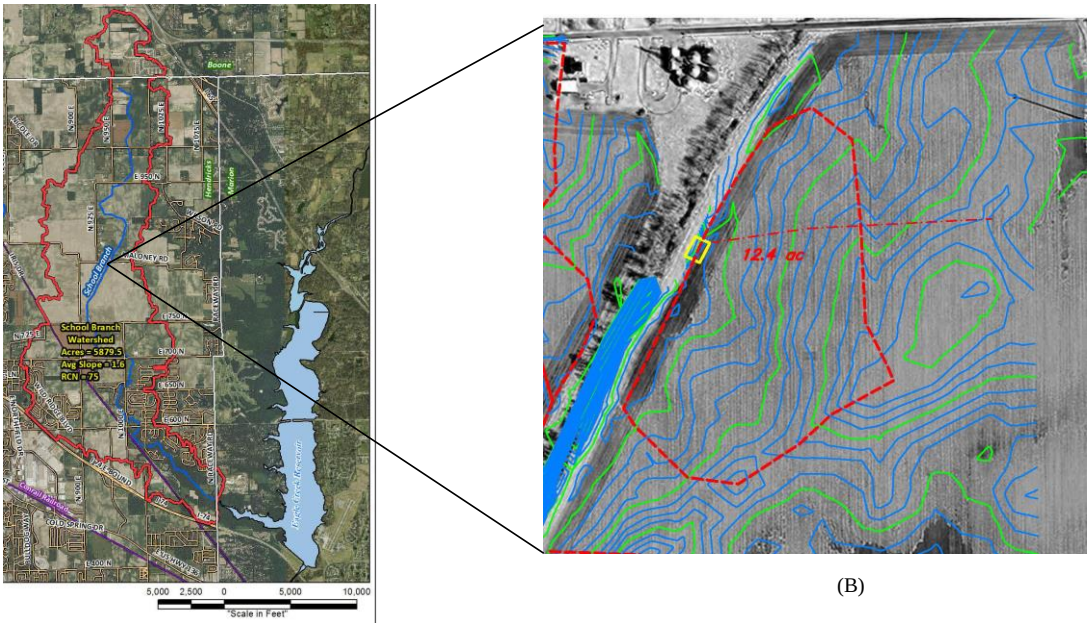
The primary objective of Work Package 3 was to evaluate the performance of semi-natural mitigation systems to attenuate the export of agricultural pollutants to surface waters. Field research was conducted to examine system performance with regard to different pollutants, and especially during high flow events when peaks in water and nutrient input are expected. This understanding could help identify future design adjustments to be made to improve the efficiency and facilitate the management of these mitigation systems. These performance criteria are critical to facilitating the adoption of these systems by farmers and other resource managers.

Chapter 2 Site Characterization

2.1 Selection and characterization of the study site, methods, and instrumentation

The study was conducted in the School Branch watershed, in Hendricks County, Central Indiana (USA) (Fig. 1). Region climate is humid continental with a long-term mean annual precipitation of 1040 mm (NOAA, 2005). For the period of study, weather data were obtained from the Midwest Regional Climate Center (http://mrcc.isws.illinois.edu/) for a weather station (id: USW00053842) located near the Eagle Creek Reservoir, about 5 km south-east of the study site. Annual rainfall totaled 1319, 953 and 1058 mm in 2011, 2012 and 2013, respectively. The year 2012 deviated significantly from normal and was characterized by a mild winter, limited snowfall, and a very dry summer. During the peak of the growing season in 2012 (May-August), a record 102 days without precipitation was observed in the area.

A topographic analysis of the watershed was conducted to identify depressional areas where wetlands would have naturally occurred if subsurface tile drainage were not installed on the landscape (Babbar-Sebens et al., 2013). Based on that analysis and subsequent consultations with a cooperating land-owner (Mr. Mike Starkey), a construction site for the bioswale was selected in a riparian area along the School Branch stream channel (39°53' N, 86°21' W). A subsurface tile (10.2 cm diameter) extending approximately 300 m into the adjacent agricultural field was diverted into the bioswale. The tile drains an area approximately 1.2 ha in size (Fig. 1). However, during major rainfall events, runoff from an area of 5 ha can potentially flow toward the bioswale. Soils at this location are poorly-drained Brookston (mesic Typic Haplaquolls) developed in Wisconsinan glacial till underlain by dolomite and limestone bedrock (NRCS, 2006). Construction of the bioswale took place between April and June 2011.



(A) (B)

Fig. 1. Location of the bioswale in the School Branch watershed (A). The potential surface drainage area of the bioswale is shown in insert B. The straight red line depicts the subsurface tile drain intercepted by the bioswale (yellow box).

In the Central Indiana region, water flow in agricultural tiles is seasonal; tiles typically flow in the spring (April - June), may completely stop in mid-summer, and start flowing again in November-December as soils rewet from autumn rainstorms. Flow events sometimes occur in mid-late winter if the soil is not frozen or a significant thaw occurs. Prior to the construction of the bioswale, periodic measurements of tile discharge were made (2007-2011) in order to determine proper dimensions for the system. Average flow was 0.0003 and 0.012 $m^3 s^{-1}$, in spring 2007 and autumn 2009, respectively. The bioswale was designed for an expected peak discharge of 0.002 $m^3 s^{-1}$.

Soil characterization was conducted in May 2010. Bulk soil and soil cores were collected from the plow layer (0-40 cm) and subsurface (40-60 cm) to determine general soil characteristics. Soil pH was measured with a pH-meter using a soil suspension (1: 2 soil to water). Finely-ground soil (150 µm) was analyzed for total C and N (dry combustion at 950 ^oC). Total soil P was determined by the ashing (550 °C, 1 h) and acid extraction (1N HCl) procedure as described by Andersen (1976). P fractionation was conducted using the Hedley (1982) procedure that separates soil P fractions based on their relative solubility in water, alkaline and acidic solutions. Using 0.5 g field moist soil sample, each fraction was sequentially extracted using 30 mL of the appropriate reagent (contact time of 16 h and filtration). The following P fractions were obtained: water extractable inorganic P (WEP), moderately labile inorganic P extracted with 0.5 M $NaHCO_3$, Fe/Al-bound P extracted with 0.1M NaOH, and Ca/Mg-bound P extracted with 1 M HCl. Inorganic P was determined using the molybdate colorimetric method (D'Angelo et al., 2001). In accord with other studies, the water extractable P (WEP) was found to be a good predictor of inorganic P flux (r^2: 0.93, $P < 0.03$) upon flooding of soils at the bioswale site (Smith and Jacinthe, 2014).

Table 1. Soil properties and P fractions at the study site. Values are means ± standard deviations. The P fractions are reported in units of mg P kg^{-1} soil.

	0-40 cm	40-60 cm
pH	6.9 ± 0.02	6.9 ± 0.02
Soil organic C (g C kg^{-1})	21.7 ± 0.8	12.4 ± 0.1
Total N (g N kg^{-1})	2 ± 0.2	1.2 ± 0.1
Total P (mg P kg^{-1} soil)	611 ± 111	352 ± 115
Water extractable P (WEP)	11 ± 0.9	4.8 ± 1.1
$NaHCO_3$ extractable P	23.2 ± 5.2	13 ± 4.1
NaOH extractable P (Fe/Al bound)	165.1 ± 13.3	115.2 ± 9.8
HCl extractable P (Ca/Mg bound)	96.9 ± 4	126 ± 21.2

The bioswale was designed to collect drainage water from a 10.2 cm diameter tile and was sized to capture discharge even after major rainfall events. It includes two cells: a subsurface flow cell (39.6 m x 4.4 m) and a surface flow cell (39.6 m x 2.7 m). A cross-sectional view of the sub-surface and surface cells is depicted in Fig. 2. The bottom of the subsurface flow cell consists of a 60-cm layer of coarse gravel (2.5 cm diameter) and bark mulch, overlaid by 30 cm of pea gravel (0.5 to 1 cm diameter) and then 60 cm of native soil on top. Near the bioswale inlet, tile discharge flows into the pea gravel layer, allowing incoming water to infiltrate into the bioreactive gravel/bark mulch layer. A 7 % slope was established underneath the subsurface flow cell to facilitate unidirectional flow (from inlet to outlet). The surface flow cell includes a gravel layer (15 cm) at the bottom and a layer (15 cm) of native soil on top. This open cell is designed to temporarily hold water. Water is intended to slowly seep from the subsurface flow cell into the surface flow cell through the central berm (Fig. 2). During high flow events, water can also transfer from the subsurface to the surface flow cell through two connector pipes located in the pea gravel layer in the subsurface cell (Fig. 3). Water levels and hydraulic retention time in the bioswale can be adjusted using the water control structure located at the outlet (Fig. 2). Taking into consideration, the length (39.6 m), width (4.4 m), depth (1.5 m) and the bulk density of construction materials (soil, mulch, gravel), the total water storage capacity (one pore volume) of the subsurface flow cell is estimated at 39.7 m^3. Including the open surface cell (maximum storage capacity of 279 m^3), the potential water storage of the bioswale is 319 m^3. Details regarding the porosity of construction materials and computation of water storage potential are reported in Appendix E.

The bioswale was instrumented with a network of piezometers, monitoring wells, flow meters, and level loggers to monitor fluctuation in water level and flow direction. Piezometers and monitoring wells were located in three transects (hereafter referred to as T1, T2, T3) down the length of the bioswale in both the subsurface and surface flow cells (Figs. 2-4). For each nest of piezometers, one was located 30 cm below the bottom of the cell, and the others at two different depths corresponding to the gravel/bark layer and the soil layer (Fig. 2). For the surface flow cell, piezometers were only located 15 cm below the cell bottom (Figs. 2-4). Level loggers were placed in selected piezometers to continuously measure water level. Flow meters were installed in the inlet and outlet of the bioswale to monitor water discharge in and out of the system.

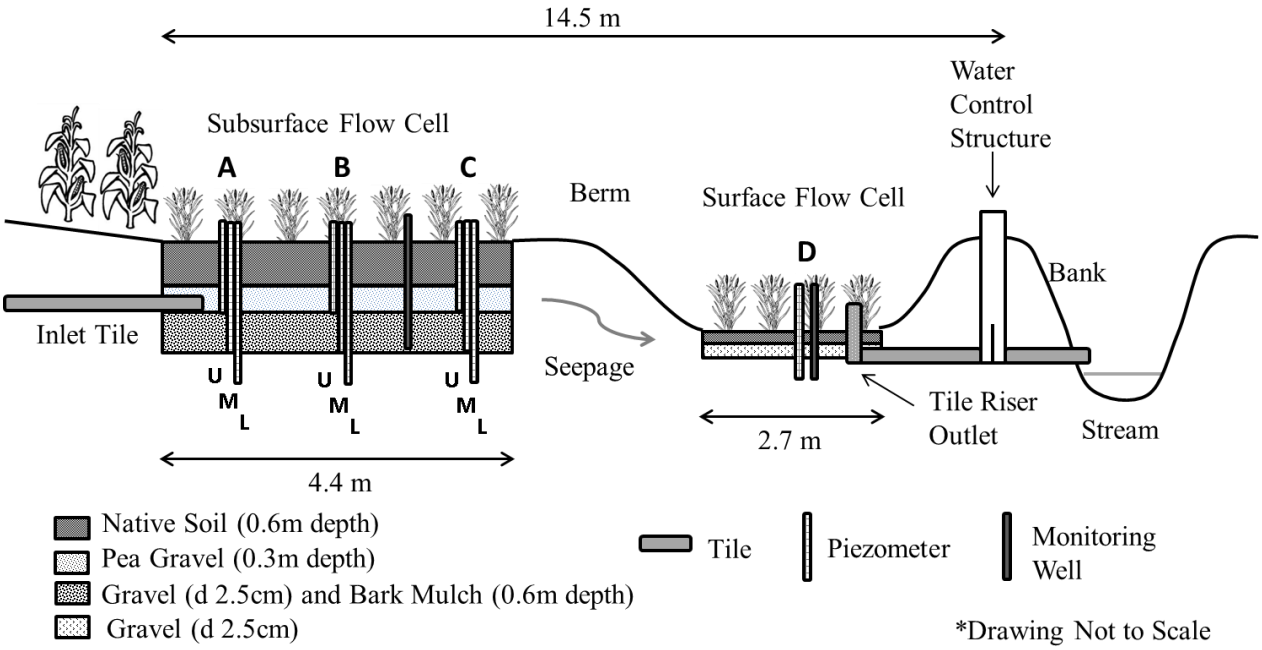


Fig. 2. Cross-sectional view of bioswale showing subsurface flow and surface flow cells, along with the approximate location of piezometers and monitoring wells.

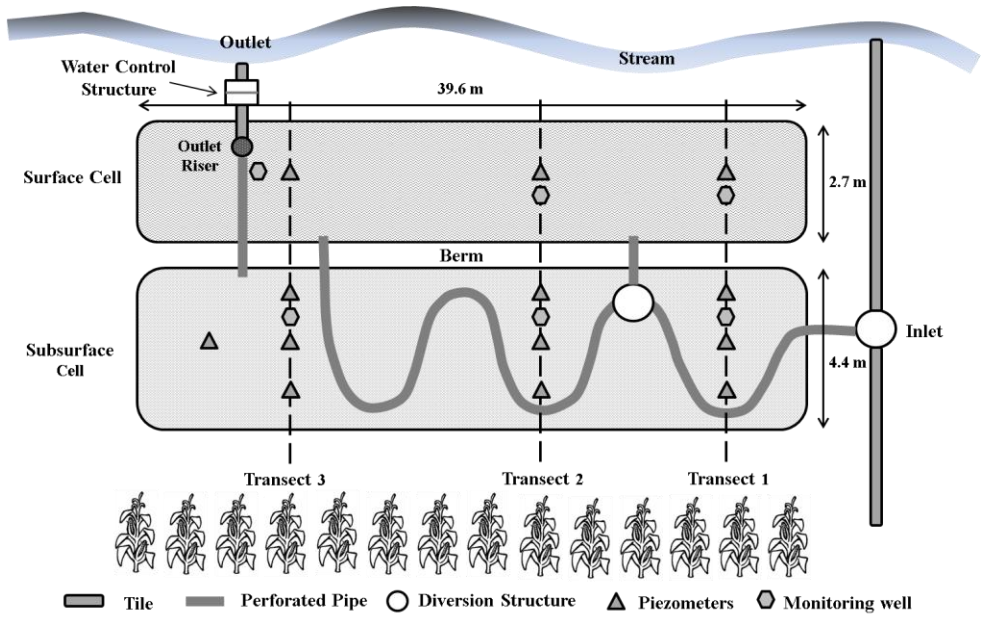


Fig. 3. Top-down view of water flow direction through the various compartments of the bioswale.

Water flowing through the bioswale was monitored in the inlet, outlet, and the monitoring wells at each transect in the subsurface cell. Parameters monitored included temperature, conductivity, salinity, dissolved oxygen, pH, and oxidation reduction potential (ORP) using YSI 600XLM multi-parameter sondes interfaced with data loggers. Data was periodically downloaded and stored on a computer. Water sampling was typically initiated once the tile was activated and flow

was detected in the piezometers and monitoring wells. During storm events, water samples were collected using ISCO automated samplers at the inlet, transect 3, surface cell, and outlet. Water was collected at transect 3 via the connection pipe between the two cells (hereafter referred to as subsurface connector). From each sampling location, water samples were withdrawn every 1 to 2 hours by the ISCO autosamplers during periods of peak flow. Sample bottles were collected immediately after each storm, normally less than 24 h. For storms lasting > 24 h, sampling bottles were removed and replaced daily by new bottles until return to near baseflow. Samples to be analyzed were selected based upon the hydrograph of flow into the bioswale. Along the descending limb of the hydrograph, sampling frequency ranged from every 4 hours to every 24 hours depending on flow pattern and duration. Once collected, water samples were taken to the laboratory for preprocessing, preservation, storage and analysis.

Water samples were filtered (0.45 µm GF/F filters), and filtrate was stored frozen if not analyzed immediately. Samples were analyzed for NO_3^-, NH_4^+, o-PO_4^{-3}, dissolved organic C (DOC), and atrazine. Analysis for $NO_3^-$and o-PO_4^{-3} was carried out using EPA methods 353.1 and 365.3 on an Aquachem Konelab 20 photometric analyzer (EST Analytical, Fairfield, OH). DOC was measured using a Vario TOC Cube analyzer (Elementar Inc., NJ). The enzyme-linked immunosorbent assay (ELISA) method was used for determination of atrazine concentration in water samples (Gruessner et al., 1995). Limits of detection of analytical methods are listed in Appendix D.

The performance of the bioswale was tested during three series of storms during the monitoring period (November 2011 - June 2013). The first series of rainstorms monitored occurred in late November/December 2011. The other flow-generating rainstorm events occurred in April/May 2012, and April/May/June 2013 (Fig. 5).

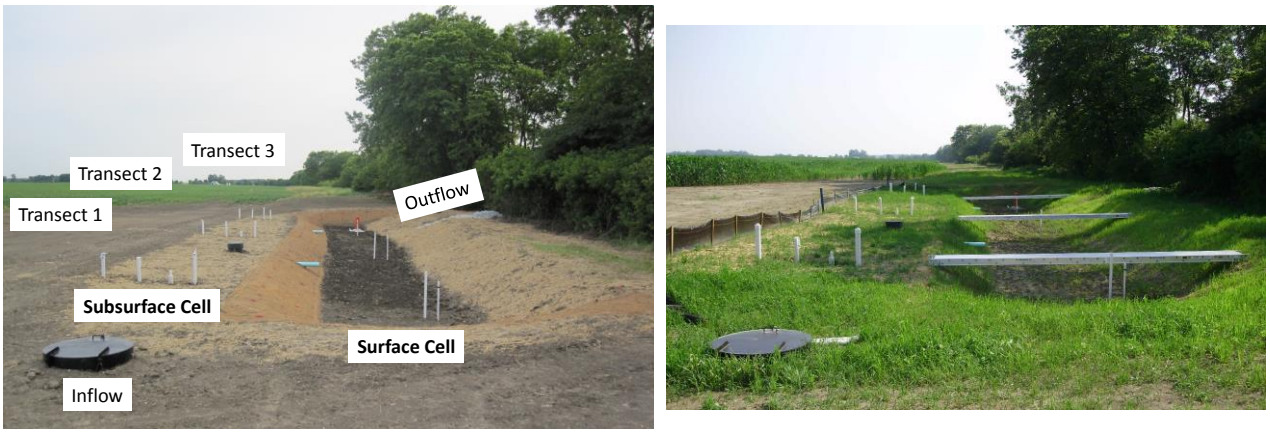


Fig. 4. The bioswale shortly after construction (*left*), and one month after installation (*right*).

2.2 Computational approach

Pollutant removal by the bioswale was computed in two ways depending on the availability of discharge data. The flow meter and level loggers installed near the inlet were frequently rendered inoperable by particulate matter that is sometimes present in tile drains.

When inlet discharge is unknown, removal (R) was computed as:

$$R = \frac{C_i - C_o}{C_i} x100 \qquad \text{(Eq. 1)}$$

in which, C_i and C_o are the inlet and outlet concentration, respectively. A positive value indicates a net removal while a negative value indicates a net release from the bioswale. This computational approach assumes that the system is at steady state and does not consider the travel time of a parcel of water through the system. Therefore, results obtained via this approach should be considered approximations. When discharge data are available, the mass (M) of a chemical element removed during an event (from initiation to termination of tile flow) was computed as:

$$M = \sum C_i Q_i - \sum C_o Q_o \qquad \text{(Eq. 2)}$$

in which, Q_i and Q_o are water discharge at the inlet and outlet, respectively.

Chapter 3 Site Hydrology

3.1 Rainfall, water level and discharge

Water level in the bioswale (Fig. 5) and tile discharge were in general very responsive to large rainfall events (> 20 mm). During the monitoring period, the highest water level in the subsurface cell (126 cm) was recorded on April 19, 2013, the day after an 80 mm rainfall event. A distinguishing feature of these rainfall events is that they tend to come as two series of consecutive storms separated by several rainless days. That is reflected in the typical bi-phasic pattern exhibited by the tile discharge into the bioswale (Fig. 6). Based on these observations, each of the 3 wet events described in this report will be analyzed as distinct periods. For example, the November and December 2011 event is divided into two periods: November 28-December 12, and December 14-23.

Hydraulic retention time (HRT) was computed as HRT = (V/Q) using the average discharge (Q) measured in the inlet and outlet, and the average water volume in the bioswale during a given period.

For the November-December 2011 wetting events, the estimated HRT was 0.97 days from Nov. 29 - Dec. 5 and 0.75 days from Dec. 14 - Dec. 25. Table 2 lists the discharge data (at the inlet and outlet) used to calculate the HRT during each of these periods.

Table 2. Mean discharge (Q) and hydraulic retention time (HRT) during the wetting events of November-December 2011.

Flow in inlet		Flow in outlet		HRT, days[†]
Period	Q, $m^3 s^{-1}$	Period	Q, $m^3 s^{-1}$	
Nov. 29 - Dec. 4	0.0015	Nov. 30 - Dec. 5	0.0034	0.97
Dec. 14 - Dec. 21	0.0006	Dec. 15 - Dec. 25	0.0057	0.75

[†]Computed as HRT = (V/Q) using the volume of water in the bioswale (201 m^3 using an average water depth of 0.7 m), and the average discharge measured in the inlet and outlet.

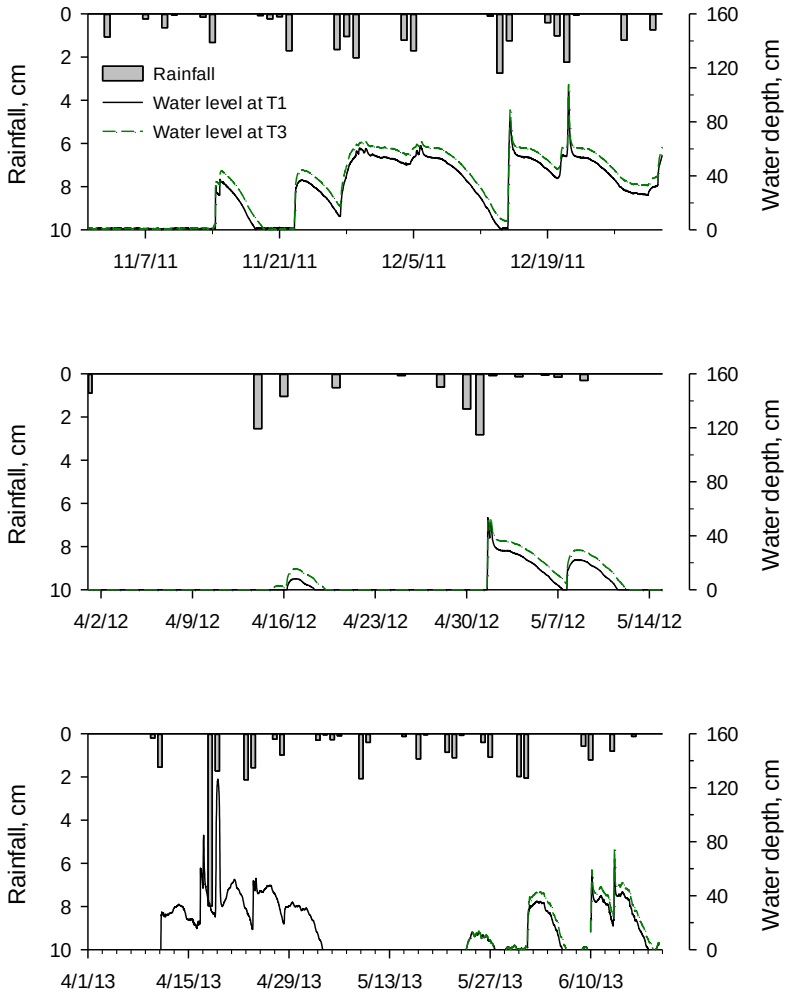


Fig. 5. Rainfall and water depth in the subsurface cell during wet events. Water level was measured in piezometers/monitoring wells installed in the first (T1) and third (T3) transect. Water level data in 2011 and 2012 was recorded using solinst level loggers in the piezometers. In 2013, water level was recorded using YSI 600XLM water quality sondes. Water level was not recorded in T3 during the April 2013 event due to instrument malfunction. Relatively higher water level at T3 compared to T1 was due to the 7 % slope at the bottom of the subsurface cell.

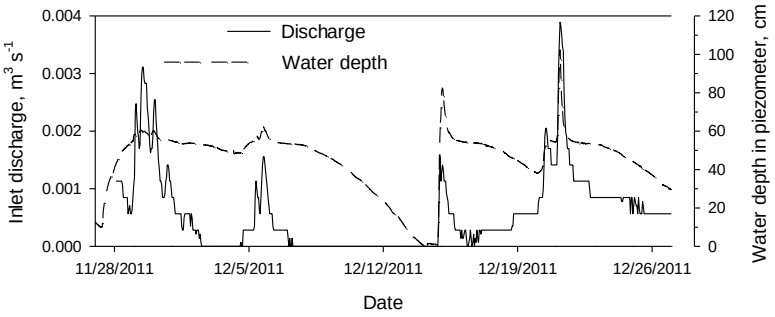


Fig. 6. Temporal variation in inlet discharge and water depth during the November-December 2011 event. Water depth was measured in the T3 piezometer.

Chapter 4 Water Quality Results

4.1 Nitrate and Ammonium

During the study, the concentration of NO_3^- in drainage water entering the bioswale ranged between 0.43 and 13 mg N L^{-1}. Peak NO_3^- concentration in the inlet was observed after fertilizer application to the corn crop in June 2013 (Fig. 7). Concentration was generally lowest in spring 2012 when soybean was grown and rainfall was below normal. Temporal variation in NH_4^+ concentration was less pronounced, ranging between 0.003 and 1.35 mg N L^{-1}. There was no clear trend between the temporal variation in NH_4^+ concentration, dissolved O_2 and temperature.

Nitrate concentration was consistently lower in the outflow to the stream than in the inflow. On the basis of observed patterns in tile discharge pattern and water level in the bioswale, the NO_3^- data was divided into periods - roughly corresponding to the time between a perceptible rise in water level and its return to base level. For each period, the average NO_3^- concentration in the inlet and outlet was computed, and NO_3^- retention was determined using Equation 1. For the period of study, N removal rate averaged 29 % (range: 16-50 %).

The mass (kg N) of NO_3^- in and out of the bioswale was computed for the period of November-December 2011 using Equation 2. This computation was not possible for the other periods due to difficulty of obtaining reliable discharge data. The cumulative amount of N delivered to the system during the two wetting events totaled 12.3 kg N (Fig. 8). Cumulative N in outflow was 6.8 kg N during that same period. That is equivalent to a 44 % retention rate. This rate is slightly higher, but well within the range of N removal (21-38 %) determined using Equation 1. The level of N removal (mean: 30 %) measured during the November-December wetting events is similar to what has been observed in an experimental (technical scale) reactive swale at UBA (German Federal Environmental Agency, Berlin) (Camilo et al. 2014).

The data presented in Table 2 show an average hydraulic residence time (HRT) of 0.9 days during the wetting events of November-December 2011. Visual inspection of the data presented in Fig. 8 further corroborates this HRT estimate. As can be seen in Fig. 8, during the first wetting event, there was a time gap of about 1 day between detection of water flow in the inlet (Nov. 29, 2 PM) and flow arrival in the outlet (Nov. 30, 3 PM). Similarly, a time gap of about 1.2 day was observed during the second wetting event [flow was detected on December 14 (8 PM) and December 16 (1 AM) in the inlet and outlet, respectively; Fig. 8].

As noted above, the flow-generating events monitored during the study tended to produce two consecutive wet periods. Close inspection of the results showed that, in all these cases, the percent N removal was greater (1.7-fold on average) in the second period than after the first wetting period (Fig. 9). There are at least two factors that could contribute to that trend. One factor is internal production of mineral N between wetting periods. Although there was no inflow, soils in the bioswale likely remained moist and within the optimum range for sustained mineralization of organic N (conversion of organic N to mineral forms such as NH_4^+ and NO_3^-). The increase in NH_4^+ concentration half-way through the wetting events of December 2011 (highest NH_4^+ concentration of 1.35 mg N L^{-1} recorded on Dec 15; Fig. 7) is consistent with that argument. If a significant amount of N was produced internally, then the amount of NO_3^- metabolized in the bioswale would be much greater than determined by considering only NO_3^- in inlet waters. Second, even when the first series of storms ended, the bioswale still held a

significant amount of water. For example, water level was ~60 cm deep when tile inflow stopped on December 6, 2011 (Fig. 6). It is therefore conceivable that prevailing wet soil conditions may have created an environment favorable for denitrifying microorganisms. In other words, the bioswale becomes better conditioned for denitrification during the second wetting period than during the first. This would explain the consistently higher N removal rate measured during the second than during the first series of storms (Fig. 9). Dilution is not likely to be an important contributor because inlet NO_3^- concentration during these consecutive wetting periods remained within a similar range.

A positive relationship was found between N removal rate (y) and the concentration of NO_3^- (x) in the inlet ($y = 5.72x - 0.96$, r^2: 0.81, $P < 0.01$; Fig. 9). This trend is consistent with past studies (Martin and Reddy, 1997; Liu et al., 2005; Beutel et al, 2009), and suggested that in the range of NO_3^- concentration processed by the bioswale denitrification proceeded as a first-order reaction. However, no significant relationship between N removal and water temperature (8.3 - 16.6 ^oC) was found. The lack of a significant effect of temperature is at variance with the results reported in previous AQUISAFE experiments (Jacinthe et al., 2009; Wicke, 2014). The limited effect of water temperature may be linked to the thermal stability of the subsurface cell where most of the NO_3^- removal occurred. Contrary to the wetland systems investigated in above-referenced studies, water temperature varied little in the subsurface cell (Spring 2013 being the exception) (Fig. 10). Greater variations in water temperature would have been observed if measurements were also made in the summer and winter. That was not done because, during the monitoring period, there was no tile flow in these seasons. In this region, tile flow typically stops during the summer and does not resume until late fall to early winter.

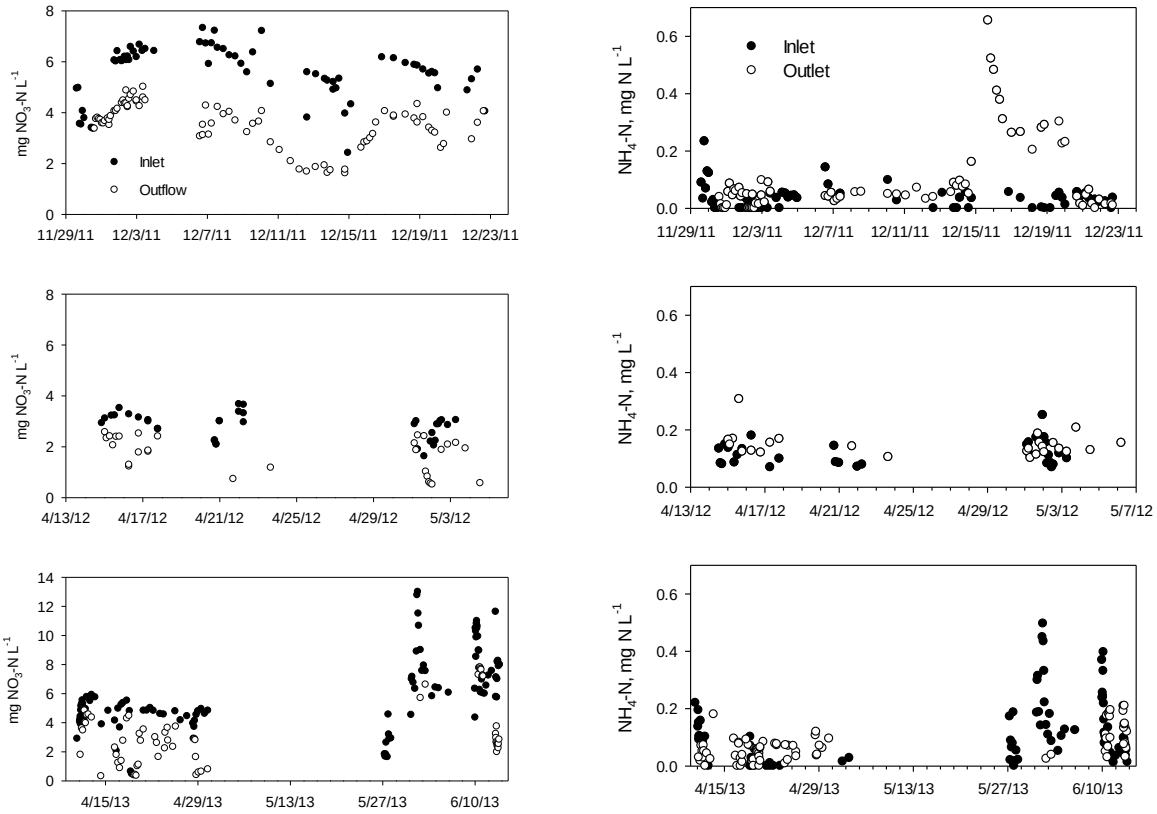


Fig. 7. Nitrate (left panel) and ammonium (right panel) concentration in the inlet and outflow of the bioswale during wet events in autumn 2011 (top), spring 2012 (middle) and spring 2013 (bottom). Note the scale difference for the spring 2013 event.

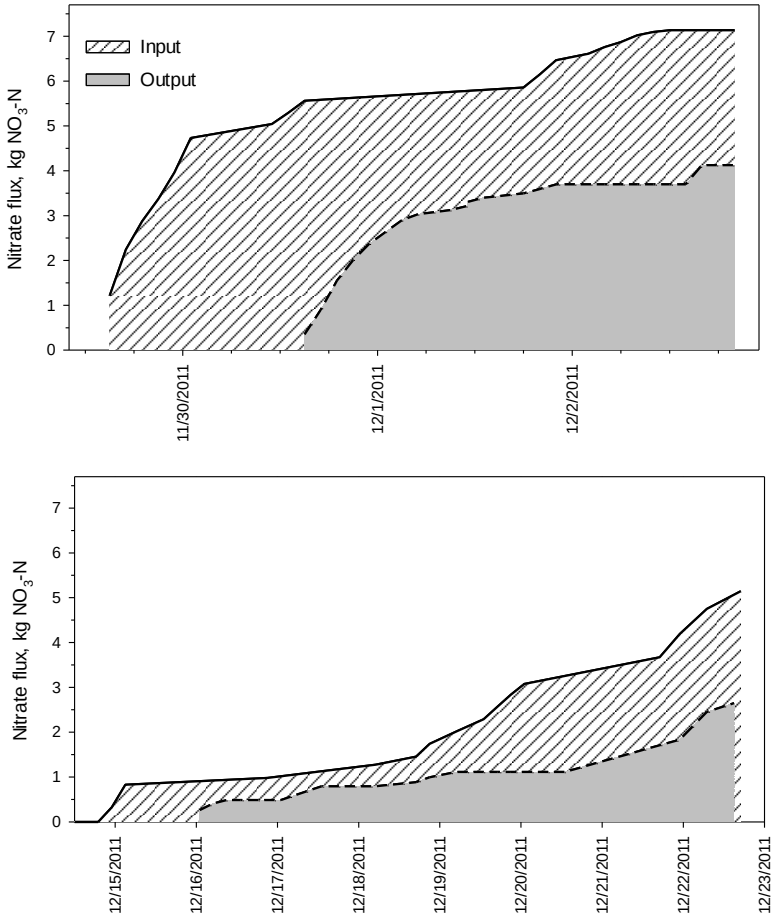


Fig. 8. Cumulative amount of N in inflow and outflow during the December 2011 events.

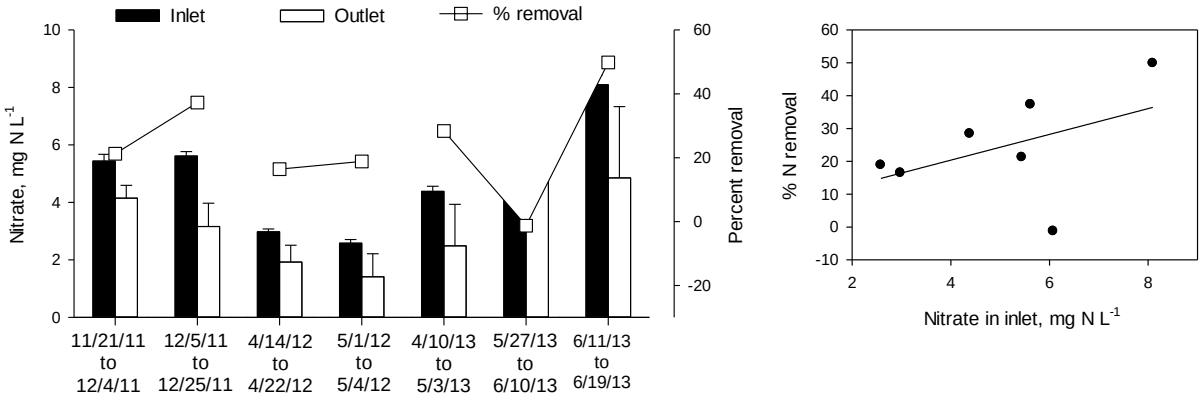


Fig. 9. Percent removal of nitrate during the course of the study (left), and relationship between NO_3^- concentration in the inlet and % N removal. Bars represent standard errors.

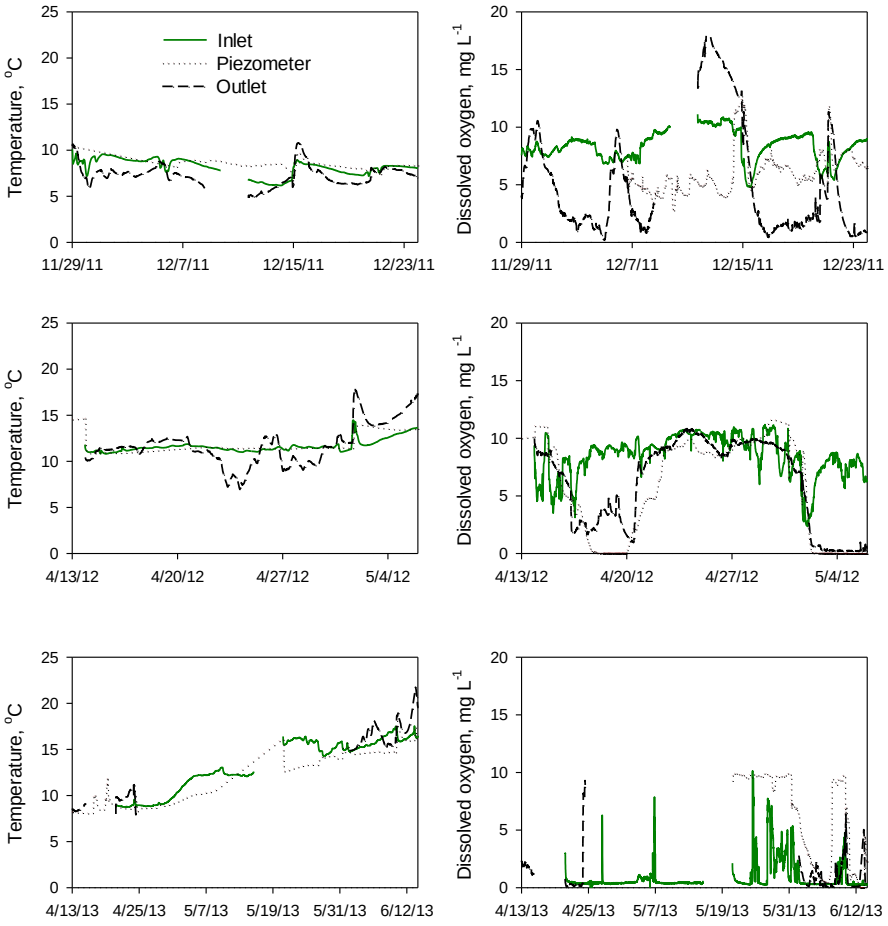


Fig. 10: Water temperature and dissolved oxygen at the inlet, T3 piezometer and outflow of the bioswale during wet events in autumn 2011 (top), spring 2012 (middle) and spring 2013 (bottom).

4.2 Soluble reactive P

SRP concentrations in the bioswale exhibited limited temporal patterns. Throughout the study, average concentration in the inlet was between 0.18 - 0.22 mg P L^{-1} (Fig. 11). No increase in SRP concentration was observed following fertilizer application to corn in June 2013. The most noticeable increase in SRP concentration was measured on December 15, 2011 (up to 2.1 mg P L^{-1}; Fig. 11), a period when the highest concentration of NH_4^+ was observed. Since ORP during that period was between 100-200 mv (Fig.14), reduction reactions could not explain these results. Assuming that NO_3^- removal observed during that period was due to denitrification, a process catalyzed by heterotrophic bacteria and in which organic carbon is used as an electron donor and thus undergoes mineralization. As organic matter is mineralized, organic N and P molecules are converted into mineral N and P. Thus, organic matter mineralization may have contributed to the pulse in NH_4^+ and SRP production during that period.

Comparison of inflow and outflow data showed that the bioswale was a net source of SRP during most sampling periods with an overall SRP release, corresponding to -13 % of input during the study (Fig. 12). The wetting event of late December 2011 resulted in the highest SRP release (-114 %). Using Equation 2, an estimated 0.63 kg P entered the system in the period of December 13-22, 2011. About 0.97 kg P left the system within the same period, corresponding to the net release of 0.36 kg P. This P release may have originated from the water soluble pool (WEP) of P in the topsoil applied during construction of the bioswale (Table 1). Assuming a soil bulk density of 1.2 g cm^{-3}, and using the information in Table 1, the amount of water soluble P in the topsoil is estimated at 0.61 kg P – an amount more than adequate to sustain the initial SRP release observed in the study. In accord with the results of a mescosom study with soils from the study site (Smith and Jacinthe, 2014), the rate of SRP release is expected to decrease as the pool of water soluble P is progressively exhausted. This expectation is supported by the observation that SRP release was much less during the second year of operation of the bioswale. Net SRP retention was observed in the third year of operation in 2013 (Fig. 12).

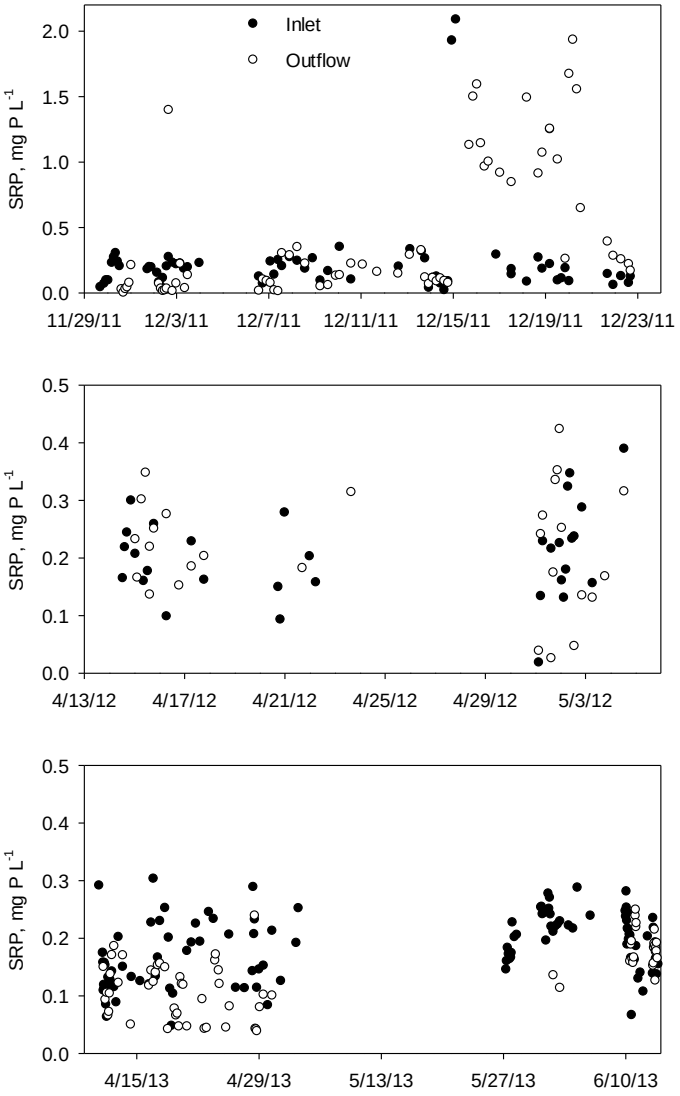


Fig. 11. Concentration of soluble reactive P (SRP) in the inlet and outflow of the bioswale during wet events in autumn 2011 (top), spring 2012 (middle) and spring 2013 (bottom). Note the scale difference for the autumn 2011 event.

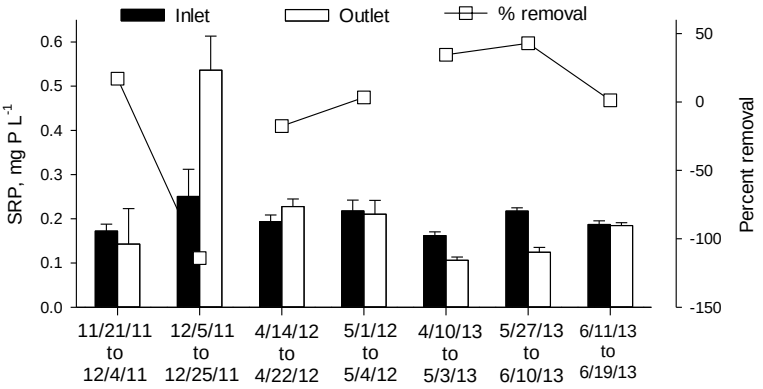


Fig. 12. Percent removal of soluble reactive P (SRP) during the course of the study. Negative values indicate a net release of SRP from the bioswale. Bars represent standard errors.

4.3 Dissolved organic carbon

The temporal trend in DOC concentration was not very well defined. During the study period, DOC concentration averaged 6.2 (range: 1.9-29.2), and 7.4 (range: 2.4-22.1) mg C L^{-1} in the inlet and outlet, respectively (Fig. 13). The highest DOC level in the outlet was recorded following a 28 mm rainfall on May 1, 2012 (Fig. 5) that may have resulted in the delivery of overland runoff to the surface cell. Compared to other locations in the bioswale, DOC was highest in the T3 piezometer, averaging 8.5 mg C L^{-1} in 2013 and reaching 52.3 mg L^{-1} in late April 2013 (Fig. 13). This elevated DOC was observed at a period when water level in the subsurface cell (25-126 cm) remained high for several consecutive days (April 1 - May 2) (Fig. 5). ORP in the monitoring well also reached its lowest level during that period (mean: -233±183 mv; Fig. 14). Increased DOC release under reducing soil conditions has been reported (Hanke et al., 2013; Smith and Jacinthe, 2014), and this has been ascribed to pH-induced dissolution of humic materials. The C-rich bark mulch material at the bottom of the subsurface cell could have further contributed to these results.

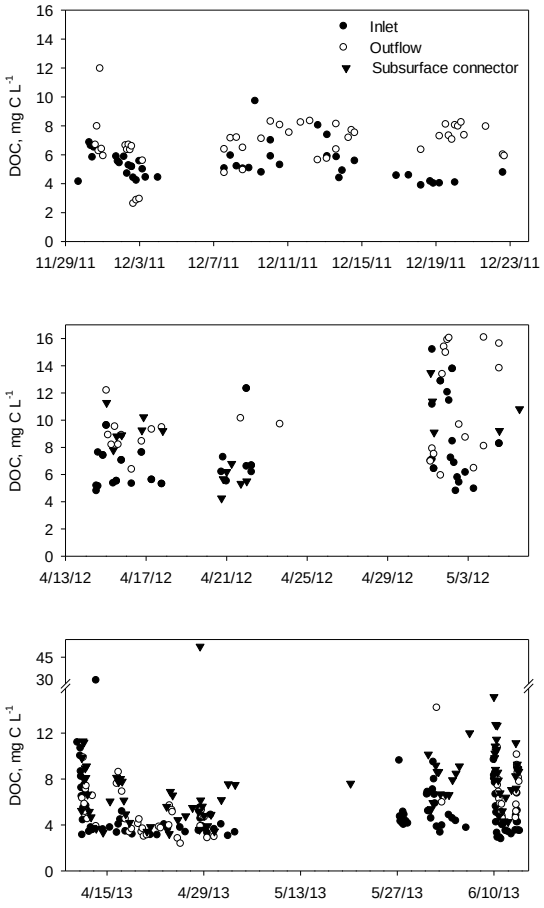


Fig. 13. Dissolved organic carbon (DOC) concentration at the inlet, T3 piezometer and outflow of the bioswale during wet events in autumn 2011 (top), spring 2012 (middle) and spring 2013 (bottom). Note the scale difference for DOC and ORP during the spring 2013 event.

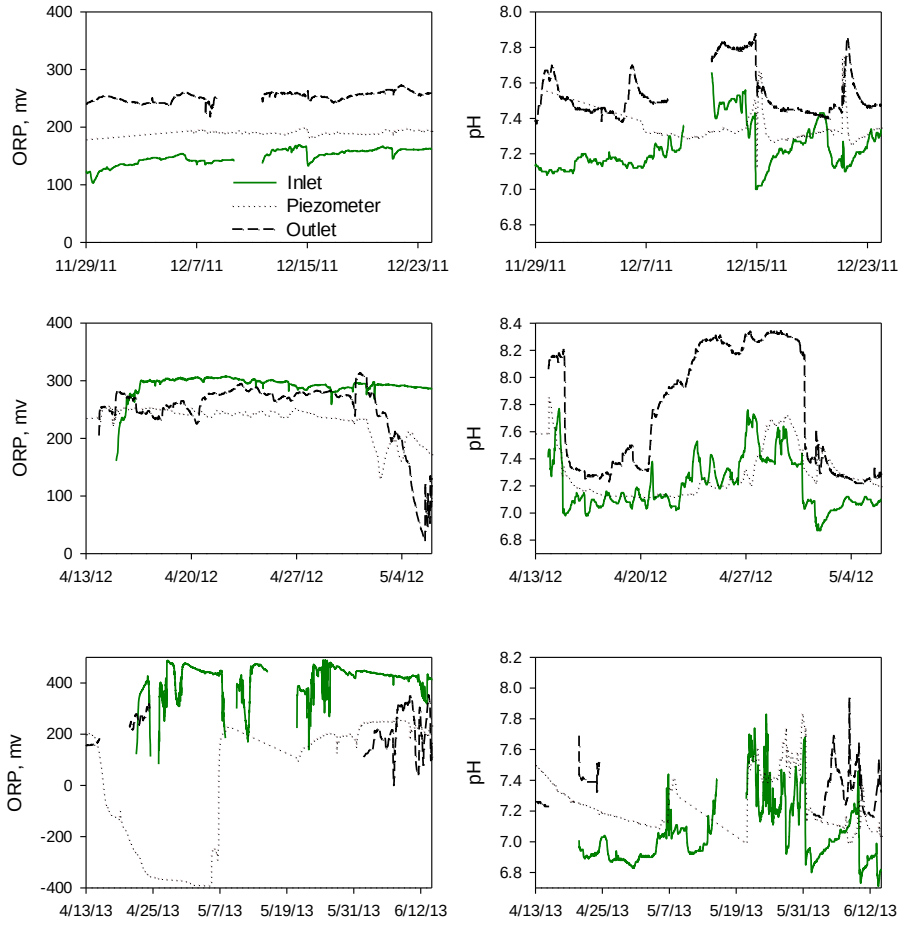


Fig. 14: Oxidation-reduction potential (ORP) (left) and pH (right) in the inlet, T3 piezometer and outflow of the bioswale during wet events in autumn 2011 (top), spring 2012 (middle) and spring 2013 (bottom).

Although N assimilation by growing vegetation may have contributed to N removal in the bioswale, most studies suggest that the process is largely driven by the activity of denitrifying microbes (Hunt et al., 1995; Eriksson and Weisner, 1997; Jaynes et al., 1998; Casey and Klaine, 2001; Kovacic et al., 2006; Reinhardt et al., 2006). Denitrifiers are facultative anaerobic microorganisms that utilize NO_3^- as an alternative electron acceptor in low-O_2 environments. Because denitrifiers are heterotrophs, it is generally assumed that their activity can be stimulated by increased availability of organic carbon. In this study however, limited to no trend was observed between % N removal and DOC concentration in the bioswale (Fig. 15), suggesting that biological activity in the bioswale was not limited by the availability of organic carbon. This interpretation is supported by the high concentration of DOC measured in the bioswale. Similarly, no relationship with DOC and trace gas production has been found in Eagle Creek reservoir - a nearby freshwater reservoir that is fed by School Branch (Jacinthe et al., 2011). Since DOC encompasses a wide range of organic compounds (carbohydrates, proteins, fulvic acids…), future studies could examine the biochemical character of DOC and determine the DOC components that most likely could fuel denitrification in the bioswale.

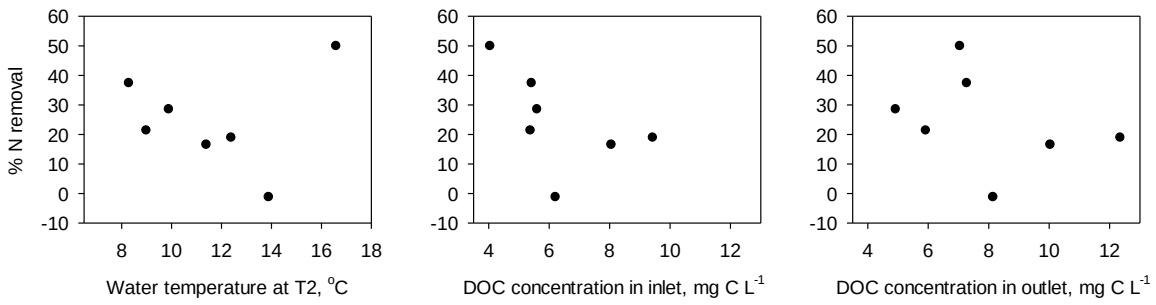


Fig. 15. Relationships between % N removal, and water temperature and dissolved organic carbon (DOC).

Overall, the bioswale was a net source of DOC to the adjacent stream (Fig. 16). In general, DOC in the outflow was 10 to 50% higher than in the inflow. In one month (Nov. - Dec. 2011), a net loss of 2 kg C as DOC was computed, equivalent to an annual loss of 0.8 Mg C ha^{-1} (using bioswale surface area of 312 m^2). While this loss can easily be compensated by vegetation growth and thus not likely to significant affect long-term biological activity in the bioswale, the most immediate concern is the ability of DOC to serve as carrier for the transport of organic agrochemicals. The positive relationship (r^2: 0.56, $P < 0.05$; Fig. 16) between atrazine and DOC concentrations in the bioswale outlet reinforces that concern. Further examination of the relationship will be conducted upon completion of the analysis of the June 2013 samples for atrazine.

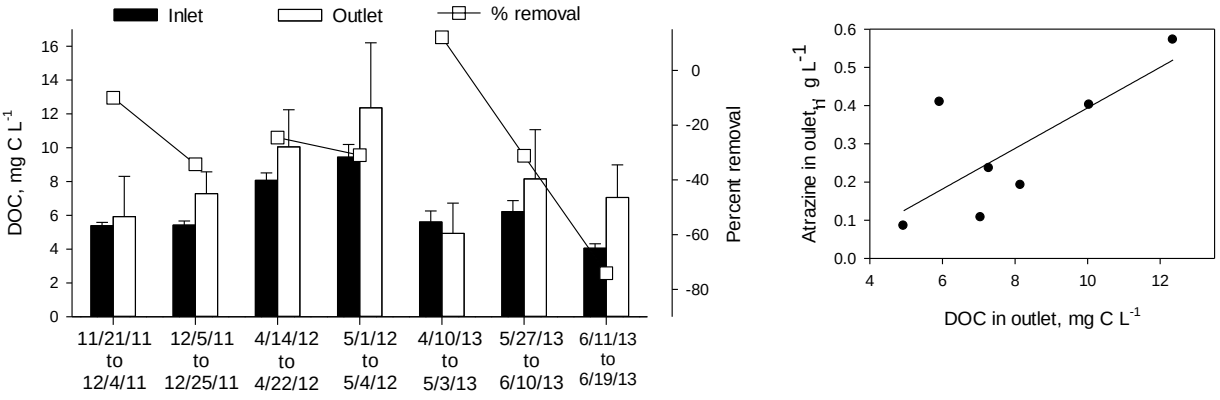


Fig. 16. Dissolved organic C (DOC) balance of the bioswale during the course of the study (left), and relationship between average atrazine and DOC concentration in the outlet (right). Positive values correspond to net gain while negative values indicate a net loss of C from the system. Bars represent standard errors.

4.4 Atrazine

During most of the monitoring period (up to early June 2013; Fig. 17 and Table 3), the concentration of atrazine in inlet waters ranged between 0.05 and 0.71 µg L^{-1}, much lower than the MCL of 3 µg L^{-1} established by USEPA for drinking water. A huge spike in concentration was observed in the days following atrazine application to corn in early June 2013. Concentration as high as 40 µg L^{-1} was measured in the inlet and subsurface cell. Analysis of the samples collected during that period is ongoing. In several diluted samples (1 to 10, and then 1 to 50), both from inlet and outflow, concentration was above the 5 µg L^{-1} detection limit of the method, suggesting that much higher concentrations (> 250 µg L^{-1}) should be expected. Potential for another method of analysis is being investigated to handle the high concentrations in the June 2013 samples.

The bioswale was a small sink (13-31 % retention) for atrazine in some sampling periods, and a net source on others (-38 % to -15 %) (Fig. 18). There did not appear to be a difference between the surface and subsurface cell with regard to atrazine retention (Table 3) which was contrary to expectations that the surface cell, due to a more aerobic environment, would be a stronger sink for atrazine than the subsurface cell (Mudhoo and Garg, 2011). Much higher (49-94 %) atrazine retention rate was found in experiments conducted at the UBA (German Federal Environmental Agency, Berlin) laboratories in a reactive swale loaded with straw and bark mulch as organic substrates, and at HRT of 1.25 days (Camilo et al., 2014). Since the bioswale evaluated in the present investigation only contained bark mulch, it is unclear if that may explain the lower atrazine retention observed. Additionally, contact between atrazine-contaminated inlet water and the organic substrate may also have contributed to the different results. In the UBA experimental design, there was likely appreciable contact between the contaminated water and organic substrate. In the Indianapolis bioswale, however, only a fraction of the contaminated water may have come in contact with the layer of bark mulch located at the bottom of the subsurface cell (Fig. 2), likely resulting in lower atrazine adsorption.

Atrazine removal rates were not computed for the periods May 27-June 10, and June 11-19, 2013 due to difficulties of obtaining accurate results for some water samples suspected of containing elevated concentration of atrazine. Although not significant, a negative relationship ($y = -6.1x + 53.7$, r^2: 0.39, $P < 0.25$) was observed between atrazine removal (y) and DOC concentration in outflow (x) suggesting that high level of DOC could result in lower atrazine retention efficiency by the bioswale. Possible mechanisms for this outcome include: (i) competition between atrazine and DOC for sorption sites on the surface of soil minerals, and (ii) facilitated-transport of atrazine through incorporation within the molecular structure of dissolved humic substances (Celis et al., 1998; Gao et al., 1998; Lima et al., 2010).

Table 3. Atrazine concentration (µg L^{-1}) in different sections of the bioswale during various periods in 2012 and 2013. Values are means with range reported in parentheses.

Period	Location				
	Inlet	Subsurface connector	Surface cell	Outlet	% Removal
2012					
April 4-30	0.46 (0.3-0.57)	0.36 (0.28-0.47)	NA	0.40 (0.39-0.41)	13
May 1-6	0.41 (0.24-0.72)	0.34 (0.19-0.51)	NA	0.56 (0.26-1.27)	-37 (overland flow)
2013					
April 11 – May 2	0.12 (0.06-0.18)	0.10 (0.06-0.13)	0.10 (0.04-0.13)	0.09 (0.005-0.42)	25
May 27 – June 4	0.22 (0.09-0.35)	NA	0.21 (0.05-0.34)	0.19 (0.06-0.42)	14
June 10-15	9 samples > DL†, 4 samples: 26.2-40.7	11 samples > DL, one sample: 45.4	38.6 (1 sample analyzed)	7 samples > DL	NA
June 16 -19	12.6-15.1 (2 samples analyzed)	3 samples > DL, one sample: 146.7	217.2 (1 sample analyzed)	2 samples > DL 2 samples: 0.09-0.11	NA

†Detection limits (DL) for the atrazine ELISA test are 0.04 µg L^{-1} (low) and 5 µg L^{-1} (high). Samples above DL of 5 µg L^{-1} were rerun at 10:1 and 50:1 dilutions. Samples listed in table as above DL have potential concentrations greater than 250 µg L^{-1}.

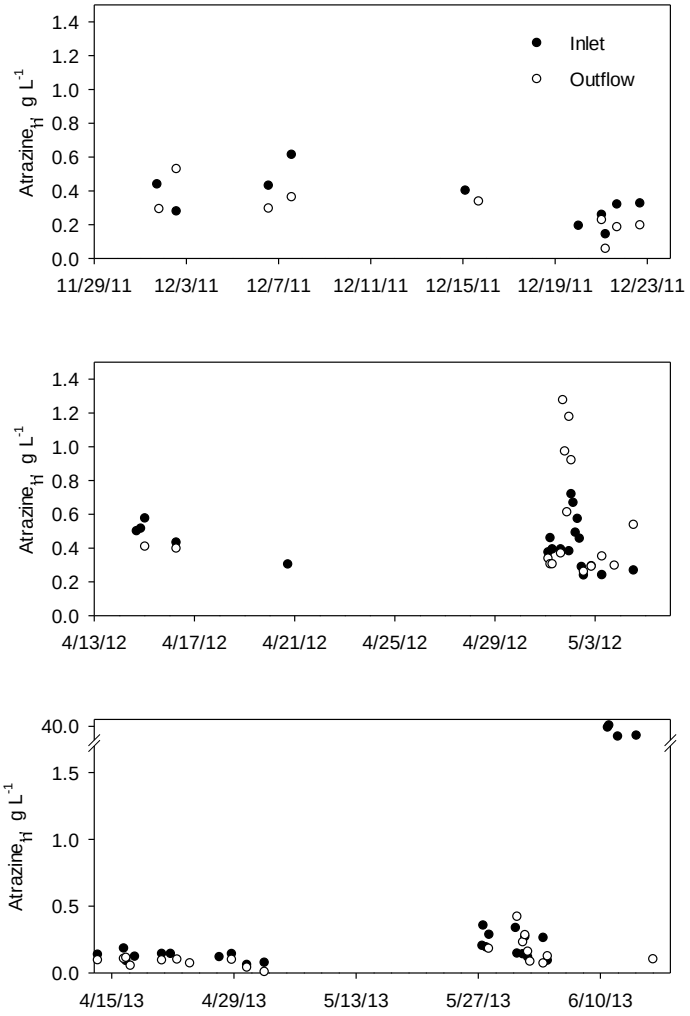


Fig. 17. Atrazine concentration in the inlet and outflow of the bioswale during wet events in autumn 2011 (top left), spring 2012 (middle left) and spring 2013 (bottom left). Atrazine concentration in the inlet and subsurface connector (top right). Note the spring 2013 event does not include the data for June 6 to June 19 due to large differences in atrazine concentration between the April and early June samples. The June 6-19 data is listed in Table 3.

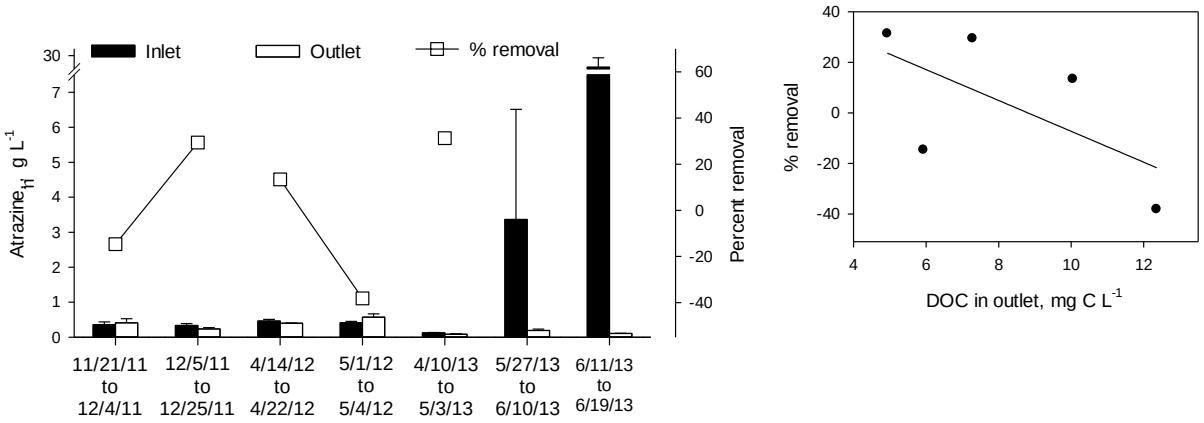


Fig. 18. Percent removal of atrazine in the bioswale during the course of the study (left), and relationship between atrazine removal rate during a period and average concentration of DOC in the outlet (right). Negative values indicate net release of atrazine from the bioswale. Bars represent standard errors.

4.5. Spatial variability of nutrient removal

Owing to its design (anaerobic subsurface and aerobic surface cells), the bioswale is well suited for studies investigating the effect of redox conditions on the fate of water pollutants. An objective of this monitoring study was to compare the efficiency of the two treatment cells. To do so, the spatial distribution of nutrients at different locations (inlet, subsurface cell connector, surface cell and outlet) across the bioswale was examined. This analysis was conducted using the data collected in spring 2013 because water samples in the surface cell were not collected in the previous seasons. The data was divided into 3 periods corresponding to the water input episodes shown in Fig. 5 (bottom graph). Due to the incompleteness of the atrazine results, only the NO_3^- and SRP data were used in this analysis.

There was an overall trend of decreasing concentration of both NO_3^- and SRP from the inlet to other locations of the bioswale (Tables 4 and 5). With regard to SRP, % retention was almost similar in the surface and subsurface cell. However, with regard to NO_3^-, the two sections of the bioswale behaved differently; periods of N retention in the subsurface coinciding with periods of N release from the surface flow cell (April 11- May 3 and June 10 – 11), and a period of N retention in the surface and N release in the subsurface cell (May 27 - June 6).

During the spring 2013 monitoring season, the overall N removal in the bioswale was 35 % of N input, with much (>85 %) of the NO_3^- removal occurring in the subsurface cell (between the inlet and the subsurface connector) (Table 4). These results therefore indicate that a subsurface flow system is the most efficient design to address NO_3^- pollution from agricultural runoff. The surface cell was a net source of N during 2 of the 3 periods. It should be noted that the sub-surface cell was a net source of N during the early June storm that was preceded by a 3-week dry-out (May 3 and May 27; Fig. 5). As discussed previously, the net N release was likely associated with mineralization of organic matter during the drying period. Some of the mineral N produced in the subsurface cell was apparently flushed out into the surface cell by the next pulse

of tile drain water that entered the bioswale starting on May 27. This was reflected in the subsequent increase in NO_3^- concentration observed in the surface cell (from 2.1 to 9.7 mg N L^{-1}). Tile discharge was brief and completed stopped during the rainless week that followed (June 1-8; Fig. 5). Consequently, water discharge in the outlet was very low, averaging 0.00083 m^3 s^{-1}. That would translate to an HRT of 2.5 days. Due to this relatively long HRT, the surface cell acted as a net N sink during the period of May 27-June 6 (Table 4). That was an interesting result because the surface cell was a net source of N at the other periods (when the HRT was ~ 1 day). In combination with the results presented in Fig. 9, this observation suggested that, in a subsurface cell, effective N removal can be achieved with an HRT of ~1 day. However, a longer HRT (> 4 days) would be required in open surface treatment wetlands. A study comparing the performance of several mitigation systems in France and Germany found that, in order to achieve NO_3^- removal >30 %, an HRT of 1 day is adequate in infiltration ditches (similar to bio-reactive swales) whereas in open water wetlands an HRT of 3 days is necessary. Therefore, conclusions reached in both studies are consistent.

Table 4. Nitrate concentration (mg N L^{-1}) and removal rate in different sections of the bioswale in Spring 2013. Values are means ± standard deviations.

	Monitoring period			Seasonal average
	Apr 11- May 3	May 27 - June 6	June 10 - 13	
Water temperature, ^oC	9.4	16.5	18.4	
Discharge, m^3 s^{-1} †	0.0034	0.0008	0.0033	
Inlet (A)	4.27 ± 0.43	6.07 ± 3.39	8.1 ± 1.92	5.07 ± 1.53
Subsurface connector (B)	2.76 ± 0.78	7.22 ± 2.56	3.97 ± 1.85	3.57 ± 1.79
Surface cell (C)	2.13 ± 0.66	8.69 ± 2.11	4.36 ± 1.59	3.38 ± 2.54
Outlet (D)	2.42 ± 0.88	6.16 ±0.65	4.85 ±2.48	3.3 ± 1.71
% N removal‡				
Bioswale overall	43	-1§	40	35
Subsurface cell	35	-19	51	30
Surface cell	-14	29	-11	2

† Discharge measured at the outlet and used to determine hydraulic retention time (HRT) in the surface cell (volume: 172 m^3, water depth 0.6 m).

‡Computations: Overall removal = 100 x (A-D)/A; removal in subsurface cell = 100 x (A-B)/A; removal in surface cell = 100 x (C-D)/C.

§Positive values indicate a net removal whereas negative values indicate a net release of nutrient from the bioswale.

Table 5. Soluble reactive P concentration (mg N L^{-1}) in different sections of the bioswale in Spring 2013. Values are means ± standard deviations.

	Monitoring period			Seasonal average
	Apr 11- May 3	May 27 - June 6	June 10 - 13	
Water temperature, ^{o}C[†]	9.4	16.5	18.4	
Discharge, m^3 s^{-1}	0.0034	0.0008	0.0033	
Inlet (A)	0.16 ± 0.02	0.21 ± 0.04	0.19 ± 0.05	0.17± 0.03
Subsurface connector (B)	0.14 ± 0.01	0.17 ± 0.07	0.21 ± 0.03	0.16 ± 0.03
Surface cell (C)	0.13 ± 0.04	0.17 ± 0.04	0.14 ± 0.04	0.14 ± 0.03
Outlet (D)	0.11±0.02	0.12 ± 0.02	0.18 ± 0.03	0.12 ± 0.03
% SRP removal[‡]				
Bioswale overall	31	43	5	29
Subsurface cell	13	19	-11[§]	6
Surface cell	15	29	-29	14

[†] Discharge measured at the outlet and used to determine hydraulic retention time (HRT) in the surface cell (volume: 172 m^3, water depth 0.6 m).

[‡]Computations: Overall removal = 100 x (A-D)/A; removal in subsurface cell = 100 x (A-B)/A; removal in surface cell = 100 x (C-D)/C.

[§]Positive values indicate a net removal whereas negative values indicate a net release of nutrient from the bioswale.

4.6 Summary of results and conclusions

This field-scale monitoring study was conducted in an effort to assess the potential and limitation of a bioswale as a management approach with regard to diffuse pollution in agricultural landscapes. Results showed that bioswale performance varied with the pollutant. With regard to NO_3^-, the system was fairly efficient (16-58 % removal rates), especially when flow (and thus wet soil) was sustained for several days to allow the development of denitrifying conditions in the wet soil environment. Nitrate removal rates measured in the study were comparable to rates reported for constructed wetlands in various world regions including Switzerland (27 %; Reinhardt et al., 2006), Australia (58 %; Bayley et al., 2003) and Illinois, USA (31-42 %, Kovacic et al., 2006). Interestingly, N removal was within the same range measured in the vegetated (58 %) and un-vegetated (15 %) slow-sand filters at UBA (Berlin) during AQUISAFE-1 (Jacinthe et al., 2009). Nitrate removal capacity increased as the concentration of NO_3^- in the bioswale increased. Activity of the denitrifying community was possibly limited by NO_3^- availability, fluctuations in oxic and anoxic conditions during storm events, and variable wetting and drying of the system, but not affected by temperature and organic carbon (Fig. 15; Table 4). The performance of the system with regard to SRP and atrazine was more variable (net retention during some periods, net release at other times), and controlling biogeochemical factors were more difficult to define. It appears however that organic matter mineralization, desorption of water soluble P from the on-site topsoil used in construction, and occasional delivery of runoff from the adjacent crop field into the bioswale may have contributed to the variable efficiency of the system with regard to SRP and atrazine attenuation.

Chapter 5 Recommendations

5.1 Proposed future work

a) *Atrazine analysis*: Completion of analysis of the 2013 samples for atrazine using an alternative method to the ELISA kit. A dilution ratio of 1:50 was used, but some June 2013 samples remain above the detection limit of the atrazine ELISA method.

b) *Dosing experiments*: Planning and implementation of dosing experiments under controlled flow conditions for more accurate determination of system performance. These experiments will help relate removal rates and HRT because near constant flow will be maintained for pre-determined periods of time. Since conservative tracers (eg. bromide) will be used in the dosing experiments, results will be examined for indication of dilution in the bioswale. In addition, a set of wells was installed next to the bioswale in summer 2013 to monitor water level and determine the likelihood of groundwater intrusion into the bioswale. If evidence of groundwater input is found, estimates of nutrient removal will have to be revised.

c) *Adjustments to system to minimize the impact of overland flow*: The bioswale design includes a pipe connecting the subsurface and surface cells near T3, the downslope end of the subsurface cell. The purpose of this pipe is to allow for the transfer of water from one cell to the other if water movement through the central berm were to become too slow. However, at the beginning of the study, this connection occasionally allowed water to flow back from the surface cell into the subsurface cell. This had created data analysis complications, and invalidated the assumption of unidirectional flow (from inlet to outlet). From observation of sheet flow marks in the nearby field, a zone of concentrated flow was identified on the southeast corner of the bioswale. During large storms, runoff from the adjacent field could bypass the subsurface cell and enter directly into the surface cell. That was observed during the spring 2012 sampling. Evidence of overland flow was also gained through examination of dissolved oxygen and water levels in the subsurface cell. During the April 2012 storm event, dissolved oxygen levels in T3 remained elevated, whereas in T2, there was a decrease in DO levels as the subsurface cell filled with water from the inlet (Fig. 19). Additionally, the water level in T3 rose before the water level in T2 (Fig. 20), suggesting that flow in the subsurface cell was from T3 to T2. This could not occur if water movement was from the inlet towards T3 at the downslope end of the subsurface cell. The original design made no provision for overland flow as a source of nutrients to the system. Yet, during this study and most notably in 2012, it became evident that overland flow must be accounted for in future evaluations of the bioswale.

In August 2012, the connection between T3 and the surface cell was modified by the installation of a one-way check valve at the junction between the surface and subsurface cells. The one-way check valve allows for water flow only from the subsurface to the surface flow cell. All of the data collected after August 2012 reflects this design modification. Future modifications should be made to either completely divert overland flow from the bioswale or to quantify the amount of nutrients delivered to the system via this pathway. This second option will be accomplished as part of a 6-year monitoring program with the Natural Resources Conservation Service (NRCS).

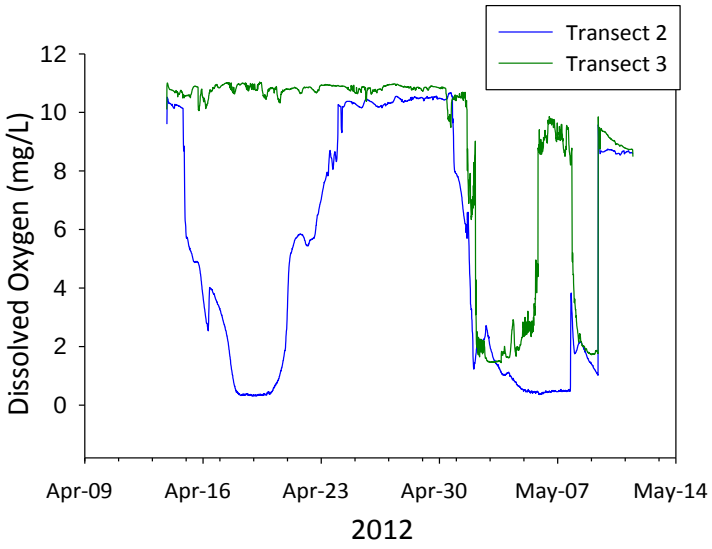


Fig. 19. Dissolved oxygen concentration (mg L^{-1}) in transects 2 and 3 between April and May 2012.

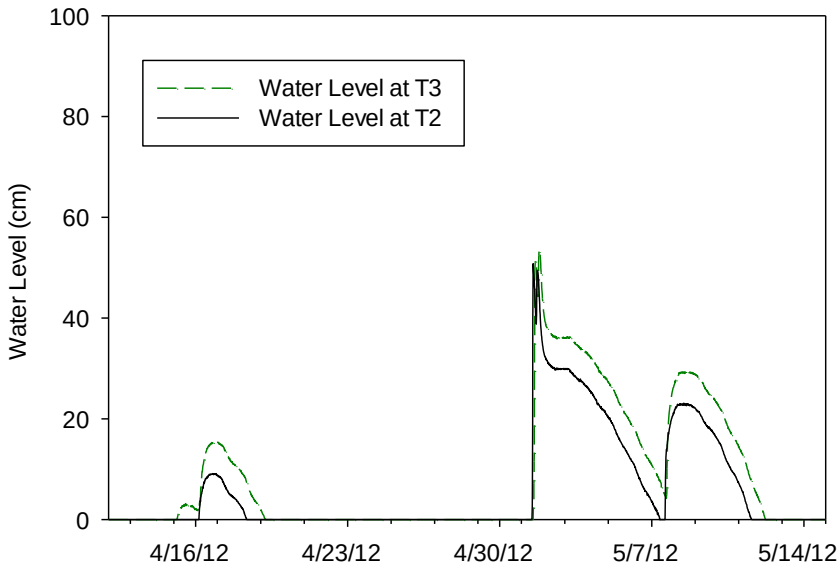


Fig. 20. Water level (cm) in transects 2 and 3 between April and May 2012.

5.2 Recommendations for future design of bioswales

The selection and design of mitigation systems to be installed in agricultural areas must take into account the multiple types of pollutants that can be present in agricultural runoff and tile waters. On the basis of that consideration, the bioswale was designed to include a subsurface and a surface cell in order to create redox environments that are suitable for the degradation of a wide range of pollutants. The evaluation project described in this report was conducted the hypothesis that NO_3^- removal will be optimized in the subsurface cell, whereas the surface cell will provide the best conditions for the degradation of atrazine. Due to unforeseen problems caused by overland flow and analytical difficulties, the atrazine data remains incomplete and the factors controlling the efficiency of atrazine retention in the bioswale are not fully understood. However, if NO_3^- is the pollutant of concern, the subsurface cell is clearly a preferred option compared to the open surface cell, especially if a short hydraulic retention time (< 1 day) is desired. The system can be improved, however. Based on the data collected and field observations, we recommend the following design and management considerations.

Management of tile-drain flow: Tile-drain flow exhibited high temporal variability - periods of high flow interspaced between periods of no flow. This pattern reflected the sporadic and intensive rainfall events observed during the study, and the succession of wetting, drying and rewetting periods observed in the bioswale. The rainfall events observed from autumn 2011 to spring 2013 were typical of the sporadic and often intensive rainstorms observed across the U.S. Midwest. This hydro-climatic variability could pose challenges to maintaining continuous anaerobic conditions conductive to denitrification and removal of NO_3^-. The higher rates of NO_3^- removal measured during the second half of monitored wetting events provided ample evidence for a possible association between prolonged wet periods and effective NO_3^- attenuation in the bioswale (Fig. 9). Therefore, with appropriate management of soil wetness, the NO_3^- removal efficiency of the bioswale can be further improved. Better control of flow discharge at the outlet could help achieve that goal. The installation of a non-permeable liner below the subsurface cell during construction would have also helped in maintaining wet conditions for longer periods of time in the bioswale.

While longer retention time could improve system effectiveness, care must be taken however to ensure that the adjacent agricultural field is not flooded due to prolonged water holding time in the bioswale. An emergency overflow system must be maintained, allowing excess tile waters to bypass the system during excessively wet periods. The monitored bioswale includes such an overflow bypass, as well as connecting pipes that allow for quick transfer of water from the subsurface to the surface cell (Figs. 2 and 3).

References

Andersen J.M. 1976. Ignition method for determination of total phosphorus in lake sediments. Water Research 10: 329-331.

Babbar-Sebens M., Barr R.C., Tedesco L.P., Anderson M. 2013. Spatial identification and optimization of upland wetlands in agricultural watersheds. Ecological Engineering 52: 130-142.

Bayley M.L., Davison L., Headley T.R. 2003. Nitrogen removal from domestic effluent using subsurface flow constructed wetlands: influence of depth, hydraulic residence time and pre-nitrification. Water Science and Technology 48: 175-182.

Beutel M.W., Newton C.D., Brouillard E.S., Watts R.J. 2009. Nitrate removal in surface-flow constructed wetlands treating dilute agricultural runoff in the lower Yakima Basin, Washington. Ecological Engineering 35: 1538-1546.

Camilo B.K., Heise, S., Litz, N. 2014. D 5.3 Extension. Report on efficient design of mitigation zones for pesticide removal. Project Acronym, Aquisafe 2.

Celis R., Barriuso E., Houot S. 1998. Sorption and desorption of atrazine by sludge-amended soil: Dissolved organic matter effects. Journal of Environmental Quality 27: 1348-1356.

Chirnside A.E.M., Ritter W.F., Radosevich M. 2009. Biodegradation of aged residues of atrazine and alachlor in a mix-load site soil. Soil Biology and Biochemistry 41: 2484–2492.

D'Angelo E., Crutchfield J., Vandiviere M. 2001. Rapid, sensitive, microscale determination of phosphate in water and soil. Journal of Environmental Quality 30: 2206-2209.

Eriksson P.G., Weisner S.E.B. 1997. Nitrogen removal in a wastewater reservoir: The importance of denitrification by epiphytic biofilms on submersed vegetation. Journal of Environmental Quality 26: 905-910.

Gao J.P., Maguhn J., Spitzauer P., Kettrup A. 1998. Sorption of pesticides in the sediment of the Teufelsweiher pond (Southern Germany). II: Competitive adsorption, desorption of aged residues and effect of dissolved organic carbon. Water Research 32: 2089-2094.

Gruessner B., Shambaugh N.C., Watzin M.C. 1995. Comparison of an enzyme immunoassay and gas chromatography/ mass spectrometry for the detection of atrazine in surface waters.Environmental Science and Technology 29: 251-254.

Hanke A., Cerli C., Muhr J., Borken W., Kalbitz K. 2013. Redox control on carbon mineralization and dissolved organic matter along a chronosequence of paddy soils. European Journal of Soil Science 64: 476-487.

Hedley M.J., Stewart J.W.B., Chauhan B.S. 1982. Changes in inorganic and organic soil-phosphorus fractions induced by cultivation practices and by laboratory incubations. Soil Science Society of America Journal 46: 970-976.

Jacinthe P.A., Vidon P., Tedesco L.P, Weigert A., Litz N. 2009. Design and operation of mitigation zones (riparian corridors and small-scale constructed wetlands) to reduce downstream loads of pollutants. Project report Aquisafe 1. Kompetenzzentrum Wasser Berlin gGmbH.

Jacinthe P.A., Filippelli G.M., Tedesco L., Raftis R. 2012. Carbon storage and greenhouse gases emission from a fluvial reservoir in an agricultural landscape. Catena 94: 53-63.

Jaynes D.B., Kaspar T.C., Moorman T.B., Parkin T.B. 2008. In situ bioreactors and deep drain-pipe installation to reduce nitrate losses in artificially drained fields. Journal of Environmental Quality 37: 429-436.

Kovacic D.A., Twait R.M., Wallace M.P., Bowling J.M. 2006. Use of created wetlands to improve water quality in the Midwest - Lake Bloomington case study. Ecological Engineering 28: 258-270.

Lima D.L. Schneider R.J., Scherer H.W., Duarte A.C., Santos E.B.H., Esteves V.I. 2010. Sorption-desorption behavior of atrazine on soils subjected to different organic long-term amendments. Journal of Agricultural and Food Chemistry 58: 3101-3106.

Liu W.X., Dahab M.F., Surampalli R.Y. 2005. Nitrogen transformations modeling in subsurface-flow constructed wetlands. Water Environment Research 77: 246-258.

Martin J.F., Reddy K.R. 1997. Interaction and spatial distribution of wetland nitrogen processes. Ecological Modelling 105: 1-21.

Mudhoo A., Garg V.K. 2011. Sorption, transport and transformation of atrazine in soils, minerals and composts: a review. Pedosphere 21: 11-25.

NOAA. 2005. Climatological Data, Indianapolis. National Oceanic and Atmospheric Administration, National Climatic Data Center, Washington, DC.

NRCS. 2006. Land Resource Regions and Major Land Resource Areas of the United States, the Caribbean, and the Pacific Basin. US Department of Agricultural Handbook 296, National Resource Conservation Service (NRCS), Washington, DC.

Reinhardt M., Muller B., Gachter R., Wehrli B. 2006. Nitrogen removal in a small constructed wetland: An isotope mass balance approach. Environmental Science and Technology 40: 3313-3319.

Robertson W.D., Merkley L.C. 2009. In-stream bioreactor for agricultural nitrate treatment. Journal of Environmental Quality 38: 230-237.

Robertson W.D., Blowes D.W., Placek C.J., Cherry J.A. 2000. Long-term performance of in situ reactive barriers for nitrate remediation. Ground Water 38: 689-695.

Smith A.S., Jacinthe P.A. 2014. A mesocosm study of the effects of wet-dry cycles on nutrient release from constructed wetlands in agricultural landscapes. Environmental Science Processes and Impacts 16: 106-115.

USDA. 2001. ARS pesticide properties database, U.S. Department of Agriculture, http://wizard.arsusda.gov/acsl/ppdb.html.

van Driel P.W., Robertson W.D., Merkley L.C. 2006. Denitrification of agricultural drainage using wood-based reactors. American Society of Agricultural and Biological Engineers 49: 565-573.

Wicke D. 2014. Pilot sites for mitigation of diffuse pollution in IC Amont catchment (Brittany). Project report Aquisafe 2. Kompetenzzentrum Wasser Berlin gGmbH.

Appendices

Appendix A. Original Site Design

Appendix B. Adaptation of Original Design

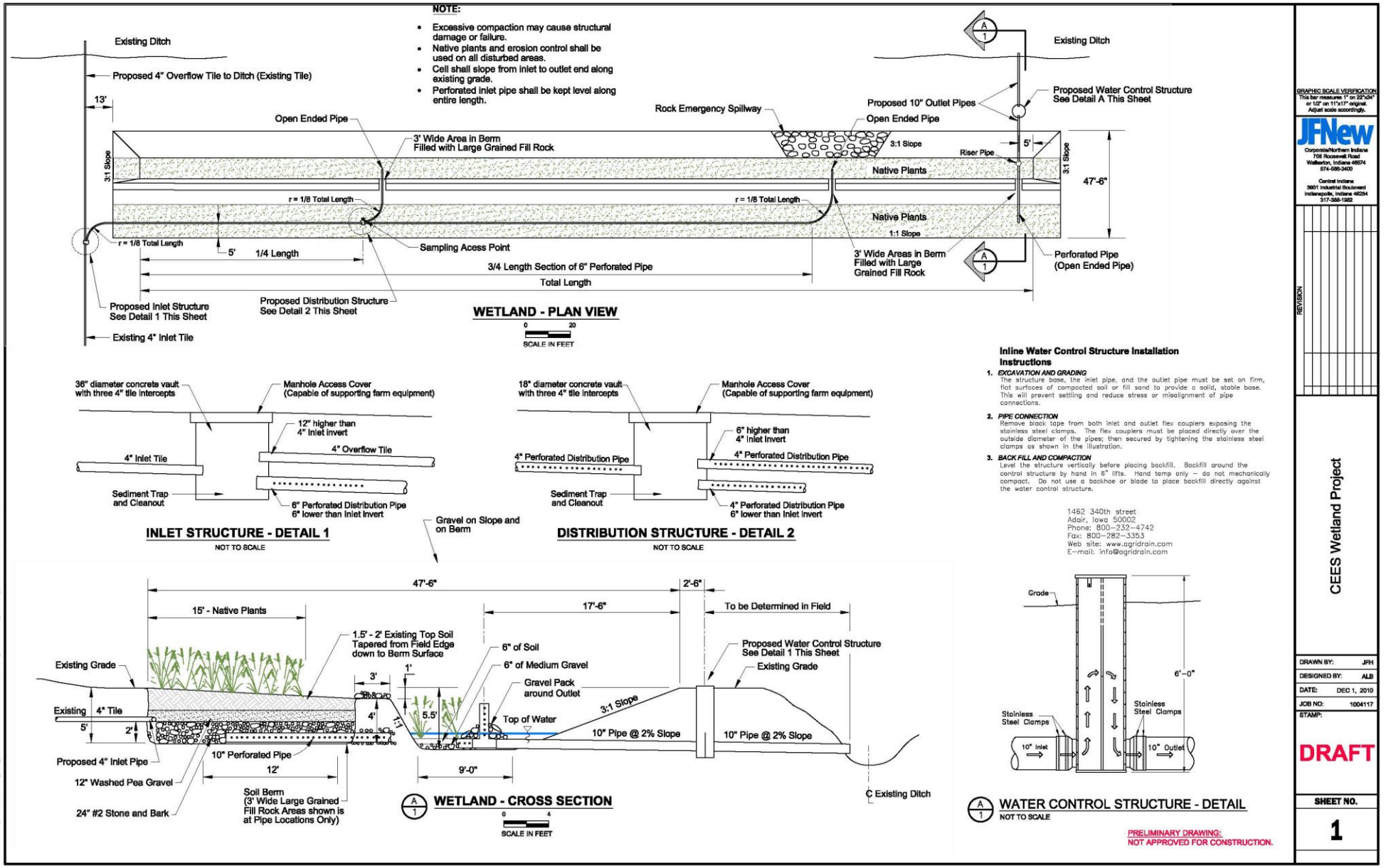
Appendix C. Instrumentation

Appendix D. Water Quality parameter measurements

Appendix E. Storage Volume Calculations

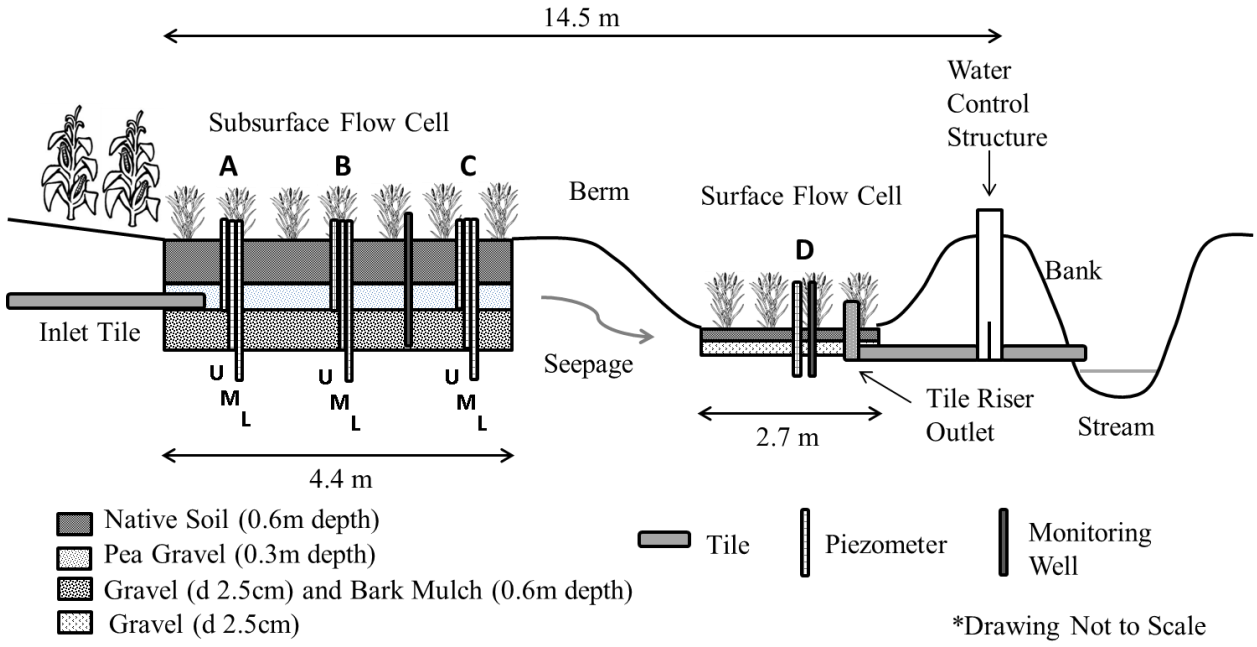
Appendix F. Additional water quality results

Appendix A. Original Site Design

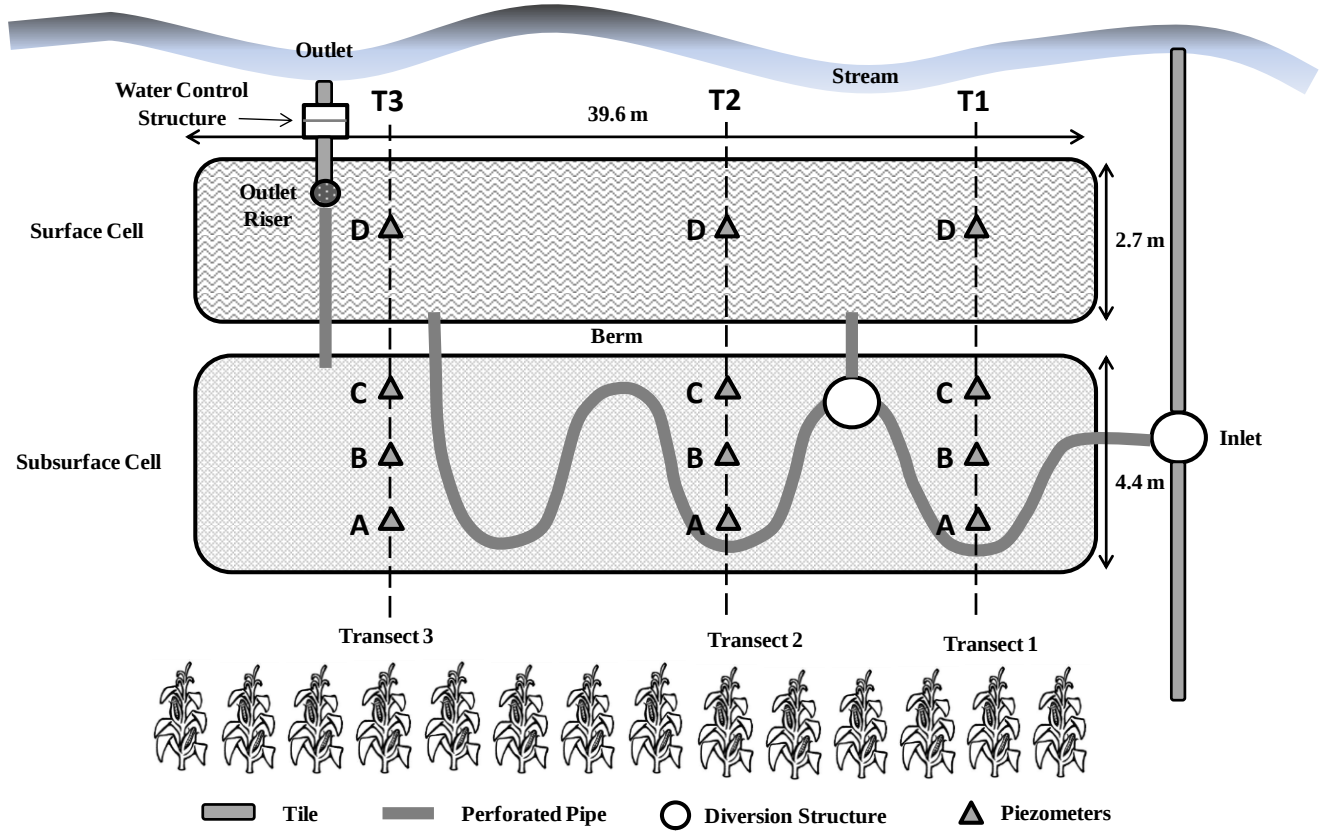


Appendix B. Adaptation of Original Design

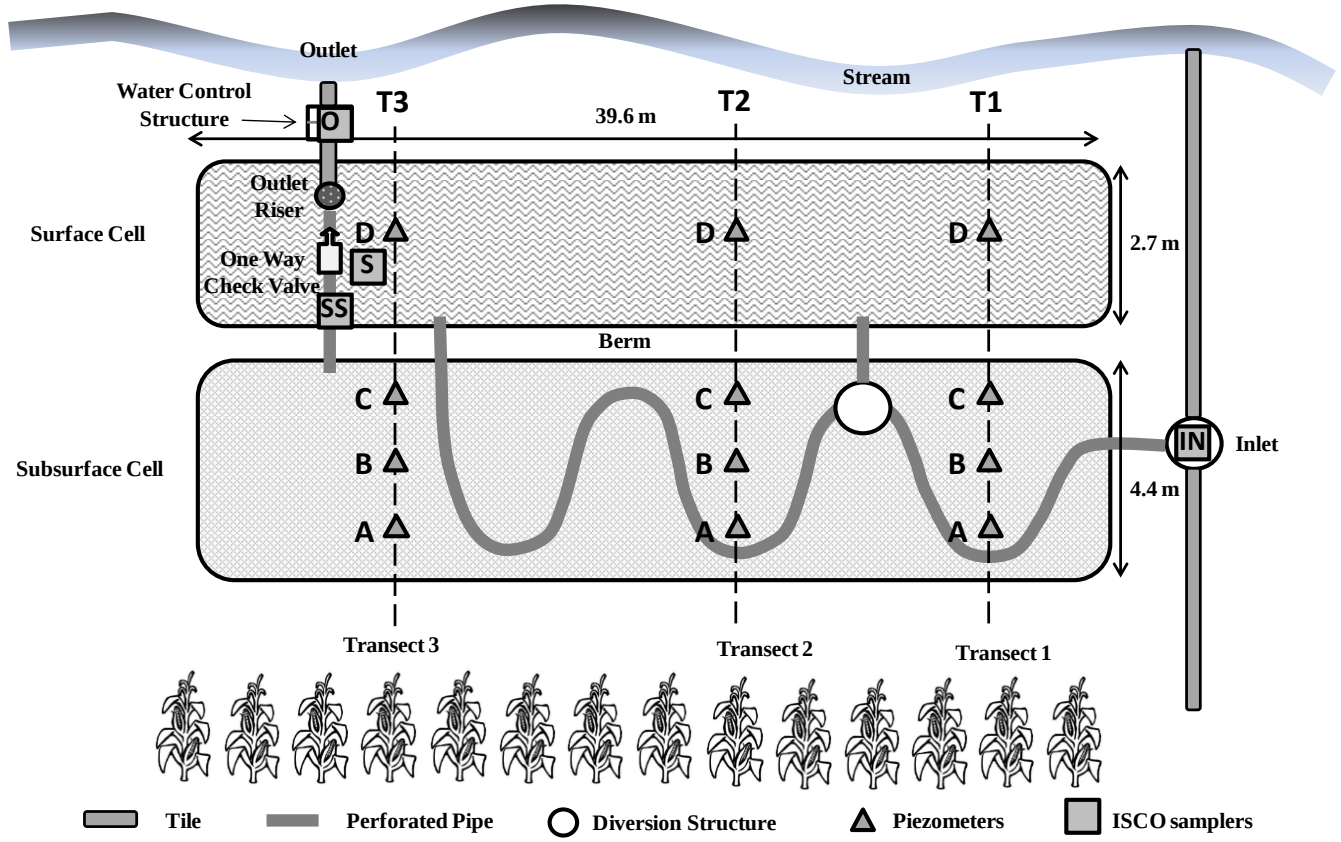
Cross-section View of Bioswale



Top-down View of Bioswale



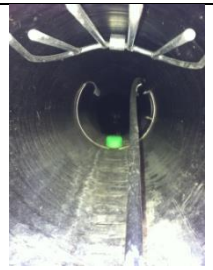
Top-down View of Bioswale with Sampling Locations and Modifications



Modifications to Bioswale

Construction	Compartment	Element	Change	Reason
Bioswale	Inlet	Flow logger	Moved from inlet tile to inlet to bioswale starting with 2012 sampling	Sensor kept clogging with sediment from tile so no flow data recorded
	Cross-over pipe connecting surface and subsurface cells	Cross-over pipe at downstream end of subsurface cell	Closed connection by replacing perforated pipe connecting cells with one way check valve in August 2012	Surface runoff water collecting in surface cell was backflowing into subsurface cell complicating results

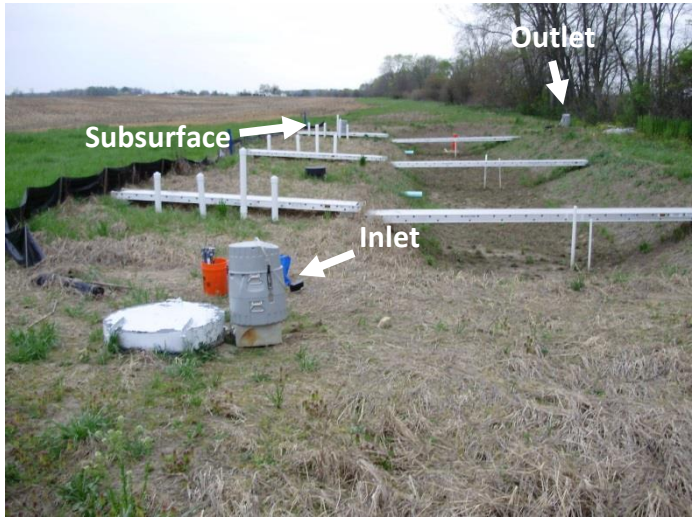
Appendix C. Instrumentation

Measuring point	**Inlet vault**	**Subsurface Cell and Surface Cell**		**Outflow**	
Sensor	Continuous flow level meter and level meter	Continuous pressure sensor	Continuous water quality meter	Continuous flow level meter and level meter	Continuous water quality meter
Model	Hach 900 Series Flow logger	Solinst levelogger	YSI 600XLM Water Quality Sonde	Hach 900 Series Flow logger	YSI 600XLM Water Quality Sonde
Technology	Doppler	Differential pressure sensor	Multiple sensors	Doppler	Multiple sensors
Parameters measured	Flow rate, velocity, level	Water level, temperature	Water level, temperature, conductivity, ORP, dissolved oxygen, pH	Flow rate, velocity, level	Water level, temperature, conductivity, ORP, dissolved oxygen, pH
Date of first installation	16.06.2011	8.06.2011	29.11.2011	16.06.2011	29.11.2011
Associated installation	Inlet tile (later moved to inlet pipe to bioswale in August 2012)	Piezometer	Monitoring Wells	Inlet tile (later moved to inlet pipe to bioswale)	Monitoring Wells
Picture		 www.solinst.com	 www.ysi.com		

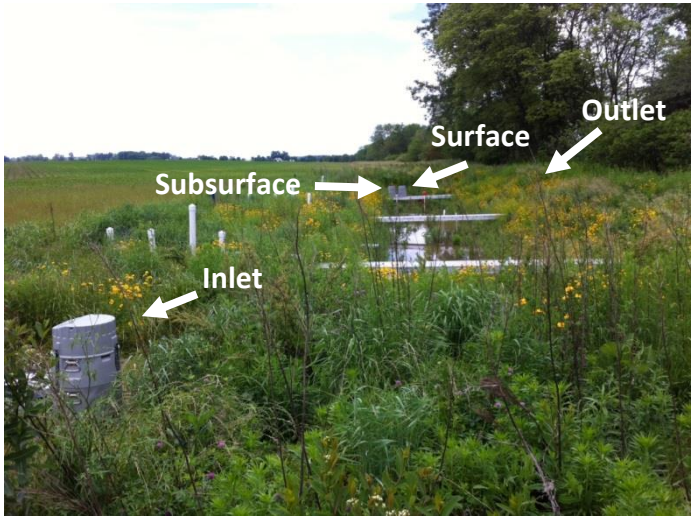
ISCO Samplers

ISCO Locations

2011 – 2012



2013



Appendix D. Water Quality parameter measurements

Parameter	Sampling	Method	Detection Limit	Max of detection	Resolution	Unit	Min	Max	Mean
Ammonium	Events	Microscale determination of NH_4 in water	0.5	10.0		mg NH_4-N/L	0	7.02	0.13
Nitrite (NO2)	Events	EPA 353.1	0.5	10.0		mg NO_3 NO_2-N/L	0.000	0.1395	0.0164
Nitrate (NO3)	Events	EPA 353.1	0.5	10.0		mg NO_3 NO_2-N/L	0.277	12.99	4.25
Orthophosphate (PO4)	Events	EPA 365.3	0.003	0.5		mg PO_4-P/L	0.002	2.09	0.210
Atrazine	Events	ELISA Microtiter Plate	0.04	5.0		Ppb	0.005	217	3.42
DOC	Events	Elementar Vario TOC Cube Analyzer	0	60,000		mg/L	2.36	52.26	6.84
Temperature	Continuous	YSI sondes/solinst levelogger	-5	50	0.01	°C			
Conductivity	Events	YSI sondes	0	100	0.001 to 0.1	mS/cm			
Redox	Events	YSI sondes	-999	999	0.1	mV			
O2	Events	YSI sondes	0	50	0.01	mg/L			
O2 % saturation	Events	YSI sondes	0	500	0.1	%			
pH	Events	YSI sondes	0	14	0.01	Units			

Parameters measured	Frequency of monitoring	Method
Dissolved Oxygen, Temperature, Conductivity, pH, Oxidation-Reduction Potential	Continuous	YSI 600 XLM Multi-Parameter Water Quality Sonde
NO_3-N, NH_4-N, SRP, DOC	Select storm sampling	Photometric method (Aquachem Konelab)
Atrazine	Select storm sampling	Enzyme-linked immunosorbent assay
DOC	Select storm sampling	Vario TOC Cube analyzer

Appendix E. Storage Volume Calculations

Cell One	Unit Length (m)	Thickness (m)	Width (m)	Specific Storage	Water Potential (m^3)
Native Soil	39.6	0.6	4.4	2%	2.09
Pea Gravel	39.6	0.3	4.4	26%	13.59
# 2 Stone w/ Bark	39.6	0.6	4.4	23%	24.05
				Total Volume	39.73

Cell Two	Unit Length (m)	Thickness (m)	Width (m)	Specific Storage	Water Potential (m^3)
Open	39.6	0.6	7.3	100.00%	173.45
Open	39.6	0.3	5.9	100.00%	70.09
Open	39.6	0.3	2.7	100.00%	32.08
Soil	39.6	0.15	2.7	2.00%	0.32
# 2 Stone w/ Bark	39.6	0.15	2.7	23.00%	3.69
				Total Volume	279.63

Theoretical HRT

Vp/Q = 1.23 Days

V = 319.36 m^3, Ponded water volume in both cells of the swale

Q = 0.003 m^3/s, Average inflow rate from flow volume measurements

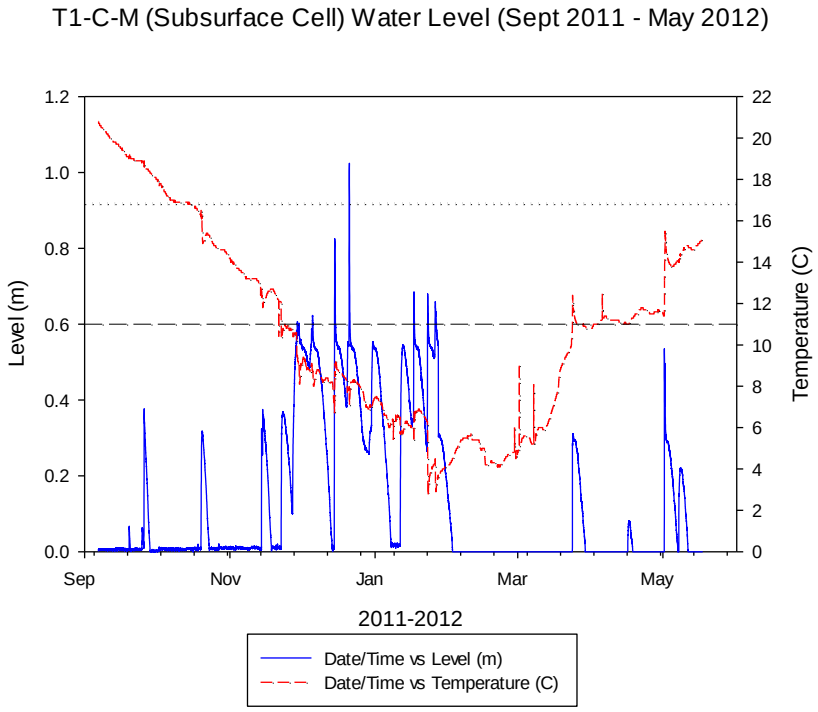
p = porosity (assume 1 for surface flow)

Appendix F. Additional water quality results

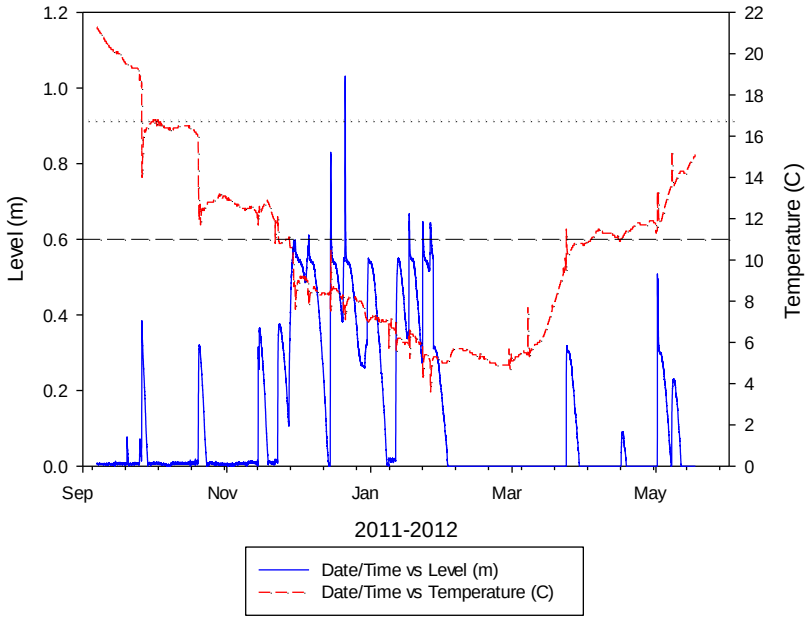
Attached excel file contains all of the water quality results from 2011 to 2013 for nitrate, nitrite, ammonium, soluble reactive phosphorus, dissolved organic carbon, atrazine, flow, water levels, and general water quality parameters (dissolved oxygen, ORP, temperature, conductivity, salinity, pH)

Water Levels and Temperatures in Bioswale during September 2011 to May 2012

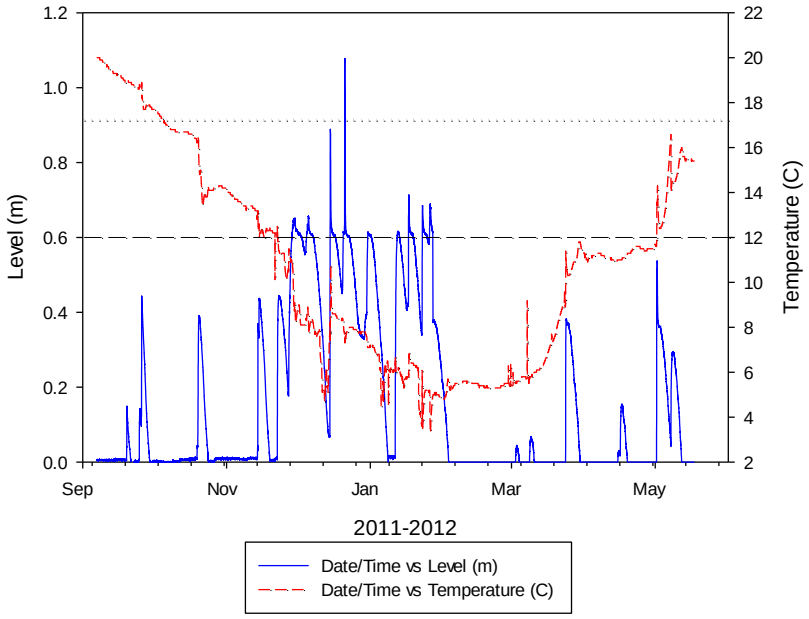
The figures below depict the water levels and temperatures measured in the subsurface and surface cells at T1, T2, and T3 piezometers using Solinst leveloggers.

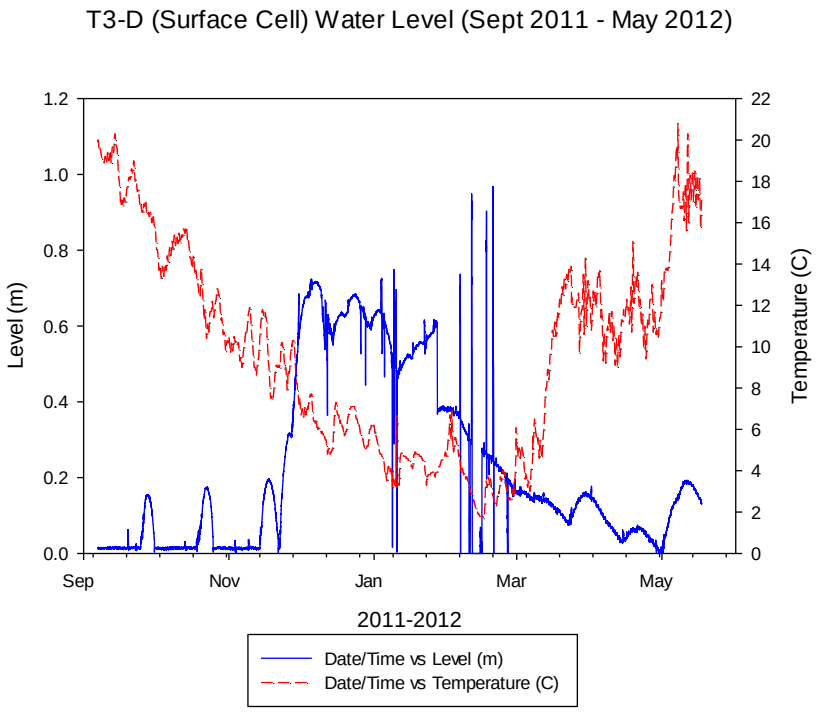


T2-C-M (Subsurface) Water Level (Sept 2011 - May 2012)



T3-C-M (Subsurface Cell) Water Level (Sept 2011 - May 2012)





Water Levels and Temperatures in Bioswale during April to June 2013

The figure below depicts the water levels and temperatures measured in the subsurface cell using the YSI 600XLM sonde. Solinst levelogger data for May 2012 to June 2013 is not available due to equipment malfunction.